LAND DEGRADATION AND GRAZING IN THE KALAHARI NEW ANALYSIS AND ALTERNATIVE PERSPECTIVES

Andrew Dougill and Jonathan Cox

INTRODUCTION

The introduction of domestic livestock to the Kalahari has been accompanied by significant changes in the rangeland ecology, reflected most obviously in a shift from herbaceous to woody plant species. The exact mechanisms behind this remain unclear, and it remains uncertain whether the vegetation changes are a direct response to increased grazing pressure, or whether they reflect more fundamental changes in ecosystem function.

Despite this uncertainty, it is common in the literature to see 'bush encroachment' used interchangeably with the more general terms 'land degradation' and 'desertification.' Whether or not this usage is always intentional, it is almost certainly misleading. Most recent definitions of land degradation (*e.g.* UNEP 1992) suggest that degradation is the result of irreversible changes, reflecting loss of ecosystem resilience. The use of forage quality as a single surrogate for land degradation is therefore questionable. On their own vegetation changes are poor indicators of damage to resilience (indeed many see the co-adaptation of vegetative communities and land use as evidence that resilience is intact), although in Botswana they are usually assumed to be *symptomatic* of loss of resilience.

In reality resilience is seldom at stake unless the vegetation changes are accompanied by soil degradation. In this paper we explore the links between grazing, ecological change and land degradation for a heavily grazed area in the Kalahari. In particular, we examine the relationship between ecological conditions and soil properties and processes. The results are striking. Data for soil nutrient dynamics and soil water redistribution show that basic soil processes are relatively unaffected by grazing pressure, with ungrazed and heavily grazed sites showing essentially similar characteristics. Certainly there is no evidence to suggest grazing leads to soil degradation, and as such links between cattle and 'land degradation' should treated with care.

ECOLOGICAL CHANGE *VS*. LAND DEGRADATION

The land degradation debate in Botswana has been in full swing for several decades, but has been especially intense since the droughts of the early 1980s (*e.g.* Cooke 1983; Ringrose *et al*. 1990; de Queiroz 1993). While we do not plan to go over old ground here, some of the principal features in the debate are worth noting.

First, the cattle industry has been an integral part of the land degradation debate. It has received most of the blame for the perceived degradation problem, but at the same time has most to lose, because declining primary productivity translates to declining secondary production (de Ridder and Breman 1993). Ecological studies have confirmed that the effects of grazing by domestic livestock on vegetation structure can be profound. Data presented by Fourie *et al*. (1987), Skarpe (1990) and Perkins (1991) for the Kalahari, and Tolsma *et al*. (1987), Dye and Spear (1982) and Bosch (1989) for elsewhere in southern Africa show that in recent years woody biomass productivity has increased at the expense of palatable grass species. With other factors remaining relatively unchanged in this period, these changes can only be explained by concomitant increases in grazing pressure (Perkins 1991).

With bush encroachment being obvious throughout large portions of east and central Botswana, it is unsurprising that some authors have painted a bleak picture of environmental conditions. The following is pretty representative:

> '*the evolving environmental conditions in Botswana are similar to those found in the Sahel twenty years ago and are consistent with trends in global desertification*'
>
> Nellis and Bussing (1989: 101)

However, while we can be pretty confident that increased grazing intensity has modified ecological conditions in the Kalahari, it is a philosophical leap to then extend the argument in the manner of Nellis and Bussing (and countless others) to include general statements about desertification and land degradation. We make this point for the following reasons.

First, changes in ecosystem structure cannot, in isolation, be used as reliable surrogates for the occurrence of land degradation. Their utility is restricted to situations where the only factor of interest is the yield of high quality forage for beef production. Most would agree that a sensible definition of land degradation would not be limited only to this criterion.

