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Implementing a patch-mosaic burning system in southern African savanna conservation areas

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I acknowledge the support of Roger Collinson (former Director of the Bophuthatswana National Parks Board) in allowing and encouraging me!, to experiment with fire in Pilanesberg National Park. I experimented with a large box of matches, and at first two game scouts, and later on my own in the park. When one of the first point-ignitions turned into a large fire (too late in the season, and fully cured fuels), and I ignited new fires on the other side of the park, and he said "Maybe in future have only one fire in the park at anytime." When beginning with point-ignitions, which were often lit in the late afternoon. The field staff were often reluctant to go home at 16h30, as they were used to watching the fires all night if need be! I thank the Wardens especially Jules Turnbull-Kemp, and Johan de Vries, and the many field staff that assisted me especially Pompie Mathebula, Thomas Sithole, and Gustaf Matjila.

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Implementing a patch-mosaic burning system in southern African savanna conservation areas

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Abstract

The implementation of a patch-mosaic burning system (PMB) in southern African savannas was investigated. Under such a system fires are point-ignited, under a range of fuel and weather conditions, and allowed to burn out by themselves. The system was applied in Pilanesberg National Park (PNP), South Africa over a period of nine years. The area burnt per year is a function of the grass fuel loads, and the number of fires per year is a function of the percentage area burnt. This should result in the creation of mosaics, which should vary in extent and existence in time. Cost effective techniques to measure grass fuel loads are therefore required. Calibrations for three techniques to estimate herbage biomass were developed: (1) disc pasture meter, (2) dry-weight ranking method, and (3) weight-estimate method. Theory underpinning the system was presented, and to implement the system a process of adaptive management was recommended. This should include both modelling and monitoring. The previous burning system was a prescribed block burning system (PBB), using fixed blocks and return periods, and was applied over nine years. Scale relationships using two grain (25 and 100 ha), and neighbourhood sizes (3 X 3, and 5 X 5), for a post-fire heterogeneity score (HS) were investigated. Linear relationships were found between them. Mean HS was higher under a PMB system than under a PBB system. For the PMB system there was a linear relationship between the HS and the amount burnt in the current year. This is likely to be explained as a function of: block sizes selected for burning, their spatial

arrangements to each other, and the percentage of park burnt. For the PMB system the HS was linearly (positively) related to the amount burnt in the current year, and negatively related to the amount burnt the previous year. A two-year comparison showed that the mean for pixels burnt biennially was higher under a PMB system than under a PBB system, as was the mean for it being unburnt for two successive years. This suggests that the pattern of fires has shifted under a PMB system, and this has resulted in variation in fire size, and therefore in fire intensity and fire frequency. The variation in fire size results in variation in post-fire spatial heterogeneity. Other landscape pattern measures should also be investigated, including their relationships with fire parameters, vegetation and herbivore relationships.

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Chapter 1

Introduction

This thesis deals with the implementation of a patch mosaic burning system in southern African savanna conservation areas. Conservation areas are defined as national parks, game and nature reserves, and protected areas. Changes in management objectives for protected areas in southern Africa over the past century, have been reflected in parallelled shifts in fire policy from anti-burning, to controlled burning, and most recently to adaptive fire management (Scott 1984; Mentis and Bailey 1990; Pyne 1997). Such changes in fire policy, objectives and practise, also reflect shifts in attitudes towards fire, as well as advances in ecological theory (Mentis and Bailey 1990; Scholes and Walker 1993; Scholes 1999). Consequently fire policies in South African savanna conservation areas have gone through a number of changes over the past century. For convenience the Kruger National Park (KNP), which is the largest conservation area in South Africa, is used to illustrate such changes. Shifts in policy are sometimes (but not always) parallelled by changes in fire policy in the United States of America, as well as in Australia (Scott 1984; Pyne 1991).

To place the thesis in perspective it is necessary to sketch a short background to protected area management, and then to discuss changes in fire policies. Against this background I then look at developments in systems theory, and savanna functioning. This is followed by a brief outline of the thesis.

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The direction and topic of this thesis is a result of that expert system. Art taught me the art of burning, and also the science of fire behaviour, and for that I thank him.

I thank my supervisor Prof. William Bond, and co-supervisors: Dr. Harry Biggs (Scientific Services, Kruger National Park), and Dr. Brian van Wilgen (CSIR). William is thanked for his insightful comments on some of the chapters. I would still like to know what was in the brown envelope that he travelled around the country with, in which he stored all his difficult questions about the theory underpinning a patch-mosaic burning system! I thank Harry Biggs for introducing me to landscape ecology.

I enjoyed working with Kate Parr on a publication in Koedoe which is reproduced in Chapter 4, and Naledi Maré for many long days of GIS work. I also thank Kate Parr, Andrew Hudak, Jessica Redfern, Tony Starfield, and Steve Higgins for comments, and Stephanie Frietag-Donaldson, Judith Kruger and Andre Potgieter for their assistance. Karin Burns, Helena Bryden Naledi Maré, and Ray Schaller, are thanked for drawing a figure. I thank Harry Biggs for providing me with accommodation (including at his house if need be) and facilities to work whenever I was in Skukuza.

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Protected area management

According to Mentis and Bailey (1990), protected areas have been and continue to be, managed to maintain what might be called their 'pristine state'. The definition of the pristine state has changed with changes in perceptions of ecosystem function, focussing on the maintenance of biotic diversity, and the biospheric processes that contribute to diversity (Huntley 1989; Mentis and Bailey 1990). Contemporary conservation management however focusses on: (1) ecosystem processes, (2) heterogeneity (and using it as a surrogate for biodiversity), and (3) adaptive management (i.e. process of learning and adjustment (Holling 1978; Walters 1986)) (Pickett pers. comm). Objectives for protected areas are now often stated as conserving biotic diversity, and recognize the importance of flux e.g. Kruger National Park (Braack 1997).

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Fire policies in Kruger National Park

In 1898 Sabi Game Reserve, in the eastern lowveld of South Africa, was proclaimed. The Kruger National Park was proclaimed (which included the Sabi Game Reserve) in 1926. In 1898 fire regime was more 'natural' than today. Fires occurred sporadically, ignited by local inhabitants or lightning (Mentis and Bailey 1990). In 1902 James Stevenson-Hamilton was appointed as the first Warden of the Sabi Game Reserve. He condoned using fire to flush-out predators, as part of an anti-predator strategy. In addition to such fires, other sources of fire would have been local inhabitants, and lightning. In addition Stevenson-Hamilton (1954) mentions that "natives on the Portuguese side of the border, invariably burned their grass in early winter, with the result that the game used to flock over to that side, and the same was the case with the shooting-farms to the west." He is referring to Portuguese East Africa (Mozambique), and the Drakensberg escarpment to the west. Therefore during the second quarter of the 20 th century autumn burning was implemented to keep the game inside the reserve (Stevenson-Hamilton 1954; Mentis and Bailey 1990). Stevenson-Hamilton (1954)[1] mentions the effects of late season hot fires, and suggested some of their biotic effects: "... controlled burning of old grass, that is grass of over twelve months old, is, I believe, essential to the welfare of the animals in a sanctuary. The burning should take place as early as possible in the year, usually between March and May, in order that the light winter rains in June and July may encourage young shoots to grow, sweet and green, on the burnt patches."

In 1916 in the USA, Clements introduced his theory of succession, in which vegetation was thought to progress towards a successional climatic climax. In Clement's thinking the community acts like an organism. Essential to this point of view is the idea that a community is a superorganism, an entity of many species that has emergent properties (Colinvaux 1973; 1986). Clement's view of a "super organism" attracted philosophers. In 1926 Jan Christian Smuts (then prime minister of South Africa) acknowledged the writings of Clements when he produced his philosophy of "holism" which was built on the "evidence of ecology." According to this philosophy the universe was made up of series of communities, each more than the sum of its parts, an idea that had some vogue (Colinvaux 1986). Professor John Phillips in 1934-35 published two papers in Ecology on: "*Succession, development, the climax, and the complex organism: An analysis of concepts*." (Colinvaux 1986). John Phillips working in South Africa in 1920's, was one of the earliest to investigate fire effects

[1]. First published in 1945; illustrated edition in 1954.

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in field trials in South Africa. One such experiment was laid-out at Groenkloof in 1918 (Scott 1984). Clement's succession theory had a major impact on fire policies worldwide. Consequently over this period there was a lot of interest in fire, with a number of field trials being laid out. Scott in 1938 laid out experiments on the effects of burning at different seasons with or without grazing near Ermelo, in the Transvaal (Scott 1984). In keeping with the Clementsian paradigm fire was regarded as unnatural, and retrogressive as it disrupted the development of climax vegetation, and there were attempts to exclude fire from the Kruger National Park (Mentis and Bailey 1990; Pyne 1997). Stevenson-Hamilton (1954) mentioned, in possible reference to these early fire experiments: "The principles which may hold good on a farm cannot always be applied to a wild game country", and "in theory old grass should be burned immediately after the first spring rains, and this may well be the most appropriate time on farms, where adequate precautions against accidental fires can be taken; but in a vast extent of wild bush country it is dangerous to delay all burning until the veld is completely dry." Pyne (1997) says ".. intellectuals schooled in European agronomy misread and distrusted veld burning. They argued that, as in England and Holland, farming and herding should proceed without fire. They even installed agricultural models for the management of game reserves and parks. Wildlife, they culled: the veld, they divided into paddocks. They sought to limit overgrazing by limiting overburning." This explains the beginnings of prescribed block burning.

In 1946 the Soil Conservation Act was passed. Under this act strict procedures were laid down regarding the use of fire, and the practise of controlled burning in arid savannas was virtually eliminated (Trollope 1984). Under this act burning in provincial nature reserves required permission from the Department of Agriculture. Changes in controlled burning as a

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result of the Soil Conservation Act, are reflected in a shift from autumn burning to spring burning for soil conservation purposes. A newly appointed Warden to the Kruger National Park even announced that the time had come to wean the park from veld burning once and for all (Pyne 1997). The National Party won the 1948 election, and this assured in changes in fire policies. In 1948 arguing from pastoral models, the National Parks Board decreed a new policy that sought to shift burning from autumn to spring, to restrict such fires to no more than once every five years (only after good spring rains), and to eliminate accidental fires (Mentis and Bailey 1990; Pyne 1997). This clearly represents the beginning of command and control thinking (see Holling and Meffe 1996; Andersen 1999). It is likely that the controlled burning policy would have resulted in high intensity fires (under high fuel loads, and late season fires), and these would have had effects on the structure of the woody component. Such changes in policy clearly relate to Clement's succession theory, and to early fire ecology (e.g. Phillips 1930;1971; Scott 1952;1971, and others).

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Approximately one quarter of the KNP burnt in 1953 (Trollope, Potgieter, Biggs, and Trollope 1998), and the following year a research program was initiated for a small-scale experiment to investigate the effects of frequency and season of burning, in all the major landscapes of the KNP. Prescribed burning was applied at a three-year interval (if possible) from 1957-1991. Clearly the earlier policy of excluding fire had failed, and a five-year interval was also too long. Van Wyk (1971) argued as followed: "historical facts, as well as the results of research, past and present, demand the regarding of fire as an integral part of the ecological complex." He argued for the retaining of the triennial burning program, saying "there seems to be little or no qualitative or quantitative changes in the grass stratum in case of triennially burned plots. Advantageous tendencies with regard to the woody stratum of the vegetation are also more pronounced on the triennially burned plots." As a result of the early results of the field trial fire continued to be applied on a fixed rotation period of 3 year interval in spring, and probably under an assumption of reduced soil erosion, and clearly as a compromise between fire exclusion on the one hand, or fire more frequently would lead to deterioration of the grass sward (van Wyk 1971), and fire less frequently which would impact on the woody component.

Developments in systems theory

It is recognized that systems consist of patches and mosaics at different scales (Weins 1976; Pickett and Rogers 1997), and these are structured as hierarchies. Hierarchies provide the logical construct around which to organize investigations of the structure and function of landscapes (Risser 1987). Hierarchy theory (Allen and Star 1982), demonstrates how processes and constraints change across scales, and how spatial and temporal scales tend to covary. This suggests that processes which operate over long temporal scales also operate over large spatial scales (long distances) (Goodchild and Quattrochi 1997), and systems at smaller scales change more rapidly, and are more variable than those at larger scales (Simmons, Cullinan and Thomas 1992). With the shift in savanna ecological theory which occurred as a result of a synthesis of savanna ecology and this resulted in a shift from stability to flux thinking (Frost, Menaut, Walker, Medina, Solbrig and Swift 1986; Westoby, Walker and Noy-Meir 1989), and this resulted in a relook at how fire should be managed in South African savannas (Scholes 1987; Mentis and Bailey 1990; Bailey, Brockett and Mentis 1993; Scholes and Walker 1993). At the same time there were major developments in landscape ecology (Turner 1989; Gooley 1989), with the importance of pattern in structuring

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systems, and of relationships between pattern and process (Turner, Dale and Gardner 1989; Turner and Gardner 1991). The recognition emerged of the important role that spatial heterogeneity plays in the structure and function of landscapes (Pickett and Rogers 1997).

Fire management policies elsewhere

Problems with the conventional approach to fire management in conservation areas which promotes late successional stages in vegetation, and low habitat diversity were recognized in Australia (Saxon 1984). In some national parks they also found the use of fire breaks to be too expensive and inconsistent with the management of extensive natural ecosystems e.g. Uluru (Ayers Rock-Mt Olga) National Park (Saxon 1984). Consequently in the early 1980's they shifted to the use of point ignitions, and mosaic burning in conservation areas (e.g. a patch-burning strategy in Uluru National Park (Saxon 1984); and an expert system for fire management in Kakaku National Park (Walker, Davis and Gill 1985)). These events in Uluru National Park, followed the 1974/75 fires in which 80% burnt (Pyne 1991).

Pilanesberg National Park in 1989, following the development of an expert system for fire management in South African savannas (see Bailey, Brockett, and Mentis 1993), shifted to a patch mosaic burning system to create a mosaic of burnt and unburnt areas shifting over space and time. This decision was also influenced by patch-burning in Australia.

Thesis outline

This thesis undertakes to explore the implementation of a patch-mosaic burning system in savanna conservation areas. Prior to developing a burning system for a protected area, it is necessary that clear objectives exist. The purpose of this thesis was not to test the theory underpinning this system, (some of which is presented in Chapter 4), but rather accepting this theory to then ask the question, how can it be applied in savanna conservation areas. Prior to these changes in fire management, management of parks often followed a command-and-control approach (Holling and Meffe 1996; Andersen 1999), and this lead to research being isolated (Andersen 1999). There is an opportunity to use a system of adaptive management in fire management in protected areas. Chapter 4 suggests that an adaptive management framework, should include both modelling and monitoring as two important parts of the management process. The system developed in a process of learning by doing is presented in Chapter 5. Under this system the amount burnt is a function of the grass fuel load. Consequently two chapters (Chapters 2 & 3), deal with measuring grass fuel loads so as to make decisions as to how much should be burnt. After the system has been applied how would one know whether it was working? At what scale should monitoring be undertaken? How does monitoring at this scale relate to monitoring at another scale? What are the relationships between scales? Chapter 6 explores relationships between scales, and ways of measuring the heterogeneity of the fire patterns, and relating these to various fire parameters. Chapter 7 is a summary.

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Chapter 2

Research note: Calibrating a disc pasture meter to estimate grass fuel loads on the Zululand Coastal Plain

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(African Journal of Range and Forage Science)

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Abstract

A disc meter was calibrated to estimate fuel load on the Zululand coastal plain. The regression model was valid for fuel load estimates within a disc height of 1 - 58 cm. The mean calibrated disc height was 11.8 cm. Estimated fuel load at mean disc height, with standard error (SE), was 4593 ± 113 kg ha^{-1}. Using this equation to estimate fuel load at the calibrated mean, the precision of the estimates varied depending on the number of disc readings. As sample size increased there was at first a rapid reduction in the SE and hence in the 95% confidence limits (CL) for the predicted fuel load. There was little reduction in the CL with increasing sample size for >100 observations. To achieve a CL ± 10% of 5 000 kg ha^{-1}, 100 samples are needed. With fuel loads of less than 4 000 kg ha^{-1}, >160 samples are required to achieve the same precision. Sampling with >100 readings is not, however, recommended owing to the poor reward (precision) per unit of sampling effort.

Additional index words: Coastal Forest and Thornveld, fire management, herbage mass, standing crop

To facilitate consistent decision making for fire management, a fuel load estimate is required as an input to a decision-aid (Bailey *et al.* 1993). The use of such aids also encourages the implementation of formal adaptive management (Holling 1978), i.e. the decision-aid can easily be refined in the light of experience. A grass fuel load estimate would, for example, help answer questions regarding which areas require burning.

Fuel load is an important determinant of fire intensity, because of its effects on the amount of heat energy released, during a fire (Luke & McArthur 1978). Most of the fuel loads in grasslands and savannas are comprised of standing grass (Trollope & Potgieter 1985; Trollope & Potgieter 1986). An estimate of fuel load therefore involves the measurement of herbage mass per unit area.

The disc pasture meter (Bransby & Tainton 1977) was calibrated to estimate grass fuel loads in the Eastern Cape (Trollope 1983) and the Kruger National Park (Trollope & Potgieter 1986). Trollope & Potgieter's (1986) regression model could not be used to estimate fuel loads on the Zululand coastal plain because of the model's limited scope. Their model was valid for fuel-load predictions between disc height readings of 1.9 cm and 31.6 cm. In sampling fuel loads on the Zululand coastal plain with a disc meter, readings would fall above this range. Conlong (1986) calibrated a standard disc meter for measuring herbage of

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five grassland communities on the eastern shores of Lake St Lucia. The maximum disc meter height recorded by Conlong in his calibrations was 23 cm.

Using Conlong's calibration data together with additional paired data collected on Eastern Shores, Cape Vidal and Sodwana State Forests, a linear regression model relating fuel load (kg ha^{-1}) to disc height (cm) was developed. The precision of the estimates of fuel load of this model are presented. Precision was expressed as the 95% confidence limits (CL) of the estimated fuel load. The relationship between the precision of the estimates of fuel load, and sample size was investigated.

St Lucia lies at the southern end of the Zululand coastal plain, in northern KwaZulu/Natal. The study area was situated in the Eastern Shores, Cape Vidal and Sodwana State Forests. The vegetation was Coastal Forest and Thornveld (Acocks 1975), which consists of a mosaic of forest, savanna, and grassland. The mean annual rainfall at Meersig of 1157 mm is distributed throughout the year, with 60% falling between October and March. Rainfall decreases in a north-westerly direction from 1200 mm at the St Lucia estuary mouth, to 600 mm to the north-west of the lake St Lucia. Mean monthly temperatures at Cape St Lucia range between 25°C and 17°C.

Bransby & Tainton (1977) provide details of the sampling procedure for calibrating a disk pasture meter. Conlong (1986) collected 213 paired disc meter samples on Eastern Shores. Additional paired samples were collected at 11 sites on Eastern Shores, Cape Vidal, and Sodwana State Forests. Sites were selected to achieve a range in fuel loads (using the fire

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records) and in grassland species composition. At each site sampling points were selected to cover the variability in fuel load. Twenty paired data samples were collected per site.

Samples were clipped at ground level using a portable generator and electric shears. The mass of each sample was determined after oven drying the sample at 70°C for at least 48 hours. Six oven dried samples were lost, leaving a total of 214 paired samples.

Using 427 paired calibration samples, a linear regression of fuel load (kg ha^{-1}) on disc height (cm) was established. The model was described in terms of (1) the mean disc height (x) for the calibration, (2) the range of the independent variable, disc height, under which the model is valid, (3) correlation coefficient (r), (4) coefficient of determination (r^2), (5) an F test, and (6) the standard error (SE_y) and 95% (CL) to the predicted fuel load (y) at mean disc height (x).

The effect of the number of disc meter readings (sample size = <u>*m*</u>) on the precision of fuel load estimates, at the mean disc height (x = 11.5 cm) was investigated. Precision was expressed as the SE_y and 95% CL to the estimated fuel load (y). The 95% CL were calculated using the formula in Rayner (1967, p378) or Sokal & Rohlf (1981, p472) (see Hardy & Mentis 1985). The relationship between the 95% CL of estimated fuel load (kg ha^{-1}) and disc height (cm) for different sample sizes (m) were also investigated, using the above equation.

The fitted linear regression equation of fuel load (kg DM ha^{-1}) on disc height (cm) was

$$y = 998.7 + 313.7\,x$$

where

y = fuel load (kg DM ha^{-1}), and

x = disc height (cm).

The linear regression equation is based on the following statistics: correlation coefficient, $r = 0.759$ ($P < 0.001$); coefficient of determination, $r^2 = 0.576$; and F ratio = 578.04 ($P < 0.005$).

The model is valid for fuel load predictions between disc height readings of 1 cm and 58 cm. Estimate of mean fuel load (y), with it's standard error (SE_y), and 95% CL are presented in Table 1. Disc settling heights above 50 cm were recorded in monospecific stands of *Imperata cylindrica*. However these paired samples did not correspond with the highest measured fuel loads.

Table 1. Number of paired samples used in calibrating a disk pasture meter (n), mean calibrated disc height (x), estimated fuel load (y) (kg ha^{-1}), ± standard error (SE_y) of the estimated fuel load (± kg ha^{-1}), and ± 95% confidence limits (CL) for y (± kg ha^{-1}).

n	x	y	± SE_y	± CL
427	11.8	4593	113	222

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Table 2. The effect of number of disc meter readings ($\underline{m}$) on the standard error (SE_y) (± kg ha^{-1}) and 95% confidence limits (CL) (± kg ha^{-1}) of the estimates of fuel load for a mean calibrated disc height of 11.8 cm.

m	SE_y	CL
20	535	1049
40	387	758
60	323	632
80	285	559
100	260	510
120	242	474
140	228	447
160	217	425
180	208	408
200	200	393

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The effect of sample size on the SE_y and the 95% CL (± kg ha^{-1}) of the estimates of fuel load for the mean calibrated disc height, showed that at small sample sizes, the 95% CL were wide e.g. $\underline{m}$ = 20: CL = ± 22.8% of predicted fuel load (Table 2). As sample size was increased there was at first a rapid reduction in the 95% CL (e.g. $\underline{m}$ = 40: CL = ± 16.5%, $\underline{m}$ = 100: CL = ± 11.1%). For greater than 100 readings there was little reduction in the 95% CL (Table 2).

The relationship between the 95% CL of the estimated fuel load and mean disc height for different sample size, illustrates how, for example, to produce a CL ± 10% of 5 000 kg ha^{-1}, 100 disc meter readings would be required (Figure 1). With fuel loads <4 000 kg ha^{-1}, the number of samples required to achieve a CL ± 10% of predicted fuel load is >160. At a mean disc height of greater than 13 cm (>5 000 kg ha^{-1}), the number of samples required to achieve the same precision is <100 (Table 3).

Hardy & Mentis (1985) assessed 29 disc meter calibrations for estimating herbage mass, and reported an average 95% confidence limit of ± 10.5%. This was with 50 readings for calibration. Danckwerts & Trollope (1980) obtained a lower SE (and thus CL), in the calibration of the disc meter in the Eastern Cape. However, as Hardy and Mentis (1985) point out, the higher degree of precision reported by Danckwerts & Trollope (1980), is influenced by the exclusion of "non representative" samples from the data set before obtaining a final regression. Trollope & Potgieter (1986) using a quadrat (4 m^2) to calibrate the disc meter, obtained 95% CL ± 7.24%, with a sample size of 100 in the Kruger National Park. In the present study, the 95% CL were ± 11.1% using the same sample size (Table 2). The high confidence limits at low disc meter readings (<7 cm) are consistent with disc meter calibrations (Danckwerts & Trollope 1980; Hardy & Mentis 1985; Trollope & Potgieter 1986).

The relationship between the 95% CL and sample size was investigated so that a user could make an informed decision as to the number of samples required, for achieving a stated level of precision. The number of disk meter readings required should vary in relation to: (1) purpose of the fuel load estimate, and (2) the precision required. As the mean disk height increased, the number of samples required to achieve a stated precision decreased (Figure 1). The purpose of this model was to aid decision making by providing an estimate of fuel load.

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Decision making can then be formalized using an expert system. Because about 1 500 kg ha^{-1} of fuel is required to carry a fire (Trollope & Potgieter 1986), fuel loads <2 000 kg ha^{-1}, are not critical for the management of fire or the prediction of fire behaviour. In situations of fuel loads >2 000 - 5 000 kg ha^{-1}, 100 samples should be taken. However, to achieve a stated precision level, sample size may need to be increased above 100 (Table 3) (Figure 1). However, the increased precision per unit sampling effort, above 100 readings, would be poorly rewarded (see Bransby & Tainton 1977; Hardy & Mentis 1985; Trollope & Potgieter 1986). For fuel loads >5 000 kg ha^{-1}, the number of samples can be reduced to less than 100. The average fuel load for the calibration was ± 5 000 kg ha^{-1}, therefore 100 samples should usually suffice. If the linear regression model developed is to be used for estimating fuel load, for predicting fire behaviour (see Trollope & Potgieter 1985; Trollope 1992), then the same guidelines should be used.

