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Rangelands at equilibrium and non-equilibrium: recent developments in the debate

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Abstract

This paper reviews the predictions and management implications of two current paradigms in the ecology and management of arid and semi-arid rangelands. The equilibrium model stresses the importance of biotic feedbacks such as density-dependent regulation of livestock populations and the feedback of livestock density on vegetation composition, cover and productivity. Range management under this model centres on carrying capacity, stocking rates and range condition assessment. In contrast, non-equilibrium rangeland systems are thought to be driven primarily by stochastic abiotic factors, notably variable rainfall, which result in highly variable and unpredictable primary production. Livestock populations are thought to have negligible feedback on the vegetation as their numbers rarely reach equilibrium with their fluctuating resource base. Recent studies suggest that most arid and semi-arid rangeland systems encompass elements of both equilibrium and non-equilibrium at different scales, and that management needs to take into account temporal variability and spatial heterogeneity.

Keywords: Range condition; Overgrazing; Key resources; Rainfall variability; Heterogeneity; Scale dependence; Mobility

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1. Introduction

For most of this century, there has been concern about the sustainability of communally grazed rangelands in Africa and other parts of the world. Pastoral systems are commonly viewed as overstocked, overgrazed, degraded and unproductive (e.g. Lamprey, 1983; Sinclair and Fryxell, 1985), and this has resulted in widespread interventions to reduce stock numbers in an attempt to halt degradation. Overgrazing is commonly thought to be inevitable in communal pastoral systems because people keep more livestock than they need for a variety of reasons (e.g. Herskovits, 1926; Lamprey, 1983), and because of the problems inherent in communal ownership of the resource, where individual benefit is maximized at the expense of the communal resource (Hardin, 1968). Increasing human population pressure, encroachment of rangelands by other land use, control of livestock diseases and the breakdown of traditional resource management structures are thought to contribute to the degradation problem.

This way of viewing and managing communal grazing systems has come under considerable criticism regarding its underlying ecological and economic assumptions, and the idea that communal rangelands are necessarily mismanaged is now widely challenged (e.g. Sandford, 1983; Homewood and Rodgers, 1987; Ellis and Swift, 1988; Abel and Blaikie, 1989; Behnke and Scoones, 1993; Behnke and Abel, 1996; Sullivan, 1996; Sullivan and Rohde, 2002). From a broader debate about interpretations of desertification, and the identification of pastoralists as its major causative agent (e.g. Homewood and Rodgers, 1987; Leach and Mearns, 1996; Dodd, 1994), a debate around the ecological dynamics and appropriate management of semi-arid rangelands has developed. This debate arose in the 1980s in response to a growing concern that interventions aimed at stabilizing spatially and temporally variable rangelands were inappropriate and damaging to pastoral livelihoods (Sandford, 1983; Ellis and Swift, 1988).

This coincided with an increasing recognition that equilibrium dynamics were difficult or impossible to demonstrate conclusively in many ecological systems (Wiens, 1977, 1984, 1989a; Connell and Sousa, 1983; DeAngelis and Waterhouse, 1987), and it was suggested that a different paradigm was necessary to describe non-equilibrium systems. Ellis and Swift (1988) and Westoby et al. (1989) applied non-equilibrium concepts to rangelands and pointed out that a fundamental misunderstanding of their ecological dynamics was leading to inappropriate and failed interventions. The debate gained momentum in the early 1990s after two international workshops around emergent new paradigms in rangeland ecology and socio-economics, which resulted in the publication of two influential books on range ecology (Behnke et al., 1993) and pastoral strategies to cope with uncertainty (Scoones, 1994). The "new rangeland ecology" posits that traditional, equilibrium-based rangeland models have not taken into account the considerable spatial heterogeneity and climatic variability of semi-arid rangelands, and that mobility, variable stocking rates and adaptive management are essential for effectively and sustainably utilizing semi-arid and arid rangelands.

Central to the debate is the relative importance of biotic and abiotic factors in driving primary and secondary production in rangelands, and the consequences of this regarding the potential for grazing-induced rangeland degradation. The equilibrium model stresses the importance of biotic feedbacks between herbivores and their resource, while the non-equilibrium model sees stochastic abiotic factors as the primary drivers of vegetation and livestock dynamics. The debate has stimulated much new research, and many researchers now agree that both equilibrium and non-equilibrium dynamics are found in rangelands, often at different times or governing different parts of the resource (e.g. Fernandez-Gimenez and Allen-Diaz, 1999; Illius and O'Connor, 1999, 2000; Desta and Coppock, 2002; Briske et al., 2003). However, the equilibrium ideas of stability and predictability have remained pervasive in ecology and range management and also in the social sciences (Scoones, 1999).

