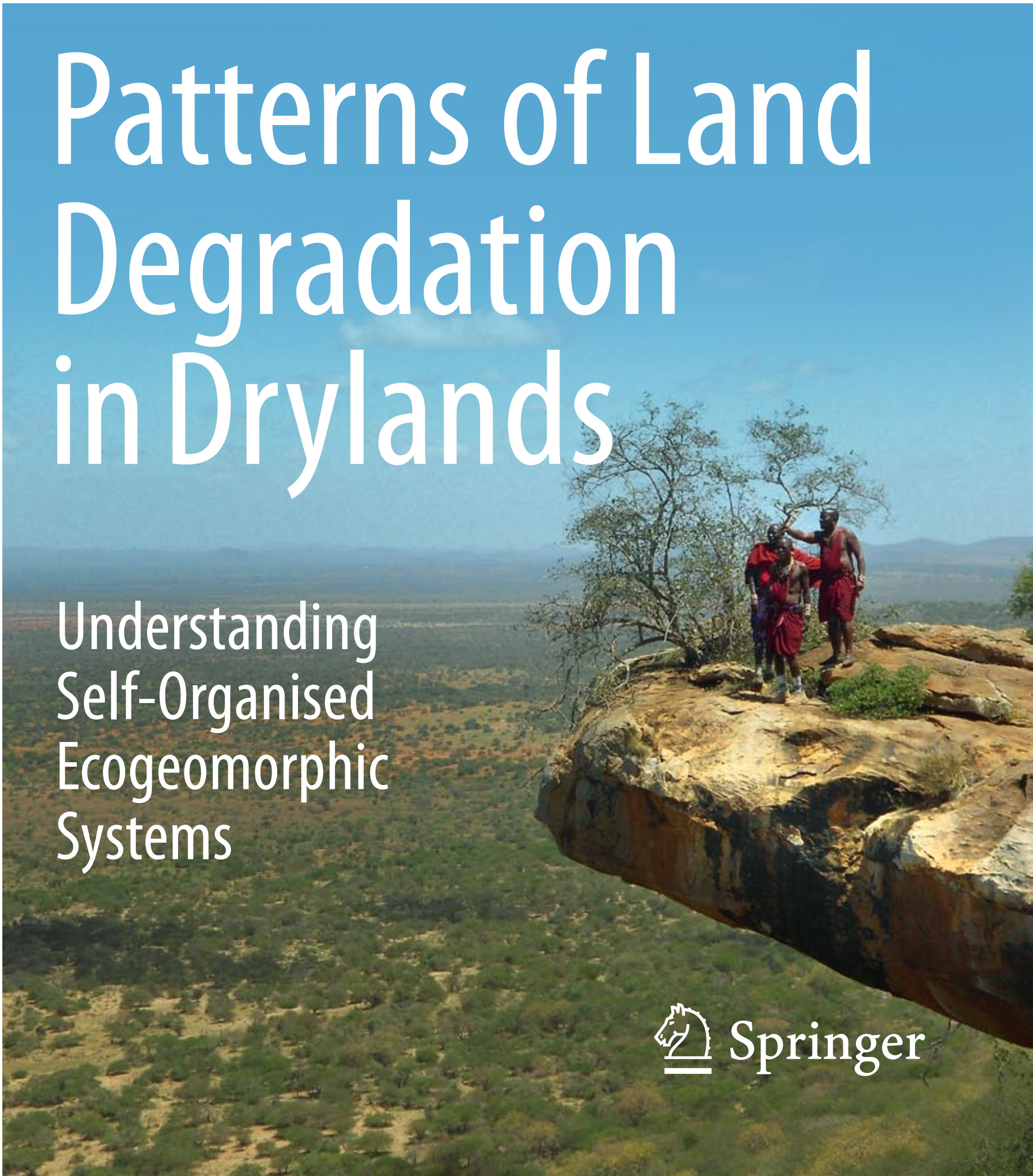
Eva Nora Mueller · John Wainwright
Anthony J. Parsons · Laura Turnbull
Editors

Patterns of Land Degradation in Drylands

Understanding Self-Organised Ecogeomorphic Systems



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Editors
Eva Nora Mueller
Institute of Earth and Environmental Science
University of Potsdam
Potsdam, Germany

Anthony J. Parsons
Sheffield Centre for International Drylands Research
University of Sheffield
Sheffield, UK

John Wainwright
Department of Geography
University of Durham
Durham, UK

Laura Turnbull
Institute of Hazards, Risk and Resilience
Department of Geography
Durham University
Science Laboratories
Durham, UK

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Preface

This book was initiated through an interdisciplinary ESF Exploratory Workshop – 'Self-organized ecogeomorphic systems: confronting models with data for land degradation in drylands' – which was held in Potsdam, Germany, on 7–10 June 2010. The workshop brought together for the first time European scientists working in drylands from multiple communities covering the fields of ecology and landscape ecology, geomorphology, hydrology, agronomy, Earth observation and mathematics to discuss their definitions of land degradation, empirical approaches, questions of corresponding spatial and temporal scales, the importance of self-organization and the application of existing modelling approaches.

During the meeting it became clear that there was a fundamental lack of common ground regarding concepts and methodological approaches between the disciplines. To address this deficiency in understanding land degradation and to provide a basis for future interdisciplinary research on ecogeomorphic systems, we harnessed the expertise of this interdisciplinary group to create this keystone manual.

Individual chapters are multi-authored, integrating research and perspectives from the workshop's participants and several invited experts. As opposed to the usual approach to producing an edited book, we took an editorial decision to select themes that emerged from the workshop and subsequent discussions and invite authors to make short contributions on those themes. We then individually and together integrated the materials produced in order to ensure the consistency and coherence of the chapters. The ordering of the authors names is according to the size of their contribution to the chapters. We would like to thank all the contributors for their full co-operation on the less than usual approach in pulling together this volume in the format we decided on.

The editors would like to thank the European Science Foundation funding scheme for fully supporting the exploratory workshop (Grant reference number EW09-011).

Potsdam, Germany Eva Nora Mueller
Durham, UK John Wainwright
Sheffield, UK Anthony J. Parsons
Durham, UK Laura Turnbull

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Contributors

Concepcion L. Alados Pyrenean Institute of Ecology, Zaragoza, Spain

Andreas Baas Department of Geography, King's College London, London, UK

Nicolas Barbier IRD/UMR AMAP, Botany and Computational Plant Architecture, Montpellier, France

Juan Bellot Depto. Ecologia, Universidad de Alicante, Alicante, Spain

Peter Biro Institute of Earth and Environmental Science, University of Potsdam, Potsdam, Germany

Richard E. Brazier Geography, College of Life and Environmental Sciences, University of Exeter, Exeter, UK

Kelly K. Caylor Department of Civil and Environmental Engineering, Princeton University, Princeton, NJ, USA

Artemi Cerdà Department of Geography, University of Valencia, Valencia, Spain

Pierre Couteron IRD/UMR AMAP, Botany and Computational Plant Architecture, Montpellier, France

Vincent Deblauwe IRD/UMR AMAP, Botany and Computational Plant Architecture, Montpellier, France

David L. Dunkerley School of Geography and Environmental Science, Monash University, Melbourne, VIC, Australia

Joan Estrany Department of Earth Sciences, University of the Balearic Islands, Palma de Mallorca, Spain

Almo Farina Instituto di Ecologia e Biologica Ambientale, University of Urbino, Urbino, Italy

Trenton E. Franz Department of Hydrology and Water Resources, University of Arizona, Tucson, AZ, USA

Francesc Gallart Institute of Earth Sciences, Jaume Almera (CSIC), Barcelona, Spain

Volker Grimm Department of Ecological Modelling, Helmholtz Center for Environmental Research UFZ, Leipzig, Germany

Tamara Hochstrasser School of Biology and Environmental Science, Agriculture & Food Science Centre, University College Dublin, Belfield, Dublin, Ireland

Philip Hunke Institute of Earth and Environmental Science, University of Potsdam, Potsdam, Germany

Florian Jeltsch Plant Ecology and Nature Conservation, University of Potsdam, Potsdam, Germany

Tobias Krueger School of Environmental Sciences, University of East Anglia, Norwich, UK

Junran Li USDA-ARS Jornada Experimental Range, New Mexico State University, Las Cruces, NM, USA

Núria Martínez-Carreras Département Environnement et Agro-biotechnologies, Centre de Recherche Public – Gabriel Lippmann, Belvaux, Luxembourg

James D.A. Millington Department of Geography, King's College London, London, UK

Eva Nora Mueller Institute of Earth and Environmental Science, University of Potsdam, Potsdam, Germany

Greg S. Okin Department of Geography, University of California, Los Angeles, CA, USA

Vasilios P. Papanastasis Laboratory of Range Ecology, Aristotle University, Thessaloniki, Greece

Robert R. Parmenter Valles Caldera National Preserve, Jemez Springs, NM, USA

Anthony J. Parsons Sheffield Centre for International Drylands Research, University of Sheffield, Sheffield, UK

Alan Puttock College of Life and Environmental Sciences, University of Exeter, Exeter, UK

Sujith Ravi Department of Environmental Earth System Science, Stanford University, Stanford, CA, USA

Pier Paolo Roggero Desertification Research Centre and Dipartimento di Agraria, Desertification Research Centre and Dipartimento di Agraria, Sassari, Italy

Stefania Scarsoglio Dipartimento di Idraulica, Trasporti ed Infrastrutture Civili, University of Turin, Torino, Italy

Britta Tietjen Institute of Biology, Freie Universität Berlin, Berlin, Germany

Laura Turnbull Institute of Hazards, Risk and Resilience, Department of Geography, Durham University, Science Laboratories, Durham, UK

John Wainwright Department of Geography, University of Durham, Durham, UK

Mareike Wieczorek Department of Geosciences, Alfred Wegener Institute for Polar and Marine Research, Potsdam, Germany

Thorsten Wiegand Department of Ecological Modelling, Helmholtz Centre for Environmental Research UFZ, Leipzig, Germany

Chapter 1
Land Degradation in Drylands: An Ecogeomorphological Approach

Eva Nora Mueller, John Wainwright, Anthony J. Parsons, and Laura Turnbull

Abstract Land degradation is particularly pernicious and pervasive in dryland regions. The dependency of local livelihoods on the services provided by ecosystems is greater in drylands than in any other ecosystems, rendering their inhabitants exceptionally vulnerable to land degradation. Current approaches to managing drylands to mitigate land degradation often fail to produce significant improvements because local knowledge is often undervalued and the complexity of underlying processes leading to land degradation is still not well understood. There remains a need to uncover the underlying dynamics and characteristic responses to environmental drivers and human-induced disturbances. The physical processes associated with land degradation in drylands fall at the interface of ecology and geomorphology. Regrettably, the disciplines of ecology and geomorphology have largely performed research in isolation of each other. The disciplines, in common with most, have a centrifugal perspective, looking outwards from themselves towards cognate disciplines. To address multidisciplinary scientific questions – such as land degradation in drylands – a centripetal approach is required in which the problem is the focus towards which the disciplines direct their attention. The purpose of this

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E.N. Mueller (✉)
Institute of Earth and Environmental Science, University of Potsdam, 14476 Potsdam, Germany
e-mail: eva.mueller@uni-potsdam.de

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

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

A.J. Parsons
Sheffield Centre for International Drylands Research, University of Sheffield,
Sheffield S10 2TN, UK
e-mail: a.j.parsons@sheffield.ac.uk

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book is to take such a centripetal approach towards the understanding of the process linkages between ecogeomorphological dryland processes and patterns to better our understanding of land degradation, and to overcome the lack of interdisciplinarity in current dryland research.

1.1 Land Degradation in Drylands

The United Nations Convention to Combat Desertification (UNCCD) defines land degradation as "a persistent reduction in biological and economic productivity" (UNCCD 1994). Land degradation is a global-scale, ongoing, and relentless problem that poses a major long-term challenge to humans in terms of its adverse impact on biomass productivity, food security, biodiversity and environmental sustainability. Land degradation is particularly pernicious and pervasive in dryland regions, which cover more than 41 % of the Earth's terrestrial surface (Millennium Ecosystem Assessment 2005). Drylands, which are sub-divided into arid, hyper-arid, semi-arid and dry sub-humid areas (Fig. 1.1) are characterized by extremely low primary productivity, nutrient-poor soils and sparse and patchy vegetation, yet have particularly high population-growth rates (MEA 2005), and are now home to over two billion people.

Land degradation in drylands is a multi-faceted problem. Recent efforts to understand land degradation have focused on the biophysical and socio-economic drivers of land degradation, human vulnerability to land degradation in terms of social, economic and political exposure to potentially harmful perturbations, poverty alleviation, and community-driven development to enhance the role of communities the sustainable management of drylands (Reynolds et al. 2007). For the most part,

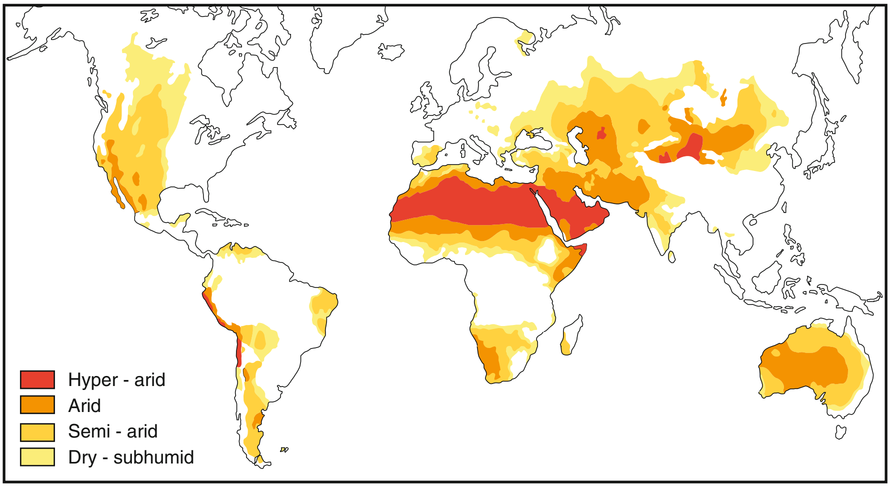

Fig. 1.1 World distribution of drylands (Adapted from Parsons and Abrahams 2009)

people living in drylands lag behind others in terms of human well being and development indicators, and more people in drylands than in any other ecosystem depend on ecosystem services for their basic needs (MEA 2005). Significant ecosystem services, such as agriculture and livestock farming, are dependent upon primary productivity. A reduction in primary production in drylands which is one of the primary characteristics of degradation, reduces the capacity of these systems to provide essential ecosystem services, and ultimately affects the resilience of these systems to future environmental pressures, thus increasing the vulnerability of people living in drylands. *In extremis*, land degradation in drylands can lead to desertification and the effective productive loss of entire landscapes. Estimates of the extent of dryland degradation vary greatly. Lepers (2003) puts it as low as 10 % of the total global dryland extent, whereas Middleton and Thomas (1997) and Dregne and Chou (1992) estimated it in the 1990s at between 20 and 70 %, respectively. At present, there are multiple initiatives seeking to understand different components of land degradation including the Land Degradation Assessment in Drylands project (LADA 2011) and the United Nations Convention to Combat Desertification (UNCCD 1994).

Diverse views are held on the complex relationship between climatic and anthropic drivers of land degradation and how these drivers affect land-degradation processes. Despite this diversity, a relatively broad consensus is presented by the MEA (2005) (Fig. 1.2). Important land-degradation processes in which include soil erosion by wind and water, depletion of soil fertility, soil salinization and changes in soil structure. Changes in soil structure can lead to crusting and compaction, enhancing desertification and anaerobism. Significant chemical processes associated with land degradation include acidification, leaching, salinization and nutrient depletion. Biological processes include alterations in the amount or diversity of natural vegetation or plant cover resulting in a decrease of biodiversity. The net effect of these physical, chemical and biological processes is an increase in the vulnerability of these systems to environmental perturbations, and a reduction in the ecosystem services that these systems can provide.

Land degradation in drylands is a problem that extends beyond the geographical boundaries of drylands. Because of the interconnectedness of many of Earth's processes (illustrated in Fig. 1.2), changes in processes operating in drylands regarding vegetation and soil structure can affect processes at broader spatial scales. For example, dust storms which occur commonly in degraded regions, have negative health implications, and cause broader-scale climate and hydrological feedbacks (Painter et al. 2010). Likewise, soil erosion in drylands mobilizes stored carbon and reduces carbon-sequestration capacity (Lal 2001), thus potentially affecting the global-scale climatic feedbacks (IPCC 2001; Meehl et al. 2007).

Land degradation is driven by a set of interlocking, socio-economic drivers, including industrial and transgenic farming, a globalized economy and capital, speculation in commodities, migration, pollution, falling environmental standards and the capture of arable land for fuel production. Population increase to around nine billion by 2050 (UN 2008) is likely to cause an increase in the extent and intensity of land degradation over the coming decades, as the magnitude and extent of these

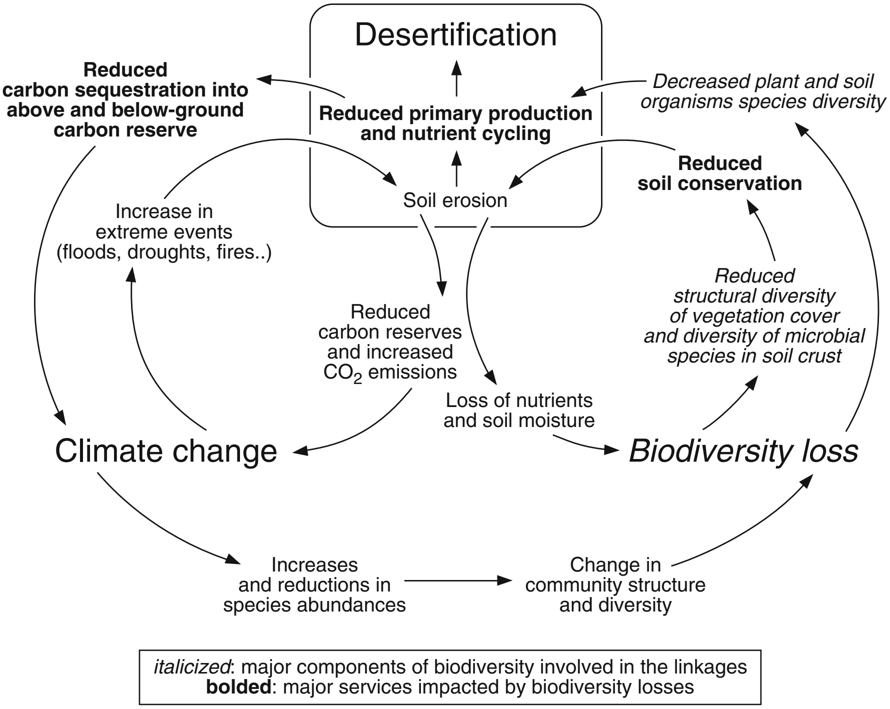

Fig. 1.2 The major components of land degradation that affect the prevision of essential ecosystem services and may ultimately lead to desertification. The *inner loops* connect biodiversity loss and climate change through soil erosion, while the *outer loop* interrelates biodiversity loss and climate change (Modified from Millennium Ecosystem Assessment 2005 and reproduced by permission of World Resources Institute)

drivers increases. For example, the transformation of rangelands to croplands will continue. Depending on the type of cultivation and irrigation methods practised on agricultural land, increases in the severity and extent of soil salinization and erosion will occur. Increases in livestock densities on rangeland will reduce vegetation cover further, exposing the soil to more erosion. Sustainable land management in drylands will play a key role in minimizing land degradation. However, current approaches to manage drylands to mitigate land degradation often fail to produce significant improvements because local knowledge is often undervalued and not included in land-management approaches, and furthermore, the complexity of underlying processes leading to land degradation is still not well understood.

Historically, empirical studies that have dominated the investigation of land degradation have focussed on physical, biological and chemical factors, at a limited range of spatial and temporal scales. The advent of remote sensing and modelling approaches over recent decades has allowed for empirical studies across multiple spatial and temporal scales, allowing greater insights into system behaviour. More recently, modelling-based studies have also played a valuable role in investigating different components of land degradation. The progression of discipline-specific empirical and modelling-based analyses by both ecologists and geomorphologists has inevitably led to specialized approaches and research agenda through which the vital interactions and feedback dynamics between the biotic and abiotic components of the dryland system cannot be explored. While there have been several attempts to overcome disciplinary boundaries in dryland research in the last decades (primary examples include the work by Thornes 1990; Schlesinger et al. 1990; Ludwig et al. 2005; Wainwright et al. 2002; and D'Odorico and Porporato 2006), research efforts considering linkages between social dimensions of land degradation, ecological, hydrological and geomorphological processes, and the structure and function of the system remains limited, and is at best, a compilation of case studies. The ongoing failure to incorporate these biophysical interactions in environmental studies limits our ability to predict the response of drylands to climate change and human-induced disturbances (Reinhardt et al. 2010) and to make sustainable land-management decisions accordingly.

1.2 Nonlinear Dynamics, Self-organization and Connectivity

In disciplines concerned with land degradation, there has been much discussion about *driving processes* and *emerging patterns*. Patterns and processes are mutually causal, but there are still many unknowns as to how these processes and patterns are connected. For a better understanding of dryland systems and land degradation, there is a need to uncover the underlying dynamics and characteristic responses to environmental drivers and human-induced disturbances.

Changes in system state arising from land degradation have been widely demonstrated to exhibit nonlinear, threshold dynamics (e.g. Laycock 1991). Because of hysteresis – a characteristic often associated with such dynamics – a system that has transitioned to a degraded stable state may not return to its former state following the removal of the driving forces of degradation. Another underlying property of drylands is self-organization (Barbier et al. 2006; Deblauwe et al. 2011), which is when larger-scale properties such as vegetation patterning emerge as a response to local-scale interactions.

Previously, there has been a focus on identifying thresholds at which sudden changes in system state occur, under the premise that if it is known when a threshold will be reached, it can be avoided. However, given the underlying complexity of dryland systems in terms of their self-organizing characteristics and system feedbacks, approaches that simply identify environmental thresholds in different types of systems are limited in their capacity to aid our understanding of land degradation. In light of this complexity, it has been recognized that to understand land degradation in drylands, attention needs to be paid to the dynamics of system state change in terms of changes in processes, self-organizing characteristics and biotic-abiotic feedbacks rather than focussing on trying to identify thresholds (Turnbull et al. 2008). It has been hypothesized that land degradation can be represented by a cusp-catastrophe model (ibid., see also Chap. 3), whereby changes in ecosystem states or dynamic régimes can be reinforced by positive stabilizing feedbacks between plants and their environment that ultimately creates high ecosystem resilience.

One key aspect of understanding changes in processes, self-organizing characteristics and biotic-abiotic feedbacks in drylands, is the extent to which system components are connected, both structurally and functionally. This concept of "connectivity" has already been the subject of considerable research (Taylor et al. 1993; Bracken and Croke 2007; Turnbull et al. 2008; Okin et al. 2009; Wainwright et al. 2011). This body of research indicates that both patterns and processes in drylands are controlled by the interaction of advective and diffusive flows of resources such as soil, water, nitrogen and propagules. Spatial heterogeneities in soil and vegetation properties are the result of these fluxes that, in turn, promote the emergence of connected pathways that modify further the advective and diffusive fluxes. The connectivity of these patterns and processes leading to system feedbacks are especially pronounced in drylands due to the high rates of processes in the abiotic régime (Turnbull et al. 2012).

There is a variety of approaches available to study the nonlinear nature of land degradation in drylands. However, all of the concepts introduced above are normally applied by separated disciplines to analyse only one (biotic or abiotic) constituent of a dryland system. That is identified here as the key problem of past and current dryland research: a lack of an integrated analysis of ecogeomorphic systems.

1.3 Ecogeomorphology in Drylands: The Purpose and Content of the Book

Processes and patterns associated with land degradation in drylands fall at the interface of ecology and geomorphology. Over recent years there has been increasing recognition of ecogeomorphology – a term we use to refer to the coupled ecological-geomorphological system in which feedbacks between biotic and abiotic components occur (Wainwright 2009; Wainwright and Parsons 2010).

Vegetation in drylands is characteristically patchy and is sometimes patterned. The multi-scale, dynamic distribution of vegetation in drylands is an emergent property that depends on complex, poorly understood nonlinear relationships and feedback dynamics between plants, soils and transport processes and human impacts. Regrettably, the disciplines of ecology and geomorphology have largely performed research in isolation of each other, "selectively picking and choosing snippets of information and theory from the other discipline when needed" (Renschler et al. 2007, p. 4). The disciplines are not alone in this behaviour. Figure 1.3 characterizes separate disciplines as having a centrifugal perspective, looking outwards from themselves towards their individual aspects. To address multidisciplinary scientific questions – such as land degradation in drylands – a centripetal approach is required in which the problem is the focus towards which the disciplines direct their attention (cf. Fig. 15.1).

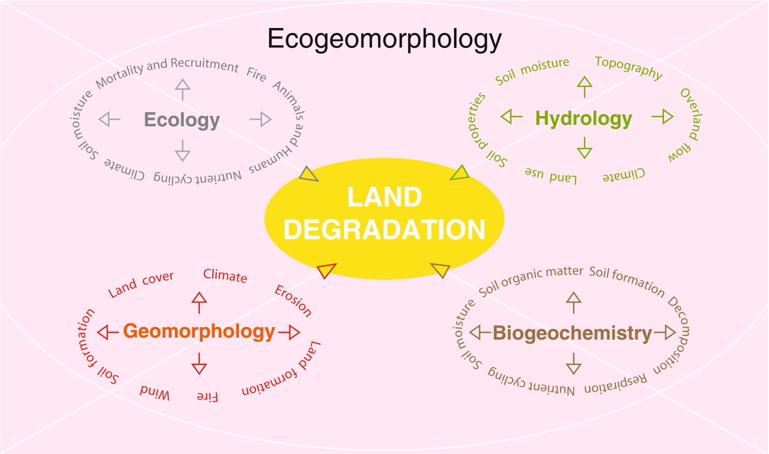

Fig. 1.3 Centrifugal approach of individual disciplines and centripetal approach of ecogeomorphic, multi-disciplinary land-degradation research

The purpose of this book is to take such a centripetal approach towards the understanding of the process linkages between ecogeomorphological dryland processes and patterns to better our understanding of land degradation, and to overcome the lack of interdisciplinarity in current dryland research. To gain a truly interdisciplinary perspective all chapters are multi-authored, drawing together the expertise of ecologists, hydrologists, geomorphologists, mathematicians, biologists, agronomists and remote sensing experts.

The first section of this book, entitled “Theory: linking process to pattern” deals with the establishment of an integrated view of current concepts of pattern formation and self-organization, abiotic and biotic interactions over a continuum of spatial and temporal scales, and process integration from both ecological and geomorphological perspectives. In the second section of this book, entitled “Methods for confronting models with data”, we bring together hitherto divergent methodological approaches to provide a fully dynamic view of the dryland system. We explore innovative ways of modelling ecogeomorphic feedback mechanisms and patterns, and uncertainty assessments are discussed. In the third section, we present four case studies from Europe, Africa, Australia and North America that present the state of the art on understanding ecogeomorphology in different dryland settings. The concluding chapter sets forth a new ecogeomorphological research agenda for land-degradation studies in drylands, which necessitates the further penetration of empirical and disciplinary boundaries and requires the focussing of research efforts between ecologists and geomorphologists and development of common research goals and research approaches.

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.

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Part I
Theory: Linking Process to Pattern

Chapter 2
The Study of Land Degradation in Drylands: State of the Art

Tamara Hochstrasser, James D.A. Millington, Vasilios P. Papanastasis, Anthony J. Parsons, Pier Paolo Roggero, Richard E. Brazier, Joan Estrany, Almo Farina, and Alan Puttock

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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T. Hochstrasser
School of Biology and Environmental Science, Agriculture & Food Science Centre,
University College Dublin, Belfield, Dublin, Ireland
e-mail: tamara.hochstrasser@ucd.ie

J.D.A. Millington
Department of Geography, King's College, London WC2R 2LS, UK
e-mail: james.millington@kcl.ac.uk

V.P. Papanastasis
Laboratory of Rangeland Ecology, Aristotle University, 54006 Thessaloniki, Greece
e-mail: v.papan@for.auth.gr

A.J. Parsons (✉)
Sheffield Centre for International Drylands Research, University of Sheffield,
Sheffield S10 2TN, UK
e-mail: a.j.parsons@sheffield.ac.uk

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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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P.P. Roggero
Desertification Research Centre and Dipartimento di Agraria, Desertification
Research Centre and Dipartimento di Agraria, 07100 Sassari, Italy
e-mail: pproggero@uniss.it

R.E. Brazier
Department of Geography, College of Life and Environmental Sciences, University of Exeter,
EX4 4RJ Exeter, UK
e-mail: r.e.brazier@exeter.ac.uk

J. Estrany
Department of Earth Sciences, University of the Balearic Islands, 07122 Palma de Mallorca, Spain
e-mail: joan.estrany@uib.cat

A. Farina
Instituto di Ecologia e Biologica Ambientale, University of Urbino, 61029 Urbino, Italy
e-mail: farina@uniurb.it

A. Puttock
College of Life and Environmental Sciences, University of Exeter, EX4 4RJ Exeter, UK
e-mail: a.puttock@exeter.ac.uk

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.

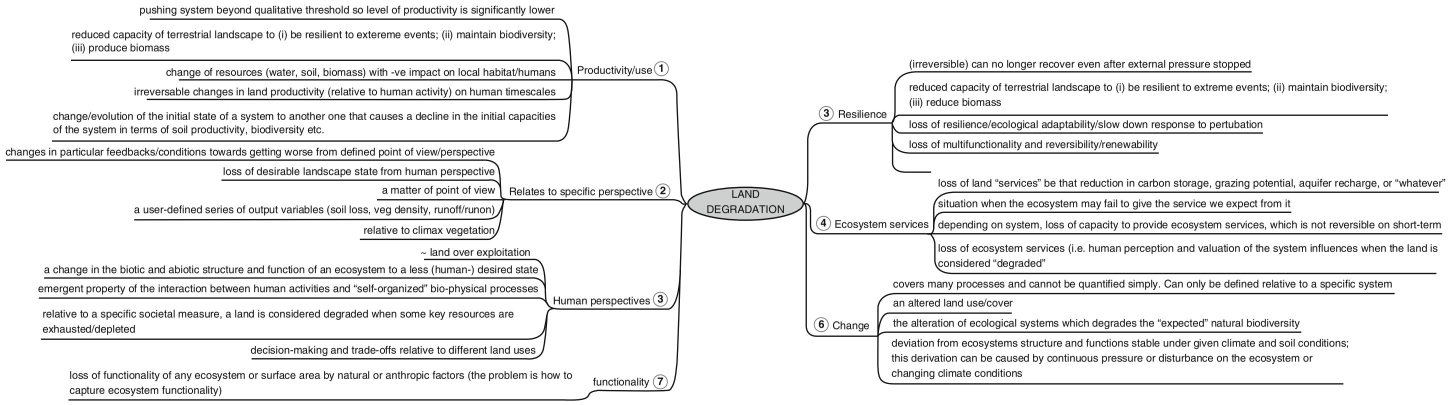

Fig. 2.1 Mind map of the concept of land degradation

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

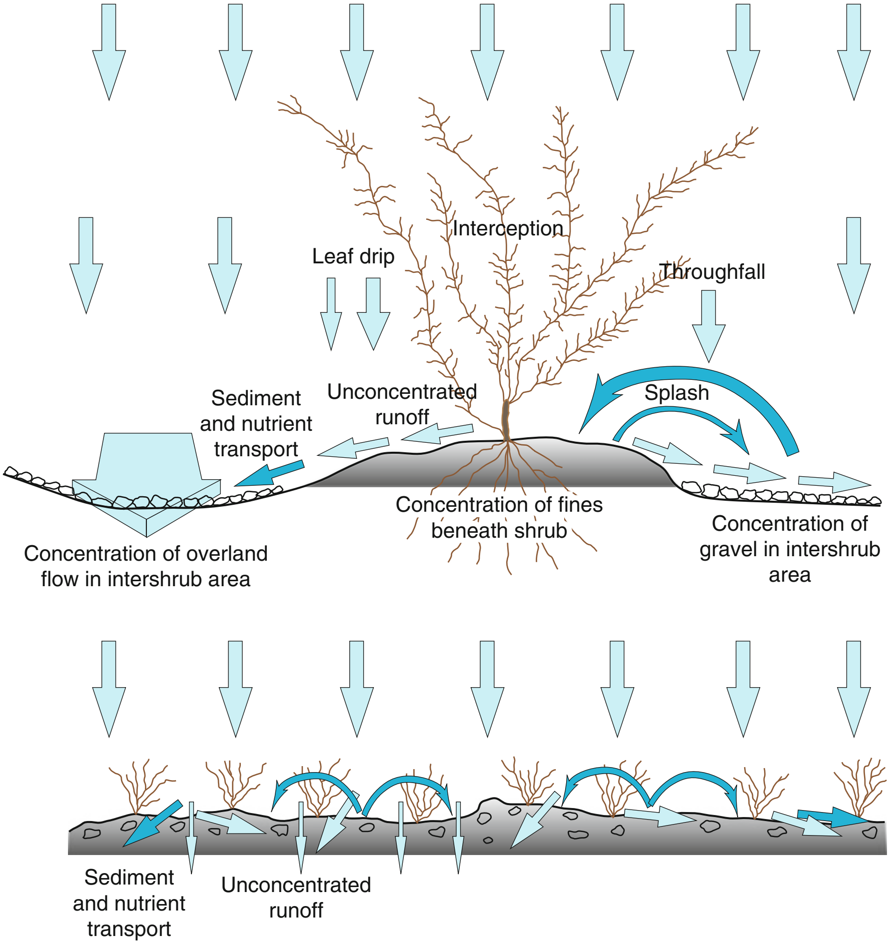

Fig. 2.2 Interactions between Islands of Fertility and water movements and erosion. Upper diagram shows runoff pattern in shrubland; lower diagram shows runoff pattern in grassland. (After Wainwright et al. 2000)

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.

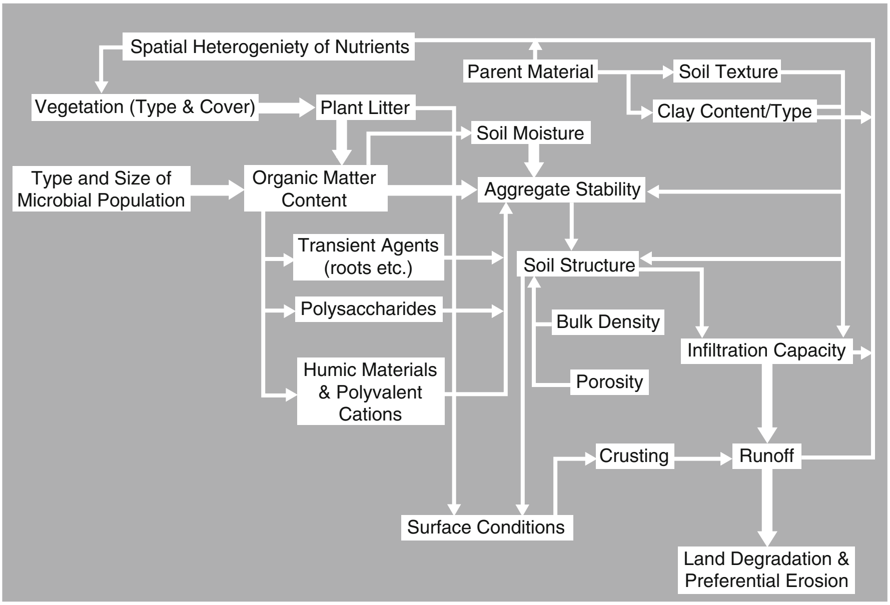

Fig. 2.3 A conceptual model depicting the linkages between soil physical and chemical indicators of land degradation (After Dickie 2006, reproduced by permission of the author)

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. Brunsden 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 ecogoemorphological 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).

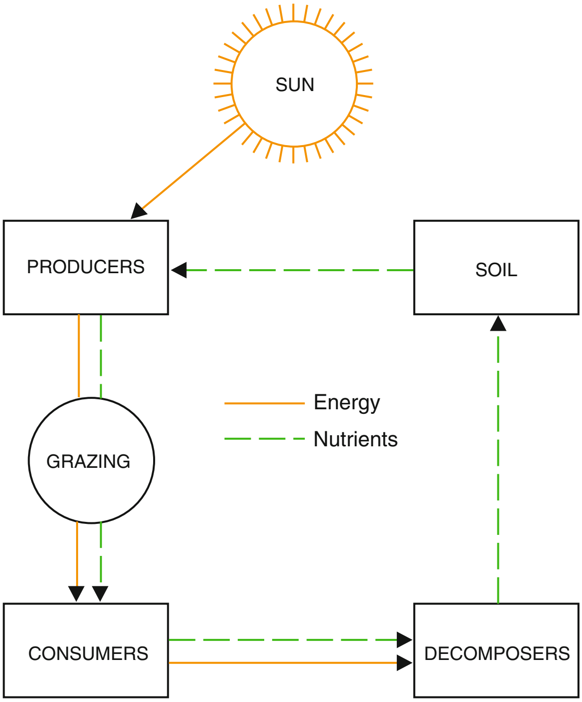

Fig. 2.4 The rangeland ecosystem

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 C_4 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 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 inspite 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 kg-m^{-2} as opposed to a typical level of 10–18 kg-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

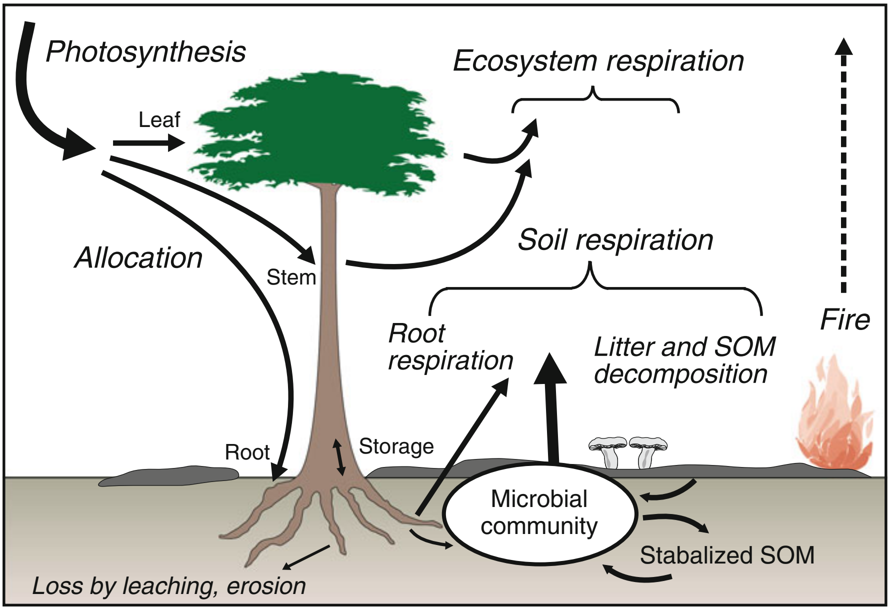

Fig. 2.7 Pathways of carbon flow through ecosystems (Reproduced from Trumbore (2006) with permission of John Wiley & Sons)

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 adaption 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, phosphoglyeraldehyde (Whitford 2002). C_4 photosynthesis (Fig. 2.8b), hypothesised to be an evolutionary adaption to low atmospheric CO_2 levels (Sage 2004; Osborne and Beerling 2006) involves the combining of CO_2 with phosphoenolypyruvate, to form four carbon molecules,

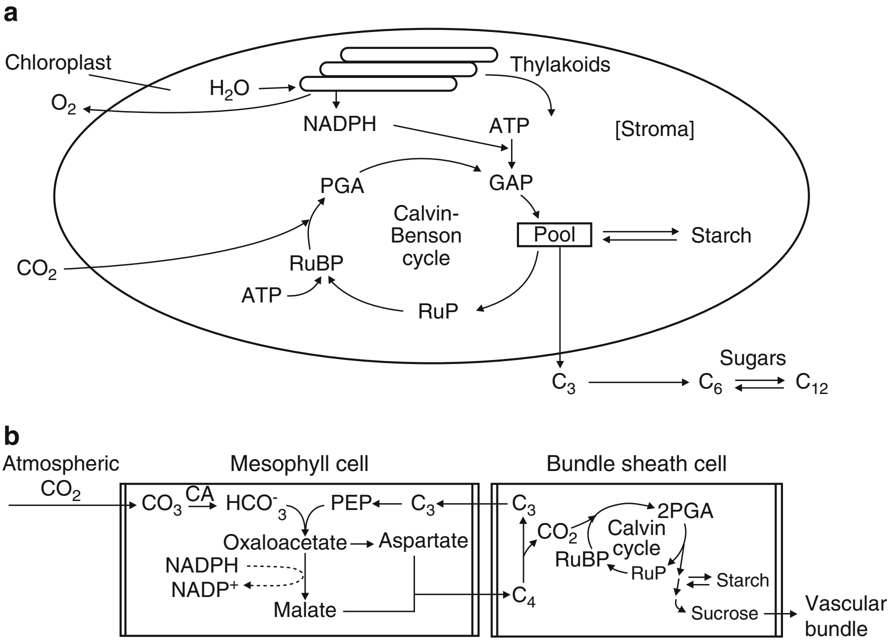

Fig. 2.8 Different photosynthesis pathways in (**a**) C_3 plants and (**b**) C_4 plants (Reproduced from Larcher 1995, with kind permission of Springer Science+Business Media)

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

	C_3	C_4	CAM
Light saturation	¼ 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).

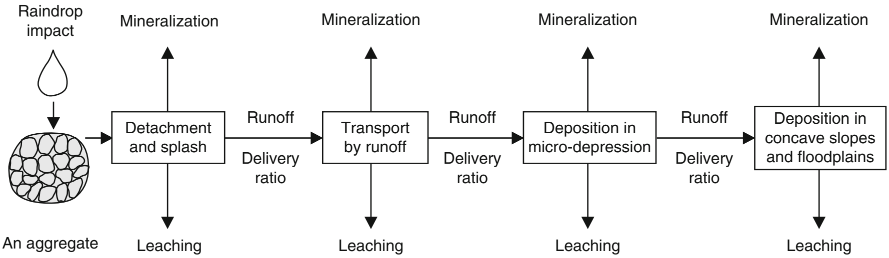

Fig. 2.9 Fate of eroded soil organic carbon (Reproduced from Lal 2005, with permission of Elsevier)

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 ± 0.5 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 (calcic 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 0.3–0.5 Pg C year^{-1} for desertification control (Lal et al. 1999) and 0.4–1.0 Pg C year^{-1} for reclaiming salt affected soils (Lal 2010). The water limitation of dry croplands constrains SOC sequestration to a range of 0–200 kg ha^{-1} 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 CO_2 rise on N_2 fixation, denitrification and leaching are also worth to be investigated further, to quantify the potential positive feedbacks between CO_2 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

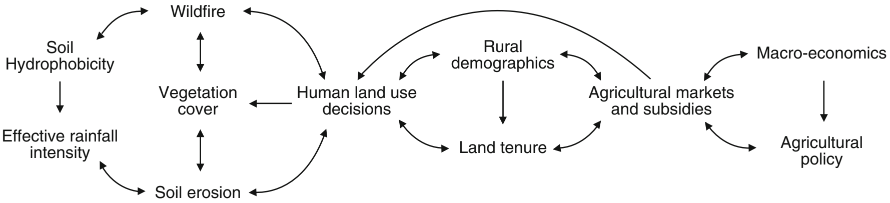

Fig. 2.10 Multiple process feedbacks exist between ecological, hydro-geomorphological and social processes. *Arrows* represent influences. Although effects of individual interactions may be well understood, reciprocal feedbacks – including human activity – are less well understood

(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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Chapter 3
Resilience, Self-Organization, Complexity and Pattern Formation

Florian Jeltsch, Laura Turnbull, Stefania Scarsoglio, Concepcion L. Alados, Francesc Gallart, Eva Nora Mueller, Nicolas Barbier, James D.A. Millington, John Wainwright, Mareike Wieczorek, and Volker Grimm

Abstract Clarity of definitions is fundamental to the successful completion of any interdisciplinary project. In this chapter, we focus on defining a number of key terms that recur throughout the volume, and thus it acts as both a foundation and glossary for understanding the material covered. Ideas of resilience, self-organization and complexity are widely used across in constituent disciplines discussed in Chap. 1, but their use varies and we attempt to define this variation and thus the context of use in the book. There is also a strong emphasis on pattern, so we provide here an initial definition, to be followed up in more detail in Chaps. 7 and 8. We then move onto the specific nature of drylands and the need to understand land

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F. Jeltsch
Plant Ecology and Nature Conservation, University of Potsdam, 14469 Potsdam, Germany
e-mail: jeltsch@uni-potsdam.de

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

S. Scarsoglio
Dipartimento di Idraulica, Trasporti ed Infrastrutture Civili, University of Turin, 10129 Torino, Italy
e-mail: stefania.scarsoglio@polito.it

C.L. Alados
Pyrenean Institute of Ecology, 50080 Zaragoza, Spain
e-mail: alados@ipe.csic.es

F. Gallart
Institute of Earth Sciences, Jaume Almera (CSIC), 08028 Barcelona, Spain
e-mail: francesc.gallart@idaea.csic.es

E.N. Mueller (✉)
Institute of Earth and Environmental Science, University of Potsdam, 14476 Potsdam, Germany
e-mail: eva.mueller@uni-potsdam.de

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degradation within them and through them. How the mode of study affects our understanding is the next theme, in particular how case studies based in different places can be generalized, given the variations in landscape and vegetation type within them. Following an initial summary of the preceding material to evaluate why self-organization and complexity are useful frameworks for understanding patterns and processes in drylands, the chapter concludes with an overview of the deterministic and stochastic frameworksfor understanding pattern.

3.1 Introduction

One issue that arose in Chaps. 1 and 2 is that of terminology and the ways in which researchers working from different disciplinary perspectives can often use the same or similar terms to mean quite different ideas or things. These differences in language can therefore have significant effects on both the ways research is carried out (e.g. Wu and Hobbs 2002), interpreted, and in the ways in which research results are communicated to stakeholders (e.g. Webb and Raffaelli 2008). These difficulties are prevalent in land-degradation studies, which involve researchers from a wide range of disciplines working on an applied topic; it was certainly a topic that cropped up on numerous occasions during the workshop from which this book was developed. These difficulties are compounded by the need to apply the results of rapidly developing research topics across a number of fields, and thus, this chapter starts off by defining what we mean by a number of key terms such as resilience, self-organization, complexity and pattern. It will become clear in doing so that there are nuances in the definitions of all of these terms, and so appreciating these nuances will be critical for understanding the rest of the book.

N. Barbier
IRD/UMR AMAP, Botany and Computational Plant Architecture, 34398 Montpellier, France
e-mail: nicolas.barbier@ird.fr

J.D.A. Millington
Department of Geography, King's College London, Strand, WC2R 2LS London, UK
e-mail: james.millington@kcl.ac.uk

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

M. Wieczorek
Department of Geosciences, Alfred Wegener Institute for Polar and Marine Research,
14473 Potsdam, Germany
e-mail: mareike.wieczorek@awi.de

V. Grimm
Department of Ecological Modelling, Helmholtz Center for Environmental Research UFZ,
04318 Leipzig, Germany
e-mail: volker.grimm@ufz.de

We then move on to more specific considerations of how we apply these terms to dryland contexts, and what specific characteristics there are of drylands that require caution in applying the general concepts of resilience, self-organization, complexity and pattern. At the same time, it is important to evaluate how one translates general concepts to specific contexts, not least so that when testing ideas, comparisons are appropriate, and avoiding, as it were, throwing the (conceptual) baby out with the (empirical) bathwater. Thus, we consider the nature of dryland landscapes, and the variability of vegetation in the world's drylands. Then, we move on to a consideration of how the concepts can be applied, and what general frameworks there are for evaluating pattern from deterministic and stochastic perspectives. Finally, we step back to consider the implications of these ideas from broader, holistic frameworks and in relation to the processes affecting dryland environments.

3.2 Definitions

3.2.1 What Is Resilience?

For centuries, natural scientists have been fascinated by the idea of stability of natural systems (see MacArthur 1955; Pimm 1991). However, 'stability' itself is an ambiguous term that is poorly defined in many fields of natural sciences and often leads to confusion rather than to understanding (e.g. Holling 1973; Grimm and Wissel 1997). As a consequence, scientists use other terms to describe certain properties of stability, such as constancy ('staying essentially unchanged', e.g. Connell and Sousa 1983), persistence ('persistence through time of a natural system', e.g. DeAngelis 1980) or resilience ('returning to a reference state – or dynamic – after a temporary disturbance', e.g. Harrison 1979; definitions after Grimm and Wissel 1997). Two types of resilience are commonly referred to that reflect two different aspects of stability. The first, termed engineering resilience (Holling 1973) focuses on stability near an equilibrium steady state, whereby resilience refers to the speed at which a system returns to equilibrium or steady state following disturbance (O'Neill et al. 1986; Pimm 1991). In contrast, ecological resilience moves beyond the time of return to equilibrium or steady state, in recognizing that a system may never return to its pre-disturbance state because of the possibility of multiple stable states where instabilities can flip a system into another state (Holling 1973; Walker et al. 2004). Therefore, ecological resilience is the magnitude of disturbance that can be absorbed before the structure of the system is changed such that it represents a new state.

Though these terms – at a first glance – seem to be more precise than just 'stability', definitions still vary among studies due to differing concepts (e.g. Grimm and Wissel 1997; Washington-Allen et al. 2008). Despite the possible impression that this is merely a semantic problem, the need to identify, quantify and compare basic stability features of natural systems remains. Given the overarching importance of disturbances for the dynamics of natural systems and their increasing vulnerability (e.g. Reynolds et al. 2007), ‘resilience’ has become the key stability property in this context (e.g. Elmqvist et al. 2003; Washington-Allen et al. 2008). However, the often-used definition of resilience as ‘*the amount of disturbance a system can absorb and still remain within the same state or domain of attraction*’ (Holling 1973, 1996; cited after Elmqvist et al. 2003) shows that it is of crucial importance to, in addition, clearly define the focal reference state or dynamics of the system (e.g. fluctuations within defined ranges) and the type of disturbance relevant for resilience (e.g. land use, climatic extremes). A key element of Holling’s (1973) notion of resilience is that the ability to “absorb” disturbances is limited. Once the capacity of absorbing, or buffering, mechanisms is exhausted, ecosystems may lose their structure and functions abruptly, rather than gradually.

3.2.2 What Is Self-Organization?

The notion of self-organization seems to have first developed in the fields of thermodynamics and cybernetics in the 1950s (Heylighen 2003). In the former, Prigogine and co-workers considered the idea of a dissipative system, in which a system uses energy to build structure and thereby exporting entropy in order to conform with the Second Law of Thermodynamics (Nicolis and Prigogine 1977). Thus, the system switches state (becomes organized) by its intrinsic behaviour, and can remain far from equilibrium. From a cybernetic perspective, this organization is defined in statistical terms, based on the probability that a particular system finds itself in a particular state. Statistical entropy then produces the definition that “every system tends to its most probably state” (Beer 1966, cited in Gershenon and Heylighen 2003), which is “a tautological law of nature, since the probabilities of the states are determined by us according to the tendencies of systems” (Gershenon and Heylighen 2003: 607). This definition would also not be able to predict future states that had never been observed (however likely they were under specific driving factors of a system). For these reasons, the cybernetic approach has not been widely taken up in the environmental sciences. More recent developments of the idea of self-organization have arisen from the ideas of complex adaptive systems (e.g. Bar Yam 1997; Kauffmann 2000). Such definitions have tended to arise from computational approaches where systems are represented as sets of discrete elements rather than Newtonian continua. Simulations using cellular automata (Wolfram 2002), neural networks (Oster 1988) or individual- or agent-based models (Grimm and Railsback 2005) demonstrate that characterizing the system based on local interactions can show self-organization of system macrostates that are not predictable from the individual system components.

A related term to self-organization in this context – often used synonymously – is that of emergence. Bedau (1997) has particularly emphasized the role of simulation in what has been called “weak emergence”, although he has subsequently expanded the definition to include all approaches to explanation. The difference between weak emergence and the systems analyst's "whole being greater than the sum of the parts" is that in the latter, the explanation is always found in the reductionist parts, while in the former, there is no direct means of providing such explanation. Thus, an important concept within self-organization is that of observer-dependence. The parts of the system that are chosen for specific analysis are chosen by the observer, and the perspective of the observer can alter whether the system in question is self-organized or disorganized (Gershenon and Heylighen 2003).

The thermodynamics approach to self-organization received a lot of interest in the 1980s as a way of explaining the development of patterns across physical, biological and chemical systems (e.g. Jantsch 1980; Haken 1983; Krinsky 1984). Kay and Schneider (1994) renewed interest in the concept from an ecosystems perspective, and Kay et al. (1999) subsequently outlined four principles of self-organization:

1. Self-organized systems capture more resources (energy and material);
2. Self-organized systems make more effective use of resources;
3. Self-organized systems build more structure; and
4. Self-organization increases resilience of the system.

These principles show how the idea of self-organization links to that of resilience (see above), complexity (see below) and adaptation. Simulations of self-organized systems show that they are typically tolerant to faults or noise (Bar Yam 1997), and thus perhaps explain why such systems tend to be resilient to external forcing, which is a characteristic of whatever definition of self-organization is used.

A particular form of self-organization is known as self-organized criticality (SOC), after the work of Bak (1996). Patterns are considered to have SOC if their log magnitude is inversely related to the log of their frequency or they show the presence of $1/f$ noise. Some recent studies (e.g. Kéfi et al. 2007) have suggested that SOC is a "normal" characteristic of dryland ecosystems, and that deviations from SOC represent degradation. Beyond the conceptual and statistical problems of evaluating SOC robustly (Frigg 2003; Lise and Paczuski 2001), the fact that such organization seems to be the exception rather than the rule (Cross and Hohenberg 1993), and the anthropocentric view on which this untested assumption is based, it assumes that dryland ecosystems are typically in equilibrium, which is a view that has been increasingly challenged over the last few decades (e.g. Wiens 1984; Sullivan and Rohde 1999; Vetter 2005; and the discussion about self-organization and far-from equilibrium conditions above).

3.2.3 What Is Complexity?

The word 'complexity' has different meanings in different disciplines, but is viewed here in a way compatible with statistical physics. A complex system is therefore a collection of microscopic parts (e.g. particles, individual plants, sediment grains).

The individual fate of these parts is intractable at system scale, because each individual's trajectory is too contingent. System properties cannot be predicted from the properties of the parts (emergence) (see also Chap. 6). However, because individuals interact only with their nearest neighbours (at the local scale) via simple positive or negative non-linear effects (feedbacks), the system as a whole may, when maintained far from thermodynamic equilibrium, display predictable emergent properties at the macroscopic level, notably in the form of ordered patterns in space and/or time. In other words, complex systems may self-organize and generate orderliness, in spite of their large number of components, and the absence of an explicit plan or blueprint. Therefore complexity, contrary to its usual meaning of complication (see also the discussion in Mulligan and Wainwright 2012), concerns the study of systems whose behaviour can be decomposed (and predicted) using a small number of simple rules. The state of the system, i.e. its emergent properties, at any time is modulated by external constraints (boundary conditions), but it also depends on previous states. Striking examples in ecogeomorphic systems include the formation of sand ripples and dunes, of the formation of spatially periodic vegetation patterns in semi-arid areas. In fact, since the pioneering works of Turing (1952) or Prigogine (e.g. Nicolis and Prigogine 1989), examples of such complex behaviours have been evidenced both theoretically and empirically, in a variety of biological (Murray 2002; Solé and Bascompte 2006), physical and chemical systems (Cross and Hohenberg 1994).

3.2.4 When Is a Pattern a Pattern?

In science, patterns are observations of any non-random structure. In ecology for example, a pattern has long been understood as the "structure which results from the distributions of organisms in, or from, their interactions with their environments" (Hutchinson 1953, p. 3, also see Watt 1947; Greig-Smith 1979). However, when identifying patterns in nature, scientists more precisely mean the identification of patterns in data about nature. Important considerations for identifying patterns, therefore, are the means by which data were collected, and most importantly the scales of measurement used to collect data. In particular, two components of scale – grain and extent – are important in determining whether a pattern is identified. Grain is the resolution of measurement (i.e. the smallest unit of measurement at which objects or states can be distinguished), whereas extent is the full scope of observation or total range over which measurements are made. As examples, different spatial patterns will be detectable in maps of vegetation configuration in semi-arid areas depending on the grain and extent of the maps (e.g. compare Figs. 3 and 6 in Barbier et al. 2006; see also Chap. 13), and different temporal patterns will be detectable in storm hydrographs depending on the resolution and duration of measurement (e.g. compare runoff for 10-min intervals with full 80-min duration, and observed runoff with simulated drainage, in Fig. 5 of Mueller et al.

2007). In other circumstances, observed structures may be described as being ‘scale-free’. These structures lack a characteristic length scale and have the same properties across any grain and extent of measurement (e.g. power-law distributions of vegetation patch sizes: Kéfi et al. 2007). These scale-free structures can also be considered to be patterns.

Because patterns are non-random, they have the potential to provide information. In natural science this information is usually understood as being about the processes that caused the pattern. Thus, identifying patterns is useful because they can be used to investigate processes (Levin 1992). Processes are typically assumed to act at a different scale from the patterns they produce, with patterns either emerging from processes at smaller scales (‘bottom-up’ processes) or imposed by constraints at larger scales (‘top-down’ processes). It is also important to consider the reciprocal effects of patterns on processes (Turner 1989). For example, the field of landscape ecology has placed an emphasis on the quantification of spatial pattern using pattern metrics (e.g. McGarigal 2006) and shown how the history of previous ecological processes can increase the strength and extent of spatial pattern (Peterson 2002). The ‘pattern-oriented modelling’ (POM) approach has been developed to use models to help decode the information present in patterns to better understand processes (Wiegand et al. 2003; Grimm et al. 2005) (see also Chaps. 7 and 9). The POM approach iteratively compares empirical and model-output patterns at multiple scales and levels of organization and for multiple models to identify most appropriate models. Approaches like POM, which place pattern at the centre of scientific investigation, are vital for improving understanding about physical processes.

3.3 What Is Specific About Drylands in Relation to Resilience and Environmental Change?

Worldwide, drylands are the regions that will likely be most affected by climate change, given their existing exposure to drought and the predicted changes in rainfall volume and distribution (IPCC 2007). Today, there is already an increase in the extent of dryland areas affected by more intense and longer droughts or higher rainfall variability (IPCC 2007), often accompanied by increasing temperatures and higher levels of atmospheric CO_2. These environmental changes further increase the inherent risk of desertification, i.e. the “land degradation in arid, semi-arid, and dry subhumid areas resulting from various factors, including climatic variations and human activities” (as defined in the UNCCD). Desertification is closely linked to high overexploitation of ecosystem services indicated by physical, biological, and social phenomena including soil erosion, reduced biodiversity and biological productivity but also by high population growth rates and unsustainable land management (Millennium Ecosystem Assessment 2005). It is accompanied by reduced income of the human population and the highest rate of infant mortality and hunger among children worldwide (Millennium Ecosystem Assessment 2005, Chapter 22). In addition, the fact that:

1. drylands cover more than one third of the of Earth's land surface and are home to more than one third of the total global population (Reynolds et al. 2007); and
2. dryland populations on average lag far behind the rest of the world on human well-being and development indicators (Millennium Ecosystem Assessment 2005),

emphasizes the need to focus more research on the resilience of dryland systems and challenges posed by desertification.

A key challenge in dryland research related to resilience and environmental change is the question for the reference state of the system. It follows from the discussion above (Sect. 3.2.2) that clearly definable equilibrium conditions can be defined and that drylands are non-equilibrium systems with different possible steady states and different domains of attraction (Reynolds et al. 2007). In non-equilibrium systems, nonlinear feedbacks, time-lags (e.g. in the interaction of species or processes) or stochastic forcings by a fluctuating environment cause continuous changes to the system. These influences may eventually drive the system across its boundaries (DeAngelis and Waterhouse 1987). Consequently, threshold dynamics are prominent features of dryland systems (e.g. Weber and Jeltsch 2000; Rietkerk et al. 2004; Safriel and Adeel 2008). Feedbacks often link abiotic and biotic components (e.g. soil moisture and vegetation) and thus typically require interdisciplinary research. While multiple steady states and thresholds already make it difficult to define a reference state of the system for resilience analyses, some authors even challenge the idea of steady states. Instead they focus on buffering mechanisms that keep the dryland system within certain system boundaries, without distinguishing certain states within these boundaries (Jeltsch et al. 2000). A defining feature of buffer mechanisms is their limited capacity. For example, buffer capacities for savannas are often large enough to maintain overall savanna structure and functions for hundreds of years despite wide fluctuations in environmental drivers. Buffer capacities can, however, be exhausted by too intensive land use, by environmental changes passing certain threshold, or by a combination of these two factors.

A further complicating feature in the dynamics of drylands is distinguishing 'pattern from noise', i.e. to identify clear long-term trends in variables that can be used to describe the state of the system (e.g. vegetation cover) in spite of the noise caused by short term responses to the highly variable environmental drivers such as rainfall (Miehe et al. 2010; Reynolds et al. 2007). The identification of long-term rangeland degradation for example is strongly limited by the difficulty to separate the effects of climatic events, such as drought, from the effects of anthropogenic activities, such as commercial livestock grazing. Long-term monitoring (e.g. using remote sensing techniques, Washington-Allen et al. 2006) as well as process-based modelling, that allows the extrapolation of available information on short-term processes and mechanisms to longer temporal scales (e.g. Jeltsch et al. 1999), are thus key elements of resilience analyses in drylands.

The problem of temporal scales is further linked to a problem of spatial scales: dryland areas do not only experience substantial differences in rainfall, both, within and between years, but rainfall is also highly variable over short geographic ranges. In combination with positive feedbacks between vegetation cover (or composition) and soil moisture, this often leads to self-organized ecosystem patchiness, differing in scale and shape (Rietkerk et al. 2004). These spatial patterns are closely linked to dryland resilience, both influencing and indicating fragility and threshold proximity (Bailey 2010; Rietkerk et al. 2004; Scheffer et al. 2009).

Last but not least, the high dependence of human well-being on the sustainable provision of ecosystem services in drylands shows that ecological and social issues are closely interwoven. Thus, any research aiming at conservation or restoration of dryland resilience has to deal with both, natural and social science (Safriel and Adeel 2008; Stringer et al. 2009). Given the separate scientific background and traditions in the related disciplines, this poses a challenge that goes beyond the problems of mere interdisciplinary research. As a first step towards integration, an overall conceptual framework for the investigation of environmental and socio-economic problems in drylands in an interdisciplinary way was recently proposed by Reynolds et al. (2007), which they call the Dryland Development Paradigm (DDP). This approach offers a comprehensive, formal framework to analyse desertification, including human-environment interactions, local knowledge and the consideration of system resilience. While this framework helps to ask the right questions it is still a long way to find the right answers to the threats posed by environmental changes in drylands.

3.4 The Rôle of Place: How Understanding Is Controlled by Different Landscapes

Any point on the land surface may exchange water and sediments with the surrounding points. In flat landscapes, this interaction is usually of short range and driven mainly by wind and raindrop energy on the ground surface and by capillary forces underground. In sloping areas, the lateral component of gravity affords the main driver for water and sediment redistribution, giving place to the adage that 'any land point belongs to a catchment': it receives water and matter from upslope points and yields them to downslope points.

Nevertheless, the nature and importance of these exchange processes largely differ between wet and dry areas. In wet landscapes, the high density and continuity of the vegetation cover limits the role of raindrop and wind processes, whereas the redistribution of water takes place mainly underground because of the low rainfall intensities and the structure of soils that promotes infiltration. Consequently, Beven and Kirkby (1979) argued that the spatial patterns of subsurface-water redistribution are controlled by the local slope as modified by the upslope basin topography (the basis of TOPMODEL, one of the more popular hydrological models), and soil-moisture patterns under these conditions are considered as driven by nonlocal controls (Grayson et al. 1997). Although it may be argued that local controls such as microtopography (Gallart et al. 2008), soil and cover properties modify the global patterns of soil moisture in these environments and provide a challenge even for distributed models (Beven 2000), the role of local controls is far less important than in dry landscapes. Finally, water erosion is usually very limited in natural wet landscapes, the geomorphic activity being usually dominated by chemical processes and mass movements on hillslopes and by sediment movement by channel processes.

Conversely, in dry landscapes, the lack of continuity of vegetation cover allows a major role of wind and raindrop energy, whereas the often higher intensity of rainfall and the poorer condition of soil surfaces favours surface runoff at the expense of infiltration. Areas of bare ground become primary local sources of water and matter whereas covered areas become sinks for both. Furthermore, overland flow is typically discontinuous in dry areas, where the concept of runon infiltration is used by hydrologists to describe the discontinuity of water transfer on hillslopes and 'transmission loss' the discontinuity of water transfer in dry streams. Local variations in gradient, aspect, soil thickness or condition and vegetation cover induce the infiltration of runoff or the return of water infiltrated upslope; spatial patterns of soil moisture are primarily driven by local controls (Grayson et al. 1997; Gómez-Plaza et al. 2001). The discontinuity of water and sediment transfers in dry landscapes provides the biggest challenge to hydrology and soil-erosion modellers but affords the best opportunity to the development of the life because the 'losses' of runoff represent 'gains' for local soil moisture and fertility.

There are particular cases of dry-like landscapes in wet areas and *vice versa*. An example of the first type is the occurrence of densely dissected areas (badlands) in wet environments due to local harsh conditions. These areas are frequent in Mediterranean mountains (Gallart et al. 2001) but the similarity of forms with dry badlands hide significant differences. Climate is sufficiently wet in these areas to allow the rapid development of vegetation and soil cover, but it is checked by the geomorphic activity itself (Regüés et al. 2000). Conversely, well vegetated areas do occur in semi-arid badlands where runoff from degraded surfaces reinfiltrates in soil covered areas as natural water-harvesting systems (Cantón et al. 2002).

The preceding paragraphs have emphasized how critical it is not to interpret dryland processes in terms of dominant concepts derived for humid landscapes (see also Bracken and Croke 2007; Bracken and Wainwright 2008). However, it is also critical to recognize that not all drylands are the same, and thus care must be taken in generalizing from a series of case studies. Drylands vary in terms of climate from the hyperarid to the sub-humid (see Chap. 1) and so they differ in terms of how they respond to climate inputs and variability, especially in the case of threshold processes and thus often the emergence of complexity and pattern. Similarly, drylands in different parts of the world are dominated by different species or functional types as a result of different evolutionary pathways since the break-up of Pangaea. The effects of such differences are covered in more detail in the following section. Furthermore, environmental systems that are sensitive to initial conditions, path dependence and contingency will also vary in their response to perturbations and variability in a way that may not be immediately obvious. Understanding the historical context of studies in different settings is therefore critical. For example, Wainwright and Thornes (2004) evaluate the ways in which changes in intensity of human impacts in Mediterranean environments can produce a whole spectrum of contrasting effects. Extreme caution is thus required in the building and application of general models – such as those considered later in this chapter – for land degradation in drylands. In testing the models, we must be aware that missing aspects of variability may cause differences between the model and observation. In some cases, differences which are usually used to decide that a model is "wrong" may simply mean that the model is incomplete or that it has not been properly applied in a specific context (e.g. by mis-specification of initial conditions or the missing effects of contingency). In our quest for models that are right for the right reasons, we must avoid rejecting ones that are wrong for the wrong reasons.

3.5 The Rôle of Place: How Understanding Is Controlled by Vegetation (Functional) Types/Species

Most of the mechanisms underlying vegetation pattern formation are based on the capacity of plants to concentrate resources around the plant canopy (Bertness and Callaway 1994) and in this way facilitate the growth of patches at expenses of the reduction of nutrients on gaps (Rietkerk et al. 2004). Furthermore, seed dispersal also plays an important role in pattern formation (Pueyo et al. 2008; Lefever et al. 2009). Species with restricted spatial dispersal (reduction of dispersal structures or with anchorage mechanisms), which are common in arid regions. Van Rheede van Outshoorn and van Rooyen (1999) showed a larger degree of spatial self-organization in comparison with developed dispersal species, from which those that disperse by wind present the lowest spatial self-organization (Fig. 3.1). Entomophile shrub species (0.182 ± 0.010) also showed larger spatial self-organization than anemophile (0.089 ± 0.026) species as expected through the influence of the positive feedback between availability of pollen resource and pollinators species that influence plant recruitment because a threshold amount of food is required to attract pollinators (Aizen and Feinsinger 1994; Klinkhamer et al. 1989). However, in spite of the restricted propagation of clonal species, they did not show larger spatial self-organization than nonclonal species (Alados et al. 2010), which must be due to the competitive expansion of ramets from clones, which counterbalance the role of facilitation on self-organized pattern formation. For example, *Stipa tenacissima* when growing in areas with low-angled slopes, which favour its clonal expansion, displace other species around its canopy. However, this displacement does not occur when it grows in slope areas where its competitive ability declines (Alados et al. 2006). In general, long-living species as shrubs present larger degree of spatial self-organization than annual herbs (Fig. 3.2), although species living under the shrub canopy as shortgrass species like *Brachipodium retusum* present similar degree of spatial self organization than shrubs and semi-shrubs species (Fig. 3.2).

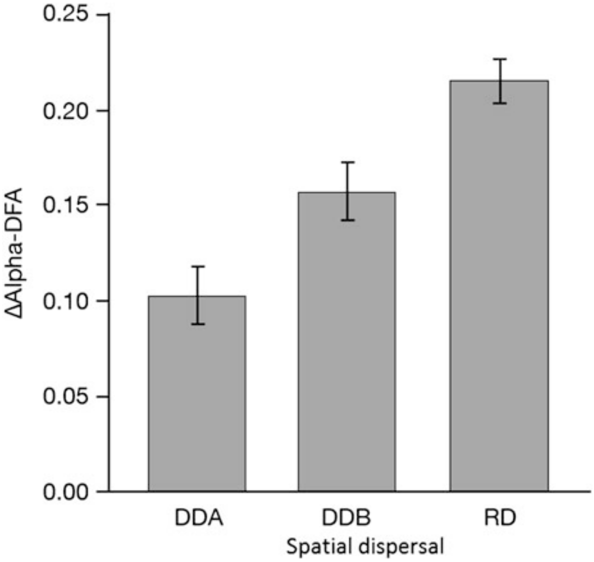

Fig. 3.1 Increases of the mean long-range spatial autocorrelation exponent (Δα-DFA) among shrub species based on their spatial dispersal ability (*DDA* developed dispersal by abiotic vectors, *DDB* developed dispersal by vertebrates, *RD* restricted dispersal) in patches larger than 900 ha in the open shrubland community of Cabo de Gata Natioinal Park (Spain) ($F_{2,80} = 26.78$, $p < 0.001$). Error bars show ± standard error (Modified from Alados et al. (2010) and reproduced by permission of Biological Journal of the Linnean Society)

The decline in self-organization also depends on the species and functional type involved. In general, it has been argued that the characteristic species of a community exhibit more self-organization in well-preserved habitats than in perturbed ecosystems (Alados et al. 2004), and Kéfi et al. (2007) have suggested the breakdown of self-organization as an indicator of degradation.

3.6 Why Are Self-Organization and Complexity Useful Frameworks for Understanding Patterns (and Processes) in Drylands?

Complex systems are defined by the nature of their main components, processes and the dynamical relationships of these components to one another, including inputs, outputs and external constraints (Cummings and Collier 2005). Complex systems thus consist of interconnected or interwoven parts (Bar-Yam 1997) across multiple levels of organization and scale (Wu 1999). Therefore, analysis of complex systems necessitates that relationships and interactions between different components of the system be examined across scales (Ascher 2001). Fundamental properties of complex systems are non-linearity, discontinuity, bifurcation, emergence and self

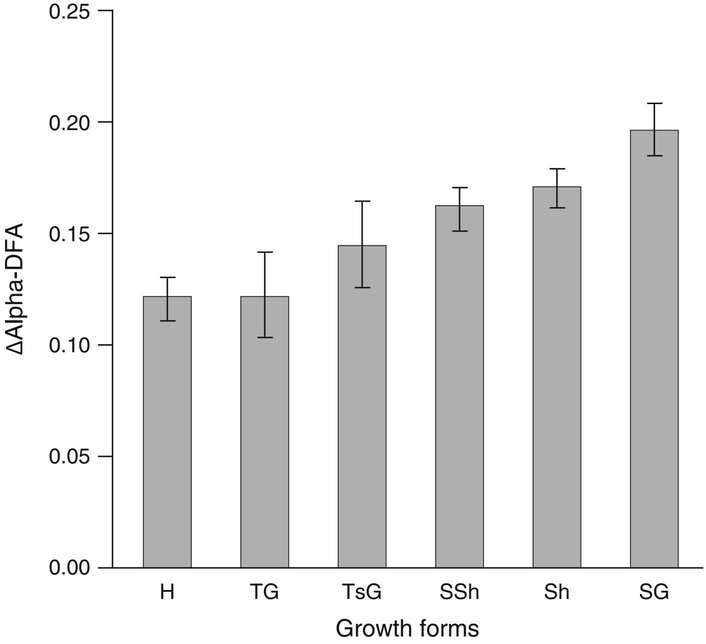

Fig. 3.2 Increases of the mean long-range spatial autocorrelation exponent (ΔαDFA) among species life forms (*H* herbs, *TG* tall grasses, *TsG* tussock grasses, *SSh* semi shrubs, *Sh* shrubs, *SG* short grasses) in patches larger than 900 ha in the open shrubland community of Cabo de Gata National Park (Spain) ($F_{5,300} = 6.53$, $p < 0.001$). Error bars show ± standard error (Modified from Alados et al. (2010) and reproduced by permission of Biological Journal of the Linnean Society)

organization; and they have multiple thresholds and attractors (or states) (Reynolds 2007; Heylighen 2009; Kay 1991; Cadenasso et al. 2006; Ascher 2001; Scheffer et al. 2001; Limburg et al. 2002). Most, if not all complex systems have multiple attractors or (dynamically) stable states that are often self-organized, and can undergo rapid transitions to a new self-organized state following a change in exogenous and endogenous forcing. These multi-state systems can be visualised by means of a stability landscape (Scheffer et al. 2001; Fig. 3.3). Each of these attractors or (dynamically) stable states are characterized by different self-organized structure that optimizes ecosystem function in accordance with the inflow of energy to the system, according to the four principles of self-organization outlined by Kay and Schneider (1994) as discussed in Sect. 3.2.2.

In complex systems, in the absence of a large perturbation that causes a sudden and direct change in ecosystem state, ecosystem-state change and the development of patterns arise from interactions between different components of the ecosystem, in particular, positive and negative feedbacks between biotic and abiotic components of the system (Turnbull et al. 2008; Pickett and Cadenasso 2008; Fig. 3.4).

Drylands exhibit key characteristics of complex systems evidenced by non-linear dynamics and self-organization, and non-linear dynamics, which arise from

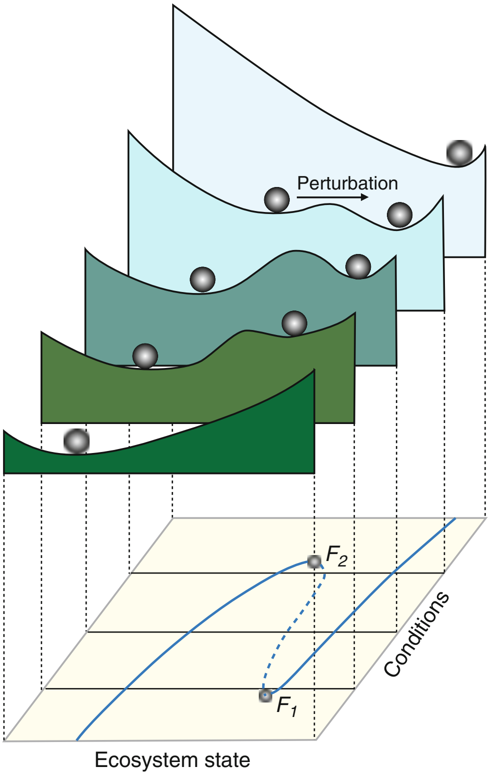

Fig. 3.3 Stability landscape illustrating how perturbations to a system can cause a system to change state in multi-state systems (the state of the system is represented by the location of the ball on the stability landscape). Sufficiently large exogenous forcing can cause a direct change in the state of the system, while mild perturbations can trigger internal feedbacks and cause the system to reorganize into a new stable state (illustrated by a change in shape of the stability landscape). Because of feedbacks, dynamics of state change are non-linear, and often display hysteresis (where a given set of environmental conditions may yield two alternate stable states), as is illustrated in the bottom plane (Source: Scheffer et al. 2001, reprinted by permission from Macmillan Publishers Ltd: Nature, Scheffer et al. 2001)

feedbacks that result in self organization and the formation of patterns over a range of spatial scales (Levin 1992) which has been observed across drylands worldwide (Fig. 3.5). Table 3.1 provides examples of ecological and geomorphic properties of complex dryland systems.

Thus, the evolution of complex dryland systems emerges from the interplay of ecogeomorphic processes occurring across multiple spatial and temporal scales (Levin 2005). Because of these cross-scale interactions, the utilization of reductionist or holistic approaches exclusively is insufficient for the study of patterns in drylands. Reductionist approaches focus on investigating isolated components of systems, and do not seek to investigate interactions between different components of the system. Holistic approaches on the other hand, look at the behaviour of the whole system and seek to identify the simplest explanatory principles of the landscape as a

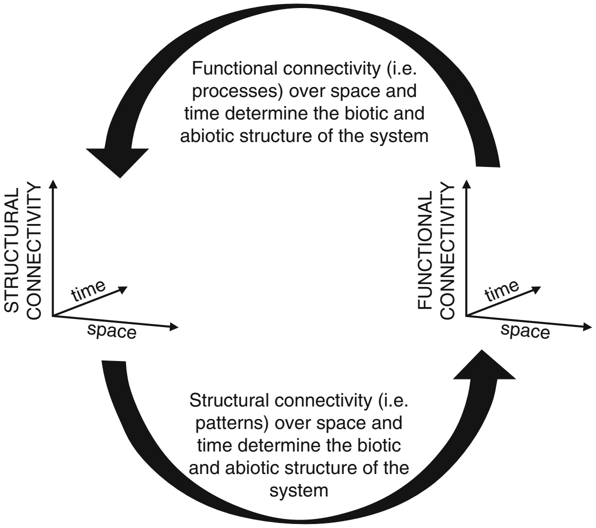

Fig. 3.4 Feedbacks between functional and structural components of the ecosystem lead to the emergence of patterns in drylands (Adapted from Turnbull et al. 2008). *Arrows* show feedbacks between ecosystem structure and ecosystem function. The interactions between ecosystem structure and ecosystem function govern ecosystem dynamics and the resulting formation of patterns in drylands

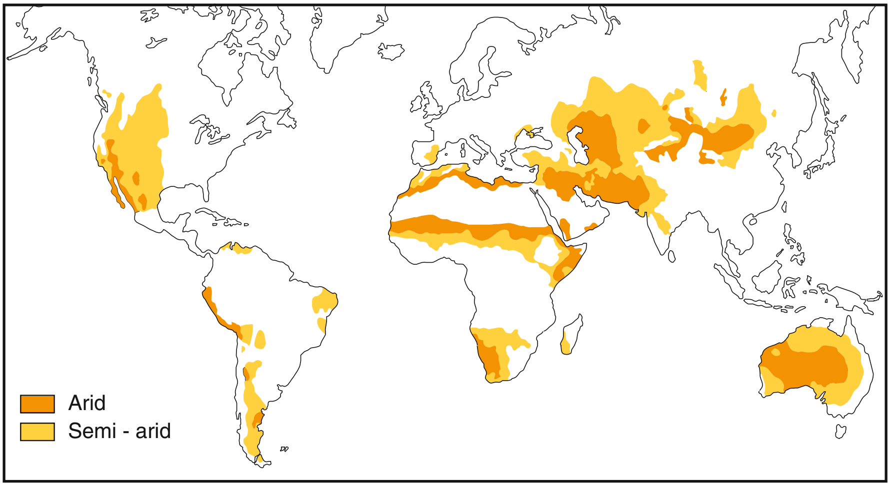

Fig. 3.5 Global distribution of banded vegetation (Source: Valentin et al. 1999) reproduced after permission of Elsevier

whole (Malanson 2004). Thus, frameworks to understand the formations of patterns in drylands that are centred on reductionist or holistic approaches only are limited in terms of the capacity to facilitate understanding of interactions between different levels or organization and across multiple spatial and temporal scales (see further discussion in Chap. 6).

Table 3.1 Ecological and geomorphic examples of common properties of drylands that are associated with complex systems

Property	Ecological example	Hydrological example
Self-organization	Microclimate effects; ecosystem engineering, soil development; vegetation patterns	Drainage network; flow path development
Threshold	Xylem embolism; leaf out; leaf fall; birth/mortality	Flooding, throughfall; snowmelt; infiltration-excess runoff; sediment entrainment
Adaptation	Species shifts; phenotype plasticity	Erosion; deposition
Nonlinearity	Temperature response curves, growth curves; plant uptake of soil moisture	Soil-water retention curves; runoff generation; preferential flow;
Irreversible	Extinction	Erosion; weathering
Scale-free behaviour	Patch distributions, patch boundaries	Stream networks
Scale-dependent behaviour	Boundary layer conductance	Dispersion
Legacy	Organismal regulation, reproductive effort	Antecedent soil-moisture conditions, soil properties; landscape configuration
Hysteresis	Nutrient/resource patches	Soil-water retention
	Soil respiration; delays in delivery of photosynthetic products to the roots due to phloem transport time	
Feedbacks	Growth	Erosion

Adapted from Jenerette et al. (2012)

In dryland ecosystems, complex interactions between plants, soils and climates and properties outlined in Table 3.1 in dryland ecosystems often make it difficult to define the specific processes that underlie observed spatiotemporal patterns of ecosystem structure (Caylor et al. 2009). Thus, in order to understand the observed patterns in dryland ecosystems and the processes that drive their spatiotemporal variability, a systems perspective is necessary that accounts for biotic-abiotic feedbacks over multiple space and time scales. A framework aimed at understanding self-organized patterns in drylands must be sufficiently broad to enable the inherent complexity of interactions between patterns and processes to emerge, by accounting for the biotic and abiotic structure of the ecosystem, ecosystem processes that connect different biotic and abiotic components of the ecosystem, and feedbacks between the structural and functional components of the ecosystem (Turnbull et al. 2008).

The conceptual framework for understanding complexity in systems outlined in Cadenasso et al. (2006) has three primary axes: (1) Heterogeneity, (2) Connectivity, and (3) Contingency (Fig. 3.6). Self-organization emerges within complex systems as a result of dynamics associated with heterogeneity, connectivity and contingency, which together determine the dynamics and characteristics of pattern formation.

