Monitoring and modelling open savannas using multisource information: analyses of Kalahari studies

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ABSTRACT

In the Kalahari of Botswana, as in many other open savannas, the main ecological change following cattle-based agricultural intensification is one of grass removal and bush encroachment. Changes to vegetation communities in Kalahari rangelands have been expressed in terms of a state-and-transition model. However, there remain uncertainties as to the mechanisms and conditions for ecological change. In part, this is due to previous (inadequate) spatial and temporal scales of data collection. This paper describes the results of ongoing ecological studies in the Makoba ranches and analyses the contribution that fine-scale ground-based surveys and interpretation of Earth observation data can make to reducing uncertainties in state-and-transition models. This detailed case study provides a mechanism for evaluating the role of multi-source information for monitoring and modeling open savannas.

Key words. Bush encroachment, environmental monitoring, open savanna, Kalahari, remote-sensing, spatial patterns, resilience, state and transition models.

INTRODUCTION: BUSH ENCROACHMENT ON SAVANNA RANGELANDS

Intensification of domestic livestock production in the Kalahari of Botswana in the last 30 years has been associated with significant ecological changes, notably the shift in vegetation dominance from herbaceous to woody plant species (Ringrose et al., 1990; Skarpe, 1990; Perkins & Thomas, 1993a,b; Ringrose et al., 1995). This process, termed bush encroachment, has been experienced similarly in open savannas across the globe, for example, elsewhere in southern Africa (Bosch, 1989), Sahelian Africa (Warren & Agnew, 1988), North America and Mexico (Archer, 1990), Australia (Andrew, 1988) and South America (Medina & Silva, 1990). Indeed, land degradation studies highlight that for many savannas the most widespread problem affecting sustainable agricultural production remains invasion by thorn scrub (Warren & Agnew, 1988; Scoones, 1995), and not the more widely cited examples of complete vegetation removal that have perpetuated myths about the extent and nature of dryland degradation (Thomas & Middleton, 1994; UNEP, 1997).

Studies reported here have a dual focus. Firstly, we review current scientific understanding on the extent and causes of bush encroachment in the Kalahari. Secondly, we provide a detailed case study of ecological research at the Makoba ranches of the Central District, Botswana (21°59’S, 25°39’E), in order to demonstrate how data collected by fine-scale ecological survey and satellite remote sensing studies are being used jointly to develop an ecological state-and-transition model. This conceptual model summarizes ecological understanding of the processes leading to changes in vegetation community structure within the dynamic Kalahari environment. In addition, the model highlights remaining uncertainties caused by complex interactions of grazing intensities, rainfall variability and fire regimes. The application of multiple studies at a single site provides the basis for a meaningful comparison of different data sources and permits an evaluation of the role of multisource information for monitoring and modelling open savannas.

Previous research in the Kalahari of Botswana has described the nature of ecological changes occurring since the widespread adoption of intensive livestock ranching, as encouraged by the national Tribal Grazing Lands Policy (TGLP) of the 1970s. These studies have demonstrated that bush encroachment generally remains confined to an area within 2 km of the boreholes on which cattle production is centred (Perkins & Thomas, 1993a,b). Given the initial adoption of an ‘eight kilometre rule’ for borehole spacing within the TGLP, there remained extensive grass-dominant areas unaffected by bush encroachment (termed ‘grazing reserve’ by Perkins & Thomas, 1993a,b). The bush-encroached zone – grazing reserve duality which characterizes TGLP ranch blocks, such as those in the Makoba ranch block, maintains vegetation community diversity on ranch blocks by providing both bush- and grass-dominant areas. This community diversity helps to explain the minimal impacts on overall agricultural productivity noted on TGLP ranch schemes to date (Vossen, 1990; White, 1993). Recent ecological and socio-economic studies in many savanna settings have shown that such mixed bush and grass communities provide the greatest flexibility for land managers in maximising agricultural outputs through matching the inherent spatial and temporal dynamism of dryland ecosystems (Behnke et al., 1993; Scoones, 1995). Possible land degradation concerns still exist in the Kalahari because, as Perkins & Thomas (1993b) note, bush encroachment is an expansive process with its greatest spatial extent occurring around older boreholes. Given this observation, and development pressures leading to a relaxation of the 8 km spacing between boreholes (White, 1993), there is potential for the coalescence of bush-dominated areas, and a real concern that this will reduce future livestock yields.