Second, we do not have sufficient information about the reversibility of the observed vegetation changes to judge whether these are permanent shifts in ecosystem characteristics. While 'desertification' and 'land degradation' have been defined in a number of ways over the years, most recent definitions suggest they apply only where (effectively) irreversible changes in the ecosystem have occurred. Warren and Agnew (1988) and others have argued convincingly that this is likely only where ecosystem resilience has been damaged. Historically, dryland ecosystems were considered fragile and such loss of resilience was seen as a very real threat. Recently, however this view has been questioned (*e.g.* Westoby *et al*. 1989; Behnke and Scoones 1993), and today the same ecosystems are considered inherently robust, and to have the ability to adapt readily to environmental instabilities (and especially to climatic fluctuations and fire). Changes in vegetation communities, whether induced by natural or land use factors, are evidence that the resilience of these ecosystems is still intact (Westoby *et al*. 1989; Abel and Blaikie 1989).

Using this argument, vegetation changes such as bush encroachment cannot be used as indicators of land degradation *per se*, as has been fashionable in Botswana. On their own they do not constitute land degradation, although they can be symptomatic. Resilience is seldom at stake unless grazing is accompanied by soil degradation, and the changes in vegetation communities that have been observed in the Kalahari are likely to be reversible provided soils have not been changed greatly (Abel and Blaikie 1989). Soil factors, and particularly soil water and nutrient availability, have been central to recent discussions on land degradation and land use sustainability (*e.g*. Parr *et al*. 1990; UNEP 1992; Stocking 1995). They are seen as particularly critical in dryland areas where, unlike vegetation, soils have relatively low resilience to degradation (Thomas 1993). Kalahari soils are a case in point. They are extremely infertile (Skarpe and Bergström 1986; Buckley *et al*. 1987) and contain negligible amounts of organic matter (Thomas and Perkins 1993b). In the light of this we could expect grazing to have significant effects on soil moisture and nutrient status, through the modification of organic inputs and outputs to and from the soil. In the following sections we assess the evidence for soil degradation in an area of rangeland that has been subject to intensive grazing for over 20 years.

Potential agents of soil degradation in the Kalahari

Soils are subject to six main types of degradational agent (Box 1). These are clearly interrelated, but in specific localities it is often possible to specify agents which are, or are not likely to be effective. In the case of the Kalahari, most of the agents listed in Box 1 can be discounted.

Lack of relief, together with high infiltration capacity and rapid hydraulic conductivity mean that water erosion in the Kalahari is extremely unlikely. Wind erosion is a greater hazard, although wind erosivity is generally low (Bhalotra 1987), and erosion is likely to be serious only in localised, stormy conditions. Thus potential erosion is high (typically 50 – 100 tonnes ha^{-1}), but actual erosion is likely to be far less significant (Cox 1994). Furthermore, susceptibility to erosion by wind or water is likely to be unaffected by vegetation changes that maintain a surface ground cover sufficient to limit erosion (Perkins and Thomas 1993b).

Because of extremely low sodium concentrations in Kalahari soils (typically 0 – 0.02 meq 100 g^{-1}), degradation in the form of salinisation or sodicity is unlikely. Likewise, structural problems are likely to be negligible. Structural instability has been observed in luvisols and planosols in eastern Botswana (*e.g*. Jones 1987), but clay contents in Kalahari soils (typically <2%) are generally insufficient to allow physical slumping, slaking or subsequent hardsetting. Given the low CECs and exchangeable sodium percentages associated with these soils, chemical dispersion is also unlikely (Cox 1994).

Box 1 Processes of Soil Degradation (after Stocking 1995).

Process	Characteristics
• Water erosion	• Splash, sheet and gully erosion. Mass movements
• Wind erosion	• Removal and deposition of soil by wind
• Excess of salts	• Salt accumulation in soil solution (salinisation) and of exchangeable sodium on soil colloids
• Chemical degradation	• Reduction in the concentrations of bases and essential nutrients, often through leaching Build up of base/nutrient concentrations to toxic levels
• Physical degradation	• Adverse changes in properties such as porosity, permeability, bulk density and structural stability
• Biological degradation	• Increase in the rate of mineralisation of humus without replenishment of organic matter

In the absence of major erosion hazard, the most likely agents of degradation in Kalahari soils are chemical degradation in the form of nutrient loss, or biological degradation in the form of organic matter loss. In this paper we focus on the influence of grazing pressure on these soil factors. In particular, and in view of the primary importance of soil water and nutrients to ecosystem structure and productivity (Frost *et al*. 1986), we concentrate on hydrochemical characteristics.