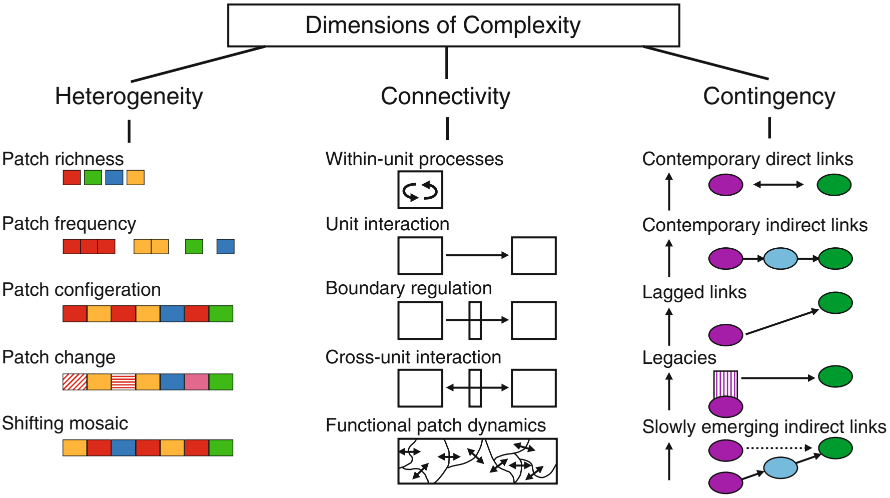

Fig. 3.6 Framework for understanding complexity (Cadenasso et al. 2006). The three dimensions of complexity are spatial heterogeneity, organizational connectivity and temporal contingencies. Components of the framework are arrayed along each axis increasing in complexity (Source: Cadenasso et al. 2006) reproduced after permission of Elsevier

The complexity framework, as outlined in Fig. 3.6, is ideal for understanding the formation of patterns and state change in drylands since it explicitly incorporates the structure of the system (*heterogeneity*), the interaction of processes across different spatial scales (*connectivity*) and changes in ecosystem processes to optimize uptake and utilization of resources and materials, in response to changes in input conditions (as per Kay 1994), and interactions across different spatial and temporal scales and variations in response times (*contingency*). Implicit in this conceptual framework is the non-linearity that arises out of system feedbacks and cross-scale interactions.

In self-organizing systems such as drylands, patterns develop as an optimized response to climatic and landscape conditions, in ways that reinforce energy use (Odum 1988) to maintain the self-organized state (Kay et al. 1999). A striking example of systems developing as an optimized response to climatic and landscape conditions is given by Valentin et al. (1999) who show how the characteristics of banded vegetation alter in response to precipitation and slope (Fig. 3.7). In systems that exhibit a higher degree of self organization, a greater amount of available energy is retained within the system (Odum 1988). This type of organizational complexity increases system resilience and the capacity of these systems to adjust to shifting external and internal conditions (Holling and Gunderson 2002). Because of this resilience, self-organized systems tend to be highly stable and can maintain the same structure even when the system is perturbed to some extent. However, if the system is perturbed to a large enough extent and the resilience of the self-organized state is exceeded, the system may switch to alternate stable state that will likely feature a different pattern that represents a different structure that optimizes the capture

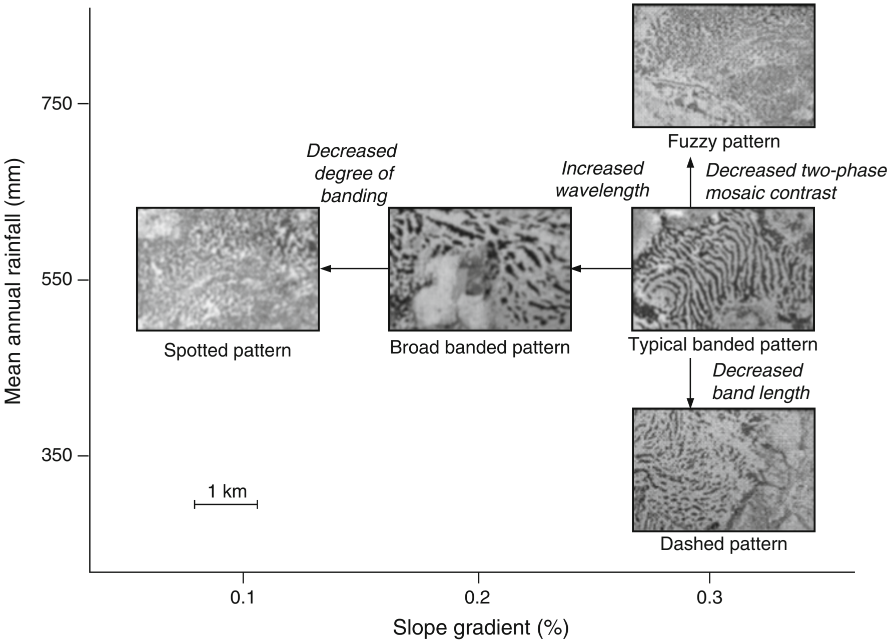

Fig. 3.7 Characteristics of banded vegetation patterns with changes in slope and mean annual precipitation (Source: Valentin et al. 1999) reproduced after permission of Elsevier

and utilization of available resources. The process of state change and ecosystem restructuring leading to the formation of patterns in drylands following exogenous or endogenous forcings are typically non-linear and often hysteretic because of strong positive feedbacks (Turnbull et al. 2012). The dynamics of ecosystem state change in response to external drivers and disturbances can be conceptualized by means of the cusp-catastrophe model, where the dynamics of state change are determined by the strength of positive feedbacks (in relation to the strength of opposing stabilizing feedbacks) that prevail within the system (Fig. 3.8).

3.7 Deterministic Frameworks for Understanding Pattern

Deterministic mechanisms of pattern formation have been widely studied (Cross and Hohenberg 1993; Newell et al. 1993; Passot and Newell 1994) with a number of applications to environmental processes (see for example Borgogno et al. 2009). Cross and Hohenberg (1993) define pattern-forming systems as falling in three spatial types with subtypes depending on the nature of temporal instability:

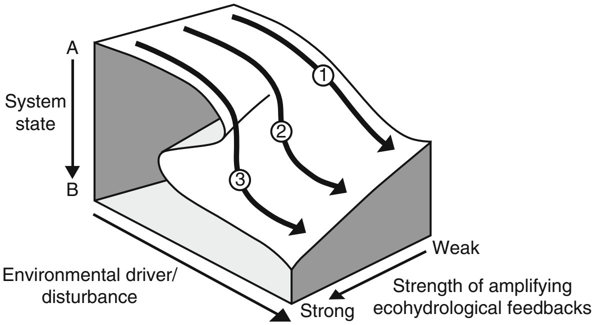

Fig. 3.8 Cusp-catastrophe model showing the potential response of an ecosystem to a change in the strength of one or more environmental drivers or disturbances, such that the trajectory of ecosystem-state change is determined by the strength of ecohydrological feedbacks. State A and state B are characterized by different self-organized patterns. The three *arrows* represent ecosystem responses in accordance with the continuous change model (*arrow 1*); the discontinuous change model without hysteresis (*arrow 2*); and the discontinuous change model with hysteresis (*arrow 3*) (Source: Turnbull et al. 2008, 2012)

Type I_s: spatially periodic, temporally stationary
Type I_o: spatially periodic, temporally oscillatory
Type II: intermediate
Type III_o: spatially uniform, temporally oscillatory

The types are defined by evaluating how a spatially uniform (equilibrium) solution to a set of nonlinear partial differential equations evolves as a result of perturbations when the system is driven further from equilibrium by changing the value of a control variable. Type I patterns occur when the wave vector of a Fourier mode perturbation $q_0 \neq 0$, while in type III patterns, $q_0 = 0$. If the wave frequency of the perturbation $\omega_0 = 0$, then the perturbations will be stable in time, while if $\omega_0 \neq 0$, the perturbations will oscillate in time.

To represent these patterns, Cross and Hohenberg (1993) take a simplified approach using *model equations*. In other words they abstract the behaviour of pattern formation to certain archetypal equations known to produce patterns of different types. These equations include four types of partial differential equation, namely the Swift-Hohenberg:

$$\frac{\partial u}{\partial t} = \epsilon\, u - \left(\nabla^2 + 1\right)^2 u - u^3 \tag{3.1}$$

the Kuramoto-Sivashinsky:

$$\frac{\partial u}{\partial t} = -\nabla^2 u - \nabla^4 u - \frac{1}{2}(\nabla u)^2 \tag{3.2}$$

reaction–diffusion models of the form:

$$\frac{\partial u}{\partial t} = f(u) + D \ \nabla^2 u \tag{3.3}$$

and Ginzburg-Landau models:

$$\frac{\partial u}{\partial t} = u + (1 + \ i \, b) \ \nabla^2 u - (1 + \ i \, c) \, |u|^2 \, u \tag{3.4}$$

where in all cases u is a spatially distributed variable; ε, b and c are parameters; D is a diffusion parameter (or matrix); and i is the imaginary unit ($\sqrt{-1}$). Model equations can also be discrete models defined as three types: (i) systems of coupled nonlinear ordinary differential equations, most notably the Lorenz equations; (ii) coupled maps, such as the logistic map:

$$u_{t+1} = R \, u_t \ (1 - u_t) \tag{3.5}$$

and (iii) cellular automata. Other discrete approaches for producing patterns are noted above; Cross and Hohenberg (1993) discuss at length how to derive and evaluate approaches to reproduce the different types of pattern, to evaluate the effects of boundary conditions, and the evolution of non-stationary patterns through time.

In using the model-equation approach, Cross and Hohenburg note that they "do not mean that all systems behave in the same way, but that certain properties, in particular those involving long-range effects, are common to a class of systems, and can therefore be understood by studying a simple member of that (universality) class" (1993: 872). In other words, in representing the phenomenological behaviour of a particular pattern-forming system using a model equation, it is not necessarily the case that we have understood the causal mechanisms explaining that particular system The model-equation approach implies significant problems of equifinality when trying to imply process from pattern.

Borgogno et al. (2009) review the use of pattern-forming models to environmental applications. They note that there are three main groups of approach. First, Turing-like instability models have been used, inspired by Turing's seminal 1952 paper on photochemical reactions. These models are based on coupled reaction–diffusion equations of the form:

$$\begin{aligned} \frac{\partial u}{\partial t} &= f(u, v) + D_u \, \nabla^2 u \\ \frac{\partial v}{\partial t} &= g(u, v) + D_v \, \nabla^2 v \end{aligned} \tag{3.6}$$

where u and v are state variables, $f()$ and $g()$ functions relating to the reaction kinetics of the system and D_u and D_v are diffusion coefficients. Instabilities can only occur if $D_u \neq D_v$. Oster (1988) uses this system to model a number of biological patterns including the shell patterns of molluscs using this approach, under the title of LALI (local activation with lateral inhibition). Cross and Hohenberg (1993) also point out that a focus on such models ignores some important approaches (e.g. the Swift-Hohenberg model) which can also produce similar patterns. While Turing himself demonstrated the use of this approach in evaluating biological morphogenesis, he was careful to point out that the model was not appropriate for use in hydrodynamic systems. Unfortunately, subsequent applications of the model in dryland degradation have not been so careful (see discussion in Stewart et al. forthcoming).

Secondly, Borgogno et al. (2009) define a set of kernel-based models, which are also known as neural models. These models use kernel functions to represent processes of short-range cooperation and long-range inhibition in biological systems, for example in the case of facilitation and competition at increasing distances from the centre of a tree or shrub canopy. A simple form of this model would be to consider biomass using a function such as:

$$\frac{\partial u}{\partial t} = h(u) + \int_{\Omega} \omega \ \left(r, r'\right) \left[u \ \left(r', t\right) - \ u_0\right] \ dr' \tag{3.7}$$

where the function $h(u)$ describes the local dynamics and $\omega(r,\ r')$ is the kernel (weighting function) representing the interaction between points r and r'. In the simple case where the kernel is a simple function of the distance between the two points, i.e. $z = |r - r'|$, local facilitation and more distant inhibition can be represented by a kernel of the form:

$$\omega(z) = b_1 \ \exp\left[-\left(\frac{z}{q_1}\right)^2\right] - b_2 \ \exp\left[-\left(\frac{z}{q_2}\right)^2\right] \tag{3.8}$$

where $0 < q_1 < q_2$ reflect the relative distances over which the facilitation and competition effects apply, respectively, and b_1 and b_2 are the magnitudes of their relative effects. If the ratios $\epsilon = b_2/b_1$ and $\chi = q_2/q_1$ are defined, then pattern formation will occur when:

$$h' \ (u_0) < 0, \ \ \frac{h' \ (u_0)}{b_1 \ q_1^2} > \frac{\pi \ \epsilon \ \chi^2 \ \left(\chi^2 - 1\right)}{(\epsilon \ \chi^2) \ \chi^2 / \ (\chi^2 - 1)}, \quad \epsilon \ \chi^4 > 1. \tag{3.9}$$

Borgogno et al. (2009) demonstrate how the Turing model and the neural/kernel models are the same in their linearized forms under conditions of steady state (as $t \rightarrow \infty$).

Thirdly, patterns can form in systems like the reaction–diffusion model of equations 3.6 even when the diffusivities are set to zero because of differential flow rates. A differential flow instability model can then be defined as:

$$\begin{aligned}\frac{\partial u}{\partial t} &= f(u,v) + p\,\frac{\partial u}{\partial t}\\ \frac{\partial v}{\partial t} &= g(u,v)\end{aligned} \tag{3.10}$$

where p is the drift velocity of the term u. Such models require the orientation of the system in the direction of drift, and thus are not independent of the observer (see Sect. 3.3). A full review of the different ecohydrological applications of these models and of stochastic models (discussed below) is given by Borgogno et al. (2009). They conclude by noting that there is a major mismatch between the theory of pattern formation by symmetry-breaking instabilities as discussed above, and the field evidence to support these ideas. Similarly, as noted in the Cross and Hohenberg review, one has to be careful about simply testing the outcomes based on patterns, as all of these models are capable of producing patterns. This problem is considered further in Chap. 8.

3.8 Stochastic Frameworks for Understanding Pattern

The study of patterns can offer useful information on the underlying ecosystem dynamics. In contrast to the more established literature on deterministic models of pattern formation, stochastic models have been developed more recently (García-Ojalvo and Sancho 1999; Sagues et al. 2007) to explain pattern formation as a noise-induced effect in the sense that patterns can emerge as a result of noisy fluctuations. Here, the main stochastic mechanisms underlying the emergence of organized spatial structures are reviewed (see also Scarsoglio et al. 2011).

A *patterned* field is considered here as one that exhibits an ordered state with spatial coherence. This general definition includes both periodic and multiscale patterns and is often adopted in the environmental sciences to account for the fact that spatial organization may emerge even when the concomitance of different processes prevents the organization of clear periodic structures.

The characteristic scales of variation of the field variable (e.g. vegetation density) are generally longer than those of the driving random forcing. As a consequence, we focus on Gaussian white noise as the driving force since it provides a valid assumption for the randomness present in environmental systems.

If we concentrate on systems with only one state variable and with no time-dependent forcing terms, the spatio-temporal dynamics can be described by a stochastic partial differential or by an integro-differential equation, expressing the temporal evolution of the field variable, ϕ, as:

$$\frac{\partial \phi}{\partial t} = f(\phi) + g(\phi)\,\xi(\vec{r},t) + DL[\phi]\,. \tag{3.11}$$

Thus, at any point $\vec{r} = (x,y)$, the dynamics of the state variable, ϕ, can be expressed as the sum of three terms: (i) a function, $f(\phi)$, representing the local dynamics, (ii) a noise component, $g\,(\phi)\,\xi\left(\vec{r},t\right)$, and (iii) a term, $DL[\phi]$, accounting for the spatial interactions with other points of the domain ($L[]$ is a differential or integral operator expressing the spatial coupling and D is the intensity of the coupling). These three components play a fundamental role in the mechanism of noise-induced pattern formation as follows: (1) the deterministic local dynamics tends to drive the field variable to a uniform steady state; (2) the noise term is able to maintain the dynamics away from its stable state; (3) the spatial coupling term provides spatial coherence. The interaction of noise with the spatial coupling is crucial, in that in the absence of noise the dynamics exhibit only transient patterns that disappear in the long term.

The local dynamics term, $f(\phi)$, is a function of local conditions. It can represent a local rate of increase/decrease of the field variable, ϕ, for example the vegetation mortality rate.

The noise term, $g\,(\phi)\,\xi\left(\vec{r},t\right)$, introduces in the dynamics a white (in time and space) Gaussian noise (the equation is interpreted in the Stratonovich sense of using the midpoint between time steps). We assume that the white noise, $\xi\left(\vec{r},t\right)$, has zero mean and correlation given by:

$$\left\langle \xi\left(\vec{\mathrm{r}},t\right)\xi\left(\vec{\mathrm{r}}',t'\right)\right\rangle = 2s\delta\left(\vec{\mathrm{r}}-\vec{\mathrm{r}}'\right)\delta\left(t-t'\right), \tag{3.12}$$

where s is the noise intensity. If the component, g, is a function of the field variable, ϕ, the noise is multiplicative, otherwise the noise is additive.

The spatial coupling operator, $L[]$, in the spatial coupling term, $DL[\phi]$, is here called *pattern-forming* if it is able to induce periodic patterns (for suitable parameter values), while non-pattern-forming operators give spatial coherence without selecting a dominant wavelength. The Laplacian operator, $L[\phi] = \nabla^2\phi$, is an example of non-pattern-forming coupling typically used to represent the effect of the diffusion mechanisms in ecosystems (e.g. the ability of vegetation to spread over the landscape). The Laplacian operator is considered a *short-range spatial coupling* in that it accounts for spatial interactions between a point of the domain and its nearest neighbours. The Swift-Hohenberg operator, $L[\phi] = -\left(\nabla^2 + {k_0}^2\right)^2\phi$, is instead able to induce periodic deterministic patterns (with wavenumber, k_0). This operator accounts for both short and long range interactions and is able to explain the formation of periodic patterns. While the Laplacian term in the Swift-Hohenberg operator expresses the effect of short-range interactions, the biharmonic term, ∇^4, accounts for long-range interactions, i.e., with points of the domain that are not only in the nearest neighbourhood of $\vec{r}$. Examples of pattern formation due to the interplay between long and short range interactions are common in nature. In landscape ecology, inhibitory effects hindering vegetation establishment typically occur in the long range (e.g. due to lateral root competition), while cooperative interactions for vegetation growth establish in the short range (e.g. due to "nurse-plant" effects under the canopy; Borgogno et al. 2009).

Some analytical techniques can be used to obtain useful indications of pattern formation (see the details in Scarsoglio et al. 2011). Linear stability analysis by normal methods assesses whether an initial perturbation of the homogeneous equilibrium state grows in time, thereby leading to the possible emergence of patterns. The short-term instability analysis determines whether there is an initial growth which can be exploited by the spatial coupling to generate a pattern. The structure function is a prognostic tool able to assess the presence of a selected wavelength in the spatial field. The classic mean-field analysis allows one to observe a possible phase transition and to determine the analytical expression of the probability distribution of ϕ at steady state. The generalized version of the mean-field analysis provides information on the loss of stability of the uniform steady state with respect to periodic perturbations.

These analytical tools can give some insights into pattern formation; however a more conclusive study of noise-induced pattern formation requires the numerical simulation of the dynamics to assess the emergence of spatial coherence. Two simple stochastic models will be proposed in Sect. 7.3 to clarify the interplay among the three fundamental terms in Eq. 3.11, namely local dynamics, noise component and spatial coupling.

3.9 By Way of Summary

This chapter has attempted to remove the ambiguity in the use of terms such as resilience, complexity, self-organization, emergence and pattern across a number of rapidly developing themes, and applied them to dryland settings in order to evaluate their potential usefulness in understanding degradation processes. One of the problems in translating cutting edge research, often with a range of esoteric or new terminologies, into practice is to ensure that these terms are as clear as possible and can be "translated" into language that is clear to stakeholders in ways that are unambiguous. This is a theme to which we will return in the later chapters of the book.

Processes were mentioned several times in this chapter; however, the discussion on self-organization and resilience were clearly governed by patterns. Nevertheless, it is a set of processes, i.e. the sequences of changes in a system that drive landform and vegetation development in drylands. Processes are studied if a systematic approach is employed to learn more about the underlying principles of a system's dynamics. Critical questions are how processes are organized in space and time across a continuum of scales and how different processes interact at different scales (Tarquis et al. 2011). The processes control the state of the ecosystem and its change regarding structure and functioning (Beisner et al. 2003). They are induced by exogenous forces which are environmental drivers and disturbances that either cause a direct change in system state or trigger changes in the internal system feedbacks that control the dynamics of an ecosystem (Turnbull et al. 2008).

One of the exciting aspects of the topics covered is the way in which cutting-edge ideas in complexity science and the mathematics of pattern formation can be applied to the understanding of a major, global problem. In presenting an overview of these novel ideas and their application to dryland evaluation, we have now set up a series of themes to be developed further in the following chapters. In terms of the processes underpinning land degradation – which are important in order for us to pass from phenomenological to mechanistic understandings – it has been shown that pattern generally evolves as a function of the interaction between processes operating over short and long distances. Thus, in Chap. 4, we consider the short-range ecogeomorphic processes, and complement them by looking at long-range processes in Chap. 5, before looking at ways of integrating them in Chap. 6. The practicalities of doing so and the frameworks for testing the outcomes of pattern-forming process is then the theme of part II of the book, again drawing on the definitions introduced here.

Two critical points arise from this chapter. The first has just been mentioned – namely the idea that process understanding is a fundamental part of taking studies of land degradation forward. Great leaps forward have been made using phenomenological approaches, but if we are to turn these advances into practical applications in specific drylands, understanding *why* particular patterns occur, and why they may (or may not) be indicators of degradation is absolutely critical. Secondly, the variety of approaches in what has generally been called complexity science shows that the results obtained are usually observer dependent. Thus, in searching for answers about land degradation, we need to be cautious in testing ideas, especially when approaching the same questions from different disciplinary backgrounds. This subjectivity may be difficult to accept for those brought up in more traditional scientific frameworks, but it is clear that we need to change the way we approach scientific problems based on the concepts discussed here. Rather than looking for the holy grail of a single answer to land degradation, it is now clear that we are looking for a suite of answers that we need to address and evaluate in critically novel ways.

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Chapter 4
Short-Range Ecogeomorphic Processes in Dryland Systems

Nicolas Barbier, Juan Bellot, Pierre Couteron, Anthony J. Parsons, and Eva Nora Mueller

Abstract Interactions between ecological and geomorphic processes in drylands operate at a continuum of spatial scales. Processes that operate at the smaller (plant-interplant up to hillslope) scales produce intrinsic patterns of vegetation and resource accumulation. Four aspects of short-range process interactions are presented here: the importance of vegetation cover of individual plants and plant patches and their interactions with resource availability; an updated account of the significance of the islands of fertility and landscape linkages; the interrelationship between facilitation and soil moisture dynamics and the importance of morphological properties, such as plant allometry, to generate stable patterns of vegetation.

4.1 The Importance of Spatial Scale

Dryland systems are driven by processes that act across a continuum of spatial and temporal scales, producing patterns of vegetation, resource accumulation and degradation. In soil science, hydrology and landscape ecology, dryland areas are frequently conceptualised into entities of certain sizes corresponding to their dominant

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N. Barbier • P. Couteron
IRD/UMR AMAP, Botany and Computational Plant Architecture, 34398 Montpellier, France
e-mail: nicolas.barbier@ird.fr; pierre.couteron@ird.fr

J. Bellot
Depto. Ecologia, Universidad de Alicante, 03080 Alicante, Spain
e-mail: juan.bellot@ua.es

A.J. Parsons
Sheffield Centre for International Drylands Research, University of Sheffield,
Sheffield S10 2TN, UK
e-mail: a.j.parsons@sheffield.ac.uk

E.N. Mueller (✉)
Institute of Earth and Environmental Science, University of Potsdam, 14476 Potsdam, Germany
e-mail: eva.mueller@uni-potsdam.de

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Table 4.1 Hydrological and ecological scale concepts (Blöschl 1996; Kirkby et al. 1998; Reynolds et al. 1997)

Scale	Scale type	Description	Size
Hydrological	Plot	Experimental plot in the field or lab	1–10 m
	Hillslope	Entity of relatively homogenous soil and vegetation along a slope	100–1,000 m
	Catchment	Assemblage of hillslopes and channel networks	10–100 km
	Regional	Assemblage of catchments	>1,000 km
Ecological	Plant-interplant/patch	Single type of plant growing on a homogenous soil type	1–100 m^2
	Mosaic	Assemblage of multiple patches	0.01–1 km^2
	Landscape	Assemblage of multiple mosaics defined by distinct geomorphic surfaces or topographic features	1–100 km^2
	Regional	Assemblage of landscapes within a general bioclimatic zone	>100 km^2

resource patterns, processes, constituencies and functionalities. In hydrological and soil-erosion studies, length scales are typically employed, (including the plot scale, hillslope scale, catchment and regional scale). Ecological extents are normally given in area, rather than length (including the plant-interplant, patch, mosaic, landscape and regional scales, see Table 4.1).

There appears to be a quantitative correspondence between the various hydrological and ecological scales, specifically: plant-interplant and plot scales, mosaic and hillslope scales and catchment and landscape scales. As land degradation involves both hydrological and ecological processes, the two scale notations, length and area, are often used interchangeably in various field and modelling studies of degradation processes.

Hydrological and ecological scales are distinguished according to different types of natural heterogeneities and patterns of their key elements and are dominated by specific redistribution or vegetation-dynamics processes. Table 4.2 summarises several of the elements and processes associated with the scales of ecogeomorphic systems, but excludes ecogeomorphic elements and processes related to wind, fire and fauna (such as the actions of cattle or rodents), as they are mainly relevant at larger scales (see Chap. 5).

To understand the relationship between ecogeomorphic feedback dynamics and the behaviour of self-organising patterns in the landscape, the elements and processes need to be described, quantified and analysed at each relevant spatial scale. The 'island of fertility' concept is a classic example at the plot or plant-interplant scale of how local water and soil resources become concentrated around a dominant vegetation species, – e.g. individual shrub canopies that form due to overland flow redistribution processes. Other examples are the concepts of facilitation and ecological fields in arid vegetation which are employed to structure and analyse ecosystem functioning. The following sections will sketch an outline

Table 4.2 Dominant properties and processes for various hydrological scales after Renschler and Harbor (2002)

	Entity	Plot	Hillslope	Catchment	Region
Elements	Topography	Microtopography		Channel network	
	Vegetation	Cover	Type	Pattern	
	Soil	Texture	Type	Lithology/geology	
	Moisture	Saturation + wilting point		Moisture availability	
	Climate	Micro-climate		Regional climate	
Processes	Rainfall	Temporal distribution		Spatial distribution	
	Water balance	Infiltration + evapotranspiration + percolation			
	Runoff	Overland + rill flow		River flow	
	Erosion	Detachment	Rill	Channel erosion	
	Sediment transport	Interrill	Gully	Channel	
	Vegetation	Establishment + mortality		Dominance	

of vegetation-resource patterns and interactions on the small scales of several metres and tens of metres, – i.e. the plot/plant-interplant to hillslope/patch scale – from both geomorphological and ecological perspectives. Processes related to evapotranspiration and heat fluxes are brought up, but interested readers are referred to the reviews of current ecohydrological literature for a full account (e.g. D'Odorico and Porporato 2006; Manfreda et al. 2010).

4.2 Interactions Between Local Water and Soil Resources and Vegetation Dynamics

At the smaller scales, dryland vegetation cover is discontinuous forming a mosaic of vegetation patches of grass and shrub species in a matrix of more or less bare soil. This mosaic is assumed to be the result of a tight coupling of soil-water availability and vegetation dynamics. The feedback dynamics between plants and soil resources (moisture and nutrients) are a critical factor in regulating the evolution of soil structure and plants communities.

By acting on vegetation, whether individual or en masse, the soil micro-environment alters the availability of resources for other organisms. Soil-vegetation interactions generate different microhabitats resulting in a spatial heterogeneity that persists through, for example, the formation of fertility islands from patches of vegetation (see next section). The scarcity of vegetation is primarily a result of adverse climate conditions that create an environment where water availability is for large parts of the year limited, intermittent and unpredictable in time and space. Precipitation events are often very isolated in times and location, with short wet weather periods followed by long dry periods, and the dynamics of plant populations often directly depend on their response to the pulses of water that they receive (Schwinning and Sala 2004). This pattern promotes the regeneration of rapid-growth species (Schütz et al. 2002), over other slower growing species, found in arid conditions that are more water-conservative (Rey and Alcantara 2000; Verdú and García-Fayos 2002). The low efficiency in plant recruitment tends to be offset by high longevity of individual shrubs which allows them to persist in this environment.

A field study in a typical semi-arid Mediterranean shrub-grass landscape (Ventos experimental area in Spain) indicated that the patches of *Quercus coccifera* shrubs have hardly changed in size and number since 1950, and their persistence is said to be due to their ecogeomorphologic position along the hillslope (Mayor et al. 2008). Large grasses such as *Stipa tenacissima* are located in the inter-shrub areas, creating a scattered patch mosaic with fine sediments located between the shrubs and bare soil areas. In the shrublands, structural and nutritional differences between bare and covered soil are significant. The preservation of water and nutrients is usually better in covered soil than in bare soil. Mayor et al. (2008) showed that positive feedback effects exist between the shrub plants and their surrounding soil. Throughout their life cycle, shrub plants modify the soil to improve its physical properties, including soil texture and fertility. The plants add organic matter to the soil, promote decomposition, increase the infiltration of rainwater and run-on from bare areas, as well as prevent erosion and facilitate deposition of material washed downslope from upslope source areas. At the catchment scale, the existence of small streams and the areas of water and sediment movement and accumulation become a factor influencing the patterns of vegetation cover. Hence, at a larger scale, the survival of shrubs is not just a function of their ecogeomorphic position, but also a function of topographical changes and resulting preferential water flow (see also Chap. 5).

A way to evaluate the interactions and benefits of vegetation cover on soil is proposed by Tongway and Hindley (1997): the Landscape Functional Analysis (LFA) Methodology. This analysis is based on evaluating two aspects of the landscape: the structure (spatial structure of the source-sink system) and the functionality (soil surface assessment). The variables used to estimate the structure are: distance between consecutive patches, patch length and width, number of patches over 10 m of linear transect, and patch area. For the functional analysis, the soil surface condition assessments use three integrated soil indicators: soil stability, infiltration/runoff balance, and nutrient cycling. For the soil stability, the variables estimated are: ground cover, surface crust fragmentation, erosion forms, deposited materials, surface compaction, and soil aggregate stability from a slake test. The estimation of the infiltration/runoff balance requires the quantification of canopy cover, litter cover, microtopography, surface compaction, and soil texture. Finally, the nutrient cycling capability is calculated by the canopy cover, litter cover, cryptogam cover and microtopography measurements.

The LFA method was applied to a semi-arid Mediterranean shrub-grass landscape to analyse the effects of vegetation cover on the dryland's ability to regulate runoff flow, water availability and vegetation persistence (Bautista et al. 2007). The field experiments showed that soils covered by patches of *Quercus coccifera* shrubs and by *Stipa tenacissima* grass plants had better structural conditions (depth, bulk density, compaction, crusting, etc.), higher fertility (amount of organic

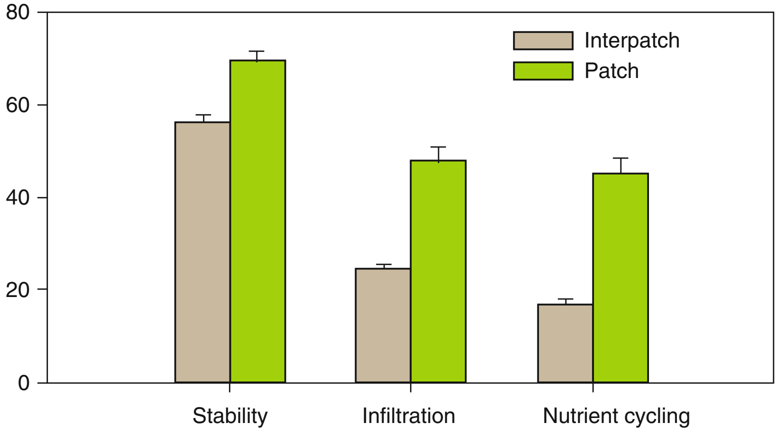

Fig. 4.1 Landscape function: soil surface assessment by patch versus interpatch analysis. All indices are statistically higher in patches than interpatches (one factor ANOVA, $n = 24$, $P < 0.001$ in all cases) (Unpublished data from AG. Mayor)

matter, nitrogen N, phosphorus P) and water content than the bare soil between patches and bushes. Figure 4.1 shows the statistically significant differences in soil stability, nutrient cycling and infiltration/runoff between covered soil (patches) and uncovered (interpatches). The reason for altered soil properties was that the vegetation that reduced soil compaction with its root systems, provided the organic matter from the aerial and ground biomass, increased porosity, aggregate stability, infiltration, inputs of water capturing fog and precipitation, and retained the fine soil material eroded from the upslope areas. Thus, the plants created conditions that favoured the retention of resources, thus forming a sink of water and materials, thereby increasing soil fertility in comparison with the bare soil.

In contrast, the runoff and soil loss rates increased in the interpatches, mainly due to surface crusting and soil compaction. Simultaneously with the improved soil conditions, the plants increased their productivity and adapted their metabolism, i.e. there were significant differences in rates of photosynthesis, transpiration, photochemical efficiency and water-use efficiency between plants located in different ecogeomorphological positions (patch or interpatch dominated) within the hillslope and watershed (Ramirez and Bellot 2009; Ramírez et al. 2008).

The functioning of ecosystems can be described in terms of both energy and resource fluxes and the ecological regulation due to interactions among organisms and their environment. In 1979, Noy-Meir found in his ‘auto-ecological’ hypothesis that for arid ecosystems the extreme environmental condition of reduced water availability determines that interactions between organisms and that feedbacks of organisms with the environment are negligible. Hence, shrub communities may be structured by the independent responses of individual species and are driven by the physical environment rather than by interactions such as competition, facilitation or predation. According to this hypothesis, there was thought to be a low connectedness, weak or intermittent interactions among population and few if any regulatory feedbacks (Graetz 1991).

However, the LAF analysis above demonstrates that plant patches improve the micro-habitat and facilitate the availability of resources for other organisms. Other dryland studies confirm that many drylands are far from being auto-ecological ecosystems (for example, Aguiar and Sala 1998). Aguiar and Sala (1998) demonstrated for dryland settings in South America that interactions among their species and the feedbacks of organisms with the physical environment are strong and numerous and that the extinction of one species or alterations of a resource pool may promote major changes in the functioning of the ecosystems.

In contrast to the auto-ecological hypothesis, there is a general consensus that there are both abiotic and biotic interactions in dryland ecosystems. The following two sections will focus in more detail on the current research frontiers in regard to these interactions: ‘island of fertility’ formations and plant interactions, such as facilitation and competition between organisms.

4.3 The Island of Fertility Concept Revisited

What exactly are islands of fertility? Nearly 40 years ago, Charley and West (1975, 1977) identified spatial patterns in the distribution of nitrogen in desert soils associated with litter fall from desert shrubs. Subsequent field studies in drylands showed that this spatial patterning applies not only to nitrogen but to a variety of plant resources, including both water and nutrients. Both biotic and abiotic processes contribute to this spatial patterning, including the deposition of windblown material beneath shrub canopies, the accumulation of fine sediments beneath shrubs due to differential splash (Parsons et al. 1992), the preservation of soils under shrubs that have invaded grasslands, burrowing by rodents which enhances infiltration beneath shrubs, the presence of nitrogen-fixing bacteria in shrub-root systems and the funnelling of rainfall by shrub canopies (Abrahams et al. 2003; Navar and Bryan 1990; Martinez-Meza and Whitford 1996). Together, all these processes give rise to what have been termed ‘islands of fertility’ associated with desert shrubs.

It has been argued that the formation of islands of fertility is a positive feedback process; and that it is one which contributes to global desertification, specifically where shrubs have invaded former grasslands. “Once begun, the increasing heterogeneity of soil resources in arid lands is likely to develop a positive feedback that will reinforce the new functional properties of the ecosystem” (Schlesinger et al. 1990, p. 247).

The focus of the islands of fertility concept is at the plant-interplant scale, and, consequently, emphasis has been placed on collecting data to show heterogeneity in the landscape at this scale. Charley and West (1977) showed a progressive increase in both soil nitrogen and carbon from the centre of plant interspaces to the point

Table 4.3 Comparison of runoff, sediment and nutrient fluxes obtained from rainfall simulation experiments and monitoring of natural events on small plots (Schlesinger et al. 1999, 2000)

		Runoff coefficient in %	Sediment in $g/m^2/min$	Total dissolved nitrogen in mg/l rain
Rainfall simulations	Grassland	24.0	0.5	4.3
	Creosotebush	46.0	1.5	1.67
	Ratio	0.52	0.33	2.57
Natural events	Grassland	3.6	0.192	0.15
	Creosotebush	18.3	0.525	0.33
	Ratio	0.2	0.37	0.45

of maximum litter accumulation beneath shrub canopies. Likewise, Abrahams and Parsons (1991), in a study of shrubland at Walnut Gulch, southern Arizona, showed infiltration to be a function of surface stone cover and argued, on the basis of previous work by Parsons et al. (1990) that this relationship is controlled by the spatial extent and density of shrub cover. Turnbull et al. (2010) undertook a study of four plots over a shrub encroachment gradient located at the Sevilleta National Wildlife Refuge, New Mexico. These authors showed that with shrub encroachment there was increased vegetation patch size and hence increased spatial heterogeneity of soil properties.

This same increase in spatial heterogeneity in soil properties as islands of fertility develop has also been identified with a change in surface microtopography, with consequent impacts on runoff and sediment and nutrient fluxes (Parsons et al. 1996). Grasslands typically have a microtopography characterised in the across-slope direction by clumps of grass and surface gravel with an amplitude of 2–5 cm and a wavelength of 20–50 cm, and in the downslope direction by treads and risers with an amplitude of 20 cm and a wavelength of 0.5–2.5 m. In contrast, shrublands show an across-slope mound-and-swale microtopography with an amplitude of 10 cm and a wavelength of 2–5 m, in which case the shrubs sit atop the mounds, and there is no pronounced downslope microtopography. Associated with these microtopographic differences, Parsons et al. (1996) showed differences in runoff hydraulics in two hillslope-scale runoff plots. On the grassland plot, downslope increases in discharge were matched more-or-less equally by increases in flow depth and velocity with flow width reducing slightly. In contrast, the shrubland plot showed that the downslope increase in discharge was accommodated largely by an increase in flow velocity, with a slight increase in flow width and negligible change in flow depth. Such differences in microtopography and runoff hydraulics have been shown to result in differences in spatial patterns of infiltration, moisture and nutrients available to plants and, hence, the spatial patterns of plant growth. Shrubland areas have been shown to have higher runoff coefficients, sediment and nutrient fluxes (Schlesinger et al. 1999, 2000), from small-plot studies using both rainfall-simulation experiments and monitoring of natural events (Table 4.3). However, the ratios of these fluxes between grassland and shrubland are not thoroughly consistent. Whereas natural runoff events show a five-fold increase in runoff for shrubland compared to grassland, the rainfall-simulation experiments shows only a doubling. And although sediment flux ratios are similar, nitrogen flux ratios are reversed: grassland yields more than twice as much nitrogen per unit rainfall than the shrubland in the rainfall-simulation experiments, but less than half as much under natural events.

It may be anticipated that shrublands produce more runoff than grasslands. The stony swales are areas of low infiltration and with the lack of downslope microtopography, runoff generated in these stony swales will directly be routed downslope. In contrast, runoff generated on the stony steps in the grassland will encounter areas of high infiltration and be subject to runon infiltration in the downslope direction.

The higher sediment flux on the shrubland is less easy to explain. Armoured stony swales are assumed to be areas of low sediment entrainment in contrast to the finer-textured mounds associated with the shrubs. The fivefold increase in runoff on the natural rainfall plots is consistent with this assumption, but the doubling of runoff and threefold increase in sediment flux on the rainfall-simulation plots is, at first sight, not. However, Abrahams et al. (2003) have shown that rain falling onto islands of fertility does not necessarily remain with the island. As a result of stemflow, effective rainfall intensities at the base of shrubs may very well exceed the local infiltration rate. Indeed, it can often be observed during a storm event that runoff is first generated at the bases of shrub mounds and transports fine sediment away from the mounds. The work of Abrahams et al. (2003) indicates that islands of fertility are leaky to some extent. Inasmuch as the leakiness of islands of fertility will increase with rainfall intensity, the observed relative values of sediment flux for the natural events and the rainfall simulations are consistent.

What is not consistent, however, is the difference in dissolved nitrogen when the rainfall simulations and natural events are compared. Runoff for example emanating from stemflow and shrub mounds should be nutrient rich compared to that emanating from stony swales so that, if the argument about the higher intensities of the rainfall-simulation leading to more runoff from shrub mounds holds, then it would be expected that higher concentrations of dissolved nitrogen would be observed from these experiments compared to the natural events. The rainfall simulations suggest that shrubland may be quite conservative of nitrogen, which would be consistent with the concept of islands of fertility.

A fundamental issue with any positive feedback process is what operates to constrain the process, or, in the present case, why has not all desert grassland been invaded by shrubs. That this is not the case implies that, though the concept may be useful in understanding small-scale patterns and processes, its role in understanding landscape-scale patterns, as envisaged by Schlesinger et al. (1990) may be somewhat less. Fluxes of runoff, sediment and nutrients measured at small scales cannot simply be extrapolated up to a landscape scale (Parsons et al. 2004, 2006; Brazier et al. 2007).

Whereas it is relatively easy to obtain data on the small scale, it is the data at larger scales that are needed to understand landscape-scale patterns. The limited data available comparing nitrogen fluxes at the landscape scale in grassland and

Table 4.4 Content of inorganic nitrogen in runoff obtained at two different scales (Schlesinger et al. 1999 and unpublished data)

	Grassland	Creosotebush
	Mg/l	Mg/l
Landscape scale	1.518	1.516
Runoff plots	0.72	0.25

shrubland (Table 4.4) suggests that the conservative nature of islands of fertility may not be apparent at a larger landscape scale. The data presented here compare inorganic nitrogen content of runoff from the same rainfall-simulation plots as used in Table 4.3 with unbounded flux measurements from 40 sites in creosotebush shrubland and 29 sites in grassland. Bearing in mind the inconsistencies between fluxes from small plots using rainfall-simulation experiments and those monitoring natural events, the differences in Table 4.4 may need to be interpreted with caution, but they are at least indicative of the need for studies of landscape-scale fluxes if landscape patterns and processes are to be understood.

In summary, the island of fertility is an important concept to understand land degradation processes at the plant-interplant scale. However for a rigorous analysis of on-going desertification, it is equally important to consider landscape linkages (Chap. 5) and biotic feedbacks of plants such as facilitation and competition.

4.4 Plant Interactions and Emergent Patterns in Ecogeomorphic Systems

Research into the lifecycle of islands of fertility has demonstrated that the spatial and temporal dynamics of vegetation may influence geomorphological processes. To understand the role of vegetation within ecogeomorphic systems, it is crucial to understand how the spatial structures of plant communities are generated. Numerous studies have focussed on the vegetated patch as the study level, leading to the concept of fertility islands as described above (Schlesinger et al. 1990). However, recent theories go further to bridge the gap between feedbacks occurring at the scale of individual plants (phytogenic or ecologic fields, cf. below) and system/landscape level dynamics (Lefever and Lejeune 1997; Lefever et al. 2009). This section therefore focuses on the plants themselves to study their interactions in an attempt to understand emergent properties at the community level. Although plant interactions are often referred to as mechanisms by which a plant affects the life of another plant, this process is clearly not independent of the physical environment. Indeed, plant interactions via local modifications of their environment can through shaping community structure at the landscape scale, eventually exert large-scale feedbacks on bio-geochemical processes. The following concepts highlight the importance of facilitation, ecological and phytogenic fields as well as the study of allometry (ratio between the parts of a single plant such as plant size to shape) for the understanding of emergent vegetation patterns.

4.4.1 The Concept of Facilitation

Competition has long had a central position in the study of plant interactions in ecology (Tilman 1982). However, competition alone cannot explain the existence of aggregative distributions of plants. Yet such distributions often occur in water-stressed (including arid) environments (Anderson 1971; Fonteyn and Mahall 1981; Phillips and MacMahon 1981), even over fairly homogeneous substrata. It was only in the 1980s that a change of paradigm started to emerge in favour of a long forgotten interaction mode: *facilitation* (Hunter and Aarssen 1988; Bruno et al. 2003). Since then, many empirical studies have confirmed the validity and importance of facilitative interactions (Callaway 1995), including those of dryland vegetation.

The concept of facilitation covers a range of possible processes affecting resource availability independently of their uptake ('non-uptake effects', sensu Goldberg 1990), or prevailing local conditions. As argued in the preceding section on islands of fertility, plants have been shown to locally influence latent heat fluxes, mineralization rates (Belsky 1994) and aeolian deposits (Leprun 1999). They also directly (through roots) or indirectly (through biological activity) improve soil structure and, in particular, macroporosity. The latter effect, coupled with topography, has often been considered as fundamental in the redistribution of water in arid landscapes (Ludwig et al. 2005). Indeed, due to the poor permeability of crusted soils in open spaces and runon effects, the water pouring during intense rainfall events would only infiltrate in vegetated patches. As a result, these patches may receive ratios of water exceeding by far the annual rainfall expected over the area (Galle et al. 2001). There is also some evidence of an increased soil water retention capacity under plant (tree) crowns, as a result of augmented organic matter content (Joffre and Rambal 1993). Some forms of facilitation between plants imply other trophic levels (e.g. via the repulsion of herbivores by specialized chemical compounds or dedicated plant structures, such as spines), or an increase in the activity of the pedofauna or pollinators. Plant facilitation can also occur via a decrease in the abundance of other plants; for instance, for shrubs through a reduced abundance and continuity of the grass layer and a decreased susceptibility to destruction by fire (Scholes and Archer 1997; D'Odorico et al. 2007).

There are many facilitation processes that exhibit intra- or inter-specific mechanisms by which the recruitment, survival, growth or reproduction of a plant may be noticeably improved through the presence of other plants. Evidencing such interactions in the field is often rendered difficult by the superposition of antagonistic effects. The reviews by Callaway (1995) and Callaway and Walker (1997) give some experimental evidence for positive interactions among plants such as root trenching to reduce competition.

Facilitation appears to have an increasing importance for plant distribution in constrained ecosystems such as dryland settings, whose functioning is conditioned by recurring water stress (Callaway and Walker 1997; Holmgren et al. 1997; Pugnaire and Luque 2001; Callaway et al. 2002). In a gapped periodic vegetation pattern in Burkina Faso, for example, Couteron and Kokou (1997) observed that drought-related mortality rates were higher for isolated woody individuals than for clumped ones. In Kenya, Belsky (1994) observed that grass yield was improved under tree crowns relative to open savannas and that ratio of grass yield improvement increased if conditions became more arid. This phenomenon was attributed to the fact that lateral roots of savannah trees extended further away from the trunk to find additional resources. As a consequence of these laterally extending roots, competition under the tree crown was reduced, facilitating grass growth. Positive interactions regarding water availability were only visible on deeper soils and disappeared entirely if the same tree species was sampled on shallower substrates. Both cases demonstrated that root architecture and the water balance changes are linked (Callaway et al. 1991, 2003) and highlight the importance of the spatial distribution of ecological interactions for a dryland system.

4.4.2 The Concepts of Ecological and Phytogenic Fields and Plant Allometry

Various attempts have been made to formalise the study of spatial interactions: for instance the theories of *ecological fields* (Wu et al. 1985; Walker et al. 1989) or *phytogenic fields* (Uranov 1965; Yastrebov 1996), or the concept of the single tree influence circle (Zinke 1962; Boettcher and Kalisz 1990). Ecological or phytogenic fields are determined as sectors in which the environment acquires new properties determined by the introduction of a given plant individual (Kryshen 2000). These theories are employed to quantify the spatial domain and spatial variation of a plant's influence on its surroundings. In general, simple and isotropous mathematical functions are used to model these interaction kernels. Modelling these interaction kernels is to be taken as an approximation at the population level as the plasticity of plant organs and, in particular, root systems (Mahall and Callaway 1991; Brisson and Reynolds 1994; Schenk et al. 1999) may produce very anisotropous and variable effects around a random individual. In practice, a proper parameterisation of ecological fields ideally requires studying the budget of growth-limiting resources in a spatially and temporally explicit manner. Resource dynamics have then to be linked to plant positions and sizes. A question that remains to be solved in the study of spatial interactions is if it is better to work in the vicinity of isolated individuals, or if it is feasible to disentangle isolated plant effects by working directly at the population level.

Ecological spatial interactions of dryland vegetation may also be studied by quantifying the intrinsic geometry between the parts of a single plant. The term allometry is often used in this context to describe the study of the relationship of plant size to shape and specific relations (length, area, volume) of individual plant parts, such as above- and below-ground plant parts. In general, these relationships are not linear: they do not vary isometrically, but allometrically (Niklas 1994).

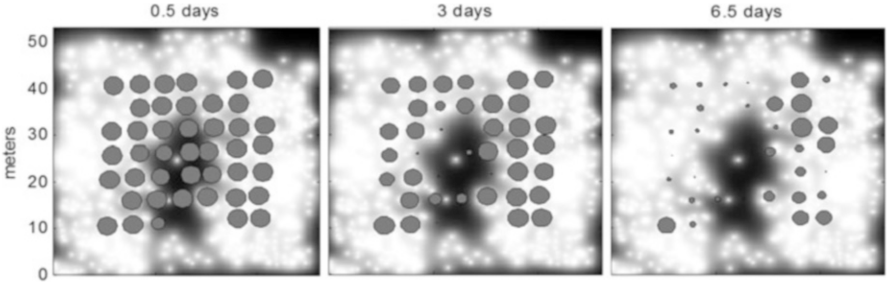

Fig. 4.2 Spatio-temporal dynamics of the soil water moisture at a 20 cm depth after a 37.5 mm rain storm within a gapped vegetation pattern in the W National Park (south-west Niger) (Modified from Barbier et al. 2008). *Grey circle* diameters are proportional to soil moisture rescaled between wilting point and field capacity. The *background grey-scale map* indicates the value of accumulated plant facilitation kernels (exponentials) with optimised parameters (see text below). Areas with most accumulated crown shadows are shown in *white*

4.4.3 Application of the Concepts to a Shrubland System

The concepts of facilitation and ecological fields have been applied by Barbier (2006) and Barbier et al. (2008) to study a semi-arid, gapped vegetation system in Niger, Africa (see also the case study in Chap. 13). This system displayed bare zones of approximately 25-m diameter periodically distributed every 50 m, within a dense shrubland matrix dominated by the shrub *Combretum micranthum G. Don* – a dominant shrub species in the West African region. The site was located on an iron-capped plateau, with shallow sandy-clayey ferruginous soil. The annual rainfall of 706 mm (La Tapoa, 1981–2002) is concentrated in 5 months around the boreal summer.

To study the effects of soil moisture dynamics on facilitation and competition fields around shrubs, they monitored the dynamics of soil moisture in the shallow root zone (20 cm) using gypsum bocks at 100 positions on a 2,500 square metre plot during one entire rainy season. A representation of the soil moisture dynamics after a storm event (37.5-mm rainfall during a few hours) is given in Fig. 4.2. About one thousand shrubs were mapped and measured and relevant soil variables, including organic matter content, bulk density and texture, were quantified.

The parameters of an exponentially decaying facilitation field around each plant were then optimised at the population level to determine the best possible fit for soil moisture and infiltration rates. The field approach showed that: (i) the facilitation range (i.e., the scale parameter of the exponential variable) corresponded to an area slightly smaller than the tree crown radius; (ii) the effect on soil-water dynamics in the root zone was not related to the recharge (at least, not after modal rainstorm types) and therefore appeared to be independent of a possible improvement of the infiltration rate. Soil-moisture recharge was shown not to differ between sensors located under bare soils and under vegetation (Fig. 4.2); (iii) direct evaporation showed a threefold decrease under shrubs and, (iv) the latter effect was more related to the crown presence (shading by crown or ground litter) than to soil variables (texture, bulk density, organic matter content).

Finding empirical evidence for the conditions of water uptake using similarly parameterised root-competition fields was difficult. As mentioned above, this may be due to the higher plasticity/anisotropy of root systems where individual roots tend to avoid each other in crowded areas. Evidencing and characterising root effects with this spatial soil-moisture sampling approach may require much larger sampling.

Root fields were therefore assessed using direct measurements from a limited number of excavated individuals of the shrub *Combretum micranthum*. It was found that its root systems were shallow and laterally extended up to a maximal distance exceeding ten times the crown radius. The measured allometric ratio linking the crown and roots for plants of different sizes was shown to play a crucial role in the emergence and dynamics of self-organising patterns (Barbier et al. 2008; Lefever et al. 2009). As the presence of a root (even below a size threshold) may not be correlated to its uptake activity, another protocol was employed using isotopic labelling of water uptake by localised irrigation with deuterated water (Barbier 2006). This approach confirmed that root uptake of water occurred up to distances of 15 m (for shrubs of height and crown diameter of about 3 m) even within bare areas.

This study contributed to the growing awareness that combined facilitative and competitive plant interactions at landscape-scale levels characterise the two-way feedback loops between the environment and vegetation. Unfortunately, the prevailing ‘patch paradigm’ has often led to neglecting plant allometry in the study of dryland patterned vegetation and relevant publications are scarce. However, it is intriguing that in the first-ever written field description of a tiger bush pattern, Gillett (1941) observed that the root systems of the dominant shrub species (*Acacia bussei Harms ex Sjöstedt*) presented a root system very similar to what was found for *C. micranthum*. Glover (1951) subsequently showed that such patterns of crown-to-root allometries are frequent for *A. bussei* in other patterned vegetations of Somalia. In any case, spatially extended root systems are a very frequent trait of plants growing in arid areas (Walter 1963; Schenk and Jackson 2002), as well as on shallow soils (Callaway et al. 2003), two recurrent characteristics found in periodic vegetation patterns (Tongway and Ludwig 1990).

Supporting the conclusion of previous sections, it is clear that a full understanding of the role of emergent patterns in degradation processes will only be possible when more field data on plant-soil moisture interactions from dryland settings become available in a spatially explicit form and ideally where the monitoring schemes of the facilitation and competition fields around shrubs (Fig. 4.2) would scale up to capturing areas of redistribution paths of plant-interplant and landscape-scale fluxes (data such as given in Table 4.3) on dryland hillslopes.

4.5 Synthesis

At the plant-interplant scale of drylands, the previous sections illustrated how geomorphologists analyse processes to comprehend patterns (Sects. 4.2 and 4.3) whereas ecologists analyse patterns to comprehend processes (Sect. 4.4), or is it the other way round? Both disciplines deal with vegetation-resource interactions and study abiotic versus biotic processes, but in a systematically different manner.

The hydro-geomorpholocical camp has found the island of fertility concept – which puts the focus of research on horizontal transfer processes of water and wind and the positive feedbacks in the soil-vegetation micro-environment – is useful in understanding small-scale patterns. However its application in studying landscape-scale patterns such as shrub encroachment appears inadequate, as it does not explain water and soil resource measurements of fluxes that are evident at larger scales.

From the ecological perspective, facilitative and competitive interactions of individual plants or plant patches are in the focus to understanding how landscape-scale patterns are sustained. Soil moisture dynamics, evapotranspiration and root uptake are here thought to be the dominant processes in regulating plant patch functioning all of them being mostly vertical processes.

Soil moisture dynamics is at the interface of the two disciplines: whether configured by stemflow, infiltration and runoff processes, or pattern-driven by root-uptake and transpiration. However, for a holistic understanding of dryland functioning, soil moisture has to be considered both as a result of flow processes *and* as a soil resource.

Both disciplines express the urgent need to obtain spatial soil-vegetation data at a larger scale as the current plot-scale data sets are not able to explain on-going degradation processes. If we now move to the next larger scale (the catchment or landscape scale), other drivers and processes become prominent, as discussed in the next chapter.

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Chapter 5
Long-Range Ecogeomorphic Processes

Artemi Cerdà, Francesc Gallart, Junran Li, Vasilios P. Papanastasis, Robert R. Parmenter, Laura Turnbull, Anthony J. Parsons, and John Wainwright

Abstract Processes operating at larger spatial scales interact with those operating at smaller scales (discussed in the previous chapter) to produce spatial patterns in drylands. Hydrological processes operating at the catchment scale, the effects

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A. Cerdà
Department of Geography, University of Valencia, 46010 Valencia, Spain
e-mail: artemio.cerda@uv.es

F. Gallart
Institute of Earth Sciences, Jaume Almera (CSIC), 08028 Barcelona, Spain
e-mail: francesc.gallart@idaea.csic.es

J. Li
USDA-ARS, Jornada Experimental Range, New Mexico State University, Las Cruces, NM 88003, USA
e-mail: junran@nmsu.edu

V.P. Papanastasis
Laboratory of Range Ecology, Aristotle University, 54006 Thessaloniki, Greece
e-mail: v.papan@for.auth.gr

R.R. Parmenter
Valles Caldera National Preserve, Jemez Springs, NM 87025, USA
e-mail: bparmenter@vallescaldera.gov

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

A.J. Parsons
Sheffield Centre for International Drylands Research, University of Sheffield, Sheffield S10 2TN, UK
e-mail: a.j.parsons@sheffield.ac.uk

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

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of grazing, fire, water and wind erosion are considered separately and in relation to their interactions with propagules and plants. The chapter concludes with a consideration of the extent to which feedbacks are central to pattern formation.

5.1 Introduction

In the previous chapter, we considered the basis of the first part of the set of interactions required to produce patterns in drylands, that of processes occurring over small spatial scales. This complementary chapter looks at the second element, namely those processes acting over large spatial scales. Of course, this definition is relative, but as seen in Chap. 3, the models that produce spatial pattern depend on the *relative* scale of process interaction, rather than an absolute one.

In terms of processes relating to water movement, hillslopes and catchments are the natural units of measurement at these larger scales. We thus start with an overview of catchment-scale processes and their monitoring. This background is then used to evaluate the effects of modifying processes such as the impacts of animal grazing and fire, both of which are significant components of dryland ecosystems across the globe. The final section on water-related processes considers how effective water-erosion processes are when moving from hillslope to catchment settings, and the relevant feedbacks in terms of water and vegetation patterning.

Emphasis then moves to wind erosion. As noted in Chap. 8, aeolian processes require a different sampling frame and it may be less obvious what the upwind area of interest is. For these reasons, aeolian geomorphologists have focussed (quite rightly) on flux estimates, an approach in which fluvial geomorphologists have only just caught up (Parsons et al. 2004). Wind flow also interacts significantly with vegetation so that long-distance transport pathways are modified.

The effects of both water and wind movement on vegetation having been noted, we then move to a more detailed assessment of the impacts on and by plants, followed by consideration of the behaviour of animals. In the final section, we consider ways of integrating all of the biotic and abiotic interactions at landscape scales.

5.2 Catchment-Scale Hydrological Processes and Their Monitoring

5.2.1 *Monitoring Design*

The catchment is the natural unit in hydrology as well as in geomorphology when water-flow processes are dominant. The adequate size of the catchment as a unit depends on the objective: for engineering applications, the ideal size is of tens or hundreds of square kilometres, whereas for process research, the need for both uniform characteristics of the basin and reasonable distances allowing field observations means that much smaller sizes are usually chosen. Most instrumented basins have a nested design, in order reconcile the need for detailed process studies and the assessment of their role at larger scales.

Even with a nested catchment design, monitoring water and sediment flows at the catchment scale furnishes just an integration of the processes occurring in the basin, but this ‘basin function’ is not simply or unequivocally invertible to determine the dominant processes at the hillslope scale because diverse sets of processes on the hillslopes may produce similar basin responses. Indeed, the history of hydrology demonstrates that wrong conceptions of hydrological processes on hillslopes can produce (apparently) acceptable simulations of water discharges at the gauging stations (Klemeš 1986; Grayson et al. 1992). On the other hand, plot and hillslope observations may not be representative of the processes relevant for the basin response (Beven 2006).

The monitoring of hydrological processes at the catchment scale therefore needs a nested network of gauging stations in channels to be matched with nets of measurements at the plot and hillslope scale. For economical or practical reasons, the stream-monitoring network is usually permanent or long-lasting, whereas plot and hillslope observations are usually made during limited periods or during experiments such as those of simulated rainfall events. In dry areas, the equipment at gauging stations must be robust to allow long dormant periods during dry spells but also to resist high water flows and sediment loads during floods. Stream-flow controls (weirs) must be self-cleaning or protected with large sedimentation ponds to prevent their malfunctioning during events; the loss of precision for low flows is usually not relevant because these flows are usually short-lived and represent a small part of the runoff volumes.

Monitoring sediment loads in temporary streams is not a straightforward task. Automatic water samplers are robust and may provide good suspended sediment sampling, but they are relatively expensive and may be objects of vandalism in remote or unattended sites. Low-cost siphon ‘stage samplers’ based on the U.S. U-59 sampler (Brakensiek et al. 1979), may provide a robust alternative, but they take water samples only during the rising limb of the hydrograph and therefore cannot provide information on the evolution of the sediment concentration-discharge relationships during events and will produce an overestimation of sediment loads on most occasions. Continuous measuring instruments like back-scattering turbidimeters or ultrasonic sensors may provide an assessment of suspended sediment dynamics during events, but the calibration functions needed to obtain sediment concentrations from sensor readings are very sensitive to some sediment properties such as the grain size of particles (Clifford et al. 1995; Soler et al. 2012). Ephemeral streams have higher efficiency for transporting bedload than permanent ones, mainly because of the poor development of armour layers (Laronne and Reid 1993). Bed load in ephemeral streams is commonly measured using deposition ponds or scour chains that allow monitoring of sediment transfers that do not necessarily result in net morphological change (Powell et al. 2007). A combination of scour chains and particle tracing may help to determine sediment volumes and distances when no other information is available (Hassan et al. 1999). Finally, more sophisticated equipment such as Birkbeck-type slot samplers have been used for continuous monitoring of bedload movement (Laronne et al. 1992).

5.2.2 *Discontinuity of Flow and Sediment Transfer*

One of the more distinctive traits of hydrological processes in dryland environments is the discontinuity of flows (see Chaps. 3 and 4). At the beginning of rainfall events, in dry areas, soils are usually dry and the water table is deep (and in many cases so deep as to be effectively non-existent). Overland flow on hillslopes is commonly produced by rainfall excess (Hortonian process) on impervious soil or bare rock patches as well as by local saturation of shallow soil patches, but it frequently reinfiltrates downslope in more permeable or thicker soil patches (Calvo-Cases et al. 2003; Wainwright and Parsons 2002; Yair and Kossovsky 2002). Connectivity between hillslopes and channels is usually poor and short-lived when it occurs during major events (Puigdefábregas et al. 1998; Bracken and Croke 2007).

When flowing water reaches streams, it usually flows over a dry bed with an abrupt water head and a significant part of the water percolates in the alluvium, a process usually known as ‘transmission loss’ in hydrology (see graph and picture examples in Renard et al. 2008), although in terms of water resources, this is a benefit rather than a loss because it usually recharges underground waters (e.g. Morin et al. 2009).

Overall, runoff coefficients typically decrease with increasing basin area in dry environments, at both the plot and basin scales (e.g. Stone et al. 2008). In terms of ecology, this means that water is redistributed by overland flow, so the lower parts of hillslopes receive not only rainfall water but also a part of the water precipitated upslope, although in soil or debris-mantled hillslopes some connection occurs only during major events and soil moisture does not usually increase downhill (Puigdefábregas et al. 1998). In sediment-filled valley bottoms, the water availability increased by redistribution makes possible the settlement of species more water-demanding than those that grow on the hillslopes (Nogueras et al. 2000). In stream valleys with several-metre-thick alluvial deposits, transmission losses allow the existence of trees or tall bushes that live only or primarily in alluvial sites where they have access to deep water (Domingo et al. 2001; Stromberg et al. 2007; Tooth and Nanson 2000).

Sediment transfer is usually subject to stronger discontinuities than water along hillslopes, giving the possibility for the development of soils and particularly colluvial deposits. In streams, fine sediments are usually transported in suspension for long distances during floods, but when erosion rates on hillslopes are high or runoff occurs only in small parts of the basin, deposition of fine sediments may happen at the end of ephemeral runoff events (Gallart et al. 2002). The analysis of the relationship between suspended sediment concentration and water discharge at the gauging stations may provide some insight on the origin of sediments, particularly when diverse events at the same location or diverse basins in the nearby area are compared (Soler et al. 2008).

Connectivity between hillslopes and channels is poor when depositional forms such as fans occur and good when erosional forms such as rills or gullies are present (Faulkner et al. 2008). In general, connectivity between hillslopes and channels and within channels is much lower for coarse sediments, and it changes in time and space depending on the activity of the sediment sources and the capacity of stream to transport bedload (Hooke 2003; see also the excellent review by Powell 2009). On the other hand, specific sediment yields in basins may be larger than those measured at the erosion plot scale, a fact attributed to the role of sediment sources not accounted for in plot studies, such as gullies, stream bank erosion and mass movements (Cantón et al. 2011).

5.2.3 Stream-Channel Aggradation and Incision

Most headwater streams in dry areas show the remnants of alternate phases of erosion and deposition in the recent past (the Holocene). The environmental changes that led to these alternations are usually attributed to climate shifts or land degradation due to human activity. Current hypotheses state that during drier conditions the vegetation cover is reduced, triggering accelerated erosion from slopes and accumulation along streams; wetter conditions favour a decrease in the sediment supply from slopes and more stable stream flow, leading to channel entrenchment (e.g. Sancho et al. 2008; Borejsza and Frederick 2010). However, Wainwright and Thornes (2003) have argued that the mechanisms are more complex and contingent, especially where human activity is involved.

Furthermore, particularly for first-order streams, it has been argued that the environmental causes for periods of deposition and incision are more intricate, because the capacity of the ephemeral flow for transporting sediments also depends on the hydraulic role of vegetation that carpets the valley bottom (e.g. Dunne and Dietrich 1980; Montgomery and Dietrich 1992, 1994). Consequently, the climatic and environmental conditions that produce channel aggradation or entrenchment may not be unique and have been subject to some debate (see Cooke and Reeves 1976, for a classic overview). Melton (1965) suggested that the degradation of hillslope vegetation due to increased grazing would increase the accumulation of sandy sediments on the sides of the valley floor, subsequently increasing the hydraulic radius of the flow and its velocity; these changes would be accompanied by the degradation of valley vegetation by grazing. Leopold (1951) and subsequently Leopold et al. (1966) suggested that an increase in the size of rainfall events at the expense of the total number of rainy days would impoverish vegetation and subsequently increase erosion, especially if this climatic swing coincided with heavy grazing activity. Corbel (1963) emphasized that an increase in summer intense rainfalls at the expense of mild winter precipitations that recharge aquifers would impoverish the vegetation cover of the valley bottoms and increase runoff peaks, triggering channel erosion. Prosser (1996) suggested that entrenchment can be promoted by extreme events or by the spontaneous change in valley-floor vegetation cover from tussock grass and sedge to aquatic plants, given the latter's lower resistance to erosion. Nogueras et al. (2000) suggested that sediment deposition in the bottoms of elementary valleys was induced by the presence of a carpet of vegetation fed by the runoff coming from the marly hillslopes; the increase of drought periods would cause a decay of this vegetation and the easier incision of the valley-bottom sediment fill (arroyo formation).

It is worth to recall that discontinuous scour-transport-fill sequences are a phenomenon in semi-arid valley bottoms that demonstrates that the conditions for incision or deposition depend on several local factors that may change along the stream path. These forms consist of linear incisions cut in the alluvial fills of valley bottoms; the topographic gradient of the incised channel is lower than the gradient of the valley floor, so when it reaches the same level, flow is spread over the valley floor and a small alluvial fan is formed at the outlet (Bull 1997; Wakelin-King and Webb 2007). These forms migrate upstream and result in synchronous incision and deposition phases in the same stream. Discontinuous erosion-deposition cells also occur disconnected from the drainage nets and are a relevant landscape feature in some Australian arid areas (Pickup 1985; Wakelin-King and Webb 2007). They also occur at smaller scales on bajada slopes (Wainwright et al. 2002).

5.3 Hydrological Processes and Grazing Activities in Ecogeomorphic Systems

5.3.1 Hydrological Processes in Rangelands

The only source of water in rangelands, particularly in drylands, is precipitation, including interception of fog. When it comes, one portion returns to the atmosphere via evapotranspiration, another falls on the ground and runs off as overland flow and a third one is retained by the site and infiltrates into the soil profile becoming available to plant growth. Part of the infiltrated water is percolated well below the root zone and recharges underground water reserves. The portion of precipitation that evaporates constitutes its largest component (Holechek et al. 2004; Wilcox et al. 2006).

Evapotranspiration is composed of interception loss, evaporation from soil and transpiration from plants. Interception loss is the amount of precipitation captured by the plant canopy and evaporates, which means that it never reaches the soil surface. Its magnitude depends on the kind of vegetation (e.g. grasses or shrubs) and the character of precipitation (small or large storms) ranging from 20 to 40 % of the water budget (Wilcox et al. 2003). Evaporation from soil depends on how much of the rangeland area is covered by bare soil ranging from 30 to 80 % of the water budget (Wilcox et al. 2003). Transpiration from plants, finally, depends on the amount of water that is available to the plant (Ansley et al. 1991).

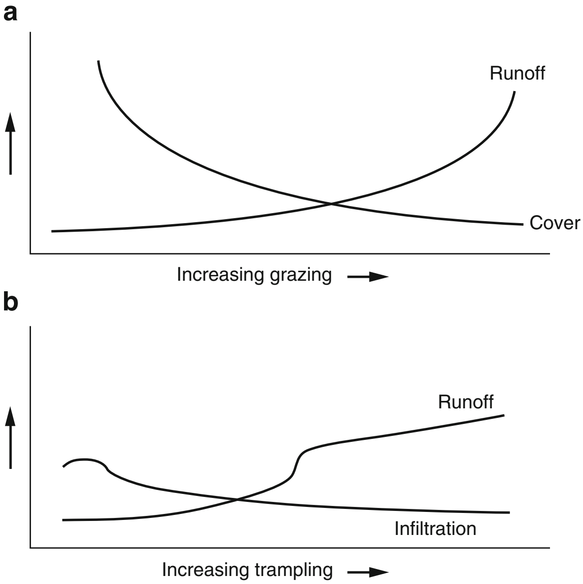
Runoff usually occurs when the amount of precipitation exceeds the infiltration and storage capacity of the soil (see Sect. 5.5). It is relatively small, usually accounting for less than 10 % of the annual water budget, but very important because it is a significant agent of erosion (Wilcox et al. 2003). The main factor that influences runoff is the amount of vegetation present to slow the water movement on the soil surface suggesting that it decreases as vegetation cover increases. For example, water runoff was increased by 15 times when vegetation cover was reduced by a factor of three times on a grassland with a 20 % slope (Helali and Nastis 1998). Excessive runoff results in soil erosion.

The water stored in the soil depends on its infiltration rate, the soil depth and texture. The infiltration rate is affected by the intensity of precipitation, the soil-surface properties and, especially, by the amount and kind of vegetation cover. The higher the infiltration rate, the higher the amount of precipitation stored in the soil for plant use and the higher is the possibility to recharge groundwater (Holechek et al. 2004). However, Wilcox et al. (2003) state that groundwater recharge is generally small in arid and semi-arid rangelands because only a few millimetres of water will move beyond the root zone each year. In some places, recharge has been demonstrated to have stopped in the later Pleistocene (e.g. Sandvig and Phillips 2006), while elsewhere, travel times of water may be so slow that recharge of the groundwater is taking place from water that infiltrated over 10,000 years ago, even if recharge from the surface has subsequently ceased (Seyfried et al. 2005).

5.3.2 Role of Vegetation Distribution

Hydrological processes in rangelands are greatly affected by vegetation distribution. In drylands, vegetation is sparsely distributed in space resulting in a two-phase mosaic composed of patches with high biomass alternating with areas of low biomass or bare soil (Noy-Meir 1973). As a result, a spatially variable infiltration field emerges with high infiltration rates in the vegetated patches and low infiltration rates in the bare patches (Tongway and Ludwig 2001). This pattern is due to the improved soil aggregation and macroporosity in the vegetated patches owing to the biological activity (e.g. termites, ants and earthworms: Whitford 2002) as well as to the vegetation roots (Ludwig et al. 2005). Moreover, bare patches act as source areas for runoff and vegetation patches as sink areas thus controlling soil erosion. According to Saco et al. (2007), the redistribution of water between these two kinds of patches is a fundamental process in drylands that may be altered if vegetation is disturbed.

Fig. 5.1 Simplified conceptual, functional relationships of the effects (**a**) of grazing on plant cover and runoff and (**b**) of trampling on infiltration and runoff (Modified from Squires 1998)



5.3.3 *Effects of Grazing Activities on Hydrological Processes*

Grazing activities that mainly affect hydrological processes in rangelands are plant defoliation and soil trampling. Removal of plant cover by grazing animals may increase the impact of raindrops, decrease the soil organic matter and soil aggregates, increase surface crusts and decrease infiltration rates. The severity of these effects depends very much on the grazing intensity or the soil trampling applied (Fig. 5.1). Grazing intensity may be light if the stocking rate (i.e. number of animals per unit area) is below the grazing capacity, moderate if the stocking rate is at the grazing capacity and heavy when the stocking rate is higher than the grazing capacity. In a review of the relevant literature, Gilford and Hawkins (1978) came to the conclusion that (a) ungrazed plots have higher infiltration rates than grazed plots; (b) light and moderate intensities have similar infiltration rates that in many situations do not differ from no grazing; and (c) heavy grazing causes higher reductions of infiltration rate than light and moderate intensities.

As a result of the reduction of infiltration rate, overland flow will increase, soil water content will decrease and runoff, sediment yields and soil erosion will also increase. This increase depends again on the grazing intensity or soil tramping (Fig. 5.1). Several authors have found that moderate grazing leads to a slight increase in soil erosion while heavy grazing also increases runoff resulting in yet higher erosion rates (e.g. Liacos 1962; Wood and Blackburn 1981; Thurow et al. 1986), not least because of the reduced soil cohesion makes the formation of rills more likely (Parsons and Wainwright 2006). More specifically, Sharma (1998) found that runoff yield increased under heavy grazing by nine and two times compared with the control and the light grazing, respectively, in the Indian arid zone; sediment concentration by two times over the light grazing and 1.5 times over the moderate grazing; and, soil loss by 41 and 11 times over the control and the light grazing, respectively. He attributed this increase to the creation of loose soil aggregates by cattle grazing and to the kinetic energy of the increased runoff rates. Thornes (2005) claims that there is a close connection between vegetation cover and erosion and suggests that erosion rate increases exponentially as vegetative cover decreases due to livestock grazing. However, the cause and effect between grazing and erosion has not been sufficiently understood yet and further studies are needed to also consider forage ecology and socio-economic aspects (Thornes 2007).

In terms of impact on species composition, heavy grazing results in the replacement of tall species by small ones and perennials by annuals. In a dryland of northern Patagonia, Bisigato and Bertiller (1997) found that grazing resulted in the replacement of large patches dominated by tall shrubs with high species richness by patches dominated by dwarf shrubs with low species richness and in the extinction of grass patches. Also, Aidoud et al. (1998) found that overgrazing in the Algerian steppe resulted in the destruction of the dominant perennial grass *Stipa tenacissima* and its replacement by weedy species. They concluded that the reduction of perennial species constitutes an efficient early-warning indicator of degradation in arid rangelands.

As far as soil compaction is concerned, livestock grazing besides disrupting the soil-aggregate stability also destroys cryptogamic crusts which are essential for hydrologic stability (Aguilar et al. 2009). Also, considerable research has been done in the USA on whether rotational grazing, which includes a rest period, has better hydrological effects than continuous grazing. Warren (1987) states that stocking rate is more important in determining infiltration rate than rotational grazing. Conversely, Savory and Butterfield (1999) claims that the consequences of trampling are a function of the time that soil and plants are exposed to animals rather than of the absolute number of animals. Comparable results are reported by Proffitt et al. (1995), who found that continuous grazing resulted in the deterioration of the topsoil physical characteristics and suggested removing the livestock for brief periods when the moisture content of the soil is close to its plastic limit. It turns out that the system that involves deferment of grazing such as the short-duration grazing one do have the potential to maintain hydrological parameters similar to the ungrazed areas (Blackburn et al. 1980; Holechek et al. 2004).

On the other hand, plant-community type seems to have more of an effect on infiltration rates than grazing treatments. From an experiment in Texas, USA that tested the effect of seven grazing treatments and three vegetation types on infiltration capacity, it was found that plant-community type explained 74 % of the variability in infiltration rates while grazing treatments explained only 34 % (Spaeth et al. 1996).

Ecohydrological interactions and associated productivity in rangelands are strongly affected by slope. Under low grazing intensities, vegetation disturbance and runoff are low on flat plains and gentle slopes but larger on steep slopes. Under

Fig. 5.2 An overgrazed area around a water trough used for watering sheep and goats at Ranti, Paphos, Cyprus in November 2007 (Source: VP Papanastasis)

overgrazing, however, not only are vegetation cover and biomass reduced but also is the rangeland's ability to conserve water even on flat plains and gentle slopes (Popp et al. 2009).

Finally, hydrological processes are affected by rangeland infrastructure. Grazing impact is higher near the sites where animals concentrate for drinking water or bedding than away from them (Fig. 5.2). Tongway et al. (2003) found that soil physical and chemical properties were improved as the distance from water increased.

5.4 Fire

5.4.1 Fire as a Natural and Cultural Agent

Fire is frequently regarded as a major agent of soil degradation. However, this is a view of the second half of the twentieth century. For millennia beforehand, fire was used as a tool to manage agricultural soils and rangelands (Naveh 1975; Pyne 1997; DeBano et al. 1998). Pasture, forest and scrubland, as well as agricultural land were managed by means of prescribed fires as this was the most economical

Fig. 5.3 *Chamaerops humilis* sprouts 1 week after the wildfire of Benicolet (1,500 ha) in Eastern Spain. April 2011. *Right*, view of the palms; and *left*, detail of the flowering

option for societies with low technological development. Moreover, fire is also a natural agent responsible for shaping vegetation in fire-prone environments (Naveh 1975; Trabaud 2002) as well as surface processes and resulting landforms (Shakesby and Doerr 2006). Although botanists found fire as an integral part of ecosystems (Bond and Keeley 2005) and understood that some landscapes are shaped by fire (Goldammer and Jenkins 1990) (Figs. 5.3 and 5.4), the view of soil scientists and geomorphologists was that fire is a disturbance that is unusual. Research findings have changed the minds of the scientist and the ideas of fire evolved during the last two decades. Now it is accepted that those disturbances are part of the ecosystem behaviour and fire should be seen as a natural and cultural agent (Cerdà and Robichaud 2009; Úbeda and Mataix 2008; Cerdà and Mataix 2009).

5.4.2 *Fire Effects on Soil and Water Losses*

Accurate measurements by means of natural and simulated rainfall demonstrated that after fire, soil erosion may increase by several orders of magnitude. This increase is due to the loss of plant cover, the increase in the raindrop-impact efficiency and the enhancement of water repellence either in surface or in the subsurface layers (DeBano 1971; Doerr and Thomas 2000; Doerr and Cerdà 2005). The infiltration capacity of soils is dramatically reduced due to the crust development by the raindrop impact, the clogging of the macropores by the ash and fine materials and the lack of vegetation (Robichaud 2000). However, is it now widely accepted that for some weeks the infiltration rates are encouraged by the ash (Cerdà 1998), although ash only lasts for several weeks (Cerdà and Doerr 2008). There have also been some suggestions that low-intensity burning may enhance infiltration due to particle aggregation and thus lead to smaller effects on the erosion rate (Lavee et al. 1995).

Fig. 5.4 Rangeland landscapes of the Mediterranean. A cultural and natural landscape shaped by the fire. *Upper pictures* show landscapes affected by fires 15 years prior and the *lower ones* show the 1 week after fire view

Table 5.1 Soil losses measured on fire-affected and control plots (Rodríguez et al. 1999–2000) under holm oak (*Quercus ilex*) and Aleppo pine (*Pinus halepensis*) cover in Zaragoza, Spain, modified after Úbeda (2000)

	Soil erosion
Vegetation cover	Mg ha^{-1} year^{-1}
Burnt oak forest	1.88
Oak forest (control)	0.00
Aleppo pine forest	11.80
Aleppo pine forest (control)	0.00

Due to the effect of fire, less water infiltrated also means more overland flow. Consequently, the lack of litter and vegetation as well as the effect of ash flowing with the overland flow contributes to increase the sediment detachment and transport. After a fire, the sediment concentration increases at least by an order of magnitude, and more water and sediment flowing results in higher erosion rates.

Table 5.1 shows an example of how the soil losses increase after fire. The contribution of the research by Rodríguez et al. (1999–2000) also suggests that the effect of fire was more intense on Aleppo pine plantations, while the natural oak forest recovered earlier and the soil losses were lower. The research done by

Table 5.2 Soil losses after forest fire: the impact of fire intensity on forests in Catalonia, modified after Úbeda (2000)

Fire intensity	Soil erosion Mg ha^{-1} year^{-1}
Burnt (low intensity)	0.20
Burnt (medium intensity)	8.46
Burnt (high intensity)	40.74
Control plot	0.08

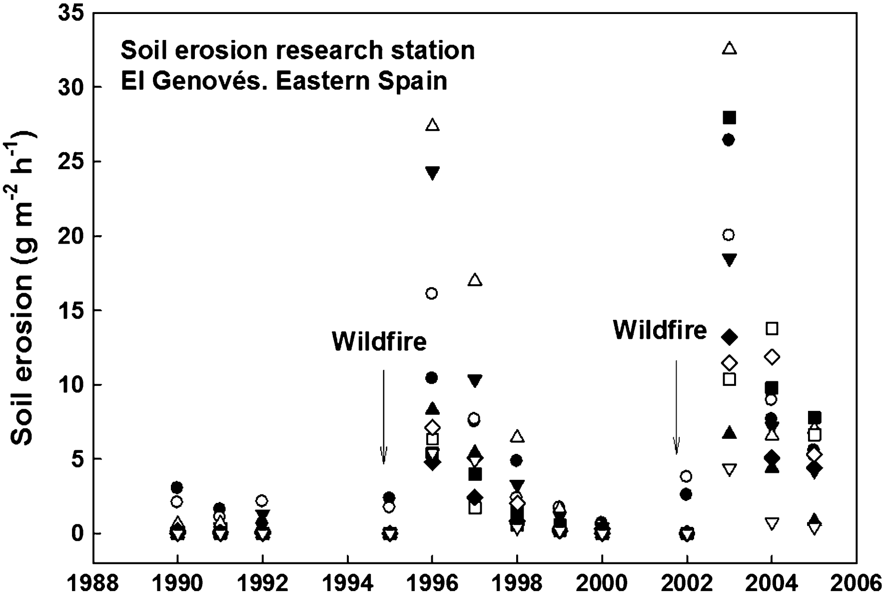


Fig. 5.5 Long-term measurements of soil erosion under simulated rainfall (55 mm h^{-1}, 1 h on 0.25-m^2 plots) show that soil erosion is triggered by the wildfires but that is controlled by the vegetation recovery after 2–5 years. Mediterranean Maquia type vegetation (*Quercus coccifera* dominant species). Soil-erosion research site of El Genoves (Data modified after Cerdà and Doerr (2005) and unpublished data)

Úbeda (2000) also suggests that the key factor in controlling post-fire soil losses is related to the fire intensity (Table 5.2). These results support the idea that traditional prescribed and recurrent fires were more sustainable than the current wildfires.

5.4.3 *Post-fire Recovery*

Researchers have demonstrated during the last two decades that post-fire soil losses are controlled by the type of vegetation recovery (Moody and Martin 2001). Sometimes the recovery is fast enough to return to the previous soil erosion rates in 4–5 years (Fig. 5.5). The post-fire soil erosion rates are determined by the rainfall.

Fig. 5.6 The recovery of vegetation is supported by adaptations to fire such as sprouting or germination, but also the patchy distribution of the vegetation is a key strategy. The abandoned terraces are extremely efficient to control the soil losses after the forest fires

Under Mediterranean climatic conditions, most of the soil and water losses are due to the high-intensity rainfall events during the summer thunderstorms or autumn rainfall events which are characterized by high intensities. At other times, it is not uncommon to have almost no erosion due to the low intensity of rainfall events.

