Managing low-output agroecosystems sustainably: the importance of ecological thresholds

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Abstract: Managing vegetation to achieve ecological, economic, and social goals is difficult. Inherent complexity among ecosystem components and unpredictable climate often limit opportunities for converting cultural inputs to harvestable products. In addition, the long lag time between treatment and financial return makes capital investment in vegetation management economically risky. One tool that can assist land managers in dealing with these constraints is the identification of ecological thresholds and elucidation of processes that signal undesirable change before it is entrained. This approach places a premium on early detection of degrading processes and implementation of management responses in the initial stages of land degradation. Managerial expertise and manipulation of naturally occurring processes, rather than cultural inputs, are key management decisions. In this paper we review current applications of the threshold concept as a management decision point and propose modifications for use in managing plant communities with low potential for annual economic return. We also propose that research and institutional programs for sustainable land management shift direction toward identifying ecological thresholds and focus on developing low-input responses to avoid, rather than restore, land degradation.

Résumé : Il est difficile de gérer la végétation pour satisfaire à la fois des objectifs écologiques, économiques et sociaux. La complexité inhérente liée aux diverses composantes de l'écosystème et la nature aléatoire du climat limitent souvent la possibilité de convertir les interventions sylvicoles en produits récoltables. De plus, le long décalage entre les interventions et les retombées financières rend l'investissement de capitaux dans la gestion de la végétation risqué du point de vue économique. L'identification de seuils écologiques et la compréhension des processus qui annoncent des changements indésirables avant qu'ils ne soient enclenchés constituent un outil qui peut aider les gestionnaires du territoire à composer avec ces contraintes. Cette approche met à l'avant-plan les processus de dégradation et l'implantation de mesures d'aménagement aux premiers signes de dégradation des terres. L'expertise en gestion et la manipulation des processus naturels, plutôt que les impératifs sylvicoles, sont des décisions clés d'aménagement. Dans cet article, nous passons en revue les applications courantes du concept de seuil critique en tant qu'élément de décision de gestion et nous proposons des modifications pour son utilisation dans la gestion des communautés végétales qui ont un faible potentiel de rentabilité économique sur une base annuelle. Nous suggérons également que la recherche et les programmes institutionnels portant sur l'aménagement durable du territoire se réorientent vers l'identification des seuils écologiques et se concentrent sur l'élaboration d'interventions à faible coût pour éviter la dégradation des terres plutôt que d'avoir à les restaurer.

[Traduit par la Rédaction]

Introduction

Vegetation management

While the term "ecosystem management" (Grumbine 1994) remains controversial and the specific practices associated with it are still contentious, the concepts have credibility (Haeuber and Franklin 1996). The core themes, managing for ecological integrity, including human activities, developing strategic partnerships, adapting management, and changing institutions are generally well accepted and supported by a broad cross section of professionals and the public (Grumbine 1997).

Received August 23, 1998. Accepted March 3, 1999.

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The most visible manifestation of ecosystem management, vegetation management, is probably the primary point of contact for most people. Vegetation is either directly harvested (forage, timber, native foods), supports consumptive industries (scenery, recreation, hunting) or affects the ecosystem level output of goods and services (biodiversity, watershed). In most cases, changes in vegetation attributes form the basis for monitoring and are used to predict other variables of interest (i.e., erosion).

Consequently, vegetation management reflects many of the uncertainties and reservations associated with ecosystem management. A constantly changing mix of ecological, economic, and social constraints and opportunities make decisions difficult. This complexity is inherent across the complete range of decisions; from deciding what mix of products and services the vegetation should yield to determining what management practices are most likely to optimize those outputs.

Ecologically, there are few ecosystems that do not exhibit nonequilibrium dynamics at some spatial and temporal level (Holling 1996). Short-term climatic variability (Conner 1994; McKeon et al. 1990), long-term climate change (Neilson 1986; Beerling and Woodward 1993; Nicholls 1997) and inherent properties of the vegetation components (Kay 1991) often interact with management and make it difficult to determine causation. In addition, the long-term nature of vegetation management means that management actions (or lack of action) implemented at one point in time may not have a discernible effect until well into the future, often beyond the working life of managers.

Economically, as the value of land and terms of trade have changed, expenditure to implement vegetation management practices has become more difficult to justify (Stuth et al. 1991). The risky and long-term nature of the return on "land improvement" practices favors safer, more short-term investments for private funds (Miller and Scanlan 1997). Government cost-sharing programs, long the source of funding for private range and forest vegetation management programs, are also being reduced and redirected, limiting the flexibility of land managers. On public land, the expenditure of funds to achieve private economic gain is no longer acceptable (see Belsky 1996).

Finally, the social contract between land managers and the general public has also changed. In addition to food and fiber production, managed ecosystems are now viewed as the source of a wide range of public benefits (Heissenbuttel 1996). While there is an ongoing debate about who pays for benefits such as clean water, habitat for endangered species, maintenance of biological diversity, and sequestration of greenhouse gases, there is little question that multiple-benefit management is, and will remain, a central focus of ecosystem management.

