INTRODUCTION

Project motivation

Elephant-induced damage to habitats seems to increase under conditions where elephant movements are restricted by neighbouring human settlements (Ben-Shahar 1993). Human modifications of the environment can intensify use of particular areas by disrupting traditional elephant pathways and prevent elephants from making large-scale movements in response to resource supply (Gadd 1997). In some conservation areas, elephant numbers increase, woody vegetation declines, and pressure mounts for controlling elephant numbers. Such a phenomenon has been termed the 'elephant problem' (Barnes 1983). Increasing densities of elephants are often blamed for decreasing woodlands but the exact dynamics of the relationship have evaded ecologists and managers for many years. A proper management policy regarding the control of elephant numbers requires information on the trends of the elephant population and, especially, their patterns of resource utilisation (Ben-Shahar 1993). Elephant control to promote the recovery of mature canopy woodland can be successful only if accompanied by effective fire protection (Pellew 1983).

That there has been a significant decline in the density of large trees in the Kruger National Park is confirmed by the visual appearance of the woody vegetation over much of the Park (Viljoen 1988). Viljoen (1988) conducted a preliminary survey on changes in the density of large trees in the *Sclerocarya birrea/Acacia nigrescens* savanna landscape of the Kruger National Park by using aerial photographs. The results showed that during the period 1944 to 1981 (37 years) the number of large trees decreased by 93.4% in the Satara area. A similar trend, but not as marked decline, was noted in the Lower Sabie area where during the period 1940 to 1977 (37 years) the large trees decreased by 49.6%. In both cases the major decline in the tree density occurred after 1965. Trollope, Trollope, Biggs, Pienaar & Potgieter (1998) found no significant changes in the density of large trees between 1940 vs. 1960 in areas with granitic soils, whereas a moderate decline in the vegetation occurred in the areas with basaltic soils on four of the major vegetation units in the Kruger National Park, i.e.

- **Landscape 5:** Mixed *Combretum/Terminalia sericea* woodland (Gertenbach 1983)
- **Landscape 12:** *Colophospermum mopane/Acacia nigrescens* savanna (Gertenbach 1983)

- **Landscape 17:** *Sclerocarya birrea* subsp. *caffra/Acacia nigrescens* savanna (Gertenbach 1983)
- **Landscape 23:** *Colophospermum mopane* shrubveld (Gertenbach 1983)

Conversely during the period 1960 to 1989 there was a dramatic decline in the density of large trees in all four above mentioned landscapes, particularly in landscape 17: *Sclerocarya birrea/Acacia nigrescens* savanna on basaltic clay soils (Trollope *et al.* 1998). Results of Trollope *et al.* (1998) suggest that the changes in the woody vegetation do not involve a decrease in species diversity but rather a change in the structural diversity where the woody vegetation is being transformed into a short woodland community with a low density of large trees. There are strong indications that the reason for the decline in the density of large trees in the Kruger National Park can be attributed to the interactions of elephants and fire on the woody vegetation. These two factors changed dramatically during the post-1960 period. Firstly, there was a dramatic increase in the elephant population in the Kruger National Park between 1960 and 1970 (Trollope *et al.* 1998) when the elephant population increased from approximately 1100 in 1960 to over 8500 in 1970 (Whyte & Wood 1995). Secondly, a controlled burning program was introduced in the Kruger National Park in 1954 and has been maintained in one form or another until 1995. Research by Trollope, Potgieter & Zimbatis (1995) concluded that the veld was being burnt too frequently in the Park during the past, based on the present condition of the grass sward. This partly forms the basis for the current *Laissez Faire* burning program introduced, in an attempt to reduce the previously high frequency of fire (Trollope *et al.* 1998).

Concern about the potential impact that elephants may have on the *Sclerocarya birrea* (marula) population of the Kruger National Park gave rise to an earlier research project (Coetzee, Engelbrecht, Joubert & Retief 1979). Although results of this research suggested that the impact at that time did not constitute a threat to the marula population, there is now a growing concern among certain research and field staff members of the Kruger National Park that the marula population is deteriorating (Whyte, Pers. comm.)[1]. There is a strong possibility that the problem is complex - elephants are probably killing the large marula trees by debarking them or pushing them over, while fire is preventing young trees from becoming established. Whyte, Biggs,

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Gaylard & Braack (1998) suggested that elephants should not be viewed in isolation, but as one component of a broader, integrated system. They recommended that the elephant impact should be managed in conjunction with other ecosystem processes such as fire, to promote biodiversity in general. Fire and large herbivores are among the principal factors determining the balance between herbaceous and woody plants in savanna ecosystems (Ruess & Halter 1990). In revision of the research objectives for the Kruger National Park, the development of an understanding of different fire regimes, and the effect of natural fire in combination with elephants and other herbivores on the biodiversity at multiple scales has been identified as a priority for nature conservation (Freitag & Biggs 1998).

