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Estimating the effects of land degradation and rainfall variation on productivity in rangelands: an approach using remote sensing and models of grazing and herbage dynamics

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Summary

1. Land degradation in non-equilibrium rangelands may be defined in terms of loss of resilience and is linked with lower economic productivity through reduction in forage consumption by stock.
2. Loss of resilience may be represented by lower water use efficiency and increased tree and shrub cover, in a simple herbage production and consumption model. The model may be calibrated from remotely sensed data.
3. Model calibration for rangeland areas in central Australia yields parameter values for degraded and undegraded situations, allowing estimation of productivity.
4. Modelling of a 50-year rainfall sequence shows that herbage production and consumption by cattle change through time because of rainfall variability. They also change with paddock layout and access to water.
5. The effect of degradation on herbage consumption is relatively small compared with the effects of rainfall variability, but it increases the chance of running out of forage during drought.

Key-words: grazing, herbage production, modelling, non-equilibrium rangelands, semi-arid zone.

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Introduction

In the International Convention on Desertification (United Nations 1994), land degradation is defined in terms of a reduction in biological or economic productivity arising from human activity. We therefore need to know what productivity is, how it might be measured and how the influence of human activity is to be separated from that of other factors when assessing land degradation.

Estimating productivity in rangelands is complex because, while biological productivity may be expressed in terms of plant production and the species composition of pasture, economic productivity depends, among other things, on the land use system, on how efficiently land managers use the resource and on external market factors. The effect of different management approaches therefore has to be filtered out when trying to assess whether a change in eco-

logical state affects economic production and equates with land degradation (Wilson & McLeod 1991).

Loss of biological and economic productivity is usually identified by comparing the current state of the soil and pastures with some kind of ideal or benchmark condition. However, explicit links with the level of animal production which is sustainable over a given time period are also required before a change in soil or vegetation can be treated as land degradation (e.g. Wilson & McLeod 1991).

Most arid and semi-arid rangelands show non-equilibrium behaviour whereby system structure and dynamics are dominated by abiotic controls, such as rainfall, which have a strong stochastic element (DeAngelis & Waterhouse 1987). Identifying soil and vegetation states which clearly constitute land degradation is particularly difficult in these systems (Pickup 1989; Friedel 1994). Short-term variations in rainfall and episodic changes due to other controlling variables, such as fire, mean that the level of animal production can vary substantially through time, except where the stocking rate is so low that animal require-

ments rarely exceed the supply of palatable herbage. The state of soil and vegetation may also vary, irrespective of the level of grazing by domestic animals (e.g. Pickup 1989; Friedel, Pickup & Nelson 1993). This background variation makes it difficult to identify grazing effects and to separate long-term impacts from those which are short term.

Given the importance of rapid recovery from disturbance in non-equilibrium rangeland ecosystems (e.g. Walker *et al.* 1981), a useful approach to identifying land degradation may be to express it in terms of some element of ecosystem resilience or ability of a system to recover after it has changed (Holling 1973). However, identifying the level of resilience is not easy when the only indicators available are measures of soil and vegetation state at a few points in time and space (e.g. Noy-Meir & Walker 1986). These measures have limited capacity to represent a spatially variable ecosystem subject to major fluctuations through time. Determining what truly constitutes degradation therefore remains a technical problem, even when the basic concepts such as ecosystem resilience and land use goals are clear.

This paper presents a method for determining how man-induced changes in the ecological state of a non-equilibrium rangeland affect biological productivity through changes in the supply and consumption of forage by grazing animals. It also examines the impact of natural variability in rainfall, and that of several different approaches to management on forage intake and supply. The study uses simulation techniques with a simple herbage production and consumption model for areas producing beef cattle in the rangelands of central Australia. The model can be calibrated from archived remotely sensed data and is therefore capable of operational use.

The link between biological and economic productivity is made by assuming that herbage consumption by grazing animals is an important element in the process of converting plant production to economic production. This approach has the advantage that herbage consumption can be estimated over large areas from remotely sensed data (e.g. McKeon *et al.* 1994; Pickup 1994). The effects of management are incorporated by modelling different stocking policies, expressed through differences in the amount and timing of herbage consumption, and examining how productivity changes with ecological state under each policy. The different policies are inferred from changes in herbage cover through time identified in remotely sensed data. They allow stocking levels to vary with forage availability and reflect observed management behaviour more closely than a single constant stocking level.

Variations in ecosystem resilience are dealt with in the model through changes in water-use efficiency which governs the extent to which rainfall is converted to plant matter. Short- and long-term changes in resilience are dealt with by modelling plant production as

a response to either fast or slow processes (Stafford Smith & Pickup 1993).

Fast processes result in changes in water use efficiency from one rainfall event to another, but are not persistent and disappear after a few significant rainfall events. They include increased vegetation response to rainfall when the previous rainfall event left a large seed bank of ephemerals, and short-term fluctuations in pasture species composition due to variations in temperature, time of year, etc. (e.g. Austin, Williams & Belbin 1981; Foran 1986). They may also include the extent to which defoliation allowed survival of perennial herbage or left enough foliage to ensure higher growth rates after the next storm (e.g. Bosch & Dudzinski 1984).

Slow processes cause shifts in water-use efficiency over sequences of rainfall events at the scale of years to decades. They include soil erosion and deposition, soil degradation and recovery, changes in the number of unpalatable trees and woody shrubs present, large-scale changes in the pattern of moisture redistribution in the landscape by run-off and run-on, and persistent shifts in the species composition of pasture (Stafford Smith & Pickup 1993).

In the model, fast processes result in a continuous feedback between grazing and water-use efficiency, and are accommodated by allowing herbage growth to vary with the extent of defoliation immediately before a rainfall event. They are therefore modelled dynamically during each run. Slow processes are represented by changes in the value of the water-use efficiency parameter itself and/or in the amount of tree and shrub cover present. They remain unchanged during a model run and have to be evaluated in separate runs with parameter values representing systems in different stages of degradation. These values are obtained from remotely sensed data for real systems using model inversion techniques (Pickup 1995.)