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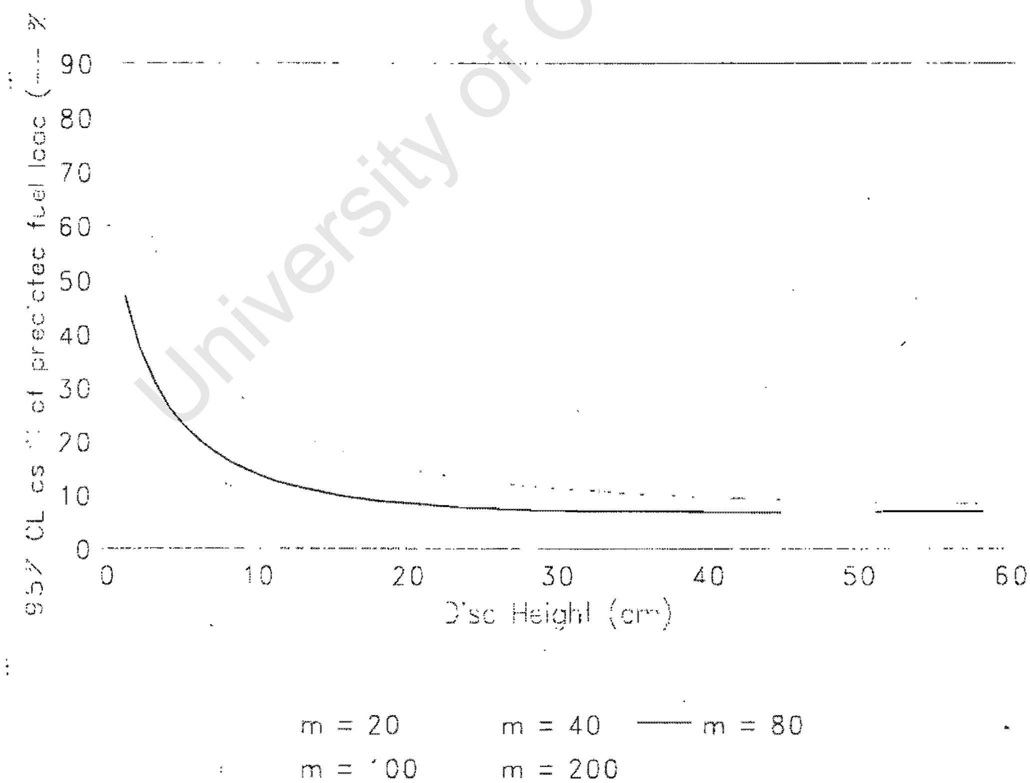

Figure 1. The relationship between the 95% confidence limits as a percentage of estimated fuel load (± %), and disc height (cm) for different sample sizes (m).

If this regression model is not precise enough to estimate grass fuel load, from the mean settling height of a disc, then alternative models should be developed and tested within the limits of precision deemed and specified important to the user.

Table 3. The sample size required to produce a fuel load estimate (kg ha^{-1}), with a 95% confidence limit ± 10 % of predicted fuel load, for various disc pasture readings (cm).

Mean Disc Height	Estimated fuel load	Sample Size (*m*)
9	3 822	220
10	4 135	160
11	4 449	140
12	4 763	120
13	5 077	100
14	5 391	80
15	5 704	80
17	6 332	60
18	6 645	60
22	7 900	40
23	8 214	40
33	11 351	20

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Chapter 3

Calibrating a dry-weight rank and a weight-estimate method for grass fuel loads in Pilanesberg National Park, South Africa

B.H. Brockett and C.L. Parr

(submitted to African Journal of Ecology)

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Abstract

A dry-weight rank (DWR), and a weight-estimate method were calibrated to measure herbage mass in Pilanesberg National Park, South Africa. The DWR method was calibrated for three landscapes (bottomland, pediments, and hills). Calibrations were conducted using a circular quadrat (1 m^2). Actual fuel load for the clipped quadrats varied from 20,287 kg ha^{-1}, to 844 kg ha^{-1}. For the DWR method a linear regression was developed for each landscape. For the weight-estimate method biomass was under-estimated by on average 29.6 ±41.6%. The percentage deviation decreased as fuel load increased. The weight-estimate technique had a lower standard error of the mean, than the actual fuel load measurements. Therefore the weight-estimate method had a higher precision than the actual fuel load. Increasing sample size will reduce variance. Each estimator would need to be calibrated, and this should also be checked for consistency between sampling, or monitoring periods.

Introduction

The determination of fuel load is often important in facilitating consistent burning decision-making. This is because a fire management decision-support system, for example an expert system, might require an estimate of grass fuel load (Bailey *et al.* 1993; Brockett, Biggs & van Wilgen 2001). Most of the fuels in grasslands and savannas comprise standing grass (Trollope & Potgieter 1985, 1986), and therefore an estimate of fuel load involves the measurement of herbage mass per unit area. A cost, or time efficient and repeatable technique to measure herbage biomass is required.

There are a number of techniques available to measure herbage biomass. These have been classified by Pieper (1988) as: (1) direct harvest methods, (2) estimation techniques, and (3) indirect methods. Estimation techniques and indirect methods are classified as non-destructive techniques. Direct harvesting is a destructive sampling procedure, and could for example involve clipping herbage in a quadrat, oven-drying, and weighing the material (Milner & Hughes 1968; Pieper 1988). These methods are often used for research purposes where accurate measures are required. However, such procedures are time consuming, relatively expensive (Pieper 1988), and would therefore be unsuitable to measuring fuel loads in extensive rangelands. Estimation techniques usually involve a double sampling procedure for calibration purposes. Such methods are designed to provide a rapid assessment of herbage mass for extensive rangeland surveys, and can also be used for research purposes. Examples include the weight-estimate method estimates (Goebel 1955), and dry-weight ranking methods (DWR) (Francis, van Dyne, and Williams 1979). An indirect method would be where precipitation is used as an independent variable to predict end-of-season biomass (see Deshmukh 1984; Snyman & Fouche 1993; Snyman 1994), and is usually applied at a broader scale. A further example of an indirect method (using dimentional analysis) is the disc pasture meter (Bransby, Matches & Krause 1977; Brandsby & Tainton 1977).

Deciding which technique to use requires that a number of steps be followed. The technique needs to be selected in relation to the stated operational objectives, and this requires an evaluation of the composition and functioning of the ecosystem. For example a technique may be suitable in one area (or veld type), but unsuitable in another. When selecting a technique it is important to bear in mind the limitations of each method. The required efficiency is related to the sensitivity (or precision) with which fuel load needs to be measured, whilst also considering staffing, and budget costs. Additional factors to be considered would be the experience of the staff implementing the monitoring program, and the time (and budget) constraints. Some techniques (e.g. a weight-estimate method) may require observers with considerable experience at estimating herbage biomass. If such observers are not available, or would be too costly, then an alternative technique needs to be selected. Most techniques require calibration and / or that a pilot survey be conducted before they can be used in specific situations (Pieper 1988). A pilot study would help establish the relative efficiency of the technique.

A disc pasture meter has been calibrated to estimate fuel loads in the Kruger National Park (Trollope & Potgieter 1986), and on the Zululand coastal plain (Brockett 1996). However, a disc meter may be unsuitable to estimate fuel loads on steep and rocky ground. This could be due to rocks affecting the settling height of the disk (cf. Pilanesberg National Park).

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In this study calibration scales were developed for a DWR method, and a weight-estimate method for fuel load were conducted, and compared with actual fuel loads derived from direct harvesting.

Study area

The study was undertaken in Pilanesberg National Park (PNP), South Africa. PNP is circular in shape with a 25-km diameter, and approximately 48,000 ha in size. It is an isolated, oval series of concentric hills and valleys composed of a unique suite of alkaline volcanic rocks. The landscape is dominated by the ranges of high rounded hills interrupted at intervals by transverse valleys or water-gaps. Some of the hill crests retain flat remnants from the original volcanic plateau, fractured initially by ring subsidence and these lines subsequently incised by stream erosion. A feature of the valley-hill profile are the well developed pediments formed chiefly of latosols, many of which have an impervious subsoil of clay or ferricrete resulting in seasonally waterlogged conditions. Valley floors meet the pediments on an even curve and exhibit a shallow U- or V-shape in profile (Farrell, van Riet & Tinley 1978). The savanna vegetation (which receives a mean annual rainfall of 630 mm) falls in Acocks's (1975) Sour Bushveld. Sourveld occurs in areas with high water supply, soils with a low base status, and where carbon assimilation is high relative to nutrient supply (Ellery, Scholes & Scholes 1995). Intrusive rocks form the hill cores; their outcropping being indicated by the denser cover of macrophyllous trees (notably *Combretum* species). Microphyllous bushes (*Acacia mellifera* and *A. tortilis*) occur on the clayey bottomlands, and extensive grassland pediments occur which are underlain by an impervious laterite layer. The area supports diverse populations of large herbivores (including many species of antelope, zebra, giraffe,

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elephant, white rhinoceros, hippopotamus, and warthog), and predators (lion, cheetah, wild dog, and leopard).

Procedure

Sampling method and land-facet type descriptions

When sampling using a quadrat there are a number of decisions that need to be made. These include the size, shape and number of quadrats to be used. One concern with quadrat size and shape is the number of borderline decisions which are directly related to the perimeter of the quadrat, and a circular quadrat has the lowest perimeter: area ratio (Pieper 1988). In addition statistical and time efficiency (Pieper 1988), and practicalities (Critchley & Poulton 1998), should also be used in selecting an 'optimum' quadrat size. Taking cognisance of the above a circular quadrat of area 1 m^2 was selected.

For the purpose of developing suitable scales to estimating fuel load, PNP was classified into three land-facet types. These were: (1) bottomlands, (2) pediments, and (3) hills. These land-facets differ in their geomorphology and pedology, and hence in grass layer species composition, cover and biomass. These follow the land-facet classification of Farrell, van Riet & Tinley (1978) used in the planning of PNP. In their classification bottomlands were classified as watercourses and valley floors; pediments as either wet phase, wet and shallow phase or dry phase pediments; and hills as steep and rocky ground, or shallow and rocky ground.

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1.*Bottomlands*

Using van Wilgen & Scholes (1997) these land-facets would be classified as arid fertile savannas. The fine fuels are patchily distributed. The main grasses contributing to fuel load are: *Panicum coloratum, P. maximum*, *Enneapogon cenchroides*, *Eragrostis rigidior, Fingerhuthia africana*, and *Urochloa mosambicensis*. Other grasses are *Aristida congesta, Tragus berteronianus*, and *Cynodon dactylon*. Trees are unimportant in carrying fire, but provide woody material for smouldering combustion, and dung adds fuel for smouldering combustion (van Wilgen & Scholes 1997).

2. *Pediments*

These land-facets would be classified as moist infertile savannas by van Wilgen & Scholes (1997), and are open grasslands on impervious laterite, with trees establishing around or on termite mounds, or where drainage occurs through the ferricrete. This classification includes the pediment savannas classification by Farrel, van Riet, & Tinley (1978), which are far less extensive and generally confined to the perimeter of the Pilanesberg complex. Fuels are fairly continuous, and trees are unimportant in carrying fire, but provide woody material for smouldering combustion (van Wilgen & Scholes 1997). Grasses that contribute to fuel load are *Cymbopgon excavatus, Digitaria eriantha, Eragrostis superba, Elionurus muticus, Heteropogon contortus*, *Hyparrhenia filipendula*, *H. dissoluta*, *Trachypogon spicatus*, and *Themeda triandra.*

3. *Hills*

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This land-facet includes two categories of north-facing (xerocline) and south-facing (mesocline) slopes. Xerocline slopes are more fertile (or sweeter), than the south-facing slopes (sour). This results in a greater fuel load accumulations on south-facing slopes. Grasses contributing to fuel load on mesocline slopes would be: *Setaria lindenbergiana*, *Eustachys paspaloides*, *Themeda triandra*, *Heteropogon contortus*, *Digitaria eriantha*, and *Diheteropogon amplectens*, and on north-facing slopes: *Loudetia simplex*, *Chrysopogon serrulatus*, and *Heteropogon contortus*.

Data collection

Paired samples of fuel rank and actual fuel load, were selected to span the range of fuel loads in each of the three land-facets. At each site the following steps were undertaken: (1) a circular quadrat was settled over the sample, (2) herbage biomass within the quadrat was given a ranking using a suitable scale, and for some of the samples the herbage biomass was visually estimated (kg ha^{-1}), (3) a photograph was taken of the fuel in the quadrat, (4) standing crop within the quadrat was then clipped at ground level using a pair of garden shears, and (5) the mass of the sample was determined after oven-drying at 70°C for at least 48 hours. The procedure was repeated at the remaining sites.

Calibrations for DWR method

Samples were ranked using the field-developed scales for each land-facet type. A reference album to be used for future assessments was created using the photos taken at each site. Using the dry-weight mass of each sample the field-developed scales were adjusted so that

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the relationship with dry matter per ha was approximately linear. For each land-facet type a fitted linear regression equation of fuel load (kg ha^{-1}) on rank scale, was developed.

Weight-estimate method for herbage biomass

To evaluate a weight-estimate technique I visually estimated fuel loads for fifteen of the quadrat samples used for the dry-weight calibration, and compared these visual estimates (kg ha^{-1}), with actual measured fuel load (kg ha^{-1}).

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Results

Calibrations for DWR method

Calibration scales were developed for each land-facet as follows: a 6-point scale for the bottomland, a ten-point scale for the pediments, and a 5-point scale for the hills (Figure 1). The highest fuel load was measured on the pediments, and the lowest on the bottomlands (Table 1). Consequently the range in the actual fuel load was highest on the pediments, and lowest on the bottomland (Table 1). The linear regressions constants, together with their coefficients of determination for the three land-facet types, are presented in Table 2.

Weight-estimate method for herbage biomass

Visual estimates of herbage biomass from the 15 quadrat samples (comprising members from the three land-facet types) were plotted against the actual biomass, together with the line of perfect agreement (Figure 2). Thirteen samples were under-estimated, and the remaining were over-estimated. Estimated fuel load was on average 3,322 ± 4131.6 kg ha^{-1}, compared with 7,122 ± 6178 kg ha^{-1} using the direct method (Table 3). The deviation from the actual fuel load was -29.6 ± 41.6%. The highest under-estimate and over-estimate were -11,787 kg ha^{-1}, and +1,816 kg ha^{-1} respectively. Fuels loads less than 1,500 kg ha^{-1} were underestimated, >1,500-2,000 kg ha^{-1} were over-estimated, and greater than 2,000 kg ha were underestimated (Figure 3). The percentage deviation decreased as fuel load increased (Figure 3). Pieper (1988) mentions that estimators tend to overestimate quadrats with low biomass, and to underestimate biomass of those quadrats with high weight.

Discussion & Conclusions

Linear regression equations for a DWR calibrations were developed for three land-facets in PNP, with coefficients of determinations of ≈0.9. Precision of sampling for herbage biomass (using the standard error of the mean) for the weight-estimated fuel load was lower than for the actual fuel load (Table 3). Therefore more precise estimates are produced using a weight-estimate method than using actual (or direct) methods. To reduce variance (and to increase precision) for the actual fuel load measurements, the number of samples could be increased. The same could be done with the estimated quadrats, however, caution should be exercised because of reduced variances on estimated quadrats (Pieper 1988). A linear regression could be fitted to the actual fuel load on estimated fuel load, and this could be used to adjust estimates. However, this would only work if an estimators estimated either consistently high or low. Each estimator would therefore need to be calibrated prior to commencing sampling, and also need to be checked for consistency between sampling, or monitoring activities. In this case the regression has no general application and hence is not presented.

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A pilot study should be conducted using the method(s) selected, and with specific observers, in order to evaluate the number of samples required per plot to achieve a stated level of precision and within acceptable sampling costs.

Table 1. Dry-weight ranking scales developed for three land-facets in Pilanesberg National Park, with maximum fuel load (Max) (kg ha^{-1}), minimum fuel load (Min) (kg ha^{-1}), range (kg ha^{-1}), and number of samples (n).

Land-facet	Bottomland	Hill	Pediment
Scale	1 - 6	1 - 5	1 - 10
Max	8857	16093	20287
Min	844	1581	1029
Range	8013	14512	19258
n	6	5	10

Table 2. Fitted linear regressions x coefficients and constants for fuel load (kg ha^{-1}) on dry-weight ranking, with degree of freedom (df), and coefficient of determination (r^2), for three land-facets types in Pilanesberg National Park.

Land-facet	Bottomland	Pediment	Hill
X coefficient	0.00062	0.00044	0.00026
Constant	0.668	1.36484	0.78689
df	4	8	3
r^2	0.991	0.968	0.989

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Table 3. Comparison between the mean (X), standard deviation (SD), and standard error (SE), and sample size (n), for a direct method and a weight-estimate method to measure herbage biomass.

Technique	Direct method	Weight-estimate method
X	7122	3800
SD	6177.9	2226.5
SE	441	159
n	15	15

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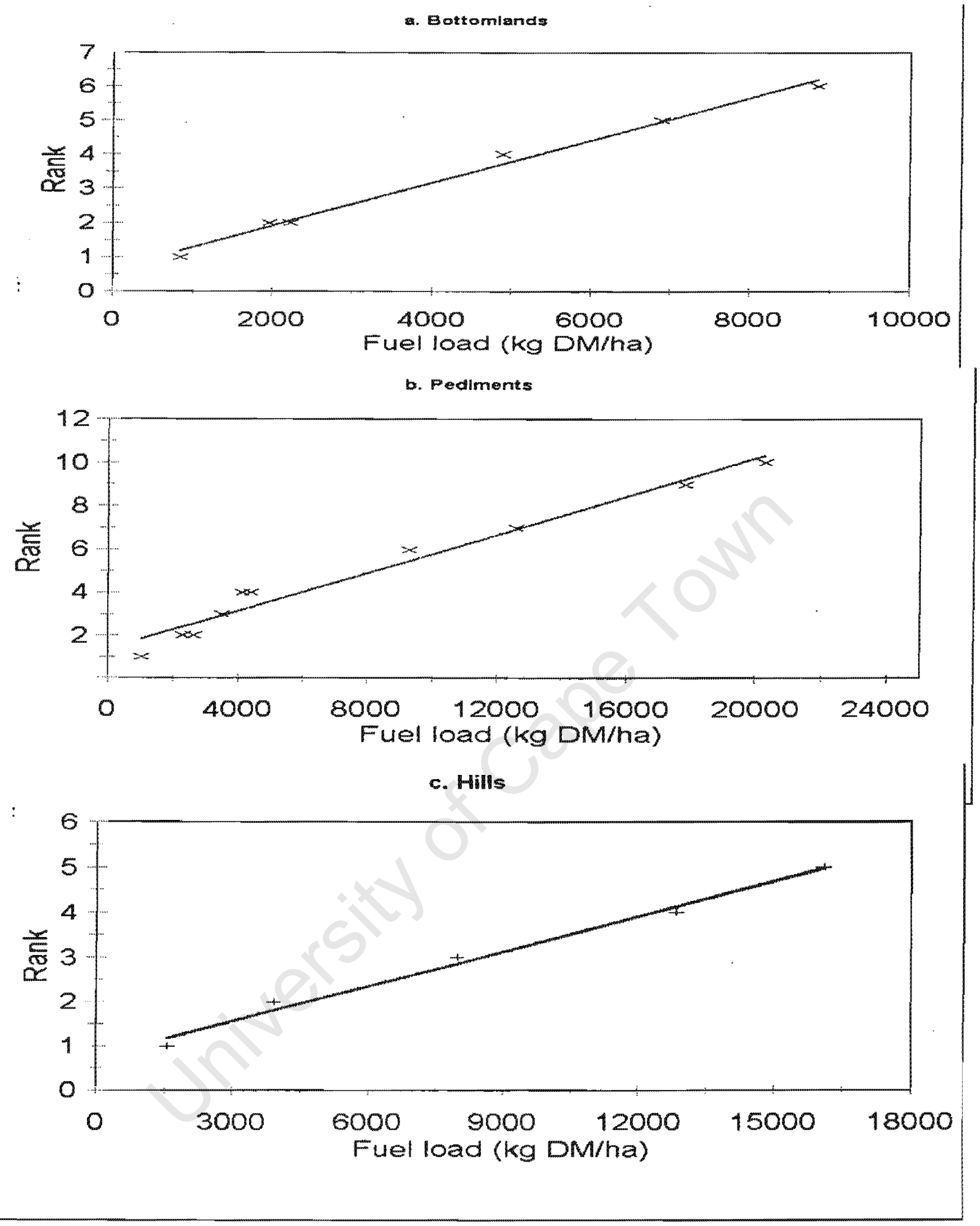

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Figure 1. Dry-weight rank calibration scales for herbage biomass estimation for three land-facets in Pilanesberg National Park, South Africa: a bottomlands, b. pediments, and c. hills.

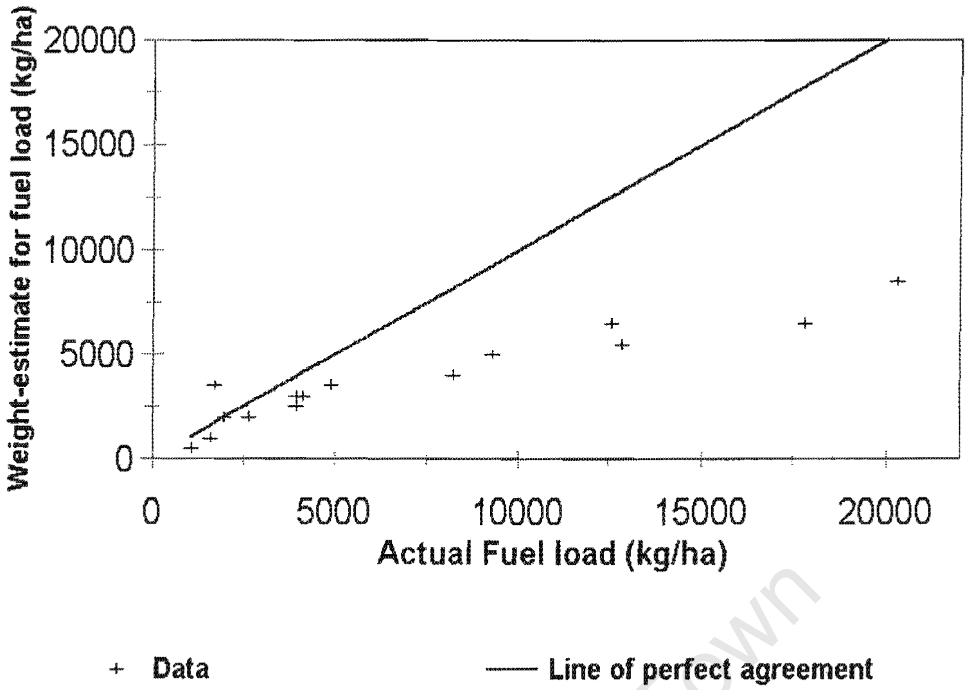

Figure 2. Direct fuel loads compared with weight-estimate for fuel load (kg ha^{-1}) in Pilanesberg National Park.

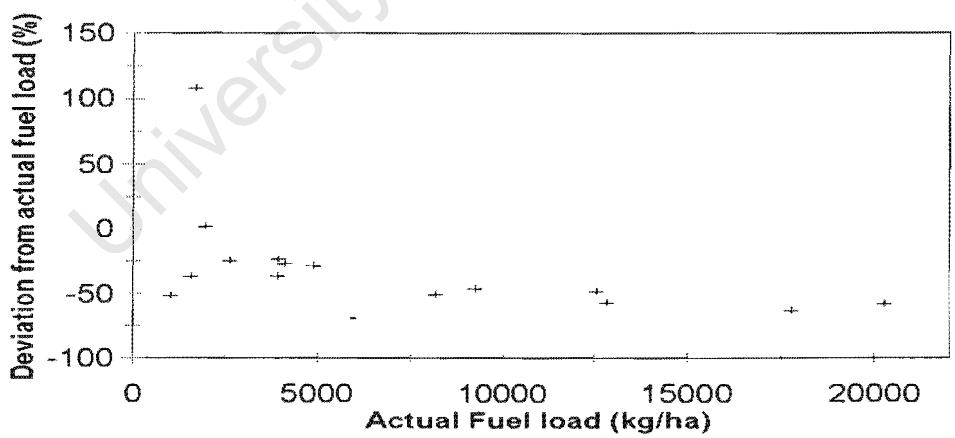

Figure 3. Deviations from actual fuel load measurements for weight-estimates (kg ha^{-1}), expressed as a percentage.

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Chapter 4

Patch-mosaic burning: a new paradigm for savanna fire management in protected areas?

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Abstract

The shift in ecological thinking, from equilibrium to non-equilibrium processes has been accompanied by a move to encourage heterogeneity rather than homogeneity in landscapes. Spatial and temporal heterogeneity is thought to be a major source of biotic diversity, and disturbances such as fire, producing heterogeneity are now recognised as being important. A patch-mosaic system of burning is based on the premise that fire pattern is a surrogate for diversity, and produces a range of patches in the landscape with unique patch characteristics and fire histories. A patch-mosaic system of burning is supported historically and empirically through field studies. However, there is a need for more research into the effects of various aspects of patch and fire variables on biotic diversity, especially in savannas where our understanding is particularly poor. Landscape-scale experiments, like those to be established in the Kruger National Park, South Africa are necessary to test different burning regimes. Challenges to patch-mosaic burning include determining the 'natural' range of variation for

fire parameters, implementing random ignitions and cost-effective fire scar mapping at the appropriate resolution. An adaptive management approach should be adopted to deal with the ignorance and uncertainties that characterise the management of savanna ecosystems. This should be applied with both modelling and monitoring as key elements in this process.

Introduction

Fire, a major disturbance force, is regarded as a crucial determinant of savanna vegetation, and acts to modify broad patterns set primarily by rainfall and edaphic factors, and hence plays a role in determining the structure and function of these systems (Walker 1987; Scholes & Walker 1993; Scholes & Archer 1997). Despite the importance of fire in savanna systems being well established, fire management in protected areas remains a contentious issue, with considerable debate as to the most appropriate burning system. In most protected areas the main goal of conservation is the maintenance of biotic diversity and ecosystem functioning. A manager of a protected area faces a number of choices, and these decisions will result in different outcomes. Regarding fire management this is no different. For example a manager could aim to implement rigid fire regimes, or could vary the seasonality, and the frequency (by varying the number of ignitions per year, and the conditions under which they are ignited), and hence also the range in fire intensities. Thus through manipulation of fire variables it is possible to change spatial and temporal diversity, and determine whether biotic diversity is maintained. Decisions made to achieve objectives thus determine the outcomes produced. Where the maintenance of biotic diversity is the primary objective of a protected

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area, it follows that the fire management should be aimed at promoting a diverse fire regime (Mentis & Bailey 1990; Scholes & Walker 1993).

A patch-mosaic burning system was developed to maximise the benefits of a diverse fire regime for savannas, and is based on the premise that fire pattern is a surrogate for biodiversity (Brockett *et al.* 2001). Using such a system the proportion of park burnt per year is a function of the grass fuel loads. Differences in rainfall both annually, seasonally and on shorter time scales (e.g. weekly) affect fuel characteristics e.g. distribution and compaction, fuel moisture, and fuel load. A patch-mosaic fire regime reflects this variation with randomly ignited fires spread over the seasons, and allowed to burn freely, unless a risk is posed to buildings or other structures, or the area to be burnt has been exceeded. This results in variation in size, number, location, seasonality and intensity of fires, and total area burnt per year.

This paper aims to examine the theory behind a patch-mosaic system, consider the empirical evidence supporting it and highlight research needs. Guidelines are offered to its implementation through adaptive management, with emphasis placed on monitoring and modelling.