The challenge is to understand under what circumstances different dynamics apply, since the two models have fundamentally different consequences for policy and management. Interventions based on the equilibrium model focus on reducing stocking rates and increasing stability, while the non-equilibrium paradigm advocates opportunistic stocking strategies and promotes mobility. The different predictions of these two paradigms also determine whether livestock numbers can be allowed to increase without threatening degradation. Possibly the most widely cited and debated argument of the non-equilibrium rangeland ecology is that plant composition and biomass in semi-arid rangelands are primarily driven by rainfall and not by grazing pressure, that animal numbers are kept below equilibrium densities by frequent droughts, and that degradation of the vegetation as a result of overgrazing is thus unlikely (Ellis and Swift, 1988; Behnke and Scoones, 1993; Sullivan and Rohde, 2002). Since overgrazing is a problem commonly attributed to communal tenure, whether or not intensive grazing is damaging to the environment has profound and far-reaching consequences for the persistence of pastoral systems. For example, many pastoralist groups have been removed from their traditional grazing areas because they are seen as a threat to wildlife conservation in East Africa (e.g. Homewood and Rodgers, 1987; Brockington, 2002). In southern Africa, the debate about degradation and low productivity in communal rangelands has influenced policy on land reform (Cousins, 1996).

This paper examines the predictions and management implications of the equilibrium and non-equilibrium paradigms, reviews some recent attempts to test their predictions and discusses the current status of the debate.

2. The ecological debate

2.1. What determines the size and productivity of livestock populations?

One of the ongoing debates in range ecology is that regarding the relative importance of density-dependent interactions and abiotic factors in determining herd productivity, reproduction and mortality from year to year (Ellis and Swift, 1988; Illius and O'Connor, 1999; Sullivan and Rohde, 2002). In a grazing system with relatively predictable rainfall and hence forage production, livestock populations are regulated in a density-dependent manner via competition for food resources. As population size nears carrying capacity, increased competition for resources leads to reduced herd productivity. A sign of density dependence is that population growth rates decrease with increasing population size because of the effects of competition on reproductive and mortality rates. This is exemplified by the Jones and Sandland (1974) model of the effect of stocking rate on cattle weight gain.

Rainfall in different years affects grass production and composition (Dye and Spear, 1982; O'Connor, 1985, 1994, 1995) and hence the effective carrying capacity at different times. If livestock populations are near carrying capacity, and hence already competing for resources, they are likely to experience population crashes in drought years when resources become scarce (Caughley, 1979). If livestock populations are well below the ecological carrying capacity, drought mortality is reduced because livestock are buffered against such stress events by greater forage and body fat reserves. Thus the recommended management practice is the maintenance of conservative stocking rates which can be maintained in drier years.

In grazing systems with very high climatic variability, forage availability varies to such a great degree with rainfall that herbivore population dynamics are driven by rainfall via its direct effect on forage availability in any given year. In such non-equilibrium systems, density-dependent interactions such as competition for resources play a minor role in regulating populations (Wiens, 1977; Ellis and Swift, 1988). Mortality is high and density-independent during severe droughts, particularly droughts lasting longer than 1 year (Homewood and Lewis, 1987; Ellis and Swift, 1988; Scoones, 1990; Oba, 2001). Livestock numbers build up during series of wet years. Population size thus fluctuates dramatically, and cannot track rainfall closely because of the time it takes populations to recover from crashes.

The dichotomy between density-dependent and abiotically driven population dynamics is an oversimplification of the range of situations found in reality. The strength of density-dependent interactions varies over time and in space. For example, density-dependent dynamics in non-drought years can alternate with density-independent mortality during droughts and subsequent recovery (e.g. Scoones (1990) in southern Zimbabwe; Desta and Coppock (2002) in southern Ethiopia). In a grazing experiment examining the relative importance of rainfall and stocking rate on plant composition and primary and secondary production, Fynn and O'Connor (2000) found that density-dependent consumer–resource coupling was largely limited to drought periods and was greater at high stocking rates. This example illustrates that a measure of grazing pressure (the number of livestock per unit of available forage) is more informative—if harder to quantify—than stocking rate (the number of animals per area) in systems where livestock numbers and rainfall vary over time.

It is exceedingly difficult to infer density-dependent mechanisms—or their absence—from livestock population census data. Part of this problem lies in the extreme difficulty and cost involved in obtaining detailed enough data on population size, mortality, fecundity and migration over a long enough time series. Simulation models are a useful tool for exploring the mechanisms which regulate livestock dynamics under different conditions in such variable systems. More fundamentally, a focus on the phenomenon of density-dependence does not in itself provide an explanation of the underlying mechanisms of the consumer–resource dynamics, for example how they are affected by seasonal variability and spatial heterogeneity in forage quality and quantity (Owen-Smith, 2002).