Forage supply and consumption models

The forage supply in arid rangelands consists of a series of short-lived, rainfall-generated plant growth pulses (e.g. Noy-Meir 1973). Forage consumption occurs mainly during the subsequent periods of cover decay as vegetation is removed by grazing and natural decline. Both supply and consumption are variable in space as well as time, so models of these processes should allow for differences in land condition and landscape heterogeneity. They must also operate at the scale of the management unit which, given the extensive nature of pastoral operations in arid Australia, means dealing with whole grazing units or paddocks. These are often 100–1000 km² in size. The model used in this paper (Fig. 1) has these properties. It operates with a time unit of 1 month to allow for short-term rainfall variability; but the version used here combines forage supply and consumption into 6-month totals to smooth out some of the short-term

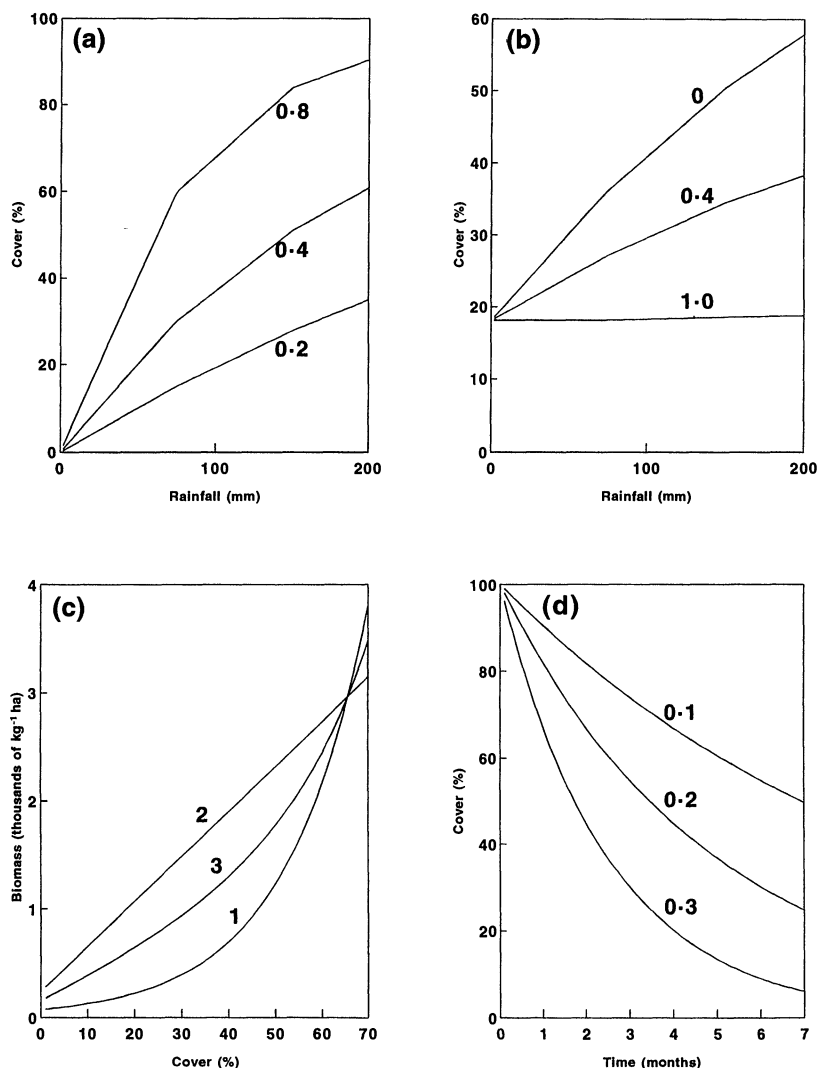


Fig. 1. Examples of functions used in the herbage production model. (a) Increase in vegetation cover assuming an initial level of zero for three different water-use efficiency values over a range of rainfalls. (b) Impact of three different fractions of tree and shrub cover on vegetation response assuming a water-use efficiency of 0.3, an initial cover of 18% and no herbage growing beneath the overstorey canopy. (c) Three cover–biomass relationships used, with 1 denoting the equation of Foran (1987), 2 indicating Bastin's relationship and 3 showing the combined relationships. (d) Cover decay over time for three different values of the exponential cover decay function k .

variability. A more detailed description, including the basic equations, is given by Pickup (1995).

The forage production model initially calculates total herbage production in terms of change in cover over time. The differentiation between herbage, not all of which is suitable for grazing, and forage is made in the consumption model. Modelling begins with rain entering a single soil moisture store from which it is gradually lost by evapotranspiration over time. Herbage production is calculated from actual evapotranspiration via a single water-use efficiency parameter, which varies with landscape type and condition, with herbage cover immediately before rain, and the amount of tree and shrub cover present. The size of a vegetation growth pulse may therefore differ over time, even though rainfall is the same. This is achieved by a combination of two processes in the model. First, as initial cover increases, potential cover

change is progressively reduced because competition for water from existing plants reduces the ability for new ones to establish and grow. Also, cover cannot, by definition, exceed 100%. Secondly, the use of a non-linear relationship between vegetation cover and biomass means that the increase in biomass for a given rainfall may be greater when growth starts from a higher initial herbage cover value. It also means that there is more plant production per unit of rainfall for larger falls. Taken together, these processes can produce a range of responses to rainfall. There is also a facility for using a linear cover–biomass relationship. This produces an increase in biomass which depends on rainfall magnitude, but not on the amount of herbage cover prior to the rainfall.

The cover–biomass relationships were both derived for vegetation types in or similar to those of the study area (Fig. 1c). The non-linear relationship (1) comes

from an area of calcareous soils dominated by ephemeral grasses and forbs, and was derived by Foran (1987). The linear relationship (2) was obtained by G.N. Bastin (unpublished data) for a run-on area occupied by perennial grasses, notably *Astrebla pectinata* (Lindley) F. Muell. While the linear pattern may be unusual it has been verified elsewhere for this community (Foran & Bastin 1982). Each may be used singly or they may be combined to produce an average relationship (3) for mixed communities.

The presence of trees and shrubs must be corrected for in the herbage cover calculations to allow for two factors. First, the overstorey canopy reduces the amount of herbage cover detected by remote sensing because it hides part of the understorey. Secondly, the presence of trees and shrubs may suppress understorey growth, although the extent of this effect may vary (e.g. Beale 1973; Scanlan & Burrows 1990). Given the linear relationship between overstorey and understorey cover found by Specht & Morgan (1981), herbage growth is reduced in direct proportion to overstorey canopy area. The resultant correction factor is specified by the model user and a value of 0.5 was used throughout this study.

The forage consumption model must allow for patterns of behaviour in both space and time which affect the ratio of vegetation cover depletion by grazing to that by natural decline. Two factors dominate behaviour through time: intake per animal and the number of animals present. Intake per animal increases up to an asymptotic value with both the amount of forage present and its greenness (e.g. Low & Wood 1979; Squires & Siebert 1983; Short 1987). The number of animals present varies with reproduction, mortality, turn-off and restocking, but can change fairly rapidly with forage availability as managers buy, sell or move animals between locations. For example, Fig. 2 shows how cattle numbers and rainfall varied in Kunoth paddock in central Australia during the period 1970–75 [see Low, Dudzinski & Müller (1981) for counting techniques and paddock details]. While numbers fluctuate, the pattern is dominated by peaks and troughs which reflect a lag in response to periods of major rainfall. The peaks occur because the increase in forage supply triggers breeding, while the troughs reflect sale of animals as forage becomes scarcer and as calves are shifted to other paddocks.