The emergent heterogeneity paradigm

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Previously much ecological thinking assumed that systems had a uniform structure, and that mechanisms that structured them were also uniform in space and time (Pickett & Rogers 1997). As a result homogeneity in the management of savannas was encouraged. Such management is regarded as a command-and-control culture, since the focus is on applying fixed prescriptions (Holling & Meffe 1996; Andersen 1999). Following major conceptual shifts in the understanding of savanna structure and functioning, the importance of inherent variability, a key characteristic in these environments, was recognised (importance of flux in addition to stability).

There remains much debate regarding the use of equilibrium theory, non-equilibrium theory, and disequilibrium concepts to explain savanna functioning (Frost *et al.* 1986; Westoby *et al.* 1989; Mentis & Bailey 1990; Scholes & Walker 1993; Illius & O'Connor 1999). Savanna are interpreted as event-driven systems (Walker *et al.* 1986), and the multiple stable state hypothesis for savannas proposes there may be two (or more) equilibria in these systems (Dublin *et al.* 1990). Disturbance events (such as fire, drought, rainfall, and herbivory) become the drivers necessary to overcome inertia between states (Mentis & Bailey 1990), and can affect a system's capacity to withstand impacts that may cause it to shift from one state to another, i.e. its resilience. Where periodic disturbance is essential to the resilience of a system, disruption of this disturbance regime can induce a loss of resilience (Holling 1981; Scholes & Walker 1993; Bullock & Webb 1995; Perrings & Walker 1997). Changes to savanna fire disturbance regimes via a prescribed block burning system, or via fire suppression (i.e. a command and control strategies) result in reduced spatial heterogeneity. This results in simplification of the system, and consequently could reduce system resilience (Mentis & Bailey 1990; Scholes & Walker 1993; Holling & Meffe 1996).

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Natural systems are typically patchy (Weins 1976), hence, it is now recognised that spatial and temporal heterogeneity, and complexity are crucial elements in the functioning of ecosystems (Christensen 1997). With the paradigm shift from homogeneity and stability thinking, to patchiness in ecology, attention has focussed on the development of patch theory, heterogeneity and patch dynamics (Weins 1997).

Variability, heterogeneity and patchiness are key elements in all system states, and as such, the effects of hierarchical, and dynamic patterns of heterogeneity are significant. Environmental patchiness (habitat and within-habitat diversity) is a major source of biotic diversity (Braithwaite 1996; Pickett & Rogers 1997). Huston (1994) describes how patchiness in pattern creates heterogeneity in resource availability, which then provides an array of opportunities for colonisation and survival. It is this existence of opportunities that maintains diversity. Individual patches support different species, or individual species require multiple habitat patch types. Species may breed, feed and shelter in different patches. Law & Dickman (1998) stress this is the case for many species of vertebrate that require multiple habitats to ensure different resources at different stages of their life-cycles. The same is also true for invertebrates (Usher & Jefferson 1991). Conservation of species requiring multiple habitats is thus enhanced in a mosaic environment.

Disturbances, such as fire, are important mechanisms for producing (and maintaining) spatial heterogeneity (Schwilk *et al.* 1997). Fire therefore plays a role in structuring ecological systems by producing a spatio-temporal mosaic of patches at different successional stages (Pickett & White 1985; Turner *et al.* 1994; Moloney & Levin 1996). It has been suggested

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that these diverse conditions may prevent community domination by one or few species (Bond & Van Wilgen 1996), and allow for the persistence of fire-sensitive species in a community subject to regular fires (Frost 1984). Forman (1995) considers a large area containing many patches in various stages of development as a 'shifting mosaic'. Patches will appear and disappear with each subsequent fire, a process that Baker (1992) refers to as 'patch population dynamics'. The character of such landscapes is determined by frequency, intensity, and spatial extent of disturbances creating patches, as well as the rate and nature of processes that result in patch succession (Christensen 1993).

The relationships between patch characteristics, fire characteristics, and landscape dynamics are complex and inter-related. Patch characteristics influence fire characteristics, which in turn influence patch characteristics in a complicated cybernetic system. Individual patches and fire events are also linked and affected by external factors such as climate. Climatic factors influencing fire behaviour vary in size and effects on a scale from months, to years to centuries. The periodicity of fire events is only partially determined by successional changes in the vegetation structure of patches (Binkley *et al.* 1993). In addition to successional changes in vegetation structure and composition, the likelihood of a fire, and patch characteristics are influenced by site history, position of a patch in the landscape, fuel load and fuel type, as well as topographical, geological and hydrological factors (Christensen 1993). Climate, ignition events, and the character of adjacent patches influence the likelihood of fire in a particular patch. Other determinants of vegetation patterns are the life-history characteristics of constituent species, previous site history, fire intensity (or severity), landscape mosaic, and the seasonality of fires. These factors are likely to result in shifts in the frequency distribution of different landscape patch types over time. Fire frequency may

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be regarded as a function of the available fuel (Van Wilgen & Scholes 1997), however this is not a simple relationship. This may be due to fuel properties, which in turn may affect variation in average fire-return intervals (Green 1982 cited by Christensen 1993). Also there is sometimes a spatial dependency between patches in a landscape, and this varies widely as a function of landscape structure, terrain, frequency of ignition events, and climatic conditions. These factors could result in variation in the fire parameters e.g. fire return periods. However, fragmentation of the landscape (for example road construction) might alter patterns of fire spread (Christensen 1993), and hence affect fire regimes.

Changes to 'natural' disturbance regimes constitute one of the major ways that humans have altered ecosystems, and thus the biological diversity that occurs within them (Bond & Van Wilgen 1996; White & Harrod 1997). Gill & McCarthy (1998) describe how there is growing concern (e.g. Keith & Bradstock 1994; Cary & Morrison 1995; Morrison *et al.* 1995) that in some Australian ecosystems regularity of burning (frequency, season, and spatial extent) could have adverse ecological effects. Hence fire management regimes that result in homogenisation of habitats should be avoided (Law & Dickman 1998). From a philosophical viewpoint, if nature is variable, any fire system aiming to mimic nature should incorporate this variability too. Gill & McCarthy (1998) and McLoughlin (1998) advocate that when managing ecosystems, the natural range of variation should be taken as the basis for fire management. Mentis & Bailey (1990) stress that fire parameters, such as fire interval, should be regarded as vectors (not scalars), where there is a frequency distribution of occurrence. In this way, ecosystem management acknowledges inherent uncertainty and ignorance, and attempts to accommodate it (Holling 1978; Christensen 1997). A burning regime that promotes heterogeneity and patchiness in the landscape is therefore preferable (Saxon 1984; Mentis & Bailey 1990; Russell-Smith *et al.* 1997).

Unfortunately, fire management in some Southern African savanna parks and in Australia land management agencies is still dominated by a command and control culture (Mentis & Bailey 1990; Anderson 1999). An alternative is offered in the form of a patch-mosaic burning system where fire parameters are varied to create a mosaic of patches representative of a range of fire histories which generates heterogeneity within the landscape (Brockett *et al. 2001*). Such a burning system is based on the premise, that patchiness is a major source of diversity. It is assumed that if fire pattern is regarded as a surrogate for biodiversity, then a diverse fire regime, produced by the patch-mosaic system, should maintain biotic diversity.

Support for a patch-burn approach: a summary of historical and empirical research

Mosaics created by fire are thought to reflect historical burning strategies. Evidence from Australia, and to a lesser extent from southern Africa, suggests that the traditional burning regimes complemented the lightning fire regime. In southern African savannas, it is thought that early Man had been using fire for 1.5 million years, and the controlled use of fire for hunting and domestic purposes has occurred as early as the Stone Age, 250 000 BP (Brain & Sillen 1988). These lightning fires would generally occur during the rainy season (October to March). Native groups such as the Bushmen/San, !Kung, Swazi, and Zulu in southern Africa, and Aborigines in Australia used fire extensively throughout the year (early dry season to late

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dry season/ spring) (Hall 1984; Braithwaite 1991; Braithwaite & Roberts 1995; Fensham 1997).

Early dry season fires were usually patchy and small in extent, increasing in size and intensity as grasses cured with the progression of the season. Small fires result in mosaic landscapes with patches of differing composition and structure, whereas large fires tend to homogenise a landscape (Binkley *et al.* 1993). Thus a traditional fire regime results in unburnt, early burnt and late burnt patches creating a landscape with a fine-grained mosaic of different seral stages (Short & Turner 1994; van Wilgen & Scholes 1997). Further to the benefits of enhanced habitat diversity, an array of patch types can act as a natural firebreak, breaking up the front of large wildfires (Minnich 1983; Saxon 1984).

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The decline and extinction of medium-sized mammals in Australia is thought to be related to a change in fire regime (Short & Turner 1994). With the movement of Aborigines out of desert areas, and the adoption of more urban lifestyles, the traditional fire regime previously imposed is being lost. It is thought that the fine-grained mosaic of different vegetation types that these species require, has not been maintained. Braithwaite (1991) advocates that in these areas fire management strategies should be based on traditional Aboriginal burning.

Botanical studies and modelling have revealed that static fire intervals may be detrimental (Keith & Bradstock 1994). Morrison *et al.* (1995) provide empirical evidence for variable fire intervals. Following analysis of plant-species compositions and recent fire histories, they conclude that it is likely that variation of inter-fire intervals through time is primarily responsible for the maintenance of biodiversity. In southern Africa it is thought that temporal variability in fire regimes promotes grass-tree co-existence, and hence the structural diversity of savanna systems (Higgins *et al.* 2000). In North American forests too, it was found that the use of mean fire interval between fires (mean fire return period) could be quite misleading in predicting ecological response (Clark 1996). Gill and McCarthy (1998) conclude that variability in fire intervals in nature is inevitable, and indeed desirable in prescribed burning plans where the aim is to conserve biodiversity. They recommend the adoption of a variable system of fire application along with targeted monitoring.

Faunal studies too indicate that at the landscape level, the ideal fire regime for the conservation of a wide range of species is one where there are a variety of burns producing a mosaic of patches. Investigating the question of whether fire patterns could be used as a surrogate for biotic diversity, Parr (1999) studied the effects of post-fire fuel age and fire frequency on ant diversity in Pilanesberg National Park, South Africa. This study demonstrated that both time since fire and frequency of burning affected ant community composition. Post-fire fuel age and fire return period are only two variables considered in this study, and it is important to consider that there are many factors (e.g. intensity and seasonality of fire, patch size, shape and adjacent patch characteristics) not tested in this study that could contribute to the maintenance of diversity.

We conducted a brief scoping study to establish which aspects of fire (e.g. frequency, intensity, season), and patch parameters (e.g. post-fire fuel age and patch size) have been researched. It was clear that not all fire and patch parameters were covered in these studies.

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Furthermore, it was evident that certain taxa have been favoured for research: birds (Mentis & Bigalke 1981; Woinarski 1990; Brooker & Rowley 1991), reptiles (Braithwaite 1987; Lunney *et al.* 1991; Trainor & Woinarski 1994) and invertebrates, e.g. ants (Andersen 1991; De Kock *et al.* 1992; York 1994; New *et al.*1996; Vanderwoude *et al.* 1997; Parr 1999) and grasshoppers (Chambers & Samways 1998). Most of these studies were based on changes in species composition and diversity (richness and abundance). This illustrates the paucity of information on faunal responses to burning, and serves to emphasise that there remains a pressing need for more studies investigating mammal, bird, reptile and invertebrate responses to burning, and the potential importance of variability in fire parameters and patch characteristics. Many fire studies are conducted opportunistically (e.g. following a wildfire through a fire exclosure), and there has been little research into the effects of extreme fire intensities which periodically stress landscape systems. As much of the research and quantitative evidence supporting the need to incorporate variability and heterogeneity originates in Australia, where fire ecology and management has perhaps advanced more rapidly than in southern Africa, there is a need to concentrate studies in southern African, and especially in savanna areas where little previous research has been conducted.

A key problem with many faunal studies is that the issue of scale is not addressed. Many fire studies are undertaken on small experimental plots where fires are often different in character to those occurring in larger, continuous areas. The problems associated with small plot size have been recognised (e.g. Gill *et al.* 1990; Carpenter 1996; Andersen *et al.* 1998) but still these studies persist. Although these studies are useful for improving our understanding, and may suggest that variability in burns is desirable, there is still a need for more objective calibration. This can only really be done by taking research a step further, and by testing

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these problems in the real environment and on a much larger landscape-scale. Hence the need for landscape-scale fire experiments (Anderson *et al.* 1998).

Implementing a patch-mosaic burning system and comparisons with other fire management systems

Determining the 'natural' range of variation of fire parameters is problematic. Acknowledging that fire intervals in nature occur with a varying frequency, Gill and McCarthy (1998) consider evidence for determining probability distributions of fire intervals. This is a complex procedure, and may be possible for some areas once life history traits of specific indicator species have been identified. However, determining specific indicator species is likely to require much research. Limits within which we should be working for specific fire variables (e.g. intensity, spatial extent and seasonality), need also to be determined and combined. Fire management policy should therefore be implemented to include as wide a range of variability as possible under the constraints imposed by alternative objectives. The variability that can be incorporated are often set not by the ecosystem but by the consumptive and/or non-consumptive use (e.g. tourism) of the protected area. An additional limit to implementing a wide range of variation in the fire parameters is the size of the protected area. The more extensive the area then the wider the range of variation in fire parameters that can be implemented.

Other burning systems may also produce patchiness and heterogeneity in the landscape. Lightning ignited fires will undoubtedly produce fires that are patchy in nature, however this

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patchiness differs fundamentally from that produced by a patch-mosaic system as fire occurrence is highly skewed towards the summer months. With a patch-mosaic system fires occur throughout the year, with greater variation in patch size as a result of the range of different conditions under which the fires are applied.

Under a prescribed block-burning system the area is divided into units for treatment (i.e. blocks or compartments), and these are burnt in a predetermined year according to schedule, and under controlled conditions, to create an equally-sized veld age mosaic (Seydack 1992). Such a system only promotes heterogeneity in the sense that the burnt areas or blocks may form a fixed mosaic. Blocks are not analogous to patches, as the mosaic is regular and uniform, both in space and in time. The mosaic produced by block burning therefore differs substantially to that produced by a patch-mosaic burning system as there is little variance in block size (hence fire size), or in the season and frequency of burn (e.g. Etosha National Park, Namibia (Stander *et al.* 1993; Du Plessis 1997)). Finally it should be noted that all burning systems produce fine scale patchiness to a greater or lesser degree e.g. small clumps of grass remaining unburnt after a fire.

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There is a tendency to liken patches in a landscape to 'islands' in keeping with the equilibrium theory of island biogeography (MacArthur & Wilson 1967). When we consider the range of patch sizes in a landscape we might try to determine at which size threshold particular species drop in or out in an effort to ascertain the minimum (or maximum) patch size that is desirable. Thus, in this way we seem to be regarding patches as islands. How much like islands are patches? With the shifting dynamic nature of the landscape under a patch-mosaic system, patches are likely to be very different to real and fragmented habitat islands. Patches are often not isolated, distinct, tangible entities in the landscape with some obvious edge. There is much scope for research into how these patches function, inter-relate, are created and for how long they exist in the landscape.

A conceptual model was developed to illustrate how fire size may be determined by fuel characteristics (e.g. fuel size, distribution, compaction, and quantity), terrain, and the number of ignitions (Fig. 1). Fire frequency is a function of the: (1) fuel characteristics (e.g., the post-fire fuel accumulation rates), (2) terrain, (3) number of ignitions, and (4) season of burning. Some of these factors influence the rate of spread of a fire, and hence fireline intensities. Variation in fireline intensities results in a variation in fire size (Fig. 1). By applying fires in autumn (before the fuels are cured), the fireline intensities would be reduced, and consequently also the fire size, and this would lead to many small fires. However, if fires were applied from late winter to early spring, then the mean fire intensity would increase, as would the mean fire size. Therefore, in applying a patch-mosaic burning system there are a number of choices to make with regard to the: (1) number of ignitions per month (i.e. the seasonal distribution), and number per year, and (2) spatial extent of fires.

On the basis of present knowledge some choices may lead to unknown outcomes. In addition there are a number of practical issues to be resolved. Some of the challenges are not unique to a patch-mosaic system e.g. zonation. Practical issues to be addressed include the organisation of the ignition points; how a random design would function, and constraints imposed on management due to the zonation of the area (Table 2). Fire scar mapping is

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essential, so as to make future decisions, and to compile accurate fire statistics. With fire scar mapping it is firstly necessary to decide on the appropriate scale (Woodcock & Strahler 1987), and secondly, it requires research effort to evaluate new technologies in relation to their cost effectiveness (Thompson & Whitehead 1992). Remotely sensed data limits management to specific scales by the very nature of the data. It is thus a combination of the spatial structure of the image and the viewed environment that determine the appropriate scales of observation (Woodcock & Strahler 1987). Mapping scales for each sensor vary considerably, and it is generally accepted that maximum operating scales for LANDSAT TM is 1:50 000, while NOAA-AVHRR 1:500 000 (Thompson & Whitehead 1992).

Applying an adaptive management approach to a patch-mosaic burning system

Scientific management involves the stating of clear objectives and their translation into goals, and making decisions to achieve objectives (Mentis 1984). A decision is a judgement; a choice between alternatives. According to Holling (1978) it is a myth that the goal of management is to produce policies that result in stable environmental behaviour. Our understanding of system's function is poor, and hence it is impossible to minimise change and to eliminate the unexpected. In many systems unexpected or surprising management impacts are in fact necessary and essential system elements (Holling 1981; Holling 1995). Therefore reserve policy must recognise the inevitability of uncertainties, and the consequent selective risk-taking (Holling 1978). Uncertainty exists with respect to the outcomes of different scenarios in terms of their effects on biotic diversity. Savanna functioning is characterized by almost unique combinations and sequences of conditions involving

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moisture, nutrients, fire, herbivory, and other disturbances. Under the current state of knowledge, and for the foreseeable future, we are ignorant of the precise impact of these scenarios of events; save that diverse scenarios not markedly different from preceding ones allowing for persistence and co-existence of species should maintain the present spatio-temporal diversity of savannas (Mentis & Bailey 1990).

Adaptive management may be used to cope with problems associated with uncertainty and imperfect knowledge. There are a number of steps in a management process. The first step is the development a set of clear objectives, and an operational strategy to achieve these objectives. This is should be followed by an evaluation of the tasks and skills required, as well as the systems procedures and budget. The system should then be implemented, and the spatio-temporal diversity monitored, and the costs established. Both Holling (1978) and Walters (1986; 1997) argue that adaptive management should begin with a concerted effort to integrate existing interdisciplinary experience, and scientific information into dynamic models that attempt to make predictions regarding alternative policies. Modelling is therefore an essential and integral part of such a process, and has an important role in: (1) the clarification of the objectives and in enhancing communication, (2) the first attempt at understanding and evaluating the consequences of different strategies, (3) how to rank the outcomes in terms of the objectives, (4) identify what to monitor and how often, and what the limits to monitoring are, and (5) could be used to study what can and cannot be deduced from empirical studies. Mentis & Bailey (1990) use an adaptive management approach, to provide suggestions regarding the implementation of a burning system in a protected area. A monitoring system including the ecological (at appropriate spatial and temporal scales), economic and social outcomes, would need to be established to determine how effective the patch mosaic burning system was at maintaining biotic diversity.

The empirical approach would be long-term, and involves applying a policy for a patch mosaic burning system (see Brockett *et al.* 2001), and monitoring spatio-temporal diversity over time. If monitoring revealed that the burning system was not performing satisfactorily, then there should be a review of this system (after an appropriate interval), and the whole process reviewed. An alternative application procedure should be selected, and the process repeated. Due to the ignorance arising from ecological processes and their interactions, field-management scale experiments are becoming popular to resolve such uncertainties at a landscape scale. However, this management by experimentation approach is generally costly, time consuming, and sometimes risky (Walters & Green 1997). A landscape-scale fire experiment was established at Kapalga, in Kakadu National Park, Australia (Anderson *et. al* 1998). In Kruger National Park, South Africa, a landscape-scale experiment (LASHFIRE) to test three fire management alternatives is to be established (Biggs & Potgieter 1999); included is a patch-mosaic burning system (Van Wilgen *et al.* 1998; Biggs & Potgieter 1999; Brockett *et. al 2001*).

If landscapes are composed of a collection of patches undergoing successional change (Pickett & White 1985), then the character of such landscapes is determined by the frequency, intensity, and spatial extent of disturbances creating patches, as well as the rate and nature of processes that result in patch succession (Christensen 1993). Hence patch models which examine how a homogenous patch or strata changes through time could be

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developed. There are three types of patch models: (1) Markov (Noble & Slatyer 1981), (2) State-and-Transition (Westoby *et. al* 1989), and (3) Frame-based (Starfield *et al.* 1993; Starfield & Chapin 1996).

A Markov matrix of transition probabilities, based on patch demography, could be used to explore the long-term effect of fire regimes on vegetation structure, and composition, and for determining which demographic traits are critical for long-term success in frequently burned savannas (Christensen 1993; Hoffmann 1999). Patches in any particular state have a probability of being transformed during a particular time by successional change or disturbance into some other state. If the transition probabilities are fixed (i.e. the matrix is stationary), the equilibrium frequency distribution of states on landscapes could be predicted with simple matrix algebra (Christensen 1993). The fire regime which leads to the equilibrated proportions of states desired by park management could then be selected. This would help order options in a hierarchy of relative priority (Bell 1984). The priority choice could then be consulted iteratively and in greater detail to simulate the heterogeneity and fire parameter outcomes. This could be used to test an option (e.g. using a flowchart or algorithm as a decision-aid) prior to implementation (Mentis 1980).

Frame-based spatial modelling may also be advantageous as more complex and long-term simulations can be undertaken to determine the possible outcomes of different management policies. This may prove particularly useful when dealing with a patch-mosaic burning system where it allows for the exploration of spatial and temporal heterogeneity indices, and issues regarding different policy (e.g. limits of acceptable change required for the

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management and monitoring systems). Starfield *et al.* (1993) used this approach to investigate the interactions between rainfall, elephants, and fire in a *Brachystegia* woodland in Zimbabwe. However, such modelling does not include the mechanisms leading to changes. Christensen (1993) mentions that the nature of change of any particular patch is dependent on the characteristics of surrounding patches. Hence the need for spatially explicit simulation models. Such spatial dependency is recognised in most fire-spread models. Higgins *et al.* (2000) showed how variance in fire intensity interacts with plant attributes, making variance in fire parameters a key factor influencing the structure of savannas. Modelling paradigms that do not simulate the interaction between environmental variance and organism response are likely to give qualitatively different predictions.

Expert systems have the potential to synthesise current wisdom, and to simultaneously structure it to facilitate application to real world problems. The structuring of current knowledge in a form that facilitates its application aids the decision-making process. This also has the effect of rendering the knowledge testable. As with simulation models, the very attempt to synthesise and integrate knowledge is of great heuristic value. It follows therefore, that expert systems can facilitate the interface between the theory behind patch burning and its practise. Hence such systems could be developed for a number of purposes (Noble 1987), and at various stages of the adaptive management process (see Mentis & Bailey 1990). For example, Noble (1987) presents an expert system to model vegetation response to different fire regimes using a mixture of qualitative and quantitative rules, and Hoare (1985) working in Kakadu National Park, Australia, modelled the biological effects of fire on plant communities, using a fire behaviour and vegetation damage expert system. Davis *et al.* (1986) developed a fire behaviour expert system for Kakadu National Park. Bailey *et al.*

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(1993) present an expert system to facilitate consistent fire management decision-making in Pilanesberg National Park, South Africa. The development of expert systems leads not only to new applications, but also to new questions and modes of research (Starfield & Louw 1986).

Conclusions

Theory behind the patch-mosaic burning system appears intuitively appealing, and research thus far seems to support this approach. There are problems however with this system, both theoretical and practical, that must be addressed. We suggest the implementation of a patch-mosaic system through adaptive management involving modelling and monitoring phases.

The patch-mosaic burning strategy advocating a flexible, variable approach to fire management based on traditional burning practices offers a useful model (Saxon 1984; Rusell Smith *et al.* 1997). In southern Africa, the challenge of adopting similar burning systems has been taken up, most notably, by Pilanesberg National Park, South Africa. A system of prescribed burning combined with natural lightening fires has been implemented since 1989 with the aim of developing a patchy landscape mosaic (Brockett *et al.* 2001).

It thus appears that through a patch-mosaic burning strategy, savanna fire ecology may be advancing into a new realm, with unexplored and exciting possibilities for research and management, with the continued protection and maintenance of biodiversity. It remains to be

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seen how long these ideas take to filter through to other fire-prone environments in southern Africa, such as the fynbos or the Drakensberg grasslands. It is possible however, that some environments where fire hazards are extreme may not be compatible with a free patch-burn regime (Morrison *et al.* 1996).