5.4.4 The Natural and Cultural Controls of Soil Erosion on Mediterranean-Type Ecosystems

On Mediterranean slopes, the high erosion rates found after forest fires are usually controlled by two main factors: one is natural, the patchy distribution of the vegetation; and the other one is cultural, the abandoned anthropic agriculture terraces.

The measurements of soil and water losses on fire-affected land suggests that the surfaces that were covered by plants before the fire show lower soil-erosion rates and the vegetation recovery is faster (Fig. 5.6, Table 5.3). The abandoned terraces are still active after the fires and they control the soil and water losses. This is a cultural landscape inherited and now in danger due to abandonment (Fig. 5.7). Moreover, it should be highlighted that the abandoned terraces are active foci of intense fire, but a policy to recover the terraces and the surrounding landscape will contribute to reduce the fire intensities and affected surface.

Table 5.3 Soil losses measured by means of 1-m^2 plots at the Bixquert fire in Eastern Spain

Soil erosion	15-year after fire		1-year after fire	
Mg ha^{-1} a^{-1}	Quercus patch	Bare patch	Quercus patch	Bare patch
1	0	0.65	0.65	5.65
2	0	0.96	0.74	4.89
3	0	2.32	0.58	6.32
3	0	1.58	0.14	1.58
3	0	1.65	1.65	2.36
3	0	0.10	2.24	4.58
3	0	0.59	0.98	9.36
3	0	0.30	2.05	7.56
3	0	4.02	1.56	8.60
3	0	1.58	1.25	4.44
3	0	1.87	1.74	5.68
3	0	1.45	1.85	8.35
3	0	1.69	2.65	9.32

Data from Cerdà and Jordán (2005) and unpublished data

The patchy distribution is efficient to control the soil losses, and this strategy is maintained after the forest fire and contribute to the recovery of vegetation as the island of fertility contribute to a faster recovery

Fig. 5.7 The land abandonment of cultural landscapes such as the Mediterranean ones contributes to wildfires. The human use of the land until the 1960s contributed to prescribed, low-intensity forest fires that were used as a management tool. The terraces were part of the systems to control fire and now are the origin of the high intensity forest fires due to the large accumulations of biomass. The abandoned terraces control the soil losses, but they are being degraded leading in places to the formation of gullies and accelerated erosion rates

5.5 Hillslope-Landscape Scale Erosion Processes: Water

To generate erosion by water, one first has to produce the water. Although rare – by definition – in drylands, the pattern of precipitation often as short, high-intensity events means that runoff and erosion are generally significant processes in all but the most hyperarid zones. Runoff production under conditions of high-intensity rainfall is generally by the Hortonian, or infiltration-excess method, in which the rainfall intensity is too high to be accommodated by the infiltration capacity of the soil. Water thus ponds at the surface until there is a sufficient amount to fill the detention storage at the surface, when it starts to flow.

Infiltration is controlled by grain size and sorting (Brooks and Corey 1964), as well as surface characteristics and vegetation. Dryland soils are typically poor in organic materials which otherwise lead to particle aggregation and the formation of bigger pores (Wainwright 2009a); sandy soils with a mixture of small amounts of finer material are prone to crusting (Poesen 1992; Valentin and Bresson 1992). Mechanical sorting by wind and water of the surface layer can create crusts (Valentin and Bresson 1992). Biological crusts also form in the presence of cyanobacteria, which are present in even the driest deserts (Belnap 2006). Stony soils are not uncommon, and the predominance of physical weathering in drylands means that soils remain stony unless other sorting mechanisms occur. Concentrations of stones at the surface can significantly reduce infiltration rates (Poesen 1992), although in some circumstances they can act as a mulch and increase infiltration (Bunte and Poesen 1994). Where vegetation is present, it can enhance infiltration by concentration of stemflow (Abrahams et al. 2003), by root pathways (Martinez-Meza and Whitford 1996; Cerdà 1997) and macropores formed by decayed roots (Whitford et al. 1988), by the production of organic materials enhancing aggregation and by feedbacks with animals burrowing in the shelter of plants (Elkins et al. 1986; Cammeraat et al. 2002). As noted above, fire and grazing animals can also have significant effects on the production of runoff.

Although other forms of runoff production such as saturation-excess or return flow are less common in drylands, they can be locally important. In particular, in shallow, stony soils in bedrock pockets, storage capacity can become filled and then flow over the bedrock. In some Mediterranean environments, there may be sufficient moisture in the winter, especially in upland catchments, to produce saturation. In areas such as badlands where pipeflow may be important, runoff is produced at the surface where pipes collapse, or emerge from a free face (Farifteh and Soeters 1999). A full description of infiltration and runoff processes and controls can be found in Wainwright and Bracken (2011).

Before the onset of overland flow, sediment movement occurs by splash, which is a relatively inefficient, diffusive process. There will typically be a net downslope movement (Savat 1981) although the effects of microtopography may cancel out the effect of a dominant slope (Torri and Poesen 1992). Most particles are moved a few tens of millimetres, and usually no more than 0.3 m (Wainwright et al. 2008), so that the dominant effect in many drylands is the differential rates of splash between and under vegetation, which can contribute to the formation of islands of fertility (Parsons et al. 1992; see also Chap. 4). In the initial phases of overland flow, sediment transport is still relatively inefficient and localized. Detachment is still only by raindrop impact (Ellison 1945; Young and Wiersma 1973), and the water layer protects the surface from some of the energy of impact (Torri et al. 1987) so that although particles may travel further once detached (Parsons et al. 1998), fewer are detached than before the onset of flow, so there is a peak of movement in the 5–10 m from the onset of flow (Parsons et al. 2004). Once transitional flows occur so

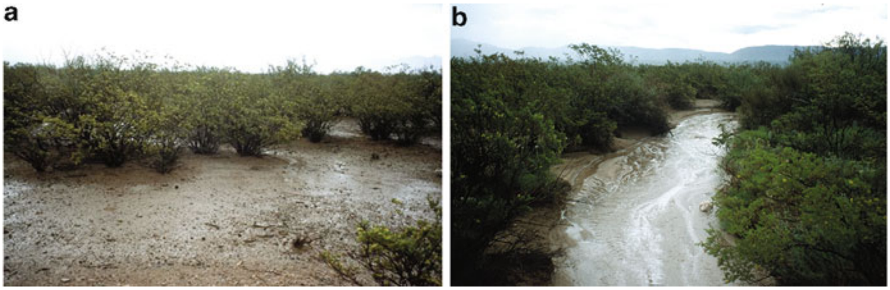


Fig. 5.8 Overland-flow generation on hillslopes at Jornada LTER site, New Mexico, USA during intense summer rainfall. (**a**) Shows the effects of higher interception and infiltration rates under shrubs in minimizing runoff production under vegetation, while the bare, stony surface in the foreground has enabled some diffuse, unconcentrated flow to form; between the shrubs in the middle ground, it is possible to see the start of concentrated flows; and (**b**) shows how flow concentrations further downslope can cause well-defined rill flows. The dominant forms of sediment movement are splash and unconcentrated overland flow transport in (**a**) and concentrated (rill) transport in (**b**) (Source: J Wainwright)

some turbulence is encountered, and especially once fully turbulent flows develop, flow detachment occurs, and there is typically an order of magnitude increase in the amount of sediment produced and the distance it travels (Poesen 1987). Flow detachment is able to produce rills and ultimately gullies by incising either through the root mat, or into bare patches between vegetation, or already-disturbed sediment, or through seepage erosion. Turbulence is controlled by the feedbacks with surface roughness, which are typically complex on dryland slopes (Abrahams et al. 1990; Smith et al. 2007, 2011). Parsons and Wainwright (2006) discuss how feedbacks between flow extremes and surface characteristics through flow-roughness produce incision in the conditions noted above when the surface becomes increasingly heterogeneous. One mechanism for the generation of such heterogeneity is through grazing of large herbivores (see also Sect. 5.3.3). However, reinfiltration of water is also a consequence of such heterogeneity (Yair and Kossovsky 2002), and high temporal variation in rainfall intensity produces even more pronounced reinfiltration effects (Wainwright and Parsons 2002), so that many dryland slopes demonstrate a buffering of the increasing downslope effectiveness of erosion (Fig. 5.8).

Where rills and gullies form, they dominate the sediment flux from hillslopes, and carry out significant amounts of geomorphic work, increasing with distance along a slope, unless supply limitations occur (Parsons et al. 2004). The extent to which the effects of this work will be transmitted to the channel system will be controlled by the specific geometry of the landscape (Wainwright et al. 2001). Hillslope-channel coupling has been used as a term in the geomorphological literature to characterize the effectiveness of this transmission (Brunsden and Thornes 1979; Brunsden 1993). Michaelides and Wainwright (2003: 1442) define hillslope-channel coupling as “the connectivity of the hydrologic and/or geomorphic processes between hillslopes and river channels. It is the effectiveness, direction and speed with which localized changes are transmitted away from the source (hillslope) and propagated throughout the hillslope system to the channel and ultimately to the catchment outlet." They go on to show how catchments with one or more floodplains can reduce the transmission of water (and sediment), and especially that the coupling is particularly sensitive to floodplain roughness. However, in some circumstances, floodplains can enhance transmission, so that an understanding of the variability as one passes from hillslope to floodplain to channel is critical. Wainwright (2006) demonstrated that coupling characteristics can have significant long-term effects on catchment evolution, so that in settings where environmental change is important, there can be major effects of "memory" of past conditions.

While flows in channels are much more effective at entraining and transporting sediment than hillslope flows (Powell 2009) and travel distances of particles are typically tens to hundreds of metres (Hassan et al. 1992), variability in flow conditions significantly affects the continuity of transport (Schumm 1973; Bull 1997; Wainwright et al. 2002). Although care must be taken to emphasize the diversity of dryland channel types, transmission losses are a common feature in dryland channels, whether they be controlled by infiltration losses through the bed or by evapo(transpi)ration (Tooth and Nanson 2011). Vegetation is often concentrated in the riparian zone, which can enhance this effect (Seyfried et al. 2005), with further feedbacks to floodplain roughness and the hillslope-channel coupling noted above.

In terms of pattern, the interconnected pathways of water flow and sediment movement mediate the presence of resources for vegetation growth (Wainwright 2009b), which then interacts with and affects water, sediment and nutrient movements. Recent emphases have been on the study of connectivity of flow pathways (Bracken and Croke 2007; Tetzlaff et al. 2009) as integrated ways of evaluating the hydrological behaviour of catchments, and their sediment-transfer processes. These connectivity-based approaches seem to show significant promise in the understanding of land degradation (Turnbull et al. 2008, 2012; Wainwright et al. 2011).

5.6 Hillslope-Landscape Scale Erosion Processes: Wind

Wind erosion, including erosion and transport of particles by wind and subsequent downwind deposition of these particles, is a fundamental abiotic process in the world's drylands. Wind erosion has direct impacts on global biogeochemical cycles, climate, human health, agricultural production and land degradation (Ravi et al. 2010). These wind-related ecogeomorphic processes occur at different scales, ranging from plant-interspace (centimetres to a few metres) to global (thousands of kilometres) (Okin et al. 2006). Here, the controls and consequences of wind erosion at hillslope-landscape scale (tens of metres to tens of kilometres) are summarized, with special consideration given to the key issues and current research frontiers of wind erosion in drylands.

At the landscape scale, wind is an important geomorphic process (Greeley and Iversen 1985). The two principal mechanisms for the transport of soil material by wind are saltation (for diameters ranging from 20 to 500 μm) and suspension (for diameters <20 μm). Both processes redistribute soil and associated nutrients and organic material at landscape scale and beyond (Field et al. 2009). During the process of wind erosion, saltating particles carrying most of the mass and momentum, therefore they can have considerable physical effects on existing vegetation, including burial, exposure of below-ground plant tissue, abrasion of plant tissue, and leaf stripping. The physical effects of saltation have been related to the reduced plant growth and mortality, and rapid changes in ecosystem structure in the northern Chihuahuan Desert (Okin et al. 2001). In addition, saltation can lead to the movement of coarse material from plant interspaces to nearby plant canopies, and the growth of mesquite dunelands into former grassland. On the other hand, soil nutrients (e.g. nitrogen and phosphorus) and organic material are disproportionately associated with smaller soil particles, therefore emission of soil particles with diameters <20 μm has significant impacts on soil-nutrient status of local soils as well as downwind ecosystems that may be thousands of kilometres away (Okin et al. 2004). The airborne fine particles may be trapped by local vegetation, working together with redistribution of soil resources by saltation, to contribute to the formation of fertile islands found throughout dryland regions (Schlesinger et al. 1990). At the Jornada Experimental Range of southern New Mexico, Li et al. (2007, 2008) showed that substantial soil-nutrient depletion and redistribution (e.g. change of autocorrelation range) occurred at the time scale of 2–3 years due to enhanced saltation and dust emission.

Wind-erosion flux has been quantified by numerous studies using different approaches, and flux equations suggested by these authors were further verified by wind tunnels, field experiments, and alternate theoretical derivations. In general, all these equations show that horizontal mass flux is strongly regulated by the wind shear velocity (u_*) and the threshold shear velocity (u_{*t}). The type, cover, and arrangement of vegetation have the strongest influence on the ability of wind to reach the soil surface. The impact of vegetation on wind erosion has been studied by Raupach et al. (1993). More recently, Okin (2008) developed a wind-erosion model which focuses more on the spatial distribution of the vegetation instead of the overall amount used by most of the previous wind erosion schemes. The new model provided satisfactory estimates of wind erosion flux over a large number of field sites with different plant communities across the western United States. In addition, at the landscape scale, a few other factors may affect the erodibility of the soil surface and the magnitude of sediment flux due to wind erosion, which may include the distance over which the erosion occurs (fetch effect), land use or management that impacts vegetation cover, agricultural activities, grazing, and off-road vehicle traffic (Okin et al. 2006).

In recent years, there is a growing interest in the scientific community to understand and model wind erosion-related processes, which is indicated by the increasing trend in aeolian publications in the past few decades (Stout et al. 2009).

Nevertheless, some fundamental aspects of aeolian transport remain unresolved, in part due to uncertainty associated with lack of reliable, long-term erosion and emission data sets from different ecosystems, and in part due to difficulty in accurately determining some key parameters affecting wind erosion (e.g. soil moisture, roughness elements). From the modelling aspect, accurate, reliable and direct measurement of the hydroclimatic factors affecting wind erosion and dust emissions are needed to validate erosion models and to assess the intensity of aeolian processes in dryland environments. In addition, the modelling of wind erosion and dust emission is still limited by the need to parameterize complex aeolian processes and by the lack of methods to determine the uncertainty associated with model predictions. Wind erosion, in combination with water erosion, is thought to be responsible for a significant proportion of the degraded land (Middleton and Thomas 1997; Lal 2001). Another major challenge ahead is to develop methodologies and models to address the impacts of wind- and water-driven erosion collectively and to develop a more integrated perspective of aeolian-hydrologic dynamics (Ravi et al. 2010; Breshears et al. 2003; Belnap et al. 2011).

Global circulation models have predicted an increase in aridity in many dryland systems around the world (e.g. Seager et al. 2009), which could increase the susceptibility of the landscape to wind erosion. In addition to climate change, more frequent and larger dust storms are likely to occur from dryland regions as more dryland areas are trampled, cleared of vegetation, ploughed, and/or converted from perennial to annual plants. Increases in wind-erosion rates due to climatic changes and land-use change result in enhanced losses of soil resources and subsequent loss of vegetation cover. The loss of vegetation cover results in the loss of vital ecosystem services, such as primary production and carbon sequestration (e.g. Chapin III et al. 1997).

5.7 Interactions with Other Long-Range Ecological Processes – Plants

A propagule is any piece of plant material that can propagate the plant. A progagule can be a woody part – hardwood or softwood – a part of a leaf, or any part of the plant that can carry out asexual reproduction. Sexual reproduction in plants is done by spores or seeds. The distribution of the propagules will determine the life of the plant and the evolution of the landscape as plants control the processes and rates of soil formation and degradation.

Land degradation can be defined as being due to the reduction of soil fertility, and as a consequence sparse vegetation cover which encourages soil erosion, which in turn further decreases soil fertility. The initiation of this land-degradation feedback can be initiated by a reduction in vegetation, but it can also be due to the lack of soil conditions that will contribute to the establishment of propagules.

The interaction between vegetation and erosion is discussed here as erosion processes can displace propagules to either fertile or unfertile locations, and

Fig. 5.9 *Left*, view of *Stipa tenacissima* tussocks in Southeastern Spain as an example of vegetation distribution and seed reallocation due to the patchy distribution of runoff generation (Cerdà 1997). *Right*, Keuper clay badlands in Eastern Spain where the intense soil losses and the lack of soil moisture reduce the ability of the vegetation to recover on the barren land (Cerdà 1999)

propagules can control the soil losses when established. Seeds are the best example of this interaction between soil, erosion and vegetation. Soil erosion takes place on slopes where vegetation is sparse (see Chap. 4). The transport of soil particles and solutes includes the reallocation of those materials and also the seeds that are transported. On vegetated slopes, the rainfall infiltrates and consequently soil erosion is negligible, but on semi-arid, arid and human-made landscapes, the low vegetation cover determines that surface erosion will redistribute the seeds, and thus where seeds are available the recovery will take place.

The redistribution of the seeds is determined by the size and the form of the seeds (Cerdà and García Fayos 2002). This removal and deposition of seeds determines the vegetation composition (Bochet et al. 2010). The importance of seed redistribution is a key factor in the pattern of growth. The vegetation distribution is due to the redistribution of seeds during surface erosion, so that some landscapes show a patchy distribution of vegetation as the location of existing plants determines the best soil conditions and also the plants act as sinks of the seeds, producing the so-called nursemaid effect (Fig. 5.9).

Badland surfaces may form other natural environments where there is a small seed bank (García Fayos et al. 1995). Most of the seeds here have strategies to avoid surface erosion, including mucilages and appendices (wings, pappi etc.) (Cerdà and García Fayos 1997; García Fayos and Cerdà 1997). The barren surfaces of the badlands are also due to other factors such as the fast drying of the soil that makes germination unviable (García Fayos et al. 2000).

Anthropic landscapes usually have low vegetation cover. Propagules can contribute to the reestablishment of vegetation and soil recovery (Tsuyuzaki and Titus 1996). The use of seeds can help vegetation recovery such as in the case of land affected by forest fires, agricultural soils and on road and railway embankments (Fig. 5.10). The dynamics of seed erosion contribute to the regeneration of degraded land (Lawson et al. 2004; Rey et al. 2005; Aerts et al. 2006).

Fig. 5.10 View of the use of seeds to control soil losses on road embankments (Photo: A Cerdà)

5.8 Interactions with Other Long-Range Ecological Processes – Animals

Vertebrate and invertebrate animals have been shown repeatedly to function as major drivers of rangeland dynamics. The fauna of rangelands include important species of herbivores, granivores, frugivores, pollinators, predators, parasitoids, fungivores, decomposers, and disease vectors. Herbivorous animals influence plant-species composition, distribution, percentage cover, and primary production, and through their urine, faeces and carcasses, redistribute nutrients and moisture. Soil-dwelling vertebrates and invertebrates create patches of soil disturbance via tunnelling and burrow construction, which alters soil texture, aeration, moisture content and vertical nutrient distributions. Pollinators facilitate successful reproduction cycles of many plant species, while granivores and frugivores disperse seeds to favourable germination sites. Predators and parasitoids may provide top-down controls on both vertebrate and invertebrate herbivores, limiting potentially devastating population outbreaks. Animals often carry bacterial, viral or protozoan disease organisms, serving as hosts and/or vectors that disperse infections to both plants and animals. Finally, the rangeland fauna includes a large number of decomposer species, which break down plant litter and animal faeces, cycling nutrients and enriching soils.

From a mechanistic viewpoint, the impacts on rangelands of these different animal taxa are often a function of their abundance, distribution and activity periods in a given area. Terrestrial animals, the vast majority of which are mobile species, typically move within and across rangelands at different spatial and temporal scales. Species with sedentary habits or low dispersal characteristics remain within localized home ranges in a suitable habitat for their entire lifetimes, whereas more mobile species often exhibit long-range dispersal or migratory behaviour among local, regional or continental land areas. As such, the density (numbers of individuals per unit area) of any given animal species varies in space and time, and can be driven by local resident population dynamics (natality and mortality rates) and/or by immigration and emigration of non-residents.

For the purposes of the following discussion, we can consider four categories of faunal assemblages that have differing spatio-temporal scales in both their movements within or among rangeland regions, and their subsequent impacts on rangeland resources. These are (1) year-round native resident species, (2) seasonally migratory native species, (3) episodic native invasive species, and (4) non-native introduced/invasive species. The impacts and examples of these four groups are detailed below.

For year-round resident animal species, their interactions on rangelands generally occur within their home ranges or territories at scales ranging from fractions of hectares (e.g. rodents) to hundreds of square kilometres (carnivores, ungulates). Densities of each species are basically regulated by food and shelter resources, with populations increasing via successful reproduction and recruitment following "good" periods (e.g. above-average rainfall leading to increased primary production or prey abundance), and declining during periods of poor food resources (i.e., the concept of "pulsed resources" (Holt 2008; Yang et al. 2008)). Herbivore populations (especially multi-voltine species like some rodents) respond quickly to favourable plant food resources, while predators exhibit time-lagged responses as their reproduction success increases following their prey's increase. Home-range size in some animal species declines with increased density, potentially due to both resource availability (e.g. abundant food resources require less movement to acquire) and/or social pressures (e.g. conspecific antagonistic encounters). However, territorial animals actively defend their land area (resource base), and often do not show large changes in resident densities (except for seasonal or annual reproductive success).

The scale and impact of activities of these resident animal species is defined by each species' movement capabilities, resource needs, and period of activities. Operating within their home range or territory, many animals behave as central-place foragers (e.g. birds in nests, rodents or rabbits in burrows, ants in underground colonies), having one or two main residences and foraging over a much wider area. This behaviour can influence the spatial distribution of rangeland resources, such as soil seed densities and plant cover (less near the central "nest", higher values further away). Other species may not use central-place foraging, but rather roam throughout the home range; however, individuals often display preferences for certain microhabitats (e.g. shrub sub-canopy areas versus shrub interspaces), with subsequent unevenly concentrated impacts on resources. Temporal differences in resident animal activities also drive ecological interactions in rangelands; species that remain active year-round have substantially different impacts on resources than species which hibernate or aestivate for portions of the year.

At larger spatial scales, migratory native species of animals have major influences on rangeland processes. Many species have adapted behaviours that optimize use of particular rangelands in an intensive manner during certain parts of the year. Migratory birds are perhaps the best example, with some non-resident breeding birds using rangelands for reproduction during spring and summer, but migrating to more favourable regions during winter; conversely, many bird species that breed in temperate ecosystems spend winters in sub-tropical and tropical rangelands, feeding on seeds produced during the summer and autumn. Given their long-distance movements and ability to find favourable resources, mixed-species winter flocks of granivorous birds can be abundant on rangelands having good rains and seed production the previous summer, and very rare on rangelands with antecedent drought conditions (Wiens 1989). Similarly, ungulate species, such as elk, buffalo, pronghorn, antelope, and wildebeest, exhibit migratory movements between different ranges (e.g. to and from summer and winter ranges in North America, or dry- and wet-season ranges in Africa) in an effort to obtain sufficient high-quality forage under varying climatic conditions (Toweill and Thomas 2002; Holdo et al. 2009).

In addition to animal impacts from resident and migratory species, rangelands may also be subjected to transient, episodic invasions by dispersing native animal species. These irregular and unpredictable events are driven by phenomena occurring outside the immediate area, and can have extremely severe impacts on the rangeland's vegetation. For example, the desert locust (Orthoptera: Acrididae: *Schistocerca gregaria*) of Africa and the Middle East continues to plague farmers and ranchers with periodic outbreaks, during which dispersing swarms composed of millions of locusts travel hundreds, and sometimes thousands, of kilometres from their original breeding grounds; such events cause major damage to crops and rangeland vegetation, and drive widespread and expensive control efforts (Lomer et al. 2001; van Huis et al. 2007). In North America, episodic outbreaks of the Mormon cricket (Orthoptera: Tettigoniidae: *Anabrus simplex*) have comparable effects on rangelands after long-distance swarm marches (Lorch et al. 2005).

Finally, long-distance movements and subsequent introduction and establishment of non-native invasive or feral animal species can have severe impacts on rangeland structure and processes. Non-native species introduced to new geographic areas can exhibit rapid, exponential population increases if their new environment is free of predators and/or disease. A classic example is the introduction of rabbits to Australia (Fenner 2010). The European rabbit (*Oryctolagus cuniculus*) was deliberately released in Australia in 1879 as a game animal, and soon spread across the continent to become a serious rangeland pest species. During 1901–1907, over 3,000 km of rabbit-proof fences were constructed to prevent rabbit dispersal (Broomhall 1991); since then, several biological control agents, including diseases and predators, have been attempted with mixed success (Fenner 2010).

The simultaneous interactions of all these faunal species groups described above (residents, migratory species, episodic dispersers, and non-native domesticated or wild invasives) on a single rangeland region can be profound, especially when combined with human politico-socio-economic activities. An example of how long-range movements of wildlife, livestock and humans interact to degrade rangelands can be found in the excellent account by West (1995) of the near extinction of the American buffalo (*Bison bison*) during the nineteenth century (1830s to 1860s) in the central plains of North America by Native American and Anglo settler activities. An estimated 3–4 million buffalo inhabited the central Great Plains (eastern Wyoming and Colorado, and western Nebraska and Kansas) in the 1830s, and were part of a much larger population of perhaps 20–30 million buffalo continent-wide. The central plains were surrounded by many Native American tribes, which controlled sections of the perimeter, but not the interior: the Cheyennes, Arapahoes and Lakota Sioux dominated in the northwestern region, the Comanches and Kiowas in the southwest, the Pawnees and Otoes in the northeast, and the Osages in the southeast. Continuous hostilities among these tribes prevented widespread hunting of the buffalo herds in this "no man's land" of the central plains, until the signing of the Great Peace of 1840, which ended the inter-tribal wars. Native American buffalo hunting then increased markedly for tribal sustenance as well as hide sales to eastern merchants, greatly reducing the herds from the Rocky Mountains towards the east. The Native American population on the central plains increased dramatically, from an estimated 10,000 in 1825 to over 20,000 by 1845, while the number of horses maintained by the tribes increased from a few thousand in 1840 to over 130,000 by 1850. Migratory buffalo herds, that had for millennia spent their summers grazing on upland short-grass rangelands and sheltering during the winters in the river valleys among tallgrass meadows and cottonwood riparian forests, gradually found their summer rangelands grazed by horses and their winter range occupied by hundreds of villages. Winter occupation by tribal villages removed the shelterwood, and horse herds usurped the valley winter pastures. With the advent of the American cowboy and the beginning of cattle drives up the river valleys to western rangelands, the tallgrass river bottom ranges were repeatedly grazed during the summers, leaving only short stubble by the onset of winter for returning buffalo. The introduction of livestock-borne brucellosis and bovine tuberculosis to the buffalo took an additional toll on the herds. On top of these human-driven pressures, a severe drought set in on the region from 1848 to 1862, further depleting the rangeland resources and forcing the remaining buffalo into closer proximity to human hunters. With the crash of the buffalo herd and the increasing effects of the drought, the Native American population declined by 25 % from 1845 to 1860, accompanied by abject poverty and starvation.

Taken collectively, all these impacts not only brought the buffalo to the brink of extinction, but also significantly degraded the rangeland resources of the central plains. The interactions of the native wildlife, introduced livestock, new diseases, climate change and human activities on the central plains more than a century ago provide clear lessons on how animal movements can influence rangeland degradation, and illustrate the need for complex planning incorporating animal ecology when developing management strategies for rangeland restoration and sustainability.

5.9 Feedbacks Between Ecological, Geomorphological and Biogeochemical Processes

The state of a system is characterized by its structure and function and by the nature of feedbacks between ecological, geomorphological and biogeochemical processes across a continuum of spatial and temporal scales. Drylands are complex and dynamic systems, which respond in various ways to a variety of exogenous forces (for example changes in climate and human-induced disturbances) (Turnbull et al. 2012). A change in one or more exogenous forces may alter the nature of feedbacks between ecological, geomorphological and biogeochemical processes and alter ecosystem structure and function, facilitating a transition to an alternative stable state (Holling et al. 1973; Wilson and Agnew 1992; Suding and Hobbs 2009; Schlesinger et al. 1990; Lawrence et al. 2007).

Feedbacks are either *amplifying* (alternatively termed positive feedbacks) or *stabilizing* (alternatively termed negative feedbacks). Within a complex system, there are likely to be multiple different types of amplifying and stabilizing feedbacks occurring simultaneously, operating across multiple spatial and temporal scales. The compound effect of these co-occurring feedbacks determines the overall resilience of the system. Amplifying feedbacks decrease the resilience of a state, rendering the system more susceptible to state change following a small disturbance or slight change in an environmental driver, while stabilizing feedbacks increase the resilience of a given state, rendering it less susceptible to state change following a disturbance or change in an environmental driver.

In many dryland regions vegetation occurs as a two-phase mosaic, consisting of vegetated and non-vegetated areas (Valentin et al. 1999). Processes and feedbacks operating within this two-phase mosaic (outlined in Chap. 4) have implications for feedbacks between ecology, geomorphology and biogeochemistry, in terms of the catchment-scale runoff response, soil-moisture distribution, land surface-climate feedbacks and wind erosion. Specific examples of these feedbacks are outlined in turn.

The extent to which the small-scale two-phase vegetation mosaic is configured in a spatially relevant pattern determines the extent to which runoff generated at the patch scale yields a hydrological response and erosion at the catchment scale (Bartley et al. 2006; Boer and Puidefabrgas 2005; Turnbull et al. 2010). In the case of shrub encroachment into grasslands, in an amplifying feedback loop, a decrease in grass cover and concurrent increase in the spatial connectivity of the bare mosaic increases the connectivity of runoff-generating areas (Wainwright et al. 2002). The higher discharge of runoff associated with this increase in [functional] connectivity increases the capacity of flow to entrain and transport sediment, leading to an increase in erosion, which increases the topographic differentiation between vegetated and non-vegetated areas. In an amplifying feedback loop, the reinforced microtopography further concentrates runoff in major flow lines, further increasing the removal of nutrients and sediment from non-vegetated areas (Turnbull et al. 2011), thus decreasing the resilience of the grass-dominated state.

Changes in runoff dynamics have a clear effect on the spatio-temporal characteristics of soil moisture. Under dry conditions, evapotranspiration governs soil moisture dynamics (Grayson et al. 1997). During runoff events however, non-local controls appear to govern the spatial distribution of soil moisture, with soil-moisture content being controlled by the lateral redistribution of water which creates particularly moist areas along drainage lines. This switch between local controls (vertical fluxes) and non-local controls (horizontal fluxes) on soil moisture dynamics has been observed over a grassland to shrubland transition with a more pronounced influence of non-local controls in shrub-dominated landscapes (Turnbull et al. 2010, see also Chap. 6). The higher antecedent soil-moisture content typically observed in non-vegetated areas increases the potential for runoff generation areas if a subsequent rainfall event occurs before substantial drying out of the soil occurs (which is sometimes the case because of land-surface—atmosphere feedbacks) and thus, an amplifying feedback increases the likelihood for future runoff generation, sediment entrainment and transport, and nutrient losses from these areas. Turnbull et al. (2010) observed that even when the landscape is characterized by relatively fine scale patch pattern (such as a grass-non vegetated mosaic), under extreme high-magnitude rainfall events, the effects of fine-scale patch patterns may be overridden and the broader scale, non-local controls may govern the distribution of soil-moisture as is more often found in shrubland.

Feedbacks occurring at the hillslope-landscape scale alter atmospheric processes by altering the portioning of sensible and latent heat fluxes due to changes in runoff, evapotranspiration and soil moisture, and also affect climate by altering gaseous exchanges with the atmosphere (such as CO_2 flux and nitrous oxide emissions) resulting from changes in biogeochemical cycling associated with vegetation change and land degradation. In terms of land-surface and precipitation interactions, the distribution of vegetation and associated spatial variations in soil moisture modifies the atmospheric energy and water budgets, thus affecting regional precipitation (Charney 1975; Hong and Pan 2000). Vegetation increases the evapotranspiration flux, since it can access deeper down soil-moisture stores (Dekker et al. 2007), causing an amplifying feedback between vegetation and precipitation. An increase in vegetation increases evapotranspiration, which increases the amount of rainfall in the area, increasing soil moisture content, thus enabling more vegetation to grow. Conversely, a decrease in vegetation cover will decrease rainfall. Decreases in vegetation cover and an increase in bare ground associated with shrub encroachment into grassland have been shown to cause increased diurnal soil heat fluxes and an increase in nocturnal upward long-wave radiation, conditions which favour shrub survival (D'Odorico et al. 2010 and He et al. 2010).

Changes in vegetation patch-patterns also affect sediment loss due to aeolian transport (Field et al. 2010; Ravi et al. 2010; Fig. 5.11). The distribution of vegetation exerts a primary control over wind erosion, by directly sheltering the soil surface from wind, by reducing wind velocities due to increased surface friction, and by trapping wind-entrained particles, reducing both vertical and horizontal fluxes (Okin et al. 2006). At the plant-patch scale wind can entrain sediment, and deposit coarse material locally, while removing dust (fine material) for long-range transport

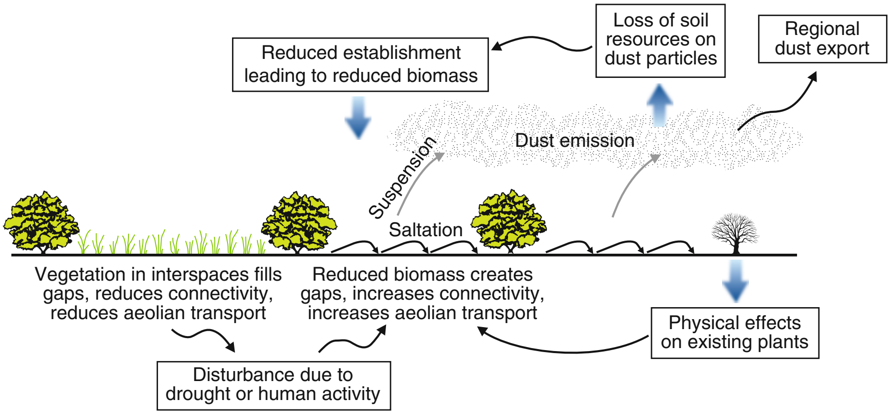


Fig. 5.11 Feedbacks between wind erosion, ecosystem function and ecosystem structure in drylands (Source: Field et al. (2009))

(Okin et al. 2001, 2006), which can subsequently affect the biogeochemistry of areas where it is deposited (Okin et al. 2004). Surface crusting in drylands also plays a significant role in surface processes. Well developed crusts can withstand shear stress exerted by the wind, and therefore help to protect soils from wind erosion. However, when crusts are disturbed, they can produce up to 500 times more sediment (Field et al. 2010).

Factors such as changes in vegetation and crust disturbance, which can increase aeolian soil erosion, have broader scale consequences. Dust emitted from these regions may be deposited on the snowpack of nearby mountains, altering their albedo, causing an increase in rates of snow melt (Painter et al. 2010). An increase in snowmelt rates during seasons of especially high dust deposition, can have the effect of shortening the duration of snowpack by a period of several weeks, which thus alters the seasonal timing and magnitude of evapotranspiration from soils that are exposed for a longer period of time (Belnap et al. 2011), and reducing the amount of water entering streams and large rivers (Painter et al. 2010).

Degrading systems are those that are on a trajectory of moving towards a more degraded state, and are generally characterized by the predominance of amplifying feedbacks which decrease the resilience of the non-degraded state and move the system towards a degraded state (Fig. 3.8). Stabilising feedbacks characterize non-degrading ecosystem states, as they tend to increase system resilience. For instance, biotic regulation of vertical water exchange and the concentration of plant-essential resources within the vicinity of plants is an example of a stabilizing feedback that maintains system resilience. Amplifying feedbacks tend to be associated with the predominance of abiotic processes (Turnbull et al. 2012) which increase in strength in degrading systems, such as increases in runoff and fluvial and aeolian erosion, which can affect the structure of vegetation, leading to propagation of these feedbacks.

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Chapter 6
Integrating Short- and Long-Range Processes into Models: The Emergence of Pattern

Kelly K. Caylor, Greg S. Okin, Laura Turnbull, John Wainwright, Thorsten Wiegand, Trenton E. Franz, and Anthony J. Parsons

Abstract The production of pattern requires feedbacks operating on different spatial and/or temporal scales and thus the integration of short- and long-range processes. More flexible models – ones able to represent the dynamics of change over more than just spatiotemporal snapshots – must be able to reconfigure their state and process representations. Scale and process are critically linked when considering the

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K.K. Caylor
Department of Civil and Environmental Engineering, Princeton University, Princeton, NJ 08544, USA
e-mail: kcaylor@princeton.edu

G.S. Okin
Department of Geography, University of California, Los Angeles, CA 90095, USA
e-mail: okin@ucla.edu

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

T. Wiegand
Department of Ecological Modelling, Helmholtz Centre for Environmental Research UFZ, 04318 Leipzig, Germany
e-mail: thorsten.wiegand@ufz.de

T.E. Franz
Department of Hydrology and Water Resources, University of Arizona, Tucson, AZ 85721, USA
e-mail: tfranz@email.arizona.edu

A.J. Parsons (✉)
Sheffield Centre for International Drylands Research, University of Sheffield, Sheffield S10 2TN, UK
e-mail: a.j.parsons@sheffield.ac.uk

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state and function of dryland ecosystems: different processes dominate at different scales. There are four scales at which land degradation in drylands is typically considered: plant-interspace, patch, landscape and region. However, cross-scale process interactions are a critical element of modelling dryland degradation. There are intimate relationships between scale, process and pattern in drylands and many of these relationships involve cross-scale interactions. One possible consequence of these cross-scale interactions is the manifestation of bistability, which may also be present at different scales. A key issue for modelling is the mismatch between temporal scales of processes, which are typically short term, and degradation scales, which are typically long term, and the feedbacks that exist between these two sets of scales. To add to this complexity, ecological, biogeochemical and geomorphological fluxes redistribute energy and materials either vertically, or horizontally, or both. Interactions between these lateral and vertical fluxes are intrinsic to ecosystem dynamics and pattern formation in drylands. In all modelling, the important challenge remains of how to strike the balance between the technical details of a particular system and the strategic simplifications necessary to maintain generality, and to employ appropriate strategies that will permit generalization from specific case studies.

6.1 Emergent Patterns of Dynamic Desert Ecosystems

To model the emergence of dynamic desert ecosystems, and their corresponding patterns, one needs to start from the basis of understanding what sorts of models can exhibit emergence. First, model outputs can be directly imposed by the equations or rules of the model. For example, many models in related areas focus on the simulation of static system properties (e.g. vegetation biomass, total runoff or erosion from a catchment), and thus are incapable of representing emergent properties. While this approach does not necessarily make such models "wrong", as all models are designed for a specific purpose (Mulligan and Wainwright 2013; Grimm et al. 2006 and see below), it means that they are not fit for *this* purpose. Second, if the model is rich enough in structure model outputs will emerge as result of the dynamic interaction of different model rules, initial conditions and environmental constraints. From an ecohydrological perspective, Newman et al. (2006) emphasized the distinction between "Newtonian" and "Darwinian" perspectives on representing coupled vegetation-water(-sediment) systems. The former emphasize prediction from the physical basis of the system, while typically ignoring the evolution of the system, whereas the latter focus on reproducing the historical response of the system with less of an emphasis on prediction due to issues of contingency. Newman et al. suggest that a combination of these sets of approach will lead to real advances in our understanding. At the heart of these two different approaches are contrasting philosophies of simplification to enable the analysis of the real world to be tractable.

The "Newtonian" perspective has come over the last half century or so to rely on systems analysis to define the characteristics of the environment as a series of interlinked subsystems. Systems analysis has failed to live up to its early promise of providing “more than the sum of the parts”, not least because of the highly reductionist explanations that have tended to be produced by breaking down the system into easily investigated parts. While modelling should provide a means for reintegrating these smaller components – and has done so notably in some cases – there have been limitations on the sorts of reintegration of knowledge that can be attained. Specifically in the context of system evolution, the “top-down” approach of systems theory makes reorganization of system elements difficult to represent. This limitation constrains the behaviour of the simulated system so that only limited numbers of evolutionary pathways can be reproduced. While this approach can simulate the development of pattern, it has usually only been possible in the case of very simple systems, or analogues of them (see discussion in Chap. 3 and Borgogno et al. 2009). One model that has received a particular amount of attention in the investigation of pattern in drylands (e.g. HilleRisLambers et al. 2001) is that of Turing (1952), originally developed to evaluate optochemical reactions. However, in using Turing’s approach as an analogue of vegetation patterns in drylands, there is a fundamental problem. Turing himself noted that the model system cannot represent hydrodynamics because it introduces physically meaningless results. Applications of alternative models (e.g. Klausmeier 1999) have produced patterns, but have done so using unrealistic parameter combinations (see discussion in Stewart et al., in press), and have thus fallen into the trap of producing the right model results for the wrong reasons.

The “Darwinian” perspective appears – at first paradoxically – to rely on complexity theory as a means of simplification. Complexity theory recognizes that complex outcomes in reality need not necessarily be the result of complicated models (see Chap. 3). Such outcomes may be the consequence of many interactions of (relatively) simple behaviour. Striking examples of such models are agent-based models (or individual-based as they are called in ecology) in which complex patterns and system behaviour can emerge from simple rules on the behaviour of the agents (Grimm and Railsback 2005; Railsback and Grimm 2012). Complexity-based approaches in this sense may be considered more “realistic” in many respects because they recognize the need to represent large-scale system development as the result of many local interactions, across multiple scales. For example, how does an individual plant or soil particle know about what else is happening in the simulation domain? Although there are perhaps processes that allow plants to obtain indirect representations of larger scale processes, e.g. through the presence of microrhizal mats (Barrow et al. 2008) or local scale genetic variability (Duran et al. 2005), these are not the processes typically represented in the systems-based approach. Solution of a series of coupled simultaneous equations of vegetation growth and water flow contains an implicit assumption that the component plants and water have information about the other plants and water in the simulation domain; clearly they do not. Cellular automata models have long been used to show the development of pattern from repeated, local rules, but more realistic models using individual-based approaches were limited for a long time by computational resources and concepts. The parallel development of computing resource (both processing speed and memory availability) and a change from the dominant procedural languages (e.g. Fortran, Basic) to object-orientated languages (e.g. C++, Java) provided both the means and techniques for developing individual-based approaches. However, as a note of caution, one might ask the extent to which the individual-based approach is a reflection of the change in programming approach rather than understanding of the environment. In the development of agent-based approaches to social simulation, O'Sullivan and Haklay (2000) have pointed out the limitations of assuming everything can be represented as a series of interacting agents representing human actors (see further discussion in O'Sullivan et al. 2012). Similar arguments may be made about patterns in drylands (see the following sections). A linked charge to which the individual-based approaches may be guilty is that of *oversimplification* of the system. Again, there needs to be a link back to process understanding to investigate this problem fully; a model is a tool for aiding the understanding of the real world rather than being an explanation in itself. One approach to overcome this problem especially in individual-based models is "pattern-oriented modelling" (Grimm et al. 2005, see also Chap. 9) which heavily relies on emergent patterns for model construction, parameterization and evaluation. The model construction is guided by patterns observed in a real system (or a generic class of real systems) in a way that the model should be in principle able to produce the observed pattern. Such a construction means that the model should incorporate state variables and equations (or rules) required to produce an output that can be directly compared with the observed pattern, making the model rich enough in structure and process to be testable. Model parameterization and model selection are then repeated based on emergent patterns by qualitatively or quantitatively comparing the observed patterns with that produced by the model. Quantitative comparisons may rely on inverse modelling, or techniques of Approximate Bayesian Computation (Bastos and O'Hagan 2009). Further consideration of the idea of emergence in modelling is considered in Chap. 3, along with the related idea of self-organization.

An area that has been receiving increasing attention in modelling circles is in producing model ontologies. A model ontology is an attempt to describe the simulated world one is aiming to represent and the knowledge that has gone into its construction (e.g. Parker et al. 2008). One clear advantage to a very explicit approach to what the model shows, both implicitly and explicitly, is that it avoids accidentally building in the answer one is trying to test. If emergence is an important characteristic of the model behaviour, then building in predictable patterns is by definition not emergence. This is a further danger of an "off-the-shelf" model use, for example, the selection of a model one knows will show patterns. The resulting patterns are not a test of a mechanism for understanding the environment, but a simplistic verification of model behaviour. However, all models require simplification and definition of a minimal level of resolution. That means that certain aspects of the model will necessarily be imposed to keep the model tractable. One of the challenges is therefore to be clear which model behaviour is intentionally imposed and which behaviour is emerging. Grimm et al. (2006, 2010) have further emphasized the need for a standardized framework for representing model structure and components, especially ones that use an individual- or agent-based approach.

Their ODD (Overview, Design, Details) framework for model description has become widely used, and can also be used on other modelling approaches (see Mulligan and Wainwright 2013).

The simulation of patterns of dryland degradation thus needs to take into account these broad conceptual considerations as well as the lessons learned from previous chapters. A model of land degradation must consider exactly what degradation actually *is* (see discussion in Chap. 2) and how it may be represented by the state variables of a model. Is vegetation pattern a straightforward indicator of degradation, or do we need to build more integrated assessments that allow mechanistic interpretations of vegetation change by considering changes and patterns in soil characteristics and water availability? For example, if one were to try to investigate the "islands of fertility" concept of land degradation (Charley and West 1975; Schlesinger et al. 1990, see also Sect. 4.3), then vegetation pattern is not a sufficient explanation; other processes and process feedbacks also need to occur (but see Stewart et al., in press, for a consideration of whether islands of fertility are of themselves an independent phenomenon for evaluating change). A key issue in environmental modelling in general, but specifically in the case of patterning in drylands (and other environments: see Belyea and Baird 2006; Diggle 1981) is that of equifinality. Multiple model structures and multiple parameterizations of those structures may reproduce patterns (or even patterns that we may be tempted to perceive that are not really there: see Chap. 3). The problem of equifinality can be reduced by attempting to reproduce multiple patterns that operate at different spatial and/or temporal scales (Grimm et al. 2005). In this way one ascertains that the model produces a behaviour that agrees with observation at multiple scales. However, a model producing pattern – even one that is statistically indistinguishable from a target pattern in the environment – can never be the final endpoint of an analysis, but similarly, neither can a conceptual explanation (which is simply another form of model).

The production of pattern requires feedbacks operating on different spatial and/or temporal scales and thus the integration of short- and long-range processes. More flexible models – ones able to represent the dynamics of change over more than just spatiotemporal snapshots – must be able to reconfigure their state and process representations. Further details of how these aims might be achieved are discussed in the following sections.

6.2 Conceptual Frameworks for Integrating Short- and Long-Range Processes into Models: Addressing Spatial Scales

Scale and process are critically linked when considering the state and function of dryland ecosystems: different processes dominate at different scales. This is not to say, however, that cross-scale process interactions are inconsequential. Indeed, the interactions between processes that work at different scales is a fundamental source of the complexity of the Earth's ecosystems and poses tremendous challenges to modelling.

This section will discuss the spatial scales at which dryland degradation is typically considered (see also Sect. 4.1) and will investigate the interaction among processes occurring at these different scales with a particular emphasis on how these interactions create patterns.

Before delving into the delineation of spatial scales, it is useful to differentiate these scales from the distance over which a particular effect may occur. For instance, facilitation and competition, as will be discussed later, appear to be critical for the formation of patches in savannas and other dryland ecosystems. In models of pattern emergence in drylands that invoke facilitation and competition, competition is typically modelled as occurring at a larger distance from an individual plant than facilitation. Thus, competition is a long-range process and facilitation a short-range process. Nonetheless, both facilitation and competition, in this case, are occurring at the same scale – the plant-interspace scale.

6.2.1 Typical Spatial Scales at Which Land Degradation in Drylands Is Considered

There are four scales at which land degradation in drylands is typically considered: plant-interspace, patch, landscape and region (Peters et al. 2006). The plant-interspace scale is the finest scale that is generally useful for understanding dryland degradation. Processes at this scale are those that involve only individual plants and their adjacent bare interspaces. An excellent example of a process that occurs almost exclusively at the plant-interspace scale is differential rainsplash to create individual mounds (e.g., Parsons et al. 1994; Bochet et al. 2000). Patches consist of multiple plant-interspaces, typically on the same soil or geomorphic unit with patches differentiated from one another by differences in plant species, densities, or sizes or soil/geomorphic properties. The patch scale is most closely aligned with the ecological 'plot' scale, the scale at which most ecological experiments are conducted. An excellent example of a process occurring at the patch scale is seedling herbivory by small rodents (Bestelmeyer et al. 2007). These animals often do not travel much beyond a single patch to procure food, resulting in significant herbivory within some patches and therefore considerable effects on vegetation growth/regeneration in the targeted patches. Aeolian saltation in vegetated areas may also be considered a patch-scale process, where a series of connected plant interspaces is necessary to allow sustained saltation of particles that generally do not move farther than the scale of the patch (Gillette et al. 2006). A landscape is made of multiple patches and may contain different soils and geomorphic units.

For the purposes here, the landscape scale is probably the least well defined, though commonly bears the connotation of internal connectedness: Ludwig et al.

(2002, 2007) consider a *landscape's* hydrological 'leakiness' (see Sect. 9.4) when considering dryland degradation and due to the implied hydrological connectivity, a watershed is typically considered a *landscape*. In this context, fire is an excellent example of a landscape-scale process. Most fires burn multiple patches and the necessity of spatial continuity for ignition makes fire a process that can significantly impact dryland degradation at the landscape scale (e.g. D'Antonio 2000).

The regional scale is coarsest scale at which land degradation in terrestrial drylands is typically considered. A region generally consists of one or more contiguous biomes, and for modelling purposes may often include both dryland and non-dryland areas (including marine areas). There are few clear examples of truly regional-scale processes that are related to dryland degradation and regions might typically be considered spatial fields in which processes at finer scales occur and interact. Nonetheless, many atmospheric processes occur at the regional scale and there are cases in which coupling of atmospheric processes with arid and semi-arid regions can influence dryland degradation. An example of this coupling will be discussed below.

When considering scale in models, it is important to remember that scale may be both explicit and/or implicit in any model. Spatially explicit models, typically grid-based models where the grids are given a specific spatial extent, differ from spatially inexplicit models that may be grid-based (i.e. two dimensional) or point-based (i.e. zero dimensional). Spatially inexplicit grid-based models are not, despite their lack of specificity, generally scale-invariant. For instance, in their model of pattern formation in drylands, D'Odorico et al. (2006b) do not specify the size of their grid cells. Nonetheless, the model invokes processes of competition and facilitation that have some "natural" distance over which they occur and therefore the model has an implicit scale. By construction, point-based models cannot create spatial patterns but they can be used effectively to elucidate the processes that lead to pattern formation. D'Odorico et al. (2006a), for example, use a probabilistic point-based model to explain the coexistence of tree and grass in savannas.

Of course a model that is spatially explicit with regard to one process can be spatially inexplicit with respect to another process. This is frequently the case in regional-scale models that are spatially explicit yet use equations within grid cells that encode processes at finer scales without being explicit about the scale at which the encoded process occurs.

6.2.2 Interacting Processes at Different Spatial Scales

Cross-scale process interactions are a critical element of modelling dryland degradation. Changing regional climate, for instance, can have important impacts on processes at finer scales. This issue is generally appreciated for vegetation modelling – vegetation responds to temperature and precipitation, which are driven by the regional scale – but it is also crucial in abiotic processes. Increasing gustiness, for instance, may enhance patch-scale saltation or increased storm intensity may increase overland flow and hillslope erosion at the patch- to landscape scales while also increasing differential rainsplash at the plant-interspace scale.

An important example of such cross-scale effects is grazing of vegetation by domestic livestock. The direct interactions, for example defoliation and trampling, occur on the plant-interspace scale but their consequence, overgrazing, is only visible at the patch or the landscape scale (Paruelo et al. 2008). An early example of a hypothesized cross-scale interaction in this direction is the Charney Hypothesis (Charney 1975; Charney et al. 1975), which suggests that increased dryland albedo due to reduced vegetation associated with desertification can alter regional precipitation. Although the detailed mechanisms hypothesized by Charney have not held up to scrutiny (Taylor et al. 2002), several modelling studies have shown that land-use change associated with desertification can impact upon regional climate (e.g. Xue and Shukla 1993; Clark et al. 2001).

6.2.3 Scale, Process and Pattern in Drylands

Several authors have shown intimate relationships between scale, process and pattern in drylands and many of these relationships involve cross-scale interactions. For instance, processes at the plant-interspace to patch scale have been hypothesized to give rise to landscape-scale patterns. These patterns can be both statistical and spatial.

An important statistical pattern often observed at the landscape scale in drylands is power-law clustering of vegetation. For instance, Scanlon et al. (2007) have suggested that facilitation at the plant-interspace scale leads to power-law size distribution of vegetation clusters in Kalahari savanna landscapes. This result has been reproduced using other models, including Manor and Schnerb (2008) as well as von Hardenberg et al. (2010). Moreno-de las Heras et al. (2011) have observed that this power-law clustering holds even for banded vegetation, because the bands themselves are divided into clumps. Kefi et al. (2007) have shown that this power-law clustering breaks down with the imposition of a distinctly patch-to-landscape scale process, such as grazing, that counteracts the positive feedbacks that cause the clustering.

Spatial patterns at the landscape scale, typically periodic distributions of vegetation that can be anisotropic (e.g. banded vegetation) or isotropic (e.g. regularly spaced plants/patches), appear to arise from tension between competition and facilitation at the patch-interspace scale (an excellent review of ecohydrological models that give rise to vegetation patterning is that by Borgogno et al. 2009). Banded vegetation, for instance, arises on hillslopes where water flow is disrupted by bands of vegetation resulting in facilitation due to increased water availability in these areas, whereas water availability is lower in inter-band areas thus inhibiting vegetation growth (e.g. Klausmeier 1999; Saco et al. 2007; Dunkerley 1999;

Tongway and Ludwig 2001). While not strictly competition, the water scarcity in the inter-band areas functions as such. When observed in this light, this phenomenon can be related to other studies in which long-range competition and short-range facilitation result in regular patterns. In a modelling study, D'Odorico et al. (2006b) have shown that short-range facilitation and long-range competition, which presumably arises from the role of water as a fundamental limitation on productivity in drylands, can give rise to patterned vegetation. Similarly, by addition of competition to a vegetation model for arid regions, Manor and Shnerb (2008) were able to go from a statistical power-law distribution (facilitation-dominated) to a spatial pattern (competition-dominated), indicating the importance of both facilitation and competition in the emergence of patterns in drylands. Barbier et al. (2008) similarly induced short-range facilitation, due to above-ground interactions, and longer-range competition, due to below-ground competition for resources, to explain the emergence of patchiness in dryland vegetation (see Sect. 4.4 and Chap. 13).

6.2.4 Hysteresis: The Emergence of Bistability in Dryland Vegetation

The emergence of bistability has been a major thrust in understanding dryland ecosystems in the past decades. Bistability in dryland vegetation modelling was first found by Noy-Meir (1975) in a model of plant–herbivore interactions based on predator–prey models. Vegetation dynamics was modelled under grazing with differential equations representing grazing as a homogeneous process in space and continuous in time. The model produced the disturbing result that a continuously grazed ecosystem may show bistability in which a small increase in grazing pressure, or a minor disturbance, may move the system from a stable state of high productivity to a stable state of low productivity. Recovery to the high productivity state is possible only after a substantial reduction of the harvest rate (i.e. hysteresis).

Bistability has been seen to emerge in drylands at ranges from the plant-interspace to the regional. At an implicit plant-interspace to patch scale, Okin et al. (2009) showed that in shrub-grass systems two stable states could exist, even when grasses were in competitive advantage with respect to shrubs, under conditions in which grass reduction leads to soil erosion. At the patch to landscape scale the feedback between fire and vegetation, in which fire favours grassland vegetation because it kill woody seedlings, has been invoked to explain shrub-grass bistability in the Chihuahuan desert (Van Auken 2000) as well as tree-grass bistability in savannas (D'Odorico et al. 2006a, 2007), among others.

At larger scales, vegetation-climate feedbacks are thought to induce bistability in some regions. The Charney Hypothesis (Charney 1975; Charney et al. 1975) and its more recent incarnations (e.g. Xue and Shukla 1993; Taylor et al. 2002) is one example of this interpretation. In addition, deMenocal et al. (2000) have hypothesized that widespread and abrupt aridification inferred from the geological record in North Africa 5000–6000 years ago arose from regional-scale vegetation-climate feedbacks functioning within a context of slowly changing solar insolation. Janssen et al. (2008) have suggested that fine-scale processes that give rise to bistability at the plant-interspace to patch scales can also increase the strength of the regional-scale climate-vegetation feedbacks, indicating not only cross-scale linkages in space, but cross-process linkages, as well.

6.3 Conceptual Frameworks for Integrating Short- and Long-Range Processes into Models: Addressing Temporal Scales

As with spatial scales, dryland ecosystem dynamics are structured by a hierarchy of temporal scales; from the momentary timescales of turbulence variations in aeolian sediment transport to decadal-scale variability in vegetation structure as successional dynamics progress in response to disturbance from herbivores and fire. However, the key issue for modelling is the mismatch between process scales, which are typically short term, and degradation scales, which are typically long term, and the feedbacks that exist between these two scales. A central challenge to understanding trajectories of land degradation and recovery is the fact that these two time scales are intimately linked by the patterns of vegetation within a dryland landscape because vegetation structure has a substantial impact on both surface dynamics and, sometimes, sub-surface processes. In the following, we demonstrate four examples of how these short- and long-range processes can be linked. In the first case, we show the critical role that hillslope-vegetation patches play in governing patterns of infiltration, and how dependent vegetation effects are on storm climatology. In the second example, we demonstrate the subtle manner by which shifts in both average rainfall - and critically, the variability of rainfall – lead to large changes in the relative abundance of woody species with divergent water use strategies. In the third case, we show that emergent hillslope vegetation patterns (i.e. the degree of patch organization within a hillslope) is strongly dependent on the degree to which vegetation patches lead to convergence of flow within inter-patch areas. In the final example, we consider the effects of aeolian erosion on the distribution of nutrients in dryland soils.

6.3.1 Short- and Long-Range Spatio-Temporal Patterns of Soil Moisture in Drylands

In the absence of topography, soil moisture is primarily controlled by the local vertical flux of water, and subsequent patterns of local water use by vegetation. Conversely, more complex terrains can lead to significant horizontal fluxes of water in both the surface and subsurface. Most notably, the rainfall-runoff-runon paradigm associated with systems where Hortonian runoff and overland flow occur may lead to significant horizontal surface fluxes of water during individual rainfall events. Moreover, locations of high surface-flow convergence (areas with large contributing areas) may receive substantially larger inputs of water into the subsurface compared to the surrounding areas. Driven by large pressure gradients, these points of local high vertical infiltration surrounded by drier soils will result in redistribution of subsurface water on a much longer timescale than surface redistribution (Franz et al. 2011).

The spatial arrangement and structure of dryland vegetation play an integral part in the redistribution of surface and subsurface water and the coupled sediment transport processes. An intriguing example of this behaviour has been observed in central Kenya. Here, a native succulent (*Sansevieria volkensii*) has recently and rapidly expanded vast areas of the landscape over the last several decades due to the elimination of perennial grasses in response to overgrazing (King et al. 2010). The colonization of the landscape has occurred in distinct expanding oval shaped patches until completely dominating entire hillslopes. The dense spear-like leaves of *S. volkensii* provide a much more suitable area for infiltration by both breaking up the soil crust and reducing the flow velocity of overland flow. By using geophysical observations (electromagnetic induction) before and after two different sized rain events (Robinson et al. 2009), the spatiotemporal patterns of infiltration and subsurface redistribution of water were characterized at this patch-interpatch spatial scale in the top 0.5 m of soil (Franz et al. 2011). The event-scale differences in electrical conductivity (ECa) will primarily be due to changes in soil moisture. The summary of results is presented in Fig. 6.1, where a 15 mm event occurred on 6 April 2009 and a 6 mm event occurred on 25 July 2009 (average size rain event is 9 mm from 40 year historical record). The 6 mm event resulted in slightly higher changes in ECa in runon vs. runoff zones, while the 15 mm event resulted in much larger differences in ECa in runon zones vs. runoff zones and inside the patch vs. outside the patch. While additional research is needed for fully validating data collected with geophysical measurements (Robinson et al. 2008), the qualitative patterns are unique datasets that will help illuminate the underlying processes governing dryland ecosystems.

6.3.2 Effects of Changes in Average Rainfall and Rainfall Variability on Dryland Vegetation Patterns

The rapid response of dryland vegetation following individual pulses of precipitation requires high temporal model resolution of rainfall. Neglecting the diurnal cycle of inputs and water use, previous work (Rodriguez-Iturbe and Porporato 2004) has utilized the marked Poisson point-process model to account for the random fluctuations of daily rainfall patterns in dryland ecosystems. This simplistic model

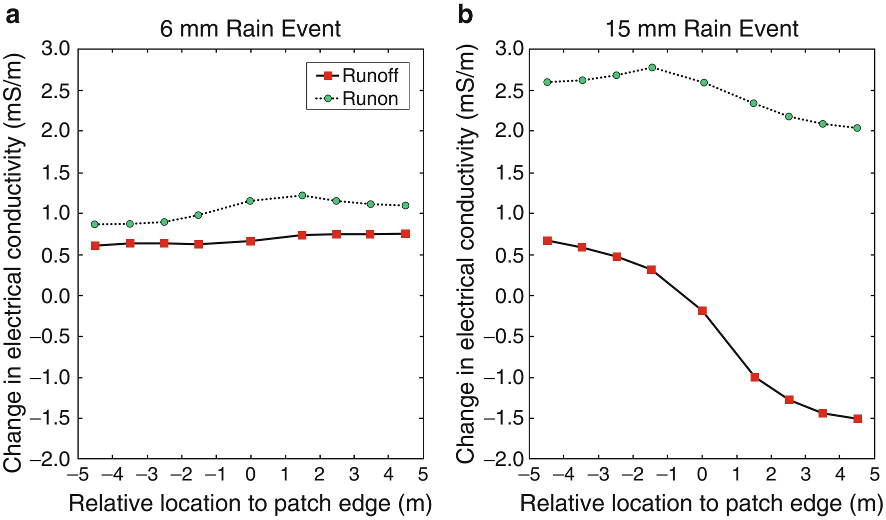

Fig. 6.1 Spatiotemporal changes of electrical conductivity around *Sansevieria volkensii* patches in central Kenya following (**a**) a 6-mm event and (**b**) a 15-mm event. Larger ECa anomalies associated with greater infiltration occurred in surface runon zones vs. runoff zones, as well as inside the patch boundary vs. outside (Adapted from Franz et al. 2011)

assumes second order stationarity, preserving the mean and variance of seasonal rainfall patterns, and has been successfully filtered through a point water-balance model to yield analytical solutions to the steady state soil moisture probability density function (Laio et al. 2001). The simplistic but powerful model is controlled by two input parameters, the mean arrival rate of storms, λ (day^{-1}), mean storm depth, α (mm), and a few soil and plant parameters controlling the vertical fluxes of water in the effective rooting zone.

Building on this existing model and utilizing the optimality trade-off hypothesis of dryland vegetation patterns whereby plants simultaneously maximize seasonal water use while minimizing seasonal plant water stress (Caylor et al. 2009), the framework was expanded into a spatial context for studying the regional distribution of woody species (Franz et al. 2010), and the spatial distribution of vegetation patterns on hillslopes (Franz et al. 2012). The studies found the shifts in regional patterns of woody vegetation species are equally sensitive to changes in both the mean and variance (holding one constant) of seasonal rainfall patterns. The regional study found the territory of the most drought-tolerant species, *Acacia tortilis*, expanded with decreases in the mean and variance of seasonal rainfall (Table 6.1), while the territory of the most intense water user, *A. xanthopholea*, decreased. In contrast, increases in the mean and variance of seasonal rainfall led to substantial territory expansion of the species with the most intensive water use.

Table 6.1 Predicted changes in regional abundance of three common woody vegetation species in central Kenya (~10,000 km^2 area)

	Rainfall		
	Mean 473 mm	Mean 393 mm	Mean 473 mm
	SD 97 mm	SD 97 mm	SD 116 mm
Acacia drepanolibium coverage	76 %	7 %	17 %
Acacia tortilis coverage	78 %	10 %	12 %
Acacia xanthopholes coverage	50 %	0 %	50 %

Adapted from Franz et al. (2010)

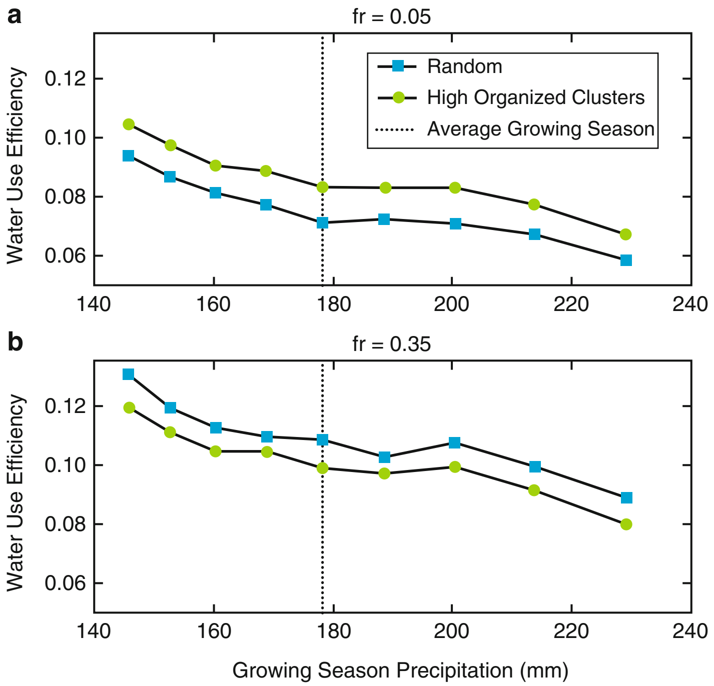

Fig. 6.2 Modelled changes of hillslope woody vegetation organization patterns assuming 20 % cover, but under changes in patch/inter-patch dynamics. Panel (**a**) illustrates the shift in most efficient vegetation clustering strategies by changing the surface-flow-convergence parameter, *fr*, associated with vegetation across a range of climates. Panel (**b**) depicts the change in organization which occurs when surface-flow convergence is reduced to equal surface-water partitioning between vegetated and bare areas

6.3.3 Effects of Flow Convergence at the Hillsope Scale

When moving to the hillslope scale, the local topography becomes a dominating influence controlling the redistribution of water as was discussed in the previous section. Here, the length scales of facilitation and competition of water control vegetation patterns (Borgogno et al. 2009). These length scales are manifested in the model parameters that control the convergence of surface-water flowpaths, *fr*. By varying this parameter, the most efficient hillslope vegetation patterns (evaluated with the optimality trade-off hypothesis averaged over multiple growing seasons) changed from a random to highly organized pattern (Fig. 6.2). The effect of changing daily rainfall properties narrowed and expanded the relative efficiency of the different vegetation patterns but did not result in a pattern shift. The changes in vegetation patterns will be moderated through the coupled feedbacks between rainfall patterns and the parameters that control the length scales of facilitation and competition. For example, increases in rainfall intensity or overgrazing will shorten the length scale of surface-flow convergence (i.e. a lower *fr* value) as a result of channelization in connected bare patches. Those changes are the likely explanation of the proliferation of *S. volkensii* in central Kenya, where vegetation pattern responded and capitalized on new system dynamics.

6.3.4 Feedbacks Between Wind Erosion and Land Degradation

Concentration of resources into islands of fertility has been identified as a characteristic of land degradation (see Chaps. 2 and 4). The timescales of the processes of concentration of resources need to be integrated with the timescales of the formation of islands of fertility if this form of land degradation is to be successfully modelled. Because wind erosion is size selective, it has the capacity to cause spatial redistribution of nutrients because the adsorption of nutrients onto soil particles is also size selective. In a study undertaken within the Jornada Experimental Range, southern New Mexico, Li et al. (2007) reported between 2.7 and 4.0 times enrichment of total nitrogen in windblown sediment compared to topsoil and between 3.2 and 6.0 times enrichment of total organic carbon. This study was undertaken on controlled plots in which grass cover was selectively removed. The study showed a dramatic increase in wind erosion at between 75 and 100 % grass cover, and that the balance between net loss and net gain in soil nutrients (due to the combined effects of loss due to wind erosion and gain due to biotic processes) shifted from net loss to net gain at between 50 and 25 % grass reduction. Based on observations over three windy seasons, the experimental study concluded that on bare areas the total organic carbon and total nitrogen of surface soils would be removed by wind erosion in about 10 years. Because the study was conducted on plots where the grass cover was artificially maintained at preselected levels, the important feedbacks between nutrient status and plant growth could not be studied. But if the significance for land degradation of nutrient loss by wind erosion is to be understood, then this feedback is crucial. Furthermore, so is the timescale of the interaction. In this study, nutrient loss was studied on an annual (windy season) timescale. Is this process timescale sufficiently short to capture feedbacks to vegetation growth and land degradation? The answer to this question may well lie in the extent to which the windy season coincides with the annual plant-growth cycle. The more that plant growth occurs outside of the windy season the answer is likely to be that it is short enough, but it is less certain to be so if the two periods are coincident.

6.4 Coupling Lateral and Vertical Fluxes into Models of Patterns in Drylands

The emergence of patterns in drylands develops from a complex interplay of processes and resulting fluxes of energy and materials over multiple spatial and temporal scales. Ecological, biogeochemical and geomorphological fluxes redistribute energy and materials either vertically, or horizontally, or both. These vertical and horizontal fluxes have, for the most part, been considered in isolation from each other, in both monitoring- and modelling-based studies. However, interactions between these lateral and vertical fluxes are intrinsic to ecosystem dynamics and pattern formation in drylands (Fig. 6.3).

The predominance of lateral and vertical fluxes of energy and materials is system specific and the critical scales over which key processes operate vary, depending on factors such as vegetation type and cover which influences processes such as photosynthesis and transpiration, soil characteristics which affect factors such as infiltration and nutrient dynamics, spatial configuration of vegetation and soil properties which affect processes such as erosion by wind and water. To model ecogeomorphic patterns and processes, accounting for the lateral and vertical fluxes of energy and materials, the structure of modelling approaches needs to be flexible enough to simulate multi-scale processes.

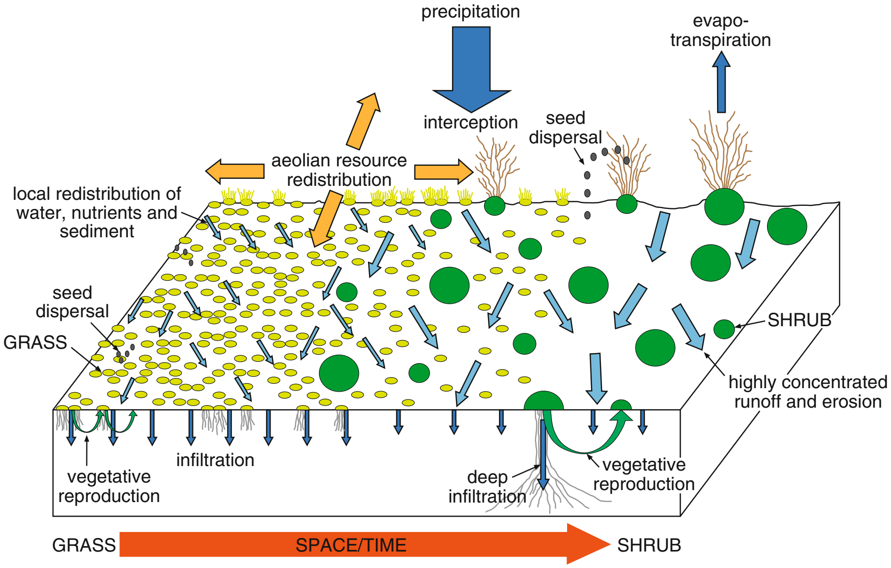

Fig. 6.3 Lateral and vertical fluxes of energy and matter that give rise to pattern formation grassland, shrubland and grass/shrub transitional stages in drylands

Since processes that govern lateral and vertical fluxes of energy and materials are not mutually exclusive, coupling these processes in models requires that spatially explicit modelling approaches be used, whereby the spatial variability of these lateral and vertical processes that facilitates the emergence of pattern is inherent within the model domain. Models that have been developed to simulate isolated components of the ecogeomorphic system (such as runoff, biogeochemistry or plant dynamics) have adopted different approaches in terms of the different space and time domains over which they operate (see Chap. 7 for a more detailed discussion). These differences have arisen because of the different spatial and temporal resolutions that are required to provide an adequate representation of the key processes. For example, in the case of water fluxes, lateral fluxes of water tend to occur over very short timescales; sometimes runoff events might last only a few minutes (Parsons and Stone 2006; Turnbull et al. 2010a; Wainwright and Parsons 2002) and the extent to which these fluxes are spatially variable depend on the soil properties and their spatial distribution (Mueller et al. 2008; Turnbull et al. 2010b). Spatially explicit models to simulate the lateral fluxes of water over hillslopes in drylands are typically event based, such as the Model for Assessing Hillslope to Landscape Runoff Erosion, Runoff and Nutrients (MAHLERAN; Wainwright et al. 2008), simulate key processes over fine spatial scales (down to 0.5 m – which is necessary for the controls of vegetation-associated microtopography on the connectivity of lateral fluxes to be accounted for), and at time increments of as short as 1 second, which is required for numerical stability of flow-routing algorithms. Spatially explicit wind-erosion models (see for example Okin 2008), can be used to simulate wind-erosion dynamics at multiple spatial scales, from an individual unvegetated gap to the landscape scale. In contrast with these models of water and wind erosion, biogeochemical cycling models such as DAYCENT (Parton et al. 1998) – which simulates carbon and nitrogen dynamics through the soil-plant-atmosphere interface, net primary production (NPP) and allocation, mortality of biomass and soil organic matter turnover – simulate vertical fluxes only and do not account for spatial interactions. Biogeochemical cycling models typically operate on daily or monthly timesteps, which are considered to be an adequate time resolution for the simulation of these processes (Parton et al. 1998). Plant-dynamics models such as ECOTONE simulate recruitment, growth and mortality of each plant on a small plot at an annual timestep (Peters 2002), but do not allow for the exchange of matter between units. Thus, a major challenge of coupling lateral and vertical ecological, biogeochemical and geomorphological fluxes in ecogeomorphic modelling approaches is overcoming their different spatial and temporal scales.

Lateral and vertical fluxes occur over multiple spatial and temporal scales, and can therefore be conceptualized as occurring at different hierarchical levels. Systems possessing hierarchical structure contain a hierarchy of smaller, lower level systems, that are, at the same time, part of an even larger higher level system (Allen and Starr 1982; De Boer 1992; O'Neill et al. 1986; Wu and David 2002). Each hierarchical level within a system is linked by the lateral and vertical exchange of energy and materials, and thus, landscapes can be construed as nested spatial hierarchies,

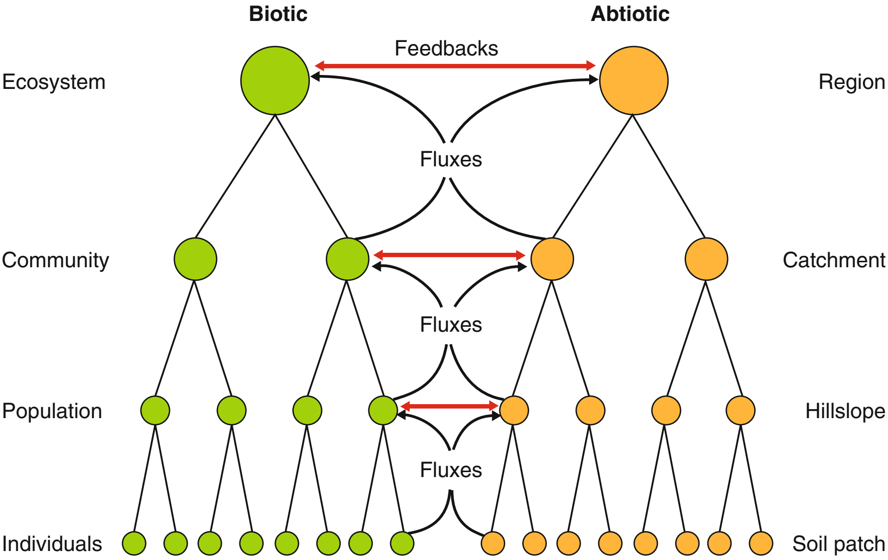

Fig. 6.4 Conceptualization of hierarchical levels that make up the ecogeomorphic system, comprising hierarchal units that are classified as either biotic or abiotic. Lateral and vertical fluxes of energy and material connect biotic and abiotic units at each hierarchical level as well as between hierarchical levels. *Arrows* indicate biotic-abiotic feedbacks that occur within each hierarchical level

in which hierarchical levels correspond with structural and functional units that control lateral and vertical fluxes at distinct spatial and temporal scales (Wu and David 2002). The hierarchical organization of a system depends on the nature of interactions between different components of a system. Ecogeomorphic systems can be considered as hierarchical levels consisting of abiotic and biotic entities that interact within those levels, and drive fluxes of materials between hierarchical levels (Fig. 6.4). Within the ecogeomorphic system, it is the network of connections via fluxes of energy and materials that give rise to higher level phenomena, which in turn modify distributions of biotic and abiotic entities at lower levels (Jenerette et al. 2011).

To model a system hierarchically, it is necessary to identifying characteristic scales of patterns and processes that control lateral and vertical fluxes. It is important to note that these characteristic scales may vary depending on the system in question. Within the hierarchical conceptualisation of an ecogeomorphic system, biotic and abiotic elements that make up a single hierarchical level will be amalgamated into a single unit at when observed at a higher hierarchical level (Poole 2002), and similarly, each element can be broken down into its component biotic and abiotic elements if observed at a lower hierarchical level or finer spatial scale.

Adopting a hierarchical modelling approach to simulate lateral and vertical exchanges of energy and materials enables key processes to be simulated at

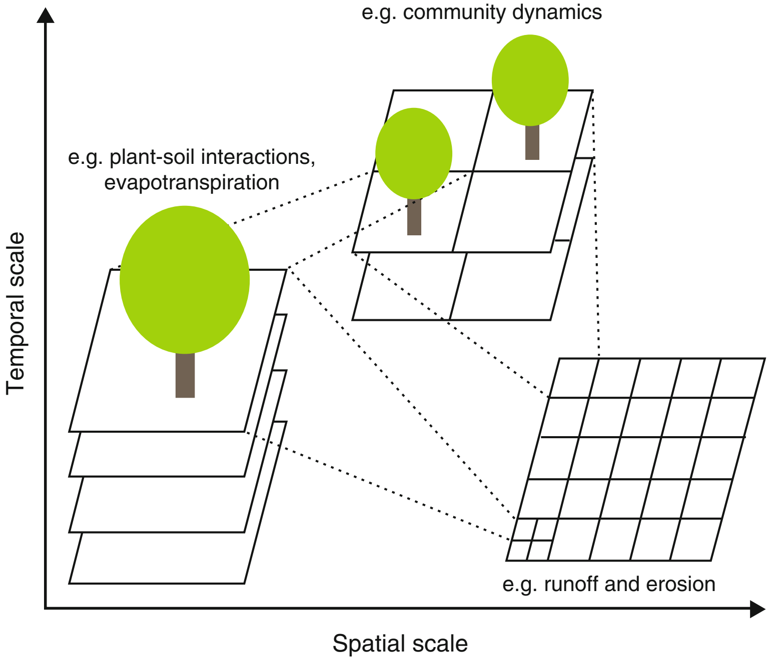

Fig. 6.5 Illustration of different space and time scales over which different ecogeomorphic processes may need to be simulated

their appropriate spatial and temporal scales. The following criteria are useful in identifying appropriate, system-specific hierarchical levels (Wu and David 2002):

1. Pattern and process are only interactive when both of them operate on the same or similar temporal scales;
2. If a spatial pattern does not change in response to a process, then only the effect of pattern on process needs to be considered;
3. When a spatial pattern changes much more rapidly than the process in question, only the spatially filtered average property is relevant to the pattern and process relationship; and
4. Unless there are interactions between pattern and process, the effects of either pattern or process can be aggregated.

A critical feature to note, however, is that in drylands, an increase in spatial hierarchy does not necessarily equate to an increase in temporal hierarchy (Fig. 6.5). It does not necessarily follow that a process that can be simulated at a large spatial resolution can also be simulated at a large temporal resolution. For instance, while the aggregate properties of vegetation may be input into a runoff model, a fine-scale temporal resolution is necessary (finer than the resolution at which vegetation dynamics need to be simulated), because of the nature of critical processes (e.g., runoff generation) that are highly dependent upon fine-scale variations in rainfall intensity.

6.5 The Problem of Generalizing from Specific Case Studies

Holling (1966) and May (1973) proposed a classification of models into strategic and tactical models. Strategic models consciously ignore many details of real systems in order to reach generality and attempt to capture the essential dynamics of a system that arises from general interactions or principles. Strategic models should provide general insights and understanding because many different systems may follow the same overall principles and have the same "essence". However, the strict simplifications made in strategic models limit their testability because they are often not tied to specific spatial and/or temporal scales. An example of a strategic model is the Noy-Meir model (see Sect. 6.2.4) where vegetation dynamics is modelled under grazing with differential equations representing grazing as a homogeneous process in space and continuous in time. Changes in plant biomass V are modelled as $\mathrm{d}V/\mathrm{d}t = G(V) - C(V)$ where $G(V) = rV(1 - V/K)$ uses the logistic equation to describe plant growth as a function of plant biomass V and a "Holling Type III" consumption function $C(V) = \beta H V^2/(V_0^2 + V^2)$ where H the stocking rate and the parameter V_0 describes animal intake capacity or digestion rate. The dependence of the equilibrium biomass on stocking rate can be calculated analytically and shows for some parameters a "folded" curve (Fig. 6.6). This model is very successful in revealing a possibly general phenomenon (bistability) based on a few general assumptions, but applications to specific systems are difficult. This problem arises partly because the parameters are difficult to measure, but more

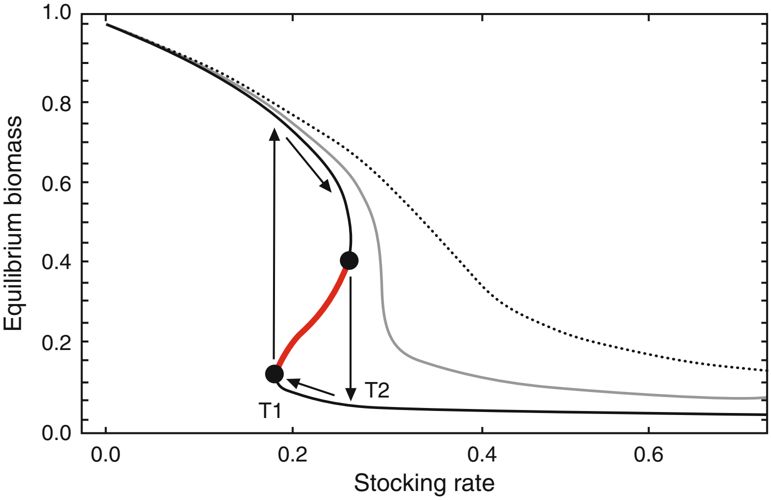

Fig. 6.6 The equilibrium biomass of the Noy-Meir model in dependence on stocking rate for different values of the parameter $\alpha = V_0/K$ (*solid* and *dashed lines*). For $\alpha = 0.1$ (*thick line*) the curve folds and an instable equilibrium and two stable equilibrium points appear. If biomass is at the higher equilibrium and stocking rate exceeds the critical value T2, the equilibrium becomes unstable and biomass collapses rapidly to the lower equilibrium

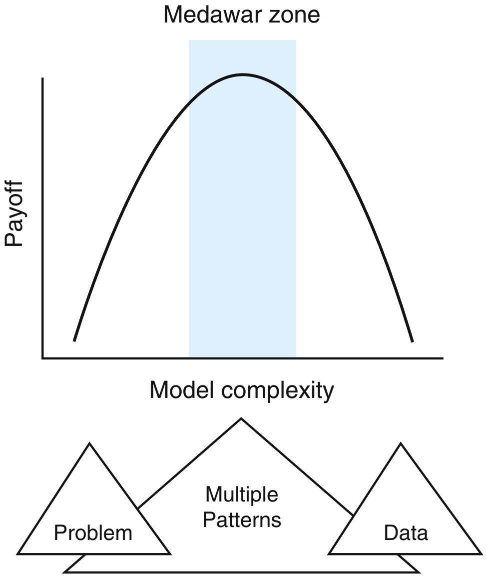

Fig. 6.7 The Medawar zone of optimal model complexity (After Grimm et al. 2005)

importantly because the model ignores a number of factors which may alter the model behaviour in specific situations. For example, grazing is not a continuous and homogeneous process, but involves discrete and stochastic defoliation events which are followed by a variable re-growth period. This series of events can lead to spatially heterogeneous grasslands.