Thus far, the progress in quantifying the extent of bush encroachment has not been paralleled by equivalent advances in our understanding of the mechanisms and processes operating to cause bush encroachment. Analysis of soil hydrochemical factors have shown that contrary to a number of early assertions (e.g. Skarpe, 1990; Perkins & Thomas, 1993a) ecological change has not been caused by, or associated with, significant changes in profile patterns of soil water and nutrient availability (Dougill et al., 1997, 1998a). These conclusions have been combined with ground-based observations of patterns of ecological change by Dougill et al. (in press), to provide a conceptual ‘state-and-transition’ model (Fig. 1) that summarizes current understanding of factors causing changes in vegetation community structure. This model highlights several remaining uncertainties (x1 – x4 on Fig. 1), chiefly regarding the combined effects of biotic pressures (grazing) and natural abiotic variability (fire and rainfall regimes) in leading to the transitions between vegetation community states. At present, these uncertainties limit model precision and inhibit its use to support changes in management strategies. Studies reported here attempt to reduce ecological uncertainties by using ground-based survey and remote-sensing methods to assess combinations of biotic and abiotic factors that cause transitions between different vegetation community ‘states’.

Ranch scale studies in the eastern Kalahari (e.g. Perkins & Thomas, 1993a,b) have identified ecological changes and current ecological heterogeneity at this scale. However, such studies are unable to explain fully the processes operating to cause the patterns of change noted (Fig. 1). Uncertainty remains because the factors causing ecological transitions operate at different scales to these ranch-scale analyses and due to interactions between these processes. Biotic effects of increased herbivore use (i.e. cattle grazing) operate at the fine scale of impacts on individual plants (Mott et al., 1992), which then combine to influence wider-scale vegetation communities. Conversely, rainfall varies at a broader regional scale, transcending ranch-scale patterns of grazing pressure. Similarly, changes to fire regimes are regulated by a combination of regional driving forces, notably rainfall and lightning occurrence, and fine-scale grazing-induced changes to vegetation communities which control the herbaceous fuel load (Frost et al., 1987). This complexity highlights the interaction of factors operating at a range of scales in determining vegetation community structure and productivity. Consequently, studies at multiple scales are required to extend understanding of key processes influencing the extent and permanence of bush encroachment.

**Ground-based ecological survey**

Recent ecological debates (e.g. O’Connor, 1991, 1994; Scholes & Walker, 1993; Belsky, 1994; Jeltsch et al.,
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Fig. 1. Ecological state-and-transition model for Kalahari open savannas (modified from Dougill et al., in press).

1996, 1997) have switched attention from the necessity for long-term ecological monitoring, to the additional information available from fine-scale spatial pattern studies of bush and grass communities. Patterns of bush and grass cover are determined by grazing, fire and rainfall regimes, in addition to competition for soil water and nutrient resources. Consequently, it is recognized that differences in fine-scale patterns of bush and grass cover could yield information on the role of changes in such ecological determinants in causing bush encroachment. For example, Jeltsch et al. (1996) conclude that increased grazing alone will lead to increased bush cover distributed evenly through an area. However, where bush encroachment is reversed by occasional fires, bushes will be clumped into patches creating thickets that exclude fire (Skarpe, 1991; Jeltsch et al., 1996). Similarly, distinct patterns of grass cover have been linked to changes in grazing pressures and interannual rainfall variability (O’Connor, 1994), in addition to competition with shrubs for light, water and nutrients (Amundson et al., 1995). Consequently, recent ecological investigations at Makoba ranch block have focused on fine-scale patterns of bush and grass cover. Analyses of these patterns aim to reduce uncertainty in the relative roles played by changes in grazing and fire regimes in causing bush encroachment.

Ecological studies have been conducted in the Makoba ranch block in the eastern Kalahari (Fig. 2) since 1988. This is an area of deep Kalahari sand soils, traditionally dominated by mixed savanna communities, with grassy plains dominated by tufted perennials including Eragrostis, Pogonarthia and Stipagrostis species and mixed shrubs, typically Lonchocarpus, Acacia, Grewia, Rhigozum and Terminalia species. Studies reported here focus on one ranch, Uwe Aboo, chosen because of previous ranch-scale ecological (Perkins & Thomas, 1993a,b) and soil hydrochemical studies (Dougill et al., 1997, 1998a).

Bush encroachment here was characterized by an increased dominance of Acacia species and Grewia flava (Dougill, Trodd & Shaw, 1998b) within 1 km of the borehole, an area that has experienced intensive grazing for over 20 years.