Thus, managers are frequently confronted with situations where they are expected to achieve poorly defined outcomes via the application of untested practices over unknown time frames. On-the-ground vegetation managers have traditionally relied heavily on field research as a basis for setting objectives and implementing practices. However, the vast majority of field research for vegetation management has been directed toward the output of a single or a limited set of products. Research for commodity production in managed ecosystems is generally seen from the narrow perspective of sustained yield of the primary product (Brown and Ash 1996). Vegetation dynamics are interpreted with a sharp focus on the product of interest and generally less focused on the processes and mechanisms that do not appear to be linked directly to production. It is not surprising that linear succession, with a strong emphasis on elimination of undesirable species, has been the basis for management (see Westoby et al. 1989). While the most obvious examples of this type of approach are management techniques for commodity production on rangelands (livestock) and forests (lumber), a similar approach dominates most land uses (e.g., military land, parks, watersheds).

The adoption and implementation of more robust models of vegetation change such as state-and-transition models (Westoby et al. 1989) have accommodated a somewhat wider view of how human activity interacts with biotic and abiotic components of an ecosystem. However, these models have largely been constructed with existing information developed in the context of commodity production and they lack sensitivity in describing interactions between specific disturbances and ecosystem function. To date, there has been little field research aimed at developing quantitative models of vegetation change that are temporally and spatially explicit enough for land management decisions. In particular, models for vegetation management lack sensitivity to the interactions of management practices and climatic fluctuations that accurately describe the conditions that initiate a nonlinear change. These ecological thresholds represent critical decision points. Early and accurate detection is essential to predicting vegetation response and therefore to the identification and selection of appropriate management options.

Ecological thresholds

The existence and importance of thresholds in the dynamics of nonequilibrium ecological systems is well established (Holling 1973; May 1977). Thresholds can be defined as the point of entry into a new domain or region of ecosystem function. In terms of the management of agroecosystems, ecological thresholds can be defined as the point in time when processes that result in a change in ecosystem function are entrained, and management actions must shift from maintaining existing processes to reversing degrading processes. While they have been identified and integrated into post-hoc analyses of ecosystem change (Archer 1989; Friedel 1991; Laycock 1991), there has been a distinct lack of the application of the concept of thresholds to land management decision-making. Perhaps this can be attributed to the notion that linear, stabilizing forces, or actions contribute inertia to the system and are important to productivity (Holling 1996). Conversely, the nonlinear destabilizing forces, or actions that result in changes in the system, are more important to maintaining diversity, resilience, and opportunity. This may be explained by the fact that we are generally interested only in the output of one product from a managed ecosystem, we tend to focus research and management only on the processes that stabilize that particular output. Processes that destabilize output, but contribute to the capacity of ecosystems to absorb and recover from stress and disturbance, are generally seen as less relevant and are often overlooked when experiments are designed, executed, and interpreted.

The management of land-based production systems to maintain environmental quality has become the focus of much policy debate (Stafford Smith et al. 1997). Unfortunately, there is little evidence on which to base decisions. Although there is an emphasis on integrating production and environmental quality, there is little useable information on which to base on-the-ground decisions or policy. If policies and programs are to enhance environmental quality and achieve a public good, research and implementation must move out of the single-cause, single-effect relationships that seem to dominate (Brown and MacLeod 1996). In addition, information should be put into a decision-making framework that identifies critical conditions and events (thresholds) and links them to managerial responses to achieve specified outcomes rather than focusing on overly simplified relationships between cause and effect.

Our objectives in this paper are to (*i*) review the threshold concept as currently used in vegetation management and (*ii*) illustrate the existence and importance of ecological thresholds in decision making for achieving vegetation management objectives.

Applying thresholds to management

Our approach pushes the concept of integrating knowledge about thresholds into decision making to a new level. To date, the most widespread application of the threshold concept in land management has been in the area of weed management on cropland (Auld and Tisdell 1987). In weed management, damage thresholds (weed populations that elicit a negative crop response), economic thresholds (weed populations that reduce net income), and period thresholds (time periods when weeds are more damaging) have been used as decision-making aids (Coble and Mortensen 1992). An assumption that is critical in cropland weed management is that the damage threshold and the economic threshold are relatively close in time and a management response can alleviate the damage

Cousens and Mortimer (1995) defined several thresholds describing the interactions between annual crops and weeds. Importantly, they extended the concept to include an "economic optimum threshold" to identify the weed density at which control decisions should be made. Even though they extended the concept to include more than 1 year, they still assumed decisions about weed management were linked to crop yield within a growing season (p. 211).

Conversely, processes that govern perennial vegetation dynamics may be entrained for many years before the change is detectable or the economic return is reduced (Brown and Ash 1996). In many cases, by the time an economic threshold is crossed, the management response to arrest and reverse the trajectory of change is both expensive and risky. In some systems, reversal of change may not be possible with existing technologies (Albaladejo et al. 1998). Thus, the ability to detect an ecological threshold and respond with appropriate management is a key element of sustainable land management. In the remainder of this paper we will discuss how the concept of ecological thresholds can be used to improve vegetation management. We will also illustrate how research can be of greater value to land management by including the identification of spatial and temporal thresholds in the analysis of existing data and in the design and implementation of new experiments.