Illius and O'Connor (1999, 2000) question the relevance of non-equilibrium concepts to arid grazing systems and argue that variability in arid and semi-arid rangelands is not the outcome of qualitatively different ecological dynamics. They propose that livestock populations in arid and semi-arid grazing areas are regulated in a density-dependent manner by key resources. Key resources are defined in terms of the key factor determining livestock populations, usually survival through the season of plant dormancy, and thus the forage available during the dry season. Herbivore populations are in long-term equilibrium with the key resources, while being largely uncoupled from forage resources that are only available in the wet season, which can be classed as non-equilibrium resources (Illius and O'Connor, 2000). So far, these models have not been tested in the field, nor has the existence and nature of key resources and non-equilibrium resources been explored in many rangeland systems outside southern Zimbabwe, where their role was highlighted by Scoones (1993, 1995).

2.2. *What determines the composition and productivity of vegetation communities?*

The equilibrium paradigm is based on the assumption that every environment has a carrying capacity determined by biophysical characteristics, such as the mean annual rainfall, soil type and other biophysical characteristics, which determine its potential primary production (East, 1984; Bell, 1982; Fritz and Duncan, 1994). The actual carrying capacity of an area at any given time is determined by range condition, which is assessed as a function of grass composition, biomass and cover and is interpreted as a stage in plant succession (Dyksterhuis, 1949; Foran et al., 1978; Trollope, 1990). The response of the vegetation to grazing pressure is linear and reversible, and can be manipulated predictably with stocking rates. No, or very light, grazing allows the vegetation to reach its climax stage, whereas heavy grazing pushes it back to a pioneer stage dominated by weedy or unpalatable grass and forb species typical of disturbed environments. Continuous intense grazing leads to vegetation changes such as the replacement of palatable grasses by less palatable plant species, replacement of perennial grasses by annuals, bush encroachment, lower standing biomass and reduced basal cover (e.g. Kelly and Walker, 1976; Coppock, 1993; Ash et al., 1995; Todd and Hoffman, 1999; Fynn and O'Connor, 2000). These in turn are predicted to result in a decrease in forage quality and quantity, increased variability of primary production, accelerated soil erosion and ultimately an irreversible decline in animal production. Rainfall is thought to affect the vegetation via a similar mechanism where drought reduces range condition by pushing the vegetation community towards a pioneer stage, while high rainfall improves range condition. Rainfall and stocking rate interact, with low rainfall exacerbating the effects of high stocking rate, and high rainfall mitigating them.

Observations that vegetation responses to grazing, drought and fire are often not linear and reversible have led to the suggestion that thresholds exist between different rangeland states (Friedel, 1991), and to the development of state-and-transition models as an alternative to the rangeland succession model (Westoby et al., 1989). These incorporate multiple successional pathways, multiple steady states, thresholds of change, and discontinuous and irreversible transitions (Stringham et al., 2003). Changes from some states, e.g. bush encroachment, can be irreversible over management time-scales and require major management inputs. Management of such systems should be opportunistic and take advantage of, or create, conditions which allow switches to a more desirable state. Although state-and-transition models are considered to characterize rangelands not at equilibrium (Westoby et al., 1989; Briske et al., 2003), they have been applied to rangelands in conjunction with succession-based models (Phelps and Bosch, 2002).

In some systems, no such distinct states can be distinguished and forage availability and composition every year are primarily determined by stochastic abiotic factors such as rainfall. Due to the short duration of the growing season, the high frequency of droughts and the great inter- and intra-annual variation of rainfall in semi-arid rangelands, the available amount of forage fluctuates considerably between years. Livestock numbers are unable to track these sharp fluctuations, and the dynamics and productivity of the vegetation and livestock are thus uncoupled most of the time. Two-year droughts, which are accompanied by severe mortalities, also occur regularly. Herd size builds up gradually in wetter years following a drought, during which time the vegetation is relatively lightly grazed. In a system such as this, degradation due to overgrazing is unlikely, since animals seldom if ever reach densities at which they provide a negative feedback on the vegetation (Ellis and Swift, 1988). Whereas the equilibrium model views drought as concentrating the effects of herbivory on scarce resources (Illius and O'Connor, 1999), the non-equilibrium model sees drought as relieving the pressure of high stocking densities, by making grazeable vegetation unavailable (Sullivan and Rohde, 2002) and by inducing density-independent livestock mortality which reduces grazing pressure (Ellis and Swift, 1988). It has been suggested that forage limitation in cold winters and density-independent livestock mortality during extreme cold events result in similar non-equilibrium dynamics in the cold rangelands of northern Asia (Kerven, 2004).

