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The effect of fire season, fire frequency, rainfall and management on fire intensity in savanna vegetation in South Africa

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Summary

1. Fire is important for the maintenance and conservation of African savanna ecosystems. Despite the importance of fire intensity as a key element of the fire regime, it is seldom measured or included in fire records.

2. We estimated fire intensity in the Kruger National Park, South Africa, by documenting fuel loads, fuel moisture contents, rates of fire spread and the heat yields of fuel in 956 experimental plot burns over 21 years.

3. Individual fires were conducted in five different months (February, April, August, October and December) and at five different return intervals (1, 2, 3, 4 and 6 years). Estimated fire intensities ranged from 28 to 17 905 kW m^{-1}. Fire season had a significant effect on fire intensity. Mean fire intensities were lowest in summer fires (1225 kW m^{-1}), increased in autumn fires (1724 kW m^{-1}) and highest in winter fires (2314 kW m^{-1}); they were associated with a threefold difference between the mean moisture content of grass fuels in winter (28%) and summer (88%).

4. Mean fuel loads increased with post-fire age, from 2964 kg ha^{-1} on annually burnt plots to 3972 kg ha^{-1} on biennial, triennial and quadrennial burnt plots (which did not differ significantly), but decreased to 2881 kg ha^{-1} on sexennial burnt plots. Fuel loads also increased with increasing rainfall over the previous 2 years.

5. Mean fire intensities showed no significant differences between annual burns and burns in the biennial, triennial and quadrennial categories, despite lower fuel loads in annual burns, suggesting that seasonal fuel moisture effects overrode those of fuel load. Mean fire intensity in sexennial burns was less than half that of other burns (638 vs. 1969 kW m^{-1}).

6. We used relationships between season of fire, fuel loads and fire intensity in conjunction with the park's fire records to reconstruct broad fire intensity regimes. Changes in management from regular prescribed burning to 'natural' fires over the past four decades have resulted in a decrease in moderate-intensity fires and an increase in high-intensity fires.

7. The highest fire intensities measured in our study (11 000 – > 17 500 kW m^{-1}) were significantly higher than those previously reported for African savannas, but were similar to those in South American cerrado vegetation. The mean fire intensity for late dry season (winter) fires in our study was less than half that reported for late dry season fires in savannas in northern Australia.

8. *Synthesis and applications*. Fire intensity has important effects on savanna vegetation, especially on the dynamics of the tree layer. Fire intensity varies with season (because of differences in fuel moisture) as well as with fuel load. Managers of African savannas can manipulate fire intensity by choosing the season of fire, and further by burning in years with higher or lower fuel loads. The basic relationships described here can also be used

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to enhance fire records, with a view to building a long-term data set for the ongoing assessment of the effectiveness of fire management.

Key-words: fire management, fuel loads, Kruger National Park, long-term ecological experiment

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Introduction

African savannas are fire-prone, and fire is important in determining the composition and structure of these ecosystems (Bond & Van Wilgen 1996; Anderson, Cook & Williams 2003). Without fire, considerable areas of African savannas could potentially develop into closed woodlands under the current climate, and the occurrence of fires over the past *c*. 8 million years has also seen the evolution of a fire-tolerant and fire-dependent flora (Bond, Woodward & Midgley 2005). The appropriate use of fire in savannas is therefore an important consideration for managing these ecosystems. Tree mortality in savannas, and the recruitment of trees into larger size classes, is strongly affected by fire intensity. An understanding of the relationship between fire intensity and tree mortality has been used for some time by managers of African savannas to decrease tree dominance and encroachment by selecting conditions that lead to more intense fires (Trollope 1974).

The ability of trees and grasses to coexist is central to the understanding of savanna ecology. This coexistence is traditionally explained by either equilibrium or disequilibrium models (Scholes & Archer 1997). Equilibrium models propose that grass–tree coexistence is possible, for example because of separation of the rooting niche, with trees having sole access to water in deeper soil horizons and grasses having preferential access to, and being superior competitors for, water in the surface soil horizons (Walter 1971). In this equilibrium model, climatic variability precludes dominance by either life form, and coexistence is possible in a variety of states (Walker & Noy-Meir 1982). Disequilibrium models, on the other hand, propose that there is no stable equilibrium and that frequent disturbances prevent the extinction through competition of either grasses or trees by periodically biasing conditions in favour of alternative competitors. Higgins, Bond & Trollope (2000) have proposed a disequilibrium model in which interactions between life-history characteristics of trees (sprouting ability, fire survival at different life stages and mortality) and the occurrence of fires (which prevent recruitment of trees into adult life classes) could explain coexistence. This model identified the critical need for variability in fire intensity as a prerequisite for grass–tree coexistence and suggested that the imposition of fire regimes of homogeneous intensity (such as those associated with regular prescribed burning) could lead to dominance by grasses.

The fire regimes that characterize fire-prone ecosystems are normally described in terms of their frequency, season, intensity and type of fire (Gill 1975). While season and frequency are relatively easily measured features of a fire regime, the accurate determination of the range of intensities of fire that occur is more problematic. There are several broad measures of fire intensity: heat per unit area, reaction intensity and fire-line intensity (Biswell 1989). Heat per unit area measures the total energy released by a fire per unit area, while reaction intensity measures the rate of release per unit area. Byram's (1959) fire-line intensity measures the rate of energy released along the fire front, and is strongly correlated with the above-ground impacts of fire. Fire-line intensity is not correlated with soil temperatures experienced during fires (and thus is not related to, for example, variations in seed germination patterns; Bradstock & Auld 1995). It is, however, significantly correlated with damage to above-ground plant parts, especially 'topkill' in woody plants (Higgins, Bond & Trollope 2000).