Managing low-input agroecosystems

Most rangelands are rangelands for a one very good reason: they cannot be farmed economically. The same is generally true of forests. Generally, the reason that they can't be farmed is edaphic, climatic, or topographic. There are very few rangelands in the United States or Australia that are not cultivated for social reasons or because they are too isolated from markets. In contrast to agronomic systems, which are developed by replacing or modifying the components of an ecosystem to improve the conversion efficiency of nutrients and water to harvestable product (Giampietro et al. 1992), range and forest ecosystems are only partially modified, leaving much of the native vegetation intact.

Even highly modified range and forest systems are low-input compared with agronomic systems. These low-input systems usually consist of multiple life forms (grasses, forbs, shrubs, trees) and management units are generally more heterogeneous. The result is that low-input systems have many more possible interactions. Consequently, the probability that increased inputs will be quantitatively reflected in increased outputs is much lower in rangelands than it is in more homogeneous agronomic systems. Because they are different from croplands in some very important attributes, we should not expect low-input ecosystems to respond as agronomic systems, and we should not expect to manage them similarly. Thus, the fundamental rule of managing rangelands, and other low-input systems, is to live within the biophysical limitations rather than trying to overcome them with inputs. This represents the difference between ecology and agronomy, as they are traditionally practiced, as a basis for management (Brown 1994).

In their attempts to overcome the limitations of the environment, rangeland researchers and managers have typically sought to increase outputs by increasing inputs (improved forage species, fertilizer additions, removal of nonforage species) or by improving harvest efficiency (genetically manipulating livestock to tolerate low forage quality and (or) quantity). Brown and Ash (1996) have reviewed the effects of input-based technologies on the ecology of Australian tropical rangelands. While the intent of technology developers was admirable, the result has been undesirable changes in native grass species composition, loss of native grass cover and increases in shrubs (both exotic and native). These problems are almost universally associated with rangelands grazed by domestic livestock.

Essentially, this approach to vegetation management was agronomic. Researchers and managers believed the application of cultural resources could overcome limitations inherent in rangeland ecosystems by stabilizing processes such as net primary productivity and livestock intake. Unfortunately, attempts to stabilize these processes in the short term ignored the need to maintain ecosystem integrity and stability that served as a basis for maintaining long-term productivity. A more logical and realistic approach is to shift the focus of management (both research and implementation) to detecting changes in destabilizing processes at critical times and critical places and responding with management to avoid undesirable change or restore ecosystem function in degraded landscapes. Thus, the first principle of managing low-input agroecosystems is to initiate management actions before undesirable changes have an effect on output.

A second principle of managing low-input agroecosystems is to limit capital investment. Because of the inherent climatic variability associated with rangelands, an investment in "improving" vegetation through chemical or mechanical additions is risky (Brown and MacLeod 1996). Additionally, investments in vegetation management generally take years to pay off and the net present value of capital investments is often negative by the time benefits are realized. Finally, the ecological complexity of most low-input systems greatly constrains a manager's ability to convert inputs into harvestable product.

Taking these two principles into account, there is a need for an approach to land management that primarily seeks to avoid degradation through early detection and response. Manipulating ecological processes by the application of managerial skill rather than cultural application also reduces the need for investment of scarce capital in risky ventures with long-term payoff. We suggest that the application of the concept of thresholds is a key element in successful and sustainable land management.

Application of the ecological threshold concept to research and management

To demonstrate the utility of ecological thresholds as an important part of land management, we will focus on the two most widespread forms of land degradation, shrub invasion and loss of perennial grass cover. In these case studies, we will demonstrate both how existing information and experiments designed specifically to define conditions associated with thresholds can be used to improve the quality of decisions.

Shrub or tree invasion

Many extensively managed rangelands are undergoing rapid invasion by exotic shrub species and (or) increase of native shrub species (see Archer 1994 for review). Shrub increase, both native and exotic, fundamentally changes the way ecosystems process nutrients (Schlesinger et al. 1996), water (Thurow 1991), and energy (Archer and Smeins 1991). The resulting vegetation often provides diminished services in the form of reduced forage production (Archer 1989), altered wildlife habitat (Humphries et al. 1991) and lower quality and quantity of water (Thurow 1991). The management response has traditionally been to wait until the increase in shrub density reduces income enough to justify the application of chemical or mechanical techniques to restore productive capacity (Scifres 1980; Noble 1998). However, the economics of rangeland management and the environmental impact of chemical and mechanical practices have forced managers to rethink the application of these technologies.

In the tropical woodlands and grasslands of Australia, exotic shrubs have created management problems that affect both conservation and pastoral values. Most of the species were introduced around the turn of the century for a wide variety of reasons. *Acacia nilotica* ssp. *indica* (Benth.) Brenan is a particularly troublesome species. It is an arborescent legume introduced to the open grasslands of western Queensland, an area naturally depauperate of trees, around the turn of the century to provide shade and increase lambing percentages (Carter and Cowan 1993). Land use in the area is almost exclusively livestock grazing, with cattle replacing sheep as the dominant grazer in the early 1970s. Stocking rates are relatively low. The recommended rate is 10–15 ha/animal unit year (AUY). An AUY is a standard measure that describes the amount of forage consumed by a cow/calf unit in 1 year. Although forage quality is relatively good, an extended dry season (April–December) limits animal performance and animals are usually 2 or 3 years old before they are marketed.