Studies were based on a grazing gradient approach, with spatial patterns of bush and grass cover being sampled at set distances along a study transect radiating from the borehole (50, 200, 400, 800 and 2600 m from the borehole – matching the exponential decline in grazing intensity with distance from the borehole) and at an ungrazed control site within the confines of the
neighbouring Makoba veterinary cordon fence. At each site, a 30 by 30 m plot was demarcated and the location of all bush species (over 0.2 m height) was mapped by the co-ordinates of their rooting points and two canopy diameters (N–S and E–W). Where canopies of individual bushes overlapped, boundaries were drawn of clumps containing the numerous coalescing bushes. Patterns of grass biomass were measured around selected bushes (at least four bushes from each of the main bush species present) within these 30 by 30 m plots at the end of the 1995 dry season. Change in grass biomass was quantified by harvesting above ground herbaceous biomass from 0.5 m by 0.5 m quadrats placed on two transects (N–S and E–W) away from a bush’s central rooting point until an unprotected ‘open’ setting was reached. The dry season timing of this study is vital, as this is the period when grass biomass is at a minimum and grasses are at the greatest risk of grazing-induced mortality (Mott et al., 1992). Consequently, such studies provide the best indicator of the ecological resilience provided by residual sub-bush-canopy grass cover (and the associated seed resource) and of the continuity of grass cover, a vital factor determining the probability of intense fires (Frost & Robertson, 1987). Unfortunately, the dry season analysis precludes detailed species identification such that ecological differences between open and sub-bush-canopy niches remains an important variable to be
Table 1. Bush clumping indices for 30 by 30 m plots in the bush-encroached zone (50, 200, 400 and 800 m from borehole), the grazing reserve (2600 m from borehole) and the ungrazed control site on Uwe Aboo Ranch, Makoba ranches

<table>
<thead>
<tr>
<th>Study sites</th>
<th>Mean no. of distinct clumps</th>
<th>Mean no. of bushes</th>
<th>Bush clumping index*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bush encroached zone (n=16)</td>
<td>76.3</td>
<td>193</td>
<td>2.62 ± 0.45</td>
</tr>
<tr>
<td>Grazing reserve (n=8)</td>
<td>90.5</td>
<td>164</td>
<td>1.80 ± 0.28</td>
</tr>
<tr>
<td>Ungrazed control (n=4)</td>
<td>51.0</td>
<td>83</td>
<td>1.63 ± 0.51</td>
</tr>
</tbody>
</table>

*(Bushes/clump – mean and 95% confidence interval).

quantified for this area, together with preliminary analysis of the grass seed bank, a factor which has only been superficially studied for this region (O’Connor & Pickett, 1992; Veenendaal et al., 1996).

Analysis of bush patterns showed that there were significantly greater number of bushes per clump at ‘bush-encroached’ sites (within 1 km of borehole) compared to ‘grazing reserve’ (2600 m from borehole) and ‘control’ (ungrazed veterinary fence) sites (Table 1), displaying the increased spatial clumping of bushes with their increased density. Recognition that bush patterning displayed a distinct lack of uniformity within sampled plots, matches analysis of aggregated patterns described for mixed bush-encroached stands in the western Kalahari (Skarpe, 1991).

The ecological resilience implied by the transition back to grass-dominance following fires (transition $X_4$ – Fig. 1) requires residual grass biomass and therefore a grass seed resource, to be present in the landscape. At sites in the Makoba ranch block, residual grass biomass is concentrated in the protected sub-bush-canopy niche of low growing dense bushes, such as *Acacia ataxacantha* and *Grewia flava*. At bush encroached sites, sub-bush-canopy niches support significantly higher ($P=0.009; n=45$; assessed by two-tailed $t$-test) grass biomass than neighbouring gaps between clumps, a pattern opposite to that found at ungrazed ‘control’ sites (Fig. 3). Findings of ecological studies in other regions, such as around Lethlakeng (Ringrose et al., 1995) and the mid-Boteti region (Ringrose et al., 1996), imply that mechanisms exist to diminish the physical protection provided by bush canopies. Reduction in this protection is likely as bushes mature or through changes in canopy form as a result of livestock browsing, such that this sub-bush-canopy refuge for grasses can be removed, so enabling a transition to a degraded, almost completely bush-dominant state (transition $X_3$).

Extending ground-based studies is essential to provide an opportunity for improving our understanding of how intensive cattle grazing links to fire frequency and ecosystem resilience. However, resolving the uncertainty about the conditions under which irreversible ecosystem changes occur (transition $X_3$) would take many years of extensive field analysis. Furthermore, it is unlikely that spatially limited ground studies will be able to investigate fully the various combinations of biotic and abiotic factors that lead to a retention of grass-dominant areas within landscapes. It is to this end that extending the spatial and temporal coverage of ecological change monitoring by investigating archives of satellite Earth observation data is considered.