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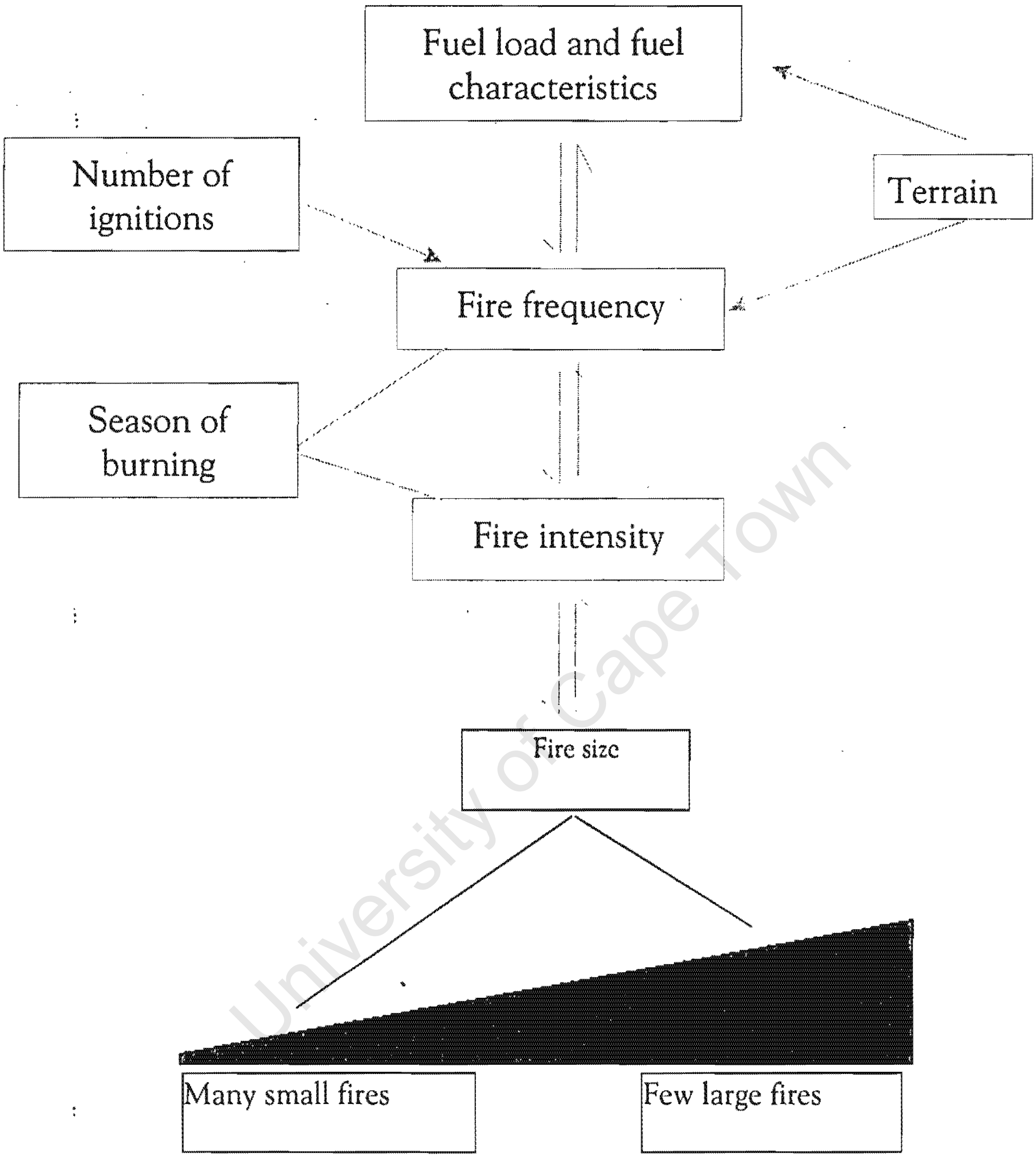

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Figure. 1. Conceptual model for the interactions between fuel load, and fire parameters (seasonality, frequency, intensity, and fire size) in producing a continuum of fire sizes.

Table. 2. Challenges to the implementation of a patch-mosaic burning system in a protected area.

Challenges
Random point ignitions
(1) ***Patchy fuel load distribution***
In arid fertile savannas in many cases the fuels may be below <1000 kg/ha, and at perhaps 50% of locations the fuel may be insufficient for a fire to spread (except under extreme fire weather conditions). A solution could be to use a stratified-random sampling design to select areas meeting fuel load criteria. Making such a choice will however result in certain outcomes, and these would need to be evaluated.
(2) ***Fuel moisture***
Due to differential curing rates of fuels across a landscape, it may be difficult to ignite fires at the selected points in certain seasons.
(3) ***Atypical locations***
Obviously atypical locations such as rocky koppies, or steep slopes with little fuel should be excluded.
(4) ***Grid-square size***
Varying the size of the grid will affect the number of ignitions per unit area, and hence the resulting mosaic. Using a larger grid size will also enable greater choice to be exercised in selecting ignitions sites (this may help with low fuel load situations).
(5) ***Map reading problems***
In flat featureless areas in may be difficult for an operator to know whether he is in the correct grid square or not. The operator could use the GOTO function on a GPS to locate the centre of the grid square.
(6) ***Zonation***

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The zonation (and hence the management) of the protected area may influence the method of ignitions used in certain zones e.g. wilderness zones. Certain zones may require aerial ignitions by helicopter.

Fire scar mapping

An essential requirement for the system is fire scar mapping after each fire. This information is required so as to make future decisions regarding whether to ignite further fires or not. In comparison with perimeter block burning systems where the mapping of fire scars are easier (as fires are ignited from roads) with the possible exceptions of wildfires. However, unburnt islands within block burns are often ignored.

Starting mosaic

The current attributes of the fuel mosaic will influence fire size. Therefore in order to establish a fire mosaic, special ignition rules and selection of fire weather conditions may be required initially.

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Chapter 5

A patch mosaic burning system for conservation areas in southern African savannas

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Abstract

Fire-prone savanna ecosystems in southern African conservation areas are managed by prescribed burning in order to conserve biodiversity. A prescribed burning system designed to maximise the benefits of a diverse fire regime in savanna conservation areas is described. The area burnt per year is a function of the grass fuel load, and the number of fires per year is a function of the percentage area burnt. Fires are point-ignited, under a range of fuel and weather conditions, and allowed to burn out by themselves. The seasonal distribution of planned fires over a year is dependent on the number of fires. Early dry season fires (May - June) tend to be small because fuels have not yet fully cured, while late season fires (August - November) are larger. More fires are ignited in the early dry season, with fewer in the late dry season. The seasonality, area burnt, and fire intensity are spatially and temporally varied across a landscape. This should result in the creation of mosaics, which should vary in extent and existence in time. Envelopes for the accumulated percentage to be burnt per month, over the specified fire season, together with upper and lower buffers to the target area are

proposed. The system was formalized after eight years of development and testing in Pilanesberg National Park, South Africa. The spatial heterogeneity of fire patterns increased over the latter years of implementation. This fire management system is recommended for savanna conservation areas of >20000 ha in size.

Keywords: biodiversity, fire, Kruger National Park, management, mosaic, Pilanesberg National Park, South Africa, spatial heterogeneity

Introduction

Protected areas in southern African savannas were often managed in the past to maintain their pristine state, and to include a full range of biological diversity. However, the definition of a desired state is problematic, and there is much debate regarding the use of equilibrium or non-equilibrium theory to explain savanna functioning (Frost *et al.* 1986; Westoby *et al.* 1989; Illius and O'Connor 1999). There are four major determinants of tropical savannas. The primary determinants are plant available moisture, and available nutrients, while fire and herbivory are superimposed as secondary determinants (Frost *et al.* 1986). Savannas can therefore be viewed as event-driven systems (Walker *et al.* 1986), which may result in multiple stable states (May 1977; Dublin *et al.* 1990). Events such as fire, drought, rainfall and grazing and their interactions, then become the drivers to overcome inertia between states (Mentis and Bailey 1990).

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Some sequences of events are unique, and are not repeated in either time or space. Mosaics accordingly arise with units of variable extent and existence in time, to create high patch diversity which is a characteristic of savanna ecosystems (Mentis and Bailey 1990). A fire regime is characterized by the frequency, seasonality, intensity and type of fire (van Wilgen, *et al.* 1997). Fire-derived mosaics could be maintained by igniting fires under diverse weather and fuel conditions, which are varied over time (Mentis and Bailey 1990). In order to maintain spatial heterogeneity it is important to vary these fire parameters spatially and temporarily across the landscape. This should mimic the prehistoric or historic fire patterns, reduce fire hazard and the costs of prescribed burning, and the costs of managing wildfires.

In this paper a patch mosaic burning system and its application in the Pilanesberg National Park are described. This system has taken its cue from Australia, where patch-burning systems have been applied in Uluru (Ayers Rock-Mount Olga) National Park (Saxon 1984), and in the Kakadu National Park (Russell-Smith 1994; Russell-Smith *et al.* 1997). It has been developed through adaptive management and through a process of learning-by-doing for use in southern Africa. The system presented here did not therefore precede the results, but rather co-evolved and was modified as results became available. The reader should bear this in mind when interpreting this paper. Patch mosaic burning constitutes a significant departure from the conventional prescribed burning currently practised in South African savanna parks (Mentis and Bailey 1990; van Wilgen *et al.* 1998). Under a fixed prescribed burning system, the area is divided into portions of equal size (termed "blocks"), and these are burned according to a schedule (e.g. a 3-year interval in spring), and under prescribed weather conditions, to create a relatively regular mosaic of post-fire ages. There are a number of problems with this approach. Refinements are often required to accommodate erratic and unpredictable rainfall, and unplanned fires (see Bailey *et al.* 1993; Stander *et al.* 1993; du Plessis 1997). In addition, a reduction in spatial heterogeneity (and thereby also in ecosystem resilience) could occur as a result of homogeneous fire regimes (Scholes and Walker 1993). Fine-scale mosaics also enhance the abundance and diversity of terrestrial vertebrate fauna, for example small mammals (Mentis and Rowe-Rowe 1979; Rowe-Rowe and Lowry 1982), ground-nesting birds such as francolins (Mentis and Bigalke 1981); and antelope (Mentis 1978). The use of habitats by faunal species takes place at a variety of spatial and temporal scales, and therefore management regimes that result in homogenization should be avoided (Law and Dickman 1998).

Study areas

The patch mosaic burning system was developed from experience gained and data gathered in two South African conservation areas: the Pilanesberg and Kruger National Parks. Both areas are covered with savanna vegetation, essentially a mixture of trees and grasses. Trees are typically 3 - 10 m in height and cover between 5 and 30 % of the area. Grasses are between 0.5 - 1.5 m tall and cover between 10 - 80 % of the area. Grass fuels could be described as anywhere between sparse and continuous, and this varies considerably between years depending on rainfall in the preceding growing season. Grass forms the majority of fuel for fires, and downed woody material and litter are relatively unimportant. Fire return periods range from 1 to 20 years, averaging around 5 years (van Wilgen *et al.* 2000).

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Pilanesberg National Park (50000 ha) is situated in the North West Province of South Africa (Figure 1). It is located in hilly country, on the remnants of an extinct volcano 25 km in diameter. The vegetation falls in Acocks' (1975) Sour Bushveld. This is a moist savanna (mean annual rainfall 630 mm falling mostly between October and March) with macrophyllous trees and bushes (notably *Combretum* species) on the rocky hills, and microphyllous trees and bushes (mostly *Acacia* species) on the clayey bottomlands. The herb layer is dominated by tufted perennial grasses which give a continuous to patchy fine-fuel distribution of 500 - 20,000 kg ha^{-1} depending on rainfall in the previous growing season. Sufficient grass-fuel to carry a fire usually accumulates after one or two growing seasons following a burn. Fires usually occur in the dry season, between April and December.

The Kruger National Park (1.9 million ha) is situated in the low-lying savannas of the eastern parts of the Northern and Mpumalanga Provinces of South Africa, adjacent to Mozambique in the east and Zimbabwe in the north (Figure 1). Mean annual rainfall varies from 750 mm in the south to 350 mm in the north, and falls predominantly between October and March. The western part of the park lies on the granitoid rocks of the Basement complex and the eastern part lies on basic and acid volcanic rocks of the Karoo sequence (Venter 1990). The vegetation of the park is diverse and falls into five of Acocks' (1975) veld types: Lowveld Sour Bushveld, Lowveld, Arid Lowveld, Mopani Veld, and Mixed Bushveld. Fires are concentrated in the dry season, between July and November (van Wilgen *et al.* 2000).

Both areas have substantial populations of large herbivores, including many species of antelope, giraffe, zebra, elephant, black and white rhinoceros, hippopotamus and warthog.

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Habitat requirements for these species differ, and are in part dependent on the fire regimes that produce them. In addition, herbivory also affects the dynamics of the fuel layer, which impacts on fire. The presence of this variety of herbivores, and the need for their conservation in the same area, was one of the prime arguments for developing the patch mosaic burning system.

The procedure for patch mosaic burning

The primary goal of patch mosaic burning is to produce heterogeneity within the landscape, as a result of applying fires in a varied manner over successive fire seasons. The procedure developed is outlined in conceptual form in Figure 2, in such a way that the basic driving principles can be clearly understood without the clutter of operational details and exceptional circumstances. Appendix 1 provides the full and rigorous flow chart for all eventualities, and is the recommended version for operational use. Readers wishing to understand the full rationale for each step to be taken in practice should thus consult this appendix, which is self-explanatory. The explanations below relate primarily to the principles in Figure 2, though occasional reference has been made to Appendix 1 when the linkages might not be intuitively clear. The versions have slightly different flow patterns, denoted by letters A to I in Figure 2 and numbers 1.0 to 2.10 in Appendix 1. Reference to letters or numbers refer to Figure 2 or Appendix 1 respectively.

Estimates of area to be burned

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In savannas, fuel loads vary enormously from year to year. This depends to a large degree on the previous seasons' rainfall and they determine, in turn, the area that will burn. Grass fuel loads are therefore a critical driver of the fire potential and are used in this system to calculate area to burn. The extent of fire in any given year is directly related to the grass biomass at the end of the rainy season; this relationship was established in the Kruger National Park (Van Wilgen *et al.* 2000). The patch mosaic system uses this relationship to determine the percentage of area to be burned (Figure 3). Grass biomass and consequently area to burn are estimated at the end of the growing season, as described below. Initial estimates (made in January) and subsequent improved estimates (made in April) are part of Step A (Figure 2), though the difference between them is not shown in this simplified version. Should early season fires occur, Step 1.3 in Appendix 1 indicates that these fires can be allowed to burn until this initial estimate is reached. Although such extensive fires are unlikely to occur at this time of the year, it is nonetheless necessary to manage these early season fires. The provisional estimate of the percentage area to be burned is obtained through using the best available seasonal rainfall forecast in January (which is around the middle of the rainfall season) from which grass biomass can be estimated using known relationships (see Deshmukh 1984; Snyman and Fouche 1993; Snyman 1994). The first estimate of percentage area to be burned is revised when the grass fuel loads are measured in April (Step A1 and Figure 3). If these measurements cannot be taken, estimates of fuel loads could be based on rainfall in the growing season (ie from July of the previous year until April). The number of fires in which the area will be burned increases as the area to be burned increases (Step A2 and Figure 4). More fires are required where large areas are to be burned, both in order to create a mosaic and to allow for the fact that fires normally burn out before reaching a large size. The data points on Figure 4 represent the actual number of fires that were required to burn the target area in Pilanesberg National Park once a mosaic had been established (ie the original large and continuous fuel beds were broken up).

Distributing fires over the fire season

Patch mosaic burning uses a large number of fires spread over the fire season rather than one or a few large fires. To maintain a fine-scale mosaic, some fires need to be ignited early in the dry season. The desired seasonality of the fires also varies with the area to be burned (Step A3 and Figure 5). This is mainly to accommodate cases with high fuel loads and large areas to burn. Here early fires (which do not become large as fuels are not fully cured) are essential to break up fuel and prevent large late season fires. Where the area to burn is large, fires should be spread over the fire season from April, peaking in June, and extending to November. With a moderate area to burn, burns should commence in April, peak in June/July, and continue until September. If the area is small and thus few fires are to be ignited, these fires should be skewed towards the latter part of the fire season, peaking in August and ending soon thereafter. The curves shown in Figure 5 represent hypothetical desired distributions. The actual distributions that were achieved are shown in Figure 9, and correspond closely to the desired distributions. For example the desired distribution for high biomass in Figure 5 corresponds to the 1996 distribution (a wet year) in Figure 9; similarly the distribution for a medium biomass year corresponds to the 1995 distribution (a year with avearge rainfall).

Location of ignitions

The locations for ignitions should be identified and plotted on a map (part of Step A3). These locations should be randomly generated, or a stratified random sampling design could be used, in which selection is based on fuel load criteria. In this system, random points are not moved to accommodate other management objectives. All ignitions are point ignitions, and once a fire has established, no further ignition is applied to assist the spread of a fire. Fires ignited in this way can burn for between one or two hours and 7 days. The resultant fire behaviour (and hence fire intensity) varies with the combination of head, back and flank fires burning during the day as well as at night.

Updating and monitoring

Beginning in January, the date of each fire is recorded, the extent mapped and corresponding area calculated. From this the accumulated number of fires and area burned to date for the year can be calculated (Step F). These running totals are continuously compared to the targets in Steps B-D in Figure 2.

Areas burned in each month

Fires will be ignited each month until the area burned falls between defined upper and lower limits (Figure 6). After each fire, the new cumulative area (Step F) is tested against these limits (Steps C and D) with ignitions for that month ending (Step I) once the upper limit is reached. Below the upper limit, Step D allows for further ignitions or unplanned fires when the target number has not been reached, and also allows further fires when the target number for that month has been reached, but not the lower limit for area burned. For practical reasons the system allows a buffer above and below the actual target. This is specified at 10

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% in Figure 6 and in Appendix 1, though some managers may wish to set the buffer as greater or smaller. This buffer allows managers some flexibility (managers will not be held to exact targets for area burned) and is in proportion to the total area to be burned (ie the larger the target area, the larger the buffer). Although 10 % is an arbitrary choice, it seemed reasonable to us based on our experience (see under discussion - "*Further development of envelopes of area burned*").

Arrangements to maximise the chance of achieving the target area

These procedures are depicted in Appendix 1. Rather than setting ignitions right up until the target, "there is a tapering off procedure" for the final designated month, which allows ignitions to stop at the lower buffer limit (Step 2.2 of Appendix 1). This serves as a measure (Step 2.6) to accommodate further unplanned burns (including lightning) without easily going over target plus the upper buffer limit (Step 2.7), and allows any shortfall in the number of planned fires to be reduced (Step 2.5). Once the upper limit is reached, all further fires should be suppressed (Step 2.8).

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Results of application of the system

The principles of the patch mosaic burning system were applied in Pilanesberg National Park over eight years, and this experience (augmented with experience and data from the Kruger National Park) provided the basis for the development of the system described above. In the sections that follow, we provide data from the Pilanesberg National Park as an illustration of the application of the principles and their outcome.

Proportion of area burned in relation to grass fuel loads

Grass fuel loads were not measured in Pilanesberg National Park, and herbaceous biomass (Table 1) was derived using a relationship between precipitation and grass peak biomass for east and southern Africa (Deshmukh 1984). We used annual rainfall in the preceding season (July to June) to estimate these fuel loads. In addition, data on herbaceous fuel loads and percentage of the area burned were available from the Kruger National Park for 1989 to 1997. These data (see data points on Figure 3) indicate that fuel loads are generally higher in Pilanesberg than in the Kruger National Park, probably due to higher rainfall at Pilanesberg. In 1995, the actual grass biomass was far lower than the predicted value due to the late arrival of the rains. As a result, only a small proportion of the park was burned in 1995 (Table 1), and this data point appears as an outlier in Figure 3.

The number of fires and area burned in Pilanesberg National Park

When patch mosaic burning was first introduced to the Pilanesberg National Park in 1989, the relationship between area burned and the number of fires did not meet the requirements of patch mosaic burning for the first 4 years. This was understood in an intuiative way, but can, after our analysis, be stated in terms of a heterogeniety index that was below 45 (Table 1). However, in the next 4 years, the mean fire size declined and the number of fires increased (Table 1). The maximum fire size (c. 10000 ha), and highest mean fire size both occurred in 1989. Both 1990 and 1991 had maximum fire sizes > 6000 ha. In the latter four years of implementation the maximum fire size was 5350 ha. Between 1989-92 the mean fire size was greater than 650 ha, while between 1993-96 it was 300 ha or less. The median fire sizes also declined over the period of implementation. For the first four years the median

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was 100 ha, and over the latter four years it was 50 ha. These changes corresponded with changes in the number of fires. In the first four years (1989-92), 63 fires were ignited which burned 72000 ha. In the latter four years, the number of fires increased to 191, and the total area burned by these fires was reduced to 61708 ha (Table 2).

The area burned by fires of different sizes for two periods (1989-92, and 1993-96) was classified into small, small plus medium, and all fires, as follows: (1) small fires: 0 - 250 ha in steps of 50 ha; (2) small and medium fires: 0 - 1000 ha in steps of 250 ha; and (3) all fires in the following categories: 0-1000 ha, 1001-3000 ha, 3001-5000 ha, 5001-10000 ha, and 10001-50000 ha. Between 1989 and 1992 a larger proportion of the area was burned by larger fires. Most of the area was burned by fires >1000 ha in size, with fires of >5000 ha constituting the greatest area burned. Few small and medium fires occurred. Between 1993-96, the number of small and medium sized fires increased, and most of the area was burned by fires >3000 ha and <5000 ha, followed closely by fires <1000 ha (Figure 7).

Seasonal distribution of fire

There was a shift in fire season over the eight years in which patch mosaic burning was applied in the Pilanesberg National Park (Figure 8). In 1989 mostly winter and spring burns (July to September) were ignited, while in later years they were spread over the year between April and December. The spread of fires over a longer period with occasional fires in summer (usually lightning ignited) assisted the development of the mosaic, which took four years to develop. If a greater number of fires had been applied in the first three years in the autumn period (April to June), then it is possible that the mosaic could have developed more

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quickly as such fires are small and patchy. However, the interim period was necessary to break up continuous fuel beds and to distribute fires over the area.

The seasonal distribution of fires between 1992 to 1996 indicates that for average (e.g. 1994) and above average (e.g. 1996) rainfall years, the distributions peaked in May or June. For a below average rainfall year (e.g. 1993), the number of early-season fires were reduced, and still peaked in June. For an average rainfall year, (but one in which >200 mm were received in autumn) the distribution peaked in late winter (1993). For a below average rainfall year (classified as < 75% of mean), the peaks were in late summer (lightning ignited fires, 1992, Figure 9).

Spatial distribution of fires

One of the major aims of the patch mosaic burning system is to create spatial heterogeneity in the distribution of fires. The impact of introducing the system at the Pilanesberg National Park was assessed based on a framework of 58 x 51 pixels, each of 500 x 500 m (25 ha). Each pixel was recorded as having been burned if more than half of its area was burned in a given year. Pixels having more than half their area lying outside the park were excluded from all analyses. For the purposes of analysis of heterogeneity patterns, a moving window neighbourhood of 9 pixels (3 x 3, covering 225 ha) was passed across the whole surface for each year, and a score (0/9, 1/9, 2/9... 9/9), was allocated to the window depending on the number of pixels burned. All windows encompassing one or more pixels outside the park were excluded, resulting in a small buffer area inside the boundary being excluded from this neighbourhood or focal analysis (Tomlin 1990). The results for each year were totalled as

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follows: scores of 0/9 and 9/9 were placed in the category "homogeneous", and all scores from 1/9 to 8/9 were placed in the category "heterogeneous". A percentage heterogeneity index was calculated as the number of heterogeneous windows expressed as a percentage of the total number of windows (Table 1).

Our choice of 225 ha (the smallest moving window that can be used based on 25 ha pixels) as a basis for calculating heterogeneity patterns was pragmatic. The basic pixel size (25 ha) was used as it had formed the basis for recording a range of features in the park by managers over many years. Managers also felt that any smaller subdivision would not have increased the accuracy of their records. In addition, it proved useful for examining the impacts on the habitat of sable antelope (a rare species whose conservation is seen as a priority). Sable antelope home ranges average just over 2000 ha (Magome 1991), and maximizing heterogeneity at a scale of 10% of this area should benefit them by allowing herds to utilise different portions of their range several times in a year. The implications of choosing different scales, resolutions, and moving window neighbourhood sizes for heterogeneity scores are being investigated further.

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Three of four recent years (1993, 1994, and 1996) showed higher heterogeneity scores than any of the first four years (Figure 10). The change in the heterogeneity (from 30.6 to 57.6) occurred between 1992 and 1993, and is related to the percentage burned, and the number of fires in these two years (Table 1). However, it is clear that the index can be influenced by area burned. For example, it was low in 1995 because of the small area burned in that year. The reason for the increase in heterogeneity in the years 1993, 1994 and 1996 lies in the wide distribution of smaller fires across the park (Figure 10), compared to the few larger fires in the earlier years (eg 1989) which left wide areas homogenous (either burned or unburned, Figure 10). There are too few data in this set to make a statistical comparison between these indices, though clearly if 1995 is dropped from the series, an elementary test would deliver a significant difference. When more data are available we suggest that a longer time series of similar indices can be tested with some form of adjustment (such as covariance analysis) being made for area burned. Alternately, a fair and objective index incorporating area burned could be derived. Minnich (1983) suggests that perhaps the most important variable affecting the spatial properties of fires is antecedent fire, as it is common for an active fire to stop at former burns because of insufficient fuel. Moreover, the extent of overlap between previous burns and active fires over some reference period should decrease as the time required for vegetation to reach flammability increases. Thus in slow-maturing communities the growth of fires is more constrained by previous fire history than in fast-maturing ones; fire size should be inversely related to fire frequency. Parr and Brockett (1999) developed a conceptual model to illustrate how fire size may be determined by fuel characteristics, terrain and number of ignitions. In this model fire size was nested under fire intensity and fire frequency. Future heterogeneity indexes that are developed should perhaps look at these multi-year fire-parameter interaction effects. This could be done by weighting the contribution of fires in the park, for example, with the current year counting 50%, the preceding year 33.3%, and the one before that 16.6%.

Discussion

Making choices

In order to implement patch mosaic burning, managers have to make certain choices. These choices are based on assumptions or conventions that we have adopted in our system. Firstly, we have chosen a single year as the basis for achieving areas burned and patterns established. We have chosen this time frame, rather than several years, because of the often marked differences between individual years within savannas. Fuel loads, for example, vary markedly from one year to the next, depending on annual rainfall; long-term rainfall trends have a far smaller effect. While this system may not be suitable for other fire-prone ecosystems (such as chaparral shrublands or boreal forests where fuel loads build up over decades or centuries, and interfire periods are similarly long), the choice suits African savannas. We have also chosen to distribute ignition points without considering spatial heterogeneity in the vegetation. Again, this is suitable for African savannas, where much of the vegetation is relatively uniformly flammable; it may not be a suitable system for landscapes with diverse mixtures of forest, grassland and riparian valleys. Our choice of using a single ignition point, rather than using multiple or supplimentary ignitions also has consequences for fire behaviour patterns - we expect a wide range of behaviours associated with patterns of head and backfires burning both during the day and at night. We also choose to suppress fires should they be likely to result in burning larger areas than defined by our system. We are thus not seeking to impose a "natural" regime on the ecosystem, but rather we are attempting to simulate a situation which may have existed when early humans ignited the vegetation in the past.