Tactical models are developed to study the detailed dynamics of particular systems with the purpose of making specific predictions, often to address management problems. They may include many detailed mechanisms or statistical relationships and require a large number of parameters. The complexity and specificity of tactical models make them difficult to analyze and can limit understanding and their ability to predict in new areas. While increasing computational power and the advent of new modelling techniques such as cellular automata or individual-based models have blurred the contrast between strategic and tactical models, the important challenge remains of how to strike the balance between the tactical details of a particular system and the strategic simplifications necessary to maintain generality. On the one hand, if the model is too simple it will be unlikely to produce understanding and novel and general results. On the other hand, if the model is too complex problems with parameterization and structural uncertainty may erode its predictive ability and understanding may not be reached because of problems with conducting comprehensive model analysis. Thus, we need to find a “Medawar zone” (Grimm et al. 2005) where the payoff of the model in terms of understanding and prediction is maximized as function of model complexity (Fig. 6.7). Given these general considerations, several strategies can be identified that allow generalising from specific case studies. The first strategy is based on a single model and reaches generalization by simplifying and generalizing a model initially developed for a specific site. In the second strategy several models from different sites are used to reach generality. Finally, the same model may be applied to several specific sites. Emergent patterns play a key role in all strategies of generalization.

6.5.1 Generalizing a Single Model for a Specific Site

The first strategy of generalization is based on one model developed for a specific case study. If the model was developed with principles of pattern-orientated modelling in mind (Sect. 9.5), it will be rich enough in structure and processes that several emerged patterns of the model can be compared to the corresponding observed patterns. This approach to model development is important because the payoff of the model in Fig. 6.7 is also determined by its structural realism which guarantees its ability to produce independent predictions that match observations. If model development was guided by multiple patterns observed at different scales and hierarchical levels, the model is likely to end up in the Medawar zone (Grimm et al. 2005). The advantage of being developed for a specific case study is that the model is explicitly tied to the spatial and temporal scales thought to be important for the question asked. When designed to reproduce multiple patterns, the model can gain structural realism because model components (e.g. individuals) correspond directly to observed objects and variables, and processes correspond to the internal organization of the real system (Grimm et al. 2005). Clearly, a model of this type may have disadvantages when it comes to generalization. First, although designed to reproduce observed patterns its complexity may make is difficult to pinpoint exactly what is driving the pattern formation, even after extensive analysis. Second, the model may contain site-specific details which were important for the concrete application at a given site, but not relevant for generalization. Finally, when designed for a specific site many rules may impose behaviour observed at the given site, but in a more general setting this imposed behaviour may not hold.

Generalization can be achieved by (i) strict simplification of the model in which site-specific detail is consequently discarded, and by (ii) replacing site-specific imposed behaviour with simple general (mechanistic) model rules. Because the original model was designed to reproduce observed patterns (e.g. the typical spatial pattern of savanna trees; Jeltsch et al. 1999), model simplification must be done in a way that the simplified model is still able to produce emerging patterns thought to be important in the generalized setting. This approach is a powerful tool of model simplification because it allows throwing away detail not required for the emergence of the key patterns, but conserves the “essence” of the model and the way in which the real system operates to produce the patterns.

An excellent example of this approach is the general savanna model by Calabrese et al. (2010) which is a strict simplification of a detailed rule-based savanna simulation model by Jeltsch et al. (1999). The underlying general question was the so-called savanna problem: what is unique about savannas that allows the continual coexistence of trees and grasses? The Calabrese model is a minimalistic and analytically tractable stochastic cellular automaton aimed to study the interaction of tree-tree competition and fire in structuring semi-arid to mesic savannas. The two major patterns used to guide both the detailed Jeltsch model and the simple Calabrese model were the tree-grass coexistence and the spatial pattern of savanna trees. Both models have the same spatial resolution; one savanna tree is approximated by an area of 5 × 5 m. This common resolution forces all model rules to the observed spatial structures and makes the models structurally realistic. The model showed that short-distance competitive effects of adult trees on juveniles, either by themselves or in combination with fire, can have powerful consequences for both the tree-grass balance in savannas and the formation of the spatial pattern of trees.

6.5.2 Generalizing from Several Models for a Specific Sites

A second strategy to generalize from specific case studies is to compare the outcome of several models designed for specific sites. However, it is naïve to assume that any collection of specific case studies will automatically produce general insight. Of course, the models used in the different case studies should be designed to address questions and phenomena that have a certain overlap. Ideally, however, the different models would address the emergence of the same general patterns (e.g. tree-grass coexistence and the spatial pattern of savanna trees in the example above) from the more idiosyncratic perspective of the specific case study. The mechanism of pattern emergence revealed by the different models will therefore be a mixture of site specific detail and more general principles which are difficult to disentangle in a singe site-specific study. Comparison of the mechanism identified by the specific studies, however, may reveal mechanisms common to all studies as well as shifts in the relative importance of the different mechanisms with environmental conditions.

6.5.3 Generalizing from Applying One Model to Different Sites

Another common strategy to generalize from specific case studies is to design one more general models with the aim to describe the dynamics at several sites. A very successful type of model based on this idea is the gap-model approach for forest dynamics (e.g. Shugart 1998; Bugmann 2001) which uses, as the basic unit, small forest patches for describing competition between trees. Gap models have also been developed for steppe vegetation (e.g. Coffin and Lauenroth 1990). Part of the success of gap models is rooted in the fact that they are structurally realistic models designed with tree fall and succession patterns in mind. Ideally, a more general model such as a gap model would be developed with respect to several emergent patterns common to all sites where it should be applied. This approach means that model behaviour that would be applied to a single site study must now be modelled as being dependent on site characteristics. To be successful, the general model must be a strict simplification when viewed from single sites, but it gains complexity by the need to accommodate more general mechanisms working with different relative importance at different sites.

6.6 Concluding Remarks

This chapter has begun and ended with consideration of models; what specific model structures may allow us to do and what we can learn from specific model applications. In between, we have examined the issues that our current understanding suggests drive the emergence of pattern in dryland vegetation. A fundamental problem that exists is the clear separation of drivers and emergent properties; be that in models or empirical understanding. Feedbacks and the complexities of a multiplicity of processes interacting at a range of spatial and temporal scales confound apparent cause-and-effect relationships.

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Part II
Methods for Confronting Models with Data

Chapter 7
Approaches to Modelling Ecogeomorphic Systems

Laura Turnbull, Tamara Hochstrasser, Mareike Wieczorek, Andreas Baas, John Wainwright, Stefania Scarsoglio, Britta Tietjen, Florian Jeltsch, and Eva Nora Mueller

Abstract Drivers of land degradation often co-occur and their effects are often non-additive because of internal system feedbacks. Therefore, to understand how drivers of land degradation alter ecogeomorphic patterns and processes, novel tools are required. In this chapter we explore different modelling approaches that have been developed to simulate pattern formation, and ecological and geomorphic processes. These modelling approaches reflect some of the best available tools at present, but notably, they tend to simulate only one or at best two components of the ecogeomorphic system. The chapter culminates with a discussion of these different modelling approaches and how they provide a foundation upon which to develop much needed ecogeomorphic modelling tools.

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

T. Hochstrasser
School of Biology and Environment Science, Agriculture & Food Science Centre,
University College Dublin, Belfield, Dublin, Ireland
e-mail: tamara.hochstrasser@ucd.ie

M. Wieczorek
Department of Geosciences, Alfred Wegener Institute for Polar and Marine Research,
14473 Potsdam, Germany
e-mail: mareike.wieczorek@awi.de

A. Baas
Department of Geography, King's College London, WC2R 2LS London, UK
e-mail: andreas.baas@kcl.ac.uk

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

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7.1 Why Model Ecogeomorphic Processes?

Land degradation in drylands is a complex phenomenon involving changes in pattern-process relationships. There are multiple drivers of land degradation in drylands which often co-occur. These include grazing, fire management, soil-surface disturbance, temperature change and precipitation change. To investigate how drivers affect pattern-process relationships, a widely used approach is single-factor experiments where one driver is experimentally manipulated at a time. However, multiple drivers co-occurring can each affect different biotic and abiotic components of the system. For instance, grazing can cause a direct reduction in biomass of forgeable plants, while other types of soil-surface disturbances can alter soil infiltrability and erodibility. Due to pattern-process and biotic-abiotic feedbacks that are inherent in drylands, the effects of co-occurring drivers tend not to be additive, thus introducing non-linear behaviour (Peters and Havstad 2006; Turnbull et al. 2008; Okin et al. 2009). Multi-factorial experiments, which enable the systematic exploration of multiple different drivers on system response, have become more widely used in ecological studies (for example Norby and Luo 2004) to tease apart the effects of different drivers. However, these experiments can become very large, especially when they are replicated. As a result, this type of experimental design tends to be favoured more by ecologists who tend to carry out investigations with greater ease at the plant-patch scale. Because multi-factorial experiments tend to be limited to small plots and a limited range of environmental conditions, extrapolating results to broader spatial scales is challenging. Multi-factorial experiments are much less widely used in geomorphic studies, because the larger spatial (and often temporal) scale of enquiry makes the implementation of replicated multi-factorial experiments virtually impossible. Since we are concerned here with understanding ecogeomorphic processes, the use of multi-factorial experimental designs has limited scope. New tools are needed to enable us to explore how drivers of land degradation affect ecogeomorphic processes across multiple spatial and temporal scales. The development of ecogeomorphic

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S. Scarsoglio
Dipartimento di Idraulica, Trasporti ed Infrastrutture Civili, University of Turin,
10129 Torino, Italy
email: stefania.scarsoglio@polito.it

B. Tietjen
Institute of Biology, Freie Universität Berlin, 14195 Berlin, Germany
e-mail: britta.tietjen@fu-berlin.de

F. Jeltsch
Plant Ecology and Nature Conservation, University of Potsdam, 14469 Potsdam, Germany
e-mail: jeltsch@uni-potsdam.de

E.N. Mueller
Institute of Earth and Environmental Science, University of Potsdam, 14476 Potsdam, Germany
e-mail: eva.mueller@uni-potsdam.de

models has great potential to provide a new generation of tools that will enable us to explore in greater depth the effects of co-occurring drivers on pattern-process relationships in drylands and thus the dynamics of land degradation. For instance, ecogeomorphic models will allow us to carry out modelling-based multi-factorial experiments for different combinations of environmental drivers and disturbances, initial conditions, parametric states and spatial scales; all of which cannot be readily undertaken using field-based experimental approaches alone. To maximise the development of ecogeomorphic modelling tools, both ecogeomorphic modelling and ecogeomorphic field experimentation should be two-way; as well as using modelling tools to guide the design of field experimentation, modelling should also be used as a tool to interpret results of field experimentation.

In developing ecogeomorphic modelling tools, perhaps the best starting place is to outline the state-of-the-art modelling tools that have already been developed to simulate biotic and abiotic patterns and processes in drylands. In this chapter, deterministic and stochastic modelling approaches of pattern formation are detailed, which are widely used in studies of pattern formation in drylands. Then, process-specific [ecological and geomorphic] modelling approaches are explored, including finite difference and finite element approaches, and rule-based approaches such as cellular automata (CA) models. The first of these models is EcoHyd which simulates soil moisture and vegetation dynamics. Next, ECOTONE is outlined, which is used to explore the effect of gap formation on vegetation dynamics. The effects of spatially variable vegetation and soil properties on runoff and erosion dynamics are then explored using MAHLERAN, which is an event-based runoff and erosion model. Wind erosion-vegetation feedbacks are investigated using the cellular automata model, DECAL. In the concluding section of this chapter these modelling approaches are discussed in terms of how they can be used to provide a foundation upon which to develop ecogeomorphic models.

7.2 Deterministic Models of Pattern Formation

Theories used to explain self-organizing vegetation patterns are often based on deterministic symmetry-breaking instability as discussed in detail in Chap. 3. Symmetry-breaking instability is a mechanism whereby the spatial dynamics of vegetation, resulting from cooperative and inhibitory interactions occurring at different spatial ranges, destabilize the homogenous state of the system in turn leading to heterogeneous distributions of vegetation and thus, pattern formation (Borgogno et al. 2009).

As an example of the deterministic approach, the now-classic example of Lefever and Lejeune (1997) is considered, who attempted to explain the origin of patterns in Tiger Bush using a form of what Borgogno et al. (2009) define as a kernel-based model (see Sect. 3.7). The basic form of the Lefever and Lejeune model is:

$$\frac{\partial V}{\partial t} = RC - M \tag{7.1}$$

where V is the vegetation growth, expressed at community level (so ignoring the effects of different species); R is a function representing the reproduction of the vegetation, C is a function representing competition and other interactions producing a limit to growth, and M is a function reflecting vegetation mortality, either by autogenic or allogenic (e.g. fire, grazing) processes. The dynamics of the vegetation community are evaluated as $V(\mathbf{s}, t)$ where s is a point in space and t is time. Each of the functions R, C and M is defined as operating in the neighbourhood of the vegetation under consideration, with the neighbouring area defined as $\mathbf{s} + \mathbf{s}'$, based on weighting functions $w_{\{R,C,M\}} = w_{\{R,C,M\}}$ ($\mathbf{s}'$, $L_{\{R,C,M\}}$). The parameters $L_{\{R,C,M\}}$ are characteristic lengths over which the reproduction, competition and mortality interactions occur, which define the extent of spatial interactions in the model.

Lefever and Lejeune define the reproduction function as:

$$R = \int_{\Omega} \mathbf{ds}' \lambda w_R \left(\mathbf{s}', L_R\right) V\left(s + s', t\right)\left[1 + \Xi V\left(s + s', t\right)\right] \tag{7.2}$$

where λ is the growth rate in the absence of interactions with other plants and Ξ is a parameter reflecting the effects of cooperation on growth (e.g. through changes in local infiltration rate or nutrients through decay of shed parts). The competition function is:

$$C = 1 - \int_{\Omega} \mathbf{ds}' w_C \left(s', L_C\right) \frac{V\left(s + s', t\right)}{K} \tag{7.3}$$

where K is the maximal density of plants in a given area. The mortality function is:

$$M = \int_{\Omega} \mathbf{ds}' \eta w_M \left(\mathbf{s}', L_M\right) V\left(s + s', t\right) \tag{7.4}$$

where the mortality rate, η, is the inverse of the average lifespan of the vegetation. The model thus has seven parameters, three of which are measureable properties of the vegetation community. The weighting functions are taken to be Gaussian, so that in a 2D model:

$$w_i \left(\mathbf{s}', L_i\right) = \frac{1}{2\pi L_i^2} \mathbf{e}^{-\left(\frac{|\mathrm{s}'|}{2L_i^2}\right)} \tag{7.5}$$

where i = {R,C,M}. Combining these equations and carrying out a linear stability analysis shows that symmetry-breaking instability can only occur to produce patterns when $L_R < L_C$, and when $K\ \Xi > 0$. In other words, the two conditions correspond to (i) the length scale over which reproduction occurs being shorter than the length scale over which competition occurs and the; and (ii) reproduction having to be cooperative, respectively. Defining the parameter $\mu = \eta/\lambda$, the stability analysis shows that patterns only occur at intermediate values $\mu_c \leq \mu \leq \mu'_{\mathrm{c}}$. Lefever

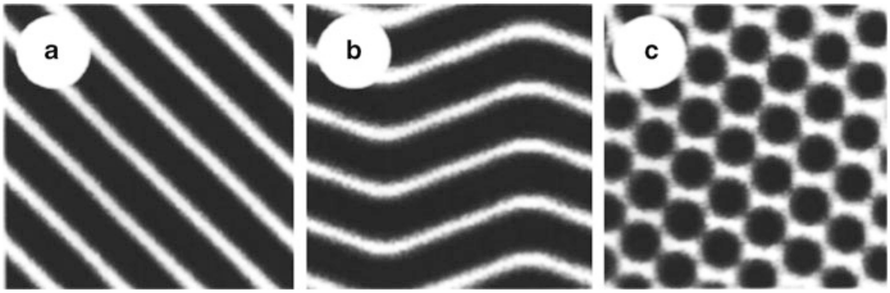

Fig. 7.1 Three examples of spatial patterns obtained for an isotropic system (*white regions* correspond to less-vegetated areas). (**a**) Pattern constituted of stripes having a uniform orientation ($\mu = 0.95$, $L = 0.1$, $\Lambda = 1$). The latter is determined by the initial condition. (**b**) Coexistence of stripes with two different orientations ($\mu = 0.95$, $L = 0.15$, $\Lambda = 1$). The relative orientation is determined by the choice of the parameters. The global orientation depends on the initial condition. (**c**) High-density *spots* arranged in hexagonal lattice on a low-density background ($\mu = 0.95$, $L = 0.1$, $\Lambda = 0.8$). In all cases, the periodicity corresponds to a wavelength approximately given by $\lambda_c = 2\pi/k_c$ (Source: Lefever and Lejeune 1997)

and Lejeune thus call μ a "switching parameter", and note that its value as the ratio of mortality to birth rates can also be considered as an index of aridity. Numerical simulation shows that the model can produce both striped and spotted patterns when the system is isotropic (i.e. not driven by fluvial or aeolian processes with a dominant direction: Fig. 7.1), and bands and arcuate features where there is anisotropy (Fig. 7.2). Key characteristics of the model are that the wavelength of the patterns decreases with vegetation density, that dynamic patterns under anisotropy can occur both in slope- and contour-parallel directions, and that contour-parallel bands tend to move upslope. All of these characteristics have been observed in Tiger Bush in the field (but see further discussions in Chaps. 12 and 13).

7.3 Basic Stochastic Processes Able to Induce Ordered Structures

The formation of vegetation patterns in drylands is commonly associated with symmetry-breaking instability in deterministic models, as outlined above. However, random fluctuations in environmental drivers may also play a critical role in the dynamics of patterns in non-linear systems (Borgogno et al. 2009; see also the discussion in Chap. 3). The emergence of new ordered states in dynamical systems, in time and in space, has been attributed to stochastic fluctuations – termed "noise-induced phase transitions" – which destabilize a homogeneous (symmetrical) state of the system (Scarsoglio et al. 2011). While random fluctuations in environmental processes are pervasive, applications of the theories of noise-induced pattern formation are rare and have only been developed more recently (Borgogno et al. 2009; Scarsoglio et al. 2011).

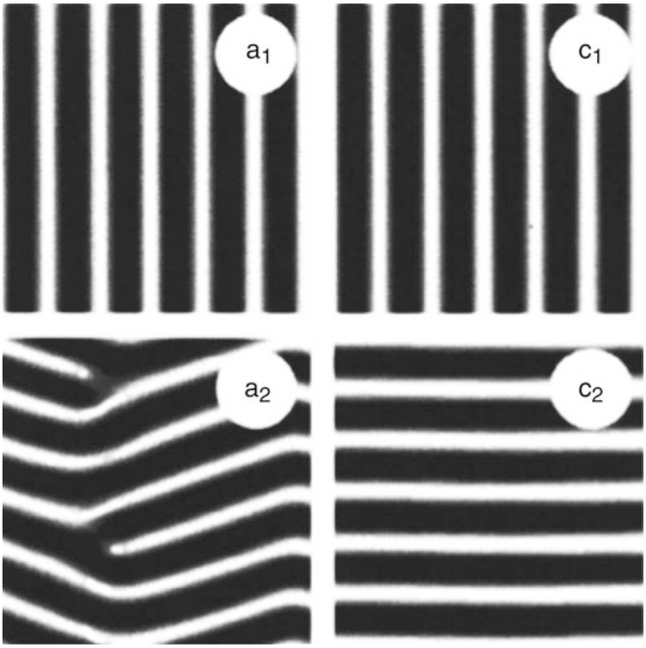

Fig. 7.2 Vegetation patterns in the anisotropic case. The influence of anisotropy (in the y direction) on reproduction (patterns a_1, c_1 obtained for $t_1 = 1$ and $t_2 = 0$) and on inhibition (patterns a_2, c_2 obtained for $t_1 = 0$ and $t_2 = 1$) is simulated. The simulations a_i ($i = 1,2$) correspond ($\mu = 0.95$, $L = 0.1$, $\Lambda = 1$), in the isotropic case, to a banded pattern (cf. Fig. 7.1a). The simulations c_i correspond ($\mu = 0.95$, $L = 0.1$, $\Lambda = 0.8$), in the isotropic case, to a pattern of hexagonal symmetry (cf. Fig. 7.1c). Reproduction anisotropy selects stripes parallel to the anisotropy direction and inhibition anisotropy selects stripes orthogonal to that direction, independently of the spatial symmetry properties of the patterns obtained in the isotropic case for the same values of parameters. Parallel stripes are static, while orthogonal stripes are moving upward, i.e. in the positive y direction (Source: Lefever and Lejeune 1997)

In order to explain the mechanisms of noise-induced pattern formation, two examples of stochastic models are outlined, which can be expressed by Eq. 3.11 presented in Sect. 3.8. The first model is:

$$\frac{\partial \phi}{\partial t} = -\phi + \xi\left(\overrightarrow{r}, t\right) + D\nabla^2\phi. \tag{7.6}$$

where ϕ is vegetation cover. In this case, deterministic dynamics ($\xi\left(\overrightarrow{r}, t\right) = 0$) damp the field variable to zero and do not exhibit steady pattern formation. The additive noise ($\xi\left(\overrightarrow{r}, t\right) \neq 0$) is able to keep the system away from the homogeneous state, while the spatial coupling induces spatial coherence. Thus, pattern formation is clearly noise-induced and arises from a synergism between additive noise and spatial coupling.

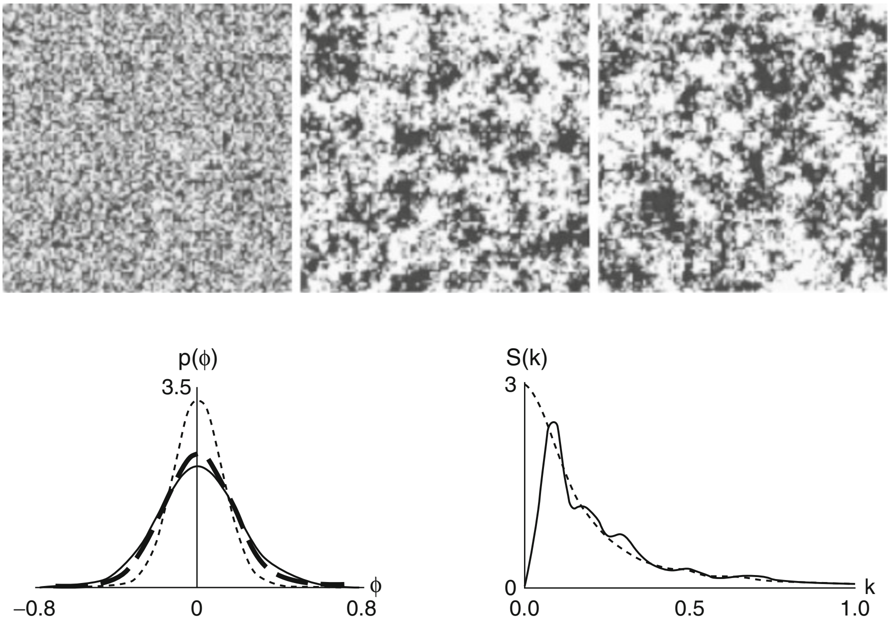

Fig. 7.3 Model (7.6.1) with $D = 50$, $s = 3$. *Top*: Numerical simulation of ϕ at $t = 0.10,100$. *Bottom*: pdf (*solid line*: numerical simulation; *dotted line*: classic mean-field analysis, *thick*: corrected mean-field analysis; Sagues et al. 2007) and azimuthal-averaged power spectrum S (*solid*: numerical simulations, *dotted*: structure function) of ϕ at $t = 100$. The numerical simulations use the Heun's scheme (Sagues et al. 2007) on a two-dimensional square lattice with 128 × 128 sites, with periodic boundary conditions, and uniformly distributed initial conditions between [−0.01, 0.01]. *Black* and *white tones* in the figures represent positive and negative values of the field, respectively, with *black* representing vegetation

Figure 7.3 shows the onset of patterns in the model (Eq. 7.6). No clear periodicity is visible but many wavelengths are present to produce multiscale patterns with irregular boundaries, which persist in the steady state. No phase transition occurs since the probability density function (pdf), which is numerically and analytically determined at steady state (see Chap. 3 for more detail), remains unimodal and with zero mean.

Numerical and analytical results in the Fourier space (see Sect. 8.4.4.2) confirm that there is no dominant wavelength different from zero. Equation 7.6 can be used to express the temporal evolution of the existing vegetation, ϕ, as the result of local linear decreasing dynamics, random rainwater availability, and vegetation's ability to develop spatial interactions. The distribution of vegetated sites in semi-arid environments exhibits spatial configurations resembling those shown in Fig. 7.3 (von Hardenberg et al. 2010; Scanlon et al. 2007).

The second model is:

$$\frac{\partial \phi}{\partial t} = -\phi - \phi^3 + \phi \xi \left(\vec{r}, t\right) - D\left(\nabla^2 + k_0^2\right)^2 \phi \tag{7.7}$$

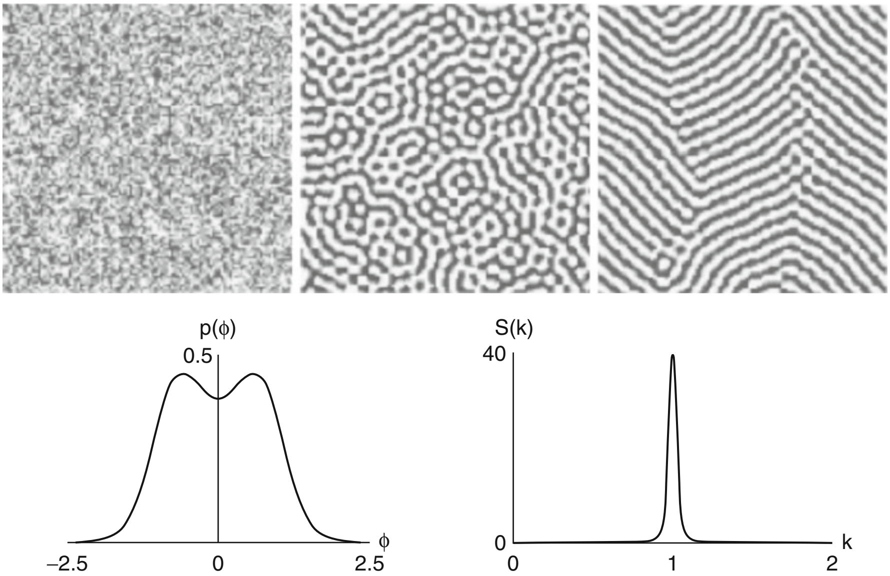

Fig. 7.4 Simulation results of the model represented by Eq. 7.7 with $D = 20$, $s = 2$, $k_0 = 1$. *Top*: numerical simulation of ϕ at $t = 0$, 10, 100. *Bottom*: pdf and azimuthal-averaged power spectrum S of ϕ at $t = 100$. See Fig. 7.3 for details on the numerical simulation

In this case, pattern formation relies on two actions: (i) multiplicative noise ($\phi\xi\left(\vec{r}, t\right)$) temporarily destabilizes the dynamics, and (ii) spatial coupling exploits this short-term instability, giving rise to the pattern. If the noise intensity is below a critical threshold ($s < s_c$) or if the noise is absent ($\xi\left(\vec{r}, t\right) = 0$), patterns are transient and fade away as the system approaches steady state. If the noise intensity exceeds the threshold ($s > s_c$), steady patterns occur.

Figure 7.4 displays an example ($s > s_c = 1$) where patterns exhibit a clear dominant wavelength and are statistically stable, although in the transient they evolve from a labyrinthine to a striped shape. At $t = 100$, the pdf of the field shows a weak bimodality with zero mean, demonstrating that no phase transition occurs, while the power spectrum has a peak corresponding to k_0, confirming that a clear periodicity is present. Equation (7.7) can be used to describe the temporal evolution of vegetation, ϕ, as the result of a local biomass loss, a disturbed local increasing dynamics, and the interplay between long and short-range interactions. A number of environmental patterns exhibit a regular and periodic spatial behaviour similar to the one shown in Fig. 7.4 (Couteron and Lejeune 2001; Lefever et al. 2009). Remarkable examples are given by peatlands (Eppinga et al. 2008), arctic hummocks and patterned ground (Gleason et al. 1986), and vegetation in semi-arid regions (e.g. Valentin et al. 1999; Esteban and Fairen 2006; Ridolfi et al. 2011).

Although this stochastic modelling approach and the deterministic modelling approach outlined previously are both able to reproduce patterns, they do not contribute to understanding the physical mechanisms that are responsible for pattern formation. In the following sections, models that attempt to address these mechanisms more directly will be considered.

7.4 Modelling Feedback Mechanisms Between Vegetation, Soil-Moisture Dynamics and Degradation

The model EcoHyD (Tietjen et al. 2010) is a combination of a two-layer soil-moisture-dynamics model, HydroVeg (Tietjen et al. 2009), and a dynamic vegetation model, which has been used previously to explore the effects of intra-annual rainfall variability and temperature on coupled water-vegetation dynamics in drylands (Tietjen et al. 2010; Jeltsch et al. 2010b). This modelling approach enables the effects of hydrological and ecological processes and their feedbacks to be disentangled. EcoHyD is spatially explicit and grid-based. Each cell has a spatial resolution of 5×5 m and is characterized by a specific topographic height. The spatial extent of the model domain is adjustable. Open boundary conditions are implemented so that water losses due to runoff are possible. Hydrological processes are simulated on an hourly to daily resolution and ecological processes on a fortnightly to yearly basis. The main processes represented in EcoHyd are presented in Fig. 7.5.

In HydroVeg, precipitation is received at hourly intervals and accumulates on the surface and infiltrates into soil layers, either as fast infiltration into deeper layers along roots and via macropores or slower infiltration into the upper layer according to the Green and Ampt (1911) approach. If the amount of surface water exceeds the hydraulic conductivity of the soil, ponding occurs and surface water is lost to the lowest neighbouring cell. Further water losses from a cell occur as evapotranspiration (ET) which is calculated daily using the Hargreaves (1974) approach. Between the two soil layers, a diffusive flux is accounted for, as well as water loss to deeper layers by drainage. Infiltration, runoff and water loss by ET depend on soil texture and the prevailing vegetation cover. The amount of surface water runoff per hour is furthermore dependent on the slope of a cell and increases with steepness (following the approach of Manning-Strickler: Dingman 1994).

EcoHyd simulates the fate of two vegetation-growth forms: grasses and shrubs/trees, which are the main life forms in the simulated savannas. Plants compete for water and space, and vegetation cover changes as a result of water-dependent growth, mortality and dispersal. Fortnightly growth follows a logistic behaviour and is reduced by limited water availability and competition. Mortality results from low soil-moisture content during the growing season, or direct removal of vegetation due to disturbances such as grazing. Grass dispersal is assumed to be homogenous in space, while shrub dispersal is limited and decreases exponentially with distance from the source cell. Both mortality and dispersal are calculated yearly, at the end of the growing season.

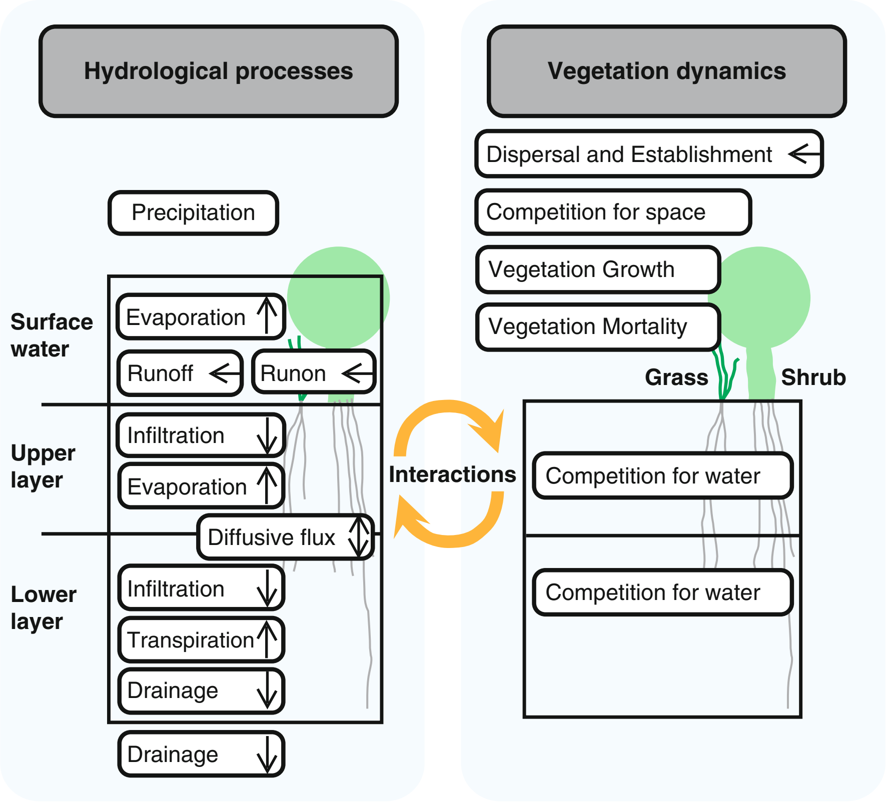

Fig. 7.5 Overview showing the main components of EcoHdy: HydroVeg (*left*) and the dynamic vegetation model (*right*). Processes represented within a single grid cell and interactions between cells are shown (Adapted from Tietjen et al. 2010). *Arrows* indicate the direction of interactions between cells and soil compartments (↓: water reaching compartment; ↑: water leaving compartment; ←: water/seed exchange between cells)

Exploratory analysis is first carried out using only HydroVeg (Tietjen et al. 2009) to determine the effects of static vegetation cover, vegetation composition and soil texture on soil-moisture dynamics for a single precipitation event on a flat site. Longer term feedbacks to growth, mortality and dispersal are not considered in this instance, but are addressed in subsequent exploratory analysis using EcoHyd. In these initial simulations, total soil depth was set to 0.80 m, with the top 0.20 m belonging to the upper soil layer (following Tietjen et al. 2010). The effects of differences in vegetation cover (grass and shrub) on soil moisture for a 15 mm precipitation event, distributed within 1 day followed by 11 dry days are presented in Fig. 7.6. Results show that moisture dynamics in the upper soil layer are hardly impacted by vegetation composition or soil texture, since the flat topography leads to negligible runoff of surface water and therefore most water eventually infiltrates. In contrast both vegetation composition and soil texture influence soil-moisture

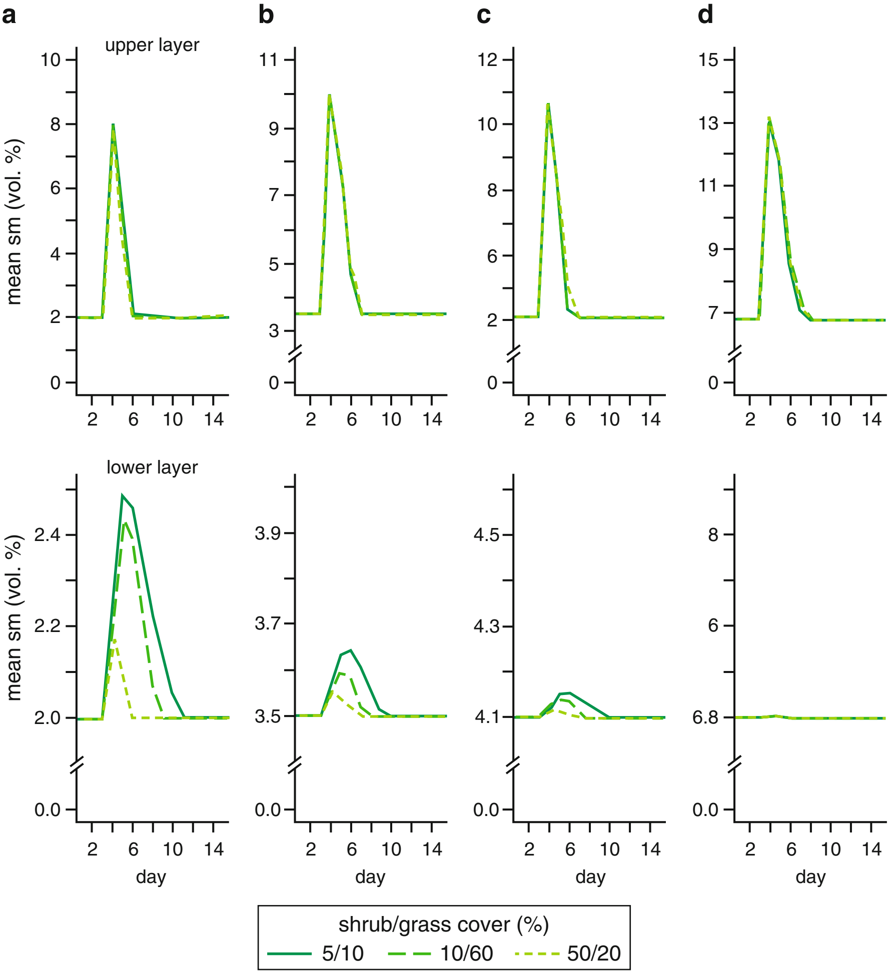

Fig. 7.6 Soil moisture (*sm*) after a 15 mm precipitation event, for two soil layers (*upper and lower panel*), three vegetation covers and four soil types (**a**) sand, (**b**) loamy sand, (**c**) sandy loam and (**d**) sandy clay loam

dynamics in the lower layer. Infiltration depth is strongly determined by soil texture. Fine-textured soils do not facilitate much deep infiltration, whereas coarse-textured sandy soils do. Therefore coarse-textured sandy soils may create favourable conditions for shrubs, potentially leading to higher encroachment on sandy than on loamy soils. Vegetation composition has a strong impact on soil-moisture dynamics in the lower soil layer. A higher shrub cover rapidly reduces soil moisture in the lower soil layer after a precipitation event via transpiration within 2 days. In contrast, with higher grass cover it takes up to 5 days until soil moisture is reduced down to the residual water content.

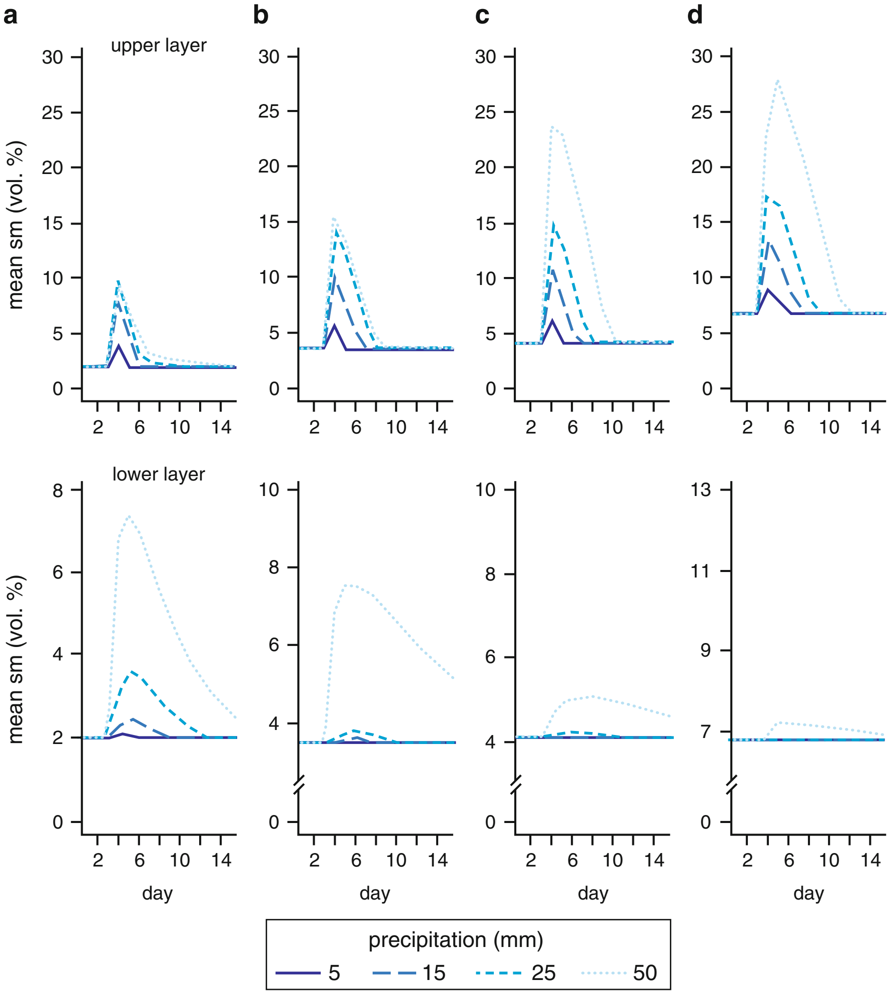

Fig. 7.7 Soil moisture (*sm*) after four different precipitation events (5, 15, 25 and 50 mm) for two soil layers (*upper and lower panel*) and four soil types (**a**) sand, (**b**) loamy sand, (**c**) sandy loam and (**d**) sandy clay loam with 60 % grass cover and 10 % shrub cover

To assess the impact of extreme events, soil-moisture dynamics were examined after single precipitation events of different intensities (totals of 5, 15, 25, and 50 mm on 1 day), followed by 11 dry days (Fig. 7.7) in simulations with 10 % shrub cover and 60 % grass cover. The results demonstrate that a shift to more extreme events as predicted in the course of climate change (Easterling et al. 2000) will influence water availability for plants in shallow and deeper soil layers differently for different soil textures. While soil texture mainly influences the lower soil layer for smaller events as shown above, high intensities of precipitation additionally influence soil moisture in the upper layer with highest changes of the water content in fine-textured soils. Although topography and runoff were not accounted for in this analysis, spatial variations in soil texture in topographically variable landscapes influences water losses further, by affecting runoff generation which is higher on soils with low permeability and sites with low vegetation cover (Martinez-Mena et al. 1998; Bartley et al. 2006).

In a further analysis using EcoHyd, feedbacks between vegetation and soil moisture are explored in the case of Namibian savannas. Four major soil-texture classes can be found in Namibia, namely sand, loamy sand, sandy loam and sandy clay loam (Schwartz 2006). Mean annual precipitation ranges from less than 50 mm to more than 600 mm (Atlas of Namibia Project 2002). Livestock farming plays a prominent role in Namibian agriculture (Ministry of Agriculture, Water and Forestry 2009). However, high grazing pressure can lead to either shrub encroachment or to a reduction in vegetation (Rietkerk and Van de Koppel 1997; Roques et al. 2001; Kuiper and Meadows 2002). This degradation is thought to be enhanced by climate change (Fischlin et al. 2007). In Jeltsch et al. (2010a) EcoHyD was used systematically to assess the impact of different combinations of climate change, soil type and grazing intensity on savanna degradation for a broad range of semi-arid Namibian savanna sites. In this analysis, EcoHyD was applied to four different soil textures and three precipitation regimes, spanning the extent of the Namibian thornbush savanna (Joubert et al. 2008), with mean annual precipitation (MAP) ranging from 200 to 500 mm (Fig. 7.8).

Grazing by cattle was varied from low to high intensity, represented as annual grazing rates of 2, 5, 10 and 20 % of the grass cover. To allow for better comparison, the same topography was applied to all areas in the model. The effects of climate change were implemented by reducing mean annual precipitation of each site by 10 % and increasing mean annual temperature by 2.25 °C, following Jeltsch et al. (2010a). An index *deg* (Jeltsch et al. 2010b) was developed to reflect possible vegetation changes and degradation trends and describes the change of perennial grass cover as a consequence of grazing in comparison to scenarios without grazing (deg = cover of perennial grasses). If the absolute value and mean increase of shrub cover after 30 years is higher than 5 %, the absolute value of deg (namely $|\text{deg}|$), is multiplied by $(+1)$, else it is multiplied by (-1). Since all scenarios in our simulations show a negative trend of grass cover under grazing, i.e. $\text{deg} < 0$, high negative values indicate a prominent risk of desertification by vegetation losses, while positive values indicate potential shrub encroachment.

Simulation results show an increasing risk of vegetation degradation with increasing grazing intensity (Fig. 7.8): grass cover decreases (quantity given by the size of the bars), while the change in shrub cover (sign of *deg*) depends on the combination of grazing intensity, precipitation, soil texture and the climate scenario. None of the scenarios shows shrub encroachment under low grazing intensity, since grass still dominates the system. However, if grazing reduces grass cover substantially by 5 % or more, shrub encroachment can occur, depending on soil texture and MAP. With finer soil texture, fewer scenarios lead to an increase of woody vegetation. If the same grazing scenario is applied under climate-change

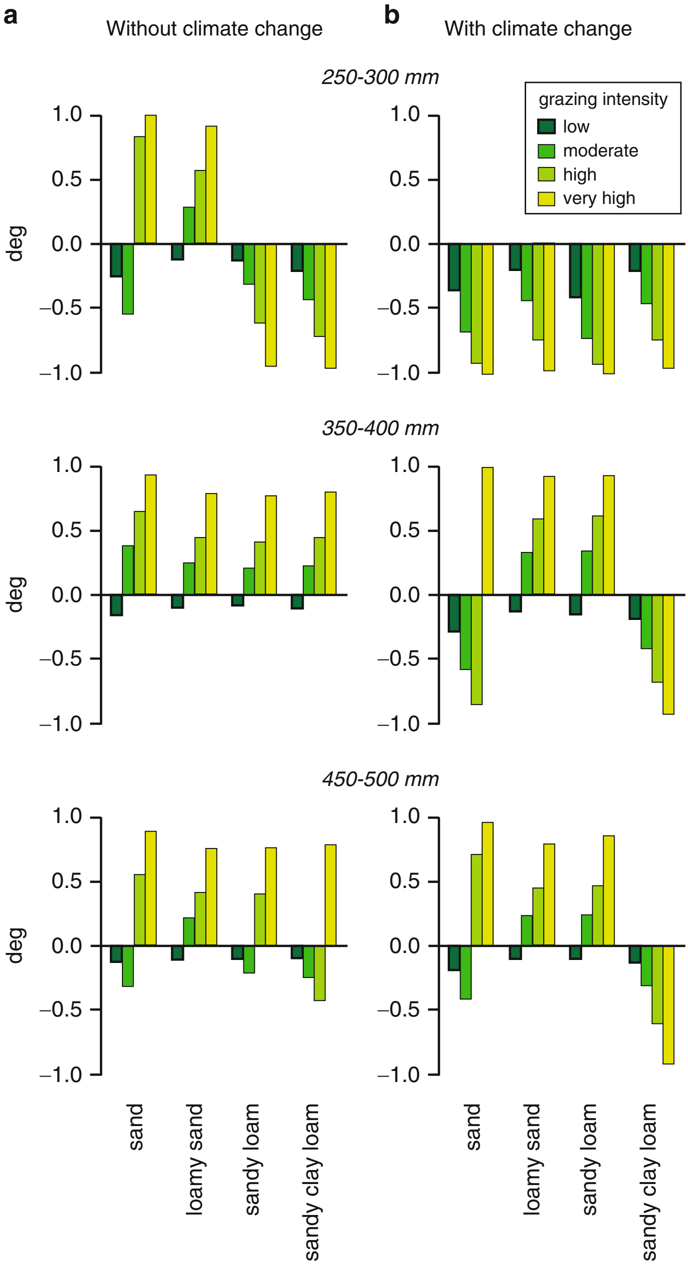

Fig. 7.8 Effects of 30 years of grazing with different intensities on the risk of savanna degradation without (**a**) and with (**b**) climate change (cc). Savanna areas are categorized according to actual mean annual precipitation (250–300, 350–400 and 450–500 mm). Degradation risks are summarized with an index *deg*, which integrates information on loss of perennial grass cover and risks of shrub encroachment (positive values indicating shrub encroachment versus negative values indicating desertification, see text for further details)

scenarios, the predicted increase of shrub cover is reversed for the two drier sites, since the low soil-moisture content does not sustain dense vegetation cover. In contrast, under more moist conditions shrub encroachment is still possible. Little water infiltrates into the deep soil layers of sandy clay-loam soils (Figs. 7.6 and 7.7). At these sites, shrub encroachment does not occur under reduced precipitation and increased temperature, even at high grazing intensities (see also Dodd and Lauenroth 1997). These results show that soil texture plays a crucial role in the response of vegetation to grazing. Furthermore, these results suggest that grass cover suffers most from grazing and climate change on soils with a high fraction of sand, which may be the result of the higher pore size distribution of sandy soils (Rawls et al. 1982) which allows for more infiltration into deeper soil layers. Less water is lost to evaporation from lower soil layers (Noy-Meir 1973) and is therefore available for plants for a longer period (Figs. 7.6 and 7.7), which may especially benefit shrubs with their deep rooting system (Walker et al. 1981).

7.5 Modelling Vegetation Dynamics Using Gap-Dynamics Models

An alternative approach to simulate vegetation dynamics is to use gap-dynamics models. Gap formation is a key process in the dynamics of plant communities (Li et al. 2005). Gaps provide opportunities for the regeneration of resident plant species and also for the establishment of newcomers. Therefore gaps are thought to play a major part in regulating species composition and plant diversity (Grubb 1977; Silvertown and Smith 1988; Li et al. 2005). Gap models simulate the establishment, growth and mortality of each individual plant on a small plot (Coffin and Urban 1993; Bugmann 2001; Perry and Enright 2006), and have become one of the most-used approaches for modelling vegetation dynamics. Gap models are based on the principle of niche differentiation between different plant species, in terms of their ability to compete for limiting resources (*competition*) and to cope with environmental fluctuation and disturbance (*mortality* and *recruitment*).

ECOTONE is a gap-dynamics model developed for simulating vegetation dynamics of grasslands, invasive species and shrublands in arid and semi-arid areas (Goslee et al. 2001; Peters 2002; Hochstrasser and Peters 2005; Goslee et al. 2006). ECOTONE simulates vegetation dynamics on small plots, which are equivalent to the size of a full-grown individual of the dominant plant. Vegetation dynamics consist of a micro-succession of individuals from different functional groups or plant species, which is induced by the mortality of plants (i.e. the opening of resource gaps) (Fig. 7.9). In ECOTONE, plants compete for water (the most limiting resource in drylands) during this micro-succession. The distribution of available water is simulated using the soil-water model, SOILWAT (SOILWAT, Parton 1978). Water dynamics are calculated on a daily time step, while competition for available water occurs on a monthly basis, and growth and mortality of plants on a yearly

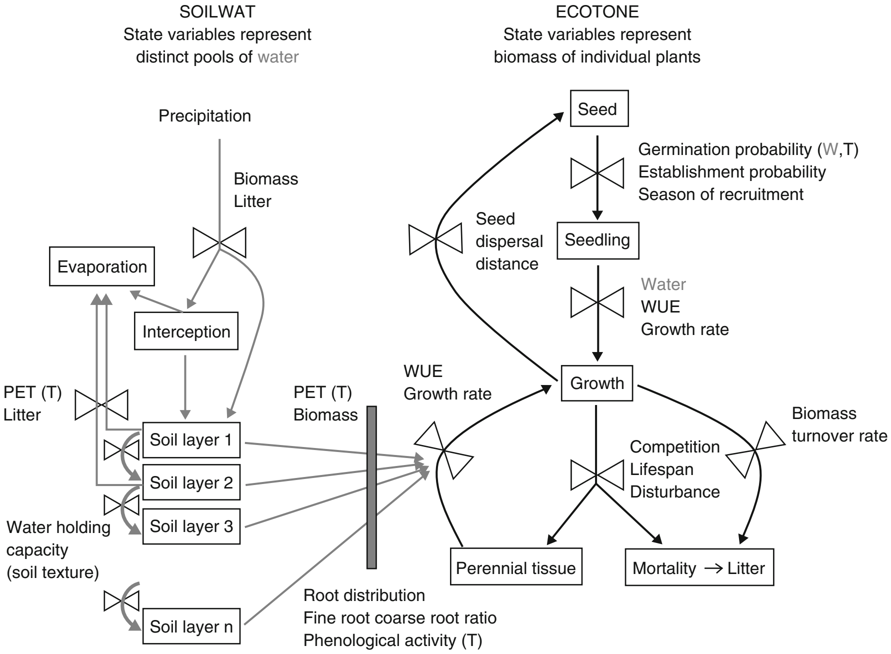

Fig. 7.9 Flow diagram of ECOTONE. *Grey arrows* indicate flows of water, *black arrows* indicate flows of biomass. Available water for plant growth depends both on climate (precipitation, temperature, PET) as well as on species characteristics (bar between SOILWAT and ECOTONE). *Letters in parentheses* after control variables indicate if they are dependent on water (*W*) or temperature (*T*) (Source: Hochstrasser 2001)

basis. The differences between these temporal resolutions of processes in the model were necessary since water dynamics can only be accurately simulated with high temporal resolution (Parton et al. 1998), whereas low temporal resolution is sufficient for simulating vegetation dynamics (Peters 2002).

Vegetation dynamics are driven by plant mortality, which opens up resource space (Fig. 7.9). Plants die due to competition, age, turnover and disturbance (Peters 2002). For example, young plants are less competitive than older plants because of their small size, while mortality increases with plant age, and therefore affects larger individuals (Coffin and Urban 1993; Bugmann 2001). Mortality may also occur due to competition and disturbance. Plant recruitment is determined stochastically, based on species recruitment probability, which is determined by seed availability multiplied by establishment probability (Peters 2002). It has been shown that recruitment can also be made a function of abiotic conditions, assuming that propagules are present in the soil (Hochstrasser 2001).

The ability of plants to acquire resources is strongly dependent on their biomass, especially leaf area for photosynthesis and root surface area for water uptake. In ECOTONE growth of plants is determined by symmetric competition for water between individual plants on the same plot. The amount of water taken up by each plant depends on its root biomass in each soil layer and its phenological activity (Peters 2002). An alternative approach (Hochstrasser 2001) uses asymmetric competition – i.e. larger plants are able to take up disproportionately more resources than they would based on the difference in biomass (Schwinning and Weiner 1998). It is assumed that all plants can take up water within the same range of soil-water potential, although there are limitations associated with this assumption. For example, creosotebush (*Larrea tridentata*) may be able to draw water from the soil at much lower water potentials than other arid land plants (Barbour et al. 1977). Water-use efficiency (WUE) determines how much the plant can grow given the water taken up. WUE can also be used to account indirectly for species-specific ranges of available water (Hochstrasser 2001).

In the following application of ECOTONE, recovery dynamics of black grama (*Bouteloua eriopoda*) grasslands versus mesquite (*Prosopis glandulosa*) shrublands following vegetation disturbance by traffic are investigated, using ECOTONE (Hochstrasser et al. 2005). ECOTONE was parameterized for black grama grasslands and mesquite shrublands in southern New Mexico. The model was driven by 80 years of daily weather data from the Jornada Experimental Range (1918–1997), southern New Mexico. Nine species and subdominant functional groups were simulated. Species parameters were derived from a literature survey of the dominant species (Hochstrasser et al. 2002). For subdominant species, existing parameterizations of the model were used (e.g. Hochstrasser 2001; Peters 2002). Vegetation composition was matched with field records of the vegetation on the soil used in this model (Hochstrasser et al. 2005). Plot-size was determined according to the resource space of a full-grown mesquite (1.0 m^2) and black grama plant (0.25 m^2). Dynamics for black grama and mesquite were investigated separately because these two life-forms operate at a different scale and this scale difference and its implications for grass-shrub interactions cannot yet be simulated well within ECOTONE. The overall objective of the study was to demonstrate how the difference in resistance (i.e. to the amount of damage done to the plant by disturbance) and resilience (i.e. the ability of the plant to recover from disturbance) between grasses and shrubs affects their recovery dynamics after disturbance. The tolerance range for pressure from disturbance was set highest for grasses, intermediate for shrubs and lowest for forbs (Hochstrasser et al. 2005).

Disturbance from vehicular traffic was simulated by partial or full mortality of the aboveground (and indirectly belowground) biomass of the plant. Effects of the disturbance were investigated at two levels: (i) the individual plant level (for disturbance intensities below the maximum tolerance of individual plants) and (ii) the population level (for disturbance intensities above the maximum tolerance of individual plants).

- A one-time disturbance was applied to an individual on a single plot to determine the effects of using a range of disturbance intensities, from a low-pressure impact (one passage of foot traffic) to a high-pressure impact just below the plant's maximum tolerance (one passage by a heavy vehicle). If the pressure

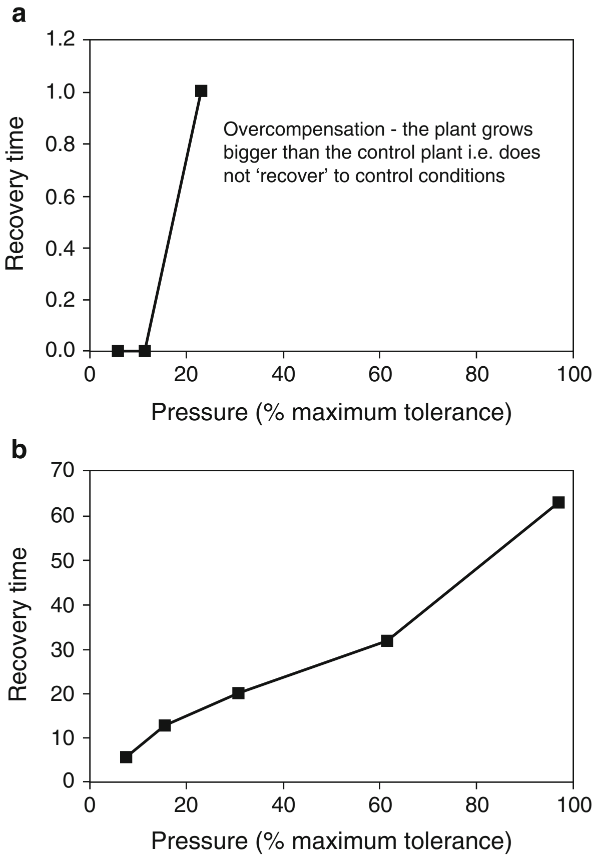

Fig. 7.10 Recovery of individual plants after a range of disturbances with different intensities. (**a**) Black grama (*Bouteloua eriopoda*) – a perennial grass with high tolerance of pressure – if the pressure applied was higher than 40 % of the maximum tolerance of plants, the impact of the disturbance on competitors of black grama resulted in the disturbed plant growing bigger than the control, thus not ever matching the control plants' biomass within 2 % (overcompensation). (**b**) Mesquite (*Prosopis glandulosa*) – a perennial shrub with low tolerance of pressure – recovery is slow in this case, but will occur – even at high pressures close to the tolerance range of the plant (Source: Hochstrasser et al. 2005)

impact was higher than the plant's tolerance, the plant died. When pressure was below the maximum tolerance, plant-recovery time (defined as the time it took the individual affected to get within 2 % of the aboveground weight of a control plant for the same year) was significantly different for black grama grass and a mesquite shrub (Fig. 7.10). Black grama recovered within a year from low-intensity disturbances and grew bigger than the control plant when disturbance intensity was higher (but below maximum tolerance) (overcompensation). The latter could be explained by the disturbance impact on competitors of black grama, which allows the already dominant plant on the plot to capture more resources than were previously available. In contrast, the recovery time of a mesquite shrub increased with the disturbance intensity. At intensities close to the shrub's maximum tolerance it takes up to 60 years to recover from the disturbance. The differences in the recovery of these two species can be explained by the amount of biomass removed by the disturbance as well as their growth rates. Black grama loses less biomass during disturbance, and has a higher growth rate than mesquite.

- At high intensities of disturbance – above the tolerance range of plants – the recovery from disturbance depends on recruitment. The latter may depend on the spatial extent of the disturbance. These effects were simulated indirectly by incorporating the effects of reduced seed availability on recruitment. A vegetation patch of 5 × 5 m in mesquite shrubland (25 plots) and 2.5 × 2.5 m (25 plots) in black grama grassland was simulated. Model results show that the black grama population was greatly reduced by the disturbance. Without dispersal limitation after the disturbance, as may occur in a small disturbance patch, the black grama population took an average of 20 years to recover. In contrast, the mesquite population took longer to recover to the control level (on average about 50 years). In the control simulation, mesquite biomass started to decline after year 150 as the population reached its maximum lifespan of 200 years. In contrast, the individuals in the disturbed populations were younger and maintained a high level of biomass at the end of the simulation run. Dispersal limitation after the disturbance, as may occur in large disturbance patches, impacted the recovery dynamics of both dominants (Fig. 7.11): the black grama population still recovered relatively rapidly, but could never attain the average biomass of the plot without the dispersal limitation. These results indicate that black grama populations are dependent on relatively high recruitment rates to maintain their population size and thus despite their high resistance and resilience to disturbance they are more vulnerable to disturbances than mesquite. In contrast, even though recovery in the mesquite population was slower than in a non-dispersal-limited situation, the population recovered and even expanded after the disturbance.

Results from this application of ECOTONE correspond with previous experiments investigating the effects of trampling disturbance on different life forms (Cole 1995; Yorks et al. 1997), demonstrating that the parameters used in gap-dynamics models to differentiate species with regard to their ability to compete for water, deal with environmental fluctuation and allogenic disturbance, can be used to forecast the behaviour of these species, and thus vegetation dominance and composition under different scenarios. While it is not possible to verify the accuracy of model simulations due to a lack of long-term experiments on vegetation recovery dynamics, these results provide valuable insight into plant-recovery dynamics and variations between species.

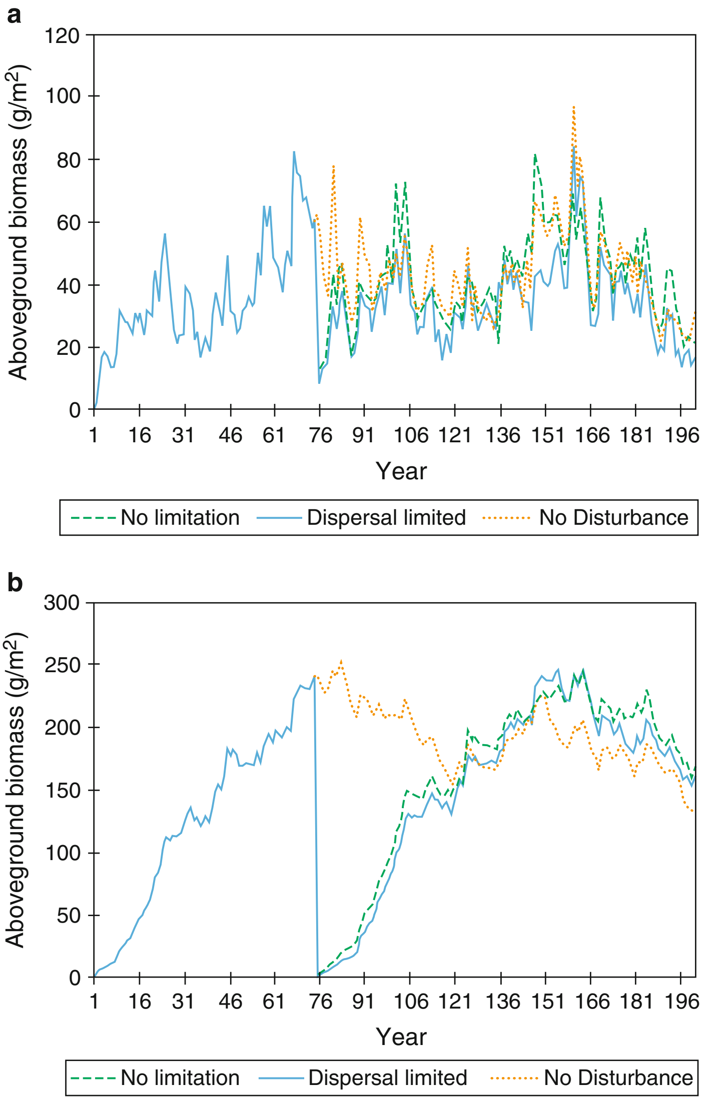

Fig. 7.11 Recovery of plant populations after high intensity disturbance (causing plant mortality). (**a**) Recovery of black grama (*Bouteloua eriopoda*) population is compared to (**b**) recovery of mesquite (*Prosopis glandulosa*) population. A situation with no dispersal limitation (as in a small disturbance) is compared to a dispersal limited situation (as in a large disturbance). The two cases are compared to a situation where no disturbance occurs to control for the natural variability of the vegetation (Modified from: Hochstrasser et al. 2005)

7.6 Modelling Redistribution of Water and Soil Resources

Spatial and temporal variation in vegetation and soil properties (discussed in the previous two sections) as well as surface microtopography have a profound influence on runoff and erosion (e.g. Wainwright et al. 2000; Abrahams and Parsons 1991; Calvo-Cases et al. 2003; Cammeraat 2004). The extent to which vegetation patches are connected and orientated in relation to predominant flow lines and the connectivity of soil properties and runoff-generating patches govern the runoff and erosion response at broader spatial scales (Bracken and Croke 2007; Müller et al. 2007a; Turnbull et al. 2008, 2010).

In many drylands, runoff is typically generated by relatively short-duration high-intensity rainfall events during which the infiltration capacity of the soil is often exceeded, leading to the generation of infiltration-excess overland flow (Horton 1945; Wainwright and Bracken 2011) which is one of the primary vectors of resource redistribution in drylands.

High-resolution timescales are necessary for modelling runoff and erosion in drylands, and therefore such models tend to be event based. Distributed modelling approaches are required because of the importance of the spatial distribution of vegetation, soil characteristics, and microtopography on runoff and erosion processes. Most spatially distributed modelling approaches divide the hillslope or catchment into a grid, which allows the effects of patterns on process to be represented. Water and soil resources are routed from cell to cell over the grid, using one of a number of standard flow-routing algorithms. In a distributed representation of a hillslope or catchment, each cell has a unique parameter value such that spatial variability of surface properties is represented. The spatial and temporal resolution of models simulating runoff and soil-redistribution processes is critical in terms of representing adequately the heterogeneity of surface characteristics and the temporal variability of rainfall characteristics, especially short bursts of especially high-intensity rainfall (Wainwright and Parsons 2002).

An example of a high-resolution event-based runoff-erosion model is MAHLERAN (Model for Assessing Hillslope to Landscape Erosion, Runoff And Nutrients). MAHLERAN is made up of three primary submodels: the runoff submodel, the erosion submodel and the nutrient submodel. The runoff submodel is the driver of erosion and nutrient dynamics. Each of these three submodels is briefly outlined here, and full details are in Wainwright and Parsons (2002), Parsons et al. (1997), Wainwright et al. (2008a, b, c), Mueller et al. (2007) and Turnbull et al. (2010).

The hydrological component of MAHLERAN uses a simple infiltration model to generate infiltration- and saturation-excess runoff (Wainwright and Parsons 2002). The infiltration rate is simulated using the Smith-Parlange approach (Smith and Parlange 1978). Runoff is routed over the hillslope using a kinematic wave approximation of the St Venant equations (Wainwright and Parsons 2002), with flow routing in the direction of steepest descent from cell to cell (in cardinal directions) over a finite-difference grid (Scoging et al. 1992), using a finite-difference solution

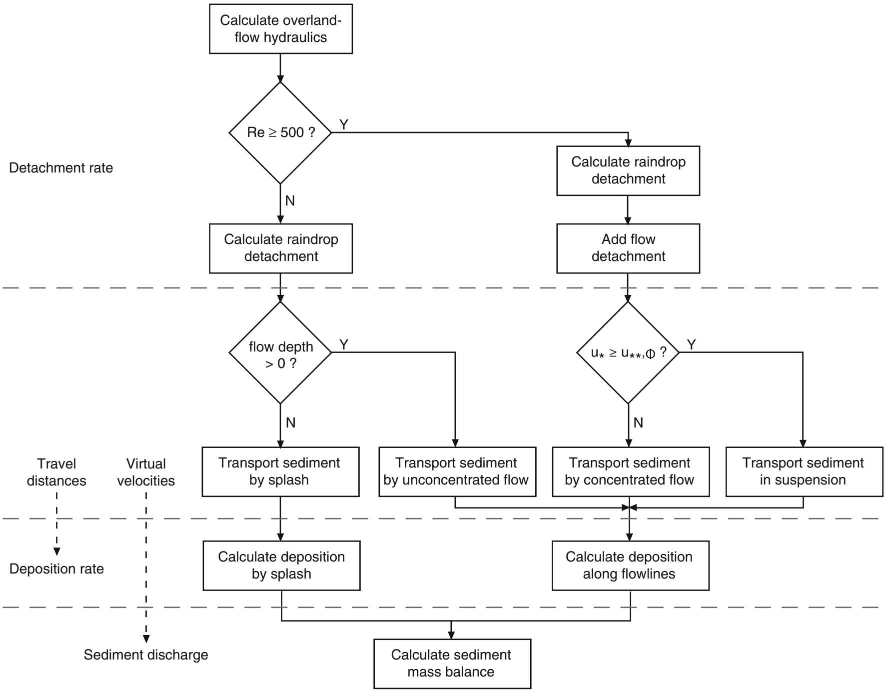

Fig. 7.12 Structure of the erosion submodel of MAHLERAN. Summary of the algorithms used in the erosion submodel of MAHLERAN. The main components in terms of detachment and the use of travel distance and virtual velocity to estimate sediment discharge are highlighted (Source: Wainwright et al. 2008a)

(Euler backward difference form; Scoging et al. 1992). Flow velocity is determined dynamically using the Darcy-Weisbach flow equation.

To achieve better prediction of soil erosion by runoff, process-based models are required (Parsons et al. 1997). The erosion submodel of MAHLERAN (Fig. 7.12), considers the interaction of both raindrop detachment, splash, unconcentrated and concentrated erosion as bedload and in suspension, which is necessary since the relative balance of these processes is the most critical control on the resulting pattern of erosion (Wainwright et al. 2008a). The erosion submodel is based on the concept of entrainment and travel distances of sediment in six particle size classes: <63 µm, 63 µm –0.25 mm, 0.25–0.5 mm, 0.5–2 mm, 2–12 mm and >12 mm. Further details can be found in (Wainwright et al. 2008a). Sediment detachment and transport is simulated for four conditions (Wainwright et al. 2008a):

- erosion as a function of raindrop detachment and transport occurs by splash when no flow is present;
- unconcentrated overland flow erosion simulated using raindrop detachment rates that are modified to account for the protective effects of the surface-water layer;

- transport by mixed unconcentrated and concentrated flow ($500 \leq Re < 2{,}000$) (Re is the flow Reynolds number);
- transport by concentrated flow ($Re > 2{,}000$).
- Sediment deposition is modelled using a transport-distance approach, whereby the distribution function of travel distances of particles transported via the different mechanisms and flow conditions enables determination of the deposition rates at each point along the transport pathway.

Dissolved nutrients are modelled conservatively according to an advection-dispersion model (Havis et al. 1992; Walton et al. 2000), in which the mass transfer of nutrients from the soil surface to runoff is driven by: (i) diffusion of dissolved nutrients from the soil interstices by movement of soil water into the overland flow; (ii) desorption of the nutrients from soil particles into the overland flow; (iii) dissolution of solid phase nutrients into the soil water or overland flow, and (iv) scouring of solid phase nutrients by hydraulic forces and subsequent transport and moving dissolution. A mass-transfer coefficient is used to lump together the mechanisms of mass transfer (Wallach and van Genuchten 1990; see Mueller et al. 2007 for more detail). Particle-bound nutrients are modelled as a function of the nutrient concentration associated with each particle-size class and the amount of sediment transported within each particle-size class. Particle-bound nutrient dynamics are modelled conservatively since it is assumed that there is no adsorption or desorption of particle-bound nutrients during transport (Viney et al. 2000).

MAHLERAN has been extensively evaluated for different conditions over dryland hillslopes with different types of vegetation, for different antecedent soil-moisture conditions and for rainfall events of varied magnitudes. Testing of the runoff submodel of MAHLERAN for a range of rainfall events over grassland and shrubland at the Sevilleta National Wildlife Refuge in central New Mexico shows that the runoff model generally performs well, although over shrubland there are some discrepancies with the timing of modelled peak discharge (Fig. 7.13). An extensive evaluation of the erosion component of MAHLERAN for sites at the Walnut Gulch Experimental watershed in southern Arizona demonstrates that MAHLERAN performs well when simulating the total amount of sediment eroded during a runoff event (Wainwright et al. 2008b, c). However, uncertainties introduced by the limited amount of data available for parameterizing detachment characteristics as a function of soil-particle size, mean that the proportion of fine sediment relative to coarse sediment is over-estimated by the model. In order to improve further the process-based understanding of erosion processes in drylands, continued laboratory-based experimentation is being undertaken to improve the ability to parameterize such models. Testing of the dissolved nutrient component of MAHLERAN indicates that the conservative modelling of event-based nutrient dynamics is inadequate because intra-event nutrient dynamics do not behave conservatively (Turnbull et al. 2011), and that parameterizing initial soil-nutrient content in event-based models is challenging because biogeochemical cycling in drylands is so temporally variable (Hartley and Schlesinger 2000; McCally and Sparks 2009). However, the approach employed to simulate particle-bound nutrients yields satisfactory results, with

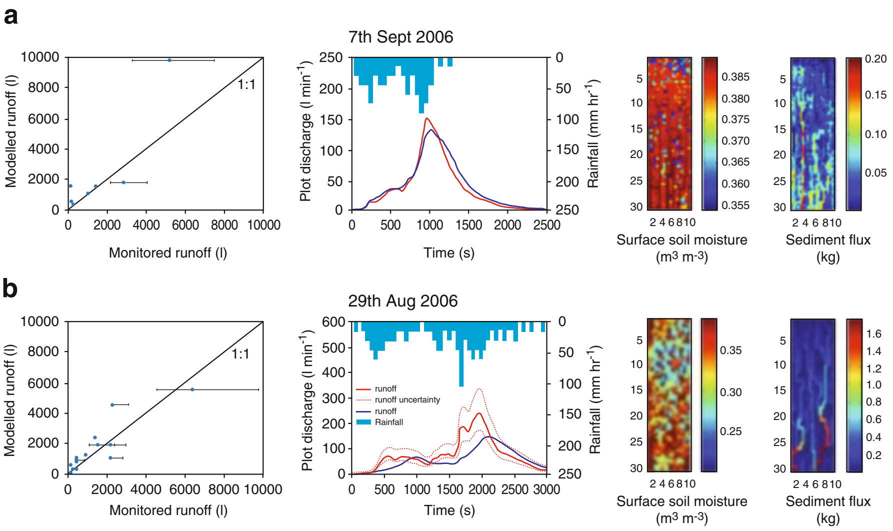

Fig. 7.13 Examples of testing the runoff submodel for (**a**) grassland and (**b**) shrubland. The plots on the *left* compare total event monitored runoff with total event modelled runoff, the plots in the *centre* show the monitored and modelled hydrographs for a single event, and the plots on the *right* show example outputs of spatial maps of water and soil resource redistribution during runoff events over 10 × 30 m grassland and shrubland plots at the Sevilleta National Wildlife Refuge, NM

excellent results for smaller runoff events, and an underestimation of modelled particle-bound nutrient yields for the largest events monitored (Turnbull et al. 2010).

Mueller et al. (2007) applied MAHLERAN to investigate the redistribution of water and soil resources over different types of vegetation boundaries (grassland, creosotebush, tarbush and mesquite; all of which differ in terms of their spatial characteristics, microtopography and influence on soil properties) in the Jornada Basin, New Mexico (32°31′N, 106°47′W; Fig. 7.14).

In this application, MAHLERAN was run with a model cell size of 10 × 10 m and a time resolution of 1 s (Mueller et al. 2007). They evaluated fluxes of water, soil and plant-essential nutrients at 20-m intervals along 60-m-wide transects through the vegetation boundaries, extending 140 m upslope of the boundary and 140 m downslope of the boundary (Fig. 7.15). Total fluxes at each point along transects were calculated as the sum of fluxes at that interval (across the 60-m width), scaled by dividing the length of strips to determine an effective average flux in m^3 flux per metre vegetation boundary (i.e. a unit flux across the boundary). Results of this study (Fig. 7.16) are presented as percentage relative changes in fluxes across the vegetation boundaries, since this metric enables the direct comparison of fluxes across different vegetation boundaries.

To investigate how the redistribution of water and soil resources changes over the vegetation boundaries, a 5-min duration rainfall event with rainfall intensity of 109.7 mm h^{-1} (a storm with a 10-year return interval at the site (Wainwright 2005))

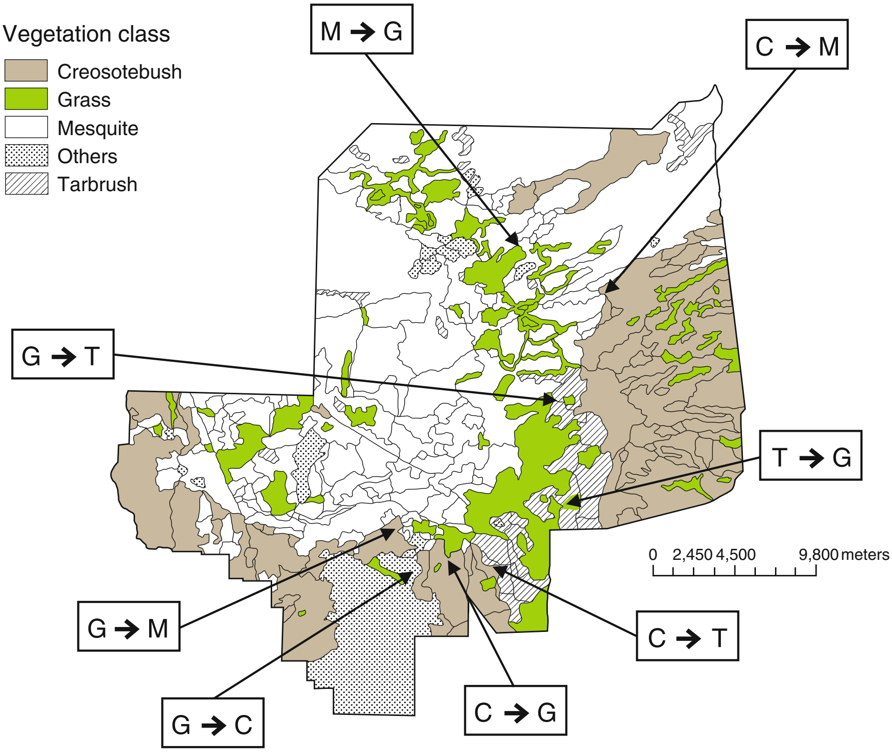

Fig. 7.14 Map showing the locations of vegetation-boundary scenarios within the Jornada Basin (map data provided by the Jornada Experimental Range Agricultural Research Service, US Department of Agriculture, Las Cruces, New Mexico). *G* is grassland, *M* is mesquite shrubland, *C* is creosotebush shrubland and *T* is tarbush shrubland (Reprinted from Mueller et al. (2007), 91–100, Copyright (2007), with permission from Elsevier)

was simulated for each of the vegetation boundaries. Simulation results show that at vegetation boundaries where shrubs are upslope of grasses, there is a substantial decrease in water flux once the boundary is crossed (Fig. 7.16), with most pronounced decreases for transitions from mesquite to grassland and from tarbush to grassland across the vegetation boundaries. Therefore, grasses that are downslope of shrublands are effectively able to capture and utilize water that is lost from upslope shrublands. These results are directly linked to those obtained from Ecohyd-HydroVeg simulations (Sect. 7.4), whereby more vegetation cover leads to less runoff, thus representing a critical feedback between runoff and erosion dynamics with vegetation growth. Changes in sediment flux over the vegetation boundaries show more complex behaviour, whereby for the shrubland to grassland vegetation boundaries, sediment flux is relatively constant over the shrubland, and increases for a short distance once the vegetation boundary is crossed because of a rise in the detachment rate because of an abrupt change in particle-size distribution

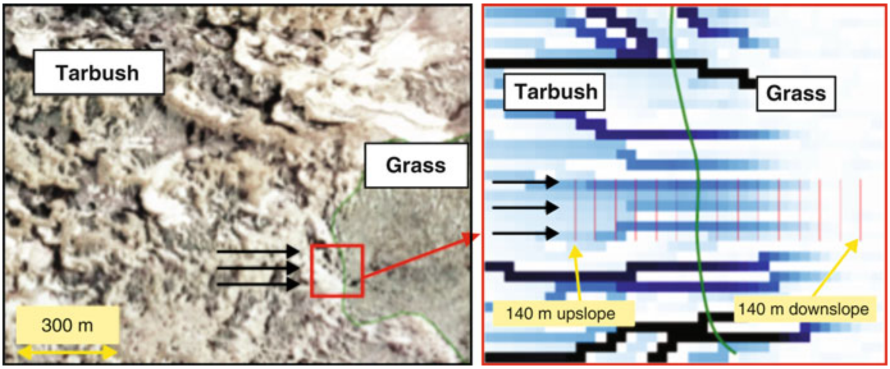

Fig. 7.15 Example of a Tarbush-Grass vegetation boundary over which water and soil resource redistribution was modelled (*left*). *Arrows* show the predominant flow direction. Close-up of the vegetation boundary (*right*) which is marked by the solid *grey line*, showing the total water flux over the boundary (modelled using MAHLERAN), and the area over which fluxes are modelled – extending from 140 m upslope of the vegetation boundary to 140 m downslope of the boundary. *Vertical lines* show the points along the transect over which fluxes were investigated (Reprinted from Mueller et al. 2007, 91–100, Copyright (2007), with permission from Elsevier)

(Mueller et al. 2007). Following this initial increase once the vegetation boundary is crossed, sediment flux declines, with grasslands retaining sediments eroded from upslope. For the grassland to shrubland vegetation boundaries, sediment flux is greatly elevated in the shrubland. The behaviour of nutrient fluxes across the vegetation boundaries is the opposite of water fluxes, with results showing an increase in nutrient fluxes moving downslope from shrubland to grassland (Mueller et al. 2007).

These modelling results suggest that the redistribution of water and nutrients during rainfall events could have great implications for the stability of vegetation boundaries. For example, a shrub to grass vegetation boundary may be stable when nutrient losses in runoff from grassland are in balance with nutrient replenishment rates by nutrient cycling in grasslands, along with the replenishment of nutrients in runoff from upslope shrublands (Mueller et al. 2007). However, a vegetation boundary may become unstable if grasses lose their ability to sequester and retain nutrients. For example, overgrazing may decrease soil infiltration rate due to soil compaction and increase the connectivity of bare areas, rendering the grassland more "leaky" with reduced ability to capture and retain resources from upslope shrubland, potentially leading to instability. These modelling results suggest that the development of islands of fertility are only one form of small-scale change associated with degradation, and that changes in connectivity are important. Thus, research efforts need to continue to focus on changes in connectivity across landscape scales, since these changes in connectivity and ecogeomorphic feedbacks associated with them are important potential driving mechanisms for catastrophic changes in these systems (Turnbull et al. 2008, 2012).