Earth observation by remote sensing

Earth observation data have been widely used to highlight variations in vegetation community characteristics from their reflectance characteristics. Many studies have observed the ability of Landsat TM and MSS imagery to assess vegetation abundance (e.g. Smith et al., 1990) or to estimate plant size and density in semiarid environments (e.g. Franklin & Turner, 1992). However, vegetation community structure (ratio of bush:grass cover) is the key dependent variable in the state-and-transition model. Importantly, ground radiometry experiments show differences in the multi-spectral reflectance of grass and bush canopies that have been attributed to their different life-forms and phenologies (Franklin et al., 1993). However, the main difficulty in assessing savanna vegetation is caused by the limited radiometric dimensionality of optical Earth observation data (Graetz & Gentle, 1982). Trodd & Dougill (1998) report the results of ground radiometry experiments on bush and grass canopies at the Uwe Aboo ranch that show that during the dry season there is limited dimensionality in visible and near-infrared reflectance data. These results imply that the presence of any vegetation – bush or grass, green or senescent – on the bright Kalahari sand soils has the effect of decreasing the reflected radiance from that surface. Subsequently, a radiometric difference between bush

and grass canopies cannot be distinguished from other spectral effects and it follows that it is impossible to make direct estimates of the relative proportions of grass and bush cover from single date optical remotely sensed data.

An alternative strategy has been to exploit information contained in a multi-temporal sequence of Earth observation data. These studies have tended to use time series of spectral vegetation indices, rather than the reflectance in individual wavebands, because they are more sensitive to variations in shrub type and phenology (Duncan et al., 1993). Analysis of temporal changes in vegetation indices based on high temporal frequency remote sensing data allows us to monitor vegetation phenology and biome seasonality (Justice et al., 1985). Vegetation indices selected for such studies are those that represent the absorption of photosynthetically active radiation, but they are seriously affected by atmospheric conditions, sensor calibration, sensor viewing conditions, solar illumination geometry and soil moisture, colour and brightness (Bannari et al., 1995; Lambin, 1996). Fortunately, some of these effects are minimized in the preparation of global data sets such as International Geosphere Biosphere Programme Global 1 km AVHRR (IGBP G1K) data (Eidenshink & Faundeen, 1994). Consequently, assuming that the soil background at a site is invariant, temporal variations in the Normalized Difference Vegetation Index (NDVI — the normalized difference of the bi-directional reflectances in the red and near-infrared bands) are the product of either variations in the angular distribution of incoming radiation or of phenological characteristics of the vegetation.

Annual and seasonal variations in AVHRR NDVI have been used to monitor regional primary productivity (e.g. Tucker et al., 1991). In a study of Botswana, Nicholson & Farrar (1994) observed different levels of productivity associated with different vegetation community types and other studies have observed differences in the phenological phase of different vegetation community types from AVHRR NDVI data (Fuller & Prince, 1996; Peters et al., 1997). A subsequent study within Central District, Botswana, investigated seasonal patterns of monthly NDVI values for three bush-dominant sites in the Makoba ranch.
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Block and two grass-dominant sites of the Central Kalahari Game Reserve (Trodd et al., 1997). AVHRR NDVI 10-day maximum value composites for the period from July 1992 to June 1993 were extracted from the IGBP G1K data set accessible through the World Wide Web – http://edcwww.cr.usgs.gov/landdaac/. NDVI values were sampled from five pixels at each site and, to reduce the effects of contamination by cloud cover (Gutman, 1987), maximum monthly values were calculated from the fifteen 10-day values for each site. Daily rainfall data that had been recorded at Uwe Abo ranch during the same period were aggregated into monthly totals. Total rainfall for the study period was 317 mm, with rainfall concentrated between October 1992 and April 1993, peaking in December 1992.

