Confronting complexity: fire policy choices in South African savanna parks

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This paper is derived from a presentation at the conference 'Fire and savanna landscapes in northern Australia: regional lessons and global challenges', Darwin, Australia, 8–9 July 2002

Abstract. Changes in ecological concepts and a new focus on biodiversity as a central objective have led to changes in fire policies in South African savanna parks. Prescribed burning using fixed fire intervals is being replaced by systems that promote more variable fire regimes and greater management flexibility. Three policy alternatives have been proposed for Kruger National Park: a lightning fire policy, patch mosaic burning, and burning based on ecological criteria. There is no agreement as yet on which policy to adopt. However there is growing consensus on the use of a management system using ‘thresholds of potential concern’ to evaluate the outcome of different policies. These thresholds have been established for numerous indicators, help focus monitoring activities, and guide managers on the need for active intervention. We discuss the applicability of the policy alternatives for preventing successional change from savanna to forest and promoting grazing lawns and their associated grazers. We conclude that none of the current policies is universally applicable. A prescriptive program of frequent, high intensity burns will be required in mesic savannas to prevent succession to forests. In arid savannas, fire regimes designed to promote variable fire frequencies and fire sizes would be preferred to maintain greater diversity of grassland swards and grazer communities. The lessons learned from fire policy debates in South African savannas are of wider relevance for managing conservation areas elsewhere.

Introduction

Fire management in conservation areas in South African savannas is currently in a state of flux. Practices maintained for nearly 50 years have been challenged and, in some parks, completely replaced by radically different alternatives, mostly in the last decade. There is widespread dissatisfaction with policies used for nearly 50 years, but no consensus on suitable replacements. At the same time, a management system has been developed, primarily in Kruger National Park, which allows much greater flexibility than in the past and which is intended to promote adaptive management. The management system is sufficiently flexible to allow diverse fire policies, once there is any agreement on a suitable policy. In this paper, we will briefly review fire policies used in savanna parks over the last century, the alternatives currently being debated, and why, in our view, the issue has been so contentious. Much of the controversy stems from adoption of new management objectives (biodiversity) and changes in conceptual thinking in ecology—from ‘balance of nature’ to ‘flux of nature’ thinking together with an appreciation for the importance of heterogeneity and scale, both spatial and temporal, for conservation. The new concepts have been widely accepted as a valid alternative to those prevailing for most of the 20th Century. However their implementation in practical conservation management is proving inordinately difficult. As a guide to considering the utility of current alternative policies, we discuss key features of the ecology of savannas that need to be considered before adopting a particular policy.

Fire management in Kruger National Park

Kruger National Park (KNP), an area of just under 2 million ha, has been the focus of recent debates on fire policy in savanna parks. Most other parks in South African savannas are smaller and have less flexibility in fire management because of their large perimeters with neighbours whose land use aims, and hence fire management, may frequently be at odds with those of park managers. However there are similar debates over fire policy in these smaller parks too. Fire policy options for KNP have been intensively debated over the last decade, along with revision of conservation policy on other aspects of park management (van Wilgen \textit{et al.} 1998; Biggs and Potgieter 1999; Whyte \textit{et al.} 1999). Fire history of the park is also well documented (Trollope 1993; van Wilgen \textit{et al.} 2000), so that KNP provides a useful case history for issues in savanna park management in the region.

The history of fire management in KNP is informative in illustrating changing conservation philosophies and their consequences (Biggs and Potgieter 1999; van Wilgen \textit{et al.} 2000).
Little is known of the fire regime before the 20th Century. After the park’s proclamation in 1926, areas were occasionally burnt for green grazing. By 1948, the park was sufficiently well staffed to allow burning of firebreaks to control wildfires. The policy from 1948 until 1956 was fire exclusion, consistent with a general antipathy to wildfire among scientists at that time. Fire suppression led to some inadvertent very large fires, resulting in a change of policy in the 1950s. In 1957, a policy of prescribed burning on a 3-year cycle in spring was introduced. The park was divided into management blocks to facilitate prescribed burning. At the same time, a comprehensive burning experiment was introduced with replicated season and frequency treatments applied in 7 ha replicates at four sites representing some of the major vegetation types in the park. These experiments have been maintained to the present day (2002, KNP Scientific Services) with some additional treatments added in the late 1970s. By 1975, it had become clear that the 3-year rotation was too short to accumulate sufficient fuel, especially during dry periods and in the more arid parts of the park. Variations of the policy were introduced with field visits to evaluate conditions before a block was burnt. These could result, for example, in a delay of several years before burning. More variable fire seasons were also introduced.