Fire-line intensity is calculated as the product of the heat yield of fuels, the amount of fuel consumed and the rate of spread of the fire. Heat yields are measured in J g^{-1}, fuel loads in g m^{-2} and rates of spread in m s^{-1}, providing units of kW m^{-1}. Of these factors, rate of spread has the greatest range in vegetation fires, varying from 0·1 to 100 m min^{-1}. The value for fuel consumed in savanna fires can vary from about 20 to 100 g m^{-2}. Heat yields vary so little (by about 10%) that they can be considered as almost constant at about 18 000 J g^{-1} (Stocks, Van Wilgen & Trollope 1997). Fire intensity in savannas thus has a potential 100-fold range of < 500 to > 50 000 kW m^{-1}, primarily because of the large variation of possible spread rates (Stocks, Van Wilgen & Trollope 1997). This variation (largely because of variation in the spread rates of fires burning in the grass layers of the vegetation) has significant consequences for the post-fire survival of trees and shrubs in African savannas. The direct measurement of fire intensity is not always possible, and post-fire indicators such as leaf and bark scorch height and percentage topkill of trees are often used as surrogate measures.

The determination of fire regimes is dependent on good fire records. Such records are seldom kept, and the reconstruction of fire histories in savannas is normally dependent on satellite remote-sensing. Where fire records are kept, they normally provide only the date and extent of fires. Both physical records (Van Wilgen *et al*. 2000) and remote sensing (Russell-Smith, Ryan & Durieu 1997) allow for the determination of frequency

and season, and more recently of intensity (Smith *et al*. 2005). A recent analysis of different approaches to fire management in an African savanna (Van Wilgen *et al*. 2004) concluded that management had little real impact on fire return periods (which were dependent on grass biomass, in turn determined by the amount of, and variability in, rainfall). On the other hand, season of fire, and possibly fire intensities, constituted elements of the fire regime that could be influenced by management.

Fire intensities have been recorded for more than two decades on experimental burning plots in the savanna ecosystems of the Kruger National Park, South Africa. These fires included a range of seasonal and post-fire age treatments, and they allowed for the derivation of general principles relating to the factors influencing fire intensity. We used fire intensity measurements from 956 experimental fires between 1982 and 2003 to derive such principles. We then used the principles to examine the probable historic effects of changing management approaches on the fire intensity regimes in the park, using the comprehensive fire records available for the park from 1957 to 2001.

Methods

STUDY AREA

The Kruger National Park is situated in the low-lying savannas of north-eastern South Africa, and covers 1948 528 ha. Altitudes range from 260 to 839 m above sea level. Mean annual rainfall varies from around 750 mm in the south to approximately 350 mm in the north, but variations about the mean can be marked from year to year. Geologically, the park is underlain by granites and their erosion products in the west, while the eastern sector is predominantly underlain by basalt. The vegetation is characterized by a well-wooded savanna, with knobthorn *Acacia nigrescens*, marula *Sclerocarya birrea*, leadwood *Combretum imberbe* and mopane *Colophospermum mopane* as the dominant trees. The flora of the park comprises 1983 species, including more than 400 tree and shrub species and more than 220 grasses. The park's fire history has been described by Van Wilgen *et al*. (2000, 2004). The mean fire return period for the park was around 4·5 years between 1941 and 1996, with intervals between fires from 1 to 34 years; fires were concentrated in the late dry season. Between 1957 and 1980, regular prescribed burning was conducted every 3 years, in spring after the first rains. From 1981 to 1991, regular prescribed burning was replaced by a regime in which intervals between prescribed fires were more flexible and were timed to take fuel loads, post-fire age and mean annual rainfall into account. From 1992 to 2001, a 'natural' fire policy was in place, in which all lightning-ignited fires were allowed to burn freely while at the same time attempts were made to prevent, suppress or contain all other fires (despite this policy, 76% of the area burnt in this period came from unplanned fires started by people). These changes in management did not fundamentally change the total area burnt (area burnt was dependant on rainfall) or the fire return periods, but did result in a change in the proportion of the area (from 38% to 72%) that burnt in the dry season (Van Wilgen *et al*. 2004).

EXPERIMENTAL BURNING PLOTS

A series of experimental burning plots was established in savanna vegetation in the Kruger National Park in 1954, and we used these to estimate fire intensities during experimental burns. In these plots, grasses provided most (74%) of the fuel (Shea *et al*. 1996). The experiment covered fires at annual (in August only), biennial and triennial intervals (in February, April, August, October and December). After 1974, quadrennial and sexennial burns in October were added to the experiment. Treatments were replicated four times in each of four major landscapes in the park (Table 1). Each plot covered approximately 7 ha. The experiment is described in detail by Biggs *et al*. (2003).

DETERMINATION OF FIRE INTENSITIES

We determined Byram's (1959) fire-line intensity at 956 experimental fires conducted between 1982 and 2003 as:

$$I = Hwr$$

where I is fire intensity (kW m^{-1}), H is heat yield (kJ g^{-1}), w the mass of fuel combusted (g m^{-2}) and r the rate of spread of the head fire front (m s^{-1}).