Research supports the suggestion that equilibrium and non-equilibrium are extremes along a continuum and that many systems encompass elements of both (Wiens, 1984, 1989a; Ellis et al., 1993; Ellis, 1994; Stafford Smith, 1996; Oba et al., 2000; Desta and Coppock, 2002; Sullivan and Rohde, 2002). Evidence from arid environments with high rainfall coefficients of variability (C.V.) suggests that these systems are well described by the non-equilibrium model (Ellis and Swift, 1988; Ward et al., 1998, 2000a; Sullivan, 1998, cited in Sullivan and Rohde, 2002; Fernandez-Gimenez and Allen-Diaz, 1999). In the arid areas studied, vegetation cover, composition and productivity were strongly determined by rainfall, while grazing intensity had a negligible influence. In more mesic areas with rainfall C.V. of less than 30%, grazing-induced changes such as bush encroachment (Desta and

Coppock, 2002) and changes in grass composition (Fernandez-Gimenez and Allen-Diaz, 1999) were found. This is consistent with the prediction (Ellis et al., 1993; Ellis, 1994) that non-equilibrium dynamics predominate in areas where rainfall C.V. exceeds 33%.

One of the reasons some arid rangelands appear to be resilient to long-term intensive grazing is that the grass sward is dominated by annual grasses, which do not germinate or establish in the absence of rainfall. Grasses grow from a seed bank in subsequent wet years, with biomass production more or less proportional to the amount of rainfall. As Sullivan and Rohde (2002) argue, there may be literally no grass to overgraze in a drought year. Studies on annual grasslands of the Sahel have, however, revealed changes in plant composition, productivity and soil characteristics in response to grazing. Turner (1999) found that long-term grazing history affected the composition and peak biomass production of annual grasslands in the Sahel, even though no vegetation responses to short-term grazing impacts could be detected. Grazing can influence annual vegetation by leading to reduced seed production of preferred or more grazing-exposed species, or by favouring species with short life cycles, heterogeneous germination patterns or competitive advantages under low litter cover (Turner, 1999 and references therein). Hiernaux (1998) similarly found changes in species composition of annual Sahelian grasslands which resulted from differences in grazing tolerance. However, there was no correlation between grazing response and palatability and productivity as predicted by the rangeland succession model. Grazing can also alter soil conditions, leading to shifts in plant composition (Turner, 1998; Hiernaux et al., 1999), although this was not found to be the case in Namibia (Ward et al., 1998). The susceptibility of soils to grazer-induced changes such as crusting, compaction and accelerated erosion is related to texture, with sandy soils being more resilient than clay soils in arid areas, and the reverse being the case at high rainfall (Walker et al., 2002). Grazing can also affect nutrient and water cycling. The loss of perennial shrubs, which accumulate nutrients in hot spot "islands" under their canopies, can lead to overall nutrient losses at the landscape scale (Schlesinger et al., 1990; Allsopp, 1999).

In systems dominated by perennial grasses, high grazing pressure can exacerbate drought mortality of grass tussocks and hinder post-drought establishment of seedlings (O'Connor, 1991, 1994; O'Connor and Pickett, 1992). Compositional changes and local extinction of grass species such as *Themeda triandra* following drought are greater under heavy grazing than under light or no grazing (O'Connor 1995; Fynn and O'Connor, 2000). Perennial grasses invest less in reproduction from seed than annual grasses, and their dispersal, recruitment and establishment is therefore often seed-limited. As grass tufts die and grasses fail to re-establish, more soil becomes exposed and hence vulnerable to erosion. O'Connor and Roux (1995) found that the long-term response to grazing was most pronounced in longer-lived plants, whereas the growth of annual grasses directly responded to rainfall from year to year.

In certain areas, long-term high grazing pressure has resulted in persistent and resilient vegetation assemblages dominated by grazing-tolerant or grazing-resistant plant species. Examples are the grazing lawns of the Serengeti (McNaughton, 1979), the shortgrass steppe of the USA (Milchunas et al., 1988), perennial *Aristida junciformis* grasslands of Transkei in South Africa (McKenzie, 1982) and annual grasslands with *Indigofera cliffordiana* dwarf shrubs in northern Kenya (Oba et al., 2000). Oba et al. (2000) found that high *Indigofera* mortality accompanied complete exclusion of ungulate herbivory for longer than 5 years, and this was followed by an increase in the percentage of bare ground after 8 years. The authors concluded that grazing was essential in maintaining the productivity and diversity of the vegetation and that a lack of grazing, rather than overgrazing, leads to rangeland degradation. However, care must be taken with the definition and assessment of degradation, and when trying to extrapolate such findings to other regions.