In 1992, the entire fire policy was changed, under the influence of a strong wilderness lobby in park management and coinciding with dissatisfaction with the existing policy among proponents of more variable fire regimes to promote biodiversity. The new policy was to permit lightning ignited fires but to suppress fires from anthropogenic sources with the intention of returning to a ‘natural’ fire regime. Where lightning-ignited fires stopped at unnatural barriers (such as tourist roads) fires were actively lit across the barriers by park staff. As an interim policy, a single lightning fire that burnt more than 50% of one of the 17 large fire management blocks was suppressed. The idea here was that past management had homogenised fuels and that this ‘lid’ on natural fires was needed to help return the system to a more likely regime of smaller fires. The lightning fire regime was maintained until 2002 but has now been abandoned. A new fire policy for the park is currently being developed.

The changes in fire policy at Kruger present, in a microcosm, changing attitudes to fire and to conservation over the last century. The debate over current policies reflects different philosophies of natural resource management and, to some extent, lack of knowledge of the ecology of the system. The block burning policy was developed when the ‘balance of nature’ was the dominant paradigm in ecology and was based on the leading ideas in rangeland science at the time. The objective resembled livestock management in using fire to promote large productive herds of mammals by maintaining a grass sward and tree densities optimal for mammal populations. At the same time, mammal populations were managed quite intensively, mostly by culling and the addition of artificial watering points. There was considerable research emphasis on suitable mammal carrying capacities to guide large mammal population management (e.g. Whyte et al. 1999 for a discussion of elephant policy).

Why was this interventionist policy changed in the 1990s? Firstly conservation biologists had shifted to ‘biodiversity’ as the preferred management objective. The focus on large mammals of earlier objectives was seen to favour some components of the biota more than others. Managing elephant populations, for example, is a larger problem than ensuring sustainable elephant populations. The animals have major impacts on other components of the ecosystem and these need to be taken into account where the objective is the broader one of maintaining ‘biodiversity’ (Cumming et al. 1997). Secondly, ecologists world-wide had begun to shift to a new paradigm, the ‘flux of nature’ (Pickett et al. 1992), embracing variability in space (e.g. Wiens 1997) and time. Savannas were seen to be non-equilibrial systems, not changing continuously as assumed by rangeland succession theory but in fits and starts with unpredictable pathways and unpredictable destinations depending on contingent factors at critical times (‘events’), such as periods immediately following the breaking of a drought (Frost et al. 1986; Westoby et al. 1989; Mentis and Bailey 1990). In the light of these new concepts, block burning was seen as undesirable because static fire frequencies had ‘homogenised’ savanna structure, large trees had been reduced in density by a condition of too frequent burning, and elephant damage and grass sward composition had deteriorated, especially in arid areas, because of too frequent burning. There was also a perception (untested) that the centre of areas ignited by perimeter burning suffered ‘unnaturally’ high fire intensities. The new paradigm has also extended to changes in mammal policy with no (or very limited) culling (e.g. Whyte et al. 1999, for elephant) and removal of artificial water points. It is interesting to note that no direct evidence for loss of biodiversity has been put forward to justify abandoning block burning. Indeed scientific evidence seems to have had very little effect on changes in fire management other than in supplying a new conceptual framework that is more appealing than its predecessor. However population declines in roan and sable, both rare antelope in South African savannas, has been attributed to habitat changes and associated changes in predation following provision of artificial water points (Harrington et al. 1999).