The two-layer model of the environmental change

Following the argument above, it can be assumed that if vegetation changes are symptomatic of soil degradation, they most likely reflect changes in soil water and nutrient availability. This is implicit in the 'two-layer model of environmental change' which has often been presented as the best explanation of bush encroachment in the Kalahari (*e.g.* Skarpe 1990a,b; 1991; Perkins and Thomas 1993a,b). In this model the balance between grass and bush production is determined by the relative availability of soil water and nutrients in different rooting zones (Walker *et al*. 1981). Savanna grasses out-compete bush species for water and nutrients in the topsoil layers, while woody species have the competitive advantage in the subsoil. According to the two-layer model, cattle grazing effects this balance by suppressing grass growth and promoting leaching to a greater depth. At the same time mineralisation of organic inputs at the soil surface is enhanced by inputs of organic matter as cattle dung (which is more readily decomposed than residual plant litter). This increase in mineralisation, especially of organic–N into nitrate, enhances the susceptibility of plant-available nutrients to leaching.

According to the two-layer model, areas of heavy grazing experience significant increases in extractable nutrient concentrations in their subsoil layers, with bush encroachment being the prime ecological symptom. If this can be proved, the implications are significant – as it would suggest that bush encroachment is more than just a superficial ecological trend, but is a manifestation of more fundamental soil changes, and as such *could* indicate land degradation(1). However, if soil water and nutrient characteristics in heavily grazed areas are *not* significantly different from those of ungrazed areas, it follows that the impact of grazing is restricted only to vegetation structure and productivity, with no evidence to support the presence of grazing-induced land degradation.

FIELDWORK

Field work was carried out between 1992 and 1994 at the Uwe Aboo Ranch in the Makoba TGLP Ranch Block C, eastern Kalahari (Figure 1). Environmental conditions at Uwe Aboo are typical of those found more generally in the Botswanan Kalahari in terms of grazing and vegetation characteristics. The ranch has undergone extensive fencing in recent years, separating the original 6 400 ha plot into a number of fenced paddocks and increasing the number of boreholes from one to three. Work reported here was restricted to an area to the north of the original

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(1) We emphasize the word *could* here because truly irreversible degradation would only be likely where the removal of grass cover has so affected hydraulic conductivity that the regeneration of the herbaceous layer is then unlikely.

borehole, which has been grazed continuously since 1973.

Vegetation and grazing

Data for vegetation structure at the Uwe Aboo site have been provided previously by Perkins (1991). Sampling along transects, he found a regular progression of vegetation types away from the borehole, which he associated with declining herbivore use intensity (HUI). HUI was greatest in the sacrifice zone adjacent to the borehole. This area contained low, degraded vegetation and extended out to about 100 m (Figure 2). Beyond this, vegetation communities were characterised by greatly increased bush density (in excess of 35 % canopy cover), and a decline in grass cover. Encroaching bush species were predominantly thorny impenetrable *Acacia* species, which gained dominance over the *Grewia* and *Lonchocarpus* bush species of the surrounding areas. This area of bush encroachment extended out to about 2 km from the borehole. Beyond, the range became increasingly dominated by low quality grass cover typical of ungrazed areas in the vicinity.

Observations in this study confirmed the earlier findings of Perkins and were consistent with data reported elsewhere (*e.g.* Andrew and Lange 1986; Tolsma *et al*. 1987; Georgiadis 1987). Indicators of HUI showed a decline in various impacts of grazing pressure with distance from the borehole (Figure 3). Dung and urine patch density increased significantly in the sacrifice zone but declined rapidly away from it, restricting the zone of significantly increased organic inputs to an area within 100 m of the borehole. These indices can be used as guides to the total time cattle spend at any one site. Cattle track density, which provides a better estimate of the time cattle spend directly grazing, was found to decline less rapidly with distance from the borehole (Figure 3).