The Elephant Factor

Management of elephants in the Kruger National Park

As the flagship of South Africa's national parks, the Kruger National Park can be seen as a model for the management of large herbivores. The largest elephant population in South Africa is found within the boundaries of this wildlife sanctuary (Smit 1997). By 1920 the Kruger National Park, which was proclaimed in 1889, was one of the last four remaining areas in South Africa with elephants (Hall-Martin 1992a). Since 1920 elephant numbers have increased in the Kruger National Park to such an extent that 13 000 surplus elephants have been culled since 1976 (Hall-Martin 1992b). The control of elephants by means of culling was implemented in the Kruger National Park to maintain the population at a density considered suitable to the area. The last management policy limited the population to 7500 elephants (Smuts 1975) after the carrying capacity of elephants in the Kruger National Park was estimated between 6000 and 8500 (Joubert 1986). During 1996 a moratorium was placed on the control of elephants by means of culling (Smit 1997), and since then the population has grown to 8896 in 1998; the largest population to be recorded in the Kruger National Park (Whyte 1998).

Reviewing the gross elephant population structure in the Kruger National Park since 1967, Whyte & Wood (1996) found the mean values for the population structure ratio as follows: bulls – 15.78% and cows – 84.22%. Whyte & Wood (1996) also found that the migration of elephant bulls is restricted to the arbitrary regional boundaries and no long-term trends were detected.

Elephant breeding clans monitored by Whyte (1993), did not move out of their home ranges into other landscapes, except under extreme drought conditions. Culling operations, however, did result in sporadic fluctuation of movement to areas outside the regional home ranges occupied by breeding herds (Whyte & Wood 1996).

Whyte *et al.* (1998) proposed a new policy for the management of the elephant population of the Kruger National Park. One of the principles on which the policy is based, is that elephant populations which are confined - but whose growth is not limited through management - are very likely to ultimately increase in number until negative impacts on the system's biodiversity results (Whyte *et al.*1998). They suggested that the Kruger National Park be divided into six zones (Figure 1). These are two botanical reserves, two high elephant impact zones and two low elephant impact zones. The management of these zones will be driven by "Thresholds of Potential Concern" (TPC's). These TPC's are specified limits of ecological change which should not be exceeded (Whyte *et al.*1998). The specified management for each zone will be followed until there are indications that one or more of the TPC's have been reached or exceeded. It is expected that the population of these zones will increase at around 7% per year. In the low elephant impact zones the population will be decreased until one or more of the TPC's have been reached or exceeded. This decrease will be achieved through the reduction of the populations within these zones by 7% per year. In the Botanical Reserves, medium densities will be maintained (Whyte *et al.*1998).

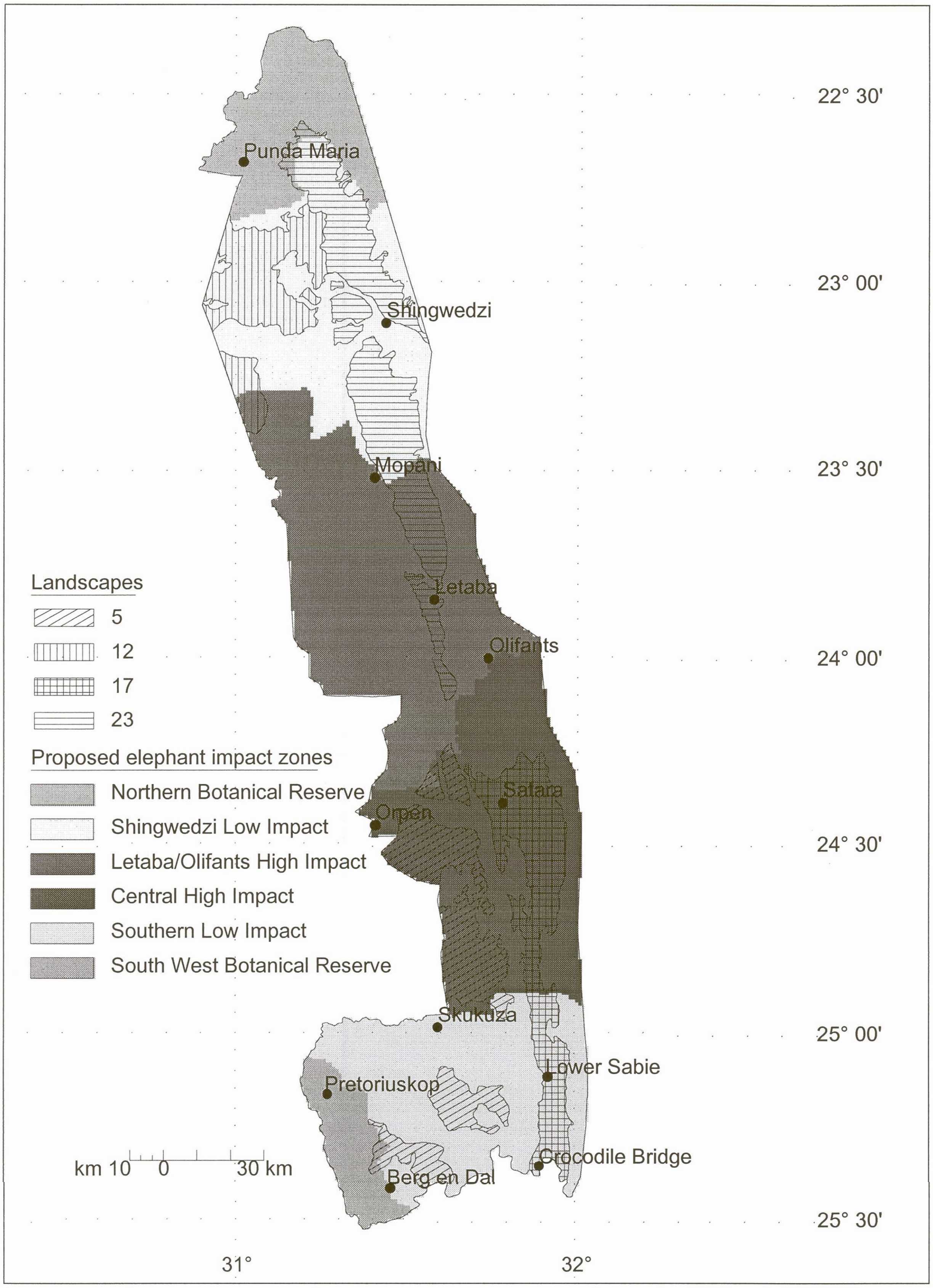
Previous studies of Elephant Impact on Vegetation