Accurate data on cattle numbers in the more extensively managed areas of Australian rangelands are not easy to get on a paddock-by-paddock basis, making it difficult to model complex patterns of change like those in Kunoth Paddock. There are also few data on forage consumption by grazing animals in large paddocks with mixed vegetation as opposed to small plots. However, extensive testing of the model with observed changes in vegetation cover derived from remotely sensed data suggests that, in many cases, an exponential cover-decay function provides a reasonable approximation of the combined effects of differ-

ing animal numbers, and of forage availability and greenness on forage consumption over the 6-month time increments used in the model (see Pickup 1995). This function is therefore employed here.

Spatial variability in forage consumption occurs because grazing in arid and semi-arid areas is centred on a limited number of natural and artificial water points to which animals must return at frequent intervals to drink. Forage consumption therefore tends to be higher close to water points, but as supply in these areas is progressively reduced, grazing shifts further out (e.g. Hodder & Low 1978; Squires 1982). This tendency is modelled by making the cover decay term a function of distance from water and results in a decrease in the ratio of consumption by grazing to loss of cover by natural decay away from the water points. Paddock layout then becomes a significant variable affecting production for, as the number of water points increases, the area within easy reach of them also increases. Improved accessibility within a paddock thus increases the total amount of forage consumed and may have a significant effect on animal production (e.g. Cridland & Stafford Smith 1993). It is allowed for in the model by a facility to change both the relative and absolute area within the paddock at each distance from water.

A further source of spatial variability in forage consumption occurs in non-uniform landscapes where there is a mixture of pasture types with different levels of palatability. More favoured areas experience higher grazing pressures and tend to be grazed out more rapidly than areas with less palatable forage, even when they are more distant from water. At the same time, as forage supply decreases in favoured areas, animals move onto other parts of the landscape (Low *et al.* 1981; Squires 1982). This pattern is dealt with by stratifying the landscape into different types on the basis of either classified remotely sensed data or published land-system maps, and using a functional relationship between cover decay and distance from water for each.

Not all herbage produced in a growth pulse is forage, so the cover decay term is made up of both forage consumption and loss of vegetation by natural decline. These processes are separated by using the rate of herbage cover decay over time at distances from water beyond the normal grazing range (6–8 km) to represent natural decay. Rates above this level in areas closer to water are then treated as the result of grazing (Pickup & Chewings 1988). This means that the function relating cover decay to distance from water contains a spatially variable element representing grazing and a spatially constant element representing natural decay. Both rates may vary with pasture species composition. Thus, the spatially variable rate tends to be higher in plant communities that are favoured for grazing than in those that are not. The spatially constant rate is also higher where ephemeral herbage dominates over perennials and reflects

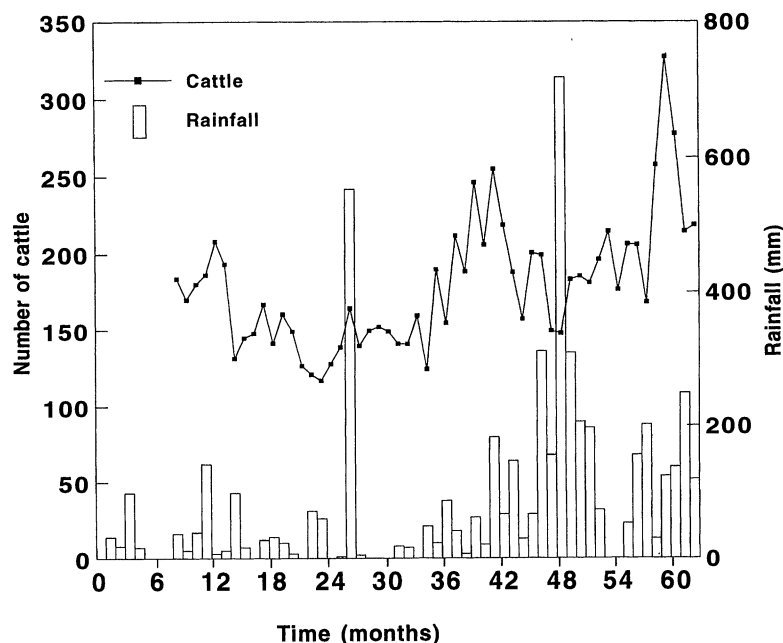


Fig. 2. Observed cattle numbers compared with monthly rainfall totals in Kunoth Paddock in central Australia over a 70-month period between 1969 and 1975. Data were obtained by aerial survey, so some of the short-term variability is due to counting errors. The main trend reflects real variations. (Data from W.A. Low: personal communication.)

more rapid loss of cover in those communities. The exponential cover decay model works well both close to and far away from water so it seems reasonable to assume that it applies to loss of cover due to both grazing and natural decay equally well.

The spatially variable element of the cover decay function includes the effects of trampling as well as grazing. Work in the USA suggests losses of 1–5% due to trampling by cattle at pasture utilization rates of 17–66% (Quinn & Hervey 1970). No data on trampling effects are available for central Australian rangelands so it has been ignored and trampling losses remain as part of consumption.

Animal production is related to the intake of total dry matter and to its nutritional value rather than directly to changes in herbage cover (e.g. Squires & Siebert 1983). The effect of differences in forage quality on animal production is, therefore, difficult to handle using remotely sensed data on vegetation production and consumption. However, it is known from observations that cattle select better quality forage where a choice is available and move from one vegetation community to another as pasture conditions deteriorate with time since the last growth-producing rainfall (Low *et al.* 1981; Squires & Siebert 1983). The cover decay functions, which are empirically derived and vary between landscape types, may therefore incorporate some effects of forage quality.

A second element determining forage quality is the amount of green leaf present (e.g. Squires & Siebert 1983; Wilson 1991). The use of an exponential function for cover decay with time may therefore incorporate some effects of forage quality since it results in greater reduction in cover immediately after a growth

event. Forage intake in the model is, therefore, at its highest when vegetation is green.

The model neglects browse. However, observations of cattle diet in central Australia indicate that browse is important for maintenance during dry periods (Chippendale 1968; Squires 1980; Squires & Siebert 1983), but most animal production comes from the consumption of herbage (Chippendale 1964; Siebert, Newman & Nelson 1968).