Patterns of vegetation structure change (Figure 3) show that bush density declined along the same vector as these HUI indices. Correlations between various HUI indices and bush density yielded significantly high coefficients (*e.g.* bush density and dung density, $r = 0{\cdot}85$; cattle track frequency, $r = 0{\cdot}90$), suggesting that physical defoliating effects of grazing may be critically important in causing bush encroachment.

Soil Properties

Field studies were carried out to relate the changes in vegetation structure described above to soil moisture and nutrient characteristics measured *in situ*. Sampling was based on the piosphere approach (Georgiadis 1987), with soil being sampled at set distances along a transect (25, 100, 400, 800, 1 600 and 2 800 m) and also in the control area beyond the cordon fence (Figure 2). In each case samples were taken at 0, 0·2 and 1 m depths.

To assess the interaction between vegetation structure and soil properties, attention was focused on soil moisture characteristics, together with inorganic (*i.e*. plant available) nitrogen species and phosphorus fractions. Together, these have been shown to be linked most closely to semi-arid ecosystem structure and productivity (Scholes and Walker 1993). Total nitrogen and phosphorus were also quantified (although less frequently) so that estimates of the basic fractionation of N and P could be made.

Soil moisture characteristics

Comparisons of soil moisture content at different sites within the piosphere showed that there was no significant variation between sites in the control area and the bush-encroached zone. These results are summarised in Figure 4.

Figure 4 indicates a sharp increase in surface and subsurface moisture contents near the borehole, which could be explained by increasing levels of organic matter or (very likely) leakage at the watering point. Alternatively, greater wetting at depth could be encouraged by reduced vegetation cover in the sacrifice zone, as is predicted by the two-layer model of environmental change. Moving away from the borehole, moisture contents declined rapidly, but levelled-out after 50 – 100 m.

Importantly, Figure 4 suggests that there is no significant difference in surface or subsurface moisture levels between the bush-encroached zone and the control (confirmed by *t* test comparison; $\rho = 0{\cdot}267$). Similar comparisons using data for field capacity, bulk density, soil organic matter and hydraulic conductivity also indicated no significant differences in hydraulic properties between the two sites.

Process studies were carried out to examine more closely the nature of profile wetting in the Kalahari sediments. These consisted of small-scale experiments using water and methylene blue/potassium permanganate solutions to stain soil water flow pathways. Results from these experiments indicated that water movement was predominantly in the form of uniform matrix flow, with negligible bypassing flow to subsoil layers. Using this information, the wetting front model (Rose *et al*. 1982) predicts that profile wetting below 0·5 m will occur only after high-magnitude rainfall events (typically > 40 mm), and suggests that wetting is unlikely ever to exceed the active rooting depth of rangeland vegetation. Nutrient leaching is less effective in matrix flow conditions because the rate of water movement allows nutrients time to be adsorbed onto soil particles, and in this scenario it is unlikely that surface nutrients from litter, dung and urine will be transported to any great depth. Recharge losses beyond the rooting zone of herbaceous species should therefore be negligible. The implications of this for the two-layer model are discussed later in this paper.

Soil inorganic nitrogen

Variations in soil inorganic–N (ammonium nitrogen, NH_4^+–N and nitrate nitrogen,

NO_3–N) with distance from the borehole are shown in Figures 5 and 6. The wide confidence intervals in both graphs indicate the high variability in inorganic–N values at each sampling point. They indicate too the problem of interpreting inorganic–N results in nutrient-poor Kalahari soils.

Problems of accuracy aside, Figures 5 and 6 illustrate some important features in field patterns. Most noticeable is the lack of inorganic–N enrichment in the sacrifice zone. Statistical tests confirm that there is no significant difference between any of the heavily grazed sites and the control site for either NH_4^+–N or NO_3–N content ($\rho = 0{\cdot}956$ and $\rho = 0{\cdot}502$ in respective *t* tests).