Table 1. Salient features of the experimental burn plots in four landscapes in the Kruger National Park, South Africa. The number of fires at which fire intensity measurements were made in each landscape is shown

Landscape	Dominant vegetation type	Underlying geology	Mean annual rainfall (mm)	Altitude range (m)	Number of fires
Mooiplaas	Savanna dominated by dense low (1–2 m) mopane (*Colophospermum mopane*) trees	Basalt	496	300–340	160
Satara	Savanna dominated by scattered tall (10–15 m) Marula (*Sclerocarya birrea*) and knobthorn (*Acacia nigrescens*) trees	Basalt	544	240–320	249
Skukuza	Savanna dominated by dense *Combretum collinum*/*Combretum zeyheri* trees	Granite	550	400–480	248
Pretoriuskop	Savanna dominated by dense tall (10–15 m) *Terminalia sericea* trees	Granite	737	560–640	299

We determined the heat of combustion of five savanna grass species using a Gallenckamp automatic adiabatic bomb calorimeter (SANYO Galleukamp, Lowghborough, Leicestershire, UK). Heat of combustion values are normally corrected to heat yields to allow for incomplete combustion in vegetation fires (Byram 1959). We collected ash remaining after head fires in winter, and determined the heat of combustion of the ash as above. Heat yields were then estimated by subtracting the heat of combustion values for ash from those for unburnt samples. It should be noted that this correction factor would not account for grass fuels that were not consumed in the fires; we assumed that all grass fuels were consumed in each fire (for example it was found that mean grass consumption in experimental fires in the Kruger National Park exceeded 95%; Shea *et al*. 1996).

We used a disc pasture meter (Bransby & Tainton 1977), calibrated for use in the Kruger National Park (Trollope & Potgieter 1986), to estimate pre-fire grass fuel loads. Before each fire, we sampled 100 points on the plot to obtain an estimate of the mean fuel load.

The rate of spread for the head fires was estimated as:

$$r = A/(L \times T)$$

where r is the rate of spread (m s^{-1}), A the area burnt as a head fire (m^2), L the mean length of fire front (m) and T the duration of the burn (s). This method provided a single estimate of the rate of spread on each of the plots. The plots were burnt as head fires, except for a small strip on the downwind side and along the edges of the plot. Calculations were based on the time taken to burn the head fire proportion and the area of the plot that burnt in the head fire.

Where it was impractical to use this procedure, the rate of spread was estimated by recording the time taken by the head fire to progress between two observable points in the plot. This procedure was less preferred, as poor visibility made such estimates less reliable, and was applied to 21% of the estimates reported here.

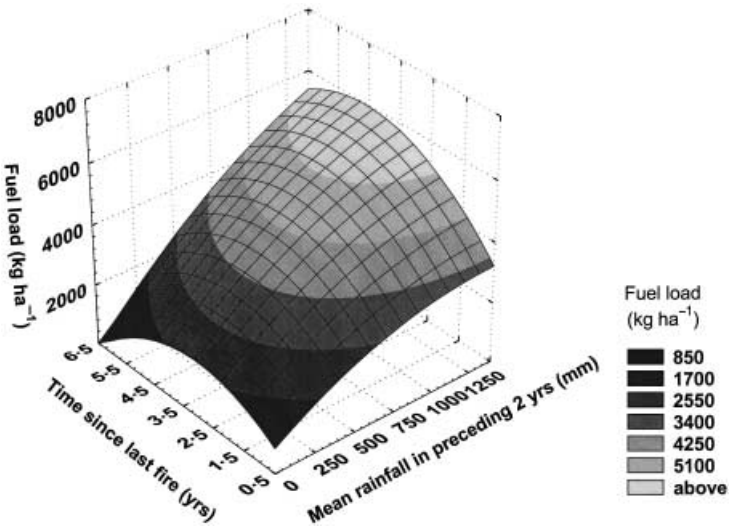

Fig. 1. The effect of time since the last fire, and mean annual rainfall in the preceding 2 years, on grass fuel loads, based on data from experimental burning plots in the Kruger National Park. The relationship is $z = 382{\cdot}9 + 3{\cdot}3x + 979{\cdot}4y - 0{\cdot}001x^2 + 0{\cdot}37xy - 161{\cdot}8y^2$ ($r^2 = 0{\cdot}169$) where z = fuel load (kg ha^{-1}), y = time since the last fire (year) and x = mean rainfall over the previous 2 years (mm).

FACTORS INFLUENCING FUEL LOADS

Fuel loads are an important variable contributing to fire intensity. We examined the influence of post-fire age and mean annual rainfall over the previous 2 years on grass fuel loads, which were measured prior to each experimental burn. Mean rainfall over the previous 2 years was used instead of a single year's total as the effect of rainfall on perennial grasses persists for more than 1 year (Van Wilgen *et al*. 2004). Rainfall data were obtained from four rainfall monitoring gauges situated in each of the four landscapes in which the experimental burning experiment was located (Table 1).

EFFECTS OF SEASON OF BURN ON INTENSITY

The moisture content of the grass sward varies considerably as a result of seasonal curing, and this in turn has a significant effect on fire intensity. Prior to each fire, the moisture content of the grass sward was estimated. Four samples (about 100 g each) of the sward were placed in air-tight bottles, weighed and dried at 65 °C for 4 days. Moisture content was expressed as a percentage of dry mass as follows:

$$M = [(W - D)/D] \times 100$$

where M id the fuel moisture content (percentage dry mass), W the wet mass of the grass sward sample and D the dry mass of the sample.

EFFECTS OF CHANGING MANAGEMENT ON FIRE INTENSITY REGIMES

In order to estimate the historic fire intensity regime of the entire Kruger National Park, we defined broad classes of fire intensity based on the mean fire intensities estimated on the experimental burning plots. For each of four seasons (summer, autumn, winter and spring), we calculated the mean fire intensity for subsets of plots supporting different categories of fuel loads at the time of the fire (the categories were < 1000, 1000–2000, 2000–4000, 4000–6000 and > 6000 kg ha^{-1}). For each combination of season and fuel load, we assigned a class of mean fire intensity as follows: very low (< 500 kW m^{-1}), low (500–1000 kW m^{-1}), moderate (1000–2000 kW m^{-1}), high (2000–4000 kW m^{-1}) and very high (> 4000 kW m^{-1}).