Comparison with fire on other continents

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The application of a patch mosaic burning system at Pilanesberg resulted in noticeable changes to the fire patterns after a few years. This result has been obtained in other ecosystems on other continents, suggesting that this approach may find wider application beyond African savannas. For example, fire mosaics in southern California and northern Baja California were found to differ with respect to the number of fires burned, but not in the total area burned (Minnich 1983). The median fire size in southern California by decade ranged from 3500 to 13500 ha, compared with 1600 ha in Baja California. Fire suppression was applied in southern California, but not in Baja California, and suppression reduced the number of fires, but not burned area; fires in southern California consequently increased in size, spread rate, and intensity and became uncontrollable in severe weather conditions. However, in Baja California there were more small fires, and less large fires. These relationships correspond with the fire statistics for Pilanesberg National Park: fewer (but larger) fires occurred in the first four years of implementation, while in the latter four years a similar (somewhat smaller) area burned in more than three times the number of fires. Similarly, the median fire size declined steadily over 15 years in Kakadu National Park in northern Australia, from 300 ha to c. 60 ha (Russel-Smith *et al.* 1997). These changes corresponded to a pronounced shift from late dry season fires, to early dry season fires (using helicopter aerial-ignitions) over the period studied. Unfortunately, we have not been able to compare our management approach to similar approaches adopted at Kakadu and Uluru (Ayres Rock-Mount Olga) National Parks, as these have not been published as far as we are aware.

Further development of envelopes of area burned

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In Figure 6, we have made a schematic proposal for the definition of limits around which decisions are made on whether to ignite further fires. These limits are shaped around the cumulative total of area burned in a given year. The actual shape of the curve will vary depending on the area that should be burned each year. As insufficient data are available to construct a family of such curves, we have not been able to develop this part of the system further. This will need to be done as more data become available. Examples of possible envelopes for targets at 50, 30, and 15 % of target area being burned are indicated in Figure 11. Once such a family of curves has been developed, an additional step (between steps 1.9 and 2.0 in Figure 2) should be added to the procedure. At this step, the appropriate curve for the year in question should be selected.

Area required to implement the system

We recommend that patch mosaic burning be considered for adoption in other African savanna conservation areas of suitable size. Pickett and Thompson (1978) have recommended that the size of a reserve should be considerably larger than the largest fire size. The size of the largest fire depends on factors such as soil fertility and rainfall. Based on the data from Pilanesberg National Park, an area at least twice the approximate area of the largest fire recorded (10 000 ha), perhaps 20 000 ha would be required to practise this type of fire management system.

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Table 1. Statistics from a patch mosaic burning system implemented in Pilanesberg National Park, South Africa over eight years (1989-1996).

Year	Annual rainfall, July-June (mm)	Predicted grass biomass (kg/ha)	% planned to burn	% actually burned	Number of fires	Mean fire size (ha)	Median fire size (ha)	Maximum fire size (ha)	Heterogeneity (%)*
1989	691.9	5676.7	62.5	61.2	16	1913	275	9875	23.7
1990	696.2	5713.8	25	27.4	21	652	50	6225	33.7
1991	735.3	6045.6	33	30.6	17	901	100	6700	39.5
1992	393.3	3142.2	20	24.9	9	1386	500	3900	30.6
1993	505.7	4096.2	27.5	19.6	42	214	50	1575	57.6
1994	583.6	4757.5	45	51.1	68	302	50	5350	51.7
1995	569.7	4640.2	4.5	2	10	98	38	450	9
1996	813.5	6709.1	50	50.7	71	282	25	4325	63.5

* Groups of 9 pixels classified as heterogenous (neither completely burned nor completely unburned) expressed as a percentage of all such groups (see text).

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Table 2. Total area burned and number of fires during two four-year periods following the application of a patch mosaic burning system in Pilanesberg National Park, South Africa.

Period	Total area burned (ha)	Number of fires
1989 - 92	72 000	63191
1993 - 96	61 708	

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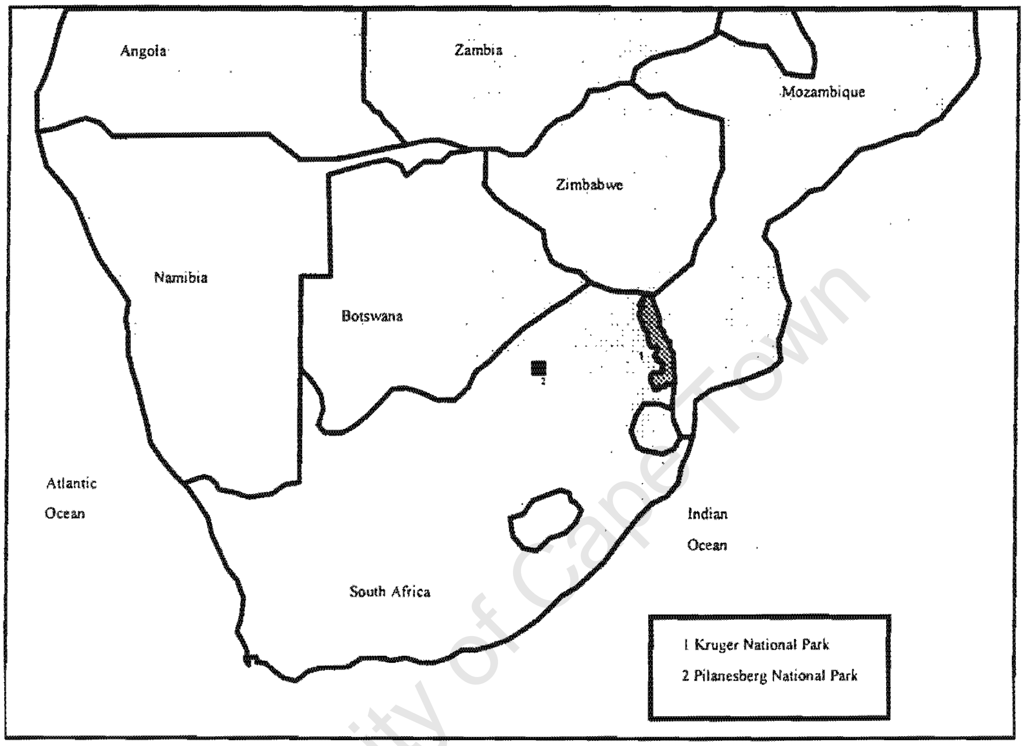

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Figure 1. Map of southern Africa showing the locations of the Kruger National Park, and Pilanesberg National Park.

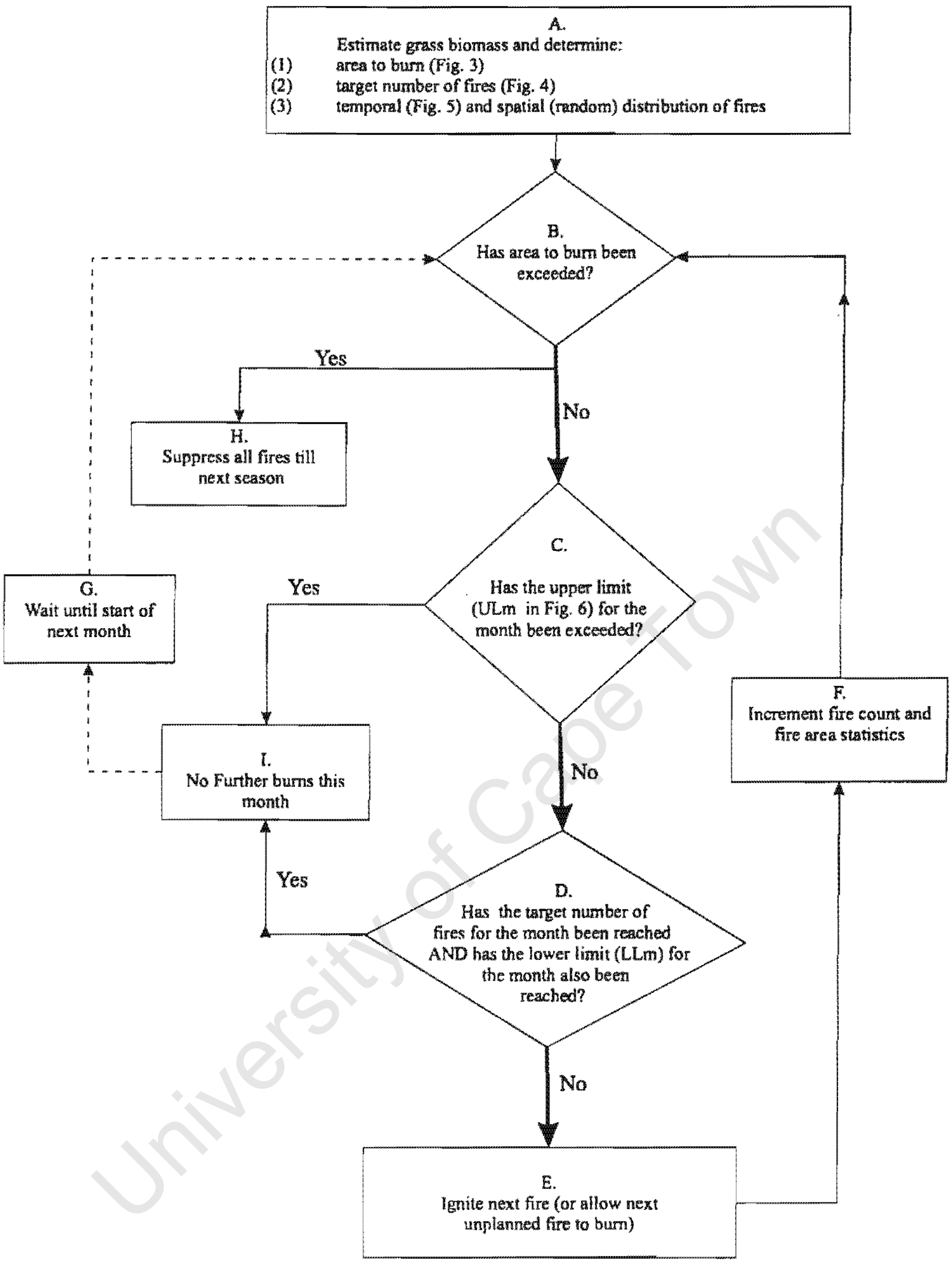

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Figure 2. Simplified conceptual procedure for a patch mosaic burning system. For the full procedure, see Appendix 1.

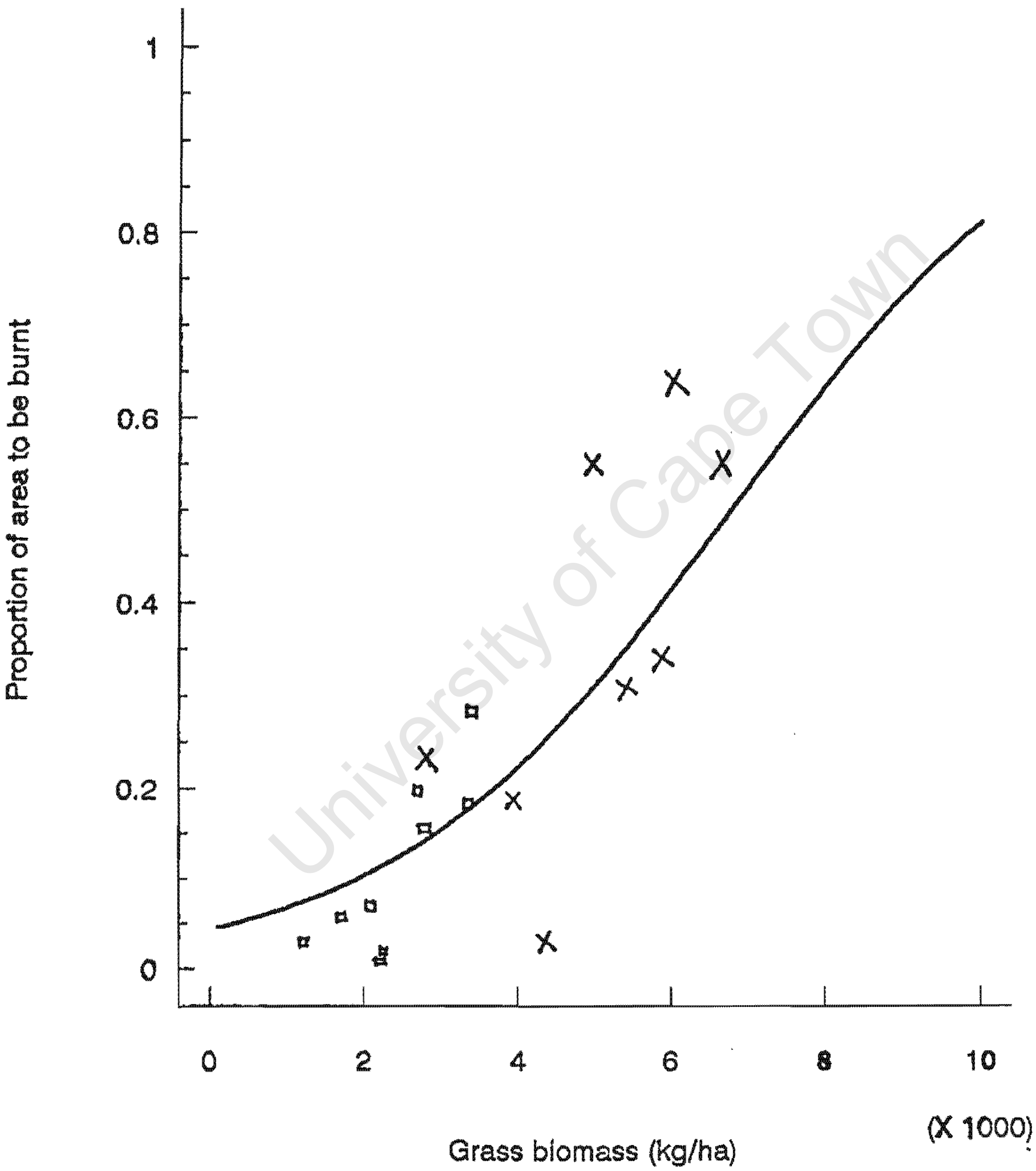

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Figure 3. The relationship between grass biomass and the proportion of area to be burned. Data points are for Pilanesberg National Park (+) over eight years (1989-96), and for the Kruger National Park (□) over nine years (1989-97). The relationship (averaged from two individual years in the Kruger National Park (see van Wilgen *et al.* 2000)) is is p = {exp (-3.7 + 0.00047b)/[1 + exp(-3.7 + 0.00047b)] + exp (-2.47 + 0.00051b)/[1 + exp(-2.47 + 0.00051b)]}/2, where p is the proportion of the area to be burned, and b is the grass biomass in kg/ha.

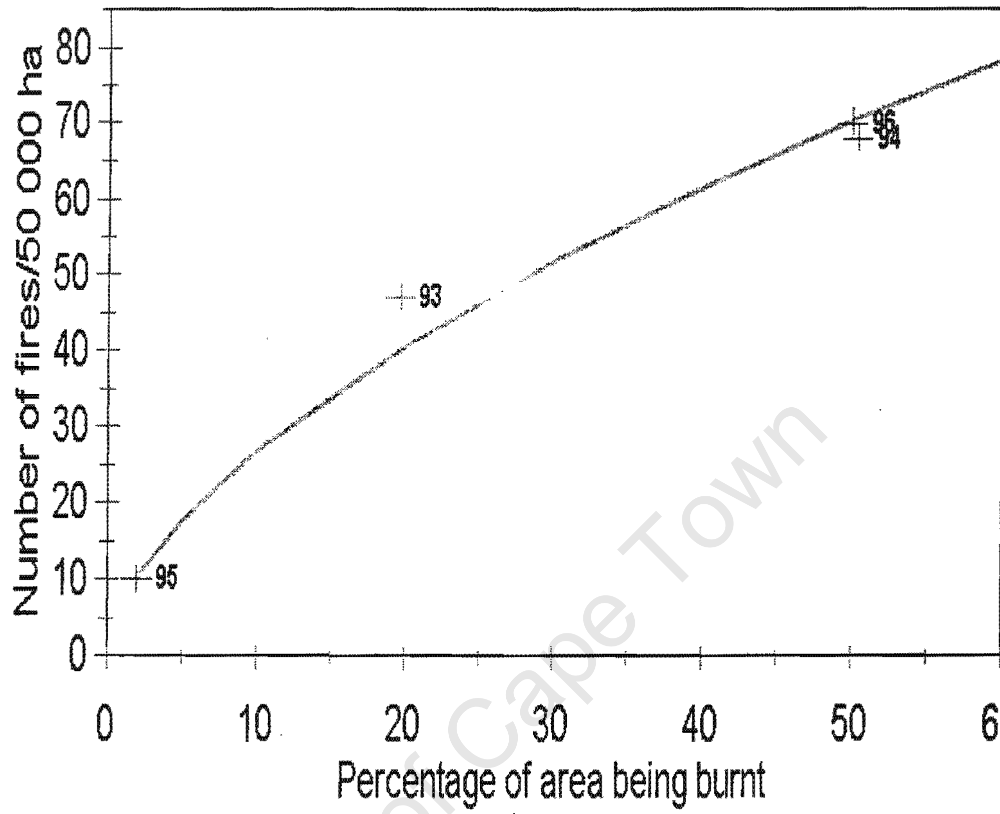

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Figure 4. The relationship between the percentage of area burned, and the number of fires per 50 000 ha. The data points are for four years (1993 - 96) in the Pilanesberg National Park. The relationship is: $y = 6.7\ p^{0.6}$, where y = number of fires per 50 000 ha, and p = percentage of area to be burned ($r^2 = 99.8$; $p = 0.0009$).

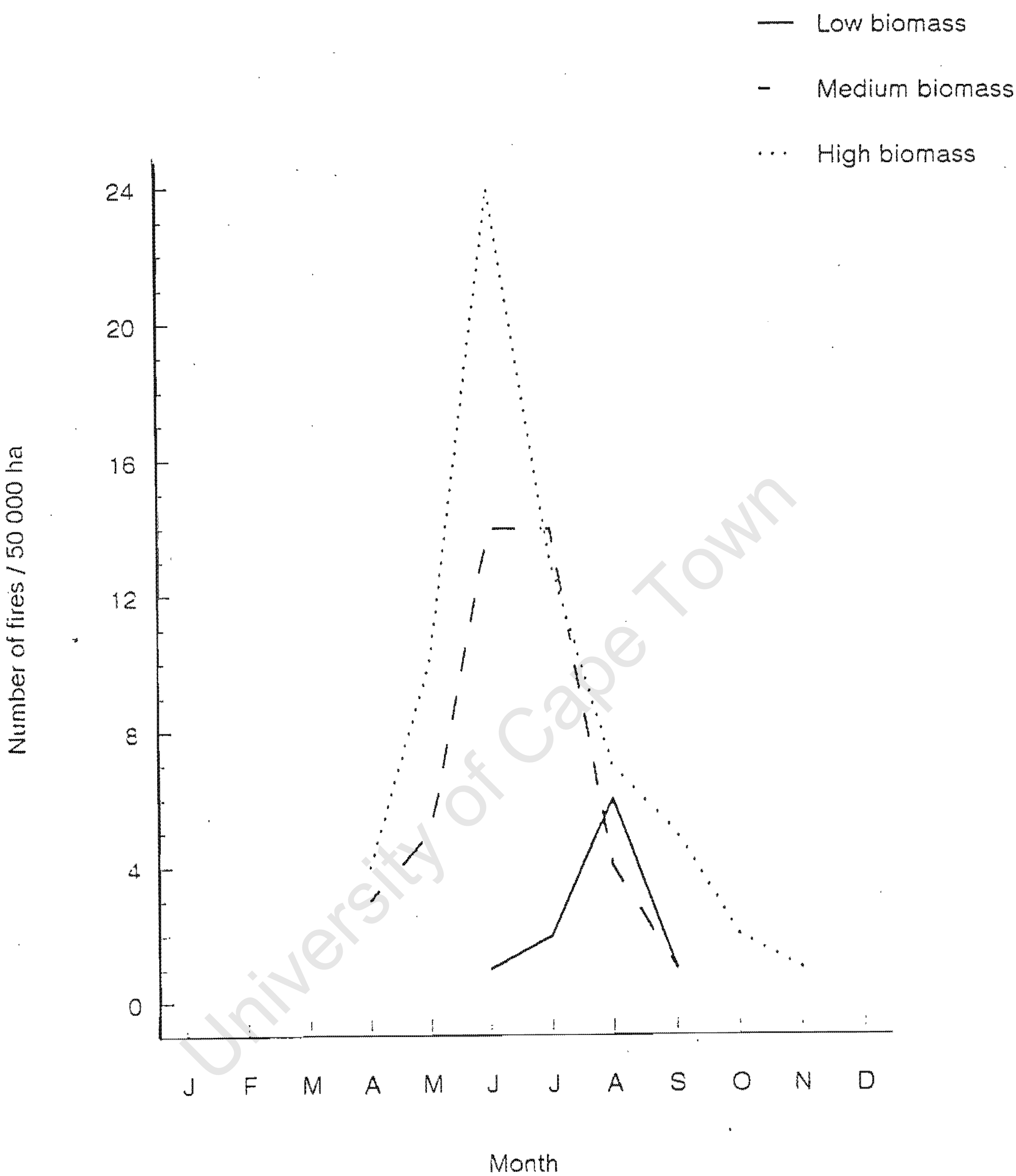

Figure 5. Hypothetical desired seasonal distribution of fires as a function of grass biomass. With high biomass, the number of fires peaks in June, and the fire season ends latest; with moderate biomass the number of fires peak in June/July, and the fire season ends sooner than for a high biomass year; and with a low biomass the number of fires peaks later (in August) and the fire season ends in September. Actual data shown in Figure 9.

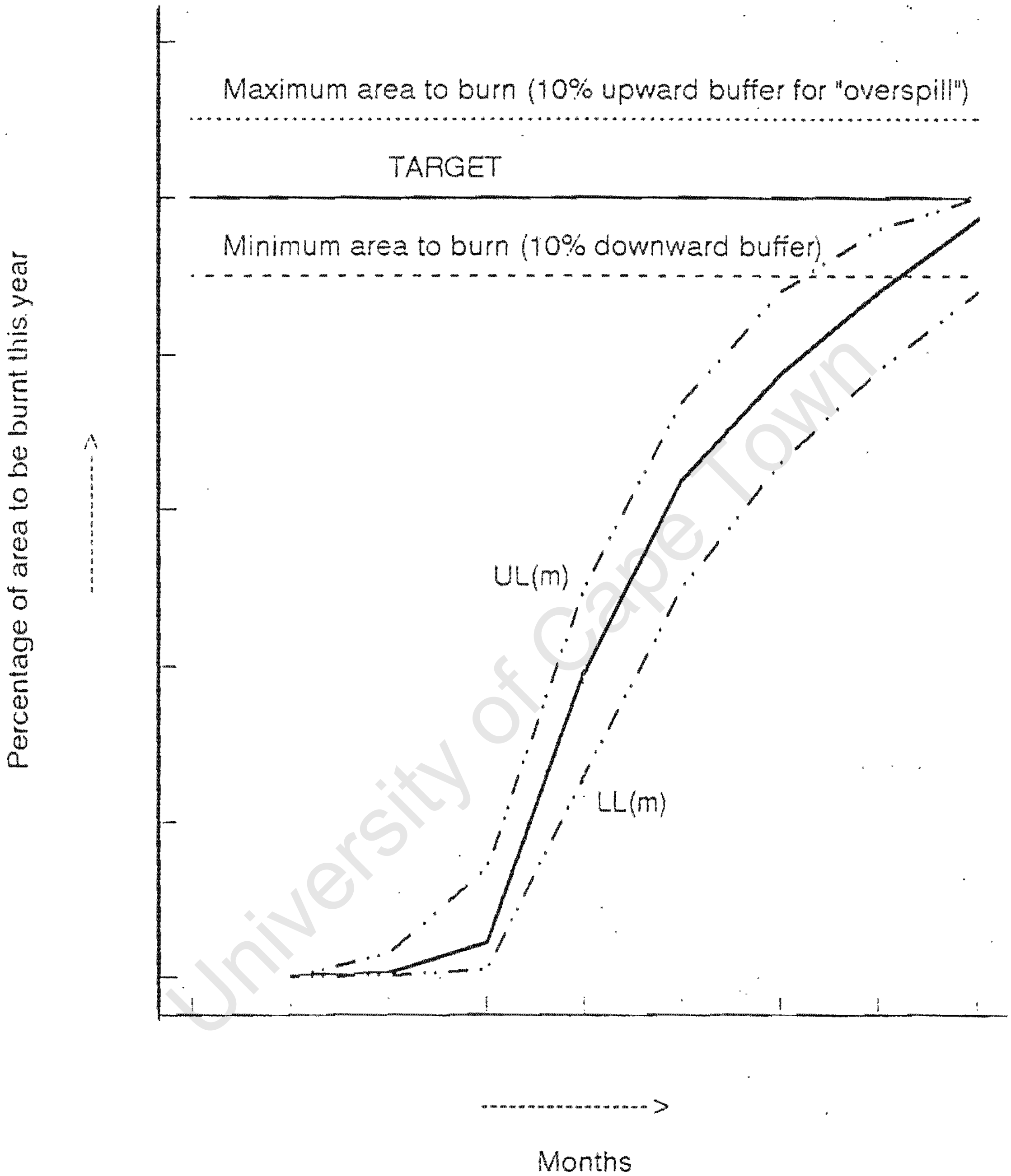

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Figure 6. Increase in percent area burned as the fire season progresses.The area to be burned, as well as proposed monthly minima and maxima are illustrated. Upper and lower monthly limits [UL(m) and LL(m)] are used when decisions on the ignition of subsequent fires are taken (see Figure 2).