**Current policy alternatives**

**Natural fire regimes**

Three alternative fire policies are currently under scrutiny in Kruger National Park (van Wilgen et al. 1998). These are ‘natural’ fire regimes, patch mosaic burning and ‘ecological criteria’. Natural fire regimes aim to re-create pre-settlement fire regimes, largely by promoting lightning fires and suppressing fires of human origin. The underlying philosophy is...
to promote ‘wilderness’ with minimal human intervention in fire and other ecological processes in the park. The wilderness philosophy is somewhat incongruous on a continent that has been the stage for hominid evolution for several million years. Our own species may have been the first hominid to regularly use fire in daily life (based on the presence of domestic hearths in caves) and has done so, according to the archaeological record, for at least 100 000 years (Deacon and Deacon 1999). We know very little of the aboriginal fire regime in South Africa. The San, hunter gatherer descendants of our African ancestors, were displaced from South African savanna regions by Bantu (from ca 500 AD) and European (mid 19th Century) settlers. There is no extant tradition of landscape burning comparable to that by Australian Aborigines and very little archaeological or ethnographic information exists to guide attempts to re-create ‘natural’ fire regimes that also incorporate the long human history of burning in the region (Hall 1984). Thus the ‘natural’ fire regime policy was based around permitting lightning fires and actively promoting their spread when fires were blocked by roads and the like, but without any pre-determined concept of what a ‘natural’ regime might look like.

**Patch mosaic burning (PMB)**

Patch mosaic burning has been developed to promote habitat heterogeneity for the explicit objective of promoting conservation of biodiversity (Parr and Brockett 1999). It was developed in reaction to the perceived homogenisation of habitats caused by block burning. The assumption is that mosaics of savanna burnt at different ages will promote biodiversity by providing more varied habitats for habitat specialists or mosaics of habitats for species with varying habitat requirements. The assumption has not, as yet, been tested in South African savanna parks. Similar arguments have been made for biodiversity and heterogeneous fire regimes in Australian savannas (e.g. Russell-Smith 1997; Keith et al. 2002). The system has been implemented for a decade in Pilanesberg, a 50 000 ha savanna park, and the aims (Parr and Brockett 1999), methods and outcome have been described in detail (Brockett et al. 2001). The reader is referred to these papers for more detail.

Fire managers set a target area to be burnt in a particular year (based on historical records of areas burnt in relation to fuel load which, in turn, is closely related to annual rainfall). Fires are set as point ignitions located at random (with some modification). Point ignitions are thought to result in more varied fire behaviour and fire severity than perimeter ignitions characteristic of block burns (Cheney and Sullivan 1997). The number of fires ignited and the area burnt in a particular month is based on the cumulative area burnt relative to the total desired for a particular year. The system is highly artificial and no attempt is made to mimic a natural fire regime. The intention is to use fires at different seasons to break up the fuel load and create a mosaic of heterogeneous areas of variable post-fire age.

It has succeeded in doing so after a decade’s application of the system in Pilanesberg (Brockett et al. 2001). As yet, there has been no study of the consequences for faunal diversity of this more heterogeneous mosaic of post-burn stages. Nor is it clear what scale of heterogeneity is appropriate and how this might vary for different species. Cheetahs, for example, might require extensive open habitats for hunting their prey. However the system could potentially be modified to change the distribution of fire sizes should the current system prove inappropriate.

**Ecological criteria**

Fire policy using ecological criteria emphasises known habitat requirements for elements of the biota and known fire regimes to manipulate plant communities to produce these requirements (Trollope 1990). It has been developed from rangeland science and extensive experimental work in both livestock farming and conservation areas. The most important departures from the block burning system used for nearly half a century in the park is the promotion of point ignitions, rather than perimeter fires, where it is considered safe to do so, and the use of rangeland condition assessments to make decisions on burning (rather than a fixed pre-determined fire regime; Trollope et al. 1989). At KNP, ecosystem condition is assessed by managers at over 500 points throughout the park. Assessments include changes in grass composition using ‘increaser/decreaser’ categories and fuel load measurements which are then used to assess whether a management block should be burnt or not (Trollope et al. 1989). In principle, the system could be adjusted to incorporate new findings on how different components of biodiversity respond to changes in the fire regime. Variations of ‘ecological criteria’ for setting burning policy are widely used in most savanna parks in South Africa.