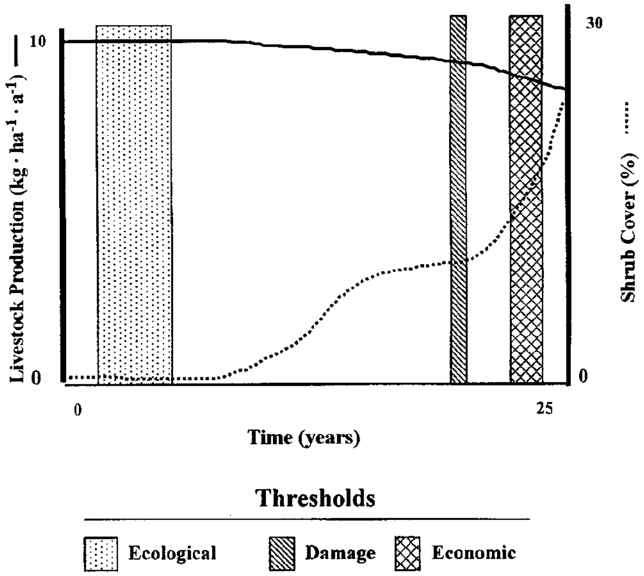
Acacia nilotica populations have expanded rapidly in the past 50 years. It establishes readily along riparian areas and open drains as well as in uplands where it is effectively converting a grassland ecosystem into open savanna and shrub thicket. A summary of the rates, patterns and causes of the invasion can be found in Brown and Carter (1998). Although the invasion is relatively widespread in range, density within its range is highly variable and more than 95% of the ecosystem is not affected or is in the early stages of invasion.

Seed production is relatively reliable when trees are in favorable habitats. Trees growing at a moderate density (25 adults/ha) across a typical landscape can produce more than 40 000 seeds/ha (I. Radford and J.R. Brown, in preparation). Seed dispersal is almost exclusively by domestic cattle (Kriticos et al. 1999), and the spatial and temporal patterns of invasion reflect grazing and land-use patterns (Brown and Carter 1998). Germination of the large seeds (50 ± 0.04 mg/seed; mean ± SE) typically exceeds 80% when they are scarified (Brown et al. 1998). Seeds associated with dung piles exceeded 1000/ha during the wet season (I. Radford and J.R. Brown, in preparation).

Based on field trials, seedling emergence and recruitment in seasons of average precipitation is common and adequate to maintain scattered, low-density stands (J.R. Brown and L. Fagan, in preparation). Rapid tap root elongation enables seedlings to escape competition with surrounding herbaceous vegetation when soil moisture is available for more than 60 days (Brown et al. 1998). In years of above-average precipitation, seedling recruitment is high (>25% survival of emerging seedlings). Once seedlings have reached a minimum size (>25 cm aboveground height), survival through the ensuing dry season is highly probable and recruitment into the juvenile (>3 years old) population occurs (J.R. Brown and L. Fagan, in preparation). While seedlings are extremely intolerant of fire (>99% mortality; I. Radford and J.R. Brown, in preparation), juveniles have very high tolerance of drought and top removal and are likely to survive into adulthood regardless of management and climate (Brown and McIvor 1993; F. Tiver, in preparation).

In these experiments, the objective was to determine the relationship between resource availability and recruitment to predict biotic and abiotic conditions associated with population expansion as a basis for early detection. Determining disturbance levels (top removal, fire, grazing) that limit successful recruitment of seedlings and juveniles is also an important part of determining an effective management response. Based on these experiments and observations, *Acacia nilotica* population dynamics, dispersal, and the ecosystem dynamics of the Mitchell Grasslands, we have developed a graphic representation of the temporal dynamics of the invasion (Fig. 1). Although the threshold between grassland and shrubland is generally defined by the damage threshold (i.e., when canopy cover has increased sufficiently to trigger a reduction in forage production), we suggest that the management threshold is more appropriately described by the ecological threshold. From an economic perspective, the economic threshold represents that point at which tree density has a negative effect on livestock performance and the point at which most land managers would take action (Miller and Scanlan 1997). During years 2–4, there is an

Fig. 1. The relationship between increases in *Acacia nilotica* cover in the Mitchell Grasslands and livestock production 1960–1994.



opportunity for management-induced actions to limit the recruitment and expansion of *Acacia nilotica*. However, once that time period has passed, there is little opportunity for management to remove individuals from the population other than through the application of high-cost chemical or mechanical treatments. By monitoring processes that are indicative of an approaching threshold (seed production, dispersal, initial recruitment, and survival through the first dry season), managers should initiate cost-effective actions to limit the expansion of this invasive weed such as controlling livestock movements and burning or manually killing scattered juvenile trees (Grice and Brown 1996).