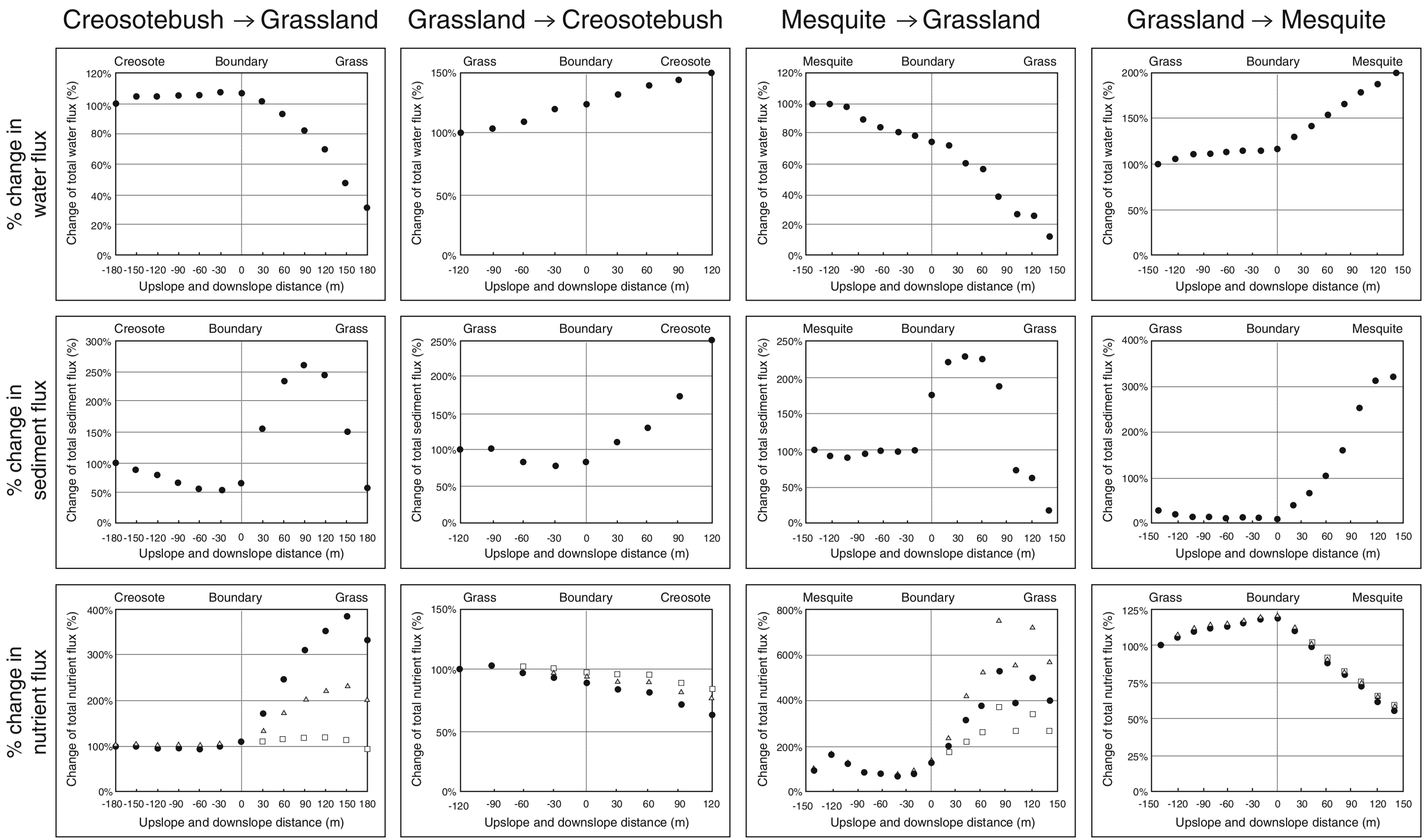

Fig. 7.16 Simulated water, sediment and nutrient fluxes across vegetation boundaries. In the *bottom row* of plots, Δ = ammonium, ■ = Nitrate, □ = Phosphorus (Reprinted from Mueller et al. 2007, 91–100, Copyright (2007), with permission from Elsevier)

7.7 Modelling the (Co-)evolution of Vegetated Aeolian Landscapes

Vegetation cover and its spatial distribution not only affect surface runoff and hydrologically mediated sediment transport, but also they affect aeolian processes. Aeolian processes play an important role in arid and semi-arid regions (discussed in Sect. 5.6). Nearly 20 % of drylands in marginal agriculture regions are underlain by aeolian sand deposits that are currently kept mostly dormant by various degrees of vegetation coverage (Thomas 1999), and it is feared that these dormant dune fields in semi-arid regions may become reactivated (Thomas et al. 2005), leading to increased soil erosion as well as dust emissions. Furthermore, regional degradation and conversion of grasslands to deteriorating shrublands is accelerating (Okin et al. 2006; Grover and Musick 1990), allowing for an increased activity in aeolian erosion, sediment transport and deposition on developing bare surfaces. This increase in aeolian erosion, sediment transport and deposition is often associated with self-organized redistribution of nutrients and sediments leading to a catastrophic shift in the ecosystem state (Scheffer et al. 2001), and the development of a spatial pattern of ‘islands of fertility’ in the form of nebkha dune fields (Barbier et al. 2006; Tengberg 1995; Wang et al. 2006). This self-organized co-evolution of shrub plant and dune landform results from positive feedbacks between plant growth and local sediment deposition – in the context of plant physiology, sediment controls, and climate.

The impact of vegetation on aeolian sand transport is primarily understood through its effects on near-surface airflow. The enhanced surface roughness of vegetated surfaces decreases the shear stress on the bed and increases the shear-velocity threshold required for sand transport as vegetation elements partially absorb the force of the wind (Lettau 1969). Furthermore, the surface area available for transport is physically reduced. These effects have been investigated on the scale of individual vegetation elements – shrubs and grass clumps (Gillies et al. 2000) – and over surfaces covered with varying degrees of vegetation density (Wolfe and Nickling 1993). Individual shrubs and/or clusters of surface plants act as sediment traps, inducing local deposition on an otherwise potentially deflating surface, and leading to the initiation and growth of shadow dunes (Hesp 1981). In dryland environments there is a great variety of plant species that induce nebkha development, including substantial woody shrubs, like *Artemisia* (sagebrush), *Prosopsis* (mesquite) and *Tamarix* (salt cedar), as well as fast-growing ground-hugging plants like *Arctotheca*, *Gazania*, *Zygophyllum*, *Ziziphus*, and *Acacia* (Fig. 7.17). Dense and vertically growing plant species, particularly the woody shrubs, pose a significant obstacle to the wind and consequently form high and relatively steep-sided nebkhas, while the ground-hugging plant species more often produce lower and more extended dome-shaped mounds (Hesp and McLachlan 2000). Nebkha shapes may be semi-circular or with an aerodynamic tail of sediment deposit if located in a unidirectional wind regime.

While it is possible to simulate the three-dimensional airflow dynamics and associated sediment transport and deposition processes on and around an individual

Fig. 7.17 Nebkhas on a deflation plane near Akhfenir, Southwest Morocco. Sand transport is predominantly from *left* toward the *right* (sand trapped inside and downwind of the plant clusters)

nebkha dune – using CFD applications for example – CA models are a potent alternative for simulating the complex and self-organizing feedback processes of initiation and evolution of nebkhas at a dunefield scale. The strength of such models is their capacity to identify and explore the key processes that underlie the complexity without involving the excessive number of coefficients, parameters, and assumptions of many detailed reductionist models.

The DECAL model (Baas and Nield 2007) is based on a CA algorithm for dune evolution developed by Werner (1995) that was adapted and extended to incorporate the effects of and feedbacks on vegetation in the aeolian environment (Baas 2002). The 3D model space consists of a grid over which discrete 'slabs' of sand are transported along a 'wind' direction between neighbouring cells. The self-organizing aggregation and migration of heaps of slabs is only limited by avalanching to maintain a maximum angle of repose and a 'shadow-zone' behind piles to mimic the forced deposition and no-erosion in the downwind wake of topography. The erosion and deposition of slabs is governed by local probabilities that are determined from the degree of vegetation cover on each cell, while the impact of the net sedimentation balance on the plants in turn is mimicked by annual growth and decline functions that increase or decrease the local vegetation coverage. The algorithm has been expanded to include the effects of multiple types of vegetation and it has proved highly successful in replicating realistic-looking parabolic dunes with trailing ridges and deflation planes as well as nebkha dunes

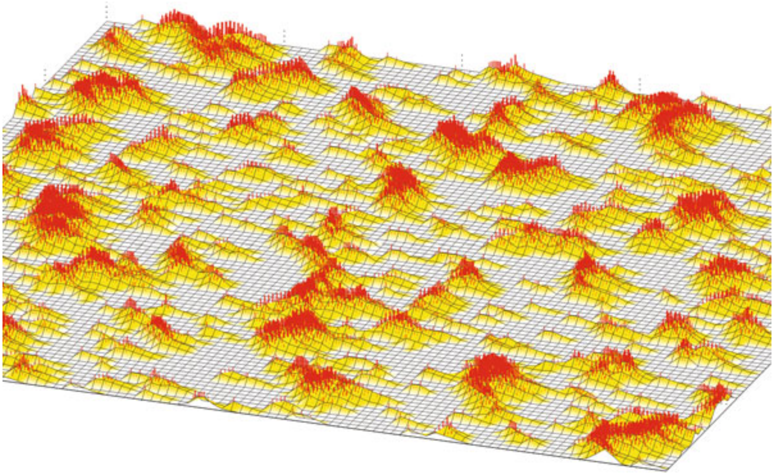

Fig. 7.18 DECAL simulation of nebkha dune field evolved from an initially flat surface under the influence of a mesquite-type shrub vegetation, showing classic aerodynamic tails. Density and size of *red* sticks indicate levels of shrub effectiveness. Sediment transport direction from *upper-left* to *lower-right* of view. Grid resolution is 1.0 m

with lee-side deposition tails (Fig. 7.18), under the influence of vegetation elements that mimic plant species such as marram grass and creeping willow in a costal environment, and mesquite or tamarisk shrubs in a semi-arid environment (Nield and Baas 2008a).

The DECAL model has been used to explore the potential evolution of dune landscapes in response to changes in sediment-transport conditions and vegetation vitality (e.g. through climate change) as well as external perturbations, such as wildfires and anthropogenic effects (Nield and Baas 2008b). Simulations of the overall change in vegetation cover as the landscape develops from a flat, barren surface, agree well with field observations of continental semi-arid dune fields in the Great Plains, Canada, as observed by Wolfe et al. (2000). Simulations over longer timescales show evolutionary sequences with thresholds, relaxation periods, and equilibration, and the model has revealed how the amplitude, frequency, and timing (relative to the evolutionary stage) of various perturbations has wildly differential effects on the resultant landscape response. Simulations of dryland environments with mesquite- or sagebrush-type vegetation, meanwhile, demonstrate the important control of initial sediment availability and supply on the subsequent size, shape and spatial distribution of nebkhas, and have also yielded more fundamental insights into the relationship between abiotic sediment-transport processes and biotic components in an ecogeomorphic system. The model reveals how the vegetation and its interactions appear to impress a characteristic scale on the dynamic system so that size and shape of vegetated dunes are fundamentally controlled by the ecological attributes of the plant species in the environment. Whereas bare sand dunes are found over several orders of magnitudes in size, dunes developing under the influence of vegetation may thus exhibit a clear restriction in size-range and shape that is fundamentally related to the biological limits of their physiological and photosynthetic potential (Baas 2007). The model is now being used to investigate rigorously the precise biotic controls on dunefield development in a quantitative framework, by linking vegetation parameters to potential system attractors and typical evolutionary trajectories determined from topographic and ecological metrics (Baas and Nield 2010).

7.8 Towards a Fully Integrated Model Framework to Simulate Feedbacks Between Biotic and Abiotic Ecosystem Structure and Function: Problems and Challenges

Having discussed sets of ecological and geomorphic processes individually in the preceding sections, we now return to the issue of linking ecological processes and geomorphic processes together more holistically. As suggested in Sects. 7.2 and 7.3 we can explore the development of patterns using deterministic and stochastic models of pattern formation; however, fully integrated ecogeomorphic models are required to explore the *processes* of pattern formation which is critical to understand land degradation in drylands. The models outlined in this chapter demonstrate that a variety of modelling tools is available to simulate pattern formation in drylands both deterministically and stochastically, and to simulate processes that lead to pattern formation in drylands. A practical limitation of the modelling approaches explored in this chapter, is that they are limited in their spatial and/or temporal extent – largely due to access to suitable computational resources, or the willingness of researchers to use them. However, the main limitation to developing fully integrated ecogeomorphic models is largely conceptual. Although great leaps have been made over recent years in understanding some of the linkages between ecological and geomorphic processes (discussed in Chaps. 4 and 5), there still remain fundamental gaps in our understanding of their interactions at different spatial and temporal scales. A critical challenge in conceptualizing the ecogeomorphic system rests in reconciling the level of process representation that is needed to simulate multi- and cross-scale feedbacks between ecological and geomorphic processes.

In terms of vegetation dynamics for example, plant-recovery time following a disturbance will be in part determined by plant phenology and rainfall seasonality. However, because of the annual time step in ECOTONE, such effects cannot be represented. Furthermore, not all disturbances impact all plants uniformly; for example, mesquite shrubs are commonly avoided both by foot, vehicle traffic, and grazers which may to an extent explain their dominance in disturbed areas, whereas grasses indigenous to the same environment are often disturbed by trampling and overgrazing. Therefore, the extent to which species-specific responses are represented in vegetation models could have a great effect on simulated vegetation dynamics.

In terms of hydrological and geomorphic processes, MAHLERAN is an event-based model that simulates runoff, erosion and nutrient dynamics during rainfall events, meaning that antecedent conditions such as soil-moisture content, soil-nutrient content and vegetation cover have to parameterized for each model run. The detailed datasets available for parameterizing antecedent conditions are rarely available, which thus highlights a great limitation of event-based modelling approaches.

The extensive expertise required to develop, parameterize and test the process-specific models explored in this chapter highlight the extent to which developing ecogeomorphic modelling tools is a major challenge, for both conceptual and technical reasons. A key issue facing the development of ecogeomorphic models, is whether or not the focus should be on making the most of existing resources by coupling existing models (legacy models) that each simulate an isolated component of the system, or if the focus should be on developing new integrated ecogeomorphic models that do not suffer from the constraints (conceptual and technical) imposed by utilizing existing models. There are many different ways in which models can be coupled, ranging from loose coupling to tight coupling (Brandmeyer and Karimi 2000). The different approaches to coupling models will have implications for simulating feedbacks across multiple spatial and temporal scales, and simulating emergent phenomena. Thus, the method used to develop an ecogeomorphic model needs to consider its desired purpose.

Loosely coupled models share a common interface, which controls data transfer between the coupled models. Advantages of loosely coupling models include: (i) it is a relatively inexpensive way of coupling models; (ii) models need not be written in the same code; (iii) individual models may be continually developed without hindering the interoperability of the coupled model; and (iv) models can be linked with few changes made to the existing code. Disadvantages of loosely coupled modelling approaches include: (i) data-conversion programs or subroutines may be required to insure data interoperability between models (for example, the spatial or temporal aggregation or disaggregation of data); and (ii) maintenance to the interface may be required when the data structure of a model is updated. In tightly coupled models, one model may be embedded inside another, or two or more (sub-) models may run in parallel. Advantages of tightly coupled modelling approaches include: (i) increased flexibility for dynamic feedbacks between model components to occur; (ii) reduced data redundancy; (iii) potential for common data storage. Disadvantages of tightly coupled approaches include: (i) the necessity for source-code modification; (ii) detailed understanding of each model; and (iii) ensuring compatibility between all common elements. Model coupling may also extend beyond tight coupling, to form a new “integrated model”, in which all model components are dynamically linked, model components have common data storage, and a single model language is used. The development of integrated models is probably the most costly form of modelling, since it usually necessitates coding from scratch, and requires a high level of expert knowledge of all processes represented within the model – hence the need for inter-/multi-disciplinarity to ensure the success of integrated models. The extent to which models need to be coupled is in part dependent on the speed and frequency of feedbacks between different processes that occur in the system to be modelled. If feedbacks occur relatively slowly, then loose coupling may suffice. However, if feedbacks occur relatively quickly, then tight coupling or integrated models are necessary.

The starting point for tightly coupled or integrated ecogeomorphic modelling must be the conceptualization of the system in question, whereby key processes and the spatial and temporal scales over which they operate are identified. A critical component of this conceptualization process is determining the elements that will link each model (for example, soil moisture, soil texture, soil-nutrient content, plant biomass). These common elements have to be compatible, for example, the ways in which water and plant biomass are represented. Furthermore, the representation of a process needs to be consistent between models that are being coupled – for example, two models might use different process descriptions to resolve soil-moisture dynamics, which might thus lead to conflicting outcomes for the same process. If models are to be meaningfully coupled or integrated, it is essential that differences in their spatial and temporal extents and scales are reconciled (Brandmeyer and Karimi 2000). Aggregation/disaggregation techniques may need to be used to reconcile differences in spatial and temporal scales, but doing so requires in-depth knowledge of how system properties and processes scale. The spatial and temporal domains of an ecogeomorphic model will depend largely on the purpose of modelling. Therefore, when conceptualizing the system, the spatial and temporal domains of the modelling study must be carefully considered. For instance, the important ecogeomorphic processes and their critical spatial and temporal scales when studying hillslope-scale processes at the timescale of rainfall-runoff events may be greatly different to those when studying the evolution of deserts over multi-decadal timescales. The coupling of two or more models could result in exceedingly high parameterization requirements, which may potentially limit the ease with which such models may be used in a meaningful way. Care therefore needs to be taken to ensure that parameterization requirements for each model component remain as low as possible. As multiple models are coupled, and as parameterization requirements increase, there is great potential for the propagation of uncertainty. Uncertainty is, to some extent, inherent in all modelling approaches, and is derived from multiple sources, such as uncertainty in process understanding, process representation and model parameterization. As models become more complicated – as is the case with coupled models – more uncertainty is introduced (Ascough et al. 2008), and the compound effects of multiple sources of uncertainty can be great (see also the discussion in Chap. 10). Identifying sources of uncertainty, and recognizing and quantifying its consequences within modelling-based studies, is a major challenge that needs to be addressed. A more detailed consideration of uncertainty is provided in Chap. 10.

Land degradation continues to occur at alarming rates in drylands, and our existing approaches to understand the effects of environmental drivers and human-induced disturbances on ecogeomorphic processes in drylands are inadequate. In order to simulate these processes in drylands, approaches used to develop ecogeomorphic models need to be as simple as possible, but no simpler, in order to expedite the process of model development, model parameterization, model testing, and be used with ease by a wide community. The abundance of models that have been developed over recent decades to simulate isolated components of the ecogeomorphic system represent an ideal starting point from which to develop coupled or integrated ecogeomorphic models. There are likely to be many advantages of pursuing tightly coupled approaches to ecogeomorphic modelling as opposed to loosely coupled or integrated approaches. Using tightly coupled modelling approaches enables feedbacks to be represented at appropriate spatial and temporal scales. Component models can be updated to reflect ongoing advances that are made in process understanding in individual disciplines (although maintenance and model testing will be required to ensure ongoing compatibility of models when components are updated). Because tightly coupled modelling approaches make use of available resources, their development may be much more rapid and more cost-effective than integrated models. A strong argument for tightly coupled modelling is that it enables evaluation of what goes wrong when it is applied outside the "comfort zone" of one's own discipline, or away from initial conceptualizations of space and time.

Ultimately, if patterns and processes are important in understanding land degradation in drylands, we can only gain limited understanding by using detailed models that do not, or cannot represent pattern, or by looking at models that look at patterns, but not the processes in operation. In developing ecogeomorphic models, either by coupling or integrating models, these different perspectives need to be brought together. Critically, the ecogeomorphic modelling and field experimentation need to be carried out in tandem, with field experimentation informing the conceptualization and development of ecogeomorphic models, and ecogeomorphic models servicing as a tool to benefit the design of a new generation of ecogeomorphic field experiments to help resolve the remaining unknowns of pattern-process linkages in drylands.

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Chapter 8
Characterizing Patterns

Pierre Couteron, Philip Hunke, Juan Bellot, Joan Estrany, Núria Martínez-Carreras, Eva Nora Mueller, Vasilios P. Papanastasis, Robert R. Parmenter, and John Wainwright

Abstract The characterization of pattern requires a solid foundation in spatial sampling, and the different direct and indirect means of obtaining relevant spatial data. The scale triplet of extent, spacing and support are used as a means for structuring the understanding of spatial data (and by extension, for looking at temporal

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P. Couteron
IRD/UMR AMAP, Botany and Computational Plant Architecture, 34398 Montpellier, France
e-mail: pierre.couteron@ird.fr

P. Hunke • E.N. Mueller
Institute of Earth and Environmental Science, University of Potsdam, 14476 Potsdam, Germany
e-mail: philip.hunke@uni-potsdam.de; eva.mueller@uni-potsdam.de

J. Bellot
Depto. Ecologia, Universidad de Alicante, 03080 Alicante, Spain
e-mail: juan.bellot@ua.es

J. Estrany
Department of Earth Sciences, University of the Balearic Islands, 07122 Palma de Mallorca, Spain
e-mail: joan.estrany@uib.cat

N. Martínez-Carreras
Département Environnement et Agro-biotechnologies, Centre de Recherche Public – Gabriel Lippmann, 4422 Belvaux, Luxembourg
e-mail: martinez@lippmann.lu

V.P. Papanastasis
Laboratory of Range Ecology, Aristotle University, 54006 Thessaloniki, Greece
e-mail: v.papan@for.auth.gr

R.R. Parmenter
Valles Caldera National Preserve, Jemez Springs, NM 87025, USA
e-mail: bparmenter@vallescaldera.gov

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

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patterns), and relevant sampling strategies are discussed in relation to each. Direct and indirect ways of collecting data on appropriate ecogeomorphic parameters – vegetation, water, soil and animals – are then discussed. The potential for collecting large extents of data using remote sensing is then considered. Emphasis is given throughout on the need to consider process as central to developing robust sampling methodologies.

8.1 Introduction

In this chapter, we investigate some of the methodologies that can be used to measure patterns resulting from relevant ecogeomorphic processes. Following an overview on the principles of spatial sampling, we discuss the ways in which vegetation, water, soil and animal patterns are measured. A variety of proxy methods is then evaluated for indirect characterization of conditions, which can be useful when direct measurements are too costly, access is difficult, or there are ethical issues (e.g. the need to carry out non-destructive methods). Finally, we consider the methods of remote sensing that can be used to evaluate spatial and temporal patterns over very large areas.

8.2 Basics of Spatial Sampling

For a full comprehension of ecogeomorphic systems, we need sampling strategies that enable us to quantify both the patterns of soil, water and vegetation resource distributions and the spatial processes that generate ecogeomorphic feedbacks, such as the driving forces of water and wind erosion or the impacts of free-moving grazers. Spatial sampling is used to determine a limited number of locations in geographic space either through field sampling or remote sensing imageries to extract information of a landscape setting that is subject to dependency and heterogeneity.

The purpose of spatial sampling in dryland settings is to understand the heterogeneity of soil, vegetation, water and climatic attributes by enabling the identification of regular or irregular pattern formations or by allowing the analysis of connectivity or fragmentation features in their spatial distributions. As the heterogeneity of drylands is often intrinsically high, and the development of heterogeneity has been used as an indicator of land degradation (see Chap. 3), capturing the heterogeneity with an appropriate sampling framework is critical to the study of ecogeomorphic processes in drylands, especially where land degradation is a key factor. Spatial data sets may be used to carry out geostatistical analysis so that spatial scales can be identified over which patterns remain constant or opposite: the scales at which significant changes in pattern and process can be detected or inferred (Turner et al. 1991). The results of spatial statistics can then be used for

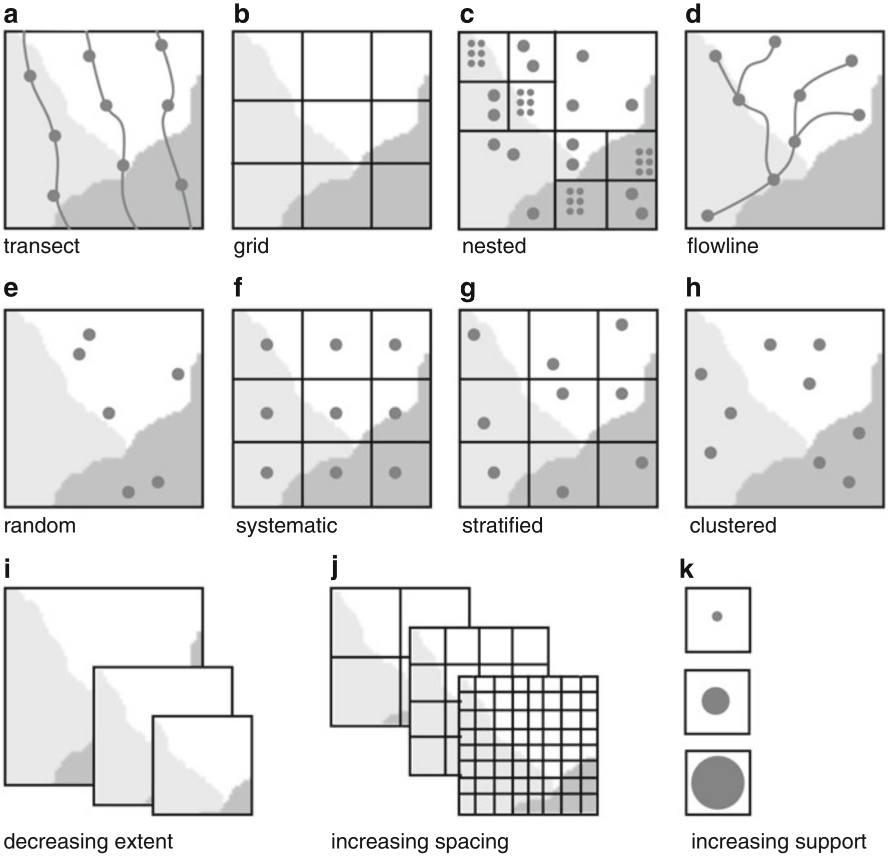

Fig. 8.1 Types of spatial sampling schemes and the effect of scaling. See text for further discussion of the different types

interpolation and extrapolation of the field data thus facilitating the continuous reproduction of the attribute surface. Finally, spatial sampling may be applied to collect information on the feedback dynamics and interactions of several landscape elements or compartments, such as the linkages between soil-moisture distribution and vegetation patterns or shrub encroachment and desert-pavement creation.

Sampling campaigns are typically carried out along parallel or perpendicular transects or within a grid structure (Fig. 8.1a, b). Multi-scale sampling schemes intrinsically take into account the variation of attributes over different spatial scales by using a nested approach (Fig. 8.1c). Flow-driven schemes position sampling locations along minor and/or major flow lines or outlets of sub-catchments, e.g. for the quantification of concentrated runoff, sediment and nutrient fluxes (Fig. 8.1d). In contrast, interrill sampling concentrates its efforts on areas which are not influenced by any concentrated transport processes. In environments where aeolian processes are significant, it would be more appropriate to sample along transects parallel to the dominant wind direction(s). In many cases, though, both wind and water are likely to be significant driving processes, so care needs to be taken with the orientation of specific sample layouts.

Sampling schemes include random, systematic, stratified, clustered sampling (and their combinations) as depicted in Fig. 8.1e–h. The quality and outcome of the analysis of the spatial field and remote sensing data directly depends on the adequate selection of a sampling scheme and the right choice in the size of the scale triplet of spacing, extent and support as defined by Blöschl and Sivapalan (1995). Spacing refers to the distance between individual samples; extent refers to the overall coverage of the study area; support refers to the integration volume or area, i.e. region of influence of a specific measurement or size of the sample and the type of measurement equipment (Fig. 8.1i–k). The term spacing is equivalent to the term resolution; for remotely sensed imageries, spacing and support are identical. For temporal scales, the space triplet can be equally employed as duration between data points of a time series (spacing), the sum of values over a specific time, e.g. daily rainfall (support, often the same as spacing) and length of the series (extent).

Poor composition of the scale triplet may lead to bias, distortion and outright errors of data interpretation: too large a spacing may prevent the detection of an underlying pattern, an extent that is too small may lead the study to disregard the presence and linkages of entire landscape components and too large a support may lead to an erroneous averaging over space. Situations may occur where for some reason no data can be obtained from a sampling unit, because a location in the field cannot be visited (e.g. due to land ownership or because hazards make it too dangerous) or when measurement is impossible due to the character of the attributes (e.g. sub-surface processes). The measurements of dryland attributes are highly limited by time and monetary constraints due to the often harsh, difficult-to-access terrain, which frequently results in a very small number of spatially distributed sampling locations (e.g. the sampling of runoff and matter fluxes at the outlet of only a small number of micro-catchments). For example, sampling of runoff and matter fluxes is normally carried out only at the outlet of a small number of micro-catchments. Similarly, climate data are frequently available from a single climate station per site (even on the meso-scale). It is easier to collect spatial field data on flora, fauna and soil properties as a snapshot sampling where simple field techniques allow collecting a large number of samples (>100) thus enabling pattern characterization using e.g. geostatistical or spectral analysis (see Chap. 9). Remote sensing techniques may help to overcome limitations in both spatial extent and spacing, as it is done for vegetation and land-use patterns. However, many variables cannot be measured neither by remote sensing nor ‘simple’ field techniques, e.g. infiltration rates in the upper unsaturated soil layers, which makes their spatial characterisation and prediction particularly challenging.

The largest source of uncertainty in the sampling of ecogeomorphic dynamics is the frequently rapid temporal changes of the system. Spatial sampling only provides a snapshot of prevailing processes and patterns and gives no information on sudden ecosystem changes triggered, e.g. through rare but critical events, such as the impacts of extensive wildfire or major erosion events due extreme storm events. The key element for representing extreme events is high-resolution process data over a long time (i.e. small temporal spacing and at the same time large extent of time series describing key driving forces of ecosystem change). Unfortunately, short and poorly spaced time series are common for dryland processes and thus a major limitation to many studies in this regard.

8.3 Direct Field Measurement

8.3.1 Vegetation

Field vegetation studies are based on direct (field measurements) and indirect measurements (aerial photograph, remote sensing) to detect its structural and functional characteristics, at two levels, plants and communities. For example, in the semi-arid Mediterranean region, Ventós, Spain (Fig. 8.2), the vegetation cover is a spatial mosaic characterized by patches with different sizes and species, which produce different patterns (Ramirez and Bellot 2009). The vegetation is composed of isolated *Pinus halepensis* trees, shrub patches of *Quercus coccifera* and *Pistacia lentiscus*, tussock grass such as *Stipa tenacissima*, and some other bushes and annual plants inserted in a wider matrix of bare and stony soil. The main species (*Q. coccifera*

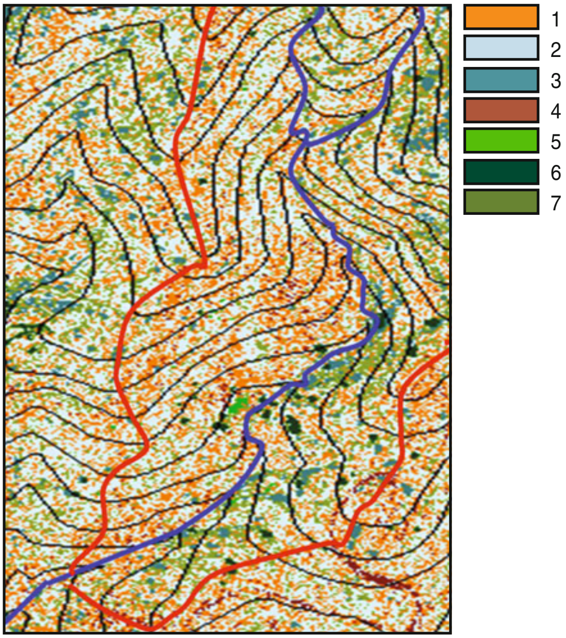

Fig. 8.2 Spatial distribution of plant-cover types in a semi-arid Mediterranean catchment (Ventós, Spain). Aerial colour photography classified as: *1* – Tussock grass, *2* – Bare soil, *3* – Oak shrub patches, *4* – Rock outcrops, *5* – Sub-shrubs (*Pistacia* sp., *Rhamnus* sp. or *Globularia* sp.), *6* – Trees (*Pinus halepensis*), and *7* – Chamaephytes vegetation (*Brachipodium* sp). In *red lines* the catchment limits, and *blue lines* the main runoff channel

and *Stipa tenacissima*) have sexual and vegetative reproduction which makes the isolation of individuals difficult. The same problem occurs elsewhere in drylands where species reproduce in both ways (e.g. *Larrea tridentata* and *Bouteloua eriopoda* in the North American deserts; *Balanites aegyptica* and *Isoberlinia doka* in Africa (Bellefontaine and Malagnoux 2008); *Eucalyptus camaldulensis* in Australia).

To characterize the vegetation pattern, the spacing of sampling should be adjusted to the most appropriate scale to capture the functional dynamics of the species. The sampling method of randomly distributed plots can be used in all cases. The appropriated size of the plots is 100 × 100 m for trees (*Q. coccifera* or *P. Halepensis* in the Ventós example), and 10 × 10 m for sub-shrubs and tussock grasses (in this case *Stipa tenacissima*), in order to capture in both cases the spatial interaction of each species. Although the minimum number of plots in each situation must be three, to allow minimal, acceptable statistical treatment, five plots are proposed as the basis for studying the pattern of the trees and shrubs, and 15 plots for tussock grass or plants of small size. In all cases, between 10 and 25 % of the study area should be sampled, distributed along an altitudinal and aspect gradient in the hillslope or watershed analyzed. Where all three types of plant are not present, this approach could be simplified. In each plot the most appropriate variables (biotic and abiotic) must be measured or estimated for the patterns that need to be characterized (Ramirez and Bellot 2009):

- **Plant cover**: each plot should be divided into four sub-units (25 × 25 m, and 5 × 5 m, respectively), and for each individual plant the maximum crown length (diameter 1) and the perpendicular line to it (diameter 2) are measured. Each individual plant-cover area is approximated using the area of an ellipse using the two measured diameters.
- **Plot leaf-area index**: An optical sensor can be used to estimate the plot leaf-area index (LAI). Eight LAI measurements were made using two optical aperture degrees (90 and 180 °) according to the sensor position in the plot.
- **Species and vertical structure**: On each plot, >100 points should be randomly or systematically sampled, and at each sample point the vegetation height is assessed at nine height levels (every 0.2 m from the ground). A graduated stick (1.8-m length) is placed vertically and the number of vegetation contacts at each height level is registered.
- **Patch-size groups (PSG)**: In the Ventós example, *Q. coccifera* patches were classified into five size groups according to average cover (C), so that PSG 0 represents $C < 0.5$ m, (1) $0.5\ \text{m} \leq C < 1.0$ m, (2) $1.0\ \text{m} \leq C < 2.0$ m, (3) $2.0\ \text{m} \leq C \leq 4.0$ m and (4) $4.0\ \text{m} < C$. The number of plants in each PSG was impossible to assess, due the reproductive characteristics of this species. In general, the classification should be modified according to the characteristics of the local species.
- **Tussock-size groups (TSG)**: Again, a classification should be carried out according to the species characteristics. At Ventós, *S. tenacissima* is the main

tussock species, and it was classified into five size groups according to average diameter (Ø), so that TSG 0 represents Ø < 0.15 m, (1) 0.15 m ≤ Ø < 0.30 m, (2) 0.30 m ≤ Ø < 0.60 m, (3) 0.60 m ≤ Ø ≤ 1.20 m and (4) 1.20 m < Ø. In this case, the relative dead foliar cover was estimated (RDFC) as an average fraction of total biomass, which increases with the tussock age.

- **Topographical variables**: The altitude should be measured, as well as the aspect in azimuth (using a compass) and the slope using a symmetrical scale (from –90 ° to +90 °, for example using a clinometers, Abney level, or more detailed survey techniques as appropriate).
- **Rock cover**: Gravel and rock cover should be systematically sampled at >100 sample points, by dividing the plot into 1 × 1 m squares, and counting the presence of rocks or gravel at each vertex (Wolman counts).
- **Soil depth**: The plot should be divided into 2 × 2 m squares. At each vertex of the grid (36 sample points) the soil depth was estimated by driving a steel stake (0.5-m length) into the ground and measuring the length of remaining stake. This approach typically produces an underestimate of soil depth, as the stake is often impeded by stone fragments. For deeper soils, indirect methods such as ground-penetrating radar may be more appropriate.
- **Analysis of the biomass-density relationship**: For each species, allometric equations fitted by specific studies are used. For each species, the number of plants and stem diameters are combined to estimate the biomass, and total biomass calculated by summing each species biomass. Further analyses can then be carried out of the spatial pattern of the biomass and its relationship with the other spatial and non-spatial variables characterized on the site.

8.3.2 Water

Water fluxes in dryland ecosystems comprise precipitation, evapotranspiration, runoff (either in the form of rill or stream flow or as unconcentrated overland flow) and moisture dynamics in the upper soil layers. Transport processes via water such as suspended sediment or bedload transport and nutrient transfer and depletion through the action of overland flow are driven by the topography and soil properties. The following hydrological variables are typically measured to characterise the water budgets of a watershed (although in some drylands the definition of a watershed itself may be problematic):

Precipitation is measured with a standard rain gauge, a weighing rain gauge or a tipping bucket type. The standard rain gauge is essentially a can that fills up with water until it is measured and emptied manually (normally every 24 h). The weighing rain gauge weighs water as it falls and records that information continuously on a chart. A tipping bucket electronically records precipitation by filling two tiny buckets that are balanced like a see-saw. Commercially available devices can typically hold 0.25 mm (0.01 in.) of water, where each tip of the buckets causes the device to record an increase of 0.25 mm of rainfall, although it is possible to use finer or coarser bucket sizes. Too large a bucket volume can cause the method to underestimate the number of small events, while too small a volume can cause an underestimation of higher intensities as the bucket “bounces” between tips. The spacing of rain gauges is normally insufficient (typically one rain gauge for an area of hundreds of square kilometres) which is problematic for describing the rainfall regime in dryland systems which are frequently characterised by high-intensity storm events of very small spatial extent of less than one square kilometres (see Renard and Keppel 1965; Wainwright 2005 for discussions of dryland storm patterns with high spatial spacing). Uncertainties of measurement include shadowing effects of the monitoring site (e.g. influence by buildings or vegetation), the effects of wind which typically leads to an underestimation of the rainfall amount and evaporation from the collector can (Chow et al. 1988). More recent advances of ground-based measurement include laser-based devices called disdrometers, which count the individual drops breaking a laser beam as they pass through, which are calibrated to give intensity and type of rainfall. These devices remain expensive so are little used in an operational sense. Remote sensing of rainfall has increasingly been used to characterize patterns in drylands. Ground-based radar can give a moderate spatial and temporal spacing (e.g. kilometre-scale at 15-min intervals), while satellite-based techniques are used for example as part of the Famine Early-Warning System (FEWS) in Africa (at daily or decadal scales over 8 km).

Direct measurements of **evaporation** are not practicable in the field. The water loss from a standard saturated surface may be measured with evaporimeters (e.g. in the form of a pan filled with water) to get an indication of the potential evaporation from natural water surfaces. Adverse effects on the measurement include the disturbing effects of heavy rain and winds, the contamination of the water surface and uncontrolled changes of the water level not due to evaporation. For the measurement of actual evapotranspiration, lysimeters may be used. A lysimeter consists of a soil-filled inner container and retaining walls or an outer container, as well as special weighing devices for measuring percolation and changes in the soil-moisture content of the container. The rate of evapotranspiration may be estimated from the water budget equation: hence evapotranspiration equals precipitation minus percolation minus change in water storage. The costs of lysimeters and their running are very high and their application in dryland setting uncommon (Chow et al. 1988).

Soil moisture is measured directly by collecting a known volume of moist soil material and estimating the weight loss after oven drying for 24 h with a temperature above 100 °C. It is a simple, but invasive approach which means that it is not possible to repeat the measurement at the same site. For *in situ* measurements, soil moisture can be derived using various radiological techniques, such as neutron scattering and gamma absorption or by using the relationship between dielectric properties of soil and their moisture content (e.g. time-domain reflectometry). Theta probes (Delta-T Devices, Cambridge UK) are often used for repeated or continuous soil moisture measurements as it is easy to use and inexpensive. The theta probe generates a 100-MHz sinusoidal signal and measures the impedance of the sampling volume which is roughly a cylinder 4 cm in diameter and 6 cm long surrounding the centre prong of the probe (Kaleita et al. 2005). Only a small volume of soil is ever evaluated, and soil contact is therefore critical. As a result, this method is excellent for laboratory or point measurements, but is likely to be subject to spatial variability problems if used on a field scale (Dirksen 1999). Soil moisture may be measured at different depths to obtain information on the influence of root uptake on water availability in different layers.

Runoff is measured along flowlines, at the outlet of watersheds or along the hillslope where runoff occurs as overland flow. In dryland settings, runoff in rills is frequently measured with a Parshall flume, a purpose-built weir where water level is recorded continuously and can be related to discharge through its hydraulic dimensions (Hudson 1993). Due to the ephemeral occurrence of runoff in dryland rills and streams, the logging device for the water level often behaves poorly as it does not start logging once runoff commences after a long period of no-flow conditions. Continuous runoff measurements in dryland settings are rare and often only available for a small number of study catchments or for selected cross-sections of larger rivers from the official gauging stations of the regional environmental protection agencies. Although the recording equipment is not very cost-intensive, the lack of on-site maintenance normally limits data availability.

Direct measurements of **overland flow** over hillslopes are normally not carried out due to lack of monitoring methods. The only runoff data available for overland flow are from plot-scale rainfall simulation experiments. The purpose of a rainfall simulator is to deliver rainfall to the soil surface in a controlled manner with realistic simulation of rainfall intensity and drop-size distributions. In a sealed-off plot area, hydraulic properties of the overland flow can then be measured such as depth of flow, overall discharge at the outlet of the plot, and estimates for the friction factor (Parsons and Abrahams 1992).

Sediments in runoff and dissolved or particulate **nutrients** are sampled during runoff events either by taking water samples manually or by using a pump sampler. An ISCO sampler is a liquid sampler that automatically takes up to 24 water samples over a specified sampling period. Both types of water samples need to be analysed in the laboratory using standard chemistry methods. Quantitative data on suspended sediments and nutrients are very rare for dryland ecosystems due to the infrequent sampling periods of the ephemeral runoff regime.

Table 8.1 summarizes the introduced sampling methods for water availability and transport processes and gives an intuitive indication of their spatial and temporal spacing (according to the scale triplet discussed in Sect. 8.2). Data of all water-related processes in dryland settings normally exhibit large spatial spacing, i.e. there are only very few sampling points available to (in)adequately quantify the processes in a very heterogeneous environment.

Table 8.1 Direct field measurements of water availability and transport processes in dryland settings

Element	Instrument	Spatial spacing	Temporal spacing	Limiting factor
Precipitation	Standard rain gauge	Large	Large (24 h)	Labour intensive
	Weighing rain gauge	Large	Small (1 s)	
	Tipping bucket	Large	Small (1 s)	
	Distrometer	Large	Small (1 kHz)	Expensive
	Remote sensing	Large	Large (days, weeks, years)	Data-processing costs
Evaporation	Evaporimeter	Large	Large (24 h)	Only actual e
	Lysimeter	Very large	Small (1 min)	Very costly
Soil moisture	Soil sample	Small	Non-continuous	–
	Theta probe, Time-domain reflectrometry, Neutron probe	Small to large	Small (10 min)	
Runoff	Parshall flume	Large	Small (1 s)	Error-prone
	Rainfall simulator	Large	Small (1 s)	One event
Sediments, nutrients	Manual sample	Large	Non-continuous	Costly, maintenance and labour intensive
	Pump sampler	Large	Medium (1–30 min)	

8.3.3 Soil Properties

Water availability, vulnerability to soil erosion, soil fertility loss and nutrient depletion are a function of the physical, biological and chemical properties of the soil. Soil sampling normally involves the development of a spatial sampling scheme (as was specified in Fig. 8.1), the collection of multiple soil samples and the subsequent analysis of soil samples in a laboratory. In contrast to field measurements of water, soil-sampling schemes normally cover a large spatial extent enabling the quantification of spatial variability and thus the heterogeneous nature of dryland ecosystems, but exhibit a very poor temporal spacing, as soil sampling is normally carried out just once or repeated only a small number of times.

Soil properties in drylands strongly depend on surface cover (vegetated or bare), land-use, disturbance regimes (e.g. trampling by livestock) and position along the hillslope and flowpaths (rill, stream, inter-rill areas). The sampling protocol in the field should thus always comprise a detailed description of the sample location and its direct surrounding regarding surface conditions which then might help to identify spatial patterns of soil.

Standard soil analysis includes the estimation of particle size distribution, organic content, bulk density, porosity, pH value, nutrient content (e.g. nitrogen and phosphorus), soil moisture (see also previous section), and soil-aggregate stability (Herrick et al. 2001), soil organic and total carbon and soluble ions (for standard soil laboratory methods see Carter and Gregorich 1993).

Soil texture, especially, plays a critical role for runoff generation, moisture capacities and sediment transport as it has a large influence on infiltration rates and the erodibility of the soil (Wainwright and Bracken 2011).

Estimations of **soil-infiltration rates** are normally carried out directly in the field, as it is rarely possible to extract an intact soil profile for a lab analysis. Simple field techniques are available for the measurement of the ponded infiltration rate using a single ring-infiltrometer, which can be carried out for larger numbers of locations in the field (Herrick et al. 2005): a ring is pressed about three centimetres into the soil and water is provided by a Mariotte siphon; the water level in the bottle is measured until the infiltration rate becomes constant. Main measurement problems occur as soon as soils show high clay or gravel contents or soil crusting, which is a typical phenomenon in arid environments. Ponded infiltration rates from ring-infiltrometers tend to be higher than the actual final infiltration rates of natural events. Several reasons lead to the overestimation of the final rates when single-ring infiltrometers are used: (a) ponded infiltration rate is not limited by rainfall amount and intensity, i.e. water is supplied to infiltrate at maximum rate; (b) the soil surface often gets disturbed when the ring is inserted into the ground, especially if a crust exists and infiltration is sped up due to the development of fissures and cracks; (c) due to high-pressure heads the water is forced into the soil resulting in an artificially increased infiltration rate; (d) lateral diffusion occurs into the dry soil surrounding the ring resulting in an overestimation of the infiltration rate (see Wainwright 1996, for a comparison of infiltration rates obtained using different methods on the same soil). To overcome the limitations of ring-infiltrometers and for the quantitative description of the final infiltration rate, i.e. the saturated hydraulic conductivity, infiltration rates obtained through rainfall simulations for individual plots (of a size of ca. 1–4 square metres, in rare cases up to several hundred square metres, Parsons et al. 1996) are used which avoid disturbing the soil surface and include agglomerations of vegetation (e.g. grass patches or individual shrubs). At the same time, rainfall experiments are considerably more time- and labour-intensive and are normally carried out in very small numbers, thus hardly enabling the characterisation of spatial variations on a hillslope or catchment. Although estimates on the infiltration process play a key role for any mathematical description of the water budgets and the inherent water deficit of drylands, there is currently a significant lack of quantitative data on infiltration rates for these ecosystems.

A combination of several soil properties establishes the vulnerability of the soil to soil fertility loss and nutrient depletion. **Soil-aggregate stability** is a measure of the vulnerability of soil aggregates to externally imposed destructive forces such as soil erosion by water or wind (Hillel 2003). Soil-aggregate stability is most commonly assessed by various standardized wet-sieving laboratory methods, where soil samples are immersed and sieved in water. The degree of aggregate stability is normally given as the fraction of the mass of the original sample that withstands destruction after the wet sieving. In general, the greater the percentage of stable aggregates, the less erodible the soil will be. Herrick et al. (2001) developed a soil aggregate stability kit with which it is possible to rate soil samples to six stability classes by sieving them in water for rangeland conditions. It classifies the aggregate stability on a scale from zero (no stability) to six (high stability). The kit is a combination of a slake test and a stability test and is designed to evaluate soil quality in drylands directly in the field which is less cost and labour intensive than other laboratory-based methods.

8.3.4 Grazers

Grazers commonly using drylands are sheep, goats and cattle. Their spatial distribution depends on several factors including forage availability, topography and water distribution, which determine the grazing pattern of each animal species. In general, all these animals tend to visit the most productive sites in terms of forage, avoid steep areas and concentrate near the watering points, but there are important differences between them. Sheep and goats, for example, can make better use of a rugged terrain, since they are of smaller size, have greater agility and a stronger climbing instinct than cattle; also, the former animals travel longer distances away from watering points than the latter (Holechek et al. 2004). These grazing patterns become accentuated when livestock are fenced or herded instead of roaming freely.

Since domestic animals are owned by people living in drylands, the easiest way to measure their numbers is to ask their owners. This approach is most straightforward in privately owned rangelands. Care must be taken in interpreting results obtained in this way, as there may be deliberate overestimation (for reasons of status) or underestimation (e.g. if respondents are suspicious about being taxed on the information provided). In the communally owned rangelands however the situation is more difficult because the same area may be grazed by more than one livestock owner. Therefore, all the farmers involved in the communal grazing must be identified and asked for the number of their animals. The same numbers can be found in the archives of the local community if the permission for grazing is given to the farmers by the local authorities.

Counting the number of livestock using a particular rangeland is not enough to evaluate the grazing impact. They have to be converted to livestock units. This is because the various animal species have different feed requirements which are a function of their bodyweight (Holechek et al. 2004). The Society for Range Management (Bedell 1998) uses a mature cow (dry or with a calf) of about 450 kg bodyweight as the animal unit (AU) that requires 12 kg day^{-1} of forage. Sheep and goats are converted to AUs by using the rates of 0.2 and 0.17, respectively (Heady and Child 1994). An adult but dairy cow weighing 600 kg is also used as a livestock unit (LSU) by the European Union (Eurostat 2010). In Australia, on the contrary, the dry sheep equivalent (DSE) is preferred as a standard livestock unit (McLaren 1997) as is also the case in the Mediterranean (Le Houérou 1981).

The grazing impact on a dryland can be expressed with the stocking rate which denotes the number of animals or animal units per unit of land over a specific time period, usually a year. Stocking rate refers to the actual number of animals grazing a particular rangeland. On the contrary, grazing capacity expresses the maximum number of animals that can graze a rangeland without deterioration of its productivity. This maximum number depends on the potential forage production, namely the forage produced by the rangeland when it is not degraded. Such forage is measured or estimated based on the grazing pattern that each animal follows suggesting that non-productive, steep and remote areas may be excluded from the rangeland area actually used. Once the grazing capacity is known then the stocking rate is used to denote overgrazing, undergrazing or proper grazing depending on whether it is higher, lower or equal to the grazing capacity. It should be noted that the high level of variability in dryland conditions means that the grazing capacity is effectively a dynamic rather than a static characteristic, and this variability can severely limit its practical use. If wild animals are also grazing drylands, alone or together with domestic animals, then more sophisticated methods should be used for their measurement, as discussed in the following section.

8.3.5 Other Animals

Many species of animals play important roles in the ecology of rangelands, influencing the spatial distributions of plants, the structure of soils, and a large suite of ecosystem processes (energy transfers, nutrient cycles, predator-prey interactions, etc.). The activities of native and non-native animals may, under some conditions, accelerate rangeland degradation, or alternatively be utilized to facilitate restoration and sustainability. Examples of common groups of animals that function as important drivers of rangeland condition include herbivorous insects (grasshoppers and locusts, beetles [e.g., weevils, scarabaeids], aphids and moth larvae [caterpillars]) and herbivorous vertebrates (rodents, rabbits), granivores (harvester ants, bruchid beetles, birds, rodents), decomposer invertebrates (termites, dung beetles, flies, isopods), pollinators (butterflies, moths, bees, flies, beetles, birds, bats), soil-disturbing vertebrates and invertebrates (burrowing species), and predators (spiders, scorpions, wasps, beetles, hawks, owls, canids and felids).

Measuring the population abundances and spatial distributions of these species is considerably more difficult than counting livestock. Unlike domesticated animals, which usually can be censused (i.e. enumerated completely), small-bodied wild animals must be sampled in the field, and the resulting sample data used in statistical models to estimate population size or density (number per hectare). The program MARK is a very commonly-used statistical package for estimating abundances and densities (White and Burnham 1999). Field sampling may involve direct observational data collection, such as visual line transects or variable circular plots for birds or mid-sized diurnal mammals, or counts within quadrats, rings or enclosure bags for such species as grasshoppers and shrub-dwelling insects

(Onsager and Henry 1978; Sanchez and Parmenter 2002). Other techniques involved capturing animals, marking them, releasing them at their point of capture, and recapturing them at some point in the future. Such data provide information for abundance or density estimates (if the "effective trapping area" is known), as well as some index to movements and home range. Radio-telemetry technology is often used to track animals with their home ranges or territories, as well as during long-distance movements during dispersal or migrations.

Just as with livestock, the impacts of animals on rangeland resources can be "standardized" across species. For livestock, an AUM (see Sect. 8.3.4 above) relates the amount of dry plant material consumed per head of livestock; for other herbivores, including insects, one can calculate a value of "consuming biomass" (Parmenter et al. 1984), which relates the amount of plant material consumed to the animal's population size or density, its metabolic energetic demand, and the ambient temperature. Consuming biomass can then be compared or summed across taxa to identify community impacts of herbivorous or granivorous animals on rangeland vegetation.

8.3.6 Faunal Impacts on Rangeland Vegetation Patterns

The effects of animals on rangeland vegetation patterns and distributions are derived through physical disturbance (digging of mounds and burrows, hoof action and trampling), trophic interactions (seed predation and dispersal, and grazing and browsing), and chemical changes to soils (deposition of faeces and urine, and carrion decomposition processes). These faunal influences may affect range vegetation directly or indirectly, and may take place over short or long time-frames. For example, an herbivorous species grazing on high-preference grass species will directly influence the cover and productivity of the selected forage species, but be reducing the competitive capacity of those species, may indirectly facilitate an increase in low-preference plant species (for example, invasive weed species). Occasional or episodic high-intensity grazing by herbivores may reduce particular plant species in the short term, but sustained heavy grazing over the long-term causes cumulative effects from which vegetation recovery becomes more difficult; the shift from desert grasslands to woody desert scrub in the American Southwest in the nineteenth and twentieth centuries is an excellent example of long-term cumulative effects on arid rangelands.

Vegetation patterns in arid and semi-arid rangelands are often influenced by the physical activities of animals in soils. The construction of burrow systems by insects and small or medium sized mammals has clear impacts on the distribution of plants, soil seed reserves, and soil physical properties. For example, harvester ant colonies (Pogonomyrmex spp.) in North American rangelands create patches of bare ground, surrounded by high densities of particular grass species; similarly, desert grassland rodents build mounds and burrow systems of various sizes, which are often dominated by annual forbs (Fig. 8.3). These colonies and mounds can

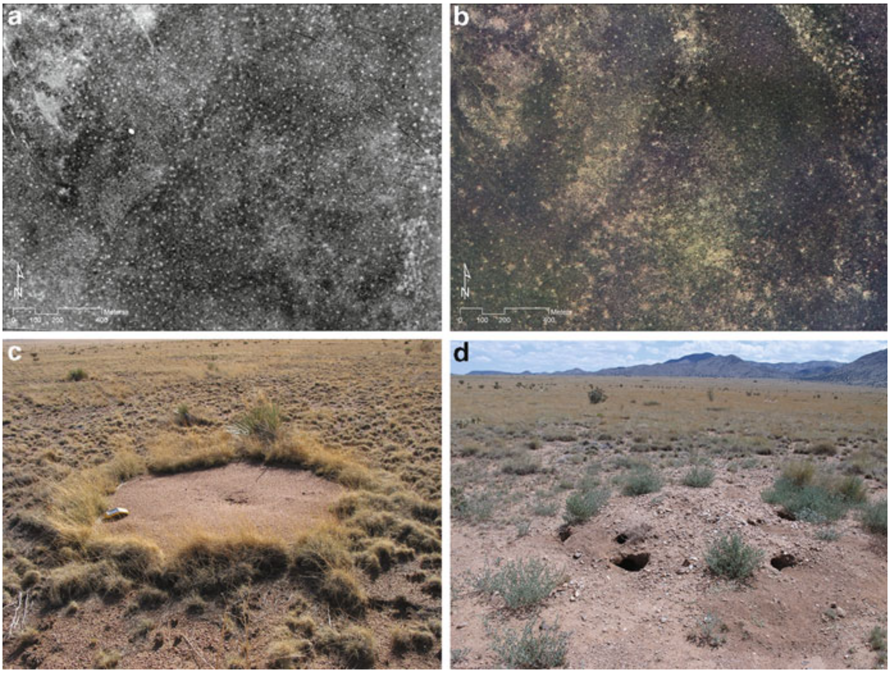

Fig. 8.3 Landscape-level rangeland vegetation spatial patterns can be driven by animal activities. In the example above, repeated aerial photographs of a desert grassland site illustrate the vegetation pattern effects of harvester ant (Formicidae: *Pogonomymex rugosus*) colonies and banner-tailed kangaroo rat (Heteromyidae: *Dipodomys spectabilis*) burrows on grass/forb distributions. The (**a**) aerial photo, taken in 1935, shows hundreds of ant colonies and rodent mounds across a grassland area of approximately 4 ha. The (**b**) photo is of the same site in 2006 (76 years later) and shows many of the same colonies/burrows, indicating the long-term persistence of these faunal landscape features. (**c**) Close-up photo of a harvester ant mound (GPS unit on left for scale). (**d**) Photo of banner-tailed kangaroo rat mound and burrow entrances (All photos from the Sevilleta National Wildlife Refuge, New Mexico, USA)

reach surprisingly high densities, and are long-lasting features on the landscape; ant colonies can survive for decades, and rodent burrows are often occupied by succeeding generations of animals for many years. Other direct physical effects include hoof action of large mammals, which can trample vegetation, and both till surface soil and litter layers, as well as compact soils. The scale of these latter effects varies with the body size of the animals involved, their numbers, and the frequency with which they re-occupy specific sites.

Herbivorous animals (wild species as well as domesticated) influence vegetation patterns via direct feeding behaviours and indirect cumulative effects (as described above). At high densities, herbivores can rapidly impact range vegetation in terms of species composition, basal and canopy cover, vegetation height (architecture), and productivity. Differences in herbivore densities in local areas can create obvious disparities in vegetation, such as "fence line" contrasts on many rangelands

Fig. 8.4 High stocking rates of livestock in confined range pastures can alter vegetation cover and plant architecture patterns, creating stark "fenceline contrasts" on the landscape. Illustrated above is a Ponderosa pine grassland in New Mexico, USA: the *left* side of the fence (Valles Caldera National Preserve land) is lightly grazed by cattle and elk, whereas the *right* side (U.S. Forest Service land) has much higher stocking densities of cattle. Dominant bunchgrass is Thurber's fescue (*Festuca thurberi*)

throughout the world (Fig. 8.4). In some arid ecosystems, vegetation communities can recover if the herbivore pressure is removed or reduced (Ryerson and Parmenter 2001), although this may take many years (Fig. 8.5). In other ecosystems, range restoration has proven extremely difficult, usually when long-lived woody plant species have become established as dominants, pre-empting establishment of soil structure and recolonizing herbaceous vegetation.

Animals also influence plant distributions through seed predation and dispersal. Many species of rodents collect and feed on the fruits and seeds of rangeland plants; these seeds are often transported away from parent plants, and cached in the soil or burrow systems for later use (Van der Wall 1990). Many seed locations are forgotten about, or the collecting individual rodent dies, and the seeds are able to germinate; clusters of young seedling plants are indicative of this phenomenon. Birds also feed on fruits and seeds, and can transport them great distances before seeds are defecated or regurgitated, again allowing for potential germination. Animals also disperse seeds that have become entangled in their fur or feathers, or encased in mud on animal appendages.

Finally, animals influence plant patterns via deposition of faeces and urine, as well as their own carcasses when they die. Faeces may contain undigested seeds, which can germinate and grow in a highly nutritious microenvironment. Faeces

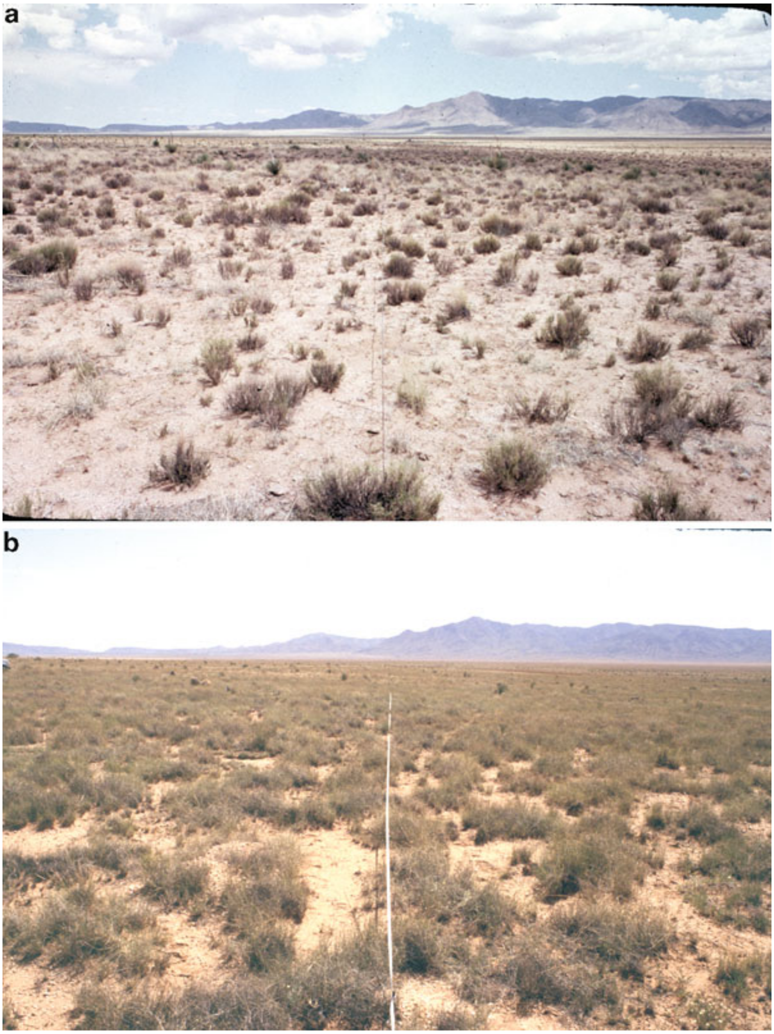

Fig. 8.5 Arid rangelands can sometimes recover from animal-induced conditions. The repeat photos above show a permanently-marked BLM range trend plot in a black grama grassland (*Bouteloua eriopoda*) in 1976 (*top*) following 150 years of heavy grazing by sheep, cattle and prairie dogs (*Cynomys gunnisoni*). Prairie dogs were removed (poisoned) in 1971–1972. The land was donated to the U.S. Fish and Wildlife Service shortly afterwards (1973), and all livestock were removed by 1975; hence, the 1976 photo illustrates the range condition at the time that major herbivores were removed. By 1996 (20 years later), the grassland vegetation had changed from dominance by annual forbs and snakeweed (*Gutierrezia sarothrae*) to a grass-dominated flora (Photos from the Sevilleta National Wildlife Refuge, New Mexico, USA (see additional details in Ryerson and Parmenter 2001))

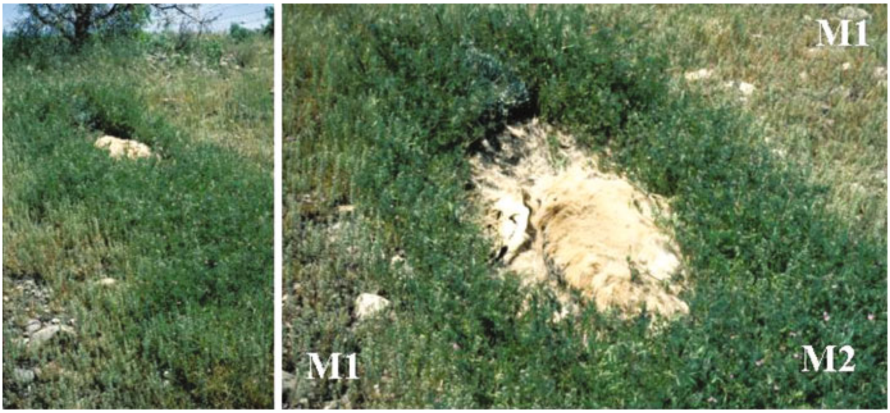

Fig. 8.6 Animal carcasses produce "islands of fertility" and influence rangeland plant productivity and distributions via the decomposition process, which includes mass transfers of organic nutrients, salts and minerals from the carcass into the soil. In the photo above, a decomposing mule deer carcass (*Odocoileus hemionus*) has a clear influence on surrounding vegetation, particularly downslope where erosional movement of nutrient-rich carrion fluids and particles have been transported. The close-up photo on the *right* shows nutrient/moisture boundary area in vegetation, with microsites outside (M1) and inside (M2) the moisture- and nutrient-enriched soil zone (See Parmenter and MacMahon (2009) and references therein for more details)

may also physically smother plants beneath them, creating bare patches of soil surrounded by otherwise healthy vegetation. Urine chemistry can have varying effects, with high concentrations and quantities killing plants directly during deposition, whereas adjacent vegetation can flourish as their roots receive a "fertilizer" effect. Animal carrion can create similar patchiness in vegetation (Fig. 8.6), with plants dying if directly under the carcass, but increasing in productivity, cover and height if sufficiently nearby to capture the additional nutrient pulse in the soil (Parmenter and MacMahon 2009).

8.4 Indirect/Proxy Measurement Techniques

8.4.1 Catchment Processes

Indirect and proxy measurement use proportions to estimate a variable when direct measurement is not possible, or record some feature that is closely related to the variable of interest. For example, pressure transducers are used for the measuring water depth, electrical resistance for temperature measurements, and signal travel time for Time Domain Reflectometry (TDR). A rating curve is then used to convert the measured variables to the variable of interest (e.g. relationship between dielectric constant and volumetric moisture content for TDR measurements). Measurements are then normally made available with a higher temporal and/or spatial resolution. Nevertheless, the conversion introduces additional measurement error, not only because the surrogate variables might have a limited degree of correlation to the variable of interest, but also because the variable of interest might not respond equally across its full measuring range (response deviates from a straight line). As a consequence, it is the greatest concern associated with using proxy or indirect measurement techniques to make sure that the measure is both valid and reliable.

In the same way, indirect/proxy measurements are used to study spatial patterns (i.e. higher number of measured points are normally needed), because it is often easier to collect spatial data than by direct field measurements (e.g. remote sensing techniques used for the estimation of land surface temperature, evaporation or near-surface soil moisture; see Sect. 8.4). However, it is often the case that as easier is the method of data collection, the larger is the measurement error (Blöschl and Grayson 2000) and compromises between both are required. More details about spatial observations and interpolations in catchment hydrology are given in Blöschl and Grayson (2000) (see also Chap. 10).

Recently, technological progress has brought a plenitude of new indirect/proxy measurement techniques. For example, Raman scattering fibre-optic distributed temperature sensing (DTS) has recently proven to be a powerful tool for accurately measuring temperature at high temporal (1 min) and spatial (1 m) spacing (Selker et al. 2006). DTS for temperature measurements have been applied in different environmental systems, such as to measure water temperature distribution along a stream (Selker et al. 2006), soil temperature (Steele-Dunne et al. 2010) and air temperature profiles in the atmospheric boundary layer (Keller et al. 2010). In the same way, portable hand-held infrared camera has recently been used to produce thermal images at the scale of 140×140 pixels over the spectral range 7.5–13 μm (Pfister et al. 2010). Another example is the use of visible and near-infrared (0.4–2.5 μm) diffuse reflectance spectroscopy to characterize the physical, chemical and biological properties of soils and sediments. Reflectance spectra can be measured directly in the field using portable spectrometers (Viscarra Rossel et al. 2006), which is a cumulative property that results from the inherent spectral behaviour of the heterogeneous combination of minerals, water and organic matter (Udelhoven et al. 2003).

The notion of indirect/proxy measurement techniques to study processes at catchment scale can be taken further to incorporate terrain parameters (Moore et al. 1991). These are now commonly computed from digital elevation models giving highly valuable information about topography, which in turn, has a major impact on the hydrological, geomorphological and biological processes at catchment scale.

Proxy measurements are specifically useful for the parameterization of numerical, spatially distributed models of the intrinsically heterogeneous dryland catchments. Information on vegetation cover and rill networks, which can be easily derived from air photos and digital elevation models, can for example be used as a proxy for the distribution of more difficult-to-measure soil-hydraulic parameters. In a study by Müller et al. (2007), vegetation cover was employed as a proxy for the spatial distribution of infiltration rate in the parameterization of an overland-flow and erosion model (Fig. 8.7a): the infiltration rate is highly sensitive and crucial for

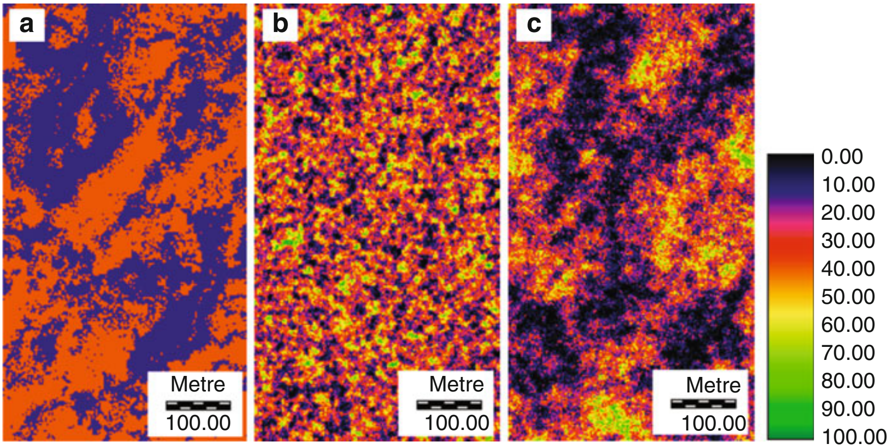

Fig. 8.7 Proxy measurements for infiltration rate of overland model: (**a**) vegetation cover as proxy for spatial distribution of infiltration rate: vegetated (*light*) cells with infiltration rate of 40 mm h^{-1} and bare (*dark*) cells with infiltration rate of 15 mm h^{-1}, (**b**) geostatistical interpolation of direct field measurements, (**c**) mixing of direct measurements with layout of rill network

model performance, exhibits significantly different values for different vegetation covers (including bare areas), and are notoriously difficult to measure directly in the field. Model parameterization can be also facilitated by mixing direct and indirect measurements. In the same study, the geostatistical interpolation of direct field measurements of infiltration rate (Fig. 8.7b) was overlaid with the layout of the rill network, as derived from a digital elevation model (Fig. 8.7c). The improved representations of pathways with low infiltration rates and thus a better reproduction of parameter connectivity considerably improved model performance in comparison to parameterization which did not use the rill network or vegetation cover as proxies for connectivity.

8.4.2 Isotopes

Isotopes are atoms of the same element that have different numbers of neutrons. The original isotopic composition of planetary systems is a function of nuclear processes in stars. Over time, isotopic compositions in terrestrial environments change by the processes of radioactive decay, cosmic ray interactions, mass-dependent fractionations that accompany inorganic and biological reactions, and anthropogenic activities such as the processing of nuclear fuels, reactor accidents, and nuclear-weapons testing. Radioactive (unstable) isotopes are nuclides that spontaneously disintegrate over time to form other isotopes. Stable isotopes are nuclides that do not appear to decay to other isotopes on geologic time scales, but may themselves be produced by the decay of radioactive isotopes.

Environmental isotopes are natural and anthropogenic isotopes whose wide distribution can assist in the coupling between biological and geomorphological systems. Typical uses of isotopes in environmental sciences include identification of mechanisms responsible for streamflow generation, testing flow-path and water-budget models developed using hydrometric data, characterization of flow paths that water follows from the time precipitation hits the ground until discharge at the streams, determination of weathering reactions that mobilize solutes along those flowpaths, determination of the role of atmospheric deposition in controlling water chemistry and assessment of biologic cycling of nutrients within an ecosystem (Kendall and Caldwell 1998).

There appears to be no single axis of causality between life and its landscape, but rather, each exerts a simultaneous influence on the other over a wide range of temporal and spatial scales. From a geomorphological perspective, a suite of stable and radioactive isotopes allow dating of sediment and bedrock ages (and erosion rates) over a wide range of temporal scales. Short lived isotopes such as ^{210}Pb, ^{241}Am and ^{137}Cs enable dating of buried sediment over the past $\approx$ 150 years (Appleby and Oldfield 1992; He and Walling 1996). New advances also allow the measurement of extremely low ^{14}C concentrations, enabling the dating of organic material up to 60 ka old (Bird et al. 1999). Radioactive cosmogenic nuclides (^{10}Be and ^{26}Al, ^{36}Cl) can determine the age of sedimentary deposits up to 5 Ma old (Granger and Muzikar 2001). From an ecological point of view, the study of ecosystems requires tools that indicate, integrate, record and trace fundamental ecological patterns and processes across contrasting scales. Isotopes (stable) can indicate, integrate, record and trace ecological processes across different temporal and spatial scales. The use of stable isotopes within networks that monitor ecological patterns and processes over broad spatial and temporal scales may reveal great insights into the changes in the ecosystems, and how they are being altered by human activities and global change (Resco et al. 2010).

The progress in landscape interpretation can be accelerated if a suite of cutting edge analytical technologies in both ecology and geomorphology are integrated. Numerous isotopic proxies allow determination of the physical and ecological structure of modern landscapes. New isotopic proxies for physical and biological processes and new methods for analyzing high-resolution data sets are emerging all the time. For instance, biological organisms fractionate isotopes such as carbon, oxygen and sulphur, while physical processes may fractionate calcium and strontium, providing a parallel set of stable isotopes. With these modern studies of where and how isotopes are fractionated, biogeochemical methods would allow us to recognize the bio-physical interactions forming landscapes (Reinhardt et al. 2010).

8.4.3 Process/Patterns

The study of vegetation patterns at landscape scales needs indirect methods and new technologies such as digital images (aerial photograph, radar or remote sensing).

The available databases often vary in frequency and quality of the images, because they are produced to others objectives, which are different of our interest in sample vegetation types. From this indirect approach, the analyzed surfaces are often ecogeomorphic units such as watershed, sub-watersheds, slopes, hydrological functional units (UHF), and large plots. These are chosen based on the type of vegetation whose pattern we want to characterize and the scale of the image that will enable us to obtain a more accurate analysis.

In the case of the sparse, dryland vegetation (e.g. the Ventós study discussed in Sect. 8.3 and illustrated in Fig. 8.2), this indirect approach has been used in order to detect general patterns of plant distribution and their relationship to the topography and ecogeomorphology. Standard aerial photographs can be scanned to permit subsequent analysis of time series of images. The scale varies with age (typically 1:50,000 for earlier photographs, and 1:20,000 in recent decades). Currently digital aerial photography at a scale of 1:20,000 is typically available, with pixel size of 0.68 m^2. Satellite imagery can also be used to analyze large areas of land, but it is not useful in general, because the support is coarse and depends on the satellite model and sensor used. The most frequently used images are from LANDSAT 5 and 7, but the pixel size of 30 × 30 m is too coarse to pick out patterns in the species present in drylands. Smaller pixel spacings (finer support) are available from the QuickBird platform, with a support of 2.5 × 2.5 m, which are more appropriate for some of the species present in drylands, but in this case, the cost of image acquisition needs to be taken into account and it is usually only practical to look at relatively small areas. In both QuickBird and especially LANDSAT imagery, it is difficult (but not impossible) to identify uniquely the reflectance characteristics of vegetation by species. Furthermore, the inaccuracy due to the errors of mixed cell occupation makes difficult to apply this technique to detect vegetation patterns. Mixture modelling can be used to unmix cells with different contents (e.g. Zhang et al. 2002; see also Sect. 8.4.3), although this approach works better at plant-type rather than species level.

The fact is the size of patch or plant species, that form the vegetation pattern we want to characterize, who determine the possibility of use these databases. In the case of Ventós, vegetation patches of *Q. coccifera* and *Pistacia lentiscus* shrubs, or *Pinus halepensis* trees, are at the limit to be analyzed with these images, because the size of patches are sometimes above and sometimes below the resolution afforded by the pixel size. Aerial photographs require prior digitizing, which add another error factor associated with the digitizing process (even when its quality is acceptable). However it may be considered a good way to ensure a match between the selected plots, forms and dimensions of species of interest. The best method for identifying patterns is the digital photography with small spacing (i.e. small pixel sizes), where is easy to classify and associate the spots of the image with species of interest. With this material it was possible to identify many of the individuals of *Stipa tenacissima* tussocks, *Q. coccifera* and *P. lentiscus* patches, isolated trees of *P. halepensis* and the bare spaces between these types of vegetation. Having the digital images, is possible to use automatic methods for classification by levels of reflectivity associated with each species, bare soil or rock outcrop (Fig. 8.2). In addition, this type of image is easily georeferenced and will provide coordinates for each pixel or vegetation patch, thereby facilitating further processing of vegetation distribution with appropriate models to detect patterns (e.g. the Wiegand model: see Chap. 6 and Sect. 9.2).

8.4.4 Remote Sensing Methods

Ecogeomorphic processes and associated dynamics of vegetation and land resources concern extensive areas of often poor accessibility. Thus, remotely-sensed data have long been considered an attractive source of information to complement and extrapolate limited field measurements. The rapid, ongoing diversification of both active (e.g. radar, laser) and passive (optic) sensors in terms of spatial and temporal spacing or spectral resolution (Wulder et al. 2004; Chambers et al. 2007) have made remote-sensing become a central tool for studying ecogeomorphic processes in drylands. The array of available data (Table 8.2) can be presented in terms of trade-offs such as spatial versus temporal spacing or spatial spacing versus extent of study area. For instance, the cost of images of very small spatial spacing (i.e. 0.5–5 m), which allow monitoring ecogeomorphic objects of small size, is still high and precludes covering very large areas or achieving high temporal frequencies. On the other hand, cheap or free data of high to medium spacing can help achieving monitoring schemes of high temporal frequencies (e.g. daily, weekly) over areas that may reach continental to global extents. In this case, the larger spatial spacing does not generally allow detecting individual ecogeomorphic objects (e.g. shrubs, gullies, etc.) and impose to base the monitoring on either land cover categories or mean-field variables expressing for instance soil surface properties or vegetation photosynthetic activity. Examples of studies analysing remotely-sensed data of different types in arid environments are summarized in Table 8.2. This table is by no means exhaustive, but rather aims at giving an overall idea of what can be done with some of the most usual data sources.

For optical sensors, which provide records of radiometric reflectance in the visible and infrared wavelengths, besides the abovementioned resolution improvements, difficulties in obtaining calibrated surface reflectance values are progressively being overcome (e.g. correction of radiometric perturbations caused by the sun-scene-sensor geometry or atmospheric effects). The tremendous efforts made within the NASA Earth Observing System (EOS) on data acquired by the MODIS sensors are a good example of the importance of the task (Huete et al. 2002). In most countries, it is now relatively affordable to build historical series using optical satellite images dating back to the 1960s while digitising older aerial photos (1930–1950s depending on the countries) can provide deeper historical baselines. The potential of such series to understand the dynamics of drylands is just revealing itself through recent studies (Barbier et al. 2006; Fairfax and Fensham 2002; Laliberte et al. 2004; Ares et al. 2003; Wigley et al. 2010).

Table 8.2 Non-exhaustive survey of the characteristic of some commonly used Earth-observation data

Object	Data type	Carrier/sensor	Spatial spacing	Spatial extent	Temporal spacing	Temporal extent	Refs.
Tree/shrub cover and abundance Crown size distribution Vegetation pattern Drainage network evolution	Very high res. optical (often panchromatic)	Airborne photos/Spaceborne (Quickbird, Ikonos, GeoEye, Orbview 2–3, Spot 1–5, Landsat 7 ETM+, Corona)	0.4–15 m	Landscape to global	1 month–10 year	50–70 year	Barbier et al. (2006), Fairfax and Fensham (2002), Laliberte et al. (2004), Ares et al. (2003), Wigley et al. (2010), Couteron et al. (2006), Couteron (2002), Deblauwe et al. (2011), LeBlanc et al. (2008)
Vegetation cover (via unmixing approaches) Land use/land cover Photosynthetic activity Soil/vegetation water content Deforestation/degradation Fire history via fire scar detections	High to medium res. optical multispectral	Spaceborne (Spot, Landsat 4–7, Aster, Hyperion)	20–30 m	Regional to global	1 month–10 year	25 year	Asner et al. (2003), Graetz and Gentle (1982), Smith et al. (1990), Couteron et al. (2001)
Fire history (frequency, timing), via active and fire scar detections Leaf phenology	Low res. optical multispectral	Spaceborne (Modis, AVHRR)	250 m–1 km	Regional to global	1 day–1 month	10–30 year	Carmona-Moreno et al. (2005), Myneni et al. (2007)

Topography	Radar	Spaceborne (SRTM, Aster)	30 m, 90 m	Regional to global	–	–	Hirano et al. (2003), Rodriguez et al. (2006)
Topography-derived moisture indices and geomorphic classes	Optical stereoscopy						
Rainfall	Radar	Spaceborne (TRMM)	0.25–5°	Regional to global	3 h–1 day	10 year	Iguchi et al. (2000)
Temperature	Medium res. optical multispectral	Spaceborne (MODIS)	250 m, 500 m, 1 km	Regional to global	1–2 day	10 year	Wan and Li (2002)
Incoming radiation	Low res. optical multispectral	Spaceborne (CERES)	1°	Regional to global	1 month	10 year	Wielicki et al. (1996)
Micro-topography (e.g., termite mounds)	Small footprint LiDAR	Airborne	0.5–2 m	Local to landscape	?	?	Levick et al. (2010)
Vertical biomass distribution							
Tree/shrub crowns							

8.4.4.1 Overview of Analysis Techniques

The diversity of data, topics and targets of the studies addressing ecogeomorphic processes obviates any complete classification, even for the only optical data. One may nevertheless roughly identify three broad types of approaches to extract thematic information from the physical signal that make the data: (i) combining raw channels to compute indices that are expected to strongly correlate with vegetation properties (e.g. vegetation indices such as the normalized difference vegetation index NDVI, the enhanced vegetation index EVI) or soil surface colour. (ii) locating in the image 'first-level' objects defined as sets of pixels satisfying predefined rules of spectral similarity or spatial adjacency (e.g. patches of vegetation or bare soil); (iii) identifying more complex 'second-level' objects or areal properties defined from features computed either on first level objects or directly from the spatial distribution of pixel values (also called second-order properties). This last point relates to the broad concept of texture on which we will elaborate in the following section.

In drylands, vegetation indices are generally not saturating since plant biomass and vegetation cover are low, but the soil surface properties and colour may substantially alter the value of the index making correlations with vegetation features unstable. First level objects or patches are classically deduced from either supervised or unsupervised classification of image pixels based on their reflectance properties. Such pixel-wise approaches, even though exemplified by a huge literature features several drawbacks related to poor calibration of reflectance values in images that impedes reproducibility of the classifying in space and time. (Actually none of the method that directly uses uncorrected reflectances is immune to this problem.) Another drawback is that relative positions of pixels are ignored by the most popular methods and algorithms constraining classification via adjacency rules are scarcely (but increasingly, e.g. Laliberte et al. 2004) used. A last problem is that a large share and often the majority of pixels may be heterogeneous and contain more than one of the land cover categories. This case is frequently encountered in drylands where vegetation and bare soil make up complex mosaics in which constitutive items may often be of smaller size than the pixels. Such a situation, which is generally prevailing if small to medium spatial spacings are used, may also occur with very small spacings.