It is increasingly recognized that the quality, quantity and seasonal availability of forage differs between parts of the landscape, and that people and livestock do not utilize all areas at the same frequency and intensity. Herded animals use the landscape differently to animals kept in paddocks, resulting in different impacts on different parts of the landscape. Some areas are more resilient to transformation than others, either because livestock cannot access them for prolonged periods (e.g. annual grasslands, grazing areas far from permanent water) or because the dominant plant species are tolerant of heavy defoliation (e.g. stoloniferous grasses). Patterns and processes in such heterogeneous landscapes are scale dependent, such that inferences about large-scale behaviour cannot reliably be made on the basis of smaller-scale observations. All patterns and processes are best described at a particular scale, but no single scale can collectively describe population, community and ecosystem level processes (Wiens, 1989b; Briske et al., 2003; Hobbs, 2003). Many range ecologists are struggling to overcome the mismatch between the scales of ecological investigation and those at which ecological processes in rangelands take place. While there is now a plethora of experimental results at the plot scale, larger-scale data from heterogeneous landscapes are still scarce.

2.3. Are non-equilibrium rangelands prone to degradation?

Much of the heat of the debate has been generated by the assertion that non-equilibrium rangelands are not vulnerable to degradation. In some cases this has been embraced so readily that concerns about degradation and the relevance of stocking rates were completely dismissed (e.g. Dikeni et al., 1996 in South Africa). This has led to concern about the ecological consequences of uncritically adopting the non-equilibrium paradigm for management, for example in areas which are not predominantly experiencing non-equilibrium dynamics (Illius and O'Connor, 1999; Fernandez-Gimenez and Allen-Diaz, 1999; Cowling, 2000; Desta and Coppock, 2002). Areas where non-equilibrium concepts would be inappropriate include less drought-prone rangelands at the more mesic end of the spectrum, but also arid, climatically variable areas where mobility of pastoralists has been severely restricted, or where the provision of seasonally scarce resources such as feed and water is reducing the temporal variation in animal growth even though rainfall and plant growth are low in drought years. Many studies suggest that the sustainability of non-equilibrium rangelands is dependent on drought (or other factors) periodically reducing livestock numbers and thus keeping grazing pressure below levels that are likely to cause degradation over the long term. A similar effect is achieved by moving livestock to less drought-affected parts of the landscape. An important question is whether this post-drought reduction in grazing pressure is a general requirement for rangeland sustainability.

Illius and O'Connor (1999) predict that areas with a greater ratio of key to non-equilibrium resources are more prone to grazing-induced degradation, as a bigger key resource allows the non-equilibrium resource to be more heavily utilized during the dry season. Introducing supplementary feed would have the same effect as increasing the key resource and thus increase the risk of degradation as this allows high grazing pressure to be maintained in an area during and after the dry season. Buying in of livestock, especially breeding stock, can speed up the recovery of the herd to its pre-drought size. For example, most livestock owners in South African communal areas have cash incomes from migrant labour, local employment, remittances and/or pensions (Cousins, 1998), making purchases of feed and livestock possible. Data from a communal area in South Africa show that high livestock numbers are increasingly being maintained through the provision of feed and buying animals after droughts (Vetter and Bond, 1999; Vetter, 2003), and that livestock numbers thus remain high during and after droughts. The provision of large amounts of subsidized supplementary feed, as is common in North Africa, the Middle East and China and was widespread in northern Asia during the Soviet era, has been observed to result in rangeland degradation (Seligman and Perevolosky, 1994; Kerven et al., 2003, 2004). An important research challenge is thus to understand the ecological consequences of restricting mobility in spatially heterogeneous areas, and of providing seasonally scarce resources such as water and feed in temporally variable environments.

It is now widely acknowledged that while many assessments of degradation were exaggerated and their attributed causes have been oversimplified, degradation has occurred in many semi-arid rangelands. The causes of this are complex, and the underlying causes can occur at larger scales than can be influenced by the land users (Ward et al., 2000b; Reynolds and Stafford Smith, 2002; Kerven et al., 2003). Common proximate causes include sedentarization of pastoralists or supplementary feeding, both of which lead to continuous, heavy utilization of parts of the range. The effects are usually spatially heterogeneous and often difficult to quantify, especially effects on secondary production which tend to be masked by spatial heterogeneity (Ash et al., 2002). Usually degradation takes place over time-scales much greater than those at which management decisions are made, and this disparity in scales has led land users not to perceive degradation as a concern (Abel, 1993; Biot, 1993; Reynolds and Stafford Smith, 2002). A recently developed synthetic framework, which recognizes the joint roles of biophysical and socio-economic factors at different scales in causing desertification, marks substantial progress in understanding the drivers and effects of degradation in arid rangelands (Stafford Smith and Reynolds, 2002).