On first inspection these results are surprising. Results from previous investigations have indicated that soil nitrogen levels are generally much higher in the sacrifice zone than in the grazed areas surrounding it (*e.g.* Tolsma *et al.* 1987; Georgiadis 1987). The simple explanation offered for this is that nitrogen in vegetation (*i.e.* potential litter) is removed from grazed areas and subsequently added to soils in the sacrifice zone through dung and urine. Cattle act as nutrient 'conduits' (Ruess 1987) through which there is net transport of nitrogen from outlying areas inwards towards the watering point. As these trends have been demonstrated clearly for total–N [(2)], we would not expect patterns for plant-available nitrogen to be any different. If anything, the differences between zones should be accentuated, as organic inputs in the sacrifice zone not only increase the supply of organic–N to the soil surface, but also enhance nitrogen mineralisation rates.

However, the simple pattern outlined above is not supported by data from this study for either NH_4^+–N or NO_3–N. The field patterns presented in Figures 5 and 6 provide no evidence to suggest either inorganic–N enrichment in the sacrifice zone or associated loss of inorganic–N in the bush-encroached area. To elucidate the reasons for this, process studies using mineralisation columns were carried out (Dougill 1995). These experiments showed that the absence of any systematic pattern in the field data could be explained by the extremely low nitrogen mineralisation rates which were common to all the sampling localities. Although increasing the supply of organic–N (to mimic dung additions) did enhance nitrogen mineralisation, resultant rates were still very low (mean $= 0{\cdot}179$ mg N 100 g^{-1} day^{-1}), and at these rates only a small portion of the total–N available in the soil can be fractionised into plant available nitrogen. This is illustrated diagrammatically in Figure 7.

The mineralisation results suggest that even if grazing leads to the loss of potential total–N inputs from the bush-encroached area (which are subsequently

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[(2)] This point is made on the basis of past findings (*e.g.* Tolsma *et al.* 1987). The limited data for total–N collected in this study also support this, although the small sample ($n = 24$) makes this interpretation tentative.

added to the sacrifice zone), the effect on the distribution of plant available nitrogen is insignificant because the supply of organic–N will always dwarf the amount of inorganic–N that can be produced at existing mineralisation rates. Nutrient availability to plants therefore appears unaffected even in areas of relatively heavy grazing.

From these data it would appear that mineralisation, rather than the total amount of nitrogen, is critical in limiting nitrogen availability to plants. Because plant uptake of nitrogen in semi-arid areas is so rapid, it is unlikely that under these conditions nutrient cycling will extend much beyond the topsoil layer, or that significant changes in nutrient profile distribution patterns will occur where there at least some plant growth remains. The latter point has been confirmed by leaching column experiments (Dougill 1995).

Soil inorganic phosphorus

Extractable PO_4^{3-}–P concentrations in soils sampled in the piosphere are summarised in Figure 8.

The most obvious feature in Figure 8 is marked PO_4^{3-}–P enrichment close to the borehole. This is particularly visible in the surface soil layer, and is largely restricted to the sacrifice zone within about 50 m of the borehole. Patterns in these data suggest a marked centripetal movement of phosphorus from grazed areas to soils adjacent to the watering point (it would appear that cattle are more effective conduits for phosphorus than nitrogen).

Intuitively, the removal of soil phosphorus from grazed areas constitutes a form of soil degradation, especially as once created, these patterns are more or less permanent.(3) However, a closer examination of the data shows that although there is a net gain of PO_4^{3-}–P in the sacrifice zone, this is not accompanied by net loss of PO_4^{3-}–P in the bush-encroached zone. Statistical tests confirm that, outside the sacrifice zone, the variation in PO_4^{3-}–P between sites is not significant.