Loss of perennial tussock grass cover

Maintaining perennial grass cover is at the core of sustainable management in most of the world's grazinglands. Perennial grass cover supports grassland productivity (Tilman et al. 1996), stability (McNaughton 1993), and other functions (Risser 1995). When degradation does occur, the response is generally to introduce government-assisted restoration programs or simply to continue to manage degraded land for a diminished set of services. Neither of these responses is sustainable or acceptable.

Perennial grasses are the basis of livestock production systems in Australian tropical rangelands (Ash et al. 1995). Among the management challenges faced by pastoralists are the highly variable climate, the extended dry season and the low quality forage. More than 80% of the average annual rainfall of 668 mm at Charters Towers, Queendland (20°11′S, 146°43′E) occurs during December–April but is highly varible. Soils are nutrient poor (soil P as low as 5 ppm; soil N as low as 0.05%; McIvor and Gardener 1991). In general, perennial C_4 tussock grasses dominate the herbaceous layer of the vegetation and forage quality is low (Brown and Ash 1996). As a result of the low-quality forage and the extreme variability in production, stocking rates are low, generally recommended at about 10 ha/AUY. Animals are harvested at 3 years of age in good years and at 4 years if conditions are poor. In an attempt to overcome these limitations, research and development in the 1950s was devoted to developing new varieties of livestock that could better tolerate low forage quality and extended drought by increasing intake (Gardener et al. 1990). This new technology (based on *Bos indicus* bloodlines) was widely adopted by land managers without a concomitant reduction in livestock numbers. The result has been widespread overgrazing, loss of perennial grass cover, soil erosion, and land degradation.

Research, development, and application of grazing management technology typically focused on productivity at the livestock or vegetation level as the measures of success. As a result, monitoring methods have tended to revolve around detecting declines in plant and animal productivity, and related decisions are intended to restore that productivity. However, recent work suggests that processes that lead to reduced productivity, and ultimately land degradation, may be entrained long before monitoring methods focused on productivity can detect changes. Ash et al. (1995) demonstrated that livestock performance can remain high even though perennial grass basal area is declining. The ability of livestock to switch to alternative forage species and maintain performance demonstrates the difficulty of detecting changes across multiple trophic levels and making meaningful management responses.

Surprisingly, monitoring changes in perennial grass productivity and (or) species composition may not provide an adequate warning, either. The ability of the vegetation to cope with redistributed water and nutrients and maintain biomass production can be misleading (B. Northup and J. Brown, in preparation). Net primary productivity of perennial tussock grasses in a long-term heavily grazed paddock quickly recovers when stocking rates are lowered (or in relatively good rainfall years) indicating recovery (Table 1). However, basal area on the degraded, lightly grazed site remained low compared with a well-managed site (0.8 vs. 1.5%, respectively) and showed little recovering over the 4-year period of the experiment. Similarly, bare ground was much higher on degraded sites (92 vs. 77%) and cover was much lower (8 vs. 21%) compared to lightly grazed communities. Basal area is the best indicator of the ability of a plant community to increase perennial grass cover in periods of resource availability (Briske 1991; McIvor and Gardener 1990). Soil cover is the best indicator of soil protection and the ability to resist erosion (Blackburn and Pierson 1994).

Perhaps the most reliable indicator of degradation and recovery status, the spatial distribution of plant and soil resources and processes at scales of less than 1 m (B. Northup and J. Brown, in preparation) illustrates an even greater disparity among the treatments. From 1993 to 1996, patch size remained constant (7 vs. 7 boundary changes) in the good condition, lightly stocked treatment (Table 1), increased dramatically (7 vs. 11 boundary changes) in the good condition, heavily stocked treatment and showed little change (11 vs. 10 boundary changes) in the poor condition, lightly stocked treatment. Even though the ecological process most closely linked to livestock production, forage supply, recovered quickly in response to reduction in grazing pressure, there was little improvement in the ability of the herbaceous plant community to resist erosion, regain ground cover, basal area

Table 1. Changes in vegetation attributes over four years (1993–1996) in an Australian grassland exposed to livestock grazing.

Characteristics	Year 1993	1994	1995	1996
S1–LG				
Yield (kg/ha)	1349 (105)	980 (47)	1403 (91)	1612 (92)
Basal area (%)	1.8 (0.2)	2.0 (0.1)	1.8 (0.1)	1.5 (0.1)
Bare ground (%)	20 (2)	48 (2)	69 (2)	79 (2)
Ground cover (%)	80 (2)	52 (3)	31 (2)	21 (2)
Boundary changes*	7	—	—	7
S1–HG				
Yield (kg/ha)	1220 (23)	370 (17)	164 (17)	79 (19)
Basal area (%)	1.8 (0.1)	1.7 (0.2)	0.2 (0.0)	0.1 (0.0)
Bare ground (%)	21 (3)	65 (3)	84 (2)	86 (2)
Ground cover (%)	79 (3)	35 (3)	16 (2)	14 (2)
Boundary changes	7	—	—	11
S2–LG				
Yield (kg/ha)	106 (8)	859 (59)	627 (112)	1353 (135)
Basal area (%)	0.6 (0.1)	0.7 (0.1)	0.9 (0.1)	0.8 (0.1)
Bare ground (%)	72 (2)	73 (1)	78 (2)	92 (2)
Ground cover (%)	28 (2)	27 (3)	22 (2)	8 (1)
Boundary changes	11	—	—	10

Note: Values are means with SE given in parentheses. Initial pasture conditions were long-term light grazing (S1) and long-term heavy grazing (S2). Grazing intensities for the period 1993–1996 were light grazing (LG; <25% of yearly production removed by grazing) and heavy grazing (HG; >75% of yearly production removed by grazing).