Of the megaherbivores, the elephant is singular in its ability to profoundly modify habitats. Where this occurs within protected areas the change may be such as to reduce biodiversity beyond the threshold of recovery in the long term. There consequently appears to be little alternative but to limit elephant numbers to commensurate with specified conservation goals (Taylor & Cumming 1993). A population limit was imposed to ensure the perpetuation of viable populations of all large mammal species in the Kruger National Park and to minimise excessive elephant damage to vegetation (Coetzee *et al.* 1979). Furthermore, vegetation structure favourable to elephant and other animals is maintained by limiting elephant impact on habitat (Coetzee *et al.* 1979). Evidence from the Tsavo National Park, Amboseli National Park and the

Addo Elephant National Park showed that in confined systems, high elephant densities could result in the extirpation of certain plant and animal species (Smit 1997). Megaherbivore populations are slow to respond to environmental changes. Elephants are capable of switching to alternative food items, so tree loss is unlikely to have an immediate effect on elephant numbers (Gadd 1997). However, less competitive animal species are more sensitive to habitat change, and are protected by implementing measures to alleviate excessive elephant utilisation of available food and water (Gadd 1997).

Elephants can subsist in virtually any habitat that provides adequate quantities of food and water but optimum habitat includes both grass and browse as they are mixed feeders (Estes 1991). The diet of elephants consists of grass, forbs, bark, twigs, leaves and fruit. Browsing involves breaking off branches and uprooting shrubs and small trees. Some elephant bulls master the technique in pushing over large mature trees. The elephants' use of browse is related to grass availability and therefore the rate at which trees are depleted is strongly dependent upon any events that alter grass availability (Gadd 1997). It is estimated that an elephant's daily dry matter forage intake is equivalent to 4 to 6% of its body mass. Thus, a mature elephant bull weighing 2800 kg will require 42 kg of forage per day (Trollope *et al.* 1998). The proportion and kind of forage consumed by elephants vary according to the season and availability of food. Elephants tend to show preference for grasses and herbs during the rainy season and for woody plants in the dry season (Trollope *et al.* 1998). According to Buss (1961), grass comprises 88% of the elephant's diet in the dry season in Uganda. In Murchison Falls National Park (Uganda), Field (1971) found that although grass was the main constituent of the diet, the amount of browse eaten ranged from 8 to 45% of the total food intake. In the Tsavo National Park (Kenya), Napier Bax & Sheldrick (1963) found that herbaceous material forms the bulk of the elephant diet, even in normal dry seasons.

In Zimbabwe and South Africa, browse is more important in the dry season than in the wet season, with a rapid increase in the amount of grass eaten soon after the first rains (Anderson & Walker 1974). In a drought, however, damage to woody species reaches a peak as elephant pressure on the woody component increases due to poorer quality of the grass (Gadd 1997).