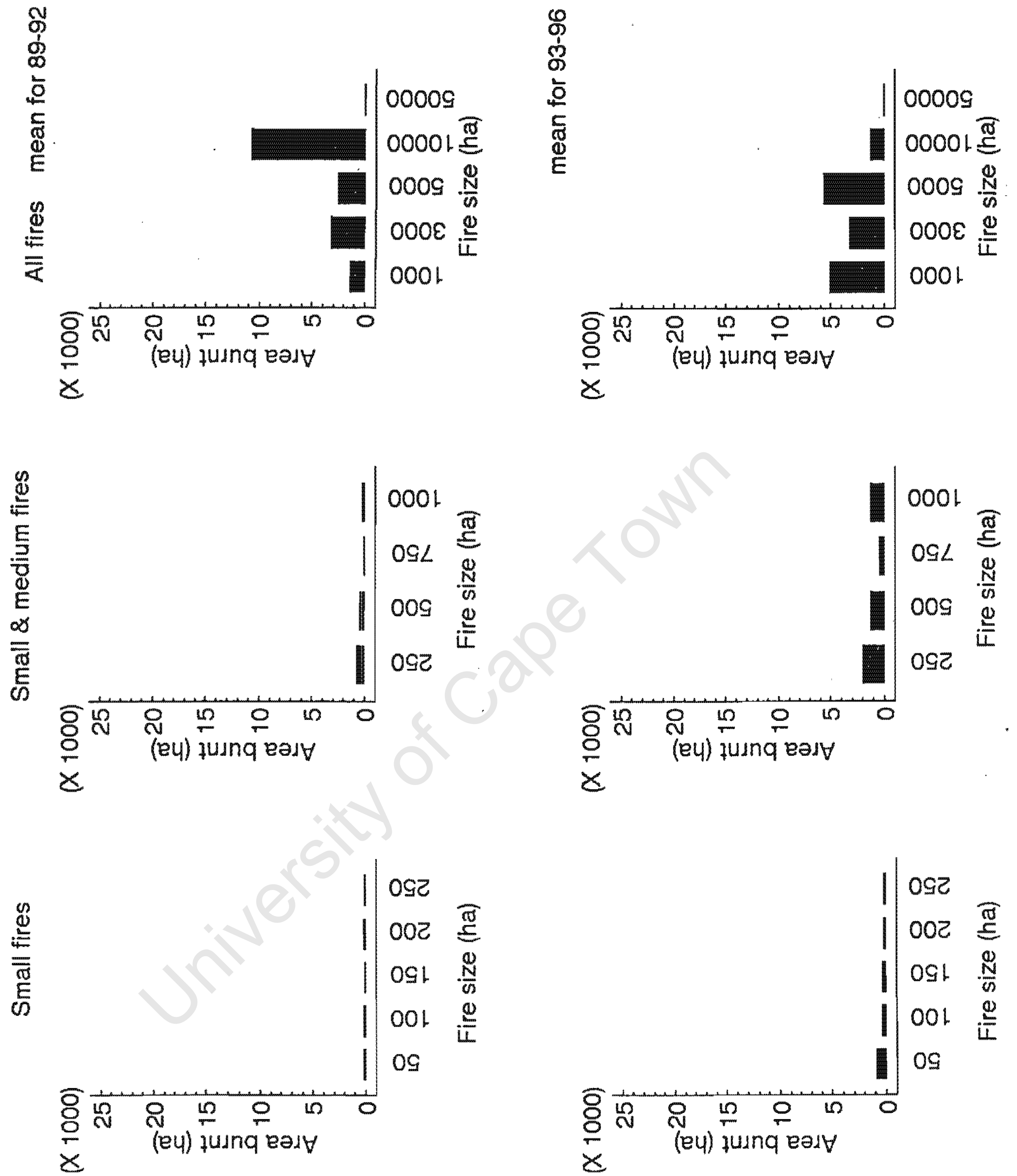

Figure 7. Total area burned (ha) by fires of different sizes in Pilanesberg National Park, South Africa, for two periods: (1) 1989-92, and (2) 1993-96.

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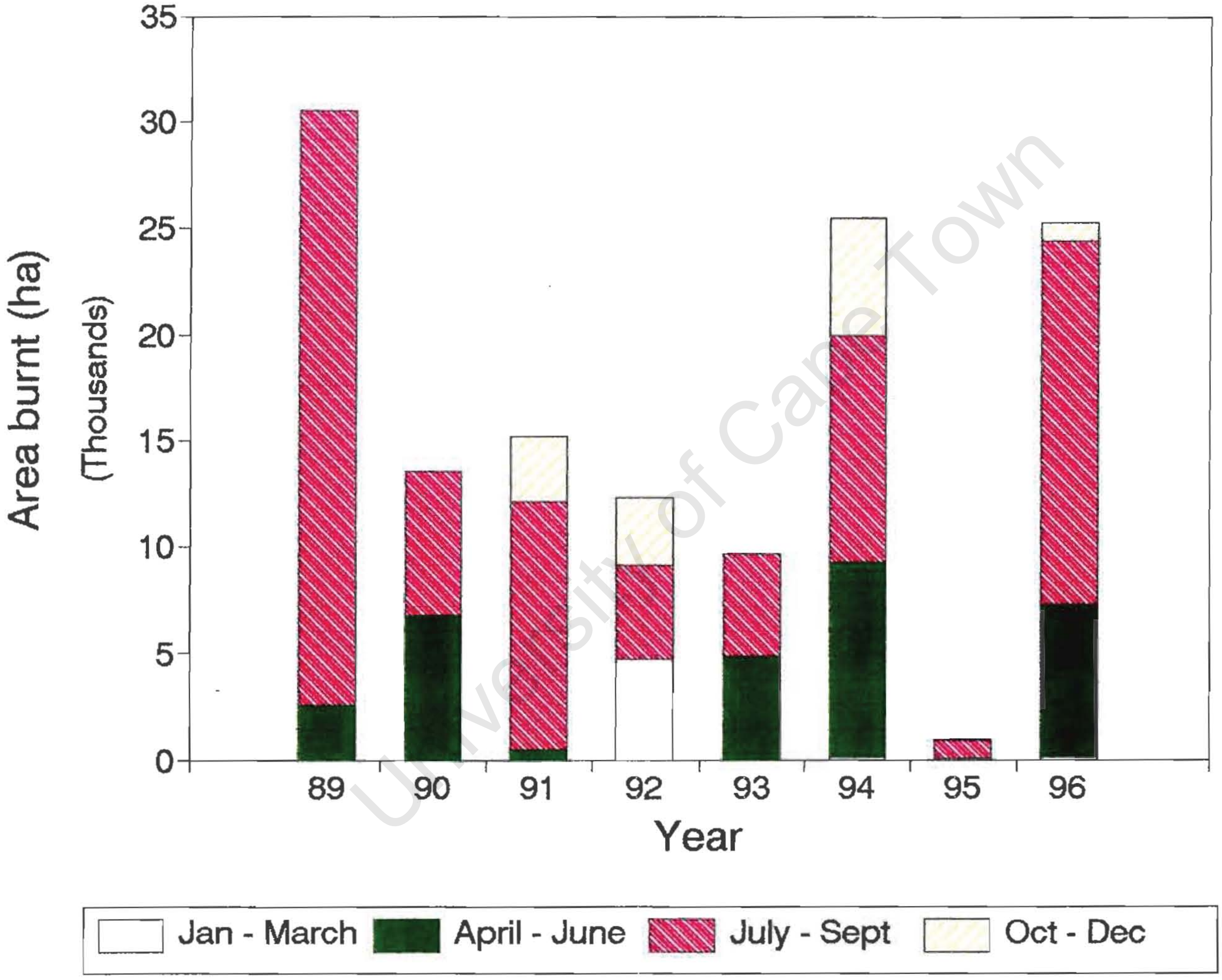

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Figure 8. Seasonal distribution of annual fires in Pilanesberg National Park, South Africa, between 1989 and 1996.

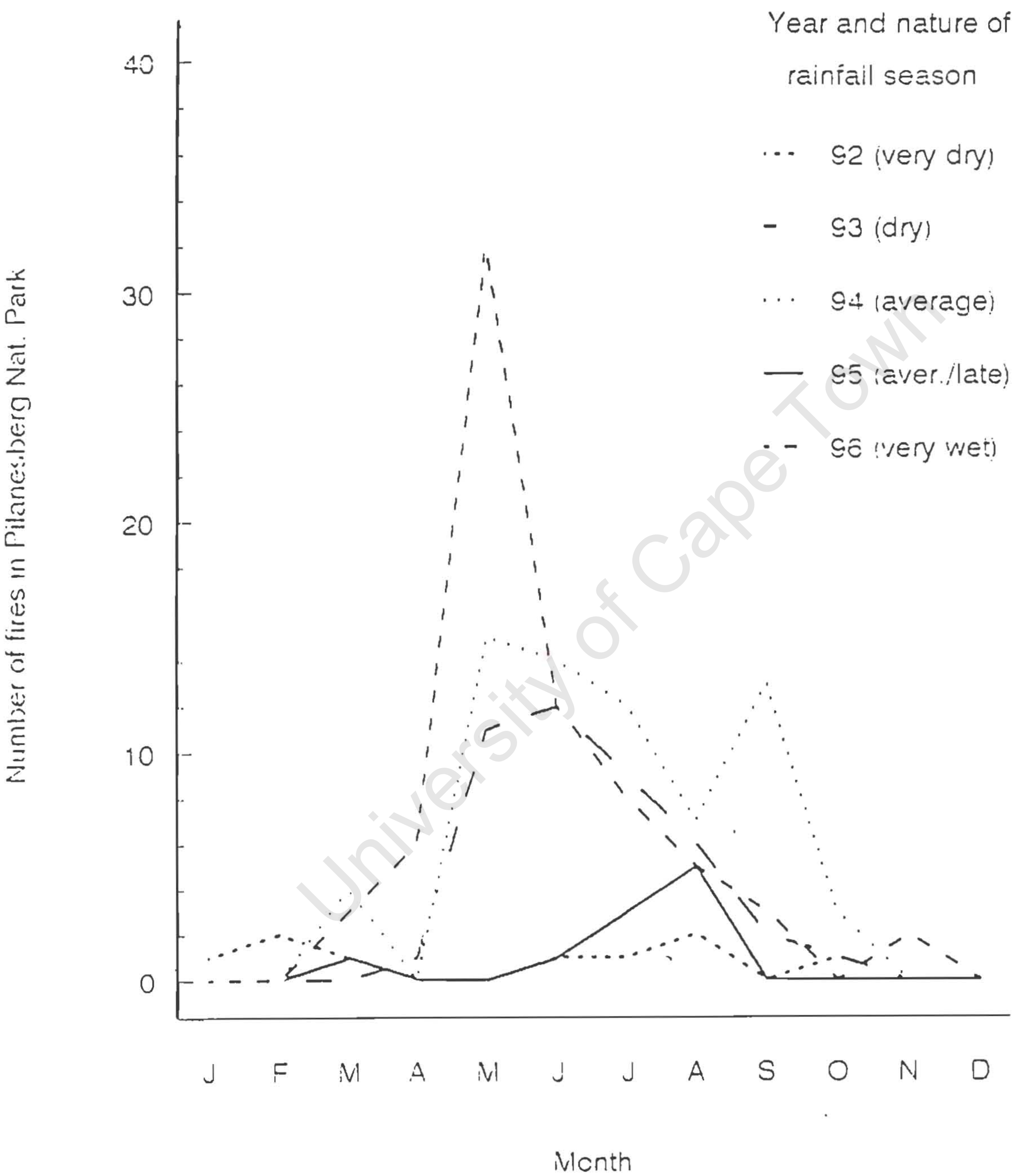

Figure 9. Monthly distribution of the number of fires in Pilanesberg National Park, South Africa, that were ignited in five years (1992 - 1996).

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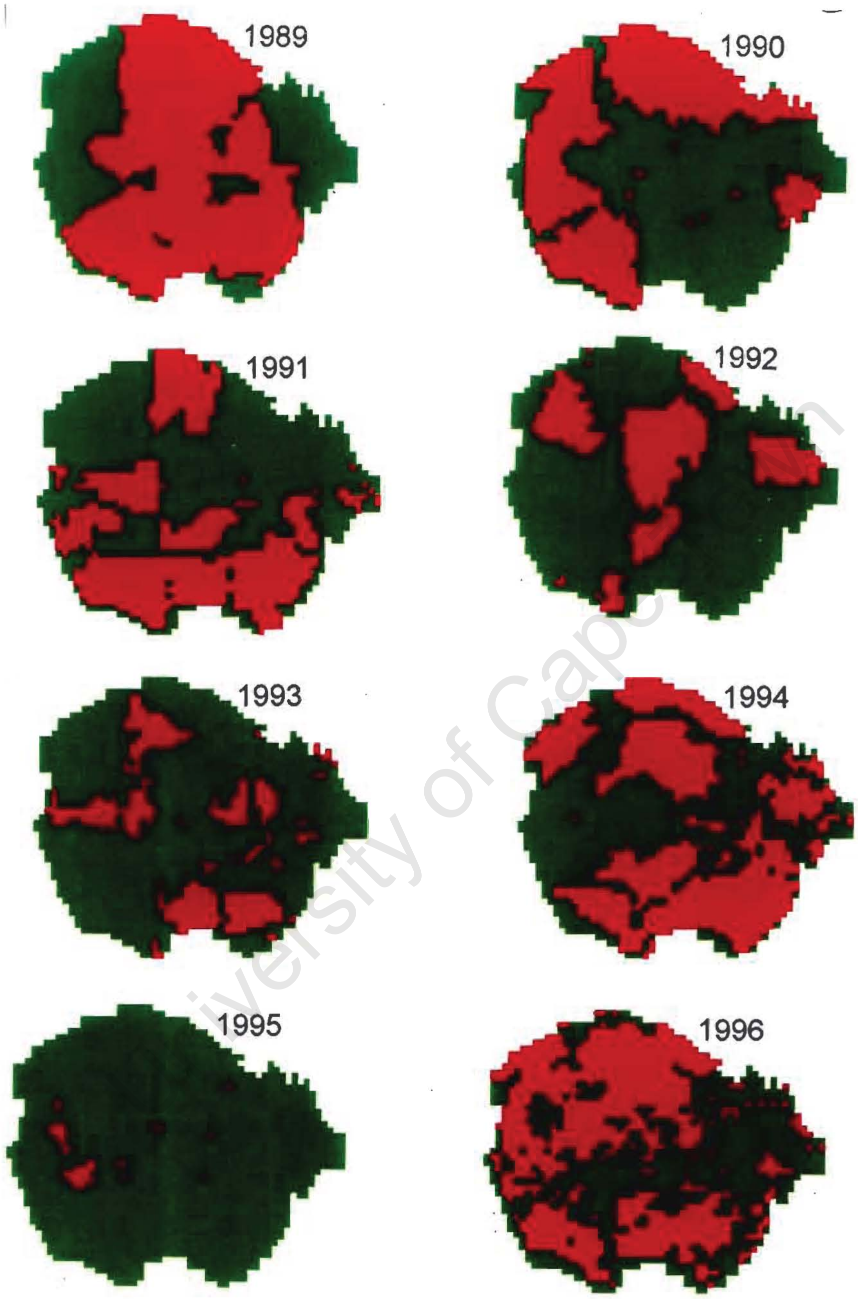

Figure 10. Annual fire maps for Pilanesberg National Park, South Africa, showing the 25 ha pixels that were burned (red) and unburned (green) over 8 years during which a patch mosaic burning system was applied to the area. An increase in heterogeneity is apparent.

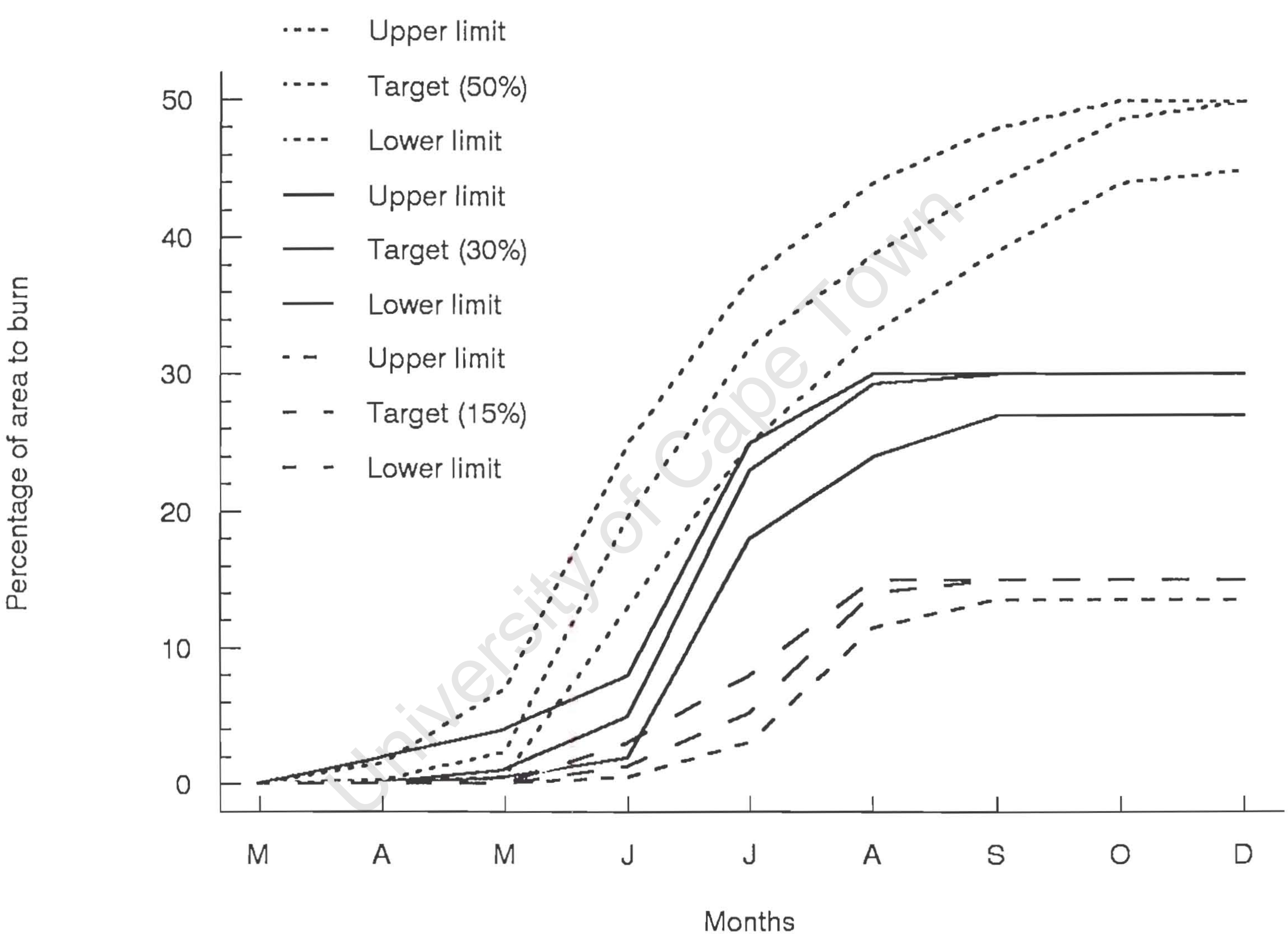

Figure 11. Proposed shapes of curves depicting the increase in percent area burned as the fire season progresses. The curves shown are those for situations where 15, 30 and 50% of the area would be burned.The area to be burned, as well as proposed monthly minima and maxima are illustrated. Families of such curves will need to be developed as more data become available.

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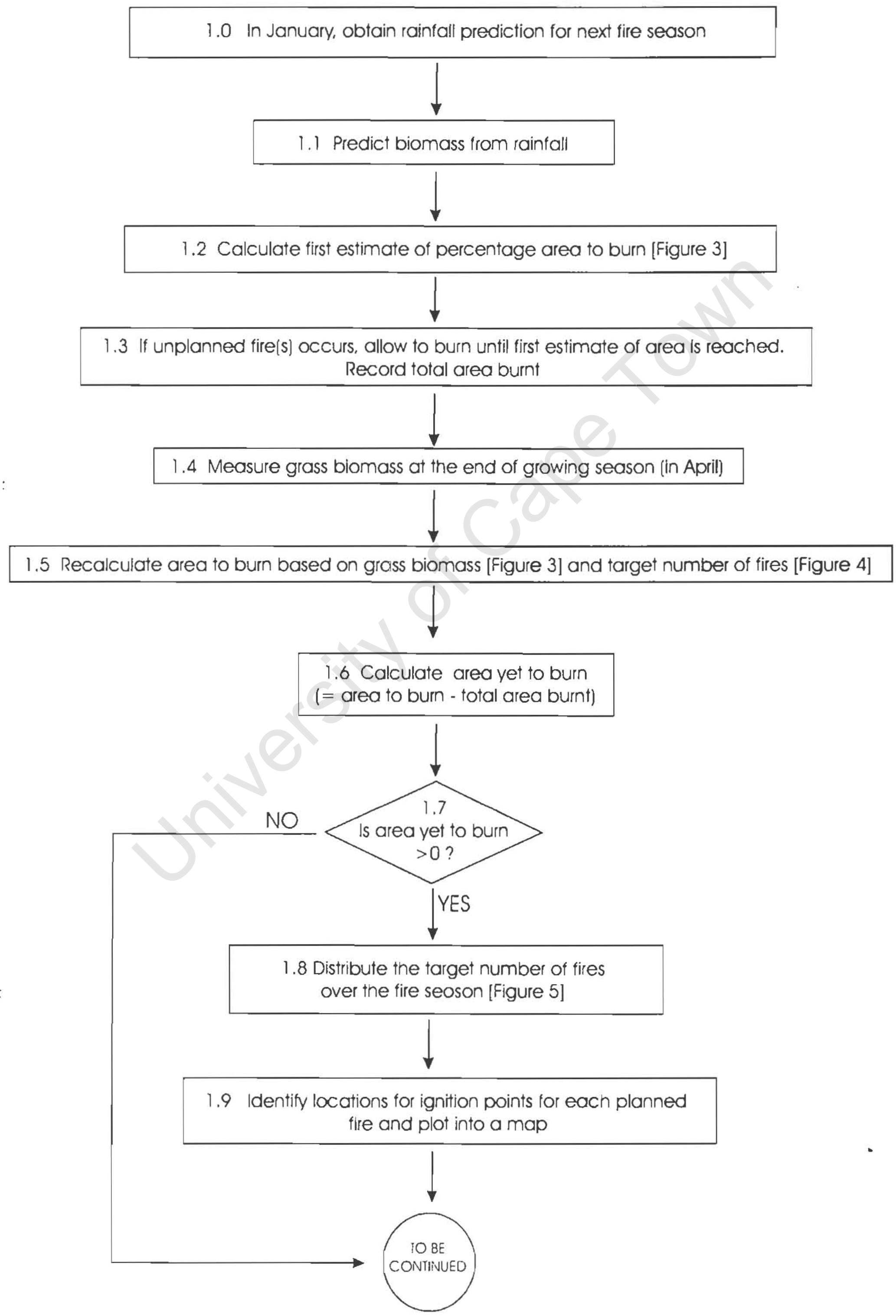
Appendix 1. Detailed procedure for the application of a patch mosaic burning system for southern African savannas.



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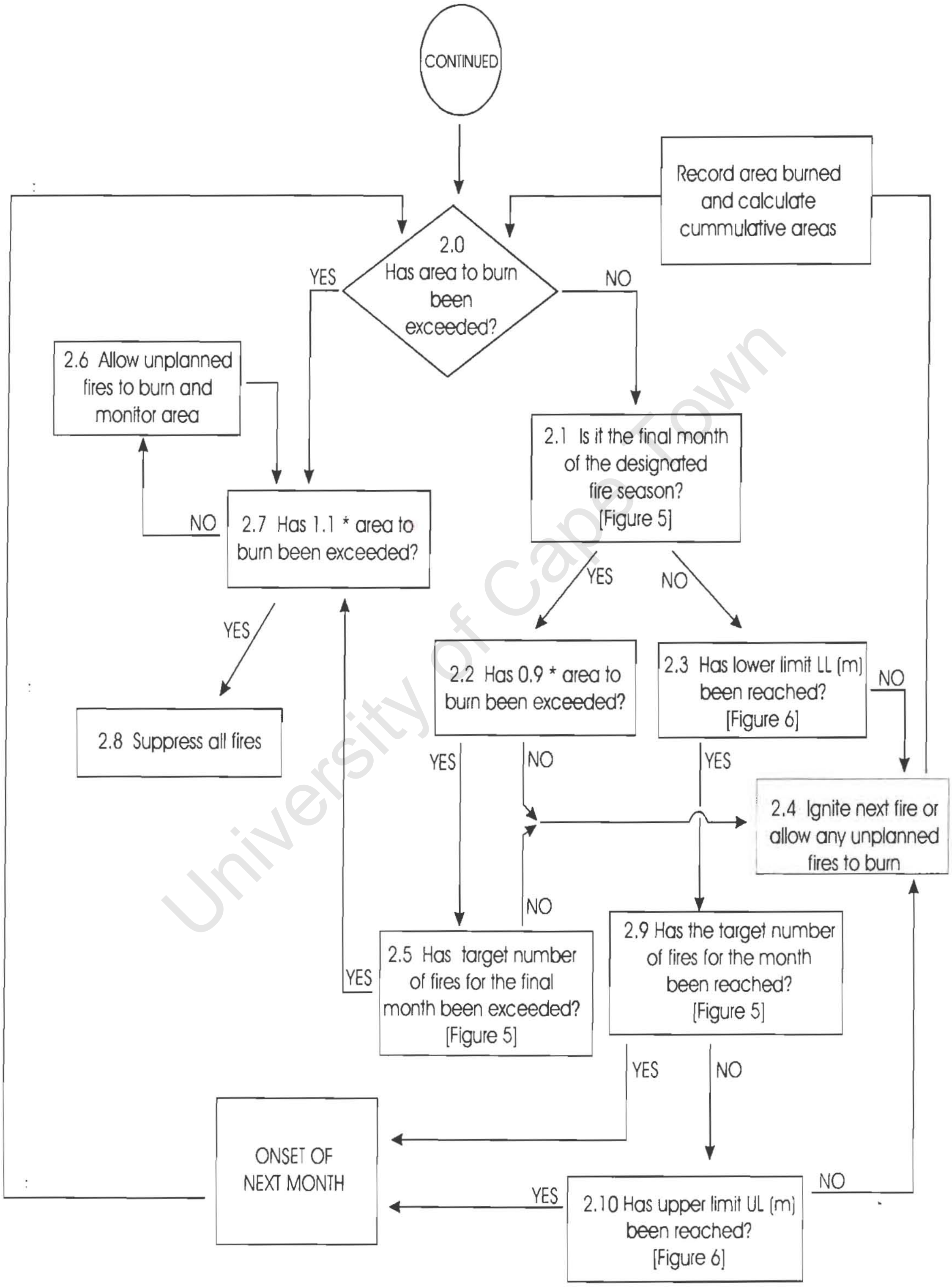

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Chapter 6

Post-fire spatial heterogeneity under two fire management systems in Pilanesberg National Park, South Africa

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Abstract

Fire patterns in Pilanesberg National Park, were examined under two burning systems and each over 9 years: a prescribed block burning system (PBB), and a patch mosaic burning system (PMB). A heterogeneity score (HS) was developed, and scale relationships for this were examined using two grain (25 and 100 ha), and neighbourhood sizes (3 X 3, and 5 X 5). Linear relationships were found between them. Average HS was higher under a PMB system, than under a PBB system. For the PBB system there was no relationship between the HS and the amount burnt the previous year. However, there was relationship with current year. This is likely to be explained as a function of: block sizes selected for burning, and their spatial arrangements to each other, and the percentage of park burnt. For the PMB system the heterogeneity score was positively related to the amount burnt in the current year, and negatively related to the amount burnt the previous year. A two-year comparison, using 25 ha raster maps, showed that the mean for pixels burnt biennially was higher under a PMB

system than under PBB, as was the mean for it being unburnt for two successive years. This suggests that the pattern of fires has shifted under PMB system, and this resulted in variation in fire frequency per pixel, and therefore in fire intensity, and this resulted in variation in fire size. The variation in fire size results in variation in post-fire spatial heterogeneity.