**The management framework—Thresholds of potential concern (TPCs)**

Contending fire policies at KNP have been, and will be, assessed in a new management framework applied since the mid 1990s. The central feature of the new policy is to assess the effects of management interventions on biodiversity by monitoring whether key indicators are approaching ‘thresholds of potential concern’ (TPCs; Rogers and Biggs 1999). Upper and lower thresholds have been defined for a wide range of biotic and abiotic ecosystem descriptors. If a threshold is reached, than either (a) management actions are implemented to prevent deleterious consequences of exceeding thresholds; or (b) the threshold is re-calibrated to a more appropriate level. TPCs have been set after extensive discussions with interested scientists, park research staff and managers and therefore reflect a degree of consensus.
within the current limits of knowledge (see e.g. Whyte et al. 1999 for a discussion of the process of setting TPCs for elephant management). They allow considerable flexibility of management action within the upper and lower thresholds, promote focussed monitoring activities, and provide feedback for managers on their success in meeting conservation objectives. The TPC framework is also intended to promote adaptive management since the results of management actions are constantly being assessed against agreed criteria. It has been developed within the framework of biodiversity as a central conservation objective and within the paradigm of ‘nature in flux’ where spatial and temporal heterogeneity are central concerns.

An example of the TPC approach in action is its use in assessing the ‘natural’ fire policy for KNP. Since this policy rested on promoting lightning ignitions and suppressing anthropogenic ignitions, a TPC for area burnt by humans was set as not more than 25% of the park in a given year. The TPC was greatly exceeded in several successive years because of numerous fires ignited by people bordering the park or illegally in transit through it. This, along with the logistic problems of fighting all anthropogenic fires, regardless of whether they were ecologically beneficial or not, resulted in the policy being abandoned in 2001.

Suggestions for appropriate thresholds of potential concern for fire management in KNP are discussed in detail by van Wilgen et al. (1998). TPCs have largely been based on analyses of historical fire regimes in the park (Trollope 1993; van Wilgen et al. 2000) to set upper and lower thresholds for different aspects of the fire regime. Like the patch mosaic burning system, the fire TPCs advocate fire regime statistics as surrogates for biodiversity. This is partly because fire scars are much easier to measure than biodiversity responses. There is also a long existing record of fire history in KNP (and most other savanna parks) to help define thresholds for different aspects of the fire regime. Like the patch mosaic burning system, the fire TPCs advocate fire regime statistics as surrogates for biodiversity. This is partly because fire scars are much easier to measure than biodiversity responses. There is also a long existing record of fire history in KNP (and most other savanna parks) to help define thresholds for different aspects of the fire regime. Though there is considerable debate over the three possible fire policy alternatives, there is very little debate over the use of the TPC system to measure the success of competing policies.

Ecological considerations for fire management in South African savanna parks
The current debate on appropriate fire policy for conservation areas has taken little cognisance of the ecological effects of fire in South African savannas. For example, data from the nearly 50 years of experimental burning treatments was not evaluated when setting up policy alternatives for KNP (Biggs and Potgieter 1999). What biological criteria should we be using and how might they influence choice of fire policy? Do different considerations apply in reserves in different geographic settings and of different sizes? We discuss current knowledge on the use of fire for managing tree densities and the influence of fire regimes on the grass sward as examples of potential ecological effects of fire regimes that should be informing management decisions.

Tree cover in savannas
A major issue in fire management is controlling woody plant density and structure. There are two distinct types of savannas in South Africa, with differing importance of fire for controlling trees. Mesic savannas and grasslands (> 650 mm rainfall per year) have the successional potential to form broad-leaved forests if fire is excluded for a decade or more (Bond 1997; Bond et al. 2003a). The forests have an entirely different suite of mammal, bird, and presumably insect species and also have an entirely different flora. In contrast, arid savannas do not show successional tendencies to fire-excluding forests though they do experience fluctuations in tree densities and sizes under different fire regimes. The different role of fires in these savannas has recently been tested using Dynamic Global Vegetation Model simulations of potential stem biomass with and without fire (Bond et al. 2003a). These DGVM simulations predict that grasslands and savannas in mesic parts of South Africa can support much higher biomass (i.e. woody plants) in the absence of fire (Fig. 1). The predictions are supported by long-term fire exclusion experiments, including those at KNP (Bond et al. 2003a). These show a successional tendency to forest at most sites where the DGVM predicts > 8000 g C m⁻² stem biomass without fire (Bond et al. 2003a). Changes in tree densities and biomass, but without colonisation of forest species, has been observed at sites with simulated biomass of < 7000 g C m⁻² (Bond et al. 2003a).