Estimating potential herbage consumption

Before the impact of land degradation can be assessed, it is necessary to understand how the influence of rainfall variability and of differences in stocking level, and in the spatial pattern of grazing are represented in the model. Four sets of trials were therefore carried out with the model (Table 1). The first two examined how the fast processes affect herbage production and consumption, and dealt with rainfall variability and changes in relative stocking rate. The third set examined how paddock layout modifies the effects of rainfall variability, since increased accessibility to water

Table 1. Specifications for model trials

Trial	Effect studied	Cover–biomass case	Paddocks
1	Rainfall variability	One	A
2	Stocking rate	One, two	A
3	Paddock layout	One, two	A, B
4	Land degradation	One, two	A

Table 2. Cover–biomass relationships used in modelling studies

Case	Slopes and rises	Aeolian sands	Run-on areas and sinks
One	Non-linear	Combined	Linear
Two	Non-linear	Non-linear	Non-linear

Table 3. Characteristics of paddocks used in the modelling studies

Paddock	Area (km ²)	Number of water points
A	576	9
B	183	2

changes the net effect of the fast processes assuming that the system remains stable. The fourth set of trials examined the impact of land degradation on herbage production and consumption, and compared it with the effects of rainfall variability and differences in paddock layout. Tests were also carried out to see how robust the model results might be, given different assumptions about cover–biomass relationships in the various scenarios examined (Table 2).

The study used two paddocks (Table 3) in the Mueller Land System of central Australia (Perry *et al.* 1962). Each paddock has several landscape types (see Pickup & Chewings 1994 for details) consisting of shrubby grassland on calcareous slopes and rises (low ridge systems), open woodland on aeolian sands, and more dense woodland in sinks (surface depressions which accumulate run-off) and watercourses. The under-

storey vegetation on the slopes and rises consists largely of ephemeral forbs and grasses, while the sinks and watercourses are characterized by often unpalatable perennial grasses. The aeolian sands have both ephemeral and perennial grasses and, like the sinks and watercourses, are heavily encroached upon by trees and shrubs in some areas.

THE EFFECT OF FAST PROCESSES

In the experiment on the impact of rainfall variability, the model was initially calibrated for the period 1982–89 using a set of sub-areas spaced at successive 1-km intervals from water for each landscape type in Paddock A (Fig. 3). Rainfall and evaporation data came from the Alice Springs and Owen Springs recording stations adjacent to the study area. Vegetation cover data for the calibration period were derived from Landsat Multispectral Scanner data using the PD54 vegetation index of Pickup, Chewings & Nelson (1993). The model provided a good fit to observed data and it is interesting to note how well the exponential cover decay function performed despite ignoring the observed time lag in response of animal numbers to rainfall.

After calibration, the model was run for a longer period using the calibrated parameter values, observed rainfalls for 1930–79 and mean monthly evaporation data. The stocking level and management approach were therefore assumed to be similar to those between 1982 and 1989. The 1930–79 period was selected because it includes both the driest and wettest years on record and shows rainfall variability typical of central Australia (Fig. 4a). Case One (Table 2) cover–biomass relationships were used.

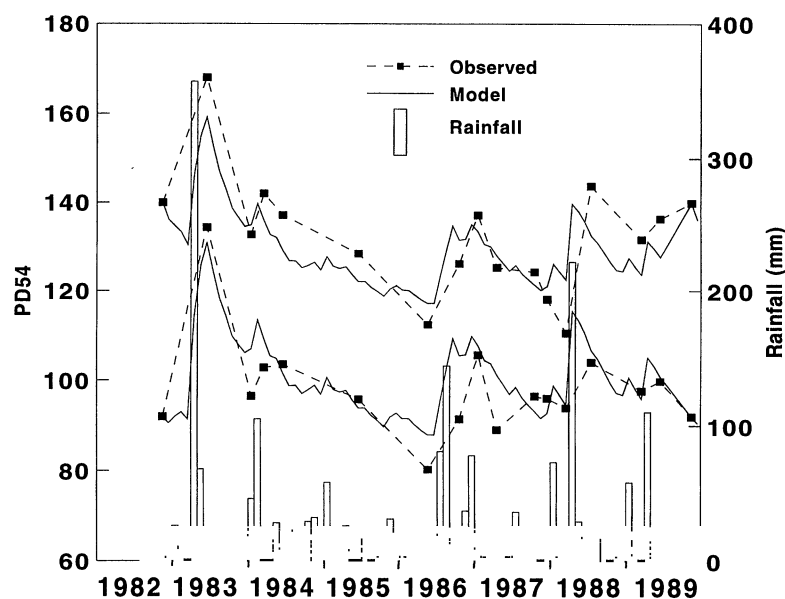


Fig. 3. Examples of results from a model calibration run. The observed and adjacent modelled data sets provide examples of the closeness of fit of the model for two different initial cover classes (calibration is actually carried out on 64 different initial cover classes). Cover is expressed in terms of the PD54 vegetation index (Pickup *et al.* 1993) which is derived from Landsat Multispectral Scanner data. The relationship between PD54 and percentage cover varies, but here the range of PD54 values on the ordinate is equivalent to 0–70% cover. Monthly rainfall totals are included for comparison.