Keywords: burning systems, grain, extent, landscape ecology, resolution, scale, spatial scale, spatial pattern

Introduction

Landscapes are spatially heterogeneous areas and may be observed from many points of view. Ecological processes in landscapes can be studied at different spatial and temporal scales, and spatial pattern plays a role in constraining many ecological processes (Risser 1987; Turner 1989; Turner, O'Neill, Gardner and Romme 1989b; Wiens 1989). Ecologists have been slow in recognize scaling (Wiens 1989). Landscape ecology emphasizes relationships between pattern and process, and the effect that changes in spatial scale have on our ability to extrapolate information across scales (Turner, Dale and Gardner 1989; Turner and Gardner 1991).

Scale can be defined as the spatial or temporal dimension of an object or process, and is characterized by both grain and extent (e.g. size of area or length of time). It is used in the context of space (geographic scale), time (temporal scale), and many other dimensions of

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research (Turner, Dale and Gardner 1989; Goodchild and Quattrochi 1997). If the pattern varies with scale, then the phenomenon is considered *scale dependent*. Scale dependency is an inherent property of landscape phenomena (Cao and Lam 1997). *Resolution* refers to the precision of measurements: grain size, if spatial. *Grain* refers to the finest level of spatial or temporal resolution available within a given data set e.g. pixel size for raster maps. *Extent* refers to the size of the study area or the duration of the study. The spatial scale of ecological data encompasses both grain and extent (Turner, Dale and Gardner 1989; Turner, O'Neill, Gardner and Milne 1989; Goodchild and Quattrochi 1997). Pickett & White (1985) viewed landscapes as a collection of patches undergoing successional change. A "patch" has therefore achieved a central focus in landscape ecology, because it can be visualized as a structural attribute of landscapes (Risser 1987). However, patch definition is often accomplished through an arbitrary selection process, and the patch is identifiable not necessarily by ecosystem properties, but rather because it is different from the matrix or from another patch. There is a high spatial auto-correlation among the entities comprising the patch, and patches defined in this way may have functional boundaries (Risser 1987).

Ecological problems often require the extrapolation of fine-scale measurements for the analysis of broad-scale phenomena (Turner, O'Neill, Gardner and Milne 1989; Turner, Dale and Gardner 1989). Methods linking spatial patterns and ecological processes at broad spatial and temporal scales, are therefore needed in research and in applied environmental problems (Turner, Dale and Gardner 1989; Turner and Gardner 1991). For example, there is a need to measure spatial-temporal heterogeneity at one suitable, and perhaps cost-effective scale, and to be able to relate it to a coarser or finer scales. Simmons, Cullinan and Thomas (1992)

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mention some of the problems facing researchers include: 1) determining how to detect change, and 2) at what appropriate scale to measure change? This requires the development of rules to translate information across scales (Turner, Dale and Gardner 1989).

To translate across scales either a 'top-down' and 'bottom-up' approach could be used (Turner, Dale and Gardner 1989). The top-down approach uses many of the ideas of hierarchy theory (Allen and Starr 1982; O'Neill, DeAngelis, Waide and Allen 1986), to extrapolate between scales. Hierarchy theory provides a logical construct, around which to organize investigations of the structure and function of landscapes (Risser 1987; Turner, Dale and Gardner 1989; Simmons, Cullinan and Thoms 1992). Hierarchy theory demonstrates how processes and constraints change across scales, and how spatial and temporal scales tend to covary i.e. processes which operate over long temporal scales also operate over large spatial scales (long distances) (Goodchild and Quattrochi 1997), and systems at smaller scales change more rapidly, and are more variable than those at larger scales (Simmons, Cullinan, and Thomas 1992). This is the so-called space-time principle (Forman 1995), which applies partly because space and time are linked through transport mechanisms (Goodchild and Quattrochi 1997). Each level of the hierarchy has its own unique properties that are not the simple summation of the disaggregated parts, and that new properties emerge when data are aggregated spatially (Golley 1989; Bian 1997). Only those properties that operate at scales larger than the size of the initial grids can be revealed. As a response to the emergence of new properties, changes are expected in model performance at the corresponding operational scales. Such changes may differ according to the biophysical features under study as well as to the aggregation methods used (Bian 1997).

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The bottom-up approach begins with individual or entity-based measurements and adds appropriate constraints to explain the resultant phenomena at broader scales. The objective here is to use information that is available at fine scales to predict phenomena at broader scales for which empirical data are lacking. Both approaches because of the suite of broad-scale environmental problems facing ecologists and the lack of data at broad scales (Turner, Dale and Gardner 1989). Hierarchy theory provides a conceptual framework for linking spatial attributes to biological functions at one or more scales, but this approach requires quantitative techniques to identify spatial properties of landscapes (Bailey, Gross, Laca, Rittenhouse, Coughenour, Swift and Sims 1996. The development of such knowledge should assist in the management of conserved areas in investigating a number of applied landscape-scale research questions e.g. post-fire habitat fragmentation and relationships to herbivore use, the relationship between fire size, spatial heterogeneity, and ecosystem function (e.g. carbon and nitrogen availability, nitrogen mineralization, and decomposition).

Spatial heterogeneity plays a key role in the structure and function of landscapes (Pickett and Rogers 1997). It exists at various scales (Turner, Hargrove, Gardner and Romme 1994), and constrains the translation of information from one scale to another (see Turner, O'Neill, Gardner and Milne 1989). This may be because heterogeneity may influence processes in nonlinear ways (Turner, Dale and Gardner 1989). Hence one of the central arguments about scale is that data may carry different information when presented at different scales (Bian 1997). This suggests that increasing the level of spatial heterogeneity also increases the difficulty of extrapolating information across scales (Turner, Dale and Gardner 1989). The relationship between ecological measurements and the grain or extent of the data may make

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it possible to predict or correct for the loss of information with changes in spatial scale (Turner, O'Neill, Gardner and Milne 1989). Thus, a consideration of patchiness at different scales in the landscape is important for understanding ecological process.

Disturbance by fire operates over a wide range of spatial and temporal scales (Christensen 1993). Landscapes subject to disturbances by, for example fire, have a patchy structure that is important to many species in these landscapes (Baker 1992). The question of how fire affects landscape heterogeneity is complicated in that spatial heterogeneity exists at various scales (Turner, Hargrove, Gardner and Romme 1994). For example, fire might affect single trees, but at the scale of the fire cycle (which integrates multiple fires), fire is essential for maintaining a mosaic of vegetational structural types (Risser 1987; Scholes and Walker 1993), and Turner, Hargrove, Gardner, and Romme (1994) provide an example of a spatial pattern of burned and unburned patches across a landscape, and of heterogeneity of burn severities within a burned patch. Turner, Hargrove, Gardner, and Romme (1994) hypothesized that the post-fire landscape heterogeneity is a function of the size of fire-created patches. The spread of fire across a landscape may be controlled by disturbance frequency when the habitat is below a critical threshold, but it may be controlled by disturbance intensity when the habitat is above the critical threshold (Turner, Gardner, Dale and O'Neill 1989). Thus, it is important to identify the pertinent scale of both the system and the disturbance events, as well as the role of landscape heterogeneity in the spread and inhibition of disturbance, as this is central to any possible predictive theory (Risser 1987). Under different fire management policies (e.g. fire exclusion, prescribed burning), a shift in landscape-heterogeneity will occur. The causes of landscape change cannot be separated

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without control landscapes that lack prescribed burning, fire exclusion, or other alterations of the 'natural' fire regime (Baker 1992).

The purpose of this paper is to compare the post-fire landscape heterogeneity under two burning systems in Pilanesberg National Park (PNP), South Africa, over a period of 18 years. A block burning system was conducted over eight years (1980-88), and a patch-mosaic burning system over eight years (1989-97). Both burning systems were compared using measures of spatial heterogeneity, and using indexes to describe the fire patterns.

Study area and park objectives

Pilanesberg National Park (PNP), which is approximately 50,000 ha in size, is situated in the North West Province of South Africa. It is hilly country, located on the remnants of an extinct volcano 25 km in diameter and that 1200 million years BP stood 7000 m above the surrounding plain. The vegetation falls in Acock's (1975) Sour Bushveld, a moist (mean annual rainfall 630 mm falling mostly between October and March) savanna with macrophyllous bushes (notably *Combretum*) on the rocky hills, and microphyllous bushes (mostly *Acacia*). Moist savannas are moisture limited, but precipitation is enough for sufficient grass-fuel to accumulate to carry fire after one or two growing seasons following a burn. Total seasonal rainfall for the period from 1980/81 - 1996/97 is presented in Table 1.

PNP is managed to conserve the system's biodiversity, abiotic resources, biophysical processes, unique landscapes and historical/ archaelogical sites, whilst at the same time utilizing the system's renewable natural resources for the enduring socio-economic benefits of the neighbouring communities primarily, and international, national and provincial stakeholders (Boonzaaier & Collinson 2000). Biotic diversity is defined using Noss (1990).

The vegetation is managed to conserve biotic diversity. According to policy fire is one of the characteristic processes of savanna, and consequently fire is being used to produce spatial and temporal patterning of the biota. It is recognised that the presence or absence of fire may temporally enhance/suppress or attract/repel populations of particular species for given areas, but the focus is on maintaining species richness over the whole park, irrespective of events at any particular site. As fire is one of the characteristic processes of savanna, it will be used to augment variation in biotic communities in space and time. However, near fire vulnerable facilities the objective of fire-security overrides that of helping to maintain biotic diversity.

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Fire management systems

Fire records

Fires occurring in PNP was mapped onto a 1:50 000 map, on which a 25 ha grid was overlain. Using the fire's initiation date as a reference, fire scar maps were entered into a spreadsheet in which each cell represented one 25 ha grid square on the 1: 50 000 map. The majority rule was applied i.e. if 50% or more of a grid square on the map was recorded as burnt, then that cell was recorded as burnt. A spreadsheet file was created for each year from 1980 to 1997.

Prescribed block-burning system: 1980-88

Under this system the area is ideally sub-divided into equally sized units of treatment (blocks), which are burnt under controlled conditions. The intention was to burn approximately 33% of the area per year, in order to maintain a three-year fire return period. Burning decision making was guided by veld condition (evidenced by decreaser perennial grasses[2] being moribund), and grass-fuel loads. Each year the oldest blocks were evaluated for burning first using these criteria to achieve the required percentage if possible. If wildfires occurred then subsequent planned block-burns were adjusted (which may have meant that some of the blocks that year would not have been burnt). The percentage of park burnt per year varied as a result of fuel loads, rainfall, veld condition, and previous burns history. The window for prescribed fires was restricted to August- early October. Prescribed fires were applied up the first week of October. There were some modifications to the prescribed fire season later in the period, with small prescribed fires applied in June-July. The objectives of these fires were to create high quality food for grazers in winter, and produce a smaller scale fire-mosaic (Bailey, Brockett & Mentis 1993).

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[2]. Decreasers are defined by Hardy, Hurt and Bosch (1999) as species that predominate in good condition veld but whose abundance declines when veld condition deteriorates through over- or under-utilization.

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Lightning and accidental fires were the major ignition source in 1980, and 1981 (Figure 1). In these years most of these fires were managed as block burns with 'back-burns' applied from the nearest road or track. Such actions by field management in affect created block burns. Prescribed block burns were applied from 1982 onwards. From 1987 onwards smaller block burns (≈25 ha in size) using streams or drainage lines and tourist roads as boundaries were ignited in June - July (Figure 1).

Patch-mosaic burning system: 1989-97

The primary goal of the patch mosaic burning system is to produce landscape-scale heterogeneity as a result of applying a variable fire regime in space and in time. This system is described by Brockett *et al.* (2001), and under this system fires are point-ignited under a range of fuel and weather conditions, and allowed to burn out by themselves. Wildfires are managed according to set of rules. A mosaic, evidenced at various scales, should result (Figure 1).

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Data Analysis

Techniques to quantify spatio-temporal mosaics

There are a number of ways to quantify spatio-temporal post-fire mosaics. Some of these may include: 1) space on time (histogram), 2) time on space, 3) simple overlay, 4) number of fires per year, 5) fire size analysis, and 6) cumulative probability curves for fire recurring at a point. Some of these techniques (e.g. simple overlays) have been available since the mid

1960's (Tomlin 1990). Examples of the cumulative probability curves for fire are for: fynbos vegetation (le Maitre and Midgley 1992), and savanna vegetation e.g. Kakadu National Park (Russell-Smith, Ryan and Durieu 1997), and Kruger National Park (van Wilgen *et al.* 2000). It is suggested that the above methods do not describe fire patterns adequately. With the development of GIS and cartographic modelling (Tomlin 1990), it is possible to explore pattern at different scales, (by aggregating and disaggregating data), and using a number of indexes (at different scales) to describe pattern (see Monmonier 1974; Murphy 1985; Turner 1989; Forman 1995).

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Turner *et al.* (1991) divides techniques in landscape ecology into two broad classes. The first category includes techniques to detect scale(s) over which a repeating pattern occurred. These techniques are generally based on variance measures, and include the use of blocking, auto-correlation, spectral analysis and trend surfaces. The second category addresses methods for patterns that are irregular or may not be repeated. These techniques are used on a neighbourhood of cells (or pixels), and uses variability in a moving pixel window to calculate different measures used e.g. fragmentation (Monmonier 1974), a binary comparison matrix (BCM), and number of different classes (NDC) (Murphy 1985). The second category of techniques will be used to measure heterogeneity of the fire patterns.

GIS packages

Two GIS packages for spatial data analysis were used: IDRISI (Clark Labs for Cartographic Technology and Geographic Analysis, Clarke University, MA, USA), and RESPAN a package that uses Tomlin (1990) cartographic modelling algebra.

Creation of 25 and 100 ha raster maps

Twenty-five hectare raster maps were created for each year using the spreadsheet files. The 100 hectare (1 km^2) raster maps were created using the IDRISI module contract, for each year's burn data two raster files were created: burnt, and unburnt. These raster maps were given a Boolean classification of 1 for burnt, and 0 for unburnt, and contracted using the aggregate function, and then reclassed using the following rules: (1) if three or more pixels were burnt then the 100 ha pixel was classified as burnt; (2) if two or more pixels were unburnt then the 100 ha pixel was classified as unburnt. The resulting files were overlaid (i.e. burnt and unburnt rasters), so as to create the 100 ha rasters.

A measure for post-fire spatial heterogeneity

Spatial heterogeneity of the fire patterns were measured using an index based on NDC (number of different classes) (Murphy 1985). Pixels having more than half their area lying outside the park were excluded from all analyses. A buffer was then created and all windows encompassing one or more pixels outside the park were excluded from this neighbourhood or focal analysis (Tomlin 1990). Then a moving window neighbourhood of, for example 9 pixels (3 x 3, covering 225 ha), was passed across the whole surface for each year. The number of different classes for each moving window were calculated. If all the cells within, for example, a 3 X 3 moving pixel window neighbourhood were burnt (9/9), or not burnt (0/9), then the window was classed as homogeneous burnt, or homogeneous unburnt respectively. For a mixture of cells burnt the window was classed as heterogeneous (Figure 2). A percentage heterogeneity index was calculated as the number of heterogeneous windows expressed as a percentage of the total number of windows (Table 1).

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Changing scale: grain size and extent relationships

For reasons of understanding scale relationships, and to develop techniques, the aim was to establish relationships across two grain sizes, using 25 ha and 100 ha pixel sizes, and using different sized neighbourhoods. The neighbourhoods were varied using different size pixel windows. These were 3 X 3 (9 pixels), and a 5 X 5 (25 pixels) windows. To compare a 3 X 3, and 5 X 5 pixel window the size of the masks needed to be the same. Therefore for the 3 X 3 moving window analysis, a mask the same size as the 5 X 5 moving window was created. The focal product function was used to generated a 5 X 5, and 3 X 3 windows that was passed across the surface. Linear regression relationships were established between these grain, and window sizes.

Heterogeneity scores under different fire management systems

Statistics for the heterogeneity scores under block burning versus patch mosaic burning were calculated, including mean, variance, standard deviation, and median. The heterogeneity scores for the two burning systems were also investigated via relationships between the heterogeneity score, and the area burnt in the current, and previous year.

Two-year comparison

A two year comparison was undertaken from 1980/81, 81/82, 82/83, ... 96/97. To conduct this comparison the number of pixels in the first year that were: (1) unburnt, or (2) burnt, were determined. Then for these pixels how many were: (a) unburnt in both years (U+U), (b)

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burnt in the first year and not burnt in the second year (B+U), (c) unburnt in the first year and burnt in the second year (U + B), and (d) burnt in both years (B + B).

Post-fire fuel age maps

Post-fire fuel age was calculated per pixel at the end of the period, for each burning system i.e. in 1988, and in 1997. A post-fire fuel age map (each pixel's value was the last year in which it burned) was then produced. Such maps are also called time-since-fire maps (see Reed, Larsen, Johnson, and MacDonald 1997; Weir, Johnson, and Miyanishi 2000).

Results

Changing scale: grain size and extent relationships

Using the same grain sizes (either 25 or 100 ha), but with different neighbourhood sizes the slope and intercepts for the linear regression equations were significant (Table 2). If one were to predict the heterogeneity score at a coarse grain (100 ha) compared with a fine grain (25 ha). The result would be approximately (see Table 2):

$y = \approx 10 + 1.2 \text{ X HS}$

where

HS = Heterogeneity Score at the finer scale.

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In the case of different sized neighbourhoods (3 X 3, and 5 X 5), slope was significant. The intercept was not significant, and consequently the regression was forced through the origin. To predict relationships between scales using a larger window (5 X 5), compared with a smaller window (3 X 3), the result would be:

$y = \approx 1.3 \times HS$

where

HS = Heterogeneity Score at smaller window (Table 2).

Correlation coefficients (r) were highest using a 25 ha, followed by 100 ha grain size. The lowest correlation was using a different grain sizes, and window size (Table 3). To make the comparison between a 3 X 3 and 5 X 5 sized windows equal, the heterogeneity scores for the 5 X 5 window could be varied so that homogenous included 0/25, 1/25, and 2/25. Then to establish relationships between the scores. The heterogeneity index used for further work were those using a 3 X 3 moving window, and a grain size of 25 ha. The bigger mask was also used i.e. not the mask used for the 5 X 5 window.

Discussion and Conclusions

Changing scale: grain size and extent relationships

Linear relationships were found between the two grain, and neighbourhood sizes investigated. The next step would be to examine these relationships using two smaller (e.g. 1 and 6.25 ha) grain sizes, in addition to the 25 and 100 ha, and using different neighbourhoods

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sizes. Then to use a square root scale i.e. 1 (1 ha), 2.5 (6.25 ha), 5 (=25 ha), and 10 (=100 ha) to relate these to each other. Satellite imagery at either 30 or 20 m resolution would be needed for the smaller grain sizes. This is currently being investigated using Landsat TM imagery.

Comparisons between the two burning systems

Average heterogeneity score under the patch mosaic burning system was approximately twice that as under prescribed block burning (35.0 ±15.11 cf. 19.3 ±6.33) (Table 4). The coefficient of variation was also higher with patch mosaic burning as compared with block burning. The maximum heterogeneity score was recorded under patch mosaic burning in 1996 (61.83%), and the lowest using a prescribed block burning in 1981 (Table 4).

For block burning there was no relationship between the heterogeneity score and the amount burnt the previous year. There was a relationship between the heterogeneity score and the area burnt in the current year (Table 5) (Figure 3). However, this later relationship is likely to be a function of the block sizes selected for burning, their spatial arrangements, and the percentage of park burnt that year.

For the patch-mosaic burning system, it was clear that the heterogeneity score was a function of a number of variables and these include the: 1. area burnt in the current year (Brockett *et al.* 2001), and 2. area burnt the previous year, 3. spatial distribution of fires, 4. number of fires, and 5. size of fires. For example, it was low in 1995 because of the small area burned

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in that year. The heterogeneity score was positively related to the amount burnt in the current year (Figure 3), and negatively related to the amount burnt the previous year (Table 5). This suggests a relationship between the pixels burnt (or unburnt) in one year, and unburnt (or burnt) in the following year, and this is explored later. Reason for the increase in heterogeneity in the years 1994 and 1996, lies in the wide distribution of smaller fires across the park (Figure 1), compared to the few larger fires in the earlier years (eg 1989) which left wide areas homogenous (either burned or unburned, Figure 1).

The two year comparison showed that in 1980/81 a high percentage (66%) of pixels that burnt, burnt in both years (Table 6). This was as a result of the large wildfires that occurred in 1981. This was also the highest number of pixels that burnt biennially over the period studied (18 yrs). With the change to a patch mosaic burning system there were a number of years for which pixels were burnt in both years (93/94, 90/91, 89/90, and 95/96). In years of droughts, or below average rainfall, and for early block burning two year comparisons, either no pixels (1982/83, 1984/85, 1994/95), or a small percentage of pixels (1981/82 (1), 1983/84 (1), 1985/86 (1), 1992/93 (3)), were recorded as being biennially burnt. Of these only 1992/93, and 1994/95, were under patch mosaic burning, and all the remaining are from a block burning system. If a pixel was burnt in year 1, then there was a greater chance (using the higher mean), that it will be burnt in year 2, with a patch mosaic burning system, than with a block burning system. Therefore there was a greater chance of pixels being burnt biennially with a patch mosaic burning system, than with a prescribed block burning system. The chances of a pixel being unburnt two years in a row, were also higher with this burning system. A patch mosaic burning system has therefore shifted the pattern of fires in PNP, and

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this has resulted in variation in the fire frequency per pixel, and fire intensity, and therefore in fire size. The variation in fire size then results in variation in post-fire spatial heterogeneity. This is illustrated by the post-fire fuel age mosaic map (calculated per pixel) which shows greater variation under a patch mosaic burning system than under a block burning system. With a proportion of the area's post-fire fuel age ranging between 0 - 8 years old. Little variation was evident under block burning with most of the pixels being between 0 - 3 years old (Figure 4).

To explain such patterns, Minnich (1983) working in California, suggested that perhaps the most important variable affecting the spatial properties of fires is antecedent fire, as it is common for an active fire to stop at former burns because of insufficient fuel. Moreover, the extent of overlap between previous burns and active fires over some reference period should decrease as the time required for vegetation to reach flammability increases (Minnich 1983). Thus in slow-maturing communities the growth of fires is more constrained by previous fire history than in fast-maturing ones; fire size should be inversely related to fire frequency. This is evidenced by the change in proportions of pixels burnt biennially over the period studied. Herbaceous production in a savanna is strongly linked to water availability (Scholes and Walker 1993), and assuming that flammability in a savanna system is primarily a function of biomass, then if water availability declined (in periods of below average rainfall), herbaceous production would decrease, and fire size would decline. This appears to have happened under a patch mosaic burning system. The patterns observed are therefore partly also a function of rainfall. Parr and Brockett (1999) developed a conceptual model to illustrate how fire size may be determined by fuel characteristics (fuel load etc.), terrain and

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number of ignitions. In this model fire size was nested under fire intensity and fire frequency. What may be needed would be to investigate these relationships on random rasters, and see what relationships are derived, and to then compare these relationships with the above (Brockett et.al 2001). Christensen (1993) suggests that high-intensity fires (cf. Yellowstone National Park at a scale of hundreds to thousands of metres) create ecologically important variation at nearly every spatial scale. The relationships between pattern and process in response to fire requires investigation in African savanna ecosystems. This may require a description of fire behaviour and post-fire effects at two spatial scales as was conducted in Kakadu National Park as part of the Kapalga landscape-scale fire experiment (see Andersen, Braithwaite, Cook, Corbett, Williams, Douglas, Gill, Setterfield, and Muller 1998; Williams, Gill and Moore 1998), and then relate the fire behaviour to a heterogeneity score. Further research should also explore pattern indexes (using an appropriately grain sizes and neighbourhood), to provide a measure of habitat suitability appropriate for different rare herbivore species e.g. sable antelope, and to relate this to the fire pattern. The complexity of landscape patterns can also be studied by using fractals. Fractals can be used to measure the spatial patterns of a diversity of quantities (e.g. patch mosaics, movement patterns, and density of organisms) that contribute to heterogeneity within a landscape. In addition, fractals offer a means of examining landscapes and ecological processes at a multitude of scales (Turner and Gardner 1991).

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Table 1. Total season rainfall from recording stations in Pilanesberg National Park, with the percentage of mean rainfall, and classification of each season rainfall using the percentage of mean. Mean annual rainfall was 630 mm.

Season	No. of recording stations	Rainfall (mm)	%-age of mean	Classification
1980/81	1	597.8	95	Below average rainfall
1981/82	3	521.6	83	Below average rainfall
1982/83	3	374.1	59	Drought
1983/84	3	466.3	74	Drought
1984/85	3	563.5	89	Below average rainfall
1985/86	3	422.6	67	Drought
1986/87	4	590.7	94	Below average rainfall
1987/88	4	658.9	105	Above average rainfall
1988/89	4	691.9	110	Above average rainfall
1989/90	4	696.2	111	Above average rainfall
1990/91	6	735.3	117	Above average rainfall
1991/92	8	284.0	45	Drought
1992/93	8	505.7	80	Below average rainfall
1993/94	8	583.6	93	Below average rainfall
1994/95	8	569.7	90	Below average rainfall
1995/96	9	813.5	129	Above average rainfall
1996/97	9	914.1	145	Abundant Rainfall

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Table 2. Fitted linear regression equations and statistics for heterogeneity indexes at different grain (pixel) sizes, and neighbourhoods: r = correlation coefficient, r^2 = coefficient of determination, using a sample size (n) of 9.