We used a spatial database of all fires occurring in the Kruger National Park (Van Wilgen *et al*. 2000, 2004) to estimate the fire intensity of each fire on the database according to the above broad classes of fire intensity. The database of fires was divided into the three periods (1957–80, 1981–91, and 1992–2001) that were subjected to different management approaches (see above). The season of fire was obtained from the date of the fire, and fuel load was estimated from the relationship between

post-fire age and rainfall (Fig. 1). As rainfall varies considerably across the park, we selected the closest of four rainfall recording stations to each fire on the database. The four rainfall stations represented a gradient of rainfall conditions across the park, and were between 4 and 21 km from the respective experimental burn sites.

Results

FIRE INTENSITIES

The mean heat of combustion value for the five grass species sampled was 18 024 J g^{-1}. After subtraction of the mean heat of combustion for ash samples, this was reduced to a heat yield (H) of 16 890 J g^{-1}. This value was assumed to be constant in all fires, as is standard practice in many other similar studies (Catchpole 2002; Williams, Gill & Moore 2003). Fire intensities were determined for 162 annual fires in August, 776 fires following fire-free intervals of between 2 and 4 years in all seasons, and 18 fires after fire-free intervals of 6 years in October (Table 2). Data for annual burns were only available for winter (August) fires, while those for sexennial burns were only available for fires in spring (October). Mean fire intensities ranged from 638 kW m^{-1} for sexennial spring burns to 2664 kW m^{-1} for winter fires at 2–4 years intervals, while the intensities at individual fires spanned three orders of magnitude, from 28 to 17 905 kW m^{-1}.

Mean fire intensities also increased with mean (calculated over the 21 years in which intensities were estimated) annual rainfall in the four landscapes in which the experimental burning plots were located (Fig. 2). The relationship suggests that an increase of 50% in mean annual rainfall (from 500 to 750 mm) corresponded to an increase in the mean fire intensity of 67%, from 1500 to 2500 kW m^{-1}, reflecting the effect of increased rainfall on grass fuel production.

FACTORS AFFECTING FUEL LOADS

Fuel loads were affected by post-fire age. Significant differences in fuel loads were found between annually burnt plots, those burnt biennially, triennially and quadrennially, and those burnt sexennially ($P < 0{\cdot}05$). Mean fuel loads in the annually burnt plots were 2964 kg ha^{-1} while those in the biennial, triennial and quadrennial burnt plots were 34% higher, at 3972 kg ha^{-1}. Fuel loads in the sexennial burnt plots were lower, at around 2881 kg ha^{-1} (Fig. 3a). Fuel loads also increased with rainfall over the previous 2 years (Fig. 1).

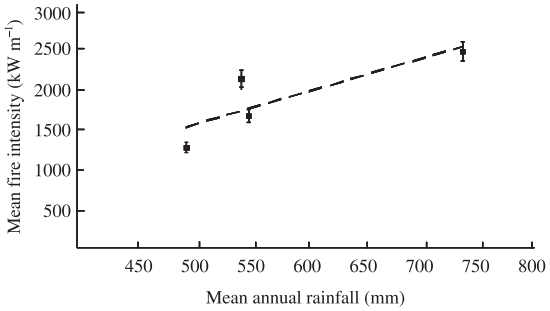

Fig. 2. The effect of mean annual rainfall on mean fire intensities estimated on experimental burning plots in four landscapes in the Kruger National Park. The relationship is: $y = 4{\cdot}13x - 558{\cdot}22$ ($r^2 = 0{\cdot}71$), where y = the mean fire intensity (kW m^{-1}) and x = mean annual rainfall (mm). Bars are the 95% confidence intervals of the mean.

FIRE SEASON AND FUEL MOISTURE CONTENT

Significant differences in fuel moisture content were found between summer, autumn and spring, and winter ($P < 0{\cdot}05$). The mean fuel moisture content of the grass sward was lowest (28%) in winter (Fig. 4a). This increased in spring and autumn (53%) and was highest (88%) in summer. Thus season had a more than threefold effect on fuel moisture content, which is, in turn, an important factor influencing fire intensity. Higher fuel moisture contents result in lower rates of combustion and consumption, and lower rates of fire spread (Pyne, Andrews & Laven 1996). In our data, 80% of fires had rates of spread between 0·01 and 0·45 m s^{-1}. Fuel moisture content had a significant effect on rate of fire spread, with rates of spread above 0·5 m s^{-1} being restricted to fuel moistures below 150% and those above 1 m s^{-1} to fuel moistures below 80% (Fig. 5). The effect of fuel load on rate of fire spread was also apparent in the data. Under conditions of low fuel moisture content (< 80%), the mean fuel load in the 142 fires that had spread rates > 0·5 m s^{-1} was more than three times that of the corresponding fuel load for the 607 fires that had spread rates < 0·5 m s^{-1} (4848 vs. 1506 kg ha^{-1}).

Fire season affected fire intensity, mainly as a result

Table 2. Mean fire intensities (kW m^{-1}) at different fire return intervals and seasons recorded in experimental plot fires in the Kruger National Park between 1982 and 2003

	Fire return period (years)		
Season of burn	1	2–4	6
Summer (fires in February and December)	No data	1279 (range 41–9621, n = 227)	No data
Autumn (fires in April)	No data	1724 (range 88–8592, n = 182)	No data
Winter (fires in August)	1891 (range 28–8365, n = 162)	2664 (range 42–9207, n = 196)	No data
Spring (fires in October)	No data	2419 (range = 136–17 905, n = 171)	638 (range 47–1752, n = 18)