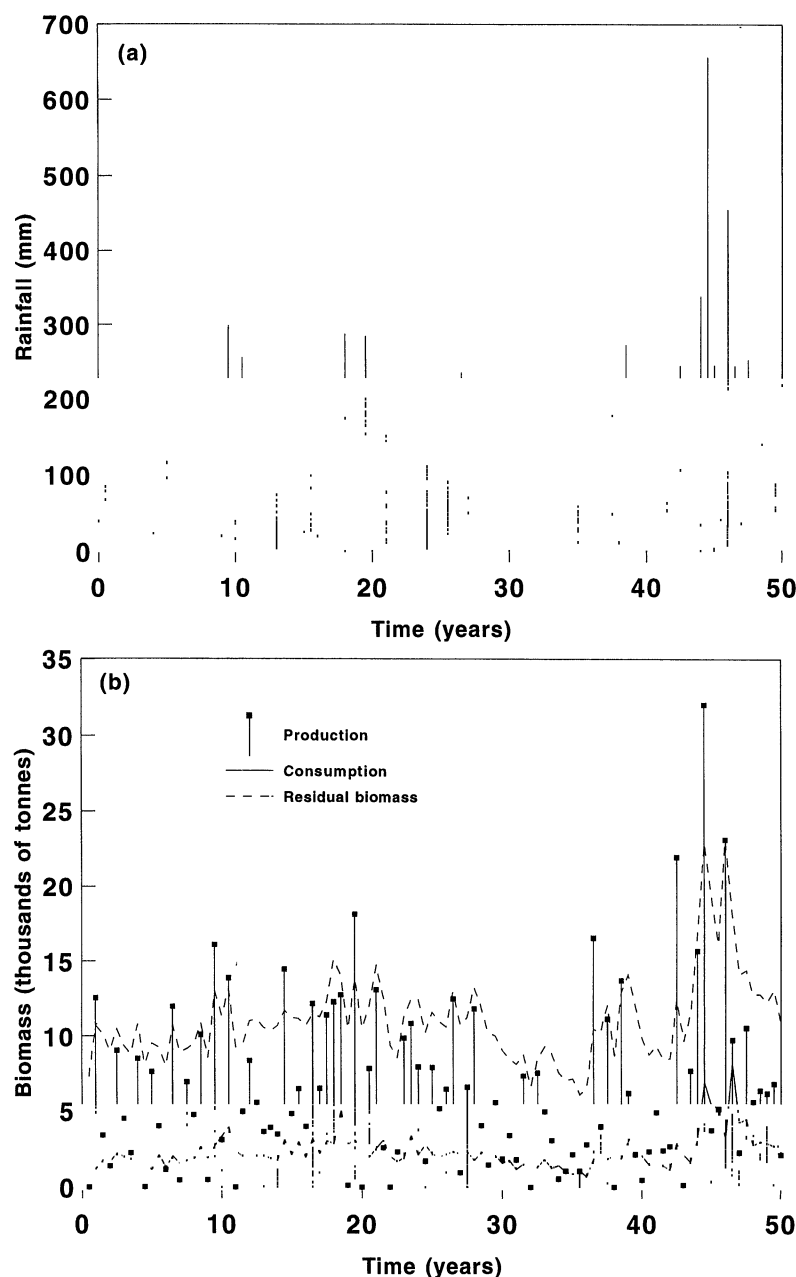


Fig. 4. (a) 6-monthly rainfall totals for the 50-year period from 1930 to 1979 used in modelling. (b) Modelled herbage production, consumption and residual biomass for a paddock of 576 km² for the same period.

When model results for the subareas were summed to provide results for the whole of the paddock for the 50-year period, they showed that herbage production, consumption and standing biomass are all highly variable and reflect the erratic rainfall (Fig. 4b). About 37% of total herbage production was consumed. However, the utilization rate was not uniform across the different landscape types in the paddock, with 75% of herbage on the calcareous slopes and rises, 49% of herbage in the sinks and only 28% of herbage on the aeolian sands being consumed. These differences reflect proximity to water, but also substantial variations in herbage palatability. Several dry periods are apparent when forage production, consumption and standing biomass declined substantially. There are

also major wet periods when these three variables rose well above average levels. Historical records show that major changes in stock numbers occurred at the district level during these times (Friedel, Foran & Stafford Smith 1990). It is therefore reasonable to assume similar changes at the paddock level. Above all, Fig. 4b shows just how much the system is dominated by rainfall.

The second set of trials with the model examined the effects of different stocking policies in an attempt to illustrate a range of management strategies. Stocking policies were changed by multiplying the spatially variable component of the cover depletion function by a factor representing the increase or decrease in stocking level (which is not necessarily constant

through time). Thus, twice the stocking level was represented by doubling the spatially variable cover depletion rate. The basic animal distribution was therefore maintained, but the number of animals at each distance from water changed proportionally as in the RANGEPAK paddock model (Cridland & Stafford Smith 1993). This procedure is supported by the results of Pickup & Chewings (1988) who found that the spatially variable component of cover changes over time, as measured on satellite imagery, is proportional to animal numbers present.

The effect of changes in relative stocking level on herbage production and consumption in Paddock A

are shown in Fig. 5a using Case One cover–biomass relationships (Table 2). The strong linear component in the cover–biomass relationship for the whole paddock means that modelled vegetation growth after rainfall is largely unaffected by prior herbage cover reduction due to grazing. Indeed, increasing the relative stocking level from 50 to 150% of the observed value only reduces herbage production by 0.27%. At the same time, total consumption rises by 114%, while average consumption per animal declines by only 29%. Higher stocking levels within this range therefore result in greater use of available forage with a less-than-proportional decline in intake per animal.