*Boundary change is the number of changes in herbaceous dominants (perennial, annual or mixed) that occur within 1 × 0.25 m quadrats along a 100 m transect.

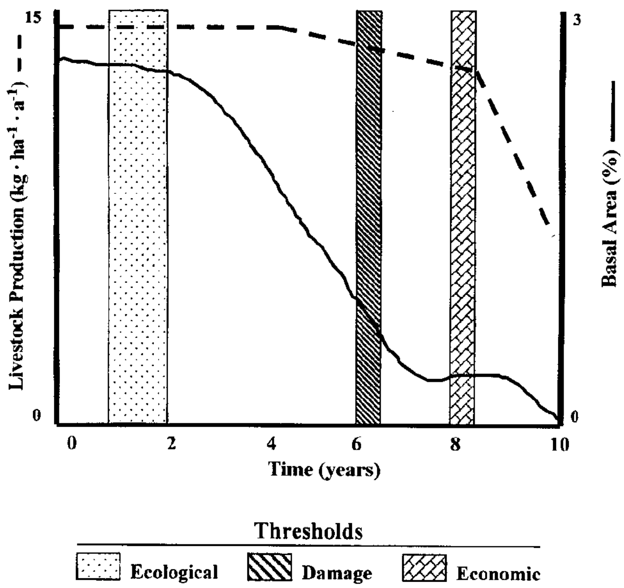
or patch structure. All of those attributes are extremely important in maintaining ecosystem function in the face of future heaving grazing and (or) drought events (McIvor and Gardener 1990).

This series of experiments was designed specifically to expose and quantify thresholds in vegetation change in these woodlands and to provide information for land managers to make reponses. In fact, there are several thresholds. The economic threshold (Fig. 2) represents the point at which livestock performance suffers from changes in the vegetation and is generally when pastoralists make a management response. The damage threshold represents the point at which changes are detectable in the vegetation by traditional measures of net primary productivity (a reduced forage supply). It is important to note that, at this point, restoration of ecosystem function (as indicated by basal area and ground cover) is difficult, if not impossible, to achieve. In reality, the ecological threshold represents the point at which management actions should be initiated if deleterious changes are to be avoided.

Conclusions

In these examples, we have demonstrated that it is possible to design and interpret experiments that define the temporal thresholds of vegetation change in a context that is usable by managers. We have also shown that a reinterpretation of existing data to identify thresholds can be of value to managers. We have identified several key elements in this approach that will greatly enhance the utility to land managers.

Fig. 2. The relationship between loss of perennial tussock grass basal area and livestock performance (redrawn from Ash et al. 1996).



In particular, the elucidation of the interaction of climatic conditions and management decisions associated with an approaching threshold will be of value to land managers. Seldom does a land manager have the resources to commit to a monitoring program that will provide the type of information we have demonstrated here. However, it is important for research scientists to remember that the precision of the data is seldom the limiting factor in decision making. More often, it is that the data lacks accuracy commensurate with the timescale of the decision. Many of the processes that support ecosystem function are highly variable and primarily reflect annual or seasonal climatic fluctuations. This short-term variability can mask the long-term direction of change and can often delay detection of threshold conditions until the change is entrained. Thus, indicators that are insensitive to short term variability and sensitive to long-term change in ecosystem function should be a goal of land management research.

Another key element of this approach is that research and implementation must focus on separating the processes that support productivity (stabilizing processes) from processes that support ecosystem function (destabilizing processes; Holling 1996). In some cases they may be the same processes operating at different rates or different timescales. Researchers should seek to define and quantify the processes that lead to degradation with as much enthusiasm as we have expended on defining the processes that support productivity.

A major shortcoming in what we have demonstrated here is that we have limited our examples to the plant community scale. Obviously, few natural and seminatural vegetation managers work solely at the plant community scale, and most of the products and services from vegetation emerge at the landscape level and higher. In addition to the temporal dynamics we have demonstrated, it is important to define what part(s) of production landscapes are most at risk and most accurately reflect the key processes on which decisions should be based.

By and large, we are still limited to making comparisons between functioning ecosystems (intact) and degraded systems. This is unlikely to change quickly. However, it makes sense that no one wants a degraded system. Conversely, we probably cannot afford to manage more than a small amount of land to retain all of the properties of pristine systems. The driving motivation behind land management research, then, should be to focus on identifying conditions that are associated with undesirable change in ecosystem function. It is unreasonable and naïve to expect land managers and the public will maintain an unchanging view of how they want ecosystems to function. Using these guidelines, we feel that the use of the concept of ecological thresholds can be an important part of informed decision making for land management.