3. Management of equilibrium and non-equilibrium rangelands

Planning and management of African communal rangelands has generally followed the equilibrium model and the assumption that these systems are overstocked and degraded. This has led to government interventions such as destocking schemes, conversion of communal areas into individually managed 'economic units' and settling of nomadic pastoralists into group ranches (Sandford, 1983; Ellis and Swift, 1988; Archer et al., 1989; Boonzaier et al., 1990; Rohde et al., 1999). The main focus of these interventions has been on preserving natural resources, with the additional intention of increasing livestock production and offtake, often for export or city markets. These schemes have met with widespread resistance, not least because they ignored the objectives of the pastoralists who derive a multitude of benefits from multi-species herds (Coughenour et al., 1985), many of which are non-consumptive (Shackleton et al., 2000). It is argued that these benefits are maximized at higher stocking rates than commercial farming objectives such as beef production (Sandford, 1983; Abel and Blaikie, 1989; Wilson and MacLeod, 1991; Behnke and Abel, 1996). Interventions often seemed to create or exacerbate, rather than solve, degradation problems and left many people economically worse off than before (Ellis and Swift, 1988; Hoffmann and Ashwell, 2001).

3.1. Dealing with temporal variability and drought

The recommended management practice for commercial farmers in semi-arid and drought-prone environments is to maintain low enough stocking rates to ensure sufficient forage in years of low rainfall. There is thus acknowledgement of climatic variability in equilibrium-based range management, but the proposed solution is to achieve stability by maintaining livestock at densities that are unlikely to exceed the reduced carrying capacity of dry years. The short-term economic benefits of keeping higher livestock numbers nevertheless encourage many commercial farmers to overstock (Ash et al., 2002), and some governments discourage this by making drought or other subsidies conditional on following recommended stocking rates.

It is argued that management based on constant and conservative stocking rates is inappropriate and costly to pastoralists in such variable systems, as they would be unable to make use of all the available forage in wet years, and would still overstock in very dry years (Sandford, 1982, 1983; Behnke and Scoones, 1993). The opportunity cost of conservative stocking rates increases with increasing rainfall variability and more conservative stocking rates (Sandford, 1982, 1983; Stafford Smith, 1996). Pastoralists employ a variety of strategies to cope with the variability of their environment (Sandford, 1983; Ellis and Swift, 1988; Scoones, 1994). Instead of aiming to keep animal numbers constant, pastoralists allow herd size to change with rainfall (Sandford, 1983, 1994; Toulmin, 1994). Drought risks are minimized not by maintaining conservative stocking rates, but rather by allowing livestock numbers to increase in wet years. While livestock owners risk substantial losses during a severe drought, having a large herd at the beginning of the drought ensures that at least some part of the herd survives. The bigger the herd belonging to an individual in a communal system, the greater is the number likely to survive, and larger herds thus provide greater security during droughts.

The effectiveness of this strategy depends on how pre-drought livestock density affects livestock survival, condition and post-drought recovery, particularly of breeding females. If mortality is completely density-independent, the number of livestock before the drought does not affect the number that die, and keeping low livestock numbers will thus not reduce drought mortality. If livestock mortality is density-dependent during drought, the effect of high pre-drought livestock numbers needs to be taken into consideration. Reducing livestock numbers before they reach densities where they exacerbate drought mortality, as suggested by Desta and Coppock (2002), would be an appropriate strategy under these conditions.

Management of livestock numbers in response to drought must take into account the variables of interest to pastoralists—i.e. the benefits derived from the livestock herd in the years between droughts, and the survival and recovery of the herd during and after the drought. To the livestock owner, the percentage mortality of the regional herd is not so much of interest as the number of animals per household that survives the drought. If the number of livestock surviving a drought is the same regardless of initial density, and the benefits derived from livestock are proportional to livestock number, a strategy of maximizing livestock numbers between droughts would be sensible. This results in a higher percentage mortality, as well as a greater number of livestock lost, but greater benefits derived between droughts and the same number of livestock after the drought. This scenario assumes that there are no short- or long-term effects on the vegetation if livestock numbers are high at the onset of drought. Short-term effects on vegetation productivity affect post-drought recovery of livestock, while long-term effects are of concern for the long-term sustainability of maintaining high stocking rates. Very often, there is a mismatch of the time-scale on which the benefits (short-term) and costs (long-term) of heavy grazing occur. In the short term, the benefits commonly exceed the costs, favouring the maintenance of high stocking rates even when there is a long-term risk of degradation (Ash et al., 2002).

It is argued that appropriate management in climatically variable grazing systems should aim at supporting flexible responses to droughts, such as pre-empting drought mortality by marketing surplus animals, and offering opportunities to re-stock by buying in animals (Sandford, 1983; Toulmin, 1994; Behnke and Abel, 1996). Opportunistic strategies are being recognized as better alternatives to constant, conservative stocking rates, even in commercial systems (Mentis et al., 1989; Danckwerts et al., 1993). However, the economic efficiency and environmental sustainability of tight tracking strategies, particularly those which rely on buying stock after droughts, are still debated (e.g. Sandford, 1994; Illius et al., 1998; Campbell et al., 2000). Rainfall is such an important driving variable in rangeland systems, and its variability so high, that simulation modelling is necessary for exploring the economic outcomes of different stocking strategies. These outcomes are influenced by the goods and commodities considered, the economic criteria used to measure outputs and the sequence of wet and dry years over which the system is modelled (Sandford, 2004).