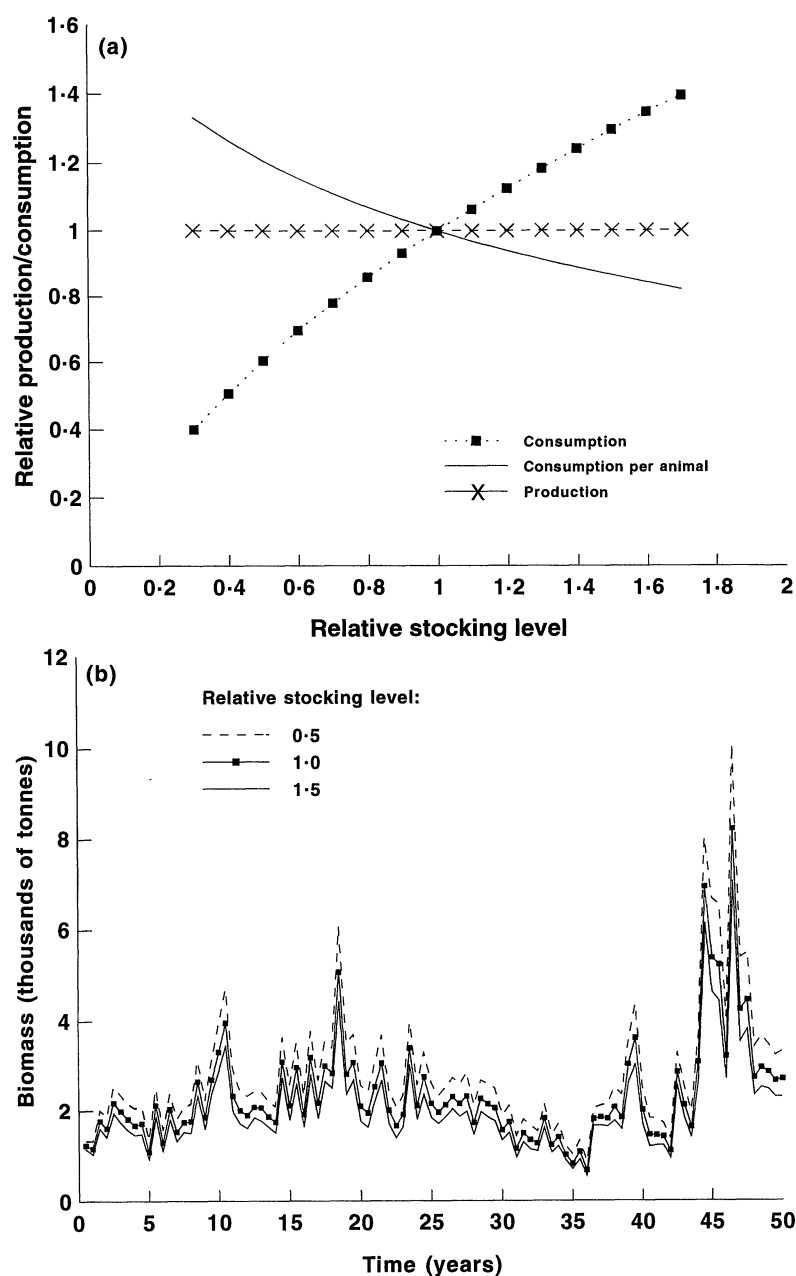


Fig. 5. (a) Effect of relative stocking level on total herbage biomass production, total consumption and consumption per animal, expressed as a fraction of the value at the observed stocking level. (b) Pattern of herbage biomass consumption from a paddock of 576 km² over the 50-year period from 1930, at 0.5, 1.0 and 1.5 of the observed stocking level.

However, only the fast processes have been modelled so there is no consideration of whether such stocking levels could be maintained over time.

Using Case Two (Table 2) cover–biomass relationships for the whole paddock gives a somewhat different result. More substantial losses in herbage production occur and a change from 50 to 150% of the observed stocking level results in 11% less herbage biomass over the 50-year period. Herbage consumption is also affected, with total consumption increasing by only 78% and intake per animal declining by 41%. Thus, while the basic pattern remains the same, the impact of changing stocking levels is greater in systems where pre-rainfall cover levels affect subsequent herbage growth rates.

While differences in stocking level affect total herbage consumption, they also change the extent to which a grazing operation is vulnerable to drought. This may, indeed, be the more significant effect where rainfall variability is high. Vulnerability to drought is best seen when relative consumption per animal is plotted through time (see Fig. 5b for Case One cover–biomass relationships; results for Case Two relationships show a similar pattern). The 0.5, 1.0 and 1.5 relative stocking rate curves are clearly separated, and there are long periods at the high stocking rate when consumption is very low and possibly not enough to sustain grazing. In contrast, herbage intake per animal at the low stocking rate is much higher than that at the 1.0 and 1.5 relative stocking rates for virtually all of the 50-year period and the three consumption curves only come close to converging (i.e. when forage runs out completely) at year 36. This was a period of catastrophic drought for the grazing industry in central Australia and animal numbers were greatly reduced throughout the region (Friedel *et al.* 1990). Lower stock numbers therefore mean less vulnerability to drought and a less frequent need to de-stock.

Changing the cover depletion rate in the model affects the amount of forage consumed, the ratio of consumption to natural decay and the size of the fodder reserve at any given time. It also assumes that, while there may be proportional differences in the number of animals present at any given time, the broad pattern of change in numbers over time remains essentially the same. Thus, the pattern of change in situations with low stocking is a highly damped and perhaps slightly lagging version of that under much higher stocking. This assumption may not always hold because fertility, mortality, pressure to de-stock and opportunities to turn off animals may be somewhat different as stocking policies change. The results derived here can therefore only be regarded as a first approximation.

Differences in paddock layout affect productivity through differences in accessibility to water and the distance animals have to walk to preferred forage. Trials dealing with the effect of paddock layout on herbage production and consumption were therefore

carried out by changing the fraction of total area in each landscape type at each distance from water within a paddock. This approach makes it possible to compare results between paddocks with different layouts or to standardize results from many paddocks to a single layout.

The impact of changing paddock layout was examined by calibrating the model for Paddock B for 1982–89 and then running it over the 50-year period. Initially, the calculations were based on the true paddock layout, but then a run was made with the same pattern of accessibility to water as that in Paddock A. The differences in layout are illustrated by the relative areas at each distance from water in the two paddocks (Fig. 6) and show that Paddock A has a larger fraction of total area closer to water than Paddock B. Using the Case One cover–biomass relationships, this improved access to water results in an 11% increase in herbage consumption, but only a 1% loss in herbage production. With the Case Two cover–biomass relationship, the increase in consumption is 13% with a loss of 4% in herbage production. The difference between the two paddock layouts arises because improved accessibility to water means that favoured areas are eaten out more quickly so loss of herbage due to natural decay is reduced. This, of course, assumes that reducing the amount of natural decay has no impact on future growth rates, which may not be the case. The results also suggest that layout can have a significant effect on animal production, so comparisons of the impact of degradation between pad-

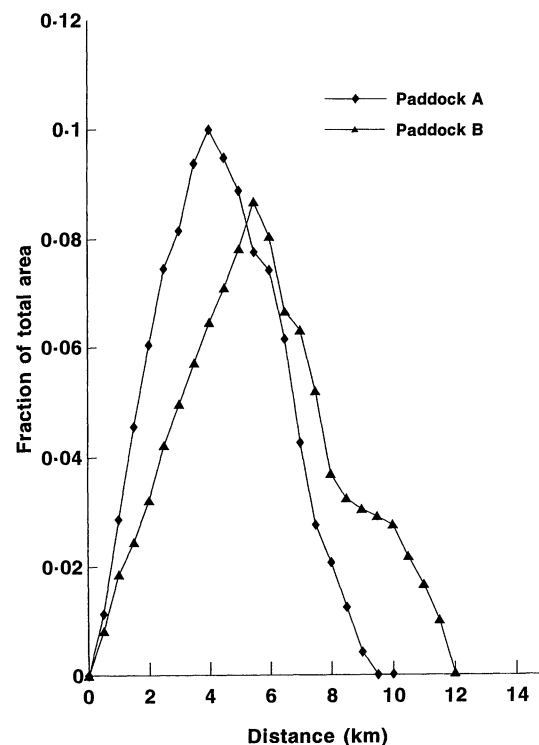


Fig. 6. Fraction of total area at the stated distance from water for Paddocks A and B.

docks based on vegetation characteristics alone must take it into account.