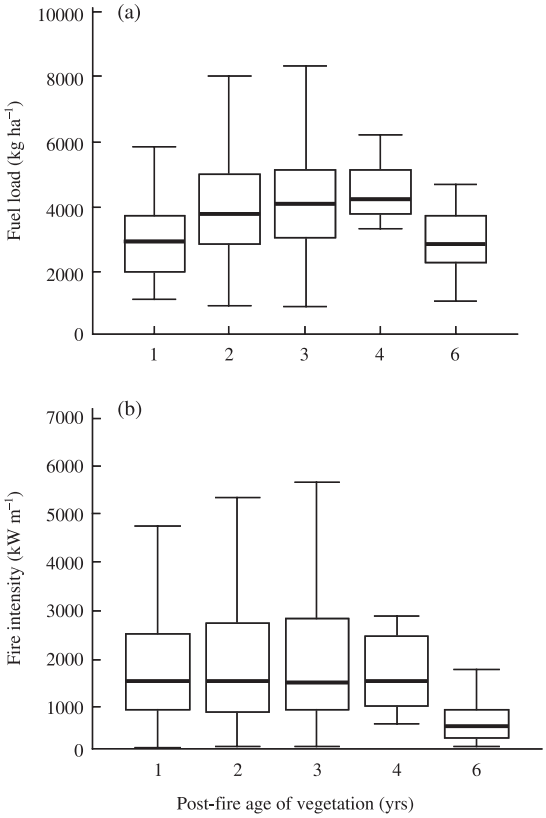

Fig. 3. (a) Grass fuel loads estimated prior to experimental fires at different post-fire ages on experimental burning plots in the Kruger National Park. Box and whisker plots show the median, quartiles and a range excluding outliers (outliers, defined as values > 1·5 × interquartile range from the upper and lower box edges, not shown). (b) Fire intensity in vegetation of different post-fire age on experimental burning plots in the Kruger National Park. Box and whisker plots show the median, quartiles and a range excluding outliers (outliers, defined as values > 1·5 × interquartile range from the upper and lower box edges, not shown).

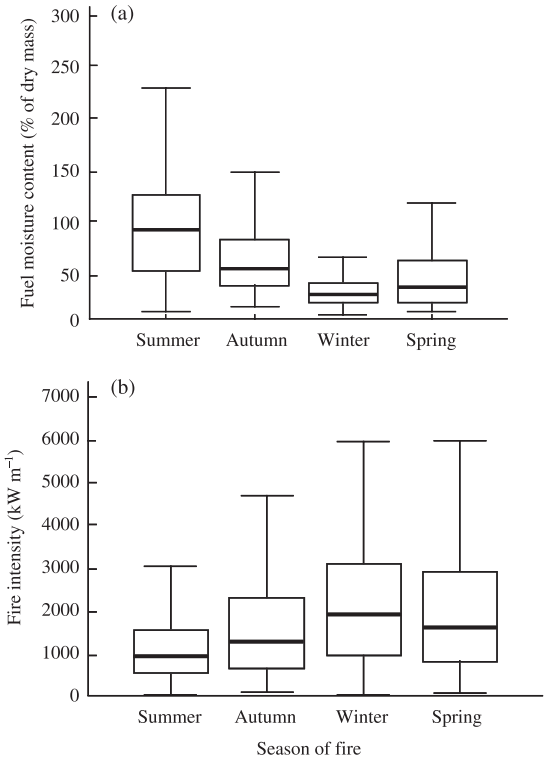

Fig. 4. (a) Grass sward moisture content estimated prior to experimental fires in different seasons on burning plots in the Kruger National Park. Box and whisker plots show the median, quartiles and a range excluding outliers (outliers, defined as values > 1·5 × interquartile range from the upper and lower box edges, not shown). (b) Fire intensities in different seasons on experimental burning plots in the Kruger National Park. Box and whisker plots show the median, quartiles and a range excluding outliers (outliers, defined as values > 1·5 × interquartile range from the upper and lower box edges, not shown).

of changes in fuel moisture. Significant differences in fire intensity were found between summer, autumn and winter fires ($P < 0{\cdot}05$). Intensities in spring fires were more variable, and were significantly different only from summer fires. Mean fire intensities were lowest (1225 kW m^{-1}) in summer fires (February and December), increased (1724 kW m^{-1}) in autumn fires (April) and highest (2314 kW m^{-1}) in winter fires (August) (Fig. 4b). However, the four most intense individual fires (> 10 000 kW m^{-1}) in our data set all occurred in spring (October) fires. Such fires are possible in October when conditions are unusually dry and fuel loads are high (fuel loads were between 5000 and 7000 kg ha^{-1} and fuel moisture contents were around 12% during these four fires), resulting in intensities from 11 210 to 17 905 kW m^{-1}. A range of fire intensities occurred in all seasons, but low-intensity fires (< 1000 kW m^{-1}) predominated in summer and autumn, with very few fires > 4000 kW m^{-1} (Fig. 6). The highest proportion of fires in winter was in the moderate category (2000–4000 kW m^{-1}). High to very high fire intensities (> 4000 kW m^{-1}) were most prevalent in winter and spring burns (Fig. 6).

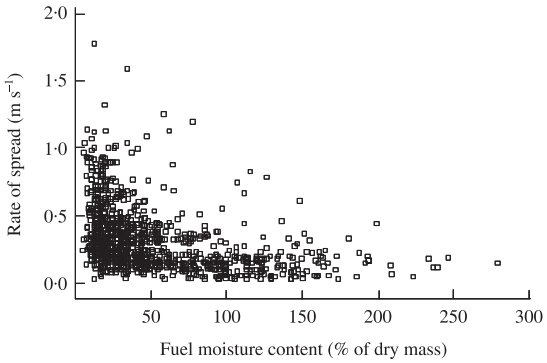

Fig. 5. The relationship between fuel moisture content and the rate of spread from 956 fires on experimental burning plots in the Kruger National Park.