References

Albaladejo, J.M.M., Roldan, A., and Castillo, V. 1998. Soil degradation and desertification induced by vegetation removal in a semiarid environment. Soil Use Manage. **14**: 1–5.

Archer, S.R. 1989. Have southern Texas savannas been converted to woodlands in recent history? Am. Nat. **134**: 545–561.

Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. *In* Ecological implications of livestock herbivory in the west. *Edited by* M. Vavra, W. Laycock, and R. Pieper. Society for Range Management, Denver, Colo. pp. 13–68.

Archer, S.R., and Smeins, F.E. 1991. Ecosystem level processes. *In* Grazing management: an ecological perspective. *Edited by* R.K. Heitschmidt and J.R. Stuth. Timber Press, Portland, Oreg. pp. 109–139.

Ash, A.J., McIvor, J.G., Corfield, J.P., and Winter, W.H. 1995. How land condition alters plant–animal relationships in Australia's tropical rangelands. Agric. Ecosyst. Environ. **56**: 77–92.

Ash, A.J., Brown, J.R., and Cowan, D.C. 1996. Transitions between vegetation states in tropical tallgrass: falling over cliffs and slowly climbing back. *In* Conference Papers, 9th Biennial Australian Rangeland Conference, 24–27 Sept. 1996, Port Augusta, South Australia. *Edited by* L.P. Hunt and R. Sinclair. Australian Rangeland Society. pp. 219–220.

Auld, B.A., and Tisdell, C.A. 1987. Economic thresholds and response to uncertainty in weed control. Agric. Syst. **25**: 219–227.

Beerling, D.J., and Woodward, F.I. 1993. Ecophysiological responses of plants to global environmental change since the last glacial maximum. New Phytol. **125**: 641–648.

Belsky, A.J. 1996. Viewpoint: western Juniper expansion: is it a threat to arid northwestern ecosystems? J. Range Manage. **49**: 53–59.

Blackburn, W.H., and Pierson, F.B., Jr. 1994. Sources of variation in interrill erosion on rangelands. *In* Variability in rangeland water erosion processes. *Edited by* W.H. Blackburn, F.B. Pierson Jr., G.E. Schuman, and R. Zartman. Soil Science Society of America, Madison Wis. pp. 1–10.

Briske, D.D. 1991. Developmental morphology and physiology of grasses. *In* Grazing management: an ecological perspective. *Edited by* R.K. Heitschmidt and J.R. Stuth. Timber Press, Portland Oreg. pp. 85–108.

Brown, J.R. 1994. Ecology as a basis for rangeland management: performance criteria for testing models. Trop. Grassl. **28**: 206–213.

Brown, J.R., and Ash.,A.J. 1996. Managing resources: moving from sustainable yield to sustainability in tropical rangelands. Trop. Grassl. **30**: 47–57.

Brown, J.R., and Carter, J. 1998. Spatial and temporal patterns of exotic shrub invasion in an Australian tropical grassland. Landscape Ecol. **13**: 93–102.

Brown, J.R., and MacLeod, N.D. 1996. Integrating ecology into natural resource management policy. Environ. Manage. **20**: 289–296.

Brown, J.R., and McIvor, J.G. 1993. Ecology of woody weed invasions in the tropical woodlands of north-eastern Australia: implications for management. *In* Proceedings of the 10th Australian Weeds Conference, Weed Society of Queensland, 6–10 Sept. 1993, Brisbane, Australia. pp. 471–474.

Brown, J.R., Scanlan, J.C., and McIvor, J.G. 1998. Herbaceous competition as a limiting factor in shrub increase: a test with different growth forms. J. Veg. Sci. **9**: 829–836.

Carter, J.O., and Cowan, D.C. 1993. Population dynamics of prickly acacia, *Acacia nilotica* subsp. *Indica* (Mimosaceae). *In* Pests of pastures: weed, invertebrate and disease pests. *Edited by* E.S. Delfosse. CSIRO Information Services, Melbourne, Australia. pp. 128–132.

Coble, H.A., and Mortensen, D.A. 1992. The threshold concept and its application to weed science. Weed Technol. **6**: 191–195.

Conner, J.R. 1994. Assessing the socioeconomic impacts of climate change on grazing lands. Clim. Change, **28**: 143–157.

Cousens, R., and Mortimer, M. 1995. Dynamics of weed populations. Cambridge University Press, Cambridge, U.K.

Friedel, M.H. 1991. Range condition assessment and the concept of thresholds: a viewpoint. J. Range Manage. **44**: 4 22–426.

Gardener, C.J., McIvor, J.G., and Williams, J. 1990. Dry tropical rangelands: solving one problems and creating another. *In* Australian ecosystems: 200 years of utilization, degradation and reconstruction. *Edited by* D.A. Saunders, A.J.M. Hopkins, and

R.A. How. Ecological Society of Australia, Canberra, Australia. pp. 185–194.