This phenomenon can best be explained by looking at phosphorus mineralisation rates. As with total–N, the supply of total–P to the soils, either in the form of dung or plant litter, is massive in comparison to the portion that is made available to plants as PO_4^{3-}–P (average ratio $< 1{:}100$). Process-based studies carried out in the Kalahari (Dougill 1995) and Northern Transvaal (Scholes and Scholes 1989) suggest this is due to the equity and synchrony that exists between mineralisation and immobilisation/fixation rates, which produce negligible net P–mineralisation rates in the soil. Thus, as with nitrogen, variation in the total supply

(3) It has been suggested that localised patterns of P-enhancement in southern African savannas can be very long-lived, with present patterns being traced to Iron Age land use practices (Fordyce 1980; Blackmore *et al*. 1990).

of phosphorus does not translate directly into changes in the amount that is made available to plants. This explains why the movement of total–P from forage in grazed areas to dung in the sacrifice zone is not reflected in a decrease in PO_4^{3-}–P in heavily grazed areas. The reason why, on the other hand, PO_4^{3-}–P levels are so much higher in the sacrifice zone than elsewhere is that dung contains a much larger proportion of extractable–P than plant litter.

The other obvious feature of the graphs in Figure 8 is that surface levels of extractable PO_4^{3-}–P are higher than those in the subsoil ($\rho < 0{\cdot}001$ in *t* test), and this profile pattern remains unchanged between sites. The pattern reflects the relative resistance of inorganic–P to leaching and implies that it cannot be incorporated into explanations of bush encroachment in the same way as soluble inorganic–N. The consistency of this pattern between all the sites outside the sacrifice zone shows grazing has no effect on surface or subsurface extractable–P contents, and brings into question the applicability of the two-layer model of environmental change. The implications of this are discussed below.

IMPLICATIONS

Ecological change as an indicator of land degradation

It is now widely accepted that significant ecological changes have occurred in the Kalahari in response to increased grazing activity. But how we should interpret these changes remains debatable, and in particular the question of whether they constitute (or, more precisely, *represent*) land degradation is still somewhat confused.

Data presented here for Uwe Aboo Ranch provide new evidence with which to evaluate the arguments presented in the literature to date. The strength of the data is that they integrate both the symptoms of ecological changes (in terms of vegetation characteristics) and also the (proposed) causes of these changes. The implications of the field results are significant:

- There appears to be no significant difference between the hydraulic properties of soils in bush-encroached zone and the control site. In both cases the predominance of water transport as uniform matrix flow suggests the susceptibility of surface nutrients to leaching is limited and that their movement to depth is unlikely. This conclusion is at odds with that postulated previously (Skarpe 1990a,b; Perkins and Thomas 1993a,b).

- Results from nutrient profile measurements and mineralisation and leaching column experiments suggest that nitrogen and phosphorus cycling is rapid, efficient and restricted to the topsoil. Low rates of mineralisation prevent the build up of inorganic nutrients in Kalahari soils, and maintain typically low levels of soil fertility in grazed and ungrazed areas alike.

There is no evidence to suggest that grazing encourages the movement of soil water and nutrients to depth, as is implicit in the two layer model of environmental change. In fact, the lack of consistency between trends in ecological parameters (and specifically the grass/bush balance) makes any model that incorporates a causal link between vegetation structure and soil properties dubious. Moreover, the apparent inapplicability of the two-layer model, together with the HUI/vegetation data presented in Figure 3, add credence to alternative explanations of bush encroachment: *i.e.* herbivory impact and fire regime. In this respect, results from environmental surveys point to two main conclusions.

- Physical defoliating effects of cattle grazing on grass species appear critical in reducing the competitive dominance of grass species over bushes, leading to bush encroachment. Expansion of intensive grazing into areas beyond the bush-encroached zone, where grass cover remains higher even through drought events, can explain the continued expansion of bush encroachment. This expansion leads to concerns over the coalescence of bush-encroached zones to form large areas of bush-dominant unproductive rangeland.