Successional change to forest is a major management problem in Hluhluwe Game Reserve in KwaZulu Natal, where just 20 years of reduced fires caused a lightly wooded mesic savanna to become a scrub forest that resists burning (Watson and MacDonald 1983; Skowron et al. 1999). Most of the higher rainfall savanna areas of this reserve are threatened with conversion to an alternative ecosystem state. Conversion

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<tr>
<th>Stem biomass (g C m⁻²)</th>
<th>Mean annual rainfall (mm)</th>
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<tr>
<td>No fire</td>
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Fig. 1. Simulated above-ground biomass with and without fire for a transect through the summer rainfall grasslands and savannas of South Africa (lat. 26.75°S). Biomass was simulated by the Sheffield Dynamic Global Vegetation Model (Figure modified from Bond et al. 2003b).
can occur as a result of tourist roads which break up fires, causing incursion of forest trees. Similar conversion from grassland or savanna to thicket and forest is widespread in the mesic eastern half of South Africa (Hoffmann and O’Connor 1999; Bond et al. 2003a). Despite the major changes in biodiversity, and the impact on the visitor experience, there is still much confusion over whether to permit forest to close over or to use fire as a tool to prevent forest incursion in mesic savanna parks. In arid savannas, fire has less dramatic effects on the tree layer. However changes in tree densities and woodland structure are still markedly affected by the fire regime (Enslin et al. 2000; Shackleton and Scholes 2000). Structural changes in arid savannas are likely to influence faunal elements, especially birds but also mammals (Page and Walker 1978; Skowron and Bond 2003).

The processes underlying tree thickening (usually referred to as ‘bush encroachment’ in South Africa) are still debated after nearly a century of research on the problem (Scholes and Archer 1997; Roques et al. 2001). Nor is there a general explanation for why the phenomenon is so widespread in South Africa (and elsewhere, van Aaken 2000). Effective management of savanna trees depends on an understanding of their life history traits and demography. Unlike shrubs in many Australian heathland, and South African fynbos, communities, savanna trees in South Africa (Rutherford 1981; Bond 1997) and Australia (Williams et al. 1999) all resprout from an early age. Some species lose the ability to sprout as adults but the sprouter/non-sprouter dichotomy seems of little relevance in population management. Instead, increases in woody plant cover depend on demographic bottlenecks at the seedling recruitment stage and, for trees, sapling release into adult size classes (Midgley and Bond 2001). The latter process has been the focus of considerable recent research. Height growth is critical in savanna trees for two reasons. Firstly, saplings below a critical height are caught in the ‘fire trap’ where stems are killed by burning and the plant resprouts from the root collar or the stem depending on the fire intensity (Trollope 1984). Secondly, plants are vulnerable to browsing which can also prevent saplings in the ‘browse trap’ from growing taller (e.g. Pellew 1983). The critical escape height is similar for both browse and fire damage: saplings above ca. 3.5 m are relatively safe from damage.

Managing the release bottleneck–potential CO₂ effects

Trollope (1984) studied the relationship between topkill and fire intensity for a number of savanna tree species in different areas of South Africa. He was able to develop predictive equations relating fire intensity to topkill (death of above-ground stems) and use these for management guidelines on controlling tree densities. The principle of maximising top-kill with high intensity fires to prevent sapling release has been widely adopted in both conservation and farming areas. Recently the demographic mechanisms involved have been developed into a simulation model (the DBM–demographic bottleneck model) based on extensive empirical data from southern African savannas (Higgins et al. 2000). Higgins et al. (2000) argued that the processes involved in the DBM provided an alternative explanation for tree/grass coexistence in savannas from the current root niche differentiation hypothesis. The mechanisms are a special case of lottery models requiring variable fire regimes and ‘storage’ of successful recruitment or release events in long-lived adult size classes.

The DBM has also been used to simulate potential effects of changing atmospheric CO₂ on tree cover in savannas. Model output is highly sensitive to changes in sapling growth rates to escape heights, in addition to changes in fire intensity. After a fire, saplings would have adequate light, water and nutrients so that atmospheric CO₂ is more likely to limit growth (Bond and Midgley 2000). Changes in tree and grass growth rate in response to changing atmospheric CO₂ have been simulated using a DGVM (Woodward et al. 2001) and incorporated into the DBM to simulate demographic responses (Bond et al. 2003b). The results of the simulations are shown in Fig. 2. They predict that trees would have been eliminated from these savannas during the last glacial when CO₂ levels fell to 180 ppm. This is consistent with the pollen record for the region (Scott 2002). The simulations also indicate that the current trend of increased tree cover may partly be explained by CO₂ fertilisation from pre-industrial concentrations (270 ppm) to current levels (360 ppm; Bond et al. 2003b). Experiments are currently in progress to test the magnitude of the growth response in several savanna species and to test the accuracy of DGVM simulations of CO₂ effects on stem growth.