Giampietro, M., Cerretelli, G., and Pimentel, D. 1992. Energy analysis of agricultural ecosystem management: human return and sustainability. Agric. Ecosyst. Environ. **38**: 219–244.

Grice, A.C., and Brown, J.R. 1996. An overview of the current status of weed management in Australian rangelands. *In* Proceedings of the 11th Australian Weeds Conference, 30 Sept – 3 Oct. Melbourne. Weed Science Society of Victoria, Melbourne, Australia. pp. 195–204.

Grumbine, R.E. 1994. What is ecosystem management? Conserv. Biol. **8**: 27–38.

Grumbine, R.E. 1997. Reflections on "What is ecosystem management?" Conserv. Biol. **11**: 41–47.

Haeuber, R., and Franklin, J. 1996. Perspectives on ecosystem management. Ecol. Appl. **6**: 692–693.

Heissenbuttel, A.E. 1996. Ecosystem management—principles for practical application. Ecol. Appl. **6**: 730–732.

Holling, C.S. 1996. Surprise for science, resilience for ecosystems, and incentives for people. Ecol. Appl. **6**: 733–735.

Holling, C.S. 1973. Resilience and stability and of ecological systems. Annu. Rev. Ecol. Syst. **4**: 1–23.

Humphries, S.E., Groves, R.H., and Mitchell, D.S. 1991. Plant invasions of Australian ecosystems: a status review and management directions. Report to the Australian National Parks and Wildlife Sevice, Endangered Species Program and CSIRO, Kowari 2, ANPWS, Canberra, Australia.

Kay, J.J. 1991. A non-equilibrium thermodynamic framework for discussing ecosystem integrity. Environ. Manage. **15**: 483–495.

Kriticos, D., Brown, J.R., Radford, I., and Nicholas, M. 1999. Biological control of invasive perennial plants in rangeland ecosystems: *Acacia nilotica* as a case study. Biol. Control. In press.

Laycock, W.A. 1991. Stable states and thresholds of range condition on North American rangeland: a viewpoint. J. Range Manage. **44**: 427–433.

May, R.M. 1977. Thresholds and breakpoints in ecosystems with a multiplicity of stable states. Nature (London), **269**: 471–477.

McIvor, J.G., and Gardener, C.G. 1990. Soil and vegetation characteristics of dry tropical rangelands for prediciting regeneration in exclosures *In* Australian ecosystems: 200 years of utilization, degradation and reconstruction. *Edited by* D.A. Saunders, A.J.M. Hopkins, and R.A. How. Ecological Society of Australia, Canberra, Australian Capital Territory. pp. 273–277.

McKeon, G.M., Day, K.A., Howden, S.M., Mott, J.J., Orr, D.M., Scattini, W.J., and Weston, E.J. 1990. Northern Australian savannas: management for pastoral production. J. Biogeogr. **17**: 355–372.

McNaughton, S.J. 1993. Biodiversity and ecosystem function of grazing ecosystems. *In* Biodiversity and ecosystem function. *Edited by* E.D. Schulze and H.A. Mooney. Springer-Verlag, New York. pp. 361–383.

Miller, E.N., and Scanlan, J.C. 1997. Linking ecology and economics for woody weed management in Queensland rangelands. *In* Proceedings of the 41st Australian Agricultural and Resource Economics Society, 20–25 Jan. 1997, Gold Coast, Queensland.

Neilson, R.P. 1986. High-resolution climatic analysis and southwestern biogeography. Science (Washington, D.C.), **232**: 27–34.

Nicholls, N. 1997. Increased Australian wheat yields due to recent climate trends. Nature (London), **387**: 484–485.

Noble, J.C. 1998. The delicate and noxious scrub. CSIRO Press, Melbourne, Australia.

Risser, P.G. 1995. Biodiversity and ecosystem function. Conserv. Biol. **9**: 742–746.

Schlesinger, W.H., Raikes, J.A., Hartley, A.E., and Cross, A.F. 1996. On the spatial pattern of soil nutrients in desert ecosystems. Ecology, **77**: 364–374.

Scifres, C.J. 1980. Brush management. Texas A&M University Press, College Station.

Stafford Smith, M., Morton, S., and Ash. A. 1997. On the future of pastoralism in Australia's rangelands. *In* Frontiers in ecology. *Edited by* N. Klomp and I. Lunt. Elsevier, Oxford, U.K. pp. 7–16

Stuth, J.W., Conner, J.R., and Heitschmidt, R.K. 1991. The decision making environment and planning paradigm. *In* Grazing management: an ecological perspective. *Edited by* R.K. Heitschmidt and J.R. Stuth. Timber Press, Portland, Oreg. pp. 201–223.

Thurow, T.L. 1991. Hydrology and erosion. *In* Grazing management: an ecological perspective. *Edited by* R.K. Heitschmidt and J.R. Stuth. Timber Press, Portland Oreg. pp. 141–159.

Tilman, D., Wedin, D., and Knops, J. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. Nature (London), **379**: 718–720.

Westoby, M., Walker, B.H., and Noy-Meir, I. 1989. Opportunistic management for rangelands not at equilibrium. J. Range Manage. **42**: 266–274.