The analysis focused on the relationship between rainfall and NDVI for each of the sites (Fig. 4). At grass-dominant sites, the NDVI increased rapidly following first rainfall. In contrast, at bush-dominant sites, the NDVI increased in the month preceding rainfall. Peak NDVI values occurred simultaneously at both site types, but were found to decrease more rapidly at grass-dominant sites compared to bush-dominant sites. These patterns correspond to seasonal dynamics of above ground biomass production proposed in an ecological growth response model (Fig. 4a) devised for southern African savannas by Scholes & Walker (1993). In this model, trees and bushes exhibit signs of growth prior to first ‘effective’ rainfall due to their ability to access subsurface moisture, whereas grasses respond to rainfall events and exhibit a shorter, more intensive growth pulse (Scholes & Walker, 1993). Inverting these relationships will permit the use of coarse resolution remote sensing data to estimate vegetation community structure providing that they can be validated by ground-based measurements. This is not an easy task, but concerted action in acquiring large area ground data (e.g. Loveland et al., 1991; De Fries & Townshend, 1994; Prince et al., 1995) and the development and application of geostatistical techniques (Chappell, 1998; Stein et al., 1998) suggest that it is possible. Furthermore, new sensors such as ESA’s Along Track Scanning Radiometer, and its successor – the Advanced ATSR, SPOT VEGETATION and the EOS Moderate Resolution Imaging Spectrometer, will acquire optical Earth observation data of greater geometric accuracy and precision and at the necessary temporal frequency to drive phenologically based models for image interpretation. These new data sources, in association with the archive of AVHRR data, will therefore provide an information source capable of monitoring changes in savanna vegetation community structure.

Of the other remaining uncertainties in the ecological model (Fig. 1), it is apparent that ground-based surveys cannot provide data on the frequency, intensity and spatial extent of fires with the required spatial and temporal coverage. Conversely, Earth observation data can map fire scar patterns (e.g. O’Neill et al., 1993) and appropriate archives are being created (e.g. Arino & Melinotte, 1998). If we accept that wildfires operate at a return interval in the order of $3 \times 10^3$ days (Graetz, 1987), then it suggests that fire regimes could be constructed for areas, such as the Kalahari, for which Landsat MSS and TM data are available from 1972 to present. In addition, problems exist with spatially representative measurement of rainfall variability patterns due to the paucity of the ground-based raingauge network and inherent spatial and temporal variability of rainfall. In this regard, Cold-Cloud-Duration data, derived from Meteosat thermal infrared images with a spatial resolution of 5 km, have been used to estimate 10 day rainfall for southern Africa since 1993 (Thorne et al., 1997), an operation that will reduce a major uncertainty in the use of temporal data sets caused by the lack, and uneven distribution, of raingauges in the Kalahari (Farrar et al., 1994). Such advances display the increasingly positive contribution that Earth observation data adds to ecological-change monitoring and modelling when this is carefully integrated with thorough and extensive ground truthing and affiliated fine scale ecological studies. The requirements for such integrated studies are discussed below.

**INTEGRATION OF GROUND-BASED AND REMOTELY SENSED INFORMATION**

This discussion of recent research at the Makoba ranch block has identified information requirements to reduce uncertainties in an ecological state-and-transition model and has demonstrated the strengths and limitations of ground-based and remote sensing sources. It is evident that the information content of either source alone is unable to meet the information requirements for assessing the combination of processes that cause transitions to a bush-dominant state. However, by integrating these sources, ecologists can expect to acquire most of the data required to test and

develop ecological models. Possible mechanisms for such integration are illustrated conceptually in Fig. 5, based on experiences of studies in the Makoba ranches and the overall need for a clear input into agricultural management strategies. Success will be most likely if integration occurs in study formulation, as opposed to attempting integration at a later stage.

Ecologists desire to improve understanding of how fine-scale vegetation patterns relate to different processes, and Earth observation scientists appreciate that subpixel vegetation patterns influence the level of reflectance (Begue, 1993). These patterns may be due to the presence of sub-bush-canopy grass or clumping of bushes, and in developing new algorithms for image interpretation, such as canopy reflectance models (Franklin & Turner, 1992), remote sensing studies will benefit by representing these ecological variables as parameters of the model. Similarly, change detection in remote sensing requires an appreciation of land cover dynamics that is embedded in ecological growth.
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Fig. 5. Monitoring, modelling and management of open savannas: an integrated research framework using multisource information.

response models. Therefore, there exists a symbiotic relationship between these two fields of research.

CONCLUSION

Ecological models of savannas have advanced in the last decade but current models still exhibit uncertainties regarding the conditions under which transitions are likely to occur and the rates of change. Reducing these uncertainties, refining and validating models, all require data, and ground-based ecological surveys, combined with existing and new Earth observation missions, are — and are likely to remain — the major sources of these data. Data are required to characterize changes in the landscape at a range of spatial and temporal scales, matching the range of scales at which determinant processes affect vegetation communities, such that ground-based studies alone are insufficient. Equally, Earth observation data provide only surrogate measures and require ground truthing and interpretation to yield ecologically meaningful information. The synthesis of studies in part of the Kalahari detailed here has shown that the full benefits will only materialize with improved understanding of Earth observation data and that, in part, this will be achieved by greater input from fine-scale ecological studies.

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