To cope with heterogeneous pixels, spectral mixture analyses techniques, also known as spectral response unmixing have long been proposed and applied to arid and semiarid landscapes (Asner et al. 2003; Graetz and Gentle 1982; Smith et al. 1990; Couteron et al. 2001). Those techniques model the response of any pixel in the scene as a linear combination of the spectral signatures of a limited number of "end-members" posited as "pure" components (e.g. green foliage, senescent foliage, bare ground, shadow). Model inversion allows assessing the relative contribution of each of the endmembers to any scene pixel. The accuracy of the assessed contributions strongly depends on the relevance of the set of endmembers and on the quality of the measure of the spectral response of the endmembers.

Texture analyses are an appealing alternative to limitation of pixelwise classifications or indices computation. Texture, along with embedded terms such as coarseness, grain and organization, is a central concept for characterizing spatial features of territories from EO data. Although the complexity of the concept proved hard to capture through a formal definition (Musick and Grover 1991) it may be convenient, following Haralick (1979), to retain two broad categories of approaches to texture: one structural, the other statistical. The structural approach relies on the prior definition of categorical classes and delineation in the image of corresponding first level objects (patches). The patches may then be counted and their geometry and spatial distribution characterized as has been done in numerous studies in landscape ecology (Turner and Gardner 1991). As mentioned above, an obvious requirement is that the thematic variable of interest (e.g., vegetation cover) may unambiguously be delimited into discrete patches and that the radiometric signal be sufficiently stable for the categorization to be reproducible through space or time. This requirement is not always granted and, in fact, arid and semi-arid landscapes have been recognized as displaying progressive transitions between vegetation types and land-use units (Puech 1994).

On the other hand, in the statistical approach, one searches to quantify the spatial (or temporal) properties of the inter-pixel variance, by the means of ‘structure functions’ that all derive from the spatial autocovariance in pixel values. The correlogram, the variogram and the spectra produced by wavelet or Fourier transforms make a large array of structure functions. Using such functions aims at a quantification of image texture at multiple scales, in terms of heterogeneity, regularity, orientations, etc., that does not requires making hypotheses on the number or discreteness of the thematic categories. To a large extent, computing structure functions, which basically deal with variances and covariances (second order properties), can even be made independently of contrast (range) and brightness (mean) values, a crucial advantage when dealing with multisource, non-calibrated data (Couteron et al. 2006).

8.4.4.2 The Use of Fourier Spectra for Texture Analysis

Some particularly useful statistical approaches for the synchronic and diachronic study of pattern in arid vegetations have been developed on the basis of Fourier spectral analysis (Couteron et al. 2006; Mugglestone and Renshaw 1998; Couteron 2002). The principle of such Fourier methods is to start by converting a signal extract of definite length (be it temporal-1D or spatial-2D) to a so-called frequency space, in which the spatial structure present in the extract is decomposed in terms of power (or variance) accounted for by discrete frequency bins along Cartesian axes (Fig. 8.8) (Rayner 1971; Diggle 1989). Frequency (the inverse of wavelength) refers to the number of times a structure repeats itself within the extract. Theoretically, this transform is obtained by fitting the signal with sine and cosine periodic functions, although in practice powerful algorithms have been developed (e.g. the Fast Fourier Transform) allowing a very efficient and rapid implementation of the Fourier

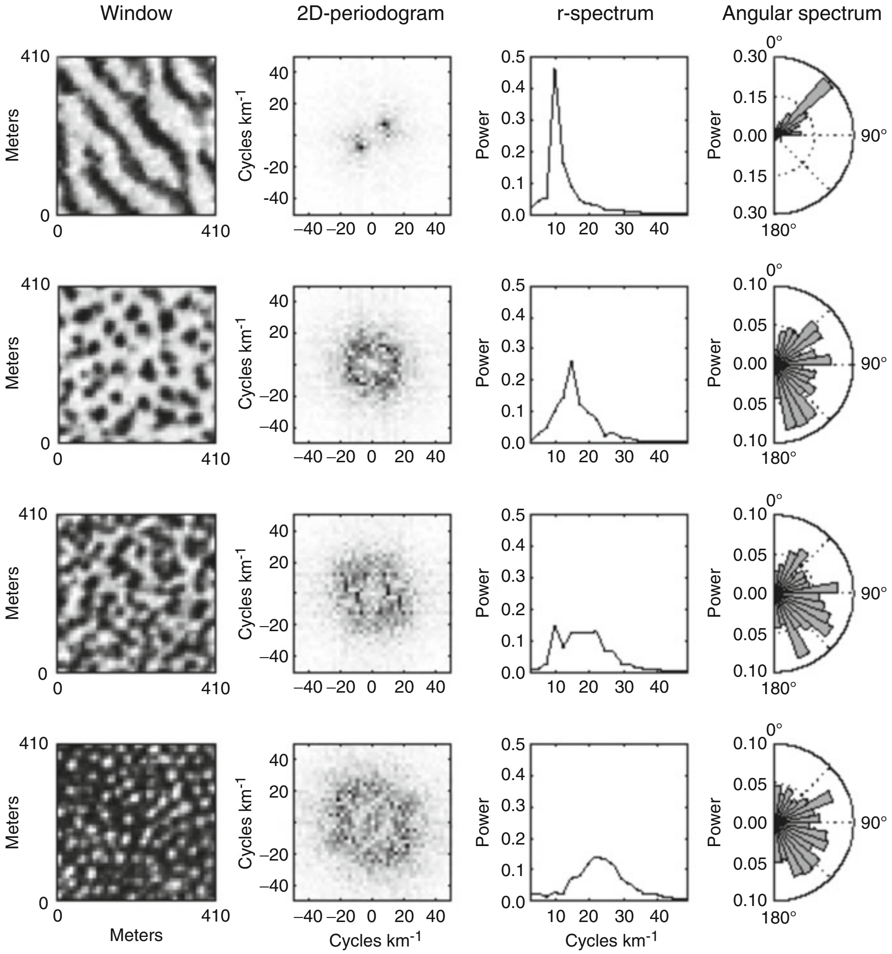

Fig. 8.8 Fourier spectral signatures of typical periodic vegetation structures encountered in semi-arid areas. The *first column* to the *left* illustrates Spot panchromatic satellite views of banded, spotted, labyrinthine and gapped patterns, in which vegetation appears in dark and bare soil in brighter tones (Deblauwe et al. 2011). These structures are all regular (spatially periodic) but vary in terms of symmetry and wavelength. These variations are apparent in the 2D periodograms (power spectra) in the second column, where pattern properties are represented in the frequency space (units of cycles km^{-1}), as the amount of variance accounted for by frequency bins along the two Cartesian directions. The 'zero frequency' is located in the *middle*. The sign of frequency (direction of wave travel) is unimportant here, and the 2D periodogram are thus symmetric about the origin. In the *third* and *fourth columns*, the same information is simplified to place the emphasis either on frequencies (averaging periodogram values across orientations: r-spectrum) or on the orientation (averaging periodogram values across frequencies: θ-spectrum)

transform on discrete signals. To reflect the duality of the transform in terms of sine and cosine functions, the result of a Fourier transform is expressed in complex numbers, and can thus be characterized by amplitude and phase values. Squared amplitude values provide the well-known power spectrum or periodogram, which presents the advantage of providing an effective (additive) partition of the image variance among frequency bins.

The widespread use of the Fourier approach in all fields of engineering and mathematics is due to several interesting properties among which is the asymptotic independence of the values making the power spectrum. This is a major difference with the otherwise closely related autocovariance function (mathematically, the Fourier transform of the autocovariance function yields the power spectrum), which allows well-mastered significance tests of the power spectrum, e.g., for assessing the significance of observed peaks against different forms of noise backgrounds (Diggle 1989; Couteron and Lejeune 2001). An important asset of the Fourier approach is that it defines pattern properties in a framework similar to that of self-organisation models. While it is of particular appropriateness for the characterization of periodic structures (Deblauwe et al. 2011; Barbier et al. 2006), it is also relevant to model and quantify scale-free fractal patterns (Keitt 2000).

Several options are available within the framework of Fourier spectral analysis when it comes to comparing patterns. The simplest one being the possibility to compute the ratio between the amplitude values of two Fourier power spectra. The ratio between the means of two sets of spectra can also be considered and tested (Barbier et al. 2006; Couteron et al. 2006; Diggle 1989). As the theoretical parametric distribution of this ratio is known (Fisher distribution), a simple statistical test is available to detect significant structural differences between the spectra and identify the range of scales at which the patterns may differ. Using this approach, Barbier et al. (2006) were able to evidence a large scale amplification of the signature (peak amplitude in the spectrum) of periodic gapped patterns after a prolonged drought episode in SW Niger. Similarly, Couteron et al. (2006) showed that the main difference between settled and unsettled locations in Northern Cameroon was the regression of the woody cover, that occasion a dwindling of the contribution of high frequencies to the spectrum.

A second approach is based on the computation of a cross-periodogram, which can be seen as a scale- and orientation-specific partition of the covariance between the compared signals. An advantage of the latter method is that it also quantifies phase shifts that may occur between the signals (Barbier et al. 2010). Thanks to this method, it was possible to demonstrate the independence between vegetation structure and potential topographic effects, at all relevant scales, in a field study (with scales between 5 and 100 m) carried out in SW Niger. In another study, (Deblauwe 2010), a series of images spanning several decades was used, and cross-spectral methods (featuring phase shift information) to systematically check for and quantify upslope migration of banded patterns at landscape to regional scales, on sites located on three continents. One of the results, which illustrates the potential of the method, was that the speed of migration, when it occurred, depended in a predictable way on pattern wavelength (see also Chap. 13).

Finally, a third option is available by referring to the framework of ordination/classification techniques to systematically compare the structure of numerous image extracts, as is often necessary when dealing with EO datasets at landscape to regional scales (Barbier et al. 2006; Couteron 2002). To reduce the dimensionality of the dataset, a feature extraction (simplification) stage is generally required. For instance, if one is only interested in pattern scales (frequencies), and not in possible dominant orientations or other forms of anisotropy in the patterns, it is possible to average out the periodogram information across spatial orientations (Mugglestone and Renshaw 1998). This results in the computation of r-spectra (‘r’ for radially averaged), which can readily serve as an input for a Principal Component Analysis, to name but one ordination/classification technique, to further decrease dimensionality. Applications over arid landscapes showed that the first two axes of such a PCA generally provide a synthetic ordering of images extracts along a coarseness-fineness gradient congruent with the visual appraisal of experienced assessors (Couteron et al. 2006). The approach also allows detecting spatially periodic patterns (singled out along the second PCA axis), which are often the signature of self-organisation phenomena (Couteron 2002; Barbier et al. 2010).

Fourier methods are sufficiently flexible to allow tailor-made analyses to characterize and classify very specific pattern properties. In Sudan, Deblauwe et al. (2011; see also Chap. 13) could analyse the modulation of self-organized periodic vegetation patterns along aridity and slope gradients, in terms of periodicity, symmetry and wavelength (Fig. 8.8). In another study (Deblauwe 2010; Koffi et al. 2007), the dominant orientation of banded patterns could be compared to the steepest slopes directions. The study concluded that systematic deviations existed from the expected orthogonality of bands vs. slope. Interestingly, these deviations pointed towards the dominant wind direction.

As the examples above attempt to illustrate, pattern description can be very simplified, if one only wants focussing on specific features (e.g., orientation or periodicity) to conduct analyses at a regional scale, or else very detailed, if one is to compare in the finest detail the output of a simulation with observed natural patterns. For instance, to validate their representation of seed propagation in patterned vegetations, Thompson et al. (2008) used such a detailed comparison of the power spectra.

As the methodological toolbox for pattern characterization develops, opportunities clearly arise to go beyond qualitative comparisons between empirical data and theoretical models. Quantitative validation is now possible, and where some parameters still need to be estimated, there are possibilities for (cautious) model inversion and indirect calibration.

8.5 Summary

An emphasis on pattern in studies of dryland degradation means that there is a need to reorientate standard methods of measurement. Evaluation of pattern in a reliable way requires much larger sample sizes, and for samples to be taken in an appropriate way. Often, random sampling will be inappropriate as it will be a relatively inefficient way of characterizing pattern, and we will usually need to employ our process knowledge in designing sampling strategies that will represent those patterns. The fact that different patterns may occur in quite different characteristics over small areas (e.g. Müller et al. 2008) suggests that extreme care is needed from the outset of a study to maximize research resources.

As well as standard field techniques for direct measurement, there is an increasing array of tools that can be used to can insight be indirect representation of key parameters. Indeed, it may be appropriate to look at a multi-phase research strategy where proxy measurements are used to guide more detailed measurement on the ground. The same approach may be true of remote sensing, and it is clear that significant advances over the last decade have enabled the observation and evaluation of pattern in drylands over significant parts of the Earth's surface.

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Chapter 9
Assessment of Patterns in Ecogeomorphic Systems

Nicolas Barbier, Juan Bellot, Pierre Couteron, Thorsten Wiegand, Volker Grimm, Vincent Deblauwe, Peter Biro, and Eva Nora Mueller

Abstract Through studying patterns we can come to understand the systematic formulae that generate them. The scale and relative stability of the processes causing dryland degradation are particularly inviting to pattern analysis. The following sections give an overview of the functioning and application of pattern assessment tools including (geo)statistical, spectral and indicator methods and explore the potential of pattern-oriented modelling.

9.1 Patterns in Drylands

It is understood that the patchiness of vegetation in dryland environments is key to the evolution of dryland systems. To analyse the mechanisms behind the formation of patchy vegetation, different kinds of typical dryland patterns are typically assessed: according to the ir/regularity of vegetation patterns; their environmental and temporal distribution; the heterogeneity of soil and water resources and their distribution; and the role of dis/connectivity in terms of material transport and

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N. Barbier • P. Couteron • V. Deblauwe
IRD/UMR AMAP, Botany and Computational Plant Architecture, 34398 Montpellier, France
e-mail: nicolas.barbier@ird.fr; pierre.couteron@ird.fr; vdeblauw@ulb.ac.be

J. Bellot
Depto. Ecologia, Universidad de Alicante, 03080 Alicante, Spain
e-mail: juan.bellot@ua.es

T. Wiegand • V. Grimm
Department of Ecological Modelling, Helmholtz Centre for Environmental Research
UFZ, 04318 Leipzig, Germany
e-mail: thorsten.wiegand@ufz.de; volker.grimm@ufz.de

P. Biro • E.N. Mueller (✉)
Institute of Earth and Environmental Science, University of Potsdam, 14476 Potsdam, Germany
e-mail: peter.biro@uni-potsdam.de; eva.mueller@uni-potsdam.de

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turnover processes. Tightly coupled feedbacks between biotic and abiotic elements determine structure and function of drylands over temporal scales from periods of seconds (e.g. runoff processes during rainfall events; landslides) to periods of several decades (e.g. vegetation pattern formation), to millennia (e.g. soil development), and over spatial scales from centimetres (cracks in soil crusts) and hectares (e.g. channel networks) to square kilometres (e.g. piedmont development).

The occurrence of patches of enhanced resource availability and productivity for vegetation is a ubiquitous aspect for drylands which are susceptible to degradation (Reynolds et al. 1999). Schlesinger et al. (1990) among others have argued that semi-arid and arid vegetation often form a self-sustaining system which concentrates water and soil resources in 'islands of fertility' (also see Sect. 4.3). Schlesinger et al. (1990) hypothesised that vegetation change – for instance, triggered by overgrazing as has happened in the south-western United States – is accompanied by a change from relatively uniform distributions of water, soil moisture, nitrogen and other soil resources in grassland to an increase of their spatial and temporal heterogeneity in shrubland. Directly linked to heterogeneous resource availability and the competition for resources is the formation of vegetation as regular stripes, spots, labyrinths or gaps with sizes from several metres to hundreds of metres surrounded by areas of bare ground. At even larger spatial scales, Deblauwe et al. (2008) identified the ubiquitous vegetational patterns at the interface of arid/semi-arid regions as a biome-scale response to environmental conditions including soil, climate and topography.

Chapters 3, 4 and 5 have already explored the self-organising nature of dryland patterns, and the short- and long-range ecogeomorphic processes that lead to the formation of dryland patterns. Anthropogenic disturbances such as grazing, deforestation, the building of dams and roads, or climatic change alter the structure and functioning of dryland patterns and/or connectivity features of transfer routes. Analysis tools are thus required to analyse past and future effects of change. This chapter presents three methods that enable the assessment of complex, altered dryland patterns: a geostatistical approach of vegetation cover in the semi-arid Patagonian steppe, a multi-scale spectral analysis of self-organised periodic vegetation patterns along aridity and slope gradients in semi-arid systems, and a connectivity assessment in a semi-arid Mediterranean shrub-grass landscape. The end of the chapter explores the potential of pattern-oriented modelling to overcome apparent limitations of current pattern assessment approaches.

9.2 Quantification of Point and Patch Patterns

Quantifying the patchiness intrinsic to dryland systems should help us understand the processes that generated these patterns, and vice versa, with understanding the influence of spatial pattering of soil resources on population or community dynamics. This section presents three methods for quantifying spatial point and patch patterns which emerge from the interactions of biotic and abiotic factors and processes. The first case deals with the impact of the physical environment on the spatial pattern of presence or absence of vegetation. The second case quantifies the small-scale spatial correlations induced by interactions among vegetation patches. The third case presents methods to quantify the spatial structure of landscapes, such as landscape fragmentation and island formations, which also play a significant part in the dynamics of dryland species and communities.

Firstly, spatial patterns of vegetation in drylands are driven by abiotic conditions which are often heterogeneous in space, or more simply, some areas are more suitable for a given plant species than others; some species require wetter environments to become established whereas other species are better adapted to establish in open areas. These observations are condensed into the niche concept (Hutchinson 1957), which is one of the most important concepts in ecology to explain the diversity of species. According to this concept, each species is adapted to a different combination of environmental conditions which allows it to explore a different combination of resources than explored by other species. A rich body of methods has been developed to investigate species–environment relationships. In the simplest case the species–environment relationships can be quantified as the probability $\lambda(x, v_i(x))$ that a given species is present at location x, given environmental conditions $v_i(x)$ at location x. The probability $\lambda(x)$ can be determined using techniques of predictive habitat or species distribution models (Guisan and Zimmermann 2000; Guisan and Thuiller 2005; Elith and Leathwick 2009) which are static and probabilistic in nature. These models relate field observations statistically to environmental predictor variables v_i. In general, the environmental variables believed to provide the main causal, driving forces for the distribution and abundance of the plant forms should be used. Species distribution models are usually applied to larger spatial scales on the population or species distribution level but, if environmental conditions are mapped on smaller scales, the same techniques can work.

Secondly, spatial patterns of vegetation in drylands are driven by biotic interactions and processes such as competition, facilitation (see Chap. 4) and seed dispersal. In plant ecology, there is increasing evidence that spatial distributions and processes play an important role in the assembly, temporal dynamics and internal functioning of plant communities (e.g., Watt 1947; Pacala 1997). It is well established that most plant species are not randomly distributed but, instead, are often locally aggregated or over-dispersed at one or several spatial scales (Watt 1947; Pielou 1977). These observations continue to stimulate theoretical work investigating the impact of the spatial arrangement of individual plants on species coexistence and the systems of biodiversity (e.g. Hurtt and Pacala 1995; Chesson 2000). Because spatial patterns emerge as a function of the interaction of location, process and mechanism, they may conserve a history of past processes that constitutes an ‘ecological archive’ (McIntire and Fajardo 2009) – a signature of the underlying processes that operated in generating these patterns. An important step in revealing the pattern-process link is therefore to quantify the spatial patterns.

Techniques to quantify spatial patterns of ecological objects that can be idealized as points (e.g., locations of grass tufts, shrubs or trees) emerged in the 1980s and are summarised under the label ‘point pattern analysis’ (e.g., Ripley 1981; Stoyan and

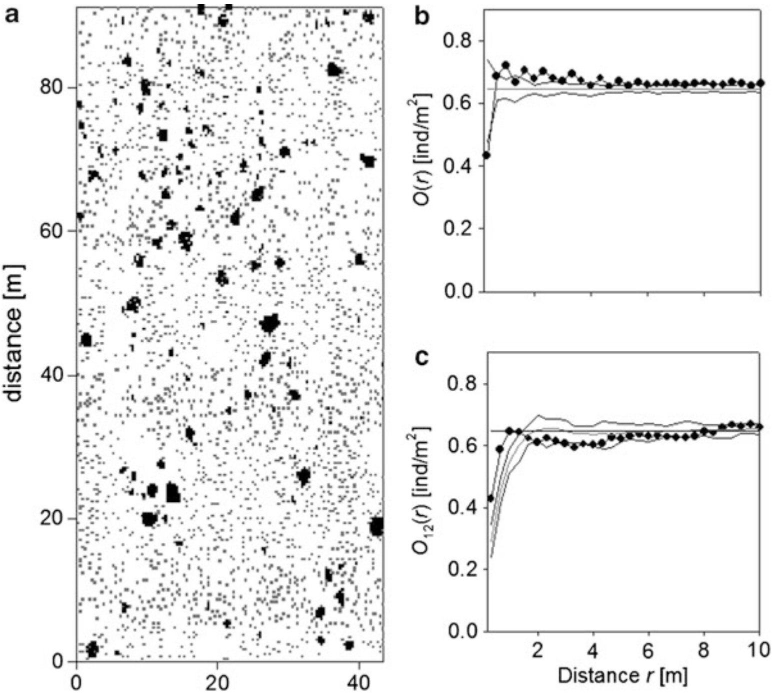


Fig. 9.1 Example of a spatial analysis of plants in a semi-arid steppe. (**a**) Spatial pattern of the grass species *Poa liguaris* (*grey*) and the shrub species *Adesmia campestris* (*black*). The area was mapped with a spatial resolution of 0.33 m and each 0.33 × 0.33 cell was assigned either to be empty, occupied by a grass tuft or occupied by a shrub. (**b**) The neighbourhood density O(r) of the point pattern of *Poa liguaris* (*dots*) as a function of distance r, expectation for a random pattern (*grey line*) and simulation envelopes of the null model of a random pattern (*black lines*). (**c**) The bivariate neighborhood density O_{12}(r) of *Poa liguaris* at distance r from area occupied by the shrub species *Adesmia campestris*

Stoyan 1994). A point pattern is a map of ecological objects (e.g., plants) within a specific observational window (e.g., a field plot or a section of an air photo, Fig. 9.1). Techniques of point pattern analysis are designed primarily to reveal small-scale spatial interactions among neighbouring points and normally assume that the environment is homogeneous, although newer approaches include some assumptions of an underlying heterogeneity by using spatially varying intensity function λ(x) (e.g., Baddeley et al. 2000; Wiegand and Moloney 2004).

The most important summary statistics of point pattern analysis is the pair correlation function g(r) which summarises information on the small-scale correlation structure among pairs of points as a function of distance r. It is derived from the quantity $O(r) = \lambda\, g(r)$ which can be interpreted as the expected number of points at distance r from a typical point of the pattern where λ is the density of the pattern (= number of points divided by area). The O(r) quantity is called the O-ring statistic and can be used directly to assess the neighbourhood density (Wiegand and Moloney

2004). In point pattern analysis, each point of the pattern is visited; its neighbours within distance band r ± dr are counted and the mean density of neighbours within the corresponding area 2π r dr is determined. In contrast to geostatistical methods such as semi-variograms which describe the small-scale spatial correlation structure of continuous variables that are measured at discrete points, the pair correlation function also quantifies the spatial structure of the points.

If the pattern reads as random (i.e., with no interactions among points), the O(r) yields the overall density λ for all distances. If points are clustered, they have a locally higher neighbourhood density: i.e., given O(r) is larger than λ; if the points show a regular spatial pattern, O(r) becomes smaller than λ.

Clearly, the value of the O(r) depends on the number of points in the study plot. If the direct interpretation of the neighbourhood density is not required, the intensity normalised version of the O(r) (the pair correlation function) is used instead. The pair correlation function yields $g(r) = 1$ for random patterns, $g(r) > 1$ for clustered patterns and $g(r) < 1$ for regular patterns. For example, Fig. 9.1a shows the spatial pattern of the grass species *Poa liguaris* in the Patagonian steppe (Wiegand et al. 2006) and Fig. 9.1b the resulting neighbourhood density O(r). The plot size was 43.3 × 91.3 m and 2,559 cells were classified as *P. liguaris*, thus the overall density of *P. liguaris* individuals yields $2{,}559/(43.3 \times 91.3) = 0.647$ individuals/m^2 (grey horizontal line in Fig. 9.1b). For the distance $r = 0.33$ m the neighbourhood density is below the expectation for a random pattern, for distances $r > 1$ m, it tends to be larger than the expectation for a random pattern, and for distances larger than 5 m, it approaches the expectation of a random pattern. The results suggest that negative interactions such as competition may act between nearby plants and a cluster mechanism at intermediate distances.

Comparison of the observed O-ring statistic or related statistics with the theoretical expectation of a random pattern is usually not sufficient because the data are a realisation of a stochastic process and assessment of the properties of a pattern requires consideration of stochastic effects induced by the finite number of points. One approach to test for non-random spatial structure is to use simulations of an appropriate null model and compare the O-ring statistic of the data to the one obtained from the realisation of the null model. In practice, e.g. 19 and 199 data sets are created as realisations of the point process underlying the null model, and simulation envelopes (e.g., the lowest and highest values of 19 simulations, or the 5th lowest and highest value of 199 simulations) show the possible range of the O-ring statistic under the null model. If the observed O-ring statistic is inside the simulation envelopes, we cannot distinguish the data from the null model. However, if the O-ring statistic is below or above the simulation envelopes of a completely random pattern, the pattern may be aggregated or regular, respectively (Fig. 9.1b). In this case, it should be noted that simulation envelopes are not confidence envelopes in a confirmatory sense. To reject or accept a given null model, more refined methods such as Goodness-of-Fit tests are required (Loosmore and Ford 2006).

Point pattern analysis also apt for studying the spatial relationship between two patterns, for instance (1) a shrub species *Adesmia campestris* and, pattern (2), a grass species *Poa liguaris* (Fig. 9.1a on a 33 × 33 sqm grid). In this case, the quantity

$O_{12}(r) = \lambda_2 g_{12}(r)$ can be interpreted as the expected number of type 2 points at distance r from a typical point of pattern 1 where λ_2 is the density of type 2 points. If the two patterns are independent (i.e., no interactions among points of the two types), the pair correlation function yields a value of one for all distances. If the two types of points are attracted, then pattern 2 occurs at locally higher neighbourhood densities around points of pattern 1. That is $g_{12}(r) > 1$ or $g_{12}(r) < \lambda$ if the points of the two patterns show repulsion. Figure 9.1c gives an example of such a bivariate analysis which shows that grass tufts occur at small distances (0.33–1 m) with higher densities in the neighbourhood of shrubs than expected by a random distribution (Fig. 9.1c). Hence, the statistics provide evidence that shrubs facilitate the growth and distribution of grass (Wiegand et al. 2006).

Many different summary statistics are used in point pattern analysis (e.g., Illian et al. 2008): the pair correlation function presented here is only one prominent and especially useful example. Other useful summary statistics include Ripley's K function, which is the cumulative version of the pair correlation function: i.e.,

$$K(r) = \int_{t=0}^{r} g(t) 2\pi t \, dt, \tag{9.1}$$

or the distribution functions $D_k(r)$ of the distances to the kth neighbour. Different summary statistics quantify different aspects of the spatial patterns and a thorough assessment of spatial structure may require use of several summary statistics. Illian et al. (2008) provides a comprehensive overview of summary statistics used in point pattern analysis.

The spatial structure of vegetation also influences the dynamics of species and communities, an example being the effect of fragmentation. Here the landscape is conceptualised as a mosaic of different habitat types and the spatial structure of one type or the spatial relationship of one type to a second type is quantified (Wiegand et al. 1999). The spatial structure and relations of different habitat types influence processes of population and community dynamics with the result that landscapes with the same composition but different spatial structure may show different dynamics. With techniques similar to that of point pattern analysis, such patch or landscape patterns can be quantified. If the landscape map is a categorical map, where each raster cell contains one category, one can formally transform the categorical raster map into a point pattern where the centres of the cells of the raster map are used as point locations and the categories define the types of points (Wiegand et al. 1999, 2006). This approach has been used in the above example to represent the patterns of the shrubs which typically occupy an area larger than one raster cell (Fig. 9.1a). Note that in the case of categorical maps the O-ring statistics require slight re-interpretation. Because one raster cell can logically accommodate only one point of a given category the $O_{12}(r)$ can be interpreted as the probability that a cell of type 2 can be found at distance r of a cell of type 1.

Uni- and bivariate O-ring statistics can be used as indices of spatial structure for a given landscape map and related with observation on variables of population or community dynamics. An advantage of using O-ring statistics for this purpose is that they are scale-dependent measures of spatial structure.

In the field of landscape ecology, a large number of landscape metrics have been developed to characterise landscape structure, summarised in the software FRAGSTATS (freely available, see McGarigal et al. 2002). In FRAGSTATS, the landscape is represented as a categorical map such as shown in Fig. 9.1. Patches are approximated by adjacent cells of the same type similar to the patches that represent individuals of *Adesmia campestris* in Fig. 9.1a. The landscape metrics focus on the composition, spatial character and distribution of patches of one or more types. Commonly used measures of composition include proportional abundance of each class, richness, evenness, and diversity. FRAGSTATS contains several calculation modi to quantify the spatial configuration of the landscape metrics, including patch size distribution (e.g., mean, median, max, variance, etc.), patch shape complexity (e.g., perimeter-to-area ratio, or fractal dimension), average isolation of the patches (mostly measured by means of nearest neighbour distances), or contagion – that is a tendency of patch types to be spatially aggregated (see Turner et al. (2001) for a thorough summary of landscape metrics).

The methods presented in this section help quantify stochastic small-scale correlation structures of plants or landscape patches but fail to a certain extent to describe anisotropic or ordered spatial patterns such as wavelike or periodic structures. In this case, spectral analysis of spatial data sets may be the better choice, as described in the following section.

9.3 Multi-scale Spectral Analysis of Patterned Vegetation

Some particularly useful statistical approaches for the synchronic and diachronic study of patterns in dryland vegetation have been developed on the basis of Fourier spectral analysis (Couteron 2002; Couteron et al. 2006; Mugglestone and Renshaw 1998). The principle of Fourier methods is to start by converting a signal extract (subset) of definite length (from either temporal-1D or spatial-2D data) to a so-called frequency space, in which the spatial structure present in the extract is decomposed in terms of power (or variance) accounted for by discrete frequency bins along the Cartesian axes (Fig. 8.8) (Rayner 1971; Diggle 1989). Frequency (the inverse of wavelength) refers to the number of times a structure repeats itself within the extract. The transformation is obtained by fitting the signal with sine and cosine periodic functions. Powerful algorithms have long been developed (e.g. FFT: Fast Fourier Transform) that allow a very efficient and rapid implementation of the Fourier transform on discrete signals. To reflect the duality of the transform in terms of sine and cosine functions, the results of a Fourier transform are expressed in complex numbers, and can thus be characterised by amplitude and phase values. Squared amplitude values provide the well-known power spectrum or periodogram, which has the advantage of providing an effective (additive) partition of the image variance among frequency bins.

The widespread use of the Fourier approach in all fields of engineering and mathematics is due to several interesting properties among which is the asymptotic independence of the values that compose the power spectrum. This facet constitutes the major difference to the otherwise closely related autocovariance function (mathematically, the Fourier transform of the autocovariance function yields the power spectrum), and allows well-mastered significance tests of the power spectrum, for example, for assessing the significance of observed peaks against different forms of noise backgrounds (Diggle 1989; Couteron and Lejeune 2001). An important asset of the Fourier approach is that it defines pattern properties in a framework similar to that of self-organisation models. The approach is particularly appropriate for the characterisation of periodic structures (Deblauwe et al. 2012; Barbier et al. 2006) and the quantification of scale-free fractal patterns (Keitt 2000). Last but not least, the Fourier framework can also be applied to point patterns (discussed in Sect. 9.2), which offers an interesting yet seldom employed alternative for the coupled analysis of raster and mapped point data. In Burkina Faso, for example, Couteron (2001) analysed the relationship between individual shrub species and the entire vegetation metrics that occurred in a banded configuration as derived from panchromatic air photos using the point pattern Fourier approach.

For the comparison of patterns, several options are available within the framework of Fourier spectral analysis. The simplest one is to compute the ratio between the amplitude values of two Fourier power spectra. The ratio between the mean spectra from two different sets can also be considered and tested (Barbier et al. 2006; Couteron et al. 2006; Diggle 1989). As the theoretical parametric distribution of this ratio is known (Fisher distribution), simple statistical tests are available to detect significant structural differences between the spectra and identify the range of scales at which the patterns may differ. Using this approach, Barbier et al. (2006) identified a large scale amplification of the signature (peak amplitude in the spectrum) of periodic gapped patterns after a prolonged drought episode in SW Niger.

A second approach for pattern comparison is based on the computation of the cross-periodogram, which can be understood as a scale- and orientation-specific partition of the covariance between the compared signal types. An advantage of this method is that it also quantifies phase shifts that may occur between the signals (Barbier et al. 2006). This property allowed Deblauwe et al. (2012) to quantify the upslope migration speed of banded patterns over large extents of semi-arid areas on three continents.

A third option uses ordination/classification techniques to systematically compare the structure of numerous image extracts, as is often necessary when dealing with earth observation datasets at landscape to regional scales (Barbier et al. 2006; Couteron 2002). To reduce the dimensionality of the dataset, a feature extraction (simplification) stage is generally required. For instance, if one is only interested in pattern scales (frequencies), and not in possible dominant orientations or other forms of anisotropy in the patterns, it is possible to average out the periodogram information across spatial orientations (Mugglestone and Renshaw 1998). This results in the computation of radially averaged spectra, which can then readily serve as an input for a Principal Component Analysis (PCA) to further decrease dimensionality. Applications over arid landscapes showed that the first two axes of such a PCA generally provide a synthetic ordering of image extracts. These principal axes represent coarseness-fineness gradients congruent with the visual appraisal of experienced assessors (Couteron et al. 2006). The approach allows detecting spatially periodic patterns (singled out along the second PCA axis), which often are the signatures of self-organisation features (Couteron 2002; Barbier et al. 2010).

Fourier methods are sufficiently flexible to allow tailor-made analyses to characterise and classify specific pattern properties. In Sudan, for example, Deblauwe et al. (2011, see also Chap. 13), analysed the modulation of self-organised periodic vegetation patterns along aridity and slope gradients in terms of periodicity, symmetry and wavelength (see Fig. 8.8). The figure illustrates Fourier spectral signatures of typical periodic vegetation structures which are often found in semi-arid areas. The first column illustrates Spot panchromatic satellite views of banded, spotted, labyrinthine and gapped patterns, in which vegetation appears in dark and bare soil in brighter tones. These structures are all regular (spatially periodic) but vary in terms of symmetry and wavelength. These variations are apparent in the 2D periodograms (power spectra) in the second column. Here, pattern properties are represented in the frequency space (units of Cycles km^{-1}) as the amount of variance accounted for by frequency bins along the Cartesian axes. The ‘zero frequency’ is located in the middle. The sign of frequency (direction of wave travel) is not important here, and the 2D periodogram is thus symmetric. In the third and fourth columns, the same information is simplified to place the emphasis either on frequencies (averaging periodogram values across orientations: radially averaged r-spectrum) or on the orientation (averaging periodogram values across frequencies: θ-spectrum).

Tools for pattern characterisation, as with the example in Fig. 8.8, are developing rapidly at the moment. The next step in pattern research will include both the qualitative comparisons between empirical data and theoretical self-organising vegetation models and the quantitative validation of modelled vegetation patterns with field or remotely sensed data, as models should be able to reproduce not only the right sequence of pattern morphologies, but also their quantitative features (see Chap. 7).

9.4 Quantification of Connectivity Patterns

The spatial formation of vegetation and the degree of its fragmentation or connectivity directly affect runoff generation. The efficiency of an ecosystem in retaining water, as well as sediment fluxes generated during rainstorm events, becomes smaller as the density and cover of perennial vegetation patches decreases and the grain size of the vegetation pattern increases (Chirino et al. 2006; Boer and Puigdefabregas 2005). Whereas the vegetated patches act as the sink for water and nutrient fluxes, the fragmentation of the vegetation or the connectivity of the bare interpatches can be used as an indicator of runoff and erosion generation.

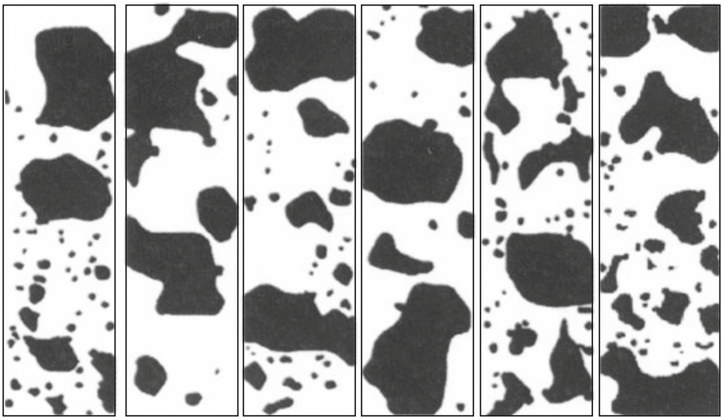
In order to quantify the effects of connectivity patterns on runoff, both at the plot and catchment scale, two indicators are introduced here: the Directional Leakiness Index and Flowlength, a spatial metric that quantifies the connectivity of runoff source areas (Mayor et al. 2008).

The *Directional Leakiness Index* (DLI) after Ludwig et al. (2002) describes how well a landscape functions to retain or leak runoff and soil resources in the direction of dominant flow down the hillslope. The index characterises maps with pixels that are classified as being either flow-obstructing patches (vegetated patches) or non-obstructing patches (bare interrill areas). The DLI is calculated for a grid with regular, rectangular cells, where each pixel on the grid is classified as being patch or non-patch. The index assesses the leakiness of patch patterns by integrating patch measures, comprising patch number, size, shape, arrangement and orientation into a simple indicator of landscape function. The leakiness index ranges from zero (no leakiness, ergo no connectivity) to one (system is totally leaky and therefore totally connected).

The *Flowlength metric* by Mayor et al. (2008) is calculated as the average of the runoff pathway lengths from all the cells in a raster-based map of a hillslope. A higher value of the metric indicates a higher hydrologic connectivity of runoff source areas. The metric is based on the assumption that the majority of bare soil areas behave as sources of runoff and sediments that are trapped by vegetation patches and topographic sinks further downslope on the runoff pathway. The potential length of the runoff path from each cell is calculated using a single flow direction algorithm that determines the gradient direction from a digital elevation model overlaid by a binary map with bare soil pixels classified as runoff sources and vegetation pixels classified as runoff sinks.

The Directional Leakiness Index DLI is a qualitative indicator related to the potential runoff generation capacity, whereas the Flowpath metric is a quantitative indicator related to the accumulated water amount in each cell. Both indices combine the information on the spatial configuration of the vegetation cover with the topography and thus improve the assessment of potential water flow connectivity in comparison to other spatial pattern metrics such as the lacunarity, proximity and connectance indices (e.g. Plotnick et al. 1993), which are based solely on plant cover data normally employed in landscape assessment.

These two indices were employed to assess the connectivity features in regard to water and sediment transfer of a typical semi-arid Mediterranean shrub-grass landscape, the Ventós experimental area in Spain (Bellot et al. 1999). Vegetation in the Ventós experimental area is an interspersed mosaic of open S. tenacissima steppes and dwarf shrublands (*Quercus coccifera*, *Pistacia lentiscus*) and scrub (*Globularia alypum*). This type of plant community is widely distributed within the semiarid and dry areas of the Mediterranean basin (Le Houèrou 2001) and derives from grasslands, as well as woodlands that have been subjected to long-term exploitation and degradation by human activities such as fibre and wood harvesting, grazing, and repeated burning (Barber et al. 1997; Puigdefábregas and Mendizabal 1998). The vegetation cover (around 40 %) is arranged in vegetated patches of one or several plant individuals and species, separated by interpatches of bare soil.



	P5	P1	P6	P2	P4	P3
DLI:	0.575	0.520	0.429	0.418	0.407	0.295
Patch density:	3.8	1.0	2.4	1.3	3.5	3.5
Functional diversity:	0.832	0.591	0.828	0.807	0.816	0.787

Fig. 9.2 Maps of vegetation patches and interpatches in the experimental plots of open steppes (Stippa tenacissima) arranged from *higher* to *lower* Directional Leakiness Index (DLI) values, i.e. *higher* to *lower* leakiness (*left* to *right*). The *black* patches are vegetation patches (Source: Bautista et al. (2007), reproduced by permission)

The interpatch areas have a high cover of rock fragments, as well as physical and biological soil crusts dominated by cyanobacteria. To quantify the effects of interpatch connectivity, the spatial distribution of perennial vegetation was classified at plot and basin scale (Fig. 9.2). At the basin scale, vegetation patterns were derived from aerial photography of high resolution; at the plot scale, the canopies of plant patches were surveyed directly in the field. The plants, stones, physical soil crust and outcrop cover were measured on several runoff plots (8 × 2 m) using the point-intercept method (point measurements of these variables were made every 10 cm along two parallel transects on each plot). Patch cover, patch density and the interpatch areas directly connected to the trough were directly derived from the vegetation maps. Five functional types (based on the five vegetation cover types and morphological characteristics that affect rainfall partitioning, infiltration, and overland flow) were recognized: shrubs, sub-shrubs, tussock grasses, sod-forming short grasses and chamaephytes.

For a 45-month period, runoff and sediment yield were measured on the plots for a total of 55 storm events. These events were used to validate the usefulness of the directional leakiness index (DLI) to assess the impact of connectivity patterns on runoff production (Bautista et al. 2007). For the analysis, it was necessary to take into account the size and intensity of the storm events and the proximity of the interpatch areas to the output channel of the catchment.

The Flowlength metric was evaluated for its use as a quantitative connectivity measure (Mayor et al. 2008). The metric Flowlength classified the runoff plots from

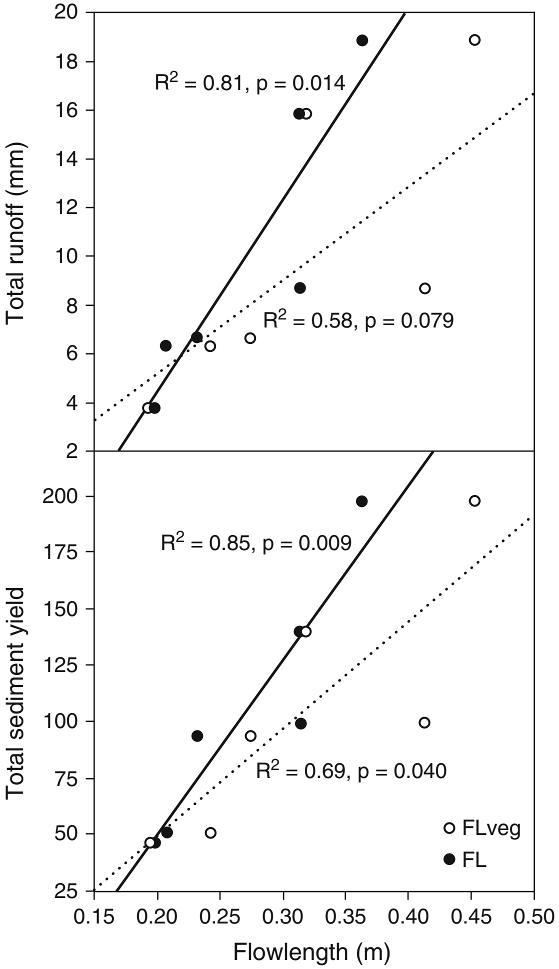


Fig. 9.3 Total runoff and total sediment yield in the experimental plots *versus* Flowlength. Flowlength values were calculated considering both the vegetation pattern and the microtopography (FL) and only considering the vegetation pattern (FLveg) (From: Mayor et al. (2008). Copyright American Geophysical Union. Reproduced and modified by permission of American Geophysical Union)

low to high connectivity (of bare soil). It was found that the connectivity increased with the grain size of the bare soil spatial pattern, which was greater in the plots where the vegetation cover was distributed in a lower number of larger patches. The Flowlength metric was linearly and positively related to both the total plot runoff and sediment yield from the experimental plots: plots with larger flowlengths produced a linearly larger runoff and sediment yield than the ones with smaller flowlengths (Fig. 9.3).

The results in the tussock grass sub-basin in the Ventós area suggested that during high-intensity storms (with rainfall intensities of more than 60–70 mm/h), interpatch connectivity as derived from the DLI index is less relevant for runoff and sediment fluxes as both bare soil and vegetation patches serve as runoff sources. However, for storms with smaller intensities (less than 60 mm/h), only the bare soil interpatches function as runoff sources and the patches act as sinks for rain and runon water – the interpatch connectivity is determinant. In addition, the proximity of the interpatch areas to the bottom output channel influences the amount of runoff that leaves the catchment: the closer they are, the smaller was the time for runon infiltration.

As with the DLI index, the size of the rainstorm events played a significant role in explaining the relation between runoff and connectivity. The strongest relationships between Flowlength and runoff were observed for moderate and high-magnitude storms with amounts larger than 20 mm; no relationship existed for small rainfall events <10 mm. Mayor et al. (2008) concluded that the Flowlength metric has the potential to serve as a surrogate for the hydrological functioning of ecosystems with patchy vegetation.

Both connectivity indices distinguished varying degrees of connectivity of vegetation as a function of topography and resulting flowpaths. They enabled linking the patterning of vegetation to lateral transport processes of water and soil resources and thus give an insight into the flow dynamics of the system.

9.5 Confronting Data with Models Through Pattern-Orientated Modelling

A variety of (geo)statistical, spectral and indicator methods has been presented here that is available to assess the intrinsic patterns of vegetation, water and soil resources of heterogeneous environments in dryland settings. Often, it is difficult to identify patterns clearly and even if patterns *are* clearly identified, these tools tend to focus on explaining one single attribute of a pattern: in focusing on the patterning of vegetation cover but not linking it to the spatial distribution of, for instance, soil properties, root growth, soil moisture or plant-available nutrient content. Several studies analysed the multivariate geostatistical relationships between vegetation and soil properties in dryland settings (e.g. Maestre and Cortina 2005; Bedford and Small 2008), however, the hitherto derived functional relationships of the studied patterns are still very limited. If we now recognise a dryland as a self-organising system containing nonlinear biotic and abiotic feedback mechanisms, we require multi-pattern analysis tools that can take these interactions into account. As empirical dryland data are notoriously limited – particularly at multiple spatial and temporal scales – one may have to resort to numerical modelling for the generation of multivariate vegetation and resource patterns. In this vein, Chap. 7 introduced complex process models for vegetation, soil and water dynamics that aim to use interacting ecogeomorphic processes to generate and ultimately explain patterns of plant and soil distributions.

The next step is to combine our efforts (of empiricists and modellers) by iteratively comparing empirical and model-output patterns at multiple scales and levels of organization and for multiple models to identify most appropriate models. What can we do to avoid focussing on possibly wrong explanations of single patterns? How can we organize model design, testing, and field data assessment to increase the chance to truly understand how an ecological or eco-geomorphic system works? Pattern-oriented modelling (POM) has been suggested in ecology in the mid-1990s as a solution (Grimm 1994; Grimm et al. 1996). POM is a general strategy of model building and use, not a specific technique. Its core idea is *multi-criteria assessment* of models: instead of designing a model that reproduces only a single pattern, it is designed to reproduce a set of patterns – typically three to five. Instead of focussing only on single striking patterns, one also includes weaker patterns that contain less information: for example in size structure, spatial distribution, response to disturbance events and changes in individual behaviour. Experience shows that making a model reproduce several "weak" patterns simultaneously more readily leads to structurally realistic models than if one focuses on strong patterns.

POM has been developed over the last decade (Railsback 2001; Wiegand et al. 2003, 2004; Grimm and Railsback 2005, 2012; Railsback and Grimm 2012) and is currently increasingly acknowledged and used also in other disciplines, for example social sciences (Janssen et al. 2009), biomedical research (Dong et al. 2010), geography, and oceanography (Cury et al. 2008). However, its application in dryland assessments is as yet fairly uncommon (but see Arnold et al. 2009 and Cipriotti et al. 2012). The following sums up the three elements of the POM methodology:

1. The selection of a model structure that, in principle, allows for patterns considered essential for characterising a system, also to emerge in the model. If in a semi-arid system vegetation shows a characteristic spatial pattern, obviously the model should be spatially explicit. If vertical layering in a forest is considered essential for capturing vertical competition for light in forests, the model should include the vertical dimension. If size and spatial distributions of plants differ between high and low densities, size should be included as a state variable. If response to extreme environmental events differs among regions with different resource levels, resource levels and dynamics need to be considered in the model. The choice of patterns for POM is experimental: experts might not agree on the importance and significance of different patterns; certain patterns might contain less information than previously assumed; new patterns might be revealed as experience is gained with the model. This is what all modelling is about: one explores, in an iterative process, the consequences of our assumptions and perceptions.
2. The derivation of alternative formulations for the model's key processes to check how well these alternative submodels will reproduce entire sets of observed patterns (Railsback 2001). The set of alternative models should include "null models", that are crude oversimplifications, to learn about the sensitivity of the model and its key submodels. Null models can be linear instead of nonlinear, constant instead of being adaptive, random instead of being mechanistic. This element of POM fully corresponds to the research programme of "strong inference" (Platt 1964).
3. The usage of multiple patterns to determine inversely entire sets of unknown parameters. This type of inverse modelling might be routine in hydrology and

other disciplines, that tend to focus more on numerical solutions, but in ecology this approach is relatively new (Wiegand et al. 2004). The protocol for this is: define the criteria for the model output that matches specific patterns; take, for each parameter that needs to be determined inversely, a certain number of samples of its meaningful range; create – using sampling algorithms like the Latin hypercube sampling – a large and representative set of possible parameter combinations; run the model for all parameter sets and check whether a certain combination fulfills all multiple patterns simultaneously. This procedure leads to a largely reduced number of parameter sets, which also considerably reduces uncertainty in model output (Wiegand et al. 2004; Cipriotti et al. 2012).

The key to POM is the multi-criteria perception of complex systems – in this case, ecogeomorphic systems thus confronting dryland models not just with data, but with multiple patterns, observed preferably at different scales and hierarchical levels. In complex systems, patterns at the system level emerge from the behaviour of the system's building blocks, e.g. individual organisms or small spatial units. On the other hand, the building block's behaviour is constrained by system-level properties. POM is a modelling strategy that acknowledges this mutual and often inseparable relation of processes at a continuum of scales. If POM is used as a 'multi-scope' approach (Grimm and Railsback 2012), cross- and interdisciplinary approaches, like ecohydrology and ecogeomorphology, will advance faster and develop more robust theory on the linkages of patterns and processes.

The presented (geo)statistical, multi-spectral and indicator techniques in the previous sections allowed new insights into pattern-process linkages of self-organised, (dis)connected or (ir)regular patterns. In combination with pattern-oriented modelling and process-based modelling (as defined in Chap. 7), these techniques should be developed further to retrieve more information from the often time and space limited dryland data sets, thus enabling the assessment of patterns which are generated by tightly coupled feedbacks between biotic and abiotic elements.

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Chapter 10
Uncertainty Assessment

Richard E. Brazier, Tobias Krueger, and John Wainwright

Abstract A number of traditional concerns as well as outside pressures on scientific understanding have led to the underplaying of uncertainty. Land-degradation studies are by no means alone in assuming that if the problem of uncertainty is ignored, it will go away. We demonstrate that such a head-in-the-sand approach is fallacious, as uncertainty underpins all our scientific activity. Field measurements and empirical observations are no less exempt than complicated numerical models. Uncertainty can be distinguished as being aleatory, or due to inherent variability, or epistemic, as a result of uncertain knowledge, although in reality both types are intimately related. Model parameters reflect the underpinning conceptual models in a discipline, and as those conceptual models change, parameter measurements may also need to change. Taking account of all of the sources of variability in measurement is critical, though, in ensuring that we do not reject models for the wrong reasons. A structure is presented for addressing uncertainty propagation in models using intervals, fuzzy membership functions and probability distributions in conjunction with stochastic simulation. The effects of model structural uncertainty are considered within a variety of Bayesian frameworks, and their relative strengths and weaknesses addressed. All aspects of uncertainty must be considered if we are to develop robust models of land degradation that incorporate ecogeomorphic feedbacks and human activity.

R.E. Brazier
Geography, College of Life and Environmental Sciences, University of Exeter,
EX4 4RJ Exeter, UK
e-mail: r.e.brazier@exeter.ac.uk

T. Krueger
School of Environmental Sciences, University of East Anglia, Norwich NR4 7TJ, UK
e-mail: t.krueger@uea.ac.uk

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

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10.1 Introduction

Until recent decades, especially in western scientific traditions, uncertainty was seen as a fundamental weakness. As research became more problem-orientated, the internal pressures were compounded by external – social and political – ones to ensure that only the “right” answer was presented. In this chapter, we will discuss the different types of uncertainty inherent in any scientific endeavour, and demonstrate how ignoring these different uncertainties is far more likely to lead us to a wrong answer. Alternatively, we may optimize our results to produce what appears to be the “right” answer in a particular context, but which turns out to lead us to make poor, sometimes catastrophic, decisions in other contexts because the underlying precepts are incorrect. Recognizing that conceptual and numerical models are always incorrect is of fundamental importance in improving our understanding of any scientific problem. Because measurements rest on implicit conceptual models, it thus follows that the issue of model and data uncertainty are interconnected. For this reason, the linkage of model and data uncertainty must be carried out in an iterative manner (e.g. Jakeman et al. 2006; Mulligan and Wainwright 2013). In this chapter we thus consider the different sorts of uncertainty and what happens when we iterate model development to confront data with models and models with data.

10.2 Uncertainty of Field Data

Very few studies of the uncertainty surrounding understanding of land degradation in drylands have been undertaken. Although this problem is certainly not restricted to drylands research, it is clear that progress must be made in understanding the limits of our empirical understanding in these environments, prior to (or at least alongside) establishing where our research priorities lie for the future.

10.2.1 Types of Uncertainty

There are underlying uncertainties in the descriptive equations of land-degradation processes, most of which, by necessity are empirically based; there are uncertainties in the parameter values or characteristics that are measured (observed) in the field, which differ with every spatial (and temporal) scale at which they are collected; and there are measurement uncertainties, associated with all field equipment that is used to quantify fluxes or measure processes that are used to describe land degradation or to evaluate models which predict elements of land degradation. Some of these uncertainties, for example those caused by inherent variability, are *aleatory* (statistical uncertainties) while others are *epistemic* (knowledge uncertainties) (Beven and Brazier 2010). However, as we will see, the quantification of these uncertainties rarely takes place. The field measurement or observation is often *believed*, which can provide a misleading sense of *quality* of understanding. In turn this may not provide models with fair datasets against which they can be evaluated. It is therefore likely that land-degradation scientists are uncertain about the types of uncertainty that exist around field-based understanding, or perhaps reluctant to confront these uncertainties. Failure to evaluate and communicate these uncertainties can have significant implications when the science is used by environmental managers and policy makers. This section aims to clarify what is meant when we talk about uncertainties in field data and therefore provide a basis from which these uncertainties can be incorporated into improved (holistic) assessments of our understanding of drylands.

10.2.2 Uncertain Process Understanding

The way in which field uncertainties are understood (or not), is guided by the development of conceptual models of the landscape, which describe the process assemblage that we believe to be the 'correct' description of that landscape and therefore direct us to make (and interpret) observations within the landscape to improve our understanding. As has been illustrated in Sect. 5.5, water erosion plays a critical role in redistributing sediment around the landscape and in heavily degrading systems, removing significant quantities of sediment (topsoil) from the system. Thus, it is interesting to consider the conceptual model proposed by Bennett (1974) which outlines an approach to understand soil erosion by water. As is discussed in detail by Wainwright et al. (2008), the central tenet of Bennett's conceptualization of soil erosion is the process-based description of how, once entrained, sediment travels downslope/stream *in suspension*. Numerous subsequent soil-erosion models have been developed on the assumption that hillslope soil erosion is dominated by sediment that travels in suspension including WEPP (Laflen et al. 1991; Risse et al. 1995; Zhang et al. 1996); KINEROS/KINEROS2 (Smith et al. 1995; Woolhiser et al. 1990; Goodrich et al. 2002) and EUROSEM (Morgan et al. 1998). However, Wainwright et al. (2008) demonstrate that, in fact, hillslope soil erosion is rarely dominated by sediment travelling in suspension, and thus the underlying process understanding represented by most modern erosion models is flawed. Furthermore, when field data, from hillslope plots or headwater catchments, for example, are collected to describe water erosion, the assumption that they also describe suspended sediment flux is also (at least partially) flawed.

Here is an example of uncertain process understanding. The work of Wainwright and Thornes (1991), Parsons et al. (1993, 1998), Rejman et al. (1999) and Cooper et al. (2012) illustrates, via empirical observations, that erosion of particles by unconcentrated flow occurs predominately as bedload movement or saltation (rolling, sliding or short hops), yet the majority of soil-erosion studies do not consider this to be the case. To overcome this problem what is required is, ironically, a return to the work of Bennett, who stated that further work was needed to understand the process basis of his assumption regarding suspended sediment.

10.2.3 Uncertain Parameters

When making observations of the properties or characteristics of drylands in the field, it is not possible to measure everything, nor will it ever be (and even if we could, we would struggle to interpret the essentially infinite amount of information produced: Mulligan and Wainwright 2013). Thus, we are forced to make decisions about the appropriate scale of measurement to characterize the inherent variability of a system. Such a choice is difficult. No single scale of observation will be appropriate to quantify the variability of all parameters of interest. Moreover, the scale at which measurements need to be taken may not be known until those measurements have already been taken! We are confronted with a paradox. We cannot know how certain our description of parameters may be, without first making very detailed measurements to describe this uncertainty.

If we consider the problem in space, numerous techniques have been used to characterize landscape properties from single point measurements within a 5-km grid at the national scale (Bellamy et al. 2005), to highly detailed sampling of plots (Turnbull et al. 2010a, b; Puttock et al. 2012). The latter approach demonstrates the need to understand the heterogeneity of a system and therefore the uncertainty, which may surround point measurements of parameters within that system. Turnbull et al. (2010a, b) use a geostatistical approach, based on the construction of semi-variograms, to describe the biotic and abiotic changes in ecosystem structure over a shrub-encroachment gradient in the Chihuahuan Desert, USA. In order to build semi-variograms, 90 sampling points were chosen within an area of 300 m^2, employing a nested approach, which permitted a very detailed description of the spatial variability of soil properties such as; bulk density, total nitrogen content, carbon: nitrogen ratio and soil organic matter content. The data showed that standard errors around mean values for each property were ±2.5, ±8, ±10 and ±3 % respectively. Thus, within a 300-m^2 area, there is significant uncertainty in our measurement of soil properties.

Though it would be churlish to suggest that all land-degradation studies focus on such detailed measurements, the lesson here is that coarser approaches to describe parameters which may exert a strong control over the structure and function of a dryland ecosystem, will not always provide a reasonable description of the properties of that system. If a detailed characterization of parameters is made, at least it is possible to be clear about how certain our understanding is. As such, we may also learn about the magnitude of variability that exists in other landscapes or in existing landscapes that may be subject to future degradation.

10.2.4 Measurement Uncertainties

Whilst it is intuitively clear that all field measurements must incorporate some level of uncertainty, it is rare in land-degradation studies (or indeed most empirically based Earth or environmental sciences) that these uncertainties are considered explicitly. A fundamental problem therefore emerges, as field measurements appear to be more *believable* than their laboratory counterparts, which are most often presented with at least some illustration of error.

One example of this problem is the persistent use of single-line hydrographs in the research literature. Typically, equipment is used to measure either flow depth or flow discharge at-a-point in the landscape on discrete time steps, which are then plotted as hydrographs (in the case of the latter) or are converted from stage (depth) to discharge employing some form of rating curve, in the case of the former. As direct measurements of flow depth are relatively simple, this approach is most common. However, discharge may vary significantly for any given flow depth, due to turbulence in the flow, backwater effects, varying viscosities of flow (especially when sediment-laden), or transport of materials such as woody debris. Indeed, the very processes of erosion we are concerned with here act to remodel the flow channel constantly, leading to substantial changes (and uncertainty) in any stage-discharge relationship. The uncertainty of the stage-discharge relationship will also increase when extrapolating to high flows beyond the range of original measurements, where taking reliable measurements is near-impossible yet predictions are most important for understanding erosion processes (see McMillan et al. 2012, for a review of stage-discharge uncertainty studies[1]). Therefore, recent work in drylands by Turnbull et al. (2010a, b) and in temperate environments (Bilotta et al. 2010; Huard and Mailhot 2008; Krueger et al. 2010; Liu et al. 2009; McMillan et al. 2010; Pappenberger et al. 2006; Westerberg et al. 2011) has employed techniques to quantify the range of discharges that are observed at each stage and to construct uncertain rating curves which describe the variance within the measurements for each flow depth (Fig. 10.1).

The resultant hydrographs offer a more transparent representation of the quality of the field measurement and the uncertainty that may be associated with it. Of course, variables that are often used to describe rates of land degradation – fluxes of sediment or rates of erosion – depend upon the accurate representation of the flow that controls them. Taking these uncertain hydrographs a step further also allows uncertain representations of sediment fluxes and then finally sediment budgets from landscapes to be calculated (see Sect. 10.3.1).

As mentioned above, a final reason for representing measurement uncertainties explicitly is that empirical data are increasingly in demand for model-evaluation purposes. If the goal of the modeller is to make predictions with constrained uncertainties, then providing empirical datasets that present a *fair* picture of their associated uncertainties is critical to model development. We run the risk of rejecting perfectly good models (i.e. those that could describe our uncertain observations), unless we describe how uncertain our field observations really are.

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[1] See also the community uncertainty inventory at http://www.experimental-hydrology.net/wiki/index.php?title=Category:Uncertainty.

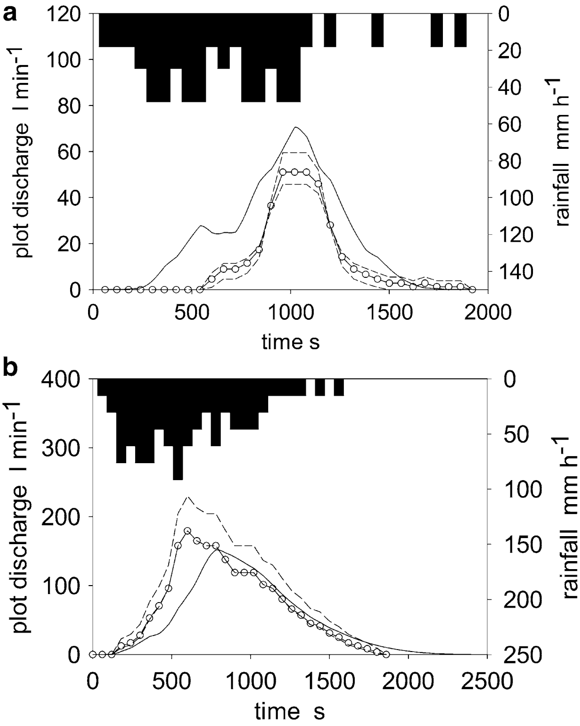

Fig. 10.1 Comparison of monitored and modelled hydrographs for semi-arid plots in New Mexico (After Turnbull et al. 2010a, b). The *solid line with dots* shows the "standard" approach of producing a single hydrograph line based on the calibration of depth data using a rating curve. The *dashed lines* are the 95 % confidence limits of this estimate. In hydrograph (**a**), the modelled hydrograph (*solid line*) falls completely outwith the uncertainty bounds except for at the very end of the falling limb, so might be considered a poor representation of observed runoff; in (**b**), the modelled hydrograph underestimates the rising limb, but accurately captures most of the falling limb (Source: redrawn, based on data originally published in Turnbull et al. (2010a, b))

10.2.5 Issues of Correlation

Another issue warranting further discussion is that of correlations between data uncertainties. For example, the same measurement technique might be used to infer two different quantities, say soil properties and vegetation properties, using remotely sensed spectral reflectance (e.g. Baret and Guyot 1991). Similarly, remote sensing characterizations of vegetation will be spatially autocorrelated because of pixel mixing as parts of the same vegetation element appear in two or more adjacent pixels because of their size in relation to image resolution. Such autocorrelation will significantly affect the testing of vegetation pattern when compared to a model that is able to have a more binary representation of vegetation presence or absence. Each of these *sets* of properties will consist of different *actual* properties again. Translating the spectral signal into the properties of interest based on field-based observations of those properties is a multivariate problem in which variables will, in general, be correlated. It follows that the *marginal* uncertainties of each individual property cannot be propagated independently through a model that also requires input of one or more of the other correlated properties. Of the prevailing uncertainty models, intervals and fuzzy membership functions do not allow correlations to be represented with ease while the repertoire of multivariate probability distributions is limited. In case of correlations it might thus be easier, using the same example, to treat the translation of the spectral signal into the properties of interest as part of the model, e.g. as part of a hierarchical Bayesian framework (see Sect. 10.4). The uncertainties of the spectral signal and the field-based observations then enter the model independently while the correlations are treated as part of the model structure. Similar issues of correlation can arise from the scaling of landscape characteristics, in which case the scaling relationships themselves are best part of an integrated or hierarchical model (e.g. Costanza and Maxwell 1994; Zhang et al. 2002). Unless we take account of the process basis underlying measurements at different scales, it is possible to introduce significant errors that are correlated. For example, Lappi (2005) has demonstrated that when trying to evaluate local tree density, competition between plants for resources will mean that errors are correlated with local density and are therefore a function of the plot size used to estimate values. These locally correlated errors will have significant consequences for evaluating vegetation patterns. In the case of soil-erosion rates, Parsons et al. (2004, 2006) have demonstrated that the measured rate is a function of plot size *ceteris paribus*, and thus errors will also be correlated if based on measurements at different scales (e.g. plots of different scales, or small-catchment studies – see Chap. 5).

10.2.6 Synopsis

It must be core to future research that process uncertainty is constantly revisited – our process understanding must be repeatedly challenged until we are happy that we can adequately describe the appropriate processes that dominate land degradation. Although we are generally not confronted with deterministic problems in drylands (see Chaps. 3 and 6), we must strive to be clear about how uncertain our process understanding is. In fact, it is precisely because the relationships between structure and function of drylands are non-deterministic, and indeed are highly non-linear, that illustration of process uncertainty must be made more explicit than is currently the case.

One way of recognizing our lack of understanding is to quantify the mismatch between our model predictions (even if just conceptual) and measurements. How large the uncertainties are in our description of the spatial variability of soil characteristics (for example), surely tells us how poor our process-understanding is. Therefore, clear descriptions of the heterogeneity of parameters may be a useful start in learning what we need to learn about the functions that bring about spatial structure (and thus pattern) in drylands.

Finally, in certain disciplines, acceptance of measurement uncertainty is common. If researchers in land degradation are content to continue without assessing measurement uncertainties, it is hard to see how honest evaluations of future models of degrading landscapes can be made. Improvements to such models will depend upon the *quality* of our empirical understanding, central to which is the illustration of measurement uncertainties.

10.3 Uncertainty of Model Data

As data enter models, it is sensible to analyze the influence of data uncertainties on model results to be able to ensure model robustness (*sensitivity analysis*). Note that small data uncertainties can have a large effect and large data uncertainties can have a small effect, depending on the specific non-linearity of a model structure, so knowing the magnitude of field-data uncertainty does not *automatically* allow inference of its effect on the model (e.g. Pacala et al. 1996; Klepper 1997). In case of large effects, data uncertainties are best propagated through to the results when applying models to provide a statement of scientific confidence, without which the results have little meaning. *Uncertainty propagation* is but one part of an overall *uncertainty analysis*, which also considers model structural uncertainty as well as parameter uncertainty resulting from calibration (see Sect. 10.4).

10.3.1 Uncertainty Propagation

A fundamental issue to consider is the *nature* of the data uncertainty, i.e. the *uncertainty model*, as this will limit the choice of propagation techniques. The prevailing uncertainty models are, in increasing order of specificity: intervals; fuzzy membership functions (or possibility distributions); and probability distributions. Higher-order uncertainties, i.e. uncertainty about the scope and formalization of uncertainty, may be considered. Theories of imprecise probability have been proposed to unify the aforementioned uncertainty models. Hall (2003) provides a good introduction. In some cases, the understanding of the relevant uncertainties might be insufficient to justify any of the above models, in which case the analyst may specify a limited number of data *scenarios* with no pretence of completeness. This approach should be viewed as a preliminary *scoping* of propagated uncertainty, yet more informative than neglecting data uncertainty altogether. See Krueger et al. (2010) for a propagation of rainfall scenarios through an ensemble of hydrological models that shows which models could be rejected under all simulation conditions and the importance of understanding variability in replicate plots.

Intervals arise typically from ranges of instrument precision, although certain manufacturers might use a range to indicate standard deviation widths around the mean of an assumed normal distribution. In order to avoid confusion, unequivocal

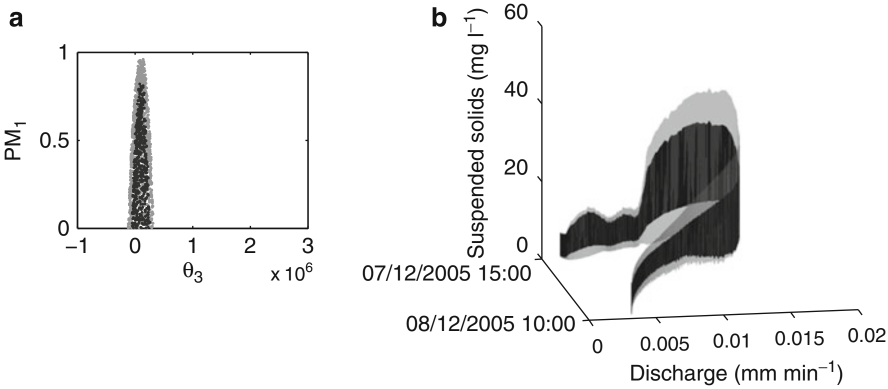

Fig. 10.2 Contribution of discharge input uncertainty to the estimated overall model uncertainty: (**a**) Model performance (PM$_1$) against values of parameter θ_3. Model realizations that include discharge uncertainty intervals are shown in *light*; those that do not (each interval was replaced with its centre) are shown in *dark*. (**b**) Hysteresis uncertainty bounds, calculated using the centre of each discharge interval and the minima/maxima of all simulations above an initial performance threshold of $PM_1 = 0.6$. Uncertainty bounds generated with discharge uncertainty are shown in *light*; those generated without are shown in *dark* (Source: Krueger et al. (2009); used with permission from *Journal of Environmental Quality*)

documentation is encouraged (McMillan et al. 2012). Intervals may also be used to indicate an analyst's low confidence about an uncertainty *distribution* while they are confident to provide a *range* encompassing that distribution. In this case, an interval may be reinterpreted as a rectangular fuzzy membership function (see below). Note that statisticians feel, in principle, uncomfortable with the crisp boundaries of intervals arguing that such extreme changes of the confidence measure are unrealistic. However, in practice interval boundaries may well reflect realistic physical limits to a quantity or may be set so far apart as to render the implicit assumptions insignificant, and certainly no worse than any imposed distributional assumptions. For simple models, independent intervals may be propagated analytically by *interval arithmetic*, while complex models require stochastic simulation (see Sect. 10.3.2). For an example derivation of uncertainty intervals in the context of stage-discharge rating curves see Krueger et al. (2010). For propagation of the resultant discharge intervals through empirical sediment models see Fig. 10.2 (Krueger et al. 2009) and Bilotta et al. (2010).

Fuzzy membership functions are typically used if the notion of specificity of probability distributions appears inappropriate or if qualitative judgements of uncertainty or vague natural language predicates are to be modelled (the original scope of fuzzy logic: Zadeh 1965). The interpretation of fuzzy membership as a *possibility measure* (e.g. Hall 2003) reflects this relaxed specificity. The membership functions, however, can be as complex as desired, ranging from rectangular, over triangular and trapezoidal, to irregularly shaped fuzzy numbers. Similar to intervals, independent fuzzy membership functions may be propagated through simple models

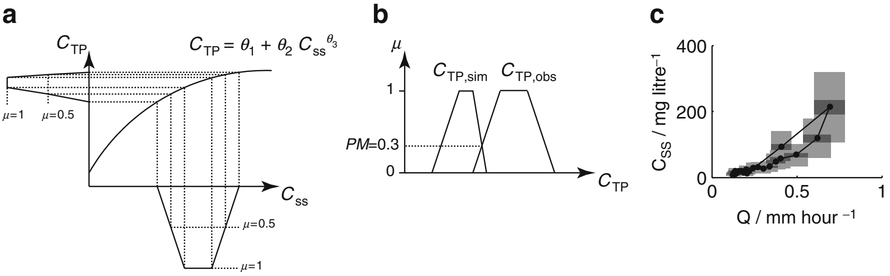

Fig. 10.3 (**a**) Propagation of fuzzy input data through an empirical sediment-transfer model by series of interval arithmetic operations for specific membership levels μ. A discretization of $\Delta\mu = 0.5$ is shown here. (**b**) Comparison of simulated (*sim*) and observed (*obs*) fuzzy numbers for this case. The level of agreement (performance measure, *PM*) is defined as μ at the point of intersection between the two membership functions. (**c**) Discharge-sediment curve estimated based on this approach. The *solid line* through the data centroids suggests a hysteretic relationship, but the data uncertainties do not support this interpretation (After Krueger et al. 2012a, b)

by a series of interval arithmetic operations at discrete membership levels, while complex models require stochastic simulation (see Sect. 10.3.2). Krueger et al. (2012a, b) provide an example derivation of trapezoidal fuzzy membership functions of flow-weighted mean sediment concentration and propagation through an empirical phosphorus model (Fig. 10.3). Torri et al. (1997) derive membership functions for soil erodibility for different combinations of mean particle size, clay content and organic matter content. An example of how the approach can be used to improve model performance against a specific dataset is provided by Tran et al. (2002).

Probability distributions are widely regarded as the most *coherent* model of uncertainty. However they will, in principle, overstate the confidence about uncertainty in cases where there is insufficient information about uncertainty such as that obtained from repeated measurements or controls. Simple models may be convoluted with data probability distribution functions (pdfs) analytically. For mildly complex models and the special case of independent and normally distributed uncertainties there is the method of Gaussian error propagation based on Taylor series approximation (e.g. Lo 2005; Goldstein et al. 2012). For complex models, as is likely to be the case for most ecogeomorphic models (see Chap. 7), stochastic simulation is again the only option.

10.3.2 *Stochastic Simulation*

Stochastic simulation is by far the most universal method of uncertainty propagation, yet is computationally expensive. The method relies on drawing random samples (*realizations*) from uncertainty ranges or distributions, whereby the chance of drawing a particular value is proportional to that value's probability or fuzzy membership degree, or is uniform in the case of intervals. All of these realizations are run through the model and the corresponding outputs combined statistically (see Fig. 8.3 for an example applied to parameterization of a runoff-erosion model, and Castrignano et al. 2008, for an example of an event-based erosion model). Technically, therefore, any uncertainty model is temporarily interpreted as a relative frequency, but reinterpreted as the original uncertainty model after propagation. Note for intervals the interest is only in the minima and maxima of output variables.

10.3.3 Inconsistency Issues

The fact that all uncertainty models are compatible *technically* is appealing but raises issues because mixing different uncertainty models in an overall uncertainty analysis will cause inconsistencies unless a generalized framework such as the Generalized Likelihood Uncertainty Estimation (GLUE) (Beven and Binley 1992; Brazier et al. 2000; Beven 2006) or a unifying theory of imprecise probabilities (see again the review by Hall 2003) is used. The reciprocal issue occurs when, for want of interpretability, all uncertainties are forced into a specific uncertainty model without being supported by the nature of and information on the uncertainties as observed in the field. From a practical perspective, i.e. the way the propagated uncertainties *look*, it might not make much difference which uncertainty model or unifying framework is used, as long as it is used consistently. However, such an assumption would have to be tested using a second-order sensitivity analysis. As always, a close collaboration between field experimentalists and uncertainty modellers, to use two coarse labels, can only help finding the appropriate uncertainty model that is consistent with both the field observations as well as the overall uncertainty analysis framework of a given model application.

10.4 Confronting Field and Model Uncertainty

The discussion above highlights the importance of understanding uncertainty in both field data and data used to drive models. There is a further source of uncertainty that it is fundamental to grasp – that of uncertainty of the model structure. Unless one takes a very naïve positivist view that the model structure one is working with is absolutely correct (see discussion in Pappenberger and Beven 2006), structural uncertainty is fundamental to the optimal use of data in both field and model settings. Indeed, uncertainty in the model structure may be the most critical control on propagation of uncertainty in model results. Beven (1996, 2006; Beven and Freer 2001) has argued that multiple model structures can lead to different parameter sets producing the same acceptable ("behavioural") output, a problem he calls equifinality, which leads to a problem of how to decide which of the multiple acceptable models is/are "right". Conversely, sensitivity to initial and boundary conditions and path dependency can lead to an acceptable model structure producing

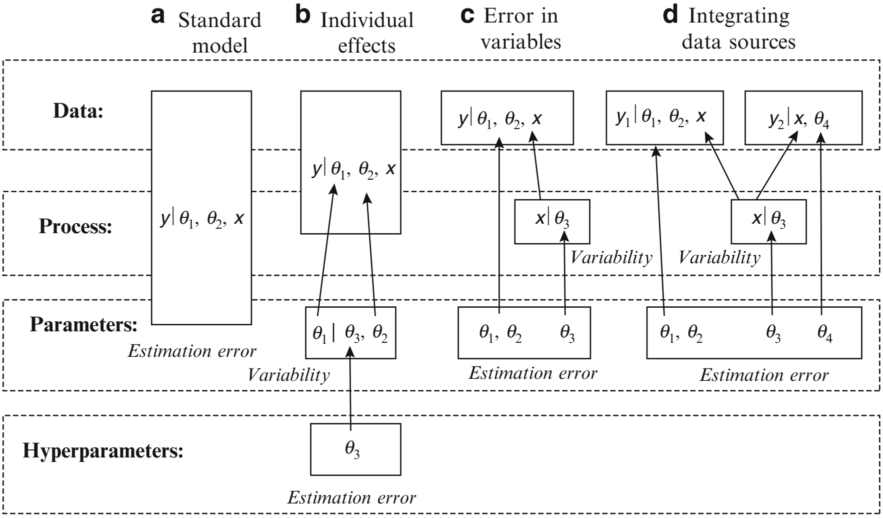

Fig. 10.4 Examples employing a Hierarchical Bayesian framework to evaluate belief in model inputs and outputs by breaking down the model into network components (Clark 2005). Although some components are known, many are unknown. (**a**) Shows the simple Bayesian approach to evaluating the effect of parameter estimation error on model output; (**b**) shows the effects of variability controlled by unobserved factors (hyperparameters), reflecting structure in space, time, or among individuals or groups; variability in a parameter θ_1 is carried out by conditioning on an additional parameter, θ_3, reflecting the fact that θ_1 is variable rather than just uncertain; (**c**) shows the effect of a hidden process controlling the model input x again controlled by an additional parameter, θ_3; and (**d**) the combined effects of variability and of integrating multiple sources of information to characterize the same process x, used as a model input (Source: Clark 2005; used with permission from John Wiley and Sons)

unacceptable (“wrong”) results simply as a result of uncertainty or variability in the inputs used (Beven and Brazier 2010). Simplistically, we can think of three levels at which we might be “right” or “wrong”: in the field measurements used to characterize the environment, in the model structure used to represent that environment, and in the model results used to demonstrate an understanding of or make predictions about that environment (the latter encompassing the former two and additional levels). More realistically, however, we are likely to have different degrees of confidence in the data and models at these different levels.

These degrees of confidence or belief in data and models have made Bayesian approaches particularly attractive to those trying to make uncertainty tangible. In general, Hierarchical Bayesian approaches can be used to unpick the different sources of uncertainty and their effects (Clark 2005; Fig. 10.4). In the simplest case, belief in model output is a function of uncertainties in model inputs and parameters, and their propagation through the model (Fig. 10.4a). As belief in the values of the parameters increases – e.g. by increasing sample size – the belief in the model output also increases. However, in more complex cases, this simple relationship does not hold. If a parameter varies according to a further parameter that is unobservable in the experimental design (what Bayesians call a hyperparameter: Clark 2005), then because it is unsampled, the increase in observations do not necessarily produce a more constrained confidence in the model results (Fig. 10.4b). Wilke (2003) demonstrates the effects of such hyperparameters on the estimation of invasive species using spatial population models. Similarly, in some cases the parameters may be observable, but the input variables may not be observable, or are measured with error (Fig. 10.4c). Although belief in the parameters converges with sample size, it does not for the input variables. In this context, consider the extent to which a simple input like precipitation – especially when incorporated in a spatial model – is subject to significant levels of error (e.g. Papamichail and Metaxa. 1996; Tsintikidis et al. 2002; Daly 2006). A final example is the case where multiple data sources are used to infer a process (Fig. 10.4d), in which case further parameters need to be introduced to describe the conditional independence of the different data sources.

Model structural error can also be treated in the form of hyperparameters. Kennedy and O'Hagan (2001) introduced a model inadequacy function whose hyperparameters are estimated jointly with the other parameters to account for the discrepancy between the "true" process and the model output with "true" inputs. The authors used a Gaussian process for this function, which is flexible, but emphasize other possible choices and the need for greater exploration of this aspect of modelling. Other model inadequacy functions have been proposed since. To cite a recent example, Reichert and Mieleitner (2009) described model structural error as a time-dependent stochastic process. While acknowledging its formal correctness, Beven (2005) criticized model structural error terms (and the Hierarchical Bayesian framework in general) for being overly optimistic about the *form* of the various error terms, which will rarely be known in practice (see also Huard and Mailhot 2008). An alternative to the model structural error term is Bayesian Model Averaging (Vrugt et al. 2006; Hsu et al. 2009) of competing model structures and its GLUE equivalent (Krueger et al. 2010). This approach is limited by our inability to define the full space of possible models *a priori*, hence the Bayesian learning paradigm will in practice have to be sequenced with classic model improvement steps. Despite its limitations, treating uncertainty in a coherent framework has significant advantages in that it allows effective syntheses of data, processes and theoretical understanding reflecting the complexity of the system (Clark and Gelfand 2006). In this way, hard-won data (and model building!) can be more effectively used in a way that is compatible with the behaviour of the complex systems that are evaluated in this volume.