Dependent variable	**Independent variable**	**Intercept**	**Slope**	**r**	**r^2 (%)**	**Significance**
5 X 5 25 ha	3 X 3 25 ha	9.150	1.259	0.992	98.41	Intercept p = 0.00562, slope p = 0.00
5 X 5 100 ha	5 X 5 25 ha	0 *	1.272	0.992	98.45	Slope p = 0.00
5 X 5 100 ha	3 X 3 100 ha	13.015	1.178	0.979	96.03	Intercept p = 0.027, slope p = 0.00
3 X 3 100 ha	3 X 3 25 ha	0 *	1.290	0.991	98.24	Slope p = 0.000

* Intercept forced through origin, because p of intercept >0.5.

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Table 3. Sample correlation coefficients, and significance levels between brackets, between a 5 X 5 moving window, and a 3 X 3 moving window, and a 25 and 100 ha grain size, with a sample size (n) of 9.

	5 X 5 25 ha	3 X 3 25 ha	5 X 5 100 ha	3 X 3 100 ha
5 X 5 25 ha	1.000 (0.0000)	0.9920 (0.0000)	0.9163 (0.0005)	0.9496 (0.0001)
3 X 3 25 ha	0.9920 (0.0000)	1.0000 ((0.0000)	0.8823 (0.0016)	0.9311 (0.0003)
5 X 5 100 ha	0.9163 (0.0005)	0.8823 (0.0016)	1.0000 ((0.0000)	0.9799 (0.0000)
3 X 3 100 ha	0.9496 (0.0001)	0.9311 (0.0003)	0.9799 (0.0000)	1.0000 ((0.0000)

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Table 4. Statistics for a post-fire heterogeneity score for a prescribed block burning versus a patch mosaic burning system in Pilanesberg National Park, South Africa, where n = number of observations, SD = Standard deviation, CV = coefficient of variation, and Max = maximum heterogeneity score, and MIN = minimum heterogeneity score.

Statistic	Block burning	Patch mosaic burning
n	9	9
Mean	19.31	35.01
Variance	40.15	228.4
SD	6.336	15.113
Median	20.7	31.1
CV (%)	32.81	43.16
Max	27.3	61.83
Min	9.35	23.59

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Table 5. Linear regression relationships between the heterogeneity score (HS), and the number of 25 ha pixels in the previous or in the current year, with sample size (n), number of degrees of freedom (df), and coefficient of determination (r^2).

Burning System	Dependent variable	Independent variable	n	df	Intercept	Slope	r2
BB	HS	Previous year	8	6	21.322	0	0.02
PMB	HS	Previous year	9	7	55.644	-0.031	0.586
BB	HS	Current year	9	7	31.652	-0.018	0.465
PMB	HS	Current year	9	7	21.776	0.0213	0.307

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Table 6. A two-year comparison between area burnt (B) or unburnt (U) expressed as a percentage of those either unburnt in the first year, or burnt in the first year.

Period	U + U/1st yr unburnt (%)	B + U/1 st yr burnt (%)	U + B/1 st yr unburnt (%)	B + B/1 st yr burnt (%)
80/81	40	60	34	66
81/82	32	68	99	1
82/83	64	36	100	0
83/84	52	48	99	1
84/85	69	31	100	0
85/86	51	49	99	1
86/87	39	61	96	4
87/88	66	34	85	15
88/89	20	80	80	20
89/90	44	56	61	39
90/91	59	41	59	41
91/92	63	37	88	12
92/93	74	26	97	3
93/94	48	52	49	51
94/95	96	4	100	0
95/96	48	52	64	36
96/97	86	14	84	16
Mean for block burning	48.11	51.88	88	12
Mean for patch mosaic burning	64.75	35.25	75.25	24.75

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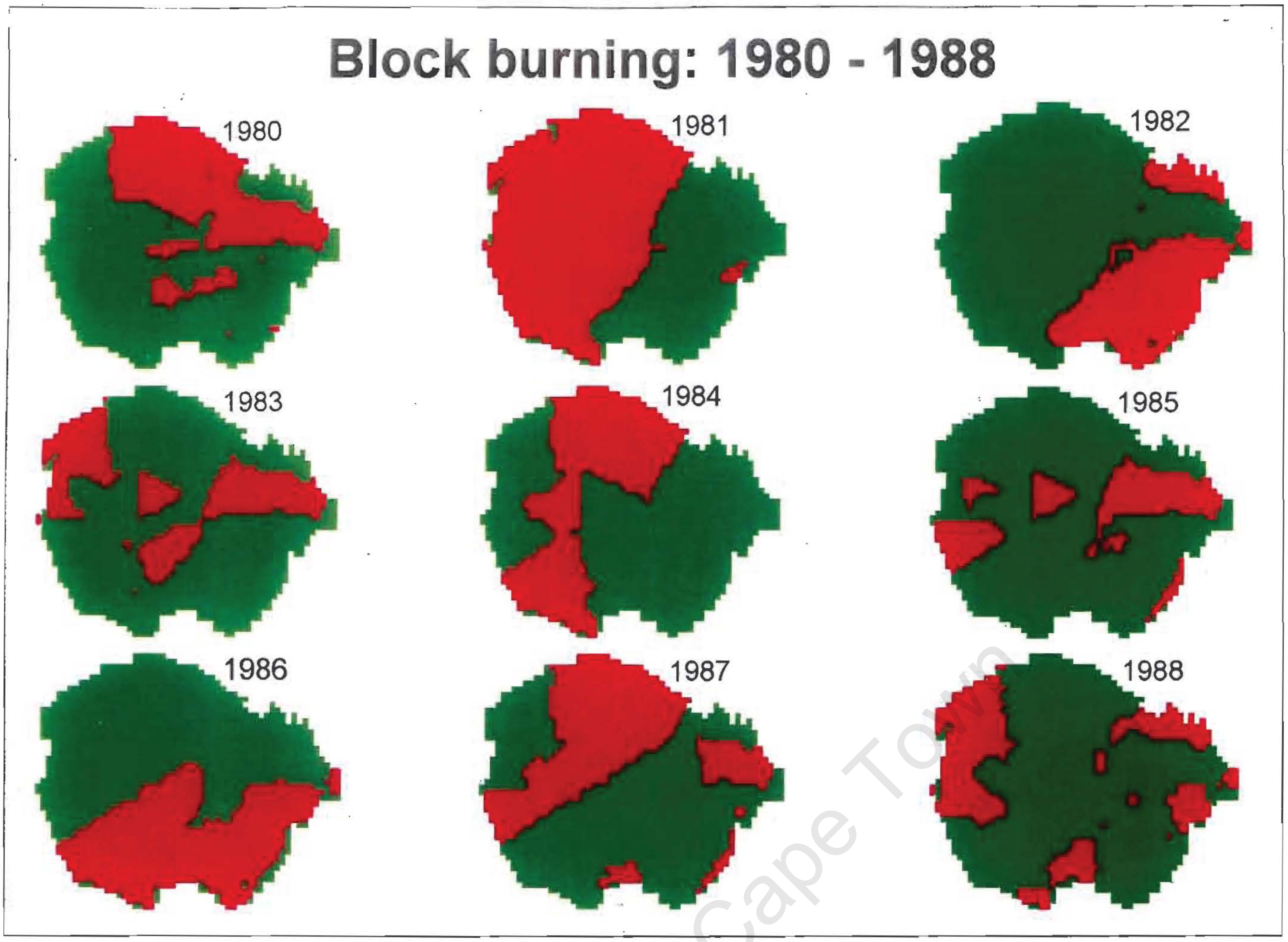

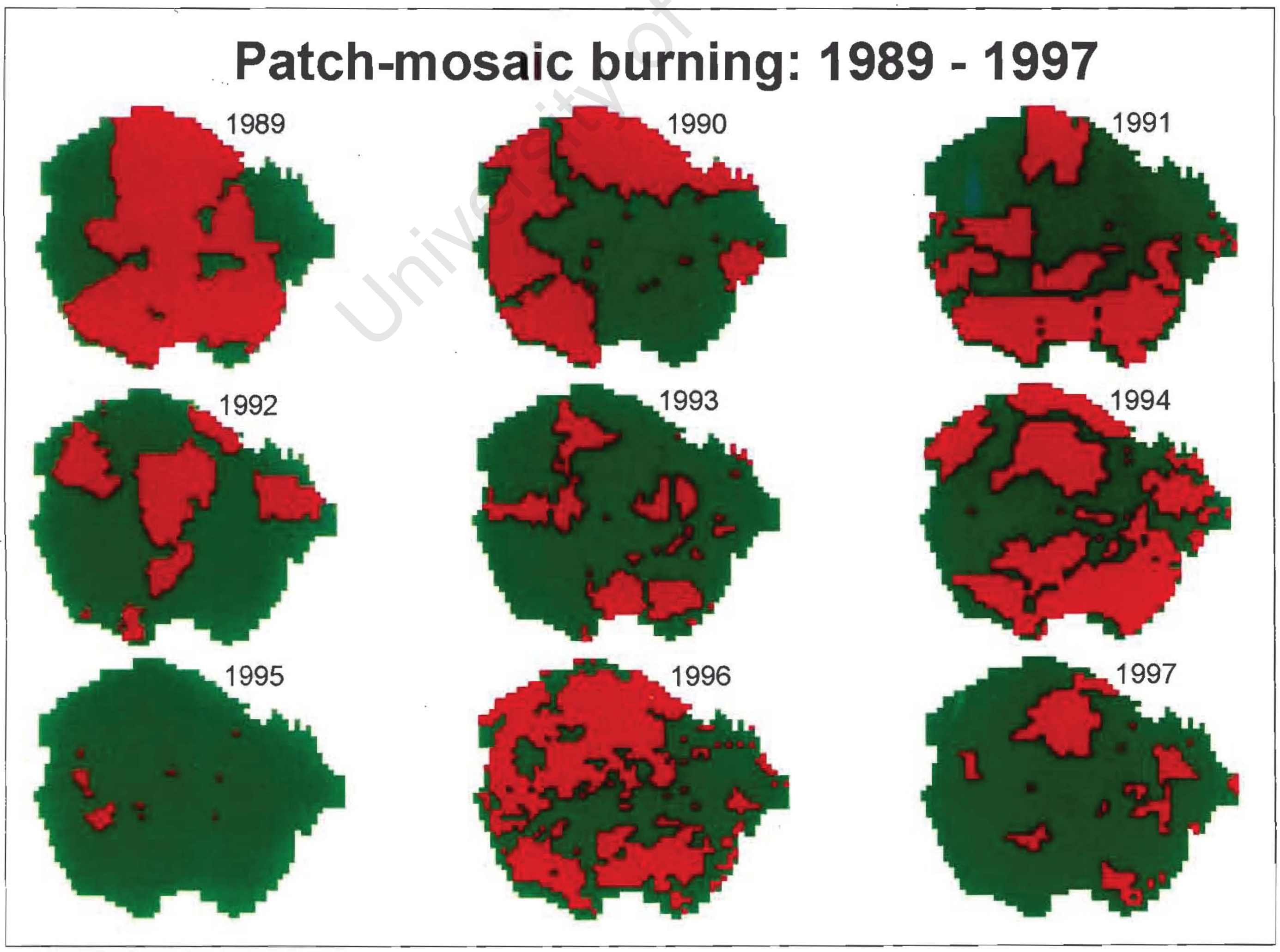

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Figure 1. Twenty-five hectare raster maps for two burning systems: block burning (1980-88), and patch mosaic burning (1989-1997).

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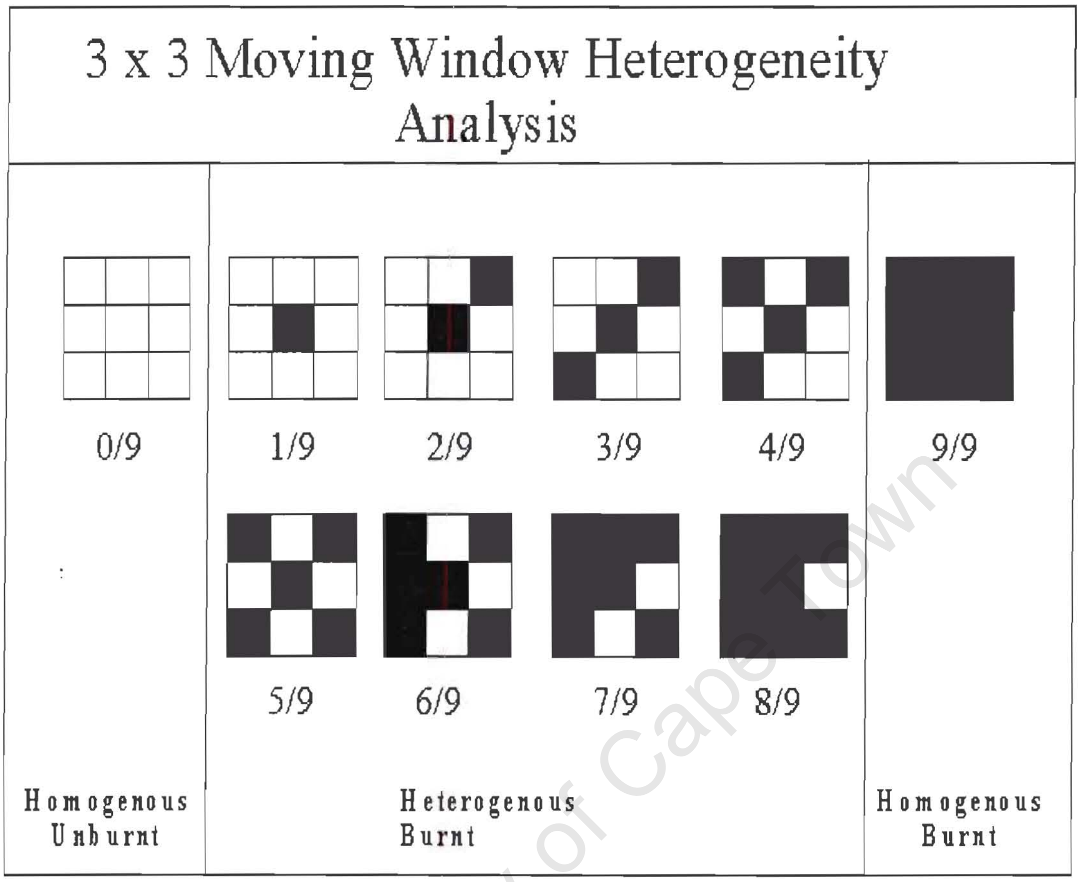

Figure 2. Classification using the variability within a 3 X 3 pixel moving window, to develop a heterogeneity score, showing the classifications used for homogenous unburnt, heterogenous burnt, and homogenous burnt.

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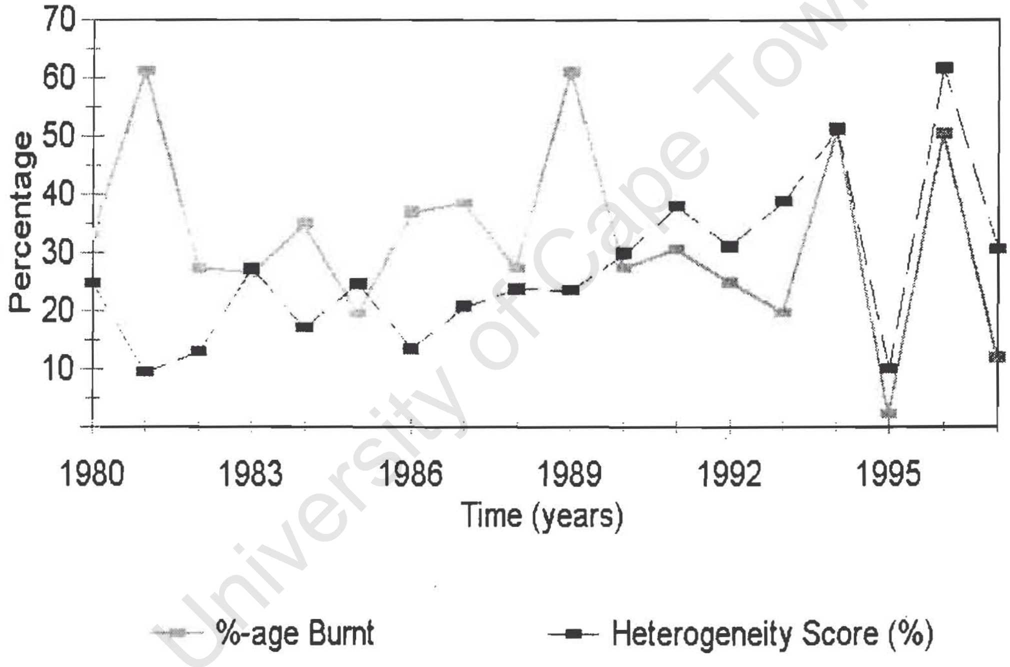

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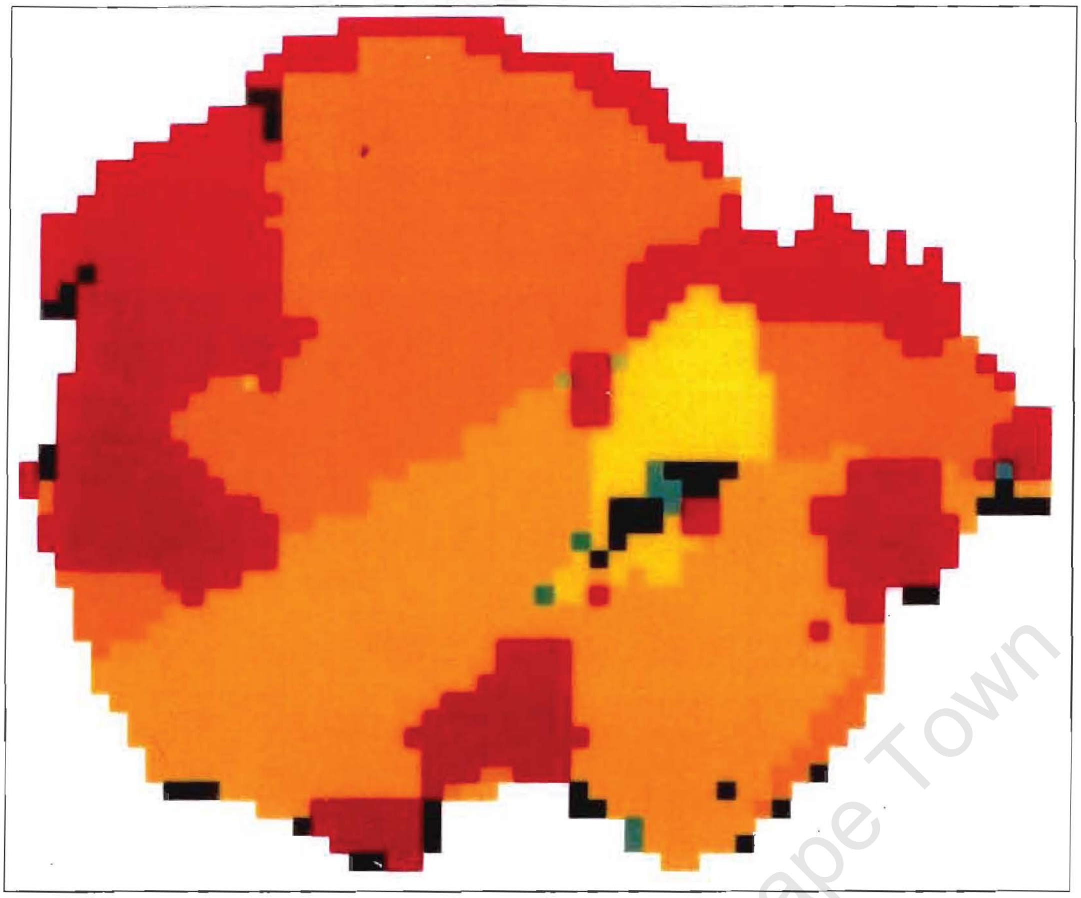
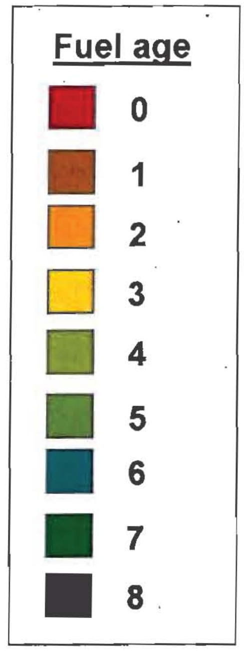
Figure 3. Percentage heterogeneity score, and percentage of park burnt per year for Pilanesberg National Park.

Post-fire fuel age from 1980 - 1988





Post-fire fuel age from 1989 - 1997

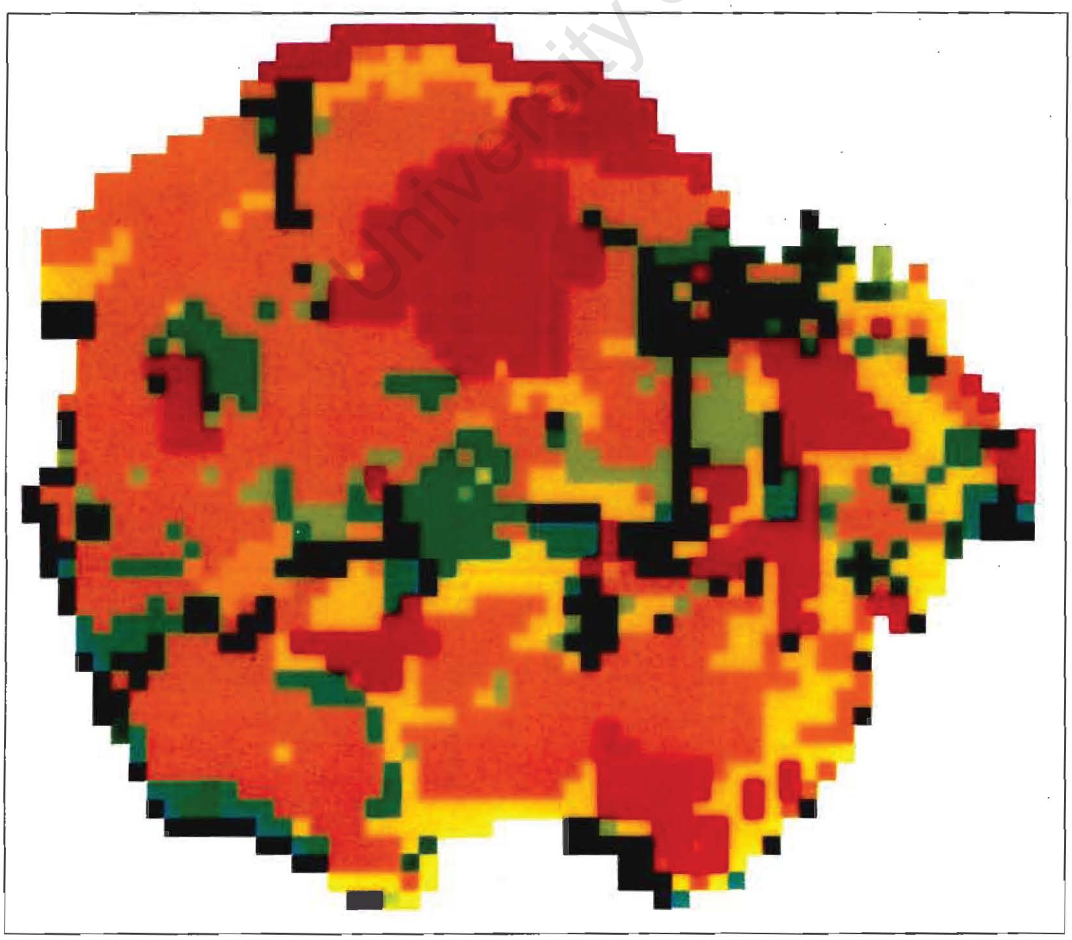


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Figure 4. Post-fire fuel age for prescribed block burning (1980-88), and for patch-mosaic burning (1989-97), for Pilanesberg National Park.

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Chapter 7

Summary

A patch mosaic burning system is achieving an aim of maintaining a variable fire pattern. This is evident by the reduction in fire size that occurred since the implementation of the system over a period of nine years. A reduction in fire size was achieved as a result of greater variation in fire frequency per 25 ha pixel, in Pilanesberg National Park, than under a prescribed block-burning system over the same period of time. A variable fire return period per pixel, should result in variation in intensity, and consequently in fire size.

The post-fire heterogeneity score developed reflects changes in some of these fire parameters. However, it is necessary to explore other landscape ecological measures and indices to describe pattern, and their relationships with fire parameters e.g. fire size. In addition it is necessary to explore fire/habitat/herbivore relationships at appropriate scales.

What the effects of such variation on biotic diversity still need addressing. Attempting to do this may require a multi-disciplinary approach, to test the assumptions in terms of the conservation of biotic diversity. This requires that a vegetation monitoring system applied at appropriate scales, and frequencies, and in relation to the specified objectives. This requires

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structural and compositional monitoring, and broad-scale pattern monitoring (using aerial photography, satellite imagery, and videography) at appropriate resolution.

Management involves making calculated, and either unprogrammed, or programmed decisions. Adaptive management of protected areas is characterized by ignorance and uncertainty. Consequently decision making involves risk taking. How can risk be reduced? One way would be to better integrate the functions of modelling, monitoring and management. By management I mean interventions e.g. burning. With respect to goal setting, modelling can assist by clarifying objectives and values, and developing measurable goals. Modelling can assist in generating a range of possible (and imaginative) solutions. The solutions can then be ranked with respect to the goals, and then one selected and applied. However, any decision analysis requires review, and updating as circumstances change. A qualified ecologist needs to be closely integrated into protected area management. Also given the lack of knowledge characteristic of savannas and to make more informed decisions (i.e. to choose between various alternatives), it may - especially for large conservation areas - require the setting-up of a landscape-scale experiments. Therefore the proposed LASHFIRE - Landscape-scale Herbivory-Fire Interaction Research Experiment in the Kruger National Park is important. It is suggested that the example of the Kapalga experiment in Kakadu National Park, northern Australia be followed in southern African savannas.

As technology develops it needs to be transferred to park management, and this may require the development and maintenance of a decision-support system. It is also recommended that managers of African conservation areas should be exposed to decision analysis procedures.

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For example, such procedures could be used as to which fires to control by back burning, and which fires to leave to burn. As experience develops so such decision-aids may become less useful, but such an aid can be used to analyse a situation in retrospect and see whether the questions asked may generate other alternatives.

Satellite imagery should soon prove useful to management to determine not only the extent of area burnt, but also the fire intensity (or severity). As this technology develops so the cost per hectare will decrease. A fire history for an area developed using satellite imagery should prove useful in investigating vegetation fire effects.

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