3.2. Dealing with spatial heterogeneity

Pastoral strategies also make use of spatial heterogeneity, as resource availability and rainfall are not evenly distributed across the landscape (Sandford, 1983; Ellis and Swift, 1988; Scoones, 1995) and spatial heterogeneity acts as a buffer to climatic variability (Ash et al., 2002). Pastoralists in arid areas are usually mobile and will move their animals to the best available grazing, covering different areas over the course of the year and between years. Such movements may be in the form of transhumance, which follows a more or less predictable pattern between wet and dry season resources, or more opportunistic movement tracking less predictable patterns of productivity, often caused by patchy rainfall patterns (Coughenour, 1991; Bayer and Waters-Bayer, 1994; Niamir-Fuller and Turner, 1999; Fernandez-Gimenez and Swift, 2003). These movement patterns are thought to enable farmers to maintain high stocking rates even in dry years without putting continuous pressure on the grazing resource throughout the year (Coughenour, 1991; Ellis et al., 1993; Stafford Smith, 1996).

Many rangelands contain small, highly productive areas which make a disproportionately large contribution to the area's total forage production. Examples are riverine areas and drainage lines, which support green grass growth throughout most of the year, or croplands where animals can graze on crop residues in the dry season. Such key resource areas may be responsible for maintaining total animal numbers considerably higher than the predicted carrying capacities, which are based on a homogeneous landscape (Scoones, 1993, 1995). In areas where semi-arid rangelands border on areas where crop production is possible, nomadic pastoralists and settled agriculturalists may have mutually beneficial arrangements where livestock use crop residues in the dry season, allowing the crop farmer to make use of manure. Where such relationships break down, the total number of livestock that can be kept is considerably reduced as exploitation of the entire rangeland by pastoralists relies on mobility and access to crop residues in the dry season (Bayer and Waters-Bayer, 1994). The same happens when the most productive grazing areas are converted to cropland, forcing pastoralists into increasingly marginal land without access to key resources (Homewood and Rodgers, 1987; Scoones, 1990; Dodd, 1994; Desta and Coppock, 2002).

Ellis and Swift (1988), Scoones (1990, p. 392), Ellis et al. (1993) and Bayer and Waters-Bayer (1994) among others also stress that to persist through droughts, pastoralists need to be able to expand their operations into areas not normally used for grazing, and to gain access to outside resources. Neighbouring pastoral groups commonly have arrangements for reciprocal grazing rights that allow movement to better pastures in drought years (Fernandez-Gimenez and Swift, 2003). In household-level studies of livestock dynamics during drought, Homewood and Lewis (1987), Scoones (1990) and Oba (2001) found that mobility during droughts was a key factor contributing to herd survival. Even in the comparatively sedentary communal livestock farming systems in South Africa, there are reports of livestock owners gaining access to wetter areas far beyond their usual pastures in a devastating drought in 2003 (Alcock, 2003 unpublished).

It seems to be widely accepted that the reduction of mobility in semi-arid and arid pastoral systems has increased the risk of degradation because of the way it concentrates grazing pressure on the resource and reduces the opportunities for resting parts of the vegetation (e.g. Coughenour, 1991; Perkins and Thomas, 1993; Oba et al., 2000; Fernandez-Gimenez and Swift, 2003; Kerven et al., 2003). At the same time, remote areas become less frequently utilized and may lose productivity in the absence of periodic grazing (Niamir-Fuller, 1999b). Constriction of mobility is associated with development interventions to settle nomadic pastoralists into ranches, encroachment of rangelands by other forms of land- use such as cultivation and conservation, increasing population densities in rangeland areas, and the proliferation of water points, often accompanied by settlements. In sparsely populated arid areas, grazing impact is often concentrated in piospheres or 'sacrifice zones' around water points or settlements (Perkins and Thomas, 1993; Sullivan, 1999; Leggett et al., 2003), while the rest of the area is largely unaffected. In more densely populated rangelands, such as the former "homelands" of South Africa where villages are only a few kilometres apart, high grazing pressure is much more continuous over the landscape.