- The decrease in the continuity of herbaceous biomass in grazed areas could reduce the extent and frequency of fires. Where fires do occur, their intensity will be diminished because of reduced fuel load. Past observations suggest that burning maintains the competitive dominance of grass species (Gillon 1983; Frost and Robinson 1987), and changes in fire regime are therefore regarded as an important contributory factor to bush encroachment. Its relative importance is uncertain however.

These observations suggest bush encroachment can be explained solely by the interactions between livestock pressure and abiotic environmental conditions, in particular rainfall variability and fire frequency. The significance of the applicability of the fire and herbivory explanations of bush encroachment is they suggest that vegetation changes are reversible. Herbivory impact is adjustable by varying husbandry practices. Drought events (Murray 1988; Perkins 1990) and fires (Gillon 1983; Harrington and Hodgkinson 1986) provide mechanisms that can cause bush die-back

Pastoral management in the Kalahari

As in other rangelands, pastoral management in the Kalahari should aim to maintain heterogeneity in fodder supplies (Scoones 1995). This can be provided by a mix of both grass-dominant and bush-dominant areas, which arguably the Kalahari range represents at the moment. The very real worry in the Kalahari remains that if the trend of borehole intensification continues at its recent pace (Tsimako 1991; White 1993) bush-encroached zones will coalesce to form large areas of bush-dominated, relatively unproductive rangeland (Perkins 1991). In this scenario, the relative importance of degraded sacrifice zones will also increase.

Studies describing the importance of physical defoliation effects of cattle grazing and natural environmental variability in leading to bush encroachment, as an inevitable consequence of the introduction of formalised cattle ranching, have a number of important implications. Specifically, they demonstrate the manner in which opportunistic strategies may be best applied to range management in the Kalahari. Prevention of the continued expansion of bush encroachment requires two main factors to be considered.

- The need to reduce physical defoliating effects of cattle on grasses during drought events. This could be achieved through provision of rapid reaction destocking programmes, or the movement of livestock to areas of adequate fodder supplies. The latter option would enable increased restocking rates in subsequent wet years.

- Limits must be imposed on the density of boreholes, if the convergence of bush-dominant areas is to be prevented, and therefore maintenance of a heterogeneity fodder landscape achieved.

Studies also tentatively imply that the role of fire could be critical in preventing the continued expansion of bush encroachment. Further studies are required to assess this link and to examine the possibilities for incorporation of prescribed burning practices to maintain grass-dominant areas and therefore allow the continuing intensification of pastoral utilisation of the area. This must be central to studies examining the possibility for sustainable development of Kalahari rangeland resources.

CONCLUSIONS

Blaikie and Brookfield (1987) made the point that degradation is a perceptual term, and its usage in Botswana is a case in point. The focus of the degradation debate is grazing and as a result forage quality has been used as the major indicator of degradation (*e.g.* van Vegten 1983; Ringrose *et al*. 1990; de Queiroz 1993). The use of such surrogates for land degradation is at best questionable at worst entirely misleading. They provide no indication of the condition of the land itself, or the productivity of the resultant biomass – rather that some commercial value has been lost with regard to the most commercially viable (but least adaptive) grazers. Like many definitions of degradation, this method of assessment does not incorporate adequately the concepts of recoverability or resilience. Previous studies have concentrated on expressions of ecological changes – whether in terms of vegetation changes (*e.g.* Skarpe 1990a,b) or livestock yield (White 1993). Only limited consideration has been given to the exact causes of these ecological changes.

Results from this study show that the over-used but under-researched association between grazing and land degradation in the Kalahari has been oversimplified. In typical Kalahari conditions, the ecological changes that have been brought about by grazing cannot be linked with more fundamental changes in ecosystem function. Basic soil processes appear relatively unaffected by grazing pressure outside the sacrifice zone, and there is no evidence to suggest that the resilience of the system has been affected through soil degradation. It should, however, be noted that this may not hold true for other semi-arid areas (including other parts of Botswana), where additional agents of soil degradation, such as soil erosion, come into play.

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