Post-fire age had less of an effect on intensity than fire season, with no significant differences between annual burns and burns in the biennial, triennial and quadrennial categories (Fig. 3b), despite lower fuel loads in annual burns (Fig. 3a). This suggested that the effects of season of fire, which were manifested in low fuel moistures in winter (and sometimes spring), overrode those of

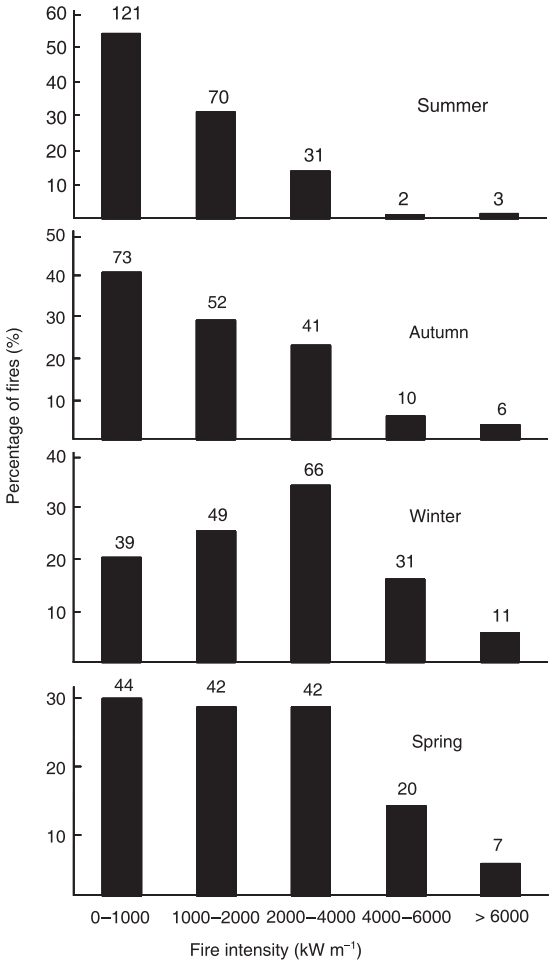

Fig. 6. Seasonal distribution of fires within five classes of fire intensity (see Table 3) from 956 fires on experimental burning plots in the Kruger National Park. The number of fires in each class is shown above the bars.

fuel load. However, fire intensities on the sexennial burnt plots were significantly lower (683 vs. 1969 kW m^{-1}; Fig. 3b) than on other plots ($P < 0{\cdot}05$), probably as a result of lower fuel loads following long interfire periods.

EFFECTS OF CHANGING MANAGEMENT ON FIRE INTENSITY REGIMES

The mean fire intensities recorded in 20 classes separating season and fuel load ranged from 194 to 5253 kW m^{-1} (Table 3). Winter fires reached the high intensity class when fuel loads exceeded 2000 kg ha^{-1}, while autumn and spring fires only reached high intensities when fuel loads exceeded 4000 kg ha^{-1}. Very high intensities were possible in both spring and winter fires when fuel loads exceeded 6000 kg ha^{-1}. Experimental burns in summer were only classed as moderate intensity at the higher levels of fuel load.

During the period between 1957 and 1980, when regular prescribed burning was practised, a total of 7·8 million ha was burnt. During this period, fires were predominantly in the moderate intensity class (59% of the area) (Fig. 7). Between 1981 and 1991, when 1·7 million ha were burnt, a more flexible approach to prescribed burning was followed. This period was dominated by fires in the moderate intensity class (71·1% of the area burnt). This decade had below-average rainfall, which resulted in a much smaller proportion of the park being burnt than in the other periods; the lower fuel loads that accompanied this period would have precluded many high-intensity fires. Between 1992 and 2001, when the 'natural' fire policy was in place, 2·8 million ha were burnt. The majority of fires in this period (50·1% of the area) were in the high intensity class. It was also only in this period that a small number fires (0·2% of the area) was classified as very high intensity, having occurred in winter or spring when fuel loads exceeded 6000 kg ha^{-1}.

Discussion

FUEL BUILD-UP AND FIRE INTENSITY

Our results suggest that fuel load (grass biomass) accumulates in proportion to rainfall for the first 4–5 years after fire. Thereafter it declines as a combined result

Table 3. Classes of fire intensity associated with different fuel loads and seasons of burn in the Kruger National Park. The mean fire intensities, and the number of fires on which these means are based, is shown

		Fuel loads (kg ha^{-1})				
Season of burn	Descriptor	< 1000	1000–2000	2000–4000	4000–6000	> 6000
Summer	Fire intensity class	Very low	Low	Moderate	Moderate	Moderate
(1 December–31 March)	Mean fire intensity (kW m^{-1})	287	578	1031	1432	1650
	Number of fires	1	17	95	83	31
Autumn	Fire intensity class	Very low	Low	Moderate	High	High
(1 April–30 May)	Mean fire intensity (kW m^{-1})	No data	732	1455	2106	1900
	Number of fires	0	19	62	79	23
Winter	Fire intensity class	Very low	Low	High	High	Very high
(1 June–31 August)	Mean fire intensity (kW m^{-1})	194	835	2082	3625	4385
	Number of fires	4	65	187	83	19
Spring	Fire intensity class	Very low	Low	Moderate	High	Very high
(1 September–30 November)	Mean fire intensity (kW m^{-1})	No data	712	1570	3066	5253
	Number of fires	0	15	103	55	16