If high CO₂ is a factor in current tree increases in some South African savannas, then a laissez faire approach to fire management is a dangerous option. Using fire to control tree cover should be a central concern. In Hluhluwe Game

[Fig. 2. Simulated median tree densities in relation to atmospheric CO₂ for an arid (300 mm p.a.), semi-arid (600 mm) and mesic (750 mm) savanna. The model simulates last glacial (180 ppm), pre-industrial (270 ppm) and current (360 ppm) CO₂ concentrations. Medians are calculated for a 5000 year simulation run using a savanna demography model with grass and tree growth rates derived from the Sheffield DGVM (from Bond et al. 2003b).]
Reserve, for example, much more effort may be required to burn under weather and fuel conditions that promote high intensity fires that would prevent the savannas from changing to woodlands and forests. Fire policies based on ‘natural’ or pre-European ideals do not take into account global change. There is nothing ‘natural’ about the increase from 270 to 360 ppm in atmospheric CO$_2$ over the last century or two. Indeed the planet has not experienced such high CO$_2$ levels for at least the last half million years (Petit et al. 1999). Management of mesic savanna reserves is likely to become even more challenging in future if we wish to maintain the fauna and flora of savannas, rather than thickets and forests.

Grazers, grasses and fire

African savanna parks differ from those in most other continents in having a largely intact large mammal fauna. Other continents experienced mass extinction of large mammals in the late Pleistocene (Owen-Smith 1989). Some authors have argued that the current prevalence of fire in grasslands and savannas in Australia and North America may be an artifact of the extinction of large mammal herbivores on these continents (Owen-Smith 1989; Flannery 1995). Africa provides unique opportunities for looking at the interplay between a largely intact mammal fauna and fire. Unfortunately this relationship is still poorly studied. The reciprocal effects of grazers on fire regime and fire on grazers has therefore not been considered in debates on fire policy at KNP (H. Biggs, personal communication).

Sustained heavy grazing can reduce grass fuel loads below the threshold needed to carry a fire. Africa has many short, grazing-tolerant grasses often forming ‘grazing lawns’ (McNaughton 1984). The existence of these grasses suggests a long evolutionary history of short grass swards adapted to heavy grazing. The lawn grasses, often members of the sub-family Chloridoideae, are capable of spreading vegetatively by stolons or rhizomes despite intense grazing pressure. Grazing lawns support high densities and a high diversity of grazing species. Grazings, lawns have been little studied in South African rangelands and are not mentioned in a recent comprehensive review (O’Connor and Bredenkamp 1997).

Unlike Australia, the existence of lawn grass species in Africa suggests that grazing mammals have long influenced the fire regime in some savannas. Grazer-created lawn areas may help create more heterogeneous landscapes for vertebrates and less extensive fires. In this respect they can be viewed as analogues to early dry season burning in Australian savannas (e.g. Williams et al. 2002). We have been studying the dynamics and distribution of lawn and bunch grasslands in Hluhluwe-Umfolozi Park (HUP) in Kwa-Zulu Natal. Both grass types occur over the full altitudinal range in the park (40–750 m asl) and over a mean annual rainfall gradient from <600 mm to 1000 mm. However, lawns are much more common in arid savannas. Both grassland types also occur over a wide range of soils in the park from relatively nutrient-poor hydromorphic sandy soils to nutrient rich, structured clay soils derived from dolerite (Zululand Grass Project, unpublished). Studies of vegetation change in the park suggest that drought, fire and mammal culling have influenced the extent of grazing lawns and plants and animals preferring this habitat (Bond et al. 2001). The implication is that both fire management and large mammal management can strongly influence the extent of alternative grassland types with the potential for landscape scale changes in biodiversity.