ESTIMATING THE IMPACT OF LAND DEGRADATION

Land degradation occurs through changes in the slow processes which reduce water-use efficiency and the amount of tree and shrub cover. It does not occur uniformly, but may be more intense closer to water and is usually concentrated in particular landscape types (e.g. Pickup & Chewings 1994). These patterns make it possible to estimate herbage production and consumption with and without degradation. Thus, water-use efficiency and tree and shrub cover may change systematically with increasing distance from water in large paddocks as grazing impact declines, until a point is reached beyond which they become relatively constant or vary without pattern (see Pickup 1995). Model parameter values obtained beyond this point represent relatively undegraded conditions and can be applied to the whole system to reconstruct its behaviour before degradation occurred. This approach is also the basis of the grazing-gradient method of assessing land degradation (Pickup, Bastin & Chewings 1994).

Potential cumulative herbage biomass production and consumption have been estimated in this way for Paddock A using Case One cover–biomass relationships and the 1930–79 rainfall sequence. Degradation produces a 4% reduction in herbage production, but has a greater effect on consumption, reducing it by 9%. The Case Two cover–biomass relationship shows a more extreme result, reducing herbage production by 6% and consumption by 13%. The reduction in consumption may be interpreted as the result of animals spending more time searching for forage where it is sparse or unpalatable, and walking to and from grazing areas further out from water.

Paddock averages can be misleading because the effect of degradation varies between landscape types. On the calcareous slopes and rises, herbage biomass production and consumption decline by 34% and 36%, respectively, for both the Case One and Case Two cover–biomass relationships. On the aeolian sands, the losses are 6 and 5% (Case One) or 9 and 8% (Case Two). In the sinks, production and consumption decline by 3 and 1% (Case One) or 11 and 5% (Case Two). The grazing impact is therefore concentrated on the slopes and rises which, even in a relatively degraded state, are still favoured by cattle. The loss of forage from this favoured landscape is partly compensated for by a shift to the sinks where a less-than-proportional reduction in consumption occurs as herbage production becomes lower.

The modelling results suggest that the impact of degradation on herbage production and consumption through time is relatively limited when compared with the natural variability associated with rainfall. For

example, Fig. 7a and 7b show herbage production and consumption in the degraded and undegraded state for the calcareous slopes and rises which are the most degraded landscape type of the three in Paddock A (Pickup & Chewings 1994). Two- to threefold variations within a 3-year period are the norm and changes can be as great as an order of magnitude when rainfall variability is particularly high. Land degradation reduces both herbage supply and consumption over the long term, but has relatively little effect on their short-term variability. Its impact in the short term is also damped by the fact that degradation effects are greatest around water points and, in large paddocks, can be partially offset as animals graze further out from water more quickly and use up herbage which would be otherwise lost by natural decay. There is also a tendency for animals to switch to areas with less palatable vegetation more quickly than before as areas with highly palatable vegetation become degraded and the forage supply is reduced.

Perhaps the greatest impact of degradation that is shown by the modelling studies is on susceptibility to drought. To illustrate this, an arbitrary threshold has been set in which it is assumed that, once consumption falls below 33% of the mean rate for the 50-year period under degraded conditions, the operation comes under stress and major de-stocking is necessary. In the degraded situation, this occurs five times, covering one 2-year period and four 6-month periods. In the undegraded situation, de-stocking is only necessary twice, covering one 18-month period and one 6-month period. Raising the threshold to 50% of the mean consumption rate for the 50-year period produces an even more extreme result. De-stocking is necessary for five 6-month periods, two 2-year periods, and one 3-year period in the degraded situation. In the undegraded situation, the threshold is crossed for only four 6-month periods and one 2-year period.

These conclusions are in keeping with experimental results obtained by Ash *et al.* (in press) from north Australian tropical grasslands, which suggest that reduced herbage growth under degraded conditions has a major effect on both pasture and animal production during drought. However, when the system is not in drought and forage supply is not a limiting factor, animal production may be similar to or higher than that from land which is in better condition. Similar conclusions were also reached by Holm (1994) in a major grazing trial in the sheep rangelands of Western Australia.

The economic impact of increased susceptibility to drought can be severe, given the substantial costs of each destocking and restocking operation. There is also the ongoing impact of having to reduce animal numbers more frequently when herd growth is mainly by natural increase. This may affect production for years to come. An indication of the level of severity can be obtained from modelling studies of drought management strategies for central Australian cattle

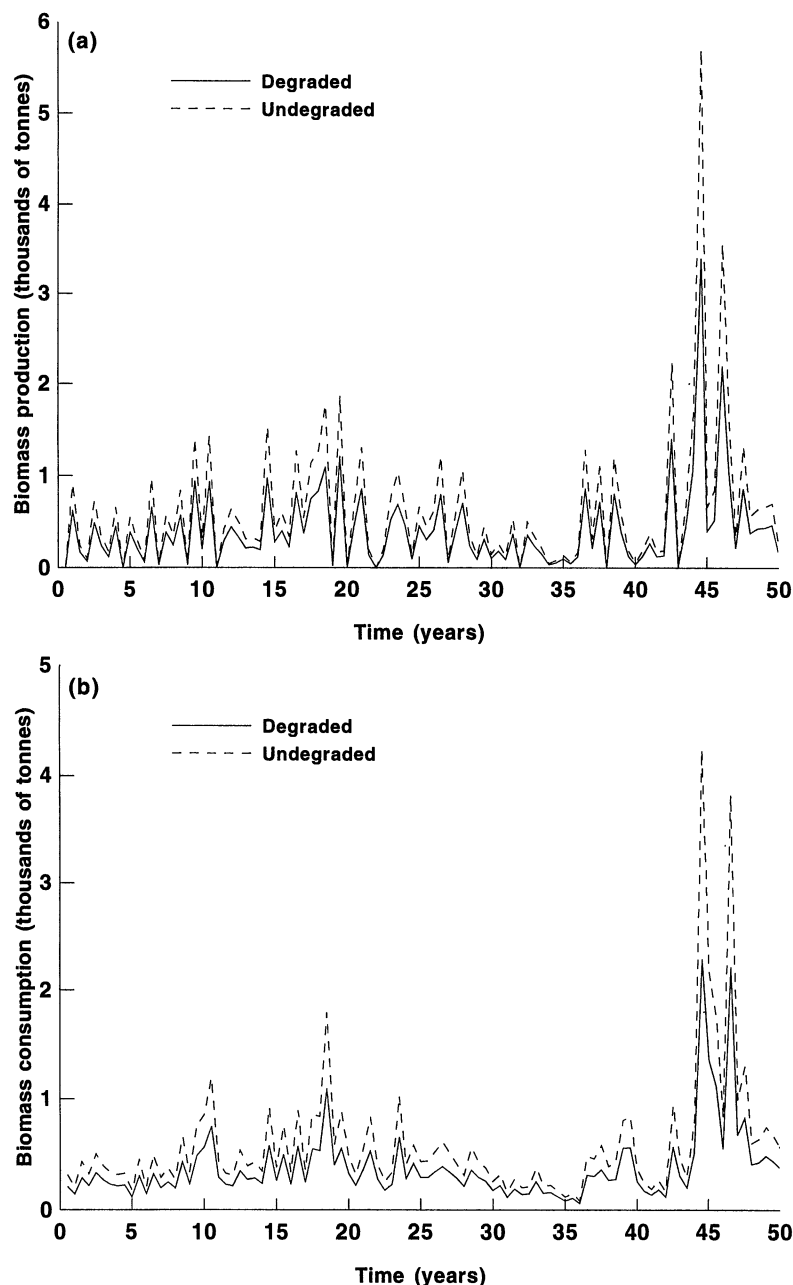


Fig. 7. (a) Herbage biomass production and (b) consumption through time for the calcareous slopes and rises in Paddock A in their current degraded state and in an undegraded state.