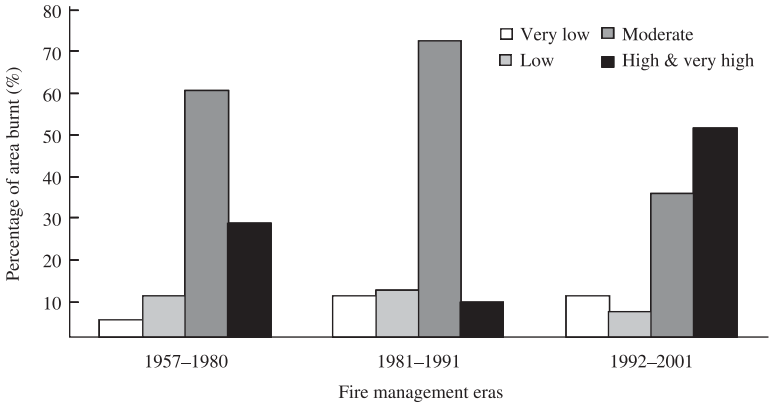

Fig. 7. The proportion of area burnt in classes of fire intensity (see Table 3) during three distinct eras of fire management. The eras are (i) 1957–80, regular prescribed burning; (ii) 1981–91, flexible prescribed burning; and (ii) 1992–2001, 'natural' fires; the areas burnt in each era were 7·8, 1·7 and 2·8 million ha, respectively.

of grazing, decomposition and the loss of grass vigour, possibly leading to equilibrium fuel loads (when decomposition balances accumulation) after > 6 years (Kessel, Good & Potter 1982; Bond & Van Wilgen 1996). Given that fire return periods are between 3 and 6 years in the Kruger National Park (Van Wilgen *et al*. 2000), fires will normally remove accumulated biomass before equilibrium is reached. Fuel build-up is also more rapid in nutrient-poor areas (on granites in the Kruger National Park). This is especially so in areas of higher rainfall (for example in the Pretoriuskop landscape; Table 1), where less palatable grasses dominate the sward, resulting in lower grazing pressure. In areas of lower rainfall, and higher soil fertility (on basalts in the Kruger National Park), more palatable grasses dominate the sward. Fuel build-up is retarded by higher grazing pressure, and fire intensities tend to be lower.

VARIATION IN FIRE INTENSITY

The classification of fire regimes for an area as large as the Kruger National Park will of necessity involve a degree of generalization, as we have done in allocating fire intensity classes to individual fires on the basis of predicted fuel loads and season of fire alone. A great deal of variation in fire intensity is possible in the same fire, especially if a large area is covered (for example, since 1957 most of the area in the Kruger National Park burnt in fires of greater than 5000 ha each; Van Wilgen *et al*. 2000). A single fire of this size will burn under a variety of changing conditions, during the day or at night, upslope or downslope, and with or against the wind, before it is extinguished, resulting in a fine-scale mosaic of varying fire intensities. Two fires in the same season may also burn under significantly different weather conditions, or in landscapes dominated by either grasses or trees. The intensities measured on experimental plots cannot encompass this variation. They also tend to exclude conditions that would lead to very high-intensity fires (for example experimental or prescribed burns are seldom conducted under very hot, dry or windy conditions for reasons of safety) and therefore do not provide a sample of the full spectrum of possible fire intensities. None the less, the principles outlined here will provide a more rigorous basis for the refinement of models that include fire intensity as an important input variable (Higgins, Bond & Trollope 2000; Van Langevelde *et al*. 2003).

COMPARISONS WITH OTHER STUDIES

The fuel loads and fire intensities reported here are generally in line with those reported from other savanna regions in Africa (Table 4). Our data, however, are based on hundreds rather than tens of observations, and some of the fires reported here had intensities more than double those reported previously. Fire intensities reported for early dry season fires in Australian savannas are similar to those reported for African savannas, while fires in the late dry season are much higher in Australian savannas (Table 4). At the Kapalga experiment in northern Australia, the mean intensity of early dry season fires was 2100 kW m^{-1}, compared with a mean of 7700 kW m^{-1} for late dry season fires (more than double the mean for winter dry season fires in our study). The difference

Table 4. Comparative fuel load and fire intensity data from savanna vegetation in Africa, Australia and South America

Place	Number of fires	Fuel loads (kg ha^{-1})	Fire-line intensity (kW m^{-1})	Source
Kruger National Park, South Africa	956	830–9214	28–17 905	This study
	10	2218–5492	480–6130	Shea *et al*. (1996)
	2	1281–5964	4048–10 906	Stocks *et al*. (1996)
Hluhluwe/Umfolozi Park, South Africa	10	1600–14 200	194–5993	Van Wilgen & Wills (1988)
Chobe National Park, Botswana	9	2000–9930	130–9474	Smith *et al*. (2005)
Western Province, Zambia (dambo vegetation)	7	1884–3314	288–5271	Hoffa *et al*. (1999)
Western Province, Zambia (miombo vegetation)	6	8953–13 233	25–5274	Hoffa *et al*. (1999)
Kasanga National Park, Zambia	4	3164–7343	1734–4061	Shea *et al*. (1996)
Early season fires, Northern Territory, Australia	15	2100–6000	500–3100	Williams, Gill & Moore (2003)
Late season fires, Northern Territory, Australia	10	3000–9800	3700–18 000	Williams, Gill & Moore (2003)
Cerrado, Brazil, South America	2	7128–10 031	2842–16 394	Kauffman, Cummings & Ward (1994)

between early and late dry season fires in Australia was mainly because of more extreme fire weather conditions in the late dry season (Williams, Gill & Moore 2003). Fuel composition also differed between the early and late dry season in Australian savannas. For example, grass made up 71·5% of the fuel load in the early dry season at Kapalga, and this dropped to 41·1% in the late dry season; the balance was made up by leaf (41·1%) and twig (12·2%) litter (Williams, Gill & Moore 2003). In African savannas, grass fuel loads are dominant, and contribute 70–98% of the total fuel (Shea *et al*. 1996). In South American cerrado vegetation, both fuel loads and fire intensities were comparable to the upper ranges found in African savannas (Table 4). Fires in these ecosystems consumed > 97% of biomass in open grasslands, compared with 72–84% in more wooded communities (Kauffman, Cummings & Ward 1994). Fire-line intensities were greatest in grasslands, which reinforced grass dominance through damage to trees.