A point of controversy is the definition of a likelihood function in Bayesian analysis. The GLUE methodology (Beven and Binley 1992; Brazier et al. 2000) was introduced allowing for "less formal" likelihood measures (i.e. those not based on probability theory), in particularly the inverse of the error variance and the Nash-Sutcliffe index in early applications, which served to expose the shortcomings of traditional model evaluation using these measures in hydrology. Mitchell et al. (2011) present one of the numerous examples of GLUE in this context, in a study of respiration of drought-stressed ponderosa pine in a semi-arid setting. Although they produce "behavioural" simulations that reproduce the respiration of the plants, these simulations often do not predict the underlying soil moisture. In this case, model structural concerns have been subsumed to those of model output; in other words, the right answer has been accepted for the wrong reason (Klemeš 1986). Mantovan and Todini (2006) criticized the use of less-formal likelihood measures for reducing the sort of information that can be extracted from the process and affecting the efficiency of the process. In particular, they suggest that these less-formal likelihood measures can lead to inconsistent results so that the estimated posterior distribution function does not necessarily converge asymptotically to the correct posterior distribution function. Results may also be inconsistent in that increasing sample size will not necessarily improve the results. Mantovan and Todini (2006) demonstrated these issues by application to a simple model where the parameters and uncertainties were known by specification. In reply, Beven et al. (2007) suggested that if the prior distribution of the error structure – as required in a formal Bayesian analysis – is mis-specified, then although the formal approach will converge to a result asymptotically, it can be demonstrated that it may be the wrong result. The less-formal approach tends to be centred on the correct result, but it is very "flat", or in other words significantly underestimates the true likelihood. They go on to suggest that less-formal approaches can be consistent (although the argument of Mantovan and Todini was actually that they cannot be *guaranteed* to be) and that formal Bayesian methods may be inconsistent because of the complexity of the information content of real data. The issue is still debated (Mantovan et al. 2007; Montanari 2005; Romanowicz and Beven 2006; Beven et al. 2008; Smith et al. 2008; Arhonditsis et al. 2008; Stedinger et al. 2008; Vrugt et al. 2009a, b; Beven 2009). It is now clear that we can often do better than employing the Nash-Sutcliffe index and similar measures as a likelihood function, as these types of measures produce less pronounced posterior likelihoods and thus causing users of GLUE to interpret the results as suggesting widespread model equifinality (e.g. Beven 1996, 2006). While equifinality can be demonstrated in a number of ecogeomorphic applications based on *a priori* considerations, the GLUE approach has in the past often overemphasized its effect. It is thus important to integrate field knowledge in the likelihood function, which is a relatively recent development in GLUE (soil-science related examples include Quinton et al. 2011; Krueger et al. 2012a, b; Ortiz et al. 2011) and other Bayesian methods (Huard and Mailhot 2008; McMillan and Clark 2009; McMillan et al. 2010).

One major issue with interpreting the results of studies on land degradation is that they are explicitly a function of human-environment interactions and can thus not be considered outwith the social context of their particular application. Not only is environmental research, like any science, prone to be biased by personal motivations and limitations, institutional agendas and societal values (e.g. Huesemann 2002), in the policy arena scientific results further collide with societal values and are filtered by vested interests (e.g. Pielke 2007). Environmental models in particular contain uncertain evidence and subjective choices and assumptions of the modeller that are largely implicit, unstructured and undocumented (Krueger et al. 2012b). While environmental scientists should at least be aware of the political filtering of their results, an issue that they may or may not wish to engage with, it is arguably a matter of professional integrity to lay open uncertainties, choices and assumptions (Ivanović and Freer 2009). However, the current trend in policy-facing research is often to promote a perspective that underplays the assumptions researchers make about “controversial, indeterminate, or contingent” premises. Fennessey (1977) has argued that a Bayesian approach is invaluable for addressing the contingent and highly contextual nature of research and ensuring that policy decisions are made on as wide a (natural and social) science base as possible. Fennessey suggests that the intuitive nature of the Bayesian approach, its ability to handle diverse data and models (which may also vary in degree of formal specification), and its ability to incorporate contingency while cumulating the knowledge acquired from a range of previous studies are invaluable for tackling these various problems. While this is certainly a scientifically structured approach to managing uncertainties, as well as choices and assumptions through multi-model methods, which has to be encouraged, it has to be kept in mind that the uncertainty assessment itself is often weakly supported by empirical evidence and relies on subjective choices and assumptions just as much (see above). There will thus be limits to the extent to which the Bayesian framework, and any uncertainty framework, can incorporate contingency and context. According to Krueger et al. (2012b) this deficit may be countered with greater transparency in environmental modelling through greater scientific rigour as well as grappling explicitly with the plurality of perspectives that exists in the science and stakeholder community. On the science side, on top of a structured uncertainty framework such as the one discussed above, the authors argue for explicit documentation of choices and assumptions, and the open source paradigm for enabling better a peer review of models than is currently achievable. On the stakeholder side, extended peer review (Funtowicz and Ravetz 1993) and co-generation of knowledge (e.g. Lane et al. 2011) are advocated. Krueger et al. (2012b) go on to review the large number of studies that have demonstrated how different types of scientific and stakeholder understanding can be incorporated into models, especially the use of expert knowledge where detailed measurements are perhaps not available. Examples in the context of drylands research include the UNEP desertification assessment (Middleton and Thomas 1997), qualitative expert systems (de la Rosa et al. 1999), fuzzy expert systems (Giordano and Liersch 2012), and probabilistic networks including Bayesian Belief Networks (e.g. Jensen and Nielsen 2007; Stassopoulou et al. 1998; Ghabayen et al. 2006; see also the overview in Krueger et al. 2012b). However, one should always be careful when applying models in inter- and transdisciplinary studies as formalized models are not always considered by all participants to be the optimal mode of approach, even if their comparison with other methodologies can lead to advances for all involved (e.g. Twyman et al. 2011).

Finally, returning to the issue of pattern that underpins much of this book, Grimm et al. (2005) argue that pattern-based approaches to modelling enable more realistic model structures to be developed that reduce sensitivity to parameter uncertainty, especially when non-integrative measures (such as plot or catchment hydrographs) are unavailable. Consequently, interactions between model components are also more realistic and thus inverse-modelling techniques can be used with more confidence to estimate parameter combinations (see also Mitchell 2003, for an application to climate prediction). As appealing as pattern-based approaches might be, Gestalt Theory as well as more recent studies into abilities of visual perception of pattern (e.g. Davis et al. 1983; Carrasco 2011) suggest that great care is needed in developing these approaches in that pattern recognition is not consistent between individuals or even by the same individual under different conditions (including repeat observations). An answer to the problem of developing pattern-based approaches undoubtedly lies in the combination of the statistical approaches discussed in Chap. 8 and integrative measurements where available (see Parsons et al. 1997; Brazier et al. 2010) with more detailed considerations of these perception-based studies. However, care also needs to be taken with statistical characterizations of pattern, as they will be strongly affected by problems of spatial and temporal autocorrelation discussed in Sect. 10.2.5 and thus simple underlying assumptions of the uncertainty of patterns will not be met. Addressing these problems will be a major challenge in linking pattern-based and uncertainty studies over the coming years.

10.5 Summary

Perhaps because of the frequent difficulty in collecting large datasets for land-degradation research, there has been a relative paucity of studies that evaluate the uncertainty of such data. Yet an uncertainty-based approach is more important than ever in such circumstances if we are to develop robust models in an efficient way. As Clark (2005) has noted, treating uncertainty as "error" causes significant misinterpretations of process. These misinterpretations may lead us to reject acceptable models because of uncertainty in measurements rather than epistemic uncertainty. If we are to avoid these and related problems, we must bring studies of uncertainty explicitly to the foreground. In confronting models with data, uncertainty cannot be ignored.

In this chapter, we have provided a brief basis for the understanding of uncertainty in this context. A differentiation is made between epistemic or knowledge uncertainties, and aleatory or statistical uncertainties relating to measurements or parameters. The former need in part to be addressed by continued reëvaluation of (conceptual) models while the latter require detailed field measurement, and the testing of assumptions (again, conceptual models) underpinning such measurements. Both need to be addressed together in a holistic approach to uncertainty, not least because the measurement of parameters also incorporates epistemic uncertainty in that all measurements are based on conceptual models. However, holistic here does not equate to a "one-size-fits-all" approach – not least because while aleatory or statistical uncertainty can effectively be dealt with in a likelihood-based framework, the same approach can only ever be an approximation for epistemic uncertainty (Beven and Brazier 2010). We think, though, that we have showcased the key ingredients of such an approach here: a structured framework for confronting models with data, methodologies for error propagation, and a transparent and explicit approach to documenting and scrutinising choices and assumptions that is inclusive to the values and expertise of stakeholders beyond the scientific community.

The limitations of these various approaches are considered to provide ways forward of reaching the right answer for the right reason – and where possible, to do so efficiently. The pressing nature of finding solutions to land-degradation problems requires that we do so. Social and political pressures to consider uncertainty as "the elephant in the room" should be avoided; doing so will only produce interventions that have immediate or delayed negative consequences, which could be catastrophic in many cases because of the underlying complexity and non-linearity of land-degradation processes. Awareness of the uncertainty in our models and data should be seen as a strength rather than as a weakness in that it facilitates decision-making about what are the most pressing problems. The promise of pattern-based approaches is considered, but uncertainty techniques to address them are in their infancy, and will require a concerted effort to overcome them.

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Part III
Case Studies

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

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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

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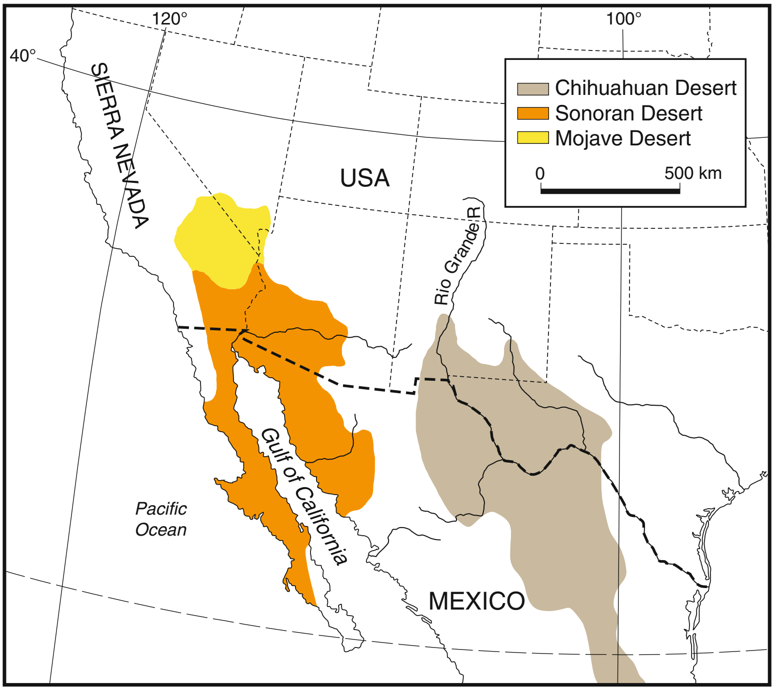


Fig. 11.1 Map showing the extent of the Sonoran, Mojave and Chihuahuan Deserts

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

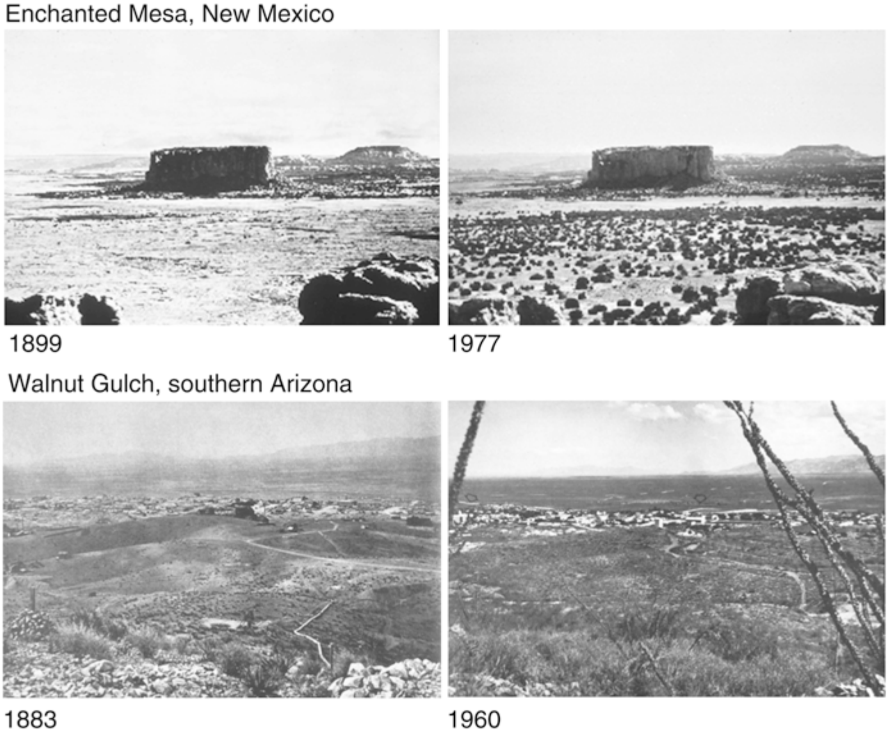


Fig. 11.2 Photographs of vegetation change in the southwestern USA. Juniper encroachment into rangeland at Enchanted Mesa, New Mexico between 1899 and 1977 (*upper*) and creosotebush encroachment into grassland and Walnut Gulch, southern Arizona between 1883 and 1960 (*lower*) (Sources: Allen et al. (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 °C) 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 ecogeomorphic 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

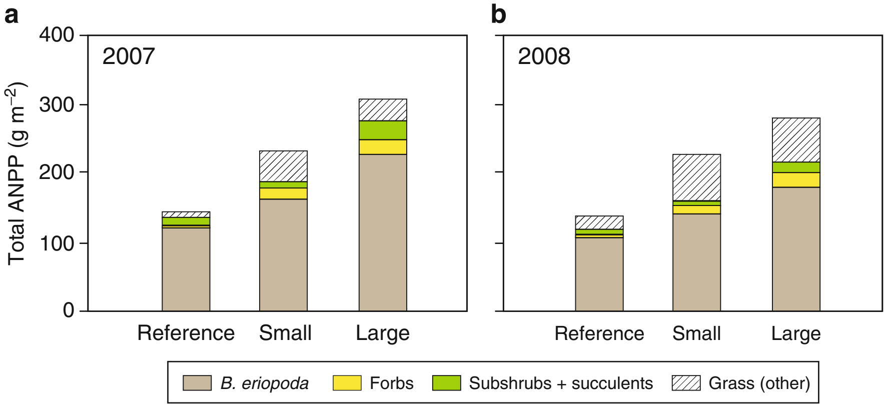


Fig. 11.3 Results of rainfall manipulation experiments carried out in black grama grassland at the Sevilleta National Wildlife Refuge, New Mexico, showing aboveground net primary productivity (ANPP) for each plant functional type in (**a**) 2007 (92.2 mm ambient rainfall) and (**b**) 2008 (125.8 mm ambient 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

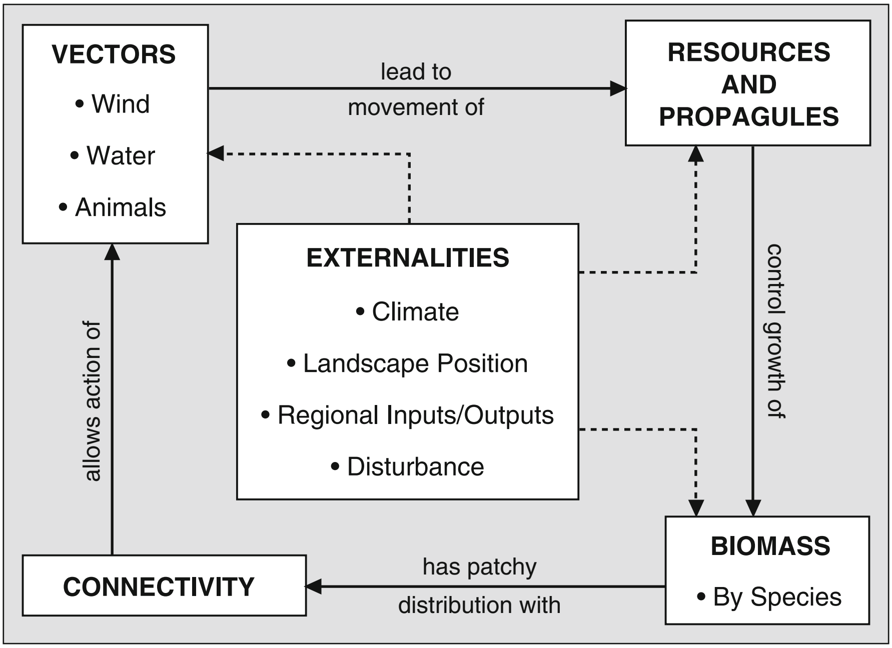


Fig. 11.4 Conceptual integrated model of Stewart et al. (perpetually in press) showing how different vectors of movement may redistribute resources and propagules in ways that may lead to pattern formation, as driven by external drivers and local feedbacks

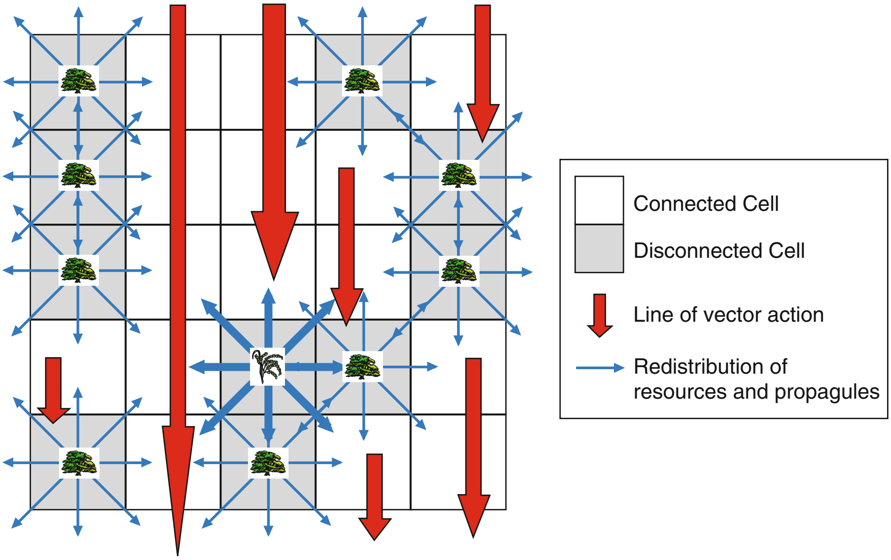


Fig. 11.5 Illustration of how different vegetation types affect the redistribution of resources and propagules as distributed by the different vectors (see Fig. 11.5) in the model of Stewart et al. (in press)

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

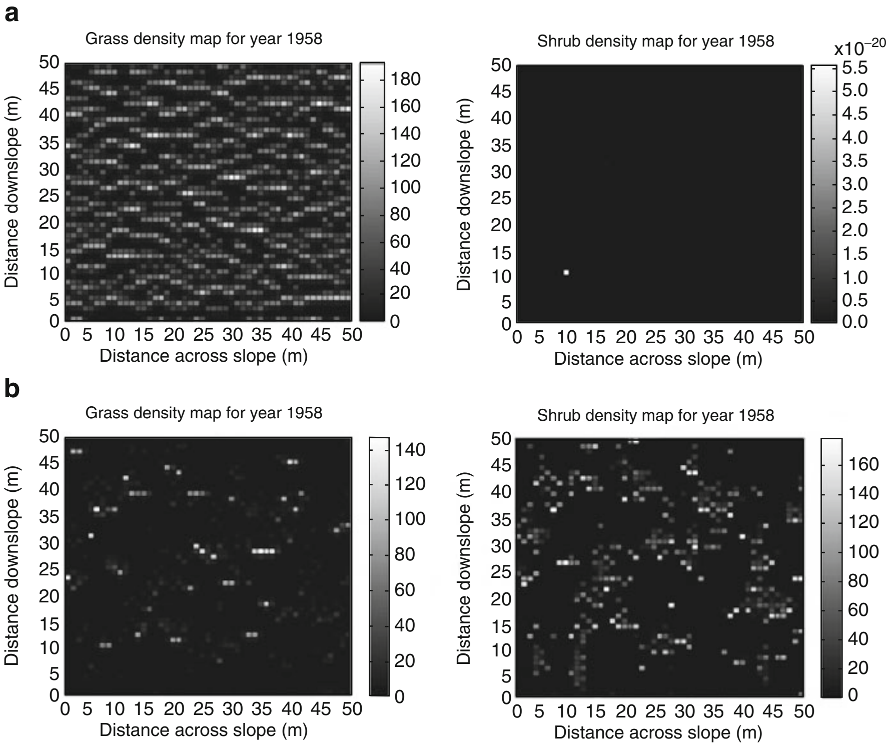


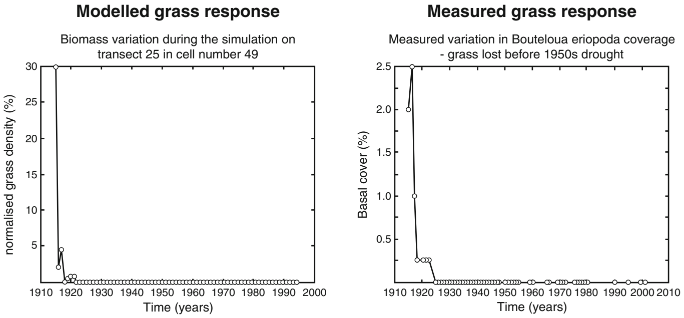
Fig. 11.6 Grass and shrub densities for example simulations using the model of Stewart et al. (in press). (**a**) Baseline simulation showing the dominant of grass patches at a variety of scales; (**b**) the combined effect of drought and high grazing pressure producing a landscape dominated by shrub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

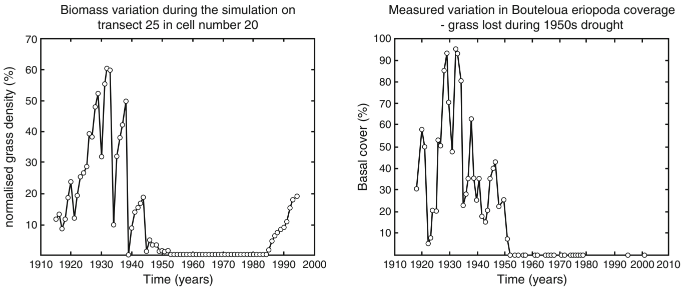
11.6 Conclusion

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

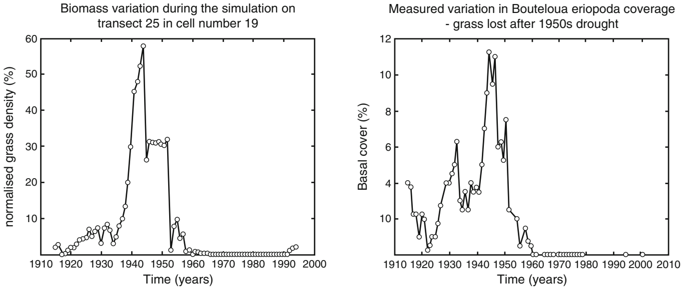
a Data illustrating grass biomass lost prior to the 1950s drought



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

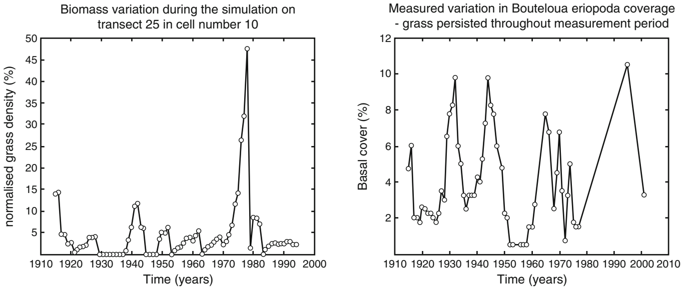


Fig. 11.7 Comparison of data simulated by Stewart et al. (in press) from the *centre line* transect of the grid, as compared with the field observations of Yao et al. (2006)

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.

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Chapter 12
Vegetation Mosaics of Arid Western New South Wales, Australia: Considerations of Their Origin and Persistence

David L. Dunkerley

Abstract The drylands of Australia contain areas of striking and spatially extensive patterned vegetation. The difficulty in understanding emergent structures in dryland eco-geomorphology springs in part from a fundamental challenge of separating function or operation in a system once fully-developed from the processes that drive the initial emergence of patterning. There is an additional challenge over much of the Australian drylands, connected with the very marked inter-annual variability in rainfall, which drives large swings in soil moisture, plant cover, and floristics, as well as in herbivore grazing pressure, abundance of soil biota, and other factors. There are reasons to believe that the rainfall variability, including particularly the ecosystem impacts of the dry years lying near the extremes of the annual series, is important to the emergence of vegetation and soil patterning. In the study area of western New South Wales discussed in this chapter, soil characteristics are tightly linked with the evolution of vegetation patterning. Diverse kinds of vegetation mosaics are present in the study area, ranging from gilgai-like depressions and mounds, in which plants surround the moist depressions, to irregular patches or groves lying within bare soil or stone mantled soil, to strikingly regular, contour-aligned vegetation banding. Groves are favourable locations for plant growth because of the combined role of the enhanced infiltrability generated within groves, and the increased soil water availability that is created by the crabholes and other shallow closed depressions. Field data suggest that crabholes and closed depressions are more likely to account for the presence of shrubs than the reverse. The production of overland flow in the intergroves within patterned vegetation can be remarkably efficient. Consequently, groves are not uniform hydrologically, and the upslope margins receive runon water more frequently than the lower parts of the groves. For the field context of western NSW, existing models fail to touch upon key mechanisms that appear to be pivotal

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D.L. Dunkerley (✉)
School of Geography and Environmental Science, Monash University, Melbourne,
VIC 3800, Australia
e-mail: david.dunkerley@monash.edu

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to the development and operation of emergent vegetation patterns, including soil shrink-swell and spatial differences in soil shear strength. In order to explore in a general way the development of vegetation banding of the kind seen in western NSW, a cellular automaton (CA) model was employed. Crabhole formation during very dry years and runs of years appears to be a very important aspect of the mechanisms that create and sustain the banded mosaic vegetation communities of the study area. Thus, it may be that soil pattern development is the leading process, and the emergence of patterned vegetation is an effect consequent upon the soil behaviour. It is likely, also, that the vegetation patterning establishes a feedback process to further strengthen the hydrologic compartmentalisation resulting from the soil collapse processes.

12.1 Introduction

The drylands of Australia contain areas of striking and spatially extensive patterned vegetation. In all cases, the vegetation patterning is associated with mosaics of differing soils and surface and subsurface hydro-geomorphic processes. Though some aspects of form and process in these landscapes have been researched, knowledge is far from complete, and this chapter will highlight some of what has been learned as well as what remains to be understood. The difficulty in understanding emergent structures in dryland eco-geomorphology springs in part from a fundamental challenge that is encountered in many fields of endeavour, namely, the task of separating function or operation in a system once fully-developed from the processes that drive the initial emergence of patterning. This is a classic ‘chicken or egg’ problem. There is an additional challenge over much of the Australian drylands, connected with the very marked inter-annual variability in rainfall, which drives large swings in soil moisture, plant cover, and floristics, as well as in herbivore grazing pressure, abundance of soil biota, and other factors (van Etten 2009; Letnic and Dickman 2010). The ENSO-related fluctuations in annual rainfall that occur strongly over much of eastern Australia (Suppiah 2004) cause conditions in many drylands there to swing from extreme drought with scant plant cover to exceptional wetness and plant cover through irregular quasi-cycles lasting some years. In this area, rare exceptionally wet years associated with La Niña conditions occur at intervals of perhaps some decades, and alternate with occasional El Niño periods of extreme dryness. In such an environment, it is not immediately clear how to characterise or parameterise the rainfall environment, nor to see clearly whether very dry or very wet years are responsible for driving the most fundamental ecosystem and landscape adjustments. As will be shown below, there are reasons to believe that the rainfall variability, including particularly the ecosystem impacts of the dry years lying near the extremes of the annual series, is important to the emergence of vegetation and soil patterning. If this is correct, then the emergence of enduring vegetation patterns in the area of eastern Australia discussed in this chapter cannot be accounted for simply by reference to mean climatological dryness or the mean annual level of water scarcity within the ecosystem; it also involves inter-annual variability as a key driver. But it will be argued below that in the study area, soil characteristics are tightly linked with the evolution of vegetation patterning.

The following discussion is limited primarily to the patterned landscapes of western NSW, Australia, though work in other environments is referred to. (For a broader review of ecogeomorphology in the Australian drylands, see Dunkerley 2010). The chapter contains three sections, which briefly consider the form and composition of the patterned landscapes, some of the eco-hydrologic factors that operate on and within them, and some explorations of their developmental pathway and persistence in the face of climate change and land-use pressures.

12.2 The Patterned Dryland Landscapes of Western New South Wales, Australia

The landscapes of western NSW considered here are all used for pastoral production, and the pressure of herbivory adds to the climatic stresses on plant cover (Read 2004). Grazing, primarily by sheep, has been carried on since the middle of the nineteenth century in many areas. In addition to the direct effects of grazing stock and introduced feral pest animals such as rabbits and goats, considerable impacts in these landscapes undoubtedly arose in the early days of European settlement from the cutting of trees for fence posts, for fuel, and for use in the mining industry. Grazing pressures are selective according to season and the palatability of various taxa, and it is therefore clear that the ecosystems studied today are to some extent modified from the conditions that would have existed prior to European settlement, in terms of floristics, ground cover, and other parameters. The probable extent of change is difficult to assess or quantify, owing to the absence of baseline data. In pre-European times, human impacts from the indigenous peoples and from native herbivores would have been of quite low intensity, though there may have been effects from the use of fire by people for clearing grass and shrub vegetation after years of very high rainfall and fuel accumulation.

Diverse kinds of vegetation mosaics are present in the area, ranging from gilgai-like depressions and mounds, in which plants surround the moist depressions, to irregular patches or groves lying within bare soil or stone mantled soil, to strikingly regular, contour-aligned vegetation banding (Fig. 12.1) such as that described by Dunkerley and Brown (1995).

The environments in which patterned vegetation occurs conform to the ranges that are reported for most global occurrences (low topographic gradients, broad relatively un-dissected surfaces, mean annual rainfalls of a few hundred millimetres or less; e.g. Lefever and Lejeune 1997). The patterned plant communities in western NSW are dominated by chenopod shrubs (e.g. Atriplex spp.), or by native perennial grasses such as Astrebla spp., but elsewhere in the Australian drylands, the most widespread patterned vegetation, including striking and extensive contour-aligned vegetation banding, is developed in woodlands of Acacia spp. with a grassy understory (Dunkerley 2002).

Fig. 12.1 Contour-aligned banded vegetation in arid western New South Wales, Australia. The *upper photo* is an aerial view, and the *lower* presents a ground-level view looking downslope. The line of trees on the horizon marks the position of an ephemeral stream channel

12.2.1 Soil Characteristics and Shrink-Swell in the Mosaic Landscapes

An aspect of the western NSW landscapes that is distinctive is their soils. These contain exogenous silts and clays delivered by aeolian transport from more arid areas of the Australian inland lying to the west (upwind). In the Barrier Ranges – Fowlers Gap area, which forms the focus for this chapter, and where many of the ridge-forming rocks are highly resistant quartzites that generate little or no soil fines, the aeolian input of exogenous clays and silts has been a key influence upon soil depth, texture, mineralogy, and physical and hydrologic behaviour (Chartres 1982). Accumulation of windblown dusts through arid glacial cycles is considered to have resulted in significant blanketing of the areas such as those flanking the low uplands of the Barrier Range, whose elevation forms a topographic barrier that lies across the path of the dominant westerly dust-transporting winds. Exogenous wind-blown fines have been reported from soils in various parts of eastern Australia. Along the Murrumbidgee River in the Wagga Wagga area, Chen et al. (2002) reported depths equivalent to up to 80 cm of dusts intermixed with dune sands, and inferred rates of dust accumulation of up to 5 cm per ka through the last glacial maximum. From the Namoi area in northern NSW, Cattle et al. (2002) reported deposition equivalent to 20 cm of dust in the last 13 ka. The dusts had illite-smectite mineralogy, and a modal diameter of 40–50 μm. Both of these locations are located much further downwind than the area discussed in this chapter, and presumably received a reduced dust flux. Dryland dusts from the Australian inland have been reported from other locations, including NW NSW (Tate et al. 2007) and Antarctica (Revel-Rolland et al. 2006).

Since the time of its accumulation in the Barrier Ranges, overland flow on the hillslopes has reworked much of this material into piedmont slopes and low-angled alluvial fans, and fluvial transport has spread it across the extensive floodplains and outwash plains of the ephemeral channel systems. Consequently, many of the areas of patterned vegetation are developed not on the relatively thin soils of the rocky hillsides but flanking footslopes on deep alluvial-colluvial soils of broadly loamy texture, which contain exogenous illite-family clay minerals. These soils exhibit marked shrink-swell behaviour, desiccation cracking, and surface seal and crust development. Many of the soils and sediments slake very rapidly when in contact with water, but the soil surface is commonly stabilised against wind and water erosion by biological soil crusts. Lichen and bryophyte crusts are locally abundant, but cyanobacterial crusts are virtually ubiquitous. Resistance to erosion by overland flow also arises from dense veneers of resistant stones, including vein quartz and quartzite, which rest on the soil surface over large areas. In extreme droughts, wind abrasion progressively disrupts and fragments the biological crusts, and undercutting and the lifting of crusts fragments become widespread.

Upton (1983) confirmed that Fowlers Gap soils also contain illite minerals, with the clay fraction tending to increase with depth. Upton's linear shrinkage tests showed that the uppermost 15 cm exhibited linear shrinkage of 1.6 %; the 15–20 cm depth, 9.2 %; and the soil below 30 cm (often to $>$100 cm) showed shrinkage of 11.6 %. Volumetric contraction would be approximately three times that of the linear shrinkage, or about 30 % in the subsoils. (Coefficients of linear shrinkage are often about 15 % (e.g. Greacen and Huon 1953) though Akpokodje (1985) reported linear shrinkage of up to 20 % for heavy-textured soils at Fowlers Gap, suggesting that locally, very large volume changes would be associated with wetting and drying cycles). Observations by the writer suggest that dry conditions are required in order for deep subsoils to dry and shrink. Thus, in El Niño years especially, marked

Fig. 12.2 Typical appearance of crabholes in vegetation mosaics of arid western NSW. The *upper right* images shows a small collapse encircled by tension cracking (see text for details)

volumetric contraction of the subsoils (where the clay content is high) results in a loss of support for the soil above, and collapse features then occur, locally called ‘crabholes’ (Fig. 12.2).

Crabholes and shallower closed depressions arising from subsidence form highly efficient traps for overland flow. Crabholes are sometimes ringed by tension cracking (Fig. 12.2) which offers a second pathway for overland flow to enter the deeper soil than would readily occur from the infiltration of rainfall through the soil matrix. Shrink-swell behaviour of the soils in some cases results in gilgai undulations in the soil surface, which may be semi-permanent. Closed depressions often result, and these gather water in wet years that follow drying and shrinkage in preceding droughts (Fig. 12.3). This is likely to prime the deep subsoils for renewed contraction when they next dry out. This effect is probably amplified by the more abundant vegetation that is able to grow in and around the water-filled depressions.

Though there are no relevant field data, it can be considered possible that the ENSO quasi-cycle is associated with a parallel quasi-cycle of shrink-swell in the deeper subsoils. Crabhole abundance and location are currently being monitored, and data to date suggest that crabholes and closed depressions that were abundant in the drought of the decade to 2009 have largely disappeared following the exceptional rainfall of 2010, which therefore appears likely to have allowed subsoil materials to increase in volume.

Fig. 12.3 Examples of closed depressions in vegetation mosaics of arid western NSW. Both photos are taken looking upslope across closed depressions that contain water from recent rain. The vegetation in each case is dominated by chenopod shrubs. Intergroves are veneered with pebbles

Table 12.1 Crabhole and closed depression volumes, Fowlers Gap Arid Zone Research Station

Grove number	Number of closed depressions (area mapped m^2)	Areal density (pits per m^2)
1	31 (75)	0.41
2	20 (37.5)	0.53
3	17 (25)	0.68
4	16 (25)	0.64
5	18 (25)	0.72

Data collected under drought conditions in March 2009, when the value of the SOI was +0.2

A key observation pertinent to understanding the hydrologic operation of these vegetation mosaics is that crabholes and closed depressions are only found within the groves. A hypothesis to account for this is that enhanced infiltration attributable to one feature (plants or closed depressions) favours co-location of the other. Mechanisms for crabhole formation have been outlined by Upton (1983) and the literature on gilgai contains many accounts of the formation of closed depressions (e.g. Knight 1980). The presence and potential importance of crabholes in the development of vegetation mosaics have been mentioned previously in the literature (Dunkerley 1997a, b; Dunkerley and Brown 1995) and will be addressed further below.

12.2.2 *Field Data on Crabholes and Shallow Closed Depressions*

In order to assess the capacity of crabholes and closed depressions to contribute to depression storage and the capture of runon water, estimates of pit and depression volume were made in the field. This was done by lining crabholes and depressions with thin, soft plastic sheeting, and filling with measured amounts of water until level with the surrounding soil surface. Table 12.1 presents some indicative data from this work.

With a mean density of about 0.5 crabhole or closed depression per m^2, and a mean volume of 3.7 l (standard deviation 1.6 l), these pits have the capacity to intercept the equivalent of about 2 mm of rainfall across the area of a typical grove. Some of the pits measured were quite large, the maximum recorded volume being 8.5 l. This static measurement using plastic sheeting is an underestimate of the true water-trapping capacity of the crabholes, neglecting as it does the unmeasured narrow crack network extending deeper into the soil, and also the ongoing seepage of water into the walls of the pit and the crack network during rainfall. Nevertheless, given that these closed depressions arise in dry years, and vanish in wet years when the subsoils are re-wetted, their significance is twofold: first, they present a large capacity to trap and retain water within the groves. Crabholes are never seen within the drier intergroves, presumably because there the subsoils are never wetted, and therefore do not display major volumetric shrinkage in dry years. Second, the crabholes are important because they compensate for the loss of vascular plant cover and litter that occurs in dry years. Crabhole formation represents the development of marked water trapping in subsequent rainfall events that has little directly to do with the vascular plant cover. Thus, in dry years, when surface plant cover becomes scant, subsoil shrinkage results in crabhole formation, and this ensures that overland flow is trapped with great efficiency even though the surface may be quite bare. In this way, the scare water resource is conserved, and conditions favourable to the persistence or re-establishment of vascular plant cover are created once adequate rainfall occurs.

Thus, an important argument of this chapter is that the groves are favourable locations for plant growth because of the combined role of the enhanced infiltrability generated within groves, and the increased soil water availability that is created by the crabholes and other shallow closed depressions. Crabholes are undoubtedly water traps and enhance the entry of water into the subsoil. In contrast, it is clear that water entry is not always enhanced in the presence of shrubs, which may in fact result in diminished soil water contents at sub-canopy sites (Moran et al. 2010). In light of the wide spacing of vascular plants in groves of the study site, which would appear to eliminate shading, sheltering from wind, or other benefits that might arise in denser groves, crabholes and closed depressions seem more likely to account for the presence of shrubs than the reverse.

12.2.3 El Niño – Southern Oscillation (ENSO) Rainfall Variability

The other distinctive feature of western NSW is its marked inter-annual rainfall variability. ENSO-related phenomena result in marked quasi-cycles of variability in annual and seasonal rainfall amounts across much of eastern Australia (Verdon et al. 2004) such that many areas show wet years in association with La Niña conditions, in which the rainfall is 50–100 % greater than in El Niño years. For much of western NSW, 1974 was the year of record rainfalls in the period of European settlement. The Southern Oscillation Index (SOI) for that year was around +20 for the late summer months (that is, trans-Pacific pressure differences were about two standard deviations from their long-term mean for those months). For September 2010, the SOI value was even higher, at +25; it was high for a number of months following June. In contrast, for much of the summer and autumn of 2010, the SOI was in the range −10 to −15. In 1974, Fowlers Gap 1974 experienced the highest annual rainfall on record (629 mm) and 2010 was the second wettest year, with 523 mm. However, the complex behaviour of annual rainfall can be illustrated by observing that 1987 was also an exceptionally wet year (408 mm) but the SOI was moderately negative all of that year. In dry years (well exemplified by 1983, when the SOI in summer lay in the extreme range −28 to −33), plant cover becomes very much reduced, and sand and dust storms become common. As mentioned earlier, sandblasting of the cyanobacterially crusted soil surface fragments or eliminates the crust over large areas. Soil moisture falls to extremely low levels (less than a few percent by volume) and occasional rainfall events result in highly sediment-laden ephemeral streamflows. In the same way, plant cover in the vegetation mosaics becomes extremely low, and only scattered individual woody plants, such as chenopod shrubs, remain, largely in a dormant condition and with diminished canopy cover. Ephemerals and grasses are completely absent. Many research questions remain about the causes of major, lengthy drought periods the affect parts of the Australian drylands, including the study area. The long and severe 1995–2010 dry period, termed the “Big Dry”, is an example (Ummenhofer et al. 2009).

12.2.3.1 The Rainfall of Fowlers Gap

When rain stops only briefly – perhaps for a few minutes – and then resumes, it is appropriate to count both periods of rain as a single storm. But as a gap in rainfall grows to many hours, perhaps with sunshine and drying winds during the gap, it makes more sense to tally two separate events. This process is customarily made rigorous by defining a minimum inter-event time, or MIT, which is a rainless period that must be reached or exceeded for two periods of rain to be counted as separate storms. There is a wide range of MIT criteria in use, but often it is set to about 6 h (Dunkerley 2008a).

Using such a definition, the Fowlers Gap rainfall record at Fowlers Gap shows that there are on average about 25 storms per year that deliver a depth exceeding 0.5 mm, and that these typically last for around 4.5 h, delivering 8.1 mm of rain at an average intensity of 3.8 mm/h. The distributions of all of these parameters exhibit strong positive skew (Fig. 12.4). However, these low rainfall intensities and moderately short storm durations in an environment of low relative humidity mean that the drying of wet foliage consumes a large fraction of the arriving rain.

Owing to the entrainment of dry air from beyond the localised rainfall cells, intra-storm evaporation can proceed at quite high rates (Dunkerley and Booth 1999; Dunkerley 2008b). Rain is registered in defined events for only about 1.3 % of the time, and the mean waiting time between successive rainfall events is about 10 days, though in the extreme it can extend to months. Owing to the low rainfall rate in this environment, and the fully dry soil surfaces and plant canopies encountered at the start of events, the effective rainfall, the part that infiltrates into the ground to add to soil moisture, will be significantly less than the gauge rainfall owing to ongoing intra-event wet canopy evaporation. Rainfall events of longer duration are therefore likely to be very important to the recharge of deep soil moisture in this environment, and hence to volumetric expansion of the subsoil clay minerals.

Annual rainfalls at Fowlers Gap can be fitted by a two-parameter gamma distribution having scale parameter of 48.83 and a shape parameter of 4.71, though the number of high-rainfall years such as 1974 and 2010 is too small to offer great confidence in the fitting of the positive tail of the distribution.

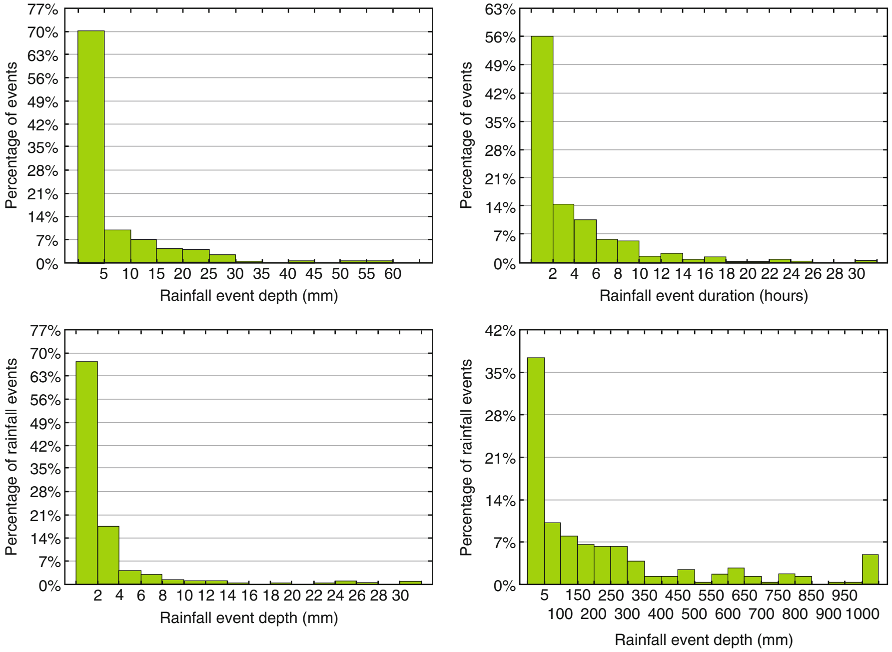

Fig. 12.4 Distribution of rainfall event depths (*top left*), rainfall rates (*bottom left*), event durations (*top right*) and event waiting times (*bottom right*) for Fowlers Gap Arid Zone Research Station. The data span 8 years of record

12.2.4 Plant Cover in the Patterned Landscapes

Plant cover in the patterned landscapes is very variable through time, as a consequence of variations in seasonal and annual rainfall, especially through the ENSO quasi-cycle just discussed. Foliar cover is low, often only reaching values of 20 %, even with the ‘groves’ or more densely vegetated phases of the mosaics. In chenopod mosaics, individual plants are not located sufficiently closely that there would be any significant synergistic or shading effect of one plant on another; rather, the plants are widely spaced. The plants are generally of low height, typically reaching no more than 30–60 cm. During very dry years, the above-ground parts of many chenopod shrubs appear brown and dead, but many individuals are capable of recovering once soil moisture is replenished, and in general these plants are long-lived. In wetter years, abundant ephemerals emerge, along with native grasses, and in suitable locations along floodplains, foliar cover can approach 100 % for short periods before dry conditions return. In the case of larger chenopod shrubs, measurements have confirmed that soil infiltrability is higher near the stem, and declines into the shrub interspace (Dunkerley 2000). Moreover, some plant-related increase in infiltrability persists even beyond the edge of the plant canopy.

12.2.5 Runoff and Runon in the Patterned Landscapes

Soil surfaces vary systematically through the vegetation mosaics. Within groves, soils are soft, and the surface is often broken by shrink-swell behaviour; biological crusts are also less developed in groves. These soils belong to the class of soils known as 'self-mulching', and they exhibit desiccation cracking (Pillai-McGarry and Collis-George 1990a, b). Infiltration rates based on cylinder infiltrometry and the application of simulated rainfall at realistic rain rates to bounded runoff plots (Dunkerley 2008c, 2009a) suggest infiltration rates of 15–30 mm/h in groves. Intergroves have soils which grade upslope somewhat diffusely into grove soils, but which become very hard and impermeable toward the lower intergrove. At lowermost intergrove locations, infiltration tests suggest rates of up to 5 mm/h. Observations made during and after rainfall have confirmed that overland flow arises on the intergroves, where shallow ponding and flow are widespread, and from there drains toward and into the groves, where soils are wet but where surface ponding is much less extensive. Surface water is primarily seen within groves at locations where preferential flow paths deliver trickles of overland flow from the intergrove lying upslope (Fig. 12.5).

The overland flow shown in Fig. 12.5 resulted from prolonged but low rate rainfall on 4 March 2010, amounting to nearly 40 mm. It is clear that ponding is primarily a feature of the impermeable intergroves, and that water is strongly absorbed within the groves. The figure also makes it clear that many plants within the grove stand on mounds and that water entering the groves passes between, rather than directly onto, the plants. Consequently, the soils exhibiting the highest infiltrability (located close to the stem) are not able to become directly active in the absorption of runon water, though in much of the literature, this effect is claimed to be a key process favouring plants in groves. For example, Yizhaq et al. (2005) include in a model of vegetation patchiness the presumption that the presence of a vascular plant induces local infiltration larger than would occur in the absence of that plant. This role for plants is cited as a key factor needed to account for the emergence of vegetation patterns.

The production of overland flow in the intergroves within patterned vegetation can be remarkably efficient. Observations by the writer have shown substantial runon water arriving at groves from the immediately upslope intergroves, from rainfall events of <3 mm depth, falling at low rain rates. Soils along the upslope margins of some groves were visibly wet when inspected 12 h after one rainfall event of just 2.8 mm depth, while downslope within the grove, soils appeared dry. In other words, these mosaic landscapes can result in important accessions of water to plants in groves that would not arise in an unstructured landscape. But clearly, groves are not uniform hydrologically, and the upslope margins receive runon water more frequently than the lower parts of the groves.

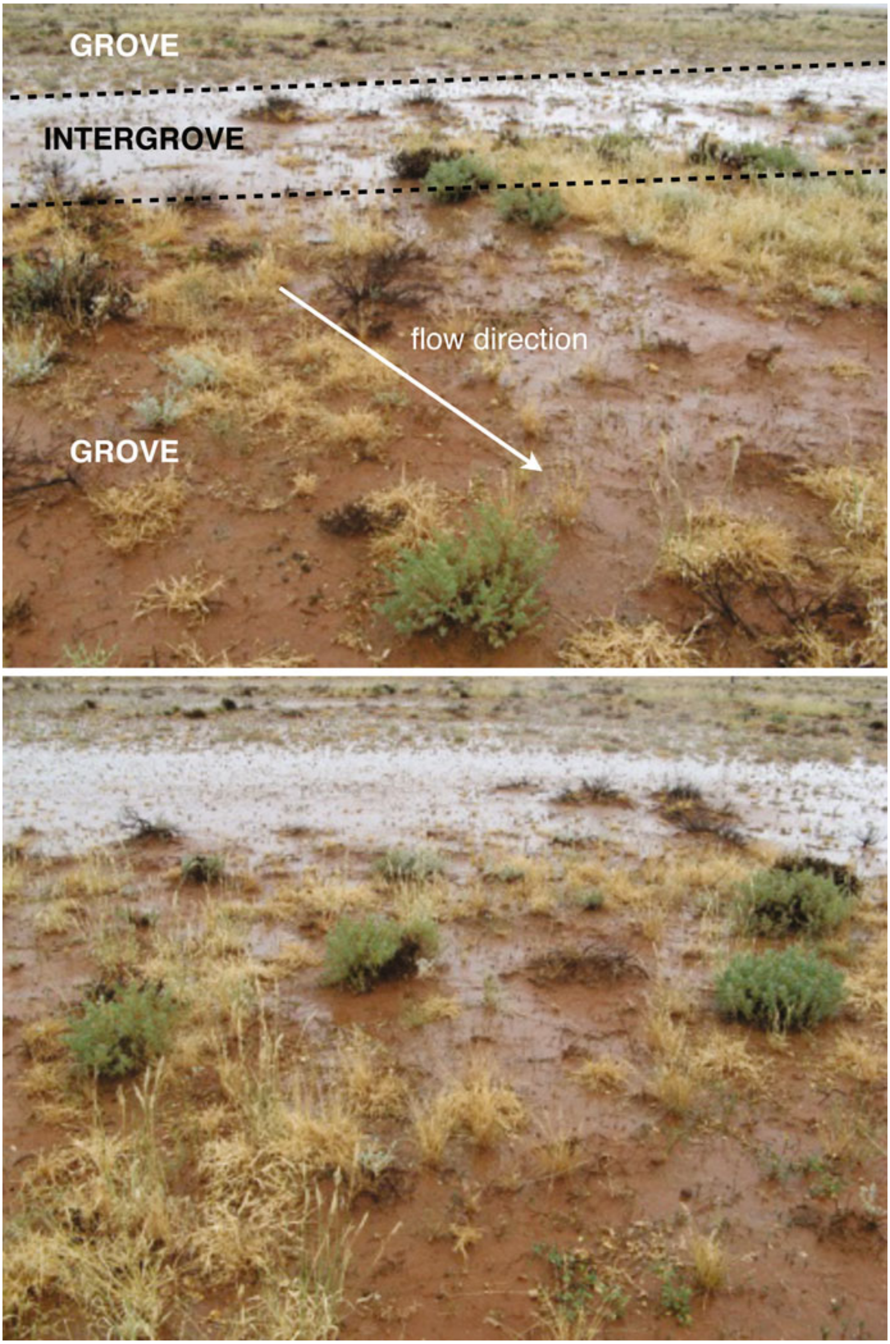

Fig. 12.5 Two examples of overland flow behaviour in a vegetation mosaic in arid western NSW. Both photos are taken looking upslope, and show a single intergrove that carries shallow surface flow, and groves upslope and downslope where ponding is virtually absent. Note the extent of bare soil event within the groves

12.3 Exploring the Origin and Development of the Patterned Landscapes

Having presented a brief outline of some of the environmental characteristics affecting patterned vegetation in western NSW, we now turn to consider what can be said about the origin of this striking mosaic of vegetation and soils.

In recent years, many hypotheses have been advanced seeking to account for the emergence of stable vegetation patterns such as those of the Fowlers Gap study sites and others in western NSW. A recent overview of some approaches was provided by von Hardenburg et al. (2010). Among the plethora of hypotheses, some consider patterns to be the outcome of an interaction of competitive and facilitative interactions among plants, acting at different spatial scales (Borgogno et al. 2009); other approaches stress the pivotal importance of the positive feedback provided by the link between vascular plant cover and soil infiltrability (Okayasu and Aizawa 2001). Some workers envisage a critical role for the nature of the rainfall climate, including seasonality and the annual variability in rainfall amounts (D'Odorico et al. 2006; Ursino and Contarini 2006; Kletter et al. 2009), while others adopt an unchanging annual rainfall or do not treat rainfall explicitly at all (Kéfi et al. 2007, 2010). Some models attempt to incorporate 'process' representations, such as canopy interception loss (Borgogno et al. 2007) or the dispersal of propagules (Thompson et al. 2008) but include no data on the properties of the soils and of the overland flow field, including infiltrability, flow depth, and flow speed, as controlled by rainfall event structure. Many theoretical approaches envisage a bistable distribution whose end-members are fully vegetated or bare states, with landscape processes of various kinds yielding a stable intermediate condition (Lejeune et al. 2002).

A striking feature of much of this literature is the extent to which it is has evolved without reference to data from particular field locations and conditions, and the extent to which hypotheses have been advanced without being constrained by validation. In particular, in light of the field context of western NSW, existing literature fails to touch upon key mechanisms that appear to be pivotal to the development and operation of emergent vegetation patterns, including soil shrink-swell. The challenge here is worth illustrating. For instance, some workers (e.g. Lefever and Lejeune 1997; Yizhaq et al. 2005; Esteban and Fairén 2006) argue, on the basis of model output, that banded vegetation patterns incrementally and progressively migrate upslope. However, there are few or no reliable data confirming this process, much less examining the mechanism or rate of the supposed upslope migration. Additionally, the models cited do not include any consideration of soil development. In the banded vegetation of western NSW, lower intergrove soils exhibit a shear strength that is far too high to allow root development, and they are so impermeable that the exceptionally dry soil environment is hostile to any propagules that might lodge there. Moreover, the system of crabholes and closed depressions tends to promote re-wetting of subsoils, and hence the future formation of new crabholes in the same location repeatedly. Through processes of this kind, it

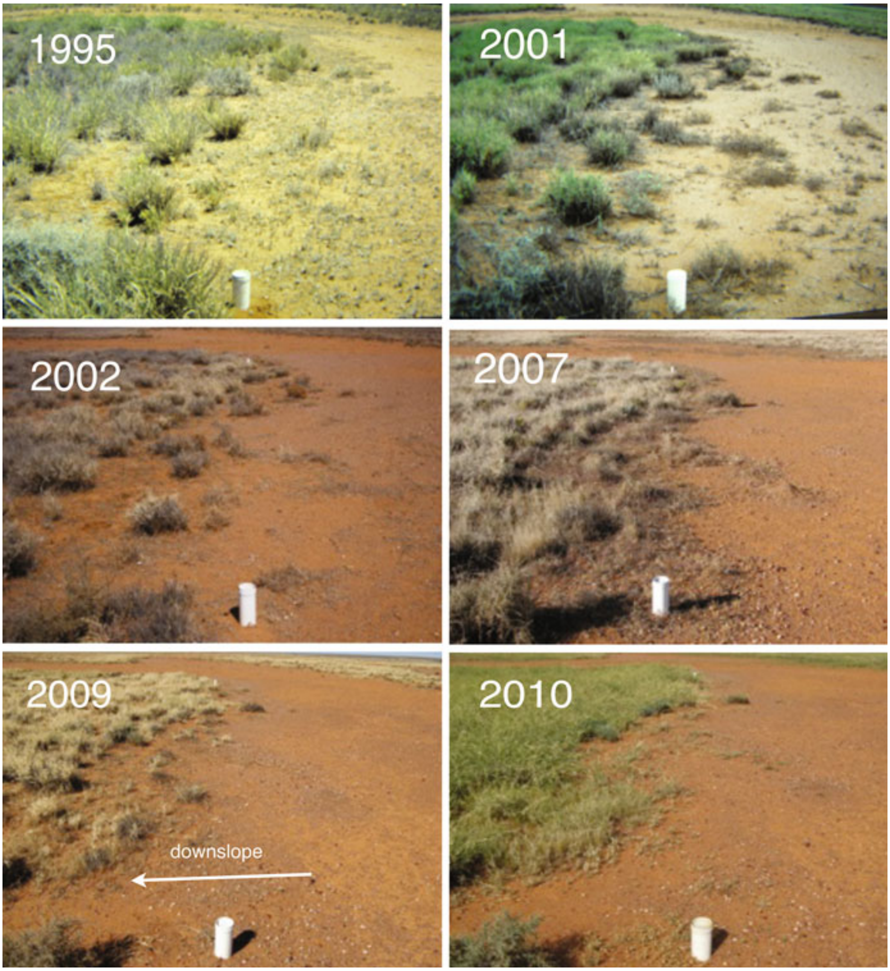

Fig. 12.6 Time series of photographs of a grove-intergrove margin in a contour-aligned banded vegetation mosaic in arid western NSW. In 1995, the upslope edge of the grove was marked with two PVC pipes 30 m apart. Though individual shrubs and grasses have been replaced during the 15 years of monitoring, there is no evident shift in the location of the grove margin

has been argued that the vegetation bands are effectively locked in a fixed position by pedologic controls (Dunkerley 2009b). By considering soils in this way as an essential component in any model dealing with vascular plants, it is possible to envisage mechanisms which would mitigate against the upslope migration of vegetation bands. More than 15 years of repeat surveys of band margins in western NSW (e.g. see Fig. 12.6) show no detectable shift in these bands, despite the notable inter-annual rainfall variability of this area, mentioned earlier.

In light of the supposed rates of upslope migration (Sherratt 2010 refers to rates of 0.2–1.0 m/a) the upslope displacement in the period of observation ought to lie in the easily detectable displacement range 3–15 m. Clearly, no such upslope displacement has occurred at the monitored locations. In considering these landscapes further, yet other inconsistencies with models divorced from field locations and field data emerge. For instance, Yizhaq et al. (2005) predicted that the spacing of vegetation bands would undergo adjustment, widening (via the elimination of some groves) when rainfall declined. Repeat surveys of 30 wavelengths of the regular vegetation banding shown in Fig. 12.6 over 15 years have shown no detectable change in grove and intergrove dimensions, despite very marked rainfall variability. Under exceptionally dry conditions, grove foliar cover declines, but grove locations do not change. This kind of discord between models and field data suggests that many models have failed to capture key ecosystem processes, and require better validation, so that modelling efforts can incrementally be developed further.

Another significant issue can be mentioned here. It has also been argued on the basis of numerical models that landscapes containing patterned vegetation may be at great risk from climate change (Kéfi et al. 2007), and this influence may be combined with increasing pressures arising from human use of these landscapes for purposes such as pastoral production. Whether this is true or not remains to be seen. In the case of the striking banded vegetation in western NSW, long-term local residents have described these areas in extreme drought as being almost bare, and yet the groves re-appear when rainfall returns to ordinary levels. Soil microtopography associated with the vegetation bands (e.g. Dunkerley and Brown 1995, 1999) and the crabhole processes already described, may well account for this stability, since they affect the retention of overland flow whether or not plants are present. Certainly, in western NSW, widespread banded grassland systems are valued by pastoralists for their resilience and productivity under climatic and grazing pressure, and this certainly does not seem to be a system under threat of imminent collapse. Some models do indeed suggest this kind of resilience. HilleRisLambers et al. (2001) used a numerical analysis to suggest that high levels of herbivory are not likely to be associated with the collapse of plant cover in patterned ecosystems. This emphasises the need for models used for scenario development to be fully tested and validated, so that their applicability in actual landscapes and ecosystems can be properly evaluated.

12.4 Development of Mosaic Vegetation in a Cellular Automaton Model

In order to explore in a general way the development of vegetation banding of the kind seen in western NSW, a cellular automaton (CA) model was employed. This modelling approach is suitable for exploring the outcomes of a range of possible drivers of vegetation pattern development, without being so demanding of data that it cannot be fully parameterised. The application of models of this kind in ecological work was reviewed by Dunkerley (1999). The model used here consists of a tessellation of 10,000 cells (100 × 100) each taken to represent perhaps a plant or a few neighbouring plants, growing on a gradient sufficient to drive overland flow or seepage across the tessellation. The model was run on time steps taken to represent a year, in the sense that the annual rainfall was delivered to the tessellation at a single model time step. These same general conditions have been widely adopted in the literature on emergent patterns in dryland vegetation (e.g. Bailey 2010). This approach completely fails to capture rainfall event properties and their role, but really too little is known about the fate (infiltration, overland flow, evaporative loss) of rainfall events of varying sizes in this landscape to attempt to model their behaviour with any confidence. Likewise, too little is known of the response of chenopod shrubs to rainfall pulses of contrasting properties to attempt to build a more process-oriented model. For these reasons, the modelling performed here is significantly abstracted and can only be used to explore the generalised influences of soil and climate on the vegetation mosaics.

12.4.1 Representing the Time Series of Annual Rainfalls in Western NSW

The frequency distribution of dryland annual rainfall depths is often positively skewed (e.g. Tilahun 2006). The longest rainfall records from western NSW span about 130 years, and show that the driest years receive 40–50 mm of rainfall. Some very dry periods have been recorded, including the 6-month period December 2004–May 2005 inclusive when the total rainfall at the Fowlers Gap Arid Zone Research Station was just 7.4 mm. Conditions for plants would be severe under such conditions, with summer daytime temperatures commonly exceeding 35 °C. Modelling with annual totals fails to capture the possible influence of such intensely dry periods. Similarly, there are often months in which the total rainfall exceeds the median annual total, and these maxima are commonly recorded from late summer or early autumn in western NSW. Some details of the frequency distributions of annual rainfall depth for five stations in western NSW are presented in Tables 12.2 and 12.3, and Fig. 12.7. These show means of about 200 mm/a and slight positive skewness. However, it cannot confidently be said that a single statistical distribution can be used to characterise these time series data. Rather, it seems equally reasonable to suggest that the distribution can be characterised by an essentially normally distributed series of totals, with outliers that are both very dry and very wet, reflecting the wide swings away from the mean rainfall that arise from the ENSO phenomena mentioned earlier. High rainfall years often constitute only one or a few percent of years, and the distributions (Fig. 12.7) are commonly discontinuous and longer rainfall time series will be needed to explore the distributions more fully.

The cellular model was run for 50–100 year sequences. Annual rainfall depths were derived from a series of normally distributed random numbers, using routines derived from Press et al. (1986). The series was truncated at an annual rainfall of

Table 12.2 Statistical measures of the distribution of annual rainfalls for five stations in western New South Wales

Meteorological station location	Years of record	Mean annual rainfall (mm)	Standard deviation of annual rainfalls (mm)	Skewness of the distribution of annual rainfalls
Broken Hill	118	251.2	114.8	1.52
Corona	130	207.1	101.6	1.35
Sturt's Meadows	57	154.7	76.5	1.28
Milparinka	127	191.7	121.7	2.43
Tibooburra	125	221.3	130.9	1.39

Source: data extracted from records of the Bureau of Meteorology, Australia

50 mm, to correspond with the data of Table 12.3. The mean was set to 200 mm/a and the standard deviation to 100 mm/a, again to correspond with typical rainfall records from western NSW (Table 12.2). To explore the influence of rainfall climate, in some model runs the random number series was modified purposively by including short runs of 3–5 dry years, to represent La Niña drought conditions, and in others, by including single very wet 'outlier' years, to represent the kind of widespread rains that were recorded across much of western NSW (and other parts of Australia) in 1974, and presumably at other times during the development of the mosaic communities, but prior to the commencement of record keeping (see Table 12.3).

12.4.2 Water Retention Rules in the Model

As is usual with CA models, a small 'neighbourhood' of cells was used as the basis for the application of the model rules, in this case a modified von Neumann neighbourhood. An initial set of randomly-selected cells covering 10 % of the tesselation was vegetated. In the operating rule set of the model, vegetated cells have a trap efficiency for rainfall that is dependent on the depth of rainfall received each year. This is intended to reflect the loss (or gain) of foliage and surface litter that occurs in dry (and wet) years. In drought conditions, when litter is scarce, the soil surface becomes increasingly bare, and it is reasoned that infiltrability is reduced owing to increasingly open and connected flow paths across the soil surface, diminished bioporosity created by soil fauna, and increasing exposure to rain, with formation of surface seals.

Given that the mean annual rainfall was defined as 200 mm, the dimensionless trap efficiency (Et; the fraction of the incident rainfall absorbed by a vegetated cell) was specified as: $\mathrm{Et} = \mathrm{R}/200$ mm for $\mathrm{R} \leq 200$ mm and $\mathrm{Et} = 1$, otherwise, in which R is the rainfall of any model year, varying through the model run.

This results in complete trapping of rainfall by vegetated cells in 'normal' years ($\mathrm{R} \sim 200$ mm), declining to only 25 % absorption in the driest years ($\mathrm{R} \sim 50$ mm).

Table 12.3 Distribution of annual rainfalls for five stations in western New South Wales

Meteorological station	Broken Hill (Patton Street)	Corona station homestead	Sturt's Meadows station homestead	Milparinka hotel	Tibooburra Post Office
Australian Bureau of Meteorology station number	47007	46003	46033	46018	46037
Years of record	1889–2010	1882–2010	1895–2010 (incomplete record)	1882–2009	1886–2010
Minimum rainfall (mm)	57.4	46.4	43.7	43.0	47.9
Mean rainfall (mm)	255.2	212.3	170.6	199.6	225.3
5th percentile	115.7	84.7	70.5	58.4	68.1
10th percentile	136.5	101.6	90.3	90.2	95.5
Median	241.6	202.6	156.3	174.3	196.0
90th percentile	396.7	335.4	291.0	322.5	396.9
95th percentile	449.9	379.9	329.4	427.9	442.1
Maximum recorded annual rainfall	838.0 (1974)	685.2 (1974)	396.0	916.4 (1974)	755.4 (1974)
Rainfall of wettest month	258.8 (March 1989)	241.4 (April 1974)	175.1 (March 1949; NB data for 1974 not available)	416.8 (January 1974)	398.4 (March 1949)

Source: data extracted from records of the Bureau of Meteorology, Australia

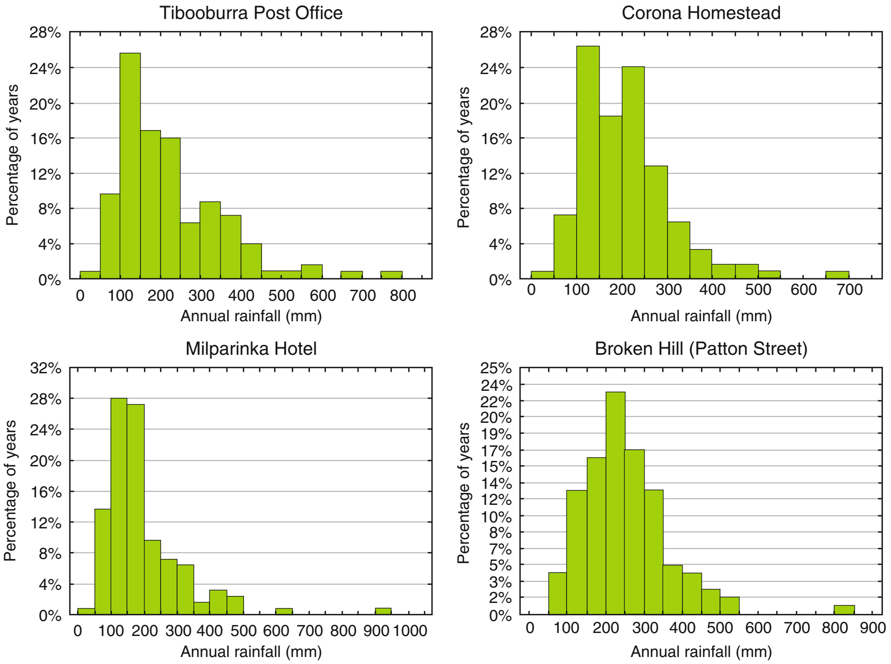

Fig. 12.7 Frequency distributions of the annual rainfall at Tibooburra, Milparinka hotel, Corona Homestead, and Broken Hill, in western New South Wales, Australia. The data are grouped into classes of width 50 mm (Source: data provided by the Bureau of Meteorology, Australia)

Vegetated cells also absorb any runon from upslope, while in contrast unvegetated cells absorb 20 % of the incident rainfall and pass the remainder downslope. Water absorbed into any cell is shared by seepage and percolation from ephemeral surface ponding among the lateral and downslope cells of the neighbourhood according to the scheme of Dunkerley (1997a). Rain was added, and other water balance calculations made, on each row of cells in turn, beginning with the farthest upslope row and working to the bottom of the slope. Water was allowed to escape from the edges and bottom of the tessellation. There are minor edge effects in the model because of this, but the behaviour of most of the tessellation is unaffected. The upslope edge (top, in the figures) also exhibits edge effects owing to the absence of runon water.

The primary model rules concern the controls on plant distribution. In the dryland being modelled, rainfall alone is insufficient to support vegetation groves, which clearly depend on the arrival of runon water from upslope. In recognition of this situation, the minimum depth of absorbed water for a cell to remain vegetated was set at 220 mm (i.e., 10 % larger than the annual mean open-field rainfall). At the end of each model ‘year’, cells with less accumulated water became unvegetated. Correspondingly, for an unvegetated cell to become vegetated, crabhole trapping and/or lateral seepage from adjoining vegetated cells were required to yield a total of 240 mm (i.e., 20 % larger than the annual mean open-field rainfall). The geometry of runoff sources and runon sinks required to yield these depths of water with the time-varying annual rainfall resulted in constant shifting of many vegetation grove boundaries, especially the lateral and downslope margins, as the model was run.

In order to account for the formation and re-healing of crabholes through subsoil contraction and expansion, a modified version of the model incorporated an increased trapping efficiency of 100 % for runon water in vegetated cells in years when the annual rainfall lay at or below the 10th percentile of the historical range of annual rainfalls (taken as 100 mm to accord with the data in Table 12.3), and the trap efficiency was returned to normal (governed by the rainfall as just described) once the rainfall returned to values above this threshold. Increasing trap efficiency under drought conditions is perhaps one of the strikingly distinctive features of the mosaic vegetation of western NSW, and appears not to have been described previously. It can be hypothesised that the mechanism of runoff trapping in crabholes, which only functions during drought conditions, would act to reduce the loss of plant cover that would otherwise occur at these times.

12.4.3 Exploring Model Operation

Given the inadequate knowledge of the ecological requirements of the vegetation in the drylands of western NSW, the sensitivity of the model to changes in the moisture threshold for vegetation to colonise or remain growing in a cell was varied from 220 mm as mentioned above, to 400 mm, in a series of steps, and the model run with an identical 50 year rainfall time series. Results show (Fig. 12.8) that as the threshold is raised, the groves become less continuous laterally, and also narrower in the direction of overland flow (top to bottom of the figure, plotted at year 21 of a 50 year simulation).

Nonetheless, contour-aligned grove structures emerge in all cases. The sensitivity of the model to a change in the moisture threshold for vegetation growth or persistence is not great. Lifting the parameter by 60 % from 220 to 400 mm resulted in a decline of only 32 % in the vegetation cover fraction, from a mean of 16.4 % to a mean of 11.2 %, averaged through all 50 years of the model run.

12.4.4 The Impact of Individual Years of Exceptionally High Rainfall

The impact of ‘outlier’ wet years was explored by modifying the 50 year runs of randomly generated annual rainfalls by inserting a single year of 800 mm (i.e., four times the mean annual rainfall, and approximately corresponding to the average of the wettest years on record at the NSW rainfall stations reported in Table 12.3, which was 798.5 mm, excluding the Sturt’s Meadows homestead record, which lacks data

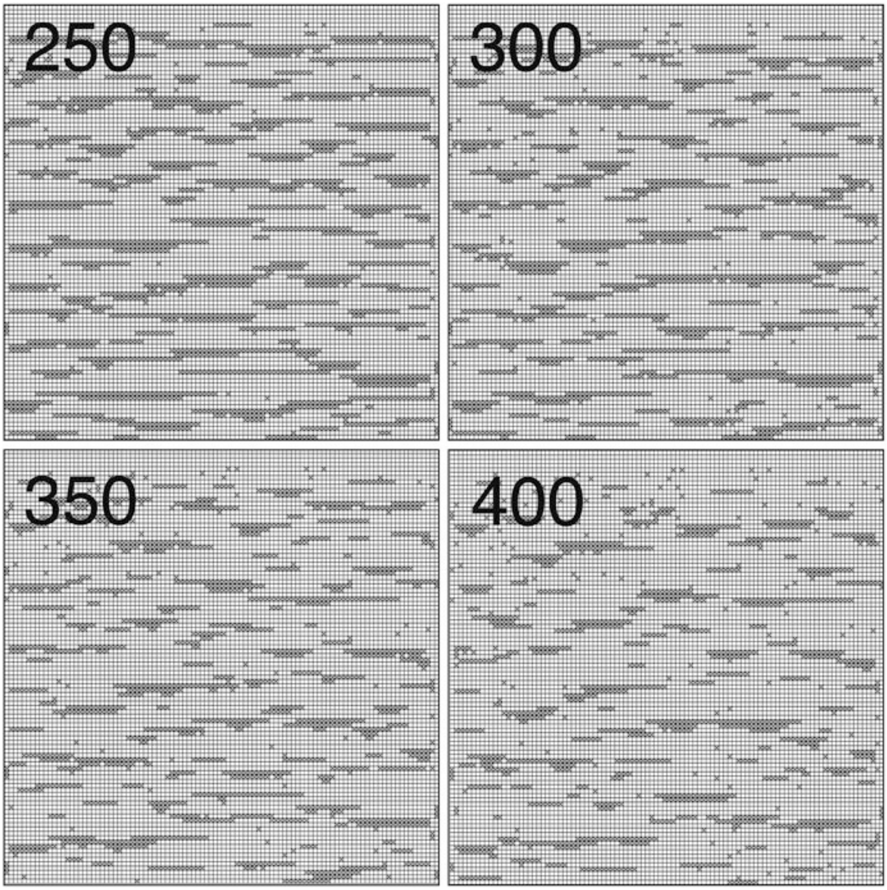

Fig. 12.8 Plotted vegetation cover across the model tessellation at year 21 of a 50 year run. The threshold accumulated infiltration depth (in mm) for a cell to become or remain vegetated is indicated in the *top left* of each plot. Effects on the grove and intergrove structure can be seen (see text for details) (Note: downslope is toward the bottom of each 100 × 100 tessellation)

for the 1974 record wet year). The wet year was inserted as year 20 of a 50 year simulation. Results showed that the vegetation groves widened considerably in the flow direction in the wet year (Fig. 12.9).

Across the 10,000 cells of the model tessellation, the plant cover jumped from 15.8 % in the preceding year (rainfall 193 mm) to 45.9 % in the wet year (rainfall 800 mm). Interestingly, owing to the persistence of the plant cover, there was a further increase in plant cover in the year following the wet year, to 47.0 % plant cover, even though the rainfall was only 290 mm. In the next year, plant cover had declined to 11.3 % with a rainfall of 141 mm. No persistent effect of the wet year on plant cover or on grove form and dimensions could be seen in the model.

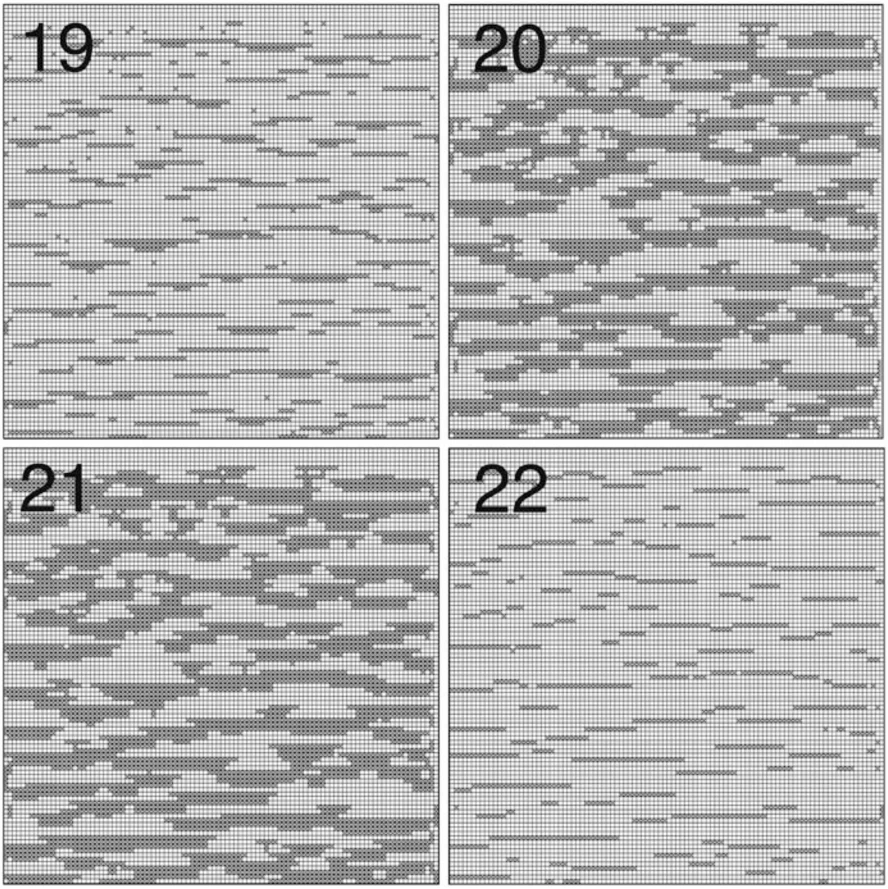

Fig. 12.9 Plotted vegetation cover across the model tessellation at years 19, 20, 21 and 22 of a 50 year simulation. Year 20 was a year of exceptionally high rainfall (800 mm). Considerable deepening of the groves can be seen in year 20 but this is rapidly lost when rainfall returns to more usual levels. See text for details (Note: downslope is toward the bottom of each 100 × 100 tessellation)

12.4.5 The Impact of Droughts and the Formation of Soil Crabholes

As noted earlier, the rainfall climate of western NSW involves very dry spells in which annual rainfalls can fall to about 50 mm, as well as rare wet years of 800 mm or more, lying at the extremes of the distribution of annual values.

In order to explore the significance of dry spells, the time series of random annual rainfalls was modified by including short runs of 3–5 years in which the rainfall was in the range 50–80 mm, i.e., lying in the lowest 5–10 % of recorded annual rainfalls (Table 12.3). Under these conditions, the model rule set incorporates the effect of subsoil contraction and crabhole formation. This is not invoked in wetter years when the subsoil clays expand and the crabholes are eliminated from the landscape. The model was run with this option and without it, in order to discern the consequences of crabhole formation for plant cover and grove and intergrove structure.

Figure 12.10 presents plots of the tessellation of cells for two versions of the cellular model, operated with and without the presence of the enhanced runon trap efficiency arising from crabhole collapse. Even though the crabholes form only intermittently in the model, and are subsequently removed, it is clear that the continuity of the vegetation banding is considerably enhanced when these are incorporated in the model than when they are not. Observation of the tessellation in successive iterations of the model also showed greater stability and persistence of the vegetation patterning when the effects of crabhole were incorporated. During year 30 of a 50 year simulation (plotted in Fig. 12.10) the vegetation cover declined to 10.4 % in the case where crabhole development was included in the model. In the same year and under the same rainfall but without crabhole formation, the plant cover declined to 5.4 %, and the vegetation groves were considerably smaller and less well connected. Significant numbers of gaps in the vegetation bands appear in this case, and in the absence of crabholes, would allow more connected overland flow paths and an increased leakiness of the landscape for water and other resources. In this situation, there is some likelihood that small flow concentrations could arise during convective storms, with associated rill or channel cutting possibly contributing to a decline in the integrity of the banded vegetation community. Interestingly, in the following year, the grove structure of the ecosystem modelled without the influence of crabhole formation shows fewer, and more widely spaced groves, than with crabholes present (Fig. 12.10f versus c). As noted earlier, this fluctuation in the spacing of groves is not seen in the field behaviour of these landscapes. Instead, groves narrow and widen in dry or wet periods, but remain in place rather than disappearing, albeit with diminished vegetation cover. This conservative behaviour can be seen successfully captured in Fig. 12.10a–c. There is a striking contrast between this behaviour and the loss of many groves between the years represented in Fig. 12.10d and e, a phenomenon that is not seen in the actual landscape.

Vegetation cover data summed over the entire tessellation (Fig. 12.11) show that in 60 year model runs with and without the incorporation of crabhole formation, there are no apparent differences in total plant cover in years with reasonable rainfall (years 1–29). A drought in years 30–32 (rainfalls 80, 84 and 88 mm respectively), vegetation cover in the absence of the crabhole effect drops to about 5 %, whilst in the presence of crabholes the cover declines only to about 12 %. For some time in years 33 and afterward, the plant cover in the non-crabhole scenario remains below that of the model with active crabholes, only recovering after about 10 years. Similar responses are seen in the second drought period (years 47–55).