stations by Foran & Stafford Smith (1991). If it is assumed that land degradation increases the number of years with 'drought' or fodder deficiency, in a given climatic sequence, from 2 to 5, as in the above circumstances, the following consequences result. Accumulated cash surpluses at the end of a 20-year modelling period under heavy, average and low stocking, respectively, fall from \$A5.11 million to \$A1.87 million, from \$A3.57 million to \$A1.31 million, and from \$A2.6 million to \$A2.05 million. Years of negative accumulated cash surplus also increase from 3 to 9 and from 1 to 8 for the high and average stocking

regimes, but there are no negative years for the low stocking situation.

Discussion

The model results allow hypotheses to be tested, but only reflect real situations when the full range of processes linking degradation and animal production are allowed for. Here, three possibly substantial effects on animal production have been either ignored or only partially accounted for. These are as follows.

1. The effect of land degradation on forage quality.

2. The linkage between animal numbers and forage availability as influenced both by management and by changing fertility and mortality. This includes the lag between periods of high rainfall and peak animal numbers which may undermine the exponential decay model.
3. The impact of grazing on the slow processes which could reduce both forage production and consumption in the long term.

The effect of ignoring differences in forage quality may not be great, for there is a growing body of literature which suggests that, in non-equilibrium systems, animal production is more closely related to total herbage production than to the species composition of pasture (e.g. Ellis & Swift 1988; Mentis *et al.* 1989; Wilson 1991; Hodgkinson 1992). Thus, while herbage production decreases as pasture condition declines (e.g. McIvor, Ash & Cook 1995) forage or diet quality remains the same or improves (Graetz 1986; Danckwerts 1989; O'Reagain & Mentis 1990; Ash & McIvor 1995). Some of these studies, however, have been restricted to analyses of the nutritional value of pasture and have not allowed for changes in the palatability of the species mixture which affects intake. It may therefore be that the impact of changes in species composition on animal production operates more through changes in palatability than through other factors. If so, this effect is allowed for in the model by differences in the cover decay rate between landscape types.

The exponential cover decay function used in the model implies that total forage intake decreases with available herbage, irrespective of the time-lagging increase in animal numbers after major rainfalls. While this may be counter-intuitive, the exponential model gives a good fit to observed data generally and has been tried out in many paddocks. However, there is some evidence that it under-estimates cover change at the onset of drought conditions. For example, Fig. 3 shows that in the dry period of late 1985 and early 1986, modelled cover was significantly greater than that observed from the remotely sensed data. However, this underprediction of cover change could also result from errors in model rainfall input. The small storms which caused the limited increase in modelled cover in late 1985 could have been very local and may not have allowed growth across the whole paddock. Thus, there are no firm grounds for rejecting the exponential cover decay approach.

If the exponential cover decay assumption holds and the major increases in animal numbers lag behind the major rainfalls, it follows that forage consumption per animal may decline faster than total forage consumption as subsequent droughts approach. This makes it difficult to establish a close linkage between modelled forage intake and animal production. There is also the problem that forage digestibility declines as it becomes drier, so animals lose condition even if intake per animal remains unchanged over time (e.g. Wilson 1991; Coates, Ash & McLean 1993).

At present the model neglects the slow processes and is limited to modelling the behaviour of a landscape whose state does not change through time. Indeed, predicting the impact of different grazing intensities on land condition remains a largely unsolved problem in non-equilibrium rangelands. It is, however, possible to speculate on the extent to which changes in land condition over the 50-year modelling period might affect forage intake if they were built into the model. Paddock A represents one of the more severe cases of land degradation in the Alice Springs District (Bastin *et al.* 1993) and has been grazed for about a century, although many of the water points were established in the last 50 years. If it is assumed that the degradation occurred gradually over the 50-year modelling period, the reduction in forage production and consumption would be approximately half that modelled in the degraded case.

Conclusions

While land degradation may be defined in terms of loss of biological and economic productivity, these consequences are difficult to measure in practice. This study has shown that simple models of herbage production and consumption can estimate the effects of degradation on some aspects of forage production and consumption which are elements of economic productivity. The models can be calibrated from widely available remotely sensed data and therefore have potential for wider use. They also have limitations, including the need for empirically derived cover–biomass relationships, estimates of the impact of tree and shrub cover on herbage growth, and limited ability to deal with the effects of variations in forage quality and animal numbers.

Modelled patterns of the production and consumption of vegetation suggest that, for much of the time, there is considerable slack in the system in the form of standing herbage biomass which is otherwise lost by natural decay. **Grazing is therefore relatively insensitive to the effects of land degradation until forage is in short supply. Land degradation comes to the fore during drought because forage runs out sooner and more frequently in degraded systems. However, this effect can partly be offset by better paddock layouts and by lower stocking rates. Another factor reducing sensitivity to land degradation relative to the effect of rainfall variability is the timing of the rainfall events themselves. If a grazing operation which does not have substantial financial or forage reserves encounters a sequence of poor years, it can fail before land degradation effects have any impact.** At the same time, if that operation experiences a sequence of good years, the financial gains made may be more than enough to offset lower forage production and consumption due to poor land condition well into the future.

The lack of sensitivity to land degradation (except in times of drought) probably becomes greater as rainfall variability increases and the forage supply becomes more volatile. Increased volatility means that forage will be in surplus for more time, since consumption cannot adjust quickly to forage production unless there is very active buying and selling or movement of animals. By contrast, systems in which rainfall variability is relatively low should feel the impact of degradation on animal production more continuously. Nevertheless, it is likely that the impact of increased susceptibility to drought is greater in areas where rainfall variability is high. Under these circumstances, the effects of degradation on animal production may be very substantial, but they are likely to be blamed on naturally occurring periods of low rainfall rather than on land condition.

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