There appears to be a need for developing management models which re-introduce mobility, to buffer pastoralists against temporal variability in forage availability, and to reduce localized degradation. When the traditional transhumant movements of cattle ranchers in the USA and South Africa became constricted by settled farmers early in the 20th century (Coughenour, 1991), the solution to perceived degradation caused by the increasingly concentrated and continuous grazing pressure was the introduction of grazing systems such as rotational grazing and resting. These were intended to mimic the evolutionary grazing patterns by native ungulates, which consisted of intense, localized defoliation followed by periods of no grazing. However, in arid areas, movement in response to variable resource availability and drought is necessary on large scales and needs to flexible. An alternative to rotational grazing and other forms of grazing management would be to restore mobility in rangelands. This would in many cases involve expanding the areas under communal tenure and re-establishing access to key resources, a strategy likely to clash with conservation agendas and other land users. The legitimacy of mobility has been questioned and undermined in many countries, and reinstating mobility thus requires a fundamental change in mindset (Niamir-Fuller, 1999b).

The causes and extent of fragmentation, its costs to pastoralists and the environment, and possible ways of reversing or mitigating it are presently the subject of policy debate (Niamir-Fuller, 1999a,b) and large-scale research such as the SCALE Project (Boone and Hobbs, 2003; Galvin et al., 2003; Reid et al., 2003). Options for buffering the effects of temporal variability are moving livestock into other areas, providing supplementary feed, selling and restocking or a combination of the above. The viability of these options in different pastoral systems, and their ecological and economic consequences need to be better explored.

4. Progress and future directions

Much of the initial heat of the debate appears to have dissipated, and recent years have seen a move from the equilibrium—non-equilibrium dichotomy towards a greater awareness of the spatial heterogeneity and temporal variability of semi-arid and arid rangelands. There is increased acknowledgement that rangeland management in drylands is complex and is influenced by spatial, bio-physical, social, cultural and economic factors at a multitude of temporal and spatial scales. The scope of enquiry has expanded from mainly communal rangelands in sub-Saharan Africa to include other continents and tenure systems, such as the pastoral regions of Asia (Fernandez-Gimenez and Allen-Diaz, 1999; Kerven et al., 2003) and commercial rangelands in Australia (e.g. Ash et al., 2002). Answers to some of the central questions in the debate seem to be emerging. Stocking rates do matter, but the grazing pressure and its timing and duration at any given time and part of the landscape are of more consequence than long-term average stocking rates (e.g. Ward et al., 1998). Degradation does not occur everywhere, but can occur in arid landscapes, particularly if grazing pressure is concentrated on certain parts of the landscapes for prolonged periods of time. Different areas and parts of the landscape differ in their susceptibility to transformation, and research needs to take into account this heterogeneity and the scale dependence which is its emergent characteristic. Management of climatically variable rangelands should be flexible and adaptive, but this flexibility is often undermined by fragmentation, increased population pressure, sedentarization and lack of access to information, markets and economic opportunities outside the rangeland system.

Progress in the debate has been hindered by a lack of clarity on the types of systems under discussion. Nomadic pastoral systems, more settled agropastoral systems and commercial ranching are all subject to temporal variability and spatial heterogeneity, but management and policy options are different in different types of rangeland systems. The research, management and policy dimensions of the debate have narrowly concentrated on two components of the system, forage and livestock, and how they interact and affect each other. This ignores other important components of livelihoods in rangelands, such as harvesting and trade in other natural resources, crop cultivation and migration in and out of the pastoral system. Apart from differences that have evolved in traditional systems under the influence of different climatic and ecological constraints (e.g. Ellis and Galvin, 1994), rangelands across the globe have been affected by a variety of other factors such as population growth, encroachment of other land use on rangelands, restriction of mobility, government policies and interventions, access to healthcare and education, urbanization and the different aspirations of the younger generation. Various combinations of these factors have led to far-reaching and often profound changes in the livelihood strategies of pastoralists, and in many areas, livestock make a decreasing contribution to livelihoods (Shackleton et al., 2000).

Despite improved consensus or at least communication among researchers, the translation of research findings into management recommendations and policy has been very slow. Some of this has to do with the difficulty in extrapolating results from controlled experiments to larger scales. It is difficult to make confident recommendations in unpredictable systems, and many researchers appear reluctant to do so. There is also considerable resistance at the policy level to communal tenure, mobility and other flexible land use practices because those too are harder to control and predict. And while the concept of adaptive management is widely considered to be sensible and appealing, changing the laws and policies to allow and facilitate it is not easy. The time-scales at which adaptive management is implemented and monitored exceed conventional research and development planning horizons, and this further explains why it is so rarely put into practice.

There has been a growing recognition of the need to integrate the ecological, economic, social and institutional dimensions of rangeland research. Nevertheless, problems communicating across disciplines still persist, and there remains a tension between those who see the debate mainly in terms of ecological theory and those who see it in a larger socio-political context. Some of the latter feel frustrated at the detached approach of many ecologists and feel that policy questions should inform the natural scientist's research agenda. A persistent problem in these discussions is that pastoralists are still under-represented at defining the research agenda with their needs, priorities and knowledge. They remain in most cases subjects of research, development and policy rather than playing an active role.

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