We are currently exploring the hypothesis that large, frequent fires indirectly reduce lawn grass formation (Fig. 3). Savanna fires are commonly used to remove old moribund growth and to attract animals to the post-fire flush of new growth (e.g. Wilsey 1996). An indirect consequence may be that grazers are drawn away from heavily grazed patches. Over the longer term, this would reduce the chance of lawn grass formation because grazers would not persist in a grazing patch long enough to cause compositional shifts from tall grass to lawn grass species (Fig. 3). The dispersive effect of burning would depend on fire size. Small burnt patches would concentrate grazers and might initiate a grazing patch. Larger fires, the norm, would act in the opposite manner. One prediction of this hypothesis is that grazing lawns would be infrequent in areas with high fire frequencies and much more common where fires are rare. Fire frequency varies greatly in HUP depending, partly, on proximity to park boundaries and active measures to reduce fire frequencies towards the centre of the reserve (Balfour and Howison 2002). Figure 4 shows the proportion of lawn patches in the reserve (derived from satellite imagery) in relation to fire return intervals over the last 40 years. Lawn grass areas are clearly associated with
These preliminary results suggest that frequent burning (fire return intervals < 4 years in our study area) inhibits the development of grazing lawns and hence indirectly reduces the density and diversity of large mammal species, primarily those (the majority) that prefer short grass habitats. Grazing lawns are only one of several, largely undocumented, outcomes of heavy grazing in South African rangelands. In contrast, a Patch Mosaic Burning system would tend to favour the development of grazing lawns. This is because it is designed to reduce fire sizes and generate a mix of fire frequencies. The effect would be to enhance the development of persistent grazing patches and promote the conversion of bunch to lawn grass communities. The lightning fire policy would also tend to promote more short grass areas. Under the lightning fire policy at KNP, the fire-return interval dropped from a median of 5 to nearly 13 years with much smaller fire sizes than those that had prevailed in the block burning era (van Wilgen et al. 2000). The ecological criteria policy could, of course, be adapted to one more consistent with managing for indigenous African mammal diversity, i.e. promoting grazing lawns, rather than promoting tall grass for beef cattle.

**Conclusions**

The challenge of managing for flux, rather than equilibrium, and for biodiversity, rather than narrower objectives, is a general one in conservation areas. It is interesting to note many areas of convergent thinking between Australian and South African biologists on appropriate conservation management (e.g. Gill and McCarthy 1998; Keith et al. 2002). Keith et al. (2002) reject the idea of managing fire regimes to maximise populations of target species, as have South African savanna park managers. They also reject ‘natural’ fire regimes as a plausible policy option. There is convergence, too, in promoting much more flexible management with limits to that flexibility set by pre-defined ‘thresholds’. However Keith et al. (2002) have called for explicitly biological criteria to be used in setting up new, flexible fire management policies in Australia whereas the current proposals for KNP have a less developed biological rationale. Part of that rationale should be an assessment of the role of fire in structuring savanna ecosystems. There is also an urgent need for information on biodiversity responses to habitat changes for a much wider range of organisms than large mammals in South African savannas.

There is abundant evidence that frequent fires are essential for maintaining mesic savannas in South Africa and preventing their conversion to forests. Frequent intense fires might be even more necessary than in the past if CO₂ fertilisation is promoting sapling growth and savanna conversion to forests.
Acknowledgements

WJB thanks Dave Balfour, Harry Biggs, Bruce Brockett, Kevin Rogers and Winston Trollope for discussions on fire management in savannas.

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Picket STA, Parker VT, Fiedler PL (1992) The new paradigm in ecology: implications for conservation biology above the species level. In ‘Conservation biology. The theory and practice of nature Parks with areas subject to conversion to forest cannot afford to adopt ‘natural’ burning policies in an unnatural setting. Promoting heterogeneity by promoting high variability in the fire regime also risks permanent loss of savannas to forests. However these policy options could be preferred in arid savannas where they would tend to favour the development of short grass swards and species dependent on them.

The current uncertainty over management directions for conserving biodiversity has led to innovative thinking in South African savannas. The TPC system offers much promise. It has helped managers develop a means for evaluating their actions and researchers to focus their work on key questions relevant to explicit management objectives. The problem of measuring biodiversity, a central conservation objective, has been reduced to measuring habitat heterogeneity as a surrogate for diversity. There are clear advantages but also risks in this approach. We need to know far more about plant and animal responses to habitat changes and how such changes are triggered by climate, fire and herbivory. However, Africa does have a particularly difficult problem to untangle. Nowhere else is there such profusion of large mammals and the interplay between these creatures, variable climate, and fire is intricate and complex with much still to be learnt about the rules.


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