Crabhole formation during very dry years and runs of years thus appears to be a very important aspect of the mechanisms that create and sustain the banded mosaic vegetation communities of the study area. The formation of crabholes within the groves during drought conditions appears to be capable of maintaining a high trap efficiency for rainfall and runon water within the groves. Without the crabholes, the

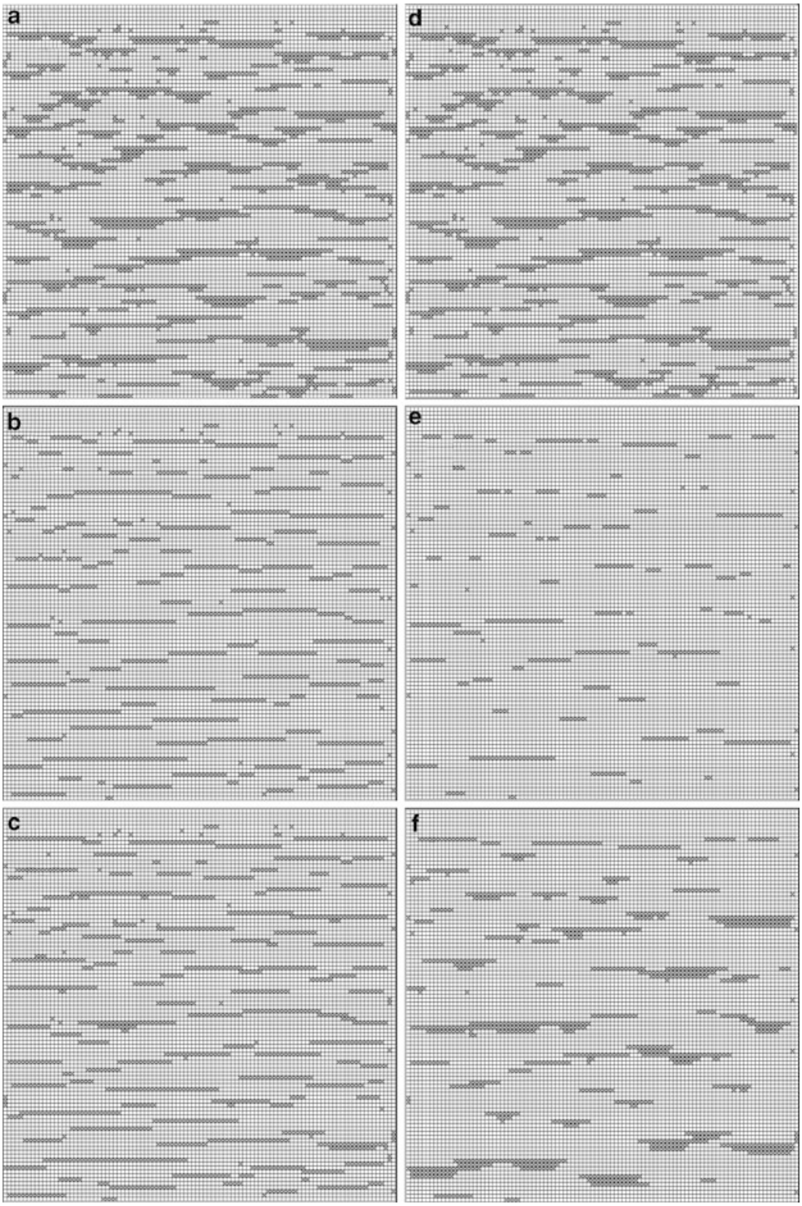

Fig. 12.10 Plotted vegetation cover across the model tessellation at years 22, 30 (drought), and 36 of a 50 year simulation. Parts (**a**–**c**) are for the model with crabhole formation included; Parts (**d**–**f**) show the model results when crabhole development was excluded. The input rainfall timeseries was the same for both model runs. Note that the pre-drought vegetation cover (Parts **a**, **d**) is the same in both models, since the crabhole development routine had not yet been invoked by years of very low rainfall. See text for details (Note: downslope is toward the bottom of each 100 × 100 tessellation)

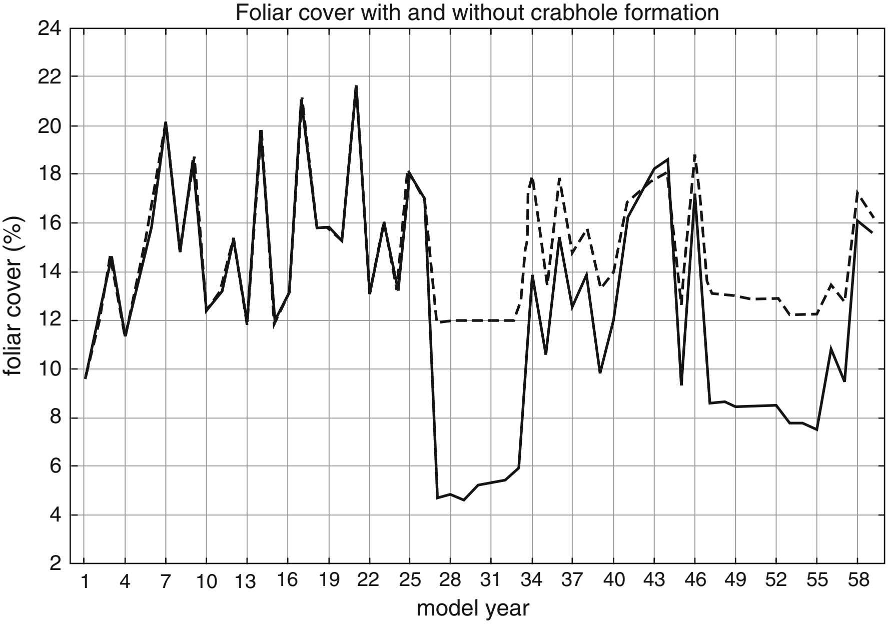

Fig. 12.11 Plotted vegetation cover summed across the model tessellation at year 30 of a 50 year simulation using a rainfall timeseries with two drought periods. The *solid line* is the vegetation cover modeled without incorporating the formation of crabholes, and the *dashed line* shows the vegetation cover with crabhole formation included. See text for details (Note: downslope is toward the bottom of each 100 × 100 tessellation)

smooth and bare soil surface, lacking organic litter and litter dams to slow runoff, and with reduced sheltering by plant canopies, would shed a larger fraction of the rainfall downslope as overland flow. This would represent a key resource loss for the ecosystem, and via positive feedback, further reduce the vegetation cover. Given that through the course of the past 160 years, grazing pressure has been added to severe, multi-year droughts, it is in some senses surprising that spectacular mosaic vegetation communities are still well represented in western NSW. However, it appears that the shrink-swell behaviour of the subsoil clays may provide at least one clue to this, resulting as they do in a critical change in the soil surface configuration during dry years that allows the ecosystem to be better supported by enhanced retention of the scarce rainwater resource.

12.5 Discussion and Conclusions

Soils and their influence on surface and subsurface hydrologic processes appear to be pivotal in the vegetation mosaics of western NSW. Many of these landscapes include exogenous dusts containing clay minerals that exhibit marked shrink-swell behaviour. This dust acts in concert with the marked interannual rainfall variability of the ENSO-dominated eastern Australian rainfall regime. One outcome is the quasi-periodic appearance of dense networks of crabholes and other closed depressions. These constitute traps for runon water that are largely unrelated to the role of the vascular plants. They may well be more important hydrologically than plants, especially after very dry years, when in their absence plant cover might be more vulnerable to collapse owing to the bare surface and high connectivity of overland flow paths across the intergroves and groves. The absence of crabholes would allow much increased loss of water from the hillslopes. Scant plant cover under such conditions would result in greatly diminished differentiation of intergrove and grove, in terms of foliar and litter cover; however, the soil-based processes may result in maximal differentiation under the same conditions.

A question not resolved here is the relative roles and importance of soil and vascular plant behaviour in the initial development of the vegetation patterning. However, a simple cellular automaton model suggests that soil collapse processes may well be able to act independently of the plant cover, and so this provides a potential mechanism for the progressive spatial organization of the soils into areas prone to repeated crabhole formation (and perhaps consequent plant colonisation) and areas that remain too dry for these processes to operate. Thus, it may be that soil pattern development is the leading process, and the emergence of patterned vegetation is an effect consequent upon the soil behaviour. It is likely, of course, that the vegetation patterning establishes a feedback process to further strengthen the hydrologic compartmentalisation resulting from the soil collapse processes. It is widely known that gilgai mechanisms operating in expansive soils can produce a wide range of vegetation patterns in association with surface pits and undulations. For instance, Maxwell (1994, Fig. 1) presented an aerial photograph of spotty vegetation in Texas, USA, that is related to soil gilgai phenomena. The pattern arising at that location from soil shrink-swell is very similar to other examples claimed to be self-organised vegetation patterns (e.g. see case study in Chap. 13). The extent to which the vegetation patterns are in fact self-emergent or are consequent upon, or interlinked with, the soil processes appears to remain unresolved, and appears to be an important but neglected aspect of the emergence of mosaic plant communities.

Consequently, further research is needed to examine these ideas in more depth. Models have successfully reproduced the kinds of emergent patterns seen in areas affected by soil freeze and thaw cycles (e.g. Kessler and Werner 2003). According to these models, bands or stripes resulting from freeze-thaw are oriented normal to the contour, reflecting differential downslope-upslope effects of gravitational loading. It has also emerged from these models that the patterns that emerge vary with the steepness of the hillslope, as for many forms of dryland vegetation patterning. Interestingly, in some environments, patterned ground and vegetation exhibit alignments that are intermediate between the common contour-aligned form described from dryland vegetation and the contour-normal form seen in cold climates (e.g. Haantjens 1965; Dunkerley and Brown 2002). It seems possible that these intermediate, oblique forms reflect a competition or interaction between hydrologic and pedologic processes. Oblique vegetation banding of the kind reported by Dunkerley and Brown (2002) remains an unresolved challenge, and the option of a hybrid origin is worthy of further examination.

The banded vegetation of western NSW has not been eliminated by drought or pastoral grazing pressure, despite being located in a region whose rainfall variability is extreme. Some of these landscapes are indeed known for their resilience and value as pasture, rather than their fragility. Ongoing field observation has drawn attention to the importance of soil processes in the development and maintenance of these systems. This includes the particular clay mineralogy that gives rise to shrink-swell behaviour in the soils, and the role of climatically-driven subsoil wetting and drying, that lead to crabhole formation and to efficient water trapping during periods when the vegetation cover is greatly diminished. Even though grove soils are much more permeable than intergrove soils, their point matrix properties (e.g. infiltrability) provide insufficient information for understanding the hydrology of the vegetation mosaics. This points to the need for more attention to be paid to physical and hydrological processes in patterned vegetation, a knowledge gap recently highlighted by von Hardenerg et al. (2010). The banded vegetation mosaics of western NSW have exhibited no upslope motion in 15 years of monitoring. Again, though many modellers assert that banded systems migrate upslope, clearly at least some actual instances do not. This suggests that either the models are incorrect, or that there are different sets of mechanisms working in different field locations. This clearly needs to be resolved by careful examination of mosaic stability in other sites, and this provides a potentially useful way to advance our understanding of these systems. The role of subsoil shrinkage and crabhole formation as explored here for the case of arid western New South Wales, Australia, provides a new mechanism that appears to be involved in the maintenance of mosaic plant communities in this area. Given that crabhole formation and disappearance in this region span multi-year to decadal timescales, linked to imperfectly understood aspects of the global climate system, the development and maintenance of the mosaic communities must be addressed using temporal scales that capture processes occurring at these timescales. In the context of these landscapes, key explanatory power lies not in a knowledge of the mean annual rainfall, but rather in an understanding of the multi-year variability of rainfall, and of the consequent changes in the soils and trap efficiency of the vegetation groves.

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Chapter 13
Case Study of Self-Organized Vegetation Patterning in Dryland Regions of Central Africa

Nicolas Barbier, Pierre Couteron, and Vincent Deblauwe

Abstract A growing body of empirical evidence supporting or opposing the mechanistic hypotheses and predictions of self-organization models exist which have been applied to the case of spatially periodic vegetation patterns found in semi-arid and arid areas around hot deserts in Africa. Overall, remarkable qualitative – and sometimes quantitative – agreement is found and presented here between the rich theoretical framework and predictions of self-organization models and the results of field and remotely sensed investigations for dryland areas in Niger, Morocco and Sudan.

13.1 Introduction

This chapter presents a case study of a self-organized semi-arid ecogeomorphic system integrating current research across disciplines and scales. It analyzes particular spatial organizations of vegetation cover in the form of spatially periodic alternations of vegetated and bare or nearly bare zones that have been described at the border of every hot desert (Deblauwe et al. 2008). The first descriptions of such striking patterns date back to the colonial period and the strategic aerial surveys that were carried out after World War II (MacFadyen 1950; Clos-Arceduc 1956). Tiger bush (i.e. banded) patterns were initially assumed to be related to geomorphic and hydrological processes, with the stripes clearly following topographic contours.

A large number of studies have since then documented the hydrology, pedology and ecology of comparable vegetation patterns around the globe. As the interactions of small-scale processes which lead to self-organization phenomena are often non-intuitive, it became possible to reproduce the whole set of pattern morphologies

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N. Barbier (✉) • P. Couteron • V. Deblauwe
IRD/UMR AMAP, Botany and Computational Plant Architecture, 34398 Montpellier, France
e-mail: nicolas.barbier@ird.fr; pierre.couteron@ird.fr; vdeblauw@ulb.ac.be

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observed in nature – whether in the form of banding or the more isotropic arrangements of labyrinthine, spotted or gapped structures – through numerical approaches in the form of spatial symmetry-breaking models (Lefever and Lejeune 1997; Rietkerk and van de Koppel 2008; Borgogno et al. 2009) (see also the discussion in Chap. 3). The models enabled an understanding of the modulation of pattern morphologies along gradients of external (environmental) constraints on the system, as well as parsimonious explanations in terms of the local processes liable to produce such emergent behaviours.

In the following section, we will summarize and test hypotheses on the mechanisms that trigger the spatial patterning. Some quantitative testing of model predictions on pattern dynamics are given in the subsequent section.

13.2 Testing the Hypothesized Ecohydrological Processes

Several self-organization models have been published recently that were able to reproduce the intrinsic vegetation patterning observed in drylands (Rietkerk and van de Koppel 2008; Borgogno et al. 2009). Functionally, these vegetation models can all be conceptually interpreted as expressing a balance between the mechanisms of short-range activation (positive feedback) and long-range inhibition (negative feedback) of biomass with surrounding vegetation modulating the local dynamics of water and vegetation biomass. The hydrological and biological processes invoked in these models differ fundamentally, especially when it comes to generating patterns in isotropous conditions (i.e. without a slope gradient and substantial water advection). Two hypothesized mechanisms for pattern generation can be distinguished:

- **M1**: a spatial diffusion of both water (unconcentrated overland flow) and vegetation with a feedback of vegetation on water infiltration. Patterns emerge because of a large difference in the diffusion coefficients of water and vegetation (HilleRisLambers et al. 2001; von Hardenberg et al. 2001);
- **M2**: the existence of positive and negative spatial feedbacks of plants on the soil-water budget via the contrasting effects induced by plant organs above ground (e.g. shade) and below ground (extended root systems). Patterns emerge because resource competition at the lateral range exceeds the facilitation range of a plant (Lefever and Lejeune 1997; Lefever et al. 2009).

Our work on isotropous gapped vegetation patterns in SW Niger (detailed in Chap. 4) shed new light on these opposing hypotheses. Using a combination of direct observation of plant organs (crowns and root systems), soil-moisture monitoring through space and time and stable isotope (^{2}H) labelling, it was found (Barbier 2006; Barbier et al. 2008) that:

1. there was no significant difference between bare and vegetated parches in recharge within the shallow root zone (350 mm mean depth above the iron pan) after average rainstorm events (30–40 mm);
2. the water half-life time in the soil was three time higher within thickets, an effect that was statistically more strongly related to the physical presence of a shrub crown above than to other soil quirks, such as improved texture, bulk density or organic matter content;
3. the root system of the dominant shrub (*Combretum micranthum*, G. Don) was mainly distributed in the upper soil layers. It was laterally very extensive and spreading under areas of bare soil. Shrubs could tap water (as confirmed through deuterium labelling) up to lateral distances equivalent to ten times their crown radius.

The above elements lend support to the second hypothesised mechanism **M2** in self-organization models and provide limited evidence for the first explanation **M1**, at least in the particular context of the study area. Moreover, it appeared that when parameterized using field data from our Niger site (i.e. adult plant size, facilitation and competition ranges and allometry exponent), the (Lefever et al. 2009) model – although deprived of any remaining degrees of freedom – is able to reproduce the observed morphology (gaps in vegetation cover) and wavelengths (the added widths of two successive sparse and dense areas) of patterns (of about 50 m). A key role in the parameterization is played by the ratio between above and below-ground plant ranges and plant size via a measurable allometry exponent (Lefever et al. 2009).

13.3 Testing Predictions on Pattern Emergence and Modulation

Whether they employ the M1 or M2 mechanism, self-organization models converge on several predictions regarding the spatial or temporal dynamics of pattern morphologies under varying environmental stressors affecting plant productivity or mortality (e.g. aridity, grazing, wood collection). Due to the size and life span of individual shrubs and the even larger space and time scales characterizing the emerging structures, testing the two mechanisms requires working at landscape to regional scales and across decadal to centurial time scales. Remote sensing surveys are starting to deliver the required coverage, while at the same time providing a sufficient horizontal resolution to allow the quantification of pattern morphology and to allow the testing of model predictions in terms of symmetry, orientation, travelling waves and dominant frequencies. Deblauwe et al. (2011) give a comprehensive summary of the predictions of self-organization models on pattern modulation:

- **P1**: Under homogeneous and isotropic conditions (i.e. on non-sloping terrain and in the absence of an influential dominant wind), stress in the form of decreased biomass productivity and/or increased mortality induced by increasing levels of aridity, grazing or woodcutting provokes a predictable succession of pattern changes. Round gaps first appear in the uniform vegetation cover and then elongate and coalesce to form a labyrinthine structure. As the aridity further increases, round spots are the only remnant of the vegetation within a bare soil matrix, preceding a final transition to bare soil (desert) (Lejeune and Tlidi 1999; Couteron and Lejeune 2001; Rietkerk et al. 2002; Meron et al. 2004; D'Odorico et al. 2006; Guttal and Jayaprakash 2007; Lefever et al. 2009).

- **P2**: Lower annual rainfall (increased aridity) results in patterns with larger wavelengths (Lefever and Lejeune 1997; Lejeune et al. 2004; Sherratt and Lord 2007).
- **P3a**: In the presence of a sufficient anisotropic environmental influence, for instance due to the presence of a slope gradient or dominant winds, all the above morphologies are forced into parallel bands elongated in a direction perpendicular to the environmental anisotropy, forming the so-called tiger bush (Lefever and Lejeune 1997; Lefever et al. 2000; von Hardenberg et al. 2001; Rietkerk et al. 2002).
- **P3b**: Runoff intensifies as the slope steepens and the competition range increases in the upslope direction, inducing an increase in the wavelength of banded patterns (Sherratt 2005).
- **P3c**: Bands are predicted to migrate in the upslope direction (Lefever and Lejeune 1997; Klausmeier 1999; Sherratt and Lord 2007).
- **P4**: Several self-organization modelling approaches have also pointed to the possibility of hysteresis loops and critical points in aridity-driven succession of vegetation states. Transitions between desert and spotted patterns (Lejeune et al. 2002), between uniform cover and gapped patterns and among the different pattern morphologies and wavelength may not occur at the same critical aridity levels during drying and wetting phases and are therefore dependent on initial conditions (Lejeune et al. 2004; Meron et al. 2004; Sherratt and Lord 2007). In this case, multiple stable vegetation states may coexist within some range(s) on the aridity scale. Since spotted patterns and bare land may represent alternative stable states, several authors have proposed that periodic vegetation may serve as warning signals of imminent and rapid ecosystem collapse or "catastrophic shift" (Rietkerk et al. 2004; Kefi et al. 2007; but see also the discussion in Chap. 3).

The map in Fig. 13.1 represents a global analysis of periodic pattern distributions (predicted and actual) obtained using pattern-occurrence data from written and digital records overlain on Landsat satellite imagery using the model MaxEnt by Phillips et al. (2006), which is a geographic distribution and habitat model. Using data on the spatial variation of a number of environmental variables related to topography, soil and climate, the MaxEnt model determines the regions where vegetation patterning might occur. Existing empirical approaches allowed sorting the relative importance of environmental variables on pattern existence (Deblauwe et al. 2008). The most important driver was the humidity index, obtained as the ratio of mean annual rainfall to mean annual evapotranspiration, with vegetation

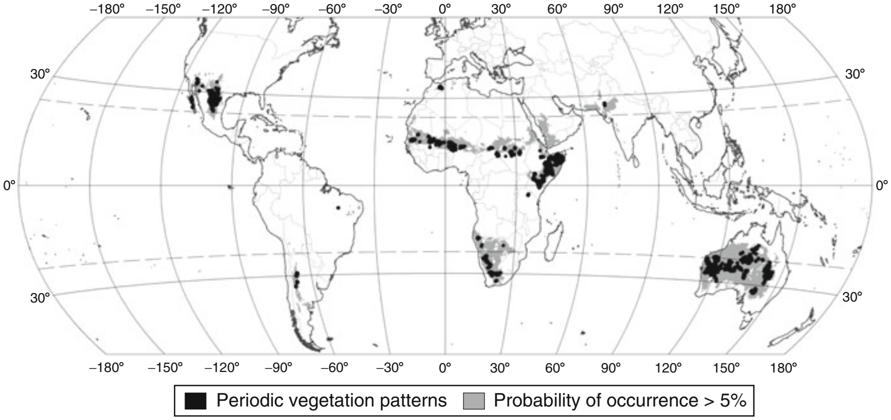

Fig. 13.1 Actual and predicted distribution of periodic vegetation patterns using the MaxEnt method

patterns occurring consistently in hot and arid to semi-arid regions. Other significant variables include the soil wilting point (mainly related to soil texture), temperature and rainfall seasonality. A number of new locations were discovered worldwide, including the first evidence of spotted patterns in Sudan and of periodic patterns in South America. Most importantly, periodic vegetation patterning appeared under predictable environmental conditions at biome scale across all longitudes, despite very different phylogenetic histories, i.e. different plant families, although possibly with a limited number of convergent life forms. These results confirmed the generality of aridity-mediated pattern emergence at the border of every tropical desert (P1).

At a smaller (regional) scale, we endeavoured to analyze pattern modulation in a quantitative way within the framework of the Fourier textural ordination methods presented in Chap. 9. In a first study, we investigated pattern emergence from homogeneous savannah cover in isotropous substratum conditions, by placing ourselves at the most humid end of the distribution of periodic patterns in the south west of the Republic of Niger (Barbier et al. 2006). The objective was to observe over time if a decrease in rainfall could trigger the emergence of gapped patterns. Using historical imagery records (from both satellite- and air-borne sensors), we were able to quantify at different points in time the proportion of gapped patterns in a 3,000 km^2 area, including a protected area (W National Park) and areas increasingly impacted by human exploitation (for wood and pasture) (Barbier et al. 2006). We thereby demonstrated that patterned areas had increased between 1956 and 1996 because of drought alone (i.e. in the National Park, where grazers and browsers density remained relatively low) and that this increase became more marked as a result of human land use and biomass uptakes. We present here data from the same protected area with a higher temporal resolution (Fig. 13.2a): from 1956 to 2010, the figure shows how patterned area first increased until early 2000s followed by a slight decrease afterwards. As a result, the area covered by gapped (and labyrinthine) pat-

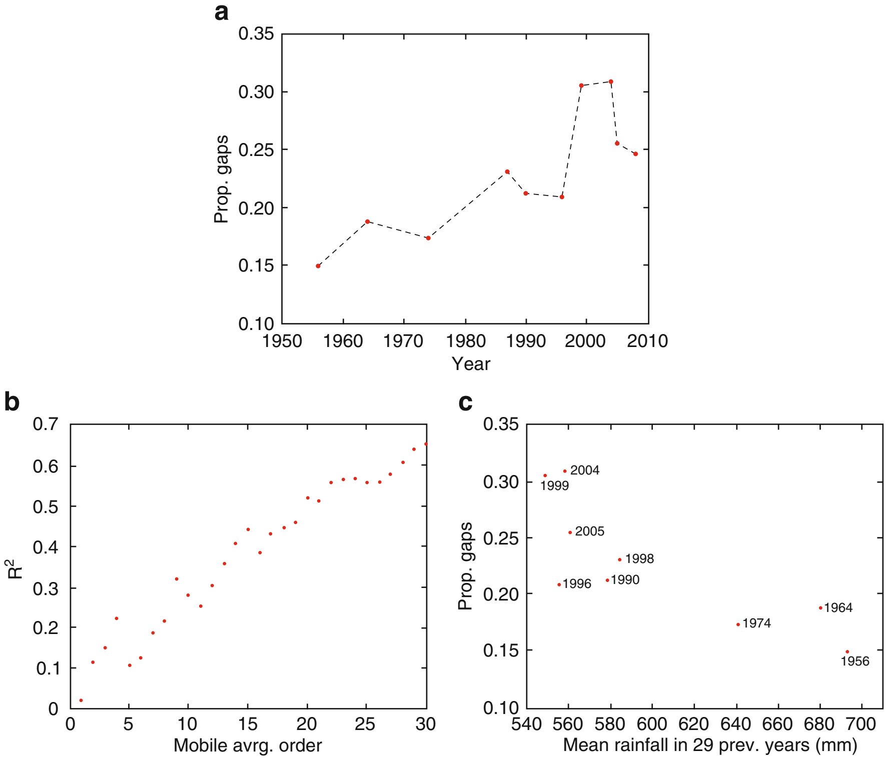

Fig. 13.2 Dynamics of gapped vegetation patterns in the W National Park (Niger) as a function of mean annual rainfall. Vegetation structure was characterized on the basis of a Fourier textural gradient analysis (see Sect. 9.3) using a set of superimposed panchromatic data sources (aerial and Spot imagery). Image spatial resolution: 10 m; analysis window size: 250 × 250 m; total window number: 14,592 windows. Rainfall statistics at the Say meteorological station (Source: DMN, Niger) (**a**) proportion of gapped patterns through time; (**b**) coefficient of determination obtained in linear regressions of gap proportion as a function of the average rainfall of previous years (number of years = mobile average order); (**c**) scatter plot of gap proportion versus the average yearly rainfall of the 29 previous years

terns had doubled in the early 2000s relative to the 1956 figure. When trying to relate pattern area statistics to mean annual rainfall, the best fits ($R^2 \sim 0.7$; $p < 0.001$) were found for the longest averaging periods, i.e. 29 years (Fig. 13.2b, c). The results suggest a notable inertia in the response of the system, which may be related to the large number of woody components in the study region. However, a potential caveat with this particular case is that, as both rainfall and pattern area present temporal linear trends, one cannot be definitive about the existence of a cause-and-effect relationship being of the observed correlation. An ideal case would have been to observe several concomitant progressions and regressions in the pattern dynamics and climatic conditions. In any case, the study brought for the first time a strong empirical support for aridity-mediated pattern emergence (part of prediction **P1**).

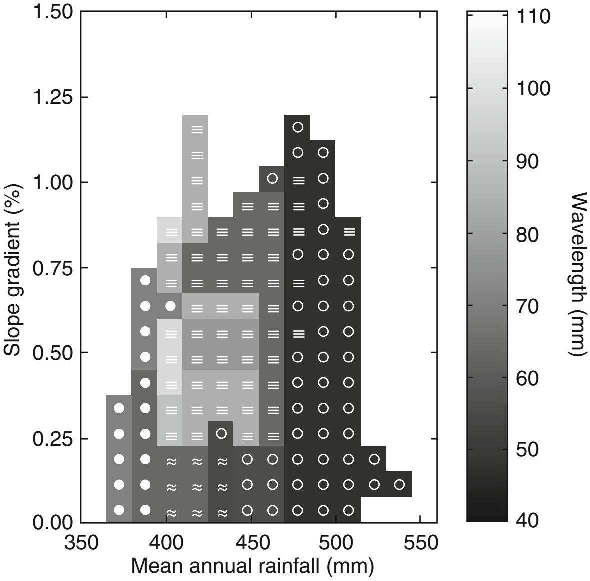

Fig. 13.3 Pattern wavelength as a function of mean annual rainfall and slope in central Sudan. Grey-scale level is proportional to the mean wavelength of the dominant pattern morphology: spotted (•), labyrinthine (≈), gapped (○) and banded pattern (≡). *White* areas represent the environmental domain where vegetation is not organised into periodic patterns (less than 15 windows featuring periodic vegetation pattern per slope–rainfall bin). Analysis based on a synchronic (2001) coverage of SPOT images (Modified from Deblauwe et al. 2011)

In a second study, in the Kurdufan province of Sudan (north-central Africa), we studied the spatial and temporal transitions between all possible morphologies (gaps, labyrinths, spots and bands) and both rainfall and slope gradients (Deblauwe et al. 2011). SPOT and Corona satellite images taken at three dates over a 35-year period were subdivided into 410 × 410-m windows and classified into the following pattern classes: non-periodic, gapped, labyrinthine, spotted and banded. The results are summarised in Fig. 13.3. The images gave some large-scale evidence that the predicted modulation sequence is valid in terms of both morphology and wavelength. Along the aridity gradient, we indeed observed the predicted sequence of gapped, labyrinthine and spotted patterns (**P1**). Above a critical slope threshold, the patterns aligned into parallel bands (**P3a**). For all slope gradients, the pattern wavelength was proportional to aridity (**P2**). Furthermore, both the sequence of morphologies and the wavelength variations were observed not only across space (synchronous comparisons) but also through time (diachronous comparisons). However, for increasing slope gradients, prediction **P3b** – which had relied on a very limited number of modelling studies – was not observed in the data, as banded pattern wavelengths appeared to decrease with slope. Hysteresis loops (multiple stable states) in pattern succession dynamics, as well as the unaccounted influence of soil factors could explain the important share of unexplained wavelength variance in the study (78 % for the banded pattern). On the other hand, no evidence supported irreversible transitions from spotted patterns to bare soils (desert) (**P4**). We even observed cases of reappearance of spotted structures after clearing for cultivation in the diachronous image series. This apparent contradiction may be due to some favourable conditions of certain soil properties (structure, texture, organic matter content, etc.) that had improved under thickets prior to cultivation. Regarding band orientation, a detailed study (Koffi et al. 2007; Deblauwe 2010) in the same region evidenced systematic deviations of band orientations in contours orthogonal to slope direction. As these deviations systematically pointed towards the direction of the dominant winds (Harmattan), this result brought some alteration to the water runoff hypothesis (**M1**), which was generally assumed applicable to sloping conditions. In particular, the role of sediment and seed redistribution by wind should be explored in more detail (see Chap. 12 for a case study on sediment transfers in Australia).

Another aspect of banded pattern dynamics that has generated long-standing debates (Tongway et al. 2001) is the prediction that bands would travel in the upslope direction (**P3c**). Multi-temporal remote sensing data allowed us to test the prediction at a number of sites in Northern America, Northern Africa and Australia (Deblauwe et al. 2012). Using Fourier analysis and a 2D cross-spectral approach (Barbier et al. 2010), we were able to quantify band-migration speed in a systematic way across regions sufficiently large enough to be statistically representative. We discovered that, not only did bands move upslope in a number of regions (up to 0.8 m year^{-1} for both a Moroccan and a Somalian sites), but also they did so at a speed proportional to pattern wavelength – a finding matching observed behaviour of some self-organization models (Sherratt and Lord 2007). Analyzing the rate of band migration in the Chihuahuan Desert (Northern America), for example, suggested that upslope colonization and downslope regression episodes did not occur synchronously, but rather in response to wet and dry spells, respectively. Upslope migration might therefore be interpreted rather as a reaction to climate variability than as an endogenous property of all banded systems. However, the question of band migration remains largely open as most of the variability in dynamics is still unexplained. For instance, the monitored dryland systems in Australia did not show any movement during the 40-years observation period.

13.4 Conclusions

What this study has highlighted is the usefulness of self-organization models as a theoretical framework allowing the guidance of empirical research. Indeed, despite their simplistic nature and the sometimes diverse mechanisms invoked, a number of convergent predictions were produced, which were very consistent – at least qualitatively and, sometimes, quantitatively (Lefever et al. 2009) – with the behaviours of real systems. Sadly, the prospect of calibrating fully quantitative models of these ecosystems seem a long way ahead, principally because of the very limited research efforts dedicated to estimating parameters in arid and semiarid landscapes within which periodic patterns extensively occur. In practice, competences from a range of fields is required, spanning ecology, hydrology, soil physics, remote sensing and non-linear systems modelling. Although studies from each of these individual fields have been carried out and published, a lack of integration has until now hampered decisive breakthroughs. Manipulative experiments in the field or in shadehouses may also help to disentangle imbricated spatial effects mediated by plants on their environment and resources.

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Chapter 14
Abandonment of Agricultural Land, Agricultural Policy and Land Degradation in Mediterranean Europe

Anthony J. Parsons

Abstract Abandonment of agricultural land has been widespread phenomenon throughout Western Europe since the middle of the last century; a trend that up until the mid 1990s was exacerbated by the European Common Agricultural Policy. Typically, in dryland parts of Europe, abandonment of these agricultural areas causes them to revert to scrubland. Land abandonment does not result in a single response. This chapter examines examine the various deleterious processes of landscape change that may result from land abandonment in dryland regions of Europe, the changing patterns of vegetation that may result, and their environmental consequences. Abandonment of agricultural land almost always results in an increase in vegetation cover. Deleterious consequences include the formation of stone pavements, soil crusting, and gully formation. In lowland settings abandonment of terracing can lead to landsliding, terrace destruction and an increase in catchment connectivity. Responses to land abandonment vary widely dependent on conditions prior to and at the time of abandonment and according to the type of vegetation that colonizes the abandoned land. In the long term agricultural abandonment leads to the development of spatial pattern in the vegetation, which can confound simple relationships between vegetation cover and runoff and erosion rates. Empirical studies have documented the complexity and diversity of changes that accompany such degradation. Modelling the effects of abandonment of agricultural land is in its infancy, but is vital for informed policy-making. A more holistic view is needed of the effects of abandoning agricultural land so that the environmentally sound policies can be implemented.

A.J. Parsons (✉)
Sheffield Centre for International Drylands Research, University of Sheffield,
Sheffield S10 2TN, UK
e-mail: a.j.parsons@sheffield.ac.uk

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14.1 Introduction

Abandonment of agricultural land has been widespread phenomenon throughout Western Europe since the middle of the last century. Traditionally, increases in agricultural production have been accommodated through increases in the area of cultivated land (Slicher van Bath 1963), but more recently they have been obtained through greater productivity. A consequence of the higher yields achievable through a combination of mechanisation and the use of pesticides and fertilizers has been rural depopulation and the abandonment of both grazing lands having low stocking rates and areas of low-input arable land. Up until the mid-1990s, this trend was exacerbated by the European Common Agricultural Policy (CAP) (Strijker 2005). Even so, Bignall and Mc Cracken (1996) estimated that 82 % of farmland in Spain still fell into the category of low-intensity systems. A second cause of abandonment of agricultural land is its misuse. Land is abandoned because productivity falls due to erosion, loss of fertility, and poor soil management. Typically, in the dryland parts of Europe (see Fig. 1.1) abandonment of these agricultural areas causes them to revert to scrubland. For example, within the Encinares del Rio Alberche y Cofio, Special Conservation area in central Spain, Romero-Calcerrada and Perry (2004) showed an approximate doubling of scrubland between 1984 and 1999 and, correspondingly, a more than halving of the pastureland, and a smaller reduction in cropland.

These pressures on low-intensity agricultural land are expected to continue into the future as the benefits of increased mechanisation of agriculture are expected to remain high (Strijker 2005). For the most part the pressures will be focused on the most marginal agricultural land, but this is not always the case. In some environments the abandonment of agriculture is considered beneficial. Increased vegetation cover may reduce soil erosion and increase plant biodiversity, thereby providing habitats for a wider range of animal species. However, in a dryland setting such benefits may not be achieved so that policies such as set-aside, whereby farmers were paid to take land out of production, pursued until 2008, far from have environmental benefits, may have resulted in further land degradation. Furthermore, the replacement of the mosaic of agricultural land with a more uniform scrubland may decrease habitat variety and hence biodiversity (Hill et al. 2008).

It is not the case that land abandonment results in a single response. Studies of the effects of land abandonment have often reported conflicting results, and their differences may often be a result of local circumstances and the nature of land abandonment. In this chapter we focus on the detrimental effects of land abandonment. We examine the various deleterious processes of landscape change that may result from land abandonment in dryland regions of Europe, the changing patterns of vegetation that may result, and their environmental consequences. The review is based on existing literature, which shows considerable spatial bias within the dryland areas of Europe. In comparison to other countries, much work has been undertaken in Spain. Consequently, many of the examples used in this chapter are drawn from this country.

14.2 Landscape Processes Consequent upon Agricultural Abandonment

Several forms of agricultural abandonment may be identified, for which the process responses may differ. However, in almost all cases it eventually results in an increase in vegetation cover. The widely held view is that an increase in vegetation cover results in a reduction in runoff and erosion (Elwell and Stocking 1976; Francis and Thornes 1990). However, significant divergences from this typical response have been observed within Mediterranean Europe. Shrub colonization following abandonment of sloping farmland in the Central Pyrenees was reported to reduce runoff and sediment yield (Ruiz-Flaño et al. 1992; Ruiz-Flaño 1993), but the use of fire to reduce shrub cover and improve grassland for grazing resulted in rill and sheet erosion with the result that the fields became stone pavements. The significance of wildfires on abandoned farmland has also been identified by Puigdefabregas and Mendizabal (1998), who reported an increase in Spain in the annual area burnt by fires by 600 % between 1960 and 1990 (Prieto 1993).

In Portugal, abandonment of lowland farmland has led to crusting of soils, high runoff coefficients and sediment yields (Nunes et al. 2010). Soils rich in sodium are particularly susceptible and show cracking and very slow rates of plant colonization. As a result of slow colonization of plants, abandoned fields in Spain have been shown to be subject to gully formation and, in some cases, the development of badlands (López-Bermúdez and Romero-Díaz 1989; Cerdà 1997). Abandonment of lowland agricultural land in southeast Spain that has been subject to terracing has been shown to result in the collapse of terrace walls due to small landslides (Lasanta et al. 2001). Whereas prior to abandonment such damage to terraces was repaired, in the absence of such maintenance it may result in the development of piping (Romero-Díaz et al. 2007), and the formation of gullies (Lesschen et al. 2008a, b). The latter authors calculated rates of erosion on abandoned terraces of 87 Mg ha^{-1} a^{-1}, which is of the order of that reported in some semi-arid badlands. Lassanta et al. (2001) showed that, following small landslides, trampling by livestock contributed to erosion of landslide scars. Although plant recolonization after abandonment was found to have occurred in a study in southeast Spain by Cammeraat et al. (2005), thereby reducing shallow landsliding, a rapid decrease in soil permeability with depth resulted in more deep-seated failures that the plant roots were powerless to prevent. A further effect that has been reported for abandoned terraced agricultural land has been an increase in connectivity in the drainage network. Whereas well maintained terraces increase infiltration, their breaching allows runoff to progress from one terrace to the next. Meerkerk et al. (2009) reported a 3.2-fold increase in the area contributing sediment to a river system in southeast Spain between 1956 and 2006, and estimated that if all terraces were removed there would be a further 6-fold increase compared to the 2006 estimate.

A key factor (*ceteris paribus*) in the fate of abandoned land may be the state of the land at the time of abandonment. Dunjó et al. (2003) examined the physicochemical properties of soils under a range of land uses in Northeast Spain. They found that cultivated and recently abandoned locations had the lowest values for soil organic matter, nitrogen and water-holding capacity, and argued that soil management and condition at the time of abandonment were crucial in maintaining soils against progressive degradation. In a study on a range of substrates in Lesvos, Komas et al. (2000) identified a critical limit for soil depth, below which shrubby vegetation would not regenerate following land abandonment. Furthermore, this critical depth varied with substrate as a result of the different water-holding capacities of soils on these substrates. These authors defined two soil depths, termed *critical* and *crucial*, that are important for the consequences of land abandonment. Below the critical depth (between 25 and 30 cm) recovery of natural vegetation is very slow and erosion processes may be very active. The crucial depth (4–10 cm) is that below which perennial vegetation cannot be supported and any soil remaining at the time of abandonment will be eroded away.

A particular type of abandonment is that reported by Piccarreta et al. (2006), which characterises the consequences of changing CAP policy during the 1990s. Following subsidies to cultivate wheat, areas of former badland in Basilicata, Italy were remodelled to facilitate their use for agriculture (Reg. CEE 1765/92). This remodelling comprised flattening of landforms, reduction of gradients and breaking up of the soil surface. The subsequent implementation of set-aside (Reg. CEE 2078/92) led to the abandonment of many of these areas of cultivation, especially the former badlands which had low productivity. As a result rills, gullies mudflows and landslides led to significantly enhanced rates of soil erosion.

What is evident from these examples is that the response to land abandonment varies widely dependent on conditions prior to and at the time of abandonment. In addition, responses may vary according to the type of vegetation that colonizes the abandoned land. Bochet et al. (2000) showed distinct differences in the ability of *Stipa tenacissima*, *Anthyllis cytisoides* and *Rosmarinus officinalis* to control runoff and sediment loss. Consequently, in response to both pre-and post-abandonment differences the effects of land abandonment on land degradation will always be locally controlled. Empirical studies of the type reviewed in this section, therefore, have done no more than build up a body of case studies. From such cases studies it is difficult, if not impossible, to make predictions about the effects of land abandonment under any one set of environmental conditions. The multidimensionality of the controls and their interactions renders a sufficiently large data set a daunting task. Furthermore, typically these studies contrast pre-and post-abandonment with little attention to the sequences of change in the long term.

14.3 Long-Term Effects of Land Abandonment

One of the most significant changes that occur when agricultural land reverts to scrubland is the development of spatial pattern in the vegetation. In studies of runoff and erosion rates in semi-natural dryland vegetation, Parsons et al. (1996) attributed the greater rates of runoff and erosion on shrubland compared to grassland largely to differences in the spatial patterns of the two vegetation communities. Similarly, Puigdefabregas (2005) argued that vegetation-driven spatial heterogeneity in the landscape could confound simple relationships between vegetation cover and runoff and erosion rates. This argument accords with the findings of Parsons et al. (1996) on shrubland and grassland which showed a greater vegetation cover on the former (44 %) than on the latter (33 %). Cobo and Carriera (2004) showed that crop abandonment and colonization by *Artemisia* enhanced soil heterogeneity which, in turn, promoted heterogeneity in the growth of *Ratama* leading to enhanced erosion in inter-spaces.

The development of spatial heretogeneity in vegetation and soil properties after abandonment of land formerly used for cereal production was studied by Lesschen et al. (2008a, b) at a site in Southeast Spain. These authors used a space-for-time approach by studying two sequences of fields ranging from currently fallow through recently abandoned, long abandoned to semi-natural vegetation (unused for more than 50 years). Their study showed that vegetation recovery in semi-arid environments appears to be very slow such that species typical of abandoned land were present even after 40 years. The recovery rate was shown to be two to three times that of more humid environments (e.g. Beaufoy 2001) and, consistent with Kosmas et al. (2000) differed with substrate. This study also showed that vegetation patchiness increased with time since abandonment. The significance of vegetation patchiness for connectivity in runoff and sediment dynamics is examined further in Sect. 9.4.

Although space-for-time substitutions of this type allow greater insight into sequential differences following land abandonment, they are incapable of providing an understanding of the dynamics of change. Furthermore, they suffer from the same fundamental weakness of the before and after studies, namely that they can be no more than case studies of a set of changes that have been shown to be extremely variable in response to local conditions.

14.4 Modelling the Effects of Land Abandonment

In contrast to the relative wealth of empirical studies of responses to land abandonment, there has been a relative dearth of modelling studies that examine the effects of land abandonment on vegetation characteristics, runoff and erosion. One of the few studies to take a modelling approach has been that by Boer and Puigdefabregas (2005) who used a distributed soil erosion model to simulate the effects of vegetation patterns on runoff and sediment production. They used autocorrelation length to characterise patch size and showed that under most conditions spatially structured vegetation led to higher runoff rates compared to a spatially uniform vegetation cover, and that this difference varied with storm intensity, being highest for intermediate (40–50 mm h^{-1}) intensities (see Fig. 14.1). Similarly, more erosion was observed from the spatially structured vegetation (see Fig. 14.2).

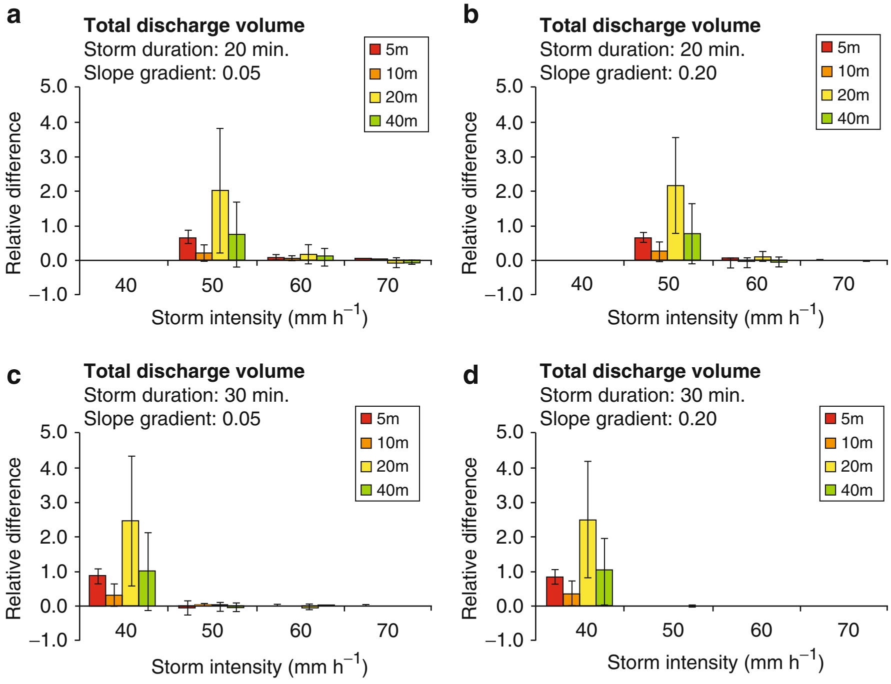

Fig. 14.1 Relative differences in predicted runoff coefficients in response to storms of 20 min (**a**, **b**) and 30 min (**c**, **d**) and different intensities among hillslopes with different spatially structured vegetation (autocorrelation lengths 5–40 m). The graphs show mean values and standard deviations for three realizations of each vegetation pattern (After Boer and Puigdefabregas 2005)

A somewhat more conceptual modelling approach is that presented by Garcia-Ruiz and Lana-Renault (2011), who propose patterns of change in runoff and erosion dependent upon the land use prior to abandonment (Fig. 14.3). These authors propose that runoff and erosion may either decrease or increase following abandonment leading in time to totally degraded stone pavement or stable shrub cover or forest.

The modelling approach presented by Boer and Puigdefabregas (2005) does allow the exploration of scenarios of land abandonment that, provided with suitable validation (as is given by these authors), and may overcome the weakness of a compendium of empirical case studies. However, what is lacking from this study is any understanding of the dynamics of change since it is based on steady-state runs of the LISEM soil erosion model with different input parameters. If the conceptual model of Garcia-Ruiz and Lana-Renault (2011) is to developed into a tool for both understanding the thresholds that govern which of the two proposed endpoints is reached at any location and for management purposes, then the dynamics of change need to be modelled. Such dynamics will need to incorporate at least all of the various processes described in Sect. 14.2.

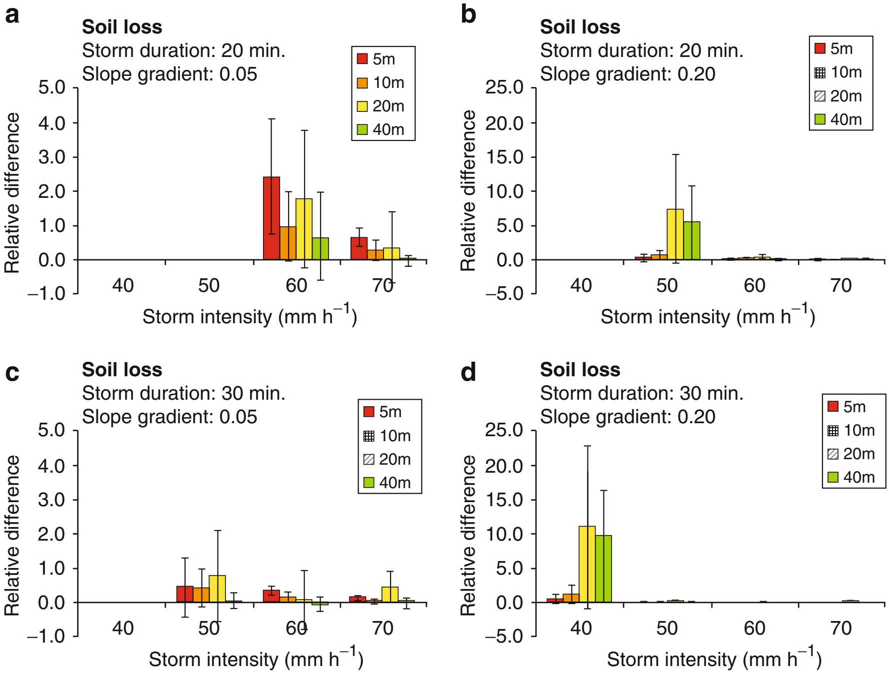

Fig. 14.2 Relative differences in predicted soil loss in response to storms of 20 min (**a**, **b**) and 30 min (**c**, **d**) and different intensities among hillslopes with different spatially structured vegetation (autocorrelation lengths 5–40 m). The graphs show mean values and standard deviations for three realizations of each vegetation pattern (After Boer and Puigdefabregas 2005, reproduced with permission of John Wiley & Sons)

14.5 Conclusion

Abandonment of agricultural land in the dryland parts of Europe has occurred in response to societal and technological changes, but also in response to agricultural policies within the European Union. The environmental changes that have taken place on this abandoned land have been unplanned, and in many instances led to unforeseen consequences in terms of land degradation. The process looks likely to continue through the coming decades, so that further land degradation is likely if current practices are continued. Against this background, there have been numerous empirical studies that have documented degradation and process changes that accompany such degradation. What these empirical studies have highlighted most of all is both the complexity and the diversity of responses to the abandonment of agricultural land. Modelling the effects of abandonment of agricultural land is in its infancy and has failed to take advantage of the wealth of empirical data to develop a dynamic understanding of land degradation. Such modelling is vital for

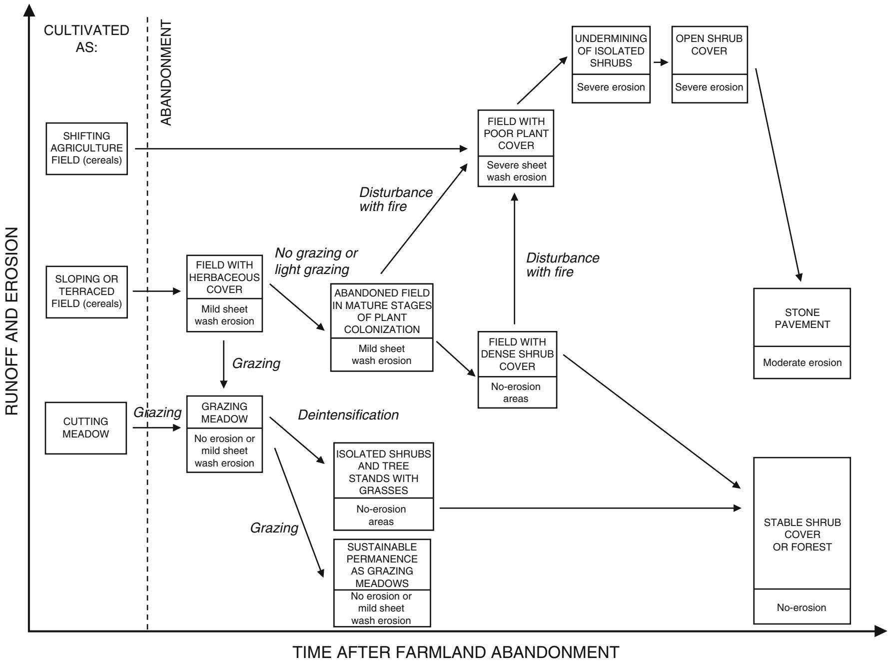

Fig. 14.3 Evolution of runoff and erosion after farmland abandonment under three scenarios: abandonment after use for shifting agriculture, sloping or terraced fields, and cutting meadows (Garcia-Ruiz and Lana Renault 2011, reproduced with permission of Elsevier)

informed policy-making. Furthermore, a more holistic view is needed of the effects of abandoning agricultural land so that the environmentally sound policies can be implemented.

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Chapter 15
Land Degradation in Drylands: Reëvaluating Pattern-Process Interrelationships and the Role of Ecogeomorphology

Eva Nora Mueller, John Wainwright, Anthony J. Parsons, Laura Turnbull, James D.A. Millington, and Vasilios P. Papanastasis

Abstract In this book we have argued that improved understanding of land degradation in drylands needs a problem-centred multidisciplinary approach. Specifically, we have argued for an ecogeomorphic approach. In this concluding chapter we review successes and shortcomings of this approach, identify key challenges that need to be overcome, and present the conceptual and methodological advances that need to be made to overcome these challenges. There has been a wealth of research investigating patterns and processes separately at small spatial scales, and, some advances in linking ecology and geomorphology have been made. However, there remains little in the way of true integration across the disciplines that deal with both ecogeomorphic patterns and processes. To overcome this weakness,

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E.N. Mueller (✉)
Institute of Earth and Environmental Science, University of Potsdam, 14476 Potsdam, Germany
e-mail: eva.mueller@uni-potsdam.de

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

A.J. Parsons
Sheffield Centre for International Drylands Research, University of Sheffield,
Sheffield S10 2TN, UK
e-mail: a.j.parsons@sheffield.ac.uk

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.D.A. Millington
Department of Geography, King's College London, Strand, London WC2R 2LS, UK
e-mail: james.millington@kcl.ac.uk

V.P. Papanastasis
Laboratory of Range Ecology, Aristotle University, 54006 Thessaloniki, Greece
e-mail: v.papan@for.auth.gr

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it is imperative that the lessons of ecology are learned – to value truly coupled eco-hydro-geomorphic studies, in which biogeochemistry, plants, geomorphology, soils and hydrology are all well represented and experimentally manipulated – and that the lessons of geomorphology and hydrology are learned: to value observational studies in which ecological measurements are coupled with hydrological and geomorphological measurements, and the role of exogenous forces is explicitly recognized. No one approach will be applicable to understanding land degradation in drylands. Unique settings, both biophysical and cultural, mean that the solutions to land degradation differ from place to place. Furthermore, evolutionary changes in drylands – degraded or otherwise – mean that methodological approaches employed to study the system may need to be fluid. We conclude the chapter by identifying five key challenges for land-degradation studies in drylands. First, a common language needs to be developed. Secondly, the problem of scale and scale interactions needs to be overcome. Thirdly, the lessons of complexity science need to be accepted and acted upon. Fourthly, the understanding of the interactions of ecogeomorphic processes and people needs to be improved. Fifthly, management strategies for combatting land degradation in drylands need to be developed taking account of scientific advances, but not waiting for an "ultimate solution" that will never arrive.

15.1 Introduction

Patterns and processes are mutually causal. Therefore, in this book, which is the primary outcome of a workshop funded by the European Science Foundation (ESF), we have attempted to take a centripetal, ecogeomorphic approach to improve understanding of pattern-process linkages in drylands (Fig. 1.3). In the preceding chapters, we have explored how an ecogeomorphic perspective – the integration of ecology and geomorphology – can improve our understanding of patterns and processes of land degradation in drylands. Multiple examples have been presented which demonstrate that the emergence of patterns develops from a complex interplay of processes and associated horizontal and vertical fluxes of energy and materials over multiple spatial and temporal scales. These processes and fluxes have, for the most part, been considered in isolation of each other, in both monitoring- and modelling-based studies. However, interactions between these fluxes, feedbacks linking different components of the ecogeomorphic system, and the existence of self-organizing states where resource patterns develop as an optimized response to climatic and landscape conditions intrinsic to dryland systems require interdisciplinary research approaches. Interdisciplinary research is also vital if the rôle of human activity in the dynamics of these systems is to be appropriately investigated and understood. It has become increasingly evident throughout the preceding chapters that the development of integrated ecogeomorphic studies need to encapsulate both field-based experimentation and model development, which together will enable the effects of drivers of land degradation and emergent phenomena to be disentangled. Consequently, this integrated approach will permit an understanding of the controls on the dynamics of land degradation which is a prerequisite for its successful management.

In this concluding chapter we review the successes and shortcomings of current ecogeomorphic approaches to land degradation in drylands, and set this approach into a broader scientific context. From this review, we identify key challenges that need to be overcome in order to advance research within an ecogeomorphic perspective and identify conceptual and methodological advances that should be made to overcome these key challenges.

15.2 Land Degradation in Drylands: State of the Art

Land-degradation studies have made great headway in recognizing the importance of interactions and feedbacks between ecological and geomorphological processes on shaping patterns and processes in drylands (e.g. Okin et al. 2006; Peters and Havstad 2006; Stewart et al., in press; Turnbull et al. 2008; Wainwright et al. 2002). However, there remains a tendency for land-degradation studies in drylands to focus on isolated components of the system or at best one or two ecogeomorphic feedbacks (following the centrifugal approach, as was depicted in Fig. 1.3). Furthermore, studies are typically undertaken at one scale of inquiry which is a major limitation since ecogeomorphic feedbacks tend to span multiple spatial and temporal scales.

Through exploring ecological and geomorphic research during the ESF workshop and through the compilation of the preceding chapters of this book, we have been able to evaluate the commonalities and disparities in ecological and geomorphic research approaches, areas where connections have been made, and areas where deficiencies remain. Here, we outline some of the key successes and shortcomings of ecogeomorphic research in drylands that have been identified in the preceding chapters and outline a research agenda in order that future research may overcome these shortfalls.

The preceding chapters of this book have demonstrated that there has been a wealth of research investigating patterns and processes separately at small spatial scales (Chap. 4), and we now have a reasonably comprehensive understanding of both ecological and geomorphic processes at these small scales. An increasing array of tools has been developed to facilitate high-resolution data collection which has been beneficial in improving the parameterization and testing of ecological and geomorphic models (Chap. 8). These data have also facilitated the study of long-range processes (Chap. 5), but there has been a general failure to integrate research on the short- and long-range scales. In addition, major advances in technology and freedom of information over the recent decades have enabled the observation and evaluation of pattern in drylands over significant parts of the Earth's surface. Furthermore, major accomplishments have been made in modelling isolated components of the ecogeomorphic system, which to varying extents, work reasonably well for simulating processes at specific spatial and temporal scales and specific environmental conditions (Chap. 7). These model developments have resulted from increases in process understanding derived from empirical studies, and in some cases, have highlighted gaps in empirically based process understanding, thus providing new directions for empirical research. These system-component models have provided a platform upon which to identify the challenges of developing integrated ecogeomorphic modelling approaches, not least those relating to the effective incorporation of uncertainty in measurement and model structure (Chap. 10). Finally, this book has provided the first interdisciplinary platform for ecogeomorphic studies in drylands, and has provided clarification of terms that are used across disciplines, but often with different meaning. The case studies show how differently disciplines deal with the analysis of vegetation-terrain interactions. For example, Barbier et al. (Chap. 13) studied the types of different vegetation patterns derived from remote sensing imagery or through numerical approaches in the form of spatial symmetry-breaking models using advanced Fourier spectral analysis and related gap proportion and pattern wavelength of vegetation patterns to regional topographic or annual climatic gradients to understand potential trigger mechanisms of spatial patterning. No process mechanisms regarding plant functioning, matter or energy fluxes were included in their approach. Similarly, Dunkerley (Chap. 12) employed a cellular automaton (CA) model on an annual time step to model explicitly the changes of vegetation establishment and growth as a function of water availability using simple rule-based algorithms. In contrast, in Chap. 11 Turnbull et al. discussed the application of a process-based ecogeomorphic model to the understanding of vegetation and surface changes in an explicit way. The model demonstrates that more process and spatial detail are often required to address the limitations imposed by equifinality.

15.3 Shortcomings of the State of the Art and How to Overcome Them: A Research Agenda

In land-degradation studies, the state of the art is a compendium of more-or-less independent pieces of research, lacking an overarching conceptual framework. Although some advances in linking ecology and geomorphology have been made through the (International) Long-Term Ecological Research ([I]LTER) programmes and through US Department of Agriculture-Agricultural Research Service experimental catchments, there remains little in the way of true integration across the disciplines that deal with both ecogeomorphic patterns and processes (but cf. Stewart et al., in press: see Chap. 11). The different research programmes that have evolved within ecology and geomorphology and contrasting research agendas have led to the development of somewhat different research approaches that now challenge the integration of these disciplines because of a lack of compatibility and acceptance of different research approaches across the disciplines. For example, many ecologists have a preference for replicated multi-factorial experiments which are not feasible in large-scale (eco)geomorphic studies that, as a result, tend to be unreplicated, not least because of the importance of the uniqueness of place as discussed in Chap. 5. There has to be a mutual acceptance and understanding of different research approaches and willingness to compromise in order to be able to carry out coupled ecogeomorphic studies. The challenge therefore is to determine how an ecogeomorphic approach can combine these different methods and research strategies to answer research questions, test hypotheses, and enable theoretical advances to be made in degradation studies in drylands. For these advances to be made, it is imperative that the lessons of ecology are learned – to value truly coupled eco-hydro-geomorphic studies, in which biogeochemistry, plants, geomorphology, soils and hydrology are all well represented and experimentally manipulated – and that the lessons of geomorphology and hydrology are learned: to value observational studies in which ecological measurements are coupled with hydrological and geomorphological measurements, and the role of exogenous forces is explicitly recognized (Turnbull et al. 2012). The optimal research strategy for land-degradation studies in drylands will apply these research approaches in tandem – replicated (where possible), experimental manipulations, combined with both short- and long-term observations and modelling-based approaches. Clearly though, to address land-degradation problems in drylands, a multitude of these methodological approaches need to be employed, with triangulation between observation, experimentation and modelling. Similarly, the different approaches to the "top-down" or "bottom-up" specification of scientific problems (see discussion in Chap. 2) need to be accommodated, and different styles of research into similar questions not dismissed out of hand.

However, any research strategy for land degradation in drylands has to recognize that no one approach will be applicable to all drylands. Unique settings, both biophysical and cultural, mean that not only may the solutions to land degradation differ from place to place, but so, too, may the research approaches needed to identify those solutions. Different styles and approaches will be particularly important for breaking the 'hierarchical relationship' in understanding physical and social causes of land degradation (Chap. 2), to ensure we move beyond understanding human and physical components of dryland systems as simply providing the initial and boundary conditions for one another. Such work may bring new uncertainties (Chap. 10) and seem risky for individual researchers, but it is necessary to move beyond narrow scientific disciplinary specialization. For example, degrading drylands are ever evolving and because of the contingency this evolution produces, replication may be impossible. The lessons of complexity science need to be learned to bring together the different components of the ecogeomorphic dryland system in a unified and appropriate way. Furthermore, place-based, comparative, and long-term research is needed to understand coupled social-ecological systems (Carpenter et al. 2009), in order to help drive social-ecogeomorphic research and the management of dryland systems.

While several modelling studies have investigated hydrological, geomorphological and ecological patterns simultaneously (e.g. Wainwright et al. 2002;

Mueller et al. 2007), some taking a complexity-based perspective (Stewart et al., in press; Chap. 11), there have been no field studies that have convincingly looked at hydrological, geomorphological and ecological processes simultaneously. Furthermore, there are a limited number of studies within ecology and geomorphology that integrate data sets from the plot scale up to regional scales. In general, long-range processes have not been so well studied as short-range processes, which is largely because of methodological limitations. In general, there has been a focus in geomorphology on the study of horizontal fluxes of materials and energy, while in ecology there has been more of a tendency to focus on vertical fluxes of materials and energy (but see Turner 1989). A pattern-based approach leads to the need to integrate across scales, because once pattern formation is viewed as the outcome of linking of short- and long-range processes (see Chaps. 2 and 3) existing scale-bound approaches are found lacking. Furthermore, the variables of interest within ecology and geomorphology are generally very different, with only soil moisture and nutrients being commonly studied within both disciplines. In fact, no strictly true ecogeomorphic case studies exist that explicitly consider the critical ecogeomorphic components outlined in Fig. 1.3. Even the studies that were presented in Chaps. 11, 12, 13 and 14 are still limited in the extent of their true ecogeomorphic coupling.

Monitoring cross-scale interactions and feedbacks using experimental approaches is inherently challenging, as current experimental approaches favour monitoring processes at one spatial scale. The continued development and refinement of ecogeomorphic modelling approaches is likely to be the most valuable research tool available for investigating cross-scale interactions and feedbacks in land-degradation studies. While the application of discipline-specific models can be successful at limited spatial and temporal scales, these models tend not to include system feedbacks between multiple elements of the ecogeomorphic systems (e.g. Baas and Nield 2010; Tietjen et al. 2010, see Chap. 7), and hence do not allow an integrated functioning of the system and emergent phenomena. Modelling becomes more challenging at larger spatial scales due to computational limitations, limited amounts of available ecological and geomorphological data that span the same spatial and temporal scales and a more limited understanding of process interactions.

Continued refinements of fine-scale deterministic models are to be encouraged, but the obstacles in translating these results to different scales need to be explored more explicitly (Marston 2010). During the ESF workshop, several key areas were highlighted that are necessary to underpin developments for the successful implementation of ecogeomorphic modelling in land-degradation studies. It was established that we need to continue advancing process understanding within ecogeomorphology, to understand feedbacks and interactions between biotic and abiotic processes and their controlling factors, and to understand linkages between hierarchical levels of organization. Incorporating mechanistic linkages between different hierarchical levels must be at the forefront of ecogeomorphic model development, where hierarchical modelling approaches are used. We need to establish the minimum parameterization requirements which can reasonably be expected to be sufficient for ecogeomorphic modelling, because inclusion of too much detail in both parameterization and process representation may increase sensitivity and error propagation (Deutschmann et al. 1997; see also Chap. 10).

Linking data from experimental approaches with models is a critical research frontier. However, both the suitability of experimental data for the desired approaches to modelling, and the feasible approaches to modelling given the practical constraints on experimental data collection need to be evaluated further. A key aspect of such investigation is the need to incorporate uncertainty analysis, especially if model-parameterization requirements increase since model output cannot be truly evaluated or compared to observations unless uncertainty in both is acknowledged and incorporated into the evaluation and comparison (see Chap. 10).

A common goal of ecologists and geomorphologists working from monodisciplinary perspectives, pertaining to issues of land degradation in drylands, is to understand the mechanisms behind pattern-process relationships. In developing this understanding, a fundamental problem is the clear separation of the roles of drivers and emergent properties; be that in models or empirical understanding. That separation is confounded by the fact that drivers and emergent properties are scale-dependent. What is a driver at one scale may be an emergent property at another, and *vice versa*. For example, islands of fertility are emergent landscape *features* of resource distribution at the plant-interspace scale (Schlesinger et al. 1990), but may be *drivers* of redistribution of resources by runoff at the hillslope scale (Parsons et al. 2004; Brazier et al. 2007). Furthermore, drivers of degradation will vary across different dryland regions. For example in Mediterranean Europe land use (i.e. agriculture and grazing) is likely to be more important than climatic drivers (e.g. Kosmas et al. 1997; Papanastasis et al. 2002; Wainwright and Thornes 2004, Chap. 14), whereas in the US Southwest, grazing alone seems more important, at least in some locations (Chap. 11). The role of human drivers of land degradation has been under-researched within the framework of ecogeomorphology (Wainwright and Millington 2010). However, humans are a fundamental part of almost all ecogeomorphic systems and need to be integrated with our ecogeomorphic understanding. In this sense, output from ecogeomorphic modelling needs to be carefully tailored so that it can be used effectively in practical applications. For example, model output needs to be at spatial scales relevant for land management. In achieving this goal, selection of the scales and relevant components of analysis need to be cognizant of the perils of observer dependence.

Any proposed approach to advancing a solution to the problem needs to be mindful of the limited resources for solving any human-environmental problem. Greater use of existing data sets (particularly long-term data sets) and techniques of data mining coupled with the use of proxy measurements (for example remote-sensing data) can lead to more efficient strategies for new data collection and the optimization of their use. A particular problem to be addressed is that of large-scale processes, both in space and through time. It is unrealistic to expect this problem to be addressed through direct experimentation, yet solved it must be.

At the end of the ESF Workshop, the participants were asked to identify the three key challenges in the study of land degradation in drylands. The results of this exercise showed there is still much to be done in terms of identifying a common research agenda: the key challenges were disappointingly numerous and there was no strong convergence of thinking apparent between the disciplines, but then again a two-day workshop might have been too short to achieve that. The following section attempts to embrace the different perspectives by setting up five key challenges for land-degradation research.

15.4 Key Challenges for Land-Degradation Studies in Drylands

In Fig. 1.3, we proposed that for a true ecogeomorphological perspective, researchers should look outward from the problem of land degradation towards contributory disciplines rather than outward from their disciplines towards problems to be addressed. In this final section, we wish to argue that an ecogeomorphological perspective, interdisciplinary as it may be, is still too narrow a perspective from which to address the problem of land degradation in drylands. Figure 15.1 places the ecogeomorphological perspective in a broader scientific perspective. From this broader perspective, and within the context of the outcome of the workshop, the key challenges for land-degradation studies in drylands have been identified.

These challenges are both conceptual and practical. We have identified five key challenges that need to be overcome for truly interdisciplinary ecogeomorphic approaches to yield a common understanding of the problem and potential solutions across the disciplines.

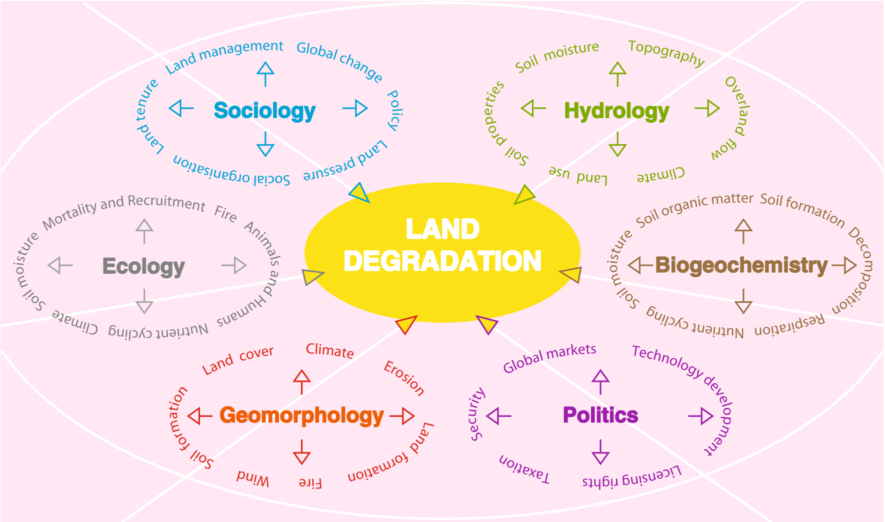

Fig. 15.1 Role of ecogeomorphology in a wider conceptualizing of land degradation in drylands

15.4.1 Develop a Common Language

Ultimately, the successful integration of ecological and geomorphological knowledge and principles within ecogeomorphology will depend on the level of theoretical compatibility as much as practical challenges and a common language, scalability of investigations, and data collection (Urban and Daniels 2006). The functioning of interdisciplinary work has been studied intensively (Balsiger 2004). Evans and Marvin (2004) noted that previous attempts at interdisciplinary research often resulted in multidisciplinary research being undertaken from a range of different (discipline-bound) perspectives, rather than leading to approaches that are truly interdisciplinary. Several problems arise frequently from recognised differences between discipline-based experts including: fundamental differences in epistemologies, knowledge and methods, a range of 'attitudes' across disciplines, different ways of formulating research questions and differences in communication (both oral and written) (Bracken and Oughton 2006). However, perhaps the biggest problem of all is that of scientists communicating without noticing that they do not understand each other. Discussions may be well-developed before it becomes apparent that a particular word or terminology has a specific disciplinary interpretation not known to scientists from other disciplines.

In the course of the ESF workshop and the compilation of this book, recurrent terminologies were clarified (such as model, scale and emergence: see Chap. 3). While formal definitions of recurrent terminologies are useful, they fail to capture the breadth and dynamism of language and multiple meanings of key words in use thus potentially inhibit the construction of knowledge. Different disciplines, and even those taking different approaches to the study of the same discipline (such as modellers and experimentalists), may tend to have different starting points for thinking about a specific word, such as watershed or connectivity, or terms such as land use or self-organization, and those working within those disciplines have been trained to think in different ways. The difference between disciplines results in conceptual boundaries being drawn in different ways and at different spatial and temporal scales. Language boundaries can be transcended if articulation is employed as a means to deconstruct one's own disciplinary knowledge in conjunction with those of other disciplines in order to understand the building blocks of an expression and thereby reconstruct a common understanding or to produce a more complex definition of a term (Ramadier 2004; Bracken and Oughton 2006). The first step in developing a common language for research may therefore be the need for a new trading, or pidgin, language which develops terms with shared meaning (McConnell et al. 2011). In turn, this pidgin can provide the basis for a creole language with a more formal grammar capable of describing more sophisticated concepts and interactions. The articulation required to overcome the boundaries of language and knowledge can be achieved by a set of truly integrated research projects. Integrated projects must allocate time to the development of shared vocabularies and way of thinking by having longer start-up phases to promote cohesion and to learn to value contributions from other disciplines: such projects require professional management to handle initial language and thought barriers (Bracken and Oughton 2006).

A common language and cross-disciplinary thinking has to be established by having all team members involved at some level in all aspects of the work, rather than just contributing to an output which it is hoped will be greater than the sum of its parts. Modellers teamed up with experimentalists, and geomorphologists teamed up with ecologists (and others identified in Fig. 15.1), should frequently test their assumptions and perceptions of setting up models, fieldwork, spatial and temporal data analysis, and so on. Even though this process is time-consuming, it enables exposure of disciplinary ignorance, methodological weaknesses and superiorities in a constructive way (Bracken and Oughton 2006). The need for developing a common language is not only true for scientists from ecology, geomorphology and other involved disciplines, but also for the communication between non-local and local scientists, inhabitants and stakeholders managing, working and living in degraded lands.

15.4.2 Transcend the Problem of Scale

It is well established that processes and patterns occurring across different scales – both spatially and temporally – affect each other (see for example Chap. 6). Although there are examples where cross-scale interactions have received attention (e.g. Peters et al. 2006; Okin et al. 2006), they are few and far between, and they have received little or no attention in ecogeomorphic, field-based, land-degradation studies.

The continued development and refinement of ecogeomorphic modelling approaches in tandem with integrated cross-scale field studies is likely to be the most valuable research tool available for investigating cross-scale interactions. Modelling of potential ecogeomorphic feedback mechanisms and self-organizational patterns will provide guidance in selecting highly variable (in space and time) and relatively fixed parameters that are critical in driving the feedback or pattern organization and which may therefore require more focussed or detailed field investigation (e.g. Grimm et al. 2005). An important component of modelling-based experimentation is to carry out sensitivity analyses to identify the most important parameters in controlling the processes being modelled (Mulligan and Wainwright 2013; Ratto et al. 2001). Modelling should thus also guide existing long-term monitoring schemes that should be enhanced and, although keeping some coherence in their principal elements, should be continuously developed in their aims and set-up. There should be a change of mind-set to envision the multi-decadal collection of data as a new standard in data mining to allow the analysis of temporal scaling in land-degradation studies. Although there are scientific movements towards adopting this type of approach – such as the NEON (National Ecological Observatory Network) project which is perhaps a striking example of a new monitoring initiative that is aimed at collecting data over the long term for a range of parameters/variables (NEON 2012) – the real problem is the allocation of sufficient institutional vision, funds, staff and infrastructure to allow the work to take place.

Modelling and process studies need to address the specific temporal scales that are attached to the recurring or non-recurring triggers and drivers of land degradation. For example, a fire may trigger ecosystem change within hours and observation of fire impacts would focus on rather short times spans. However, fires may have multi-scale effects over a range of timescales such as persistent drought because of changes to the soil hydrology over several years that might alter soil-vegetation functioning in such a way that the system does not recover from the disturbance, up to evolutionary timescales where the form and/or function of species evolve to tolerate or adapt to repeated burning episodes. Landscapes characterized by a fire-adapted ecology will thereby have different geomorphic feedbacks from those where plants are not fire-adapted. These feedbacks will have differing spatial consequences because short- and long-range erosion processes (see Chaps. 4 and 5) will be triggered in different ways. Management decisions might cause an immediate change of land use, but changes to soil-vegetation interactions might only become apparent decades later such as with the effects of clear-cutting of *Eucalyptus* woodland in southwest Australia to provide land for irrigation agriculture, leading to severe salinization problems as a result of increased groundwater recharge (Cramer and Hobbs 2005; see also Chap. 14 on land abandonment in the Mediterranean region). Future management decisions may also be influenced by prior spatial patterns of land use and management which constrain ecological and geomorphic processes. Furthermore, temporal variability in drivers might vary over relatively short cycles such as the effects of El Niño on annual temperature and rainfall regimes (e.g. Dahm and Moore 1994; Wainwright 2005) or over much longer periods of centuries to millennia due to climate variations. A large array of records is readily available in some areas such as the US Southwest to assess temporal change of land degradation and their drivers, such as continuous coverage of multi-spectral remote sensing imageries, old air photography, erosion-pin analysis of hillslope evolution or regional records of livestock rates and subsidy measures. However, even where it is available, the degree to which this information is actually used to address questions of temporal scale is very limited. In many areas of the world, comparable data sources are not available. In the same way as pictures of the blue planet and subsequent large-scale development of remote sensing have allowed us to take on board large spatial scales and incorporate them in our studies, there is a need to develop techniques that allow us to conceptualize processes and their interactions over timescales longer than our perceptual basis – and certainly for longer than the artificial basis of the three-year research or PhD project (see also Klemeš 1997).

15.4.3 Take on Board the Lessons of Complexity Science

A related point is the current predominance of short-term, hypothesis-driven research. While such a deductive framework to research undoubtedly has its uses, one needs a more fluid approach to deal with research questions in land-degradation studies. Complexity science suggests that it is fundamental to address the roles of thresholds, path dependence, contingency and emergent behaviour in our understanding. In some cases, inductive approaches will be required to evaluate the effects of contingency and path dependence; in others "play" with a range of "toy" models may be the best way to interpret emergent behaviour such as pattern formation. Comparison of the behaviour of toy and process-based models will enable us to determine the necessary conditions for the formation of patterns in specific environments. Iterations between these two modelling approaches and the collection of field data at appropriate spatio-temporal scales are the only way to improve our understanding of process and system evolution. There should no longer be a barrier between nomothetic and idiographic approaches when solving complex problems like land degradation. Furthermore, given the need to incorporate thresholds, contingency and path dependence into our topics of investigation, it is fundamental that we design a research approach that is serendipitous. Major advances are rarely made when the scientific endeavour is locked into a phase of static mundanity (e.g. Eco 1998).

The more fluid approach to research is to advance logic that allows us to include uncertainty and allows us to include belief. Uncertainty (Chap. 10) enters our research process at every step along the way – whether we are prepared to admit it or not. Techniques are available to evaluate uncertainty of measurements and the propagation of error within a model, but the means of evaluating epistemic uncertainty, when the model structure itself is in error are much less clear, not least because the problem of equifinality means that simply comparing model results against measurements is no basis of providing an unequivocal result (see discussion in Oreskes et al. 1994; Mulligan and Wainwright 2013). To address the issue of epistemic uncertainty, we also need to consider the extent to which our belief in the behaviour of a particular system is moulded – notwithstanding the homogenization of a rigorous "scientific" training – by our experiences and also our usually insular disciplinary backgrounds (see discussion above and in Chap. 1). Unique settings, both biophysical and cultural, mean that not only may the solutions to land degradation differ from place to place, but so, too, may the research approaches needed to identify those solutions. Accepting this uniqueness and how it moulds our beliefs may also help in transcending the scale problem.

More flexible ways of looking at problems are also vital. For example, qualitative data are not only fundamental for evaluating the human interactions with land degradation (see Sect. 15.4.4), but also they can provide invaluable insights either into system behaviour (addressing issues of epistemic uncertainty) or data sources where information would otherwise be unavailable (see Chap. 10). We should therefore not be shy about using any source of information useful for solving the problems of land degradation, for the benefit of the billions who depend on drylands for subsistence. If there are things out there that would be beneficial to your studies, make use of them. For example, old photographs can be used to study land-use change in a qualitative way in order to get information on long-term change (e.g. Hastings and Turner 1965; Nyssen et al. 2009), or diaries of early travellers can be used to lengthen the available climate records before instrumentation was in place (e.g. Nash and Endfield 2008).

15.4.4 Improve the Understanding of the Interactions of Ecogeomorphic Processes and People

The integration of biophysical processes that is represented by ecogeomorphology provides only a partial perspective on the problem of land degradation in drylands. A more complete perspective of the land-degradation problem must include socio-economic and political processes and their interactions with ecogeomorphological processes and patterns, as depicted in Fig. 15.1. In many ways the solution to this challenge is similar to that in Sect. 15.4.1 on the establishment of a common language. Part of the solution is to have integrated research projects in which the scientists from all disciplines fully engage in all aspects of the research and participate where feasible to gain an appreciation of the full range of different perspective involved as well as the broader scale context. Without understanding the politics, socio-cultural characteristics and economics of a region, it is impossible to determine suitable land-management strategies that will achieve the desired goals, and the amount of resources that a region might be able to invest to do so (e.g. Green and Lemon 1996). While some regions might be able to diversify, to reduce dependence on activities that exacerbate degradation, others may not, and these are the regions that need more creative solutions. However, there is a risk that fully interactive research projects become unmanageable due to the greater diversity of processes involved.

Given the importance of modelling for understanding large-scale changes (both in space and time), an integrated modelling framework that can include all of the mutual dynamics of all disciplines shown in Fig. 15.1, as contributory to understanding land degradation in drylands, is essential. In modelling, for example, an extension of continuum (differential) models that may be appropriate for the biophysical processes and patterns of ecogoemorphology will not usually be suitable for modelling socio-economic and political processes. However, simple rule-based approaches – the so-called "generative social science" approach (Epstein 2007) – have been demonstrated to be useful in this context. For example, Millington et al. (2008) used a rule-based approach to evaluate the different impacts that Spanish farmers with traditional versus modern worldviews have on vegetation and fire regimes, and thus their potential impacts on land degradation. There is a conceptual advantage with using this approach, using agent-based techniques, which strongly overlap with the individual-based approaches used in many pattern-based models (Grimm and Railsback 2005). This sort of modelling approach also shows great potential for the integration of knowledge from different sources using participatory model building (e.g. Castella et al. 2005; Nguyen-Duc and Drogoul 2007), in which different stakeholders and experts are questioned in order to evaluate the key parameters and processes, or interact with initial versions of the model to evaluate its performance. However, given the qualitative modes of analysis in many aspects of social science, care must be taken in their application, evaluation and reporting. Millington et al. (2012) discuss ways in which narrative methods may be used to discuss and test the results of this sort of model, for example by acting as an intermediary between formal descriptions of model structure and quantitative analyses of model output. Twyman et al. (2011) note that there may often be a paradox in that the model analyses may be given too much emphasis by policy-makers because their quantitative nature may seem more "exact" than the qualitative aspects of the work (but see Chap. 10), while at the same at the same time as they are seen as incommensurate with qualitative techniques. They suggest that this paradox underlies a "healthy tension" between the two approaches that essentially improves both approaches through a triangulation of results, not least when scenario- and narrative-based methods underpin both aspects of the interdisciplinary research.

Thus, a commonality of approaches (complexity-based science, agent/individual-based modelling, narratives, triangulation) can be seen to underpin a common approach to the understanding of the human and environmental aspects of the land-degradation problem. It is exactly these sorts of methodological and conceptual advances that are needed to address the (interdisciplinary) scientific problems that are more fundamentally required to improve approaches to managing land-degradation problems.

15.4.5 Develop Management Strategies for Combatting Land Degradation in Drylands

Notwithstanding the gaps in our understanding of land degradation in drylands, there is a pressing need to use the integrated understanding discussed above to develop management strategies to stop, prevent or reverse this degradation. As it has been previously stated, the world's drylands are home to two billion of the world's population, including many of the poorest. Many of the world's drylands have undergone land degradation in the recent past affecting the livelihoods of 250 million people, and many are projected to be affected by further desertification in the coming decades as a result of climate change (Schlesinger et al. 1990; Okin et al. 2004). People cannot wait for scientists to formulate what they may consider to be an adequate explanation of land degradation before action can begin. Nor is this approach an appropriate model for the interaction of science and society. Rather, the management agenda is one that has to be developed in parallel with scientific understanding. A symbiotic relationship between improved scientific understanding and improved management tools is one that will benefit both: developments in scientific understanding will be grounded in place and management will be at the forefront of scientific understanding. Implicit in such an approach is that there is not ONE scientific understanding of land degradation in drylands but that there are many that are both locationally and culturally contingent. That is not to say that there are not similarities across locations and cultures, but just that such similarities are not a *necessary* part of understanding land degradation.

If developing management strategies is to be symbiotic with developing scientific understanding, then this development needs to be an integral part of research programmes to improve scientific understanding. There are two challenges to this integration. First, to identify those management strategies that are compatible with the scientific understanding and, secondly, to have those strategies adopted by land users and managers. Unless the latter is achieved, then neither will the goal of developing strategies to combat land degradation be achieved, nor will the benefits of successful (or otherwise) evaluation of scientific understanding be possible.

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