ECOLOGICAL CONSEQUENCES OF CHANGES IN FIRE INTENSITY

Fire intensity has an appreciable effect on African savanna trees. Relatively high fire intensities will kill the aerial parts of medium-height woody plants, forcing them to resprout from the base after fire. It is only when these saplings grow tall enough to escape the flame zone (which increases with increasing intensity), and are able to continue growing from aerial parts, that they are recruited into larger size classes of trees. For example, fires of 3000 kW m^{-1} will topkill 90% of tree saplings 1 m tall but only 40% of those that are 2 m tall (Van Wilgen, Everson & Trollope 1990). The number of times that these critical intensities are attained is a key determinant of recruitment by trees into larger size classes; trees that are unable to grow to heights sufficient to escape fires of higher intensity are considered to be in a 'fire trap' (Higgins, Bond & Trollope 2000). In our study, intensities above 3000 kW m^{-1} were more often reached in winter and spring fires (26% and 24% of fires, respectively), than in summer and autumn fires (6% and 14% of fires, respectively). Fire managers who wish to reduce the recruitment of trees to larger size classes could therefore plan to set fires mainly in winter and spring, in years when grass fuel loads are high, in order to increase fire intensity.

In his definition of fire regimes, Gill (1975) used the term fire type to distinguish between fires that burn in organic layers of the soil (ground fires), those burning in fuels contiguous with the ground (surface fires) and those burning in the canopies of trees (crown fires). Fires in African savannas are surface fires, burning in grass layers below the tree canopies. The term fire type has sometimes been used in the literature on African savanna fires to distinguish between head fires (those burning with the wind or upslope) and back fires (those burning against the wind or downslope; Trollope 1978; Trollope 1999). Head fires and backfires are not types of fire in terms of Gill's (1975) definition. They differ in their intensity, but the distinction was made in terms of fire type because head fires and back fires of the same intensity have different effects on grasses (Trollope 1978). More recently, the term fire severity (Waldrop & Brose 1999; Diaz-Delgado, Lloret & Pons 2003) has been used to distinguish between measures of fire behaviour (intensity) and effects as a result of the alteration of soil properties and below-ground processes (severity). Fire severity is also used to describe fire duration (Jacoby, Ansley & Trevion 1992; Perez & Moreno 1998). African savanna back fires have longer residence times than head fires (Trollope 1978), and thus differ in their severity in this respect. The use of back fires to decrease fire intensity (for example to reduce the topkill of trees) will increase fire severity, and managers of African savannas need to understand these differences and their effects.

OPTIONS FOR MANAGERS

Managers of conservation areas in South Africa are currently reviewing their fire management policies, in response to changes in ecological concepts and a new focus on biodiversity conservation (Bond & Archibald 2003). Despite recent advances in understanding, the ecological impacts of fires on all elements of the biota are not known in sufficient detail to be able to prescribe appropriate fire regimes with confidence. The managers of some areas (including the Kruger National Park) have, in response, opted to use fire patterns as surrogate measures of achieving diversity goals, on the assumption that a diversity of fire patterns (including a diversity of fire intensities) will promote the conservation of biological diversity (Van Wilgen, Richardson & Seydack 1994; Van Wilgen, Biggs & Potgieter 1998). The need for variation in fire intensity was also supported by a simulation model that suggested that such variation is essential for tree–grass coexistence in savannas (Higgins, Bond & Trollope 2000). Managers in the Kruger National Park have therefore recently adopted an approach that will seek to diversify the range of fire intensities achieved in the application of management fires.

When the targets for achieving a range of fire intensities were originally conceived, the concern was that management fires over the past four decades were too frequent, and too intense, and that this, in turn, retarded the recruitment of savanna trees into larger size classes (Van Wilgen *et al*. 2003). In addition, fire and browsing by elephants interacted to increase the mortality of large trees (Eckhardt, Van Wilgen & Biggs 2000; Van Wilgen *et al*. 2003). As a result, vegetation structure was being homogenized into a landscape dominated by short (< 2–3 m) trees (Van Wilgen *et al*. 2003). Our analysis suggests that there has been a recent shift towards higher intensity fires (Fig. 7) in the last decade, despite the desire to move away from a dominance of high-intensity fires.

The response of the vegetation to fire treatments on the experimental burning plots has not yet been

comprehensively analysed. However, a concerted effort is now being made to redress this situation (Freitag 1998). Preliminary results suggest that fire has a marked effect on the structure, but not the composition, of woody vegetation. Trees and shrubs survive and mature in non-fire treatments but are significantly smaller where regularly burnt (Van Wilgen *et al*. 2003). It was also suggested, for one set of experimental plots at least, that seasonal effects of fire manifest themselves through fire behaviour (Enslin *et al*. 2000). On these plots, there was a continuum from the least multistemmed coppicing (in low-intensity summer fires) to the most (in high-intensity winter fires). High-intensity fires could therefore be contributing to the widespread phenomenon of dominance by trees of low, multistemmed stature.

Given the absence of a thorough understanding of fire effects, managers need to adjust their approaches to fire management using the best available information (adaptive management; Rogers 2003). The regime between 1992 and 2001 was dominated by intense, late dry-season burns, and this may have exacerbated the problem of dominance by trees of low, multistemmed stature. By shifting a larger proportion of fires to summer and autumn (the growing season), managers could hope to reduce the intensity of many fires. Whether this approach will be effective could be assessed by including estimates of fire intensities into fire records. The relationships between season of fire, fuel loads and fire intensity described here could be used for that purpose.

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