Landscape 5=Mixed *Combretum/Terminalia sericea* woodland, 12= *Colophospermum mopane/Acacia nigrescens* savanna, 17= *Sclerocarya birrea/Acacia nigrescens* savanna, 23= *Colophospermum mopane* shrubveld

Figure 1. The proposed elephant management zones in the Kruger National Park to be used to control the impact of the elephants on the system's biodiversity (Source: Whyte *et al.* 1998).

Generally the proportion of browse in the elephant diet is much higher than in East Africa, having been estimated between 74 and 86% (dry season figures) from stomach contents of elephants in the Chizarira and Mana Pools Game reserves (Anderson & Walker 1974). Van Wyk & Fairall (1969) estimated the proportion of browse to be 50% for elephants in the Kruger National Park, and stated that woody species were far more important in the diet of elephants living in the Kruger National Park than those living in the more open savanna areas of East Africa. Pienaar, Van Wyk & Fairall (1969) found that elephants in the Kruger National Park utilise herbaceous grasses and forbs mainly during the rainy season. During the dry winter months the elephants congregate along the rivers and permanent watering points where they consume reeds, grasses, forbs and the leaves, bark and twigs of riverine trees and shrubs (Trollope *et al.* 1998). Further away from water the elephants browse the dry leaves of preferred trees and shrubs. In areas where fires have occurred that minimized grazing opportunities, they push over and utilise trees that have escaped the fires and retained their leaves. Tree species that are particularly selected for by elephants are *Adansonia digitata* (baobab), marula, *Acacia nigrescens* (knobthorn), *Combretum imerbe* (leadwood) and *Pterocarpus angolensis* (kiaat) (Trollope *et al.* 1998).

Elephant impact on vegetation is known to be selective, though a wide variety of woody plants may be involved. Pellew (1983) found that elephants largely ignore trees <1 m, while Jachmann & Bell (1985) found that elephants feed mostly between the height of 2–3 m. Although elephants rather forage on smaller than larger stems, trees up to the height of 7 m were found to be highly susceptible to elephant damage (Jachmann & Bell 1985; Gadd 1997). The consequences of elephant impact on any particular species depends on the nature of the scars, the ability of the plant to recover, its demography and role in various plant communities and the interrelationships between the latter and other ecosystem components. Many factors, therefore, contribute in a complex manner to determine the effect of elephant impact on any particular plant species and/or plant community and the management of elephant numbers (Coetzee *et al.* 1979). Caughley (1976) found that elephant impact has been preventing recruitment of *Colophospermum mopane* (mopane) into taller size classes, but the impact have been found not to effect the regeneration thereof. Whyte & Wood (1996) found that 159 baobabs died between 1993 and 1996. Whyte, Nel, Steyn & Whyte (1996) expressed their concern over the decline in the baobab population due to elephant utilisation. Van Wyk & Fairall (1969) reported severe utilisation of marula by elephants especially in the southern region of the Kruger National Park. Joubert (1982) noted severe debarking and uprooting of marula trees by elephant bulls along tourist roads and Owen-Smith (1988) noted that the extent of elephant bark damage and felling of marula trees in the Kruger National Park was a source of concern to park managers.

In a vegetation impact study on elephants in protected areas adjacent to the Kruger National Park, Gadd (1997) reported that a minor part of the elephants' diet (0.14% of all feeding events) comprised woody vegetation with stems smaller than 2 cm in diameter. Seedlings were eaten in extremely small amounts by comparison to their overall availability. Lewis (1987) suggested that the mortality of marula seedlings in the Luangwa Valley should be attributed to browsers other than elephants, in particular impala (*Aepyceros melampus*). Walker, Stone, Henderson & Vernede (1986) found that marula seedlings in particular are highly palatable and may be killed by herbivores when not protected by other vegetation. Whyte (Pers. comm.)[2] has not observed any marula seedlings being utilised by specific elephant bulls over a two-year period during his study on vegetation utilisation by elephants. Gadd (1997) found that 45% of the marula trees (with a stem diameter >10 cm) that were surveyed suffered elephant impact, mainly branch breakage. Gadd (1997) also concluded that marula trees are capable of surviving any branch breakage if less than 75% of the tree is damaged. Branch breakage did not have a significant impact on the marula population surveyed, as most branch breakage was less than 50% to individual trees. Main stem breakage and bark stripping was the major cause for the 2% mortality of the marula population observed (Gadd 1997).

Opposed to the destructive impact that elephants may have on marula trees, Lewis (1987) concluded that elephants may have a positive impact on the marula population in the Luangwa Valley as they play a role in both tree recruitment and the dispersion of marula seeds. Results of this study showed that seeds that passed through an elephant's digestive tract had a much higher germination rate during their first year than those that did not (Lewis 1987).

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The feeding preference of elephants for certain species can cause a net decline of those species while permitting expansion of less palatable species (Laws 1970). Many studies have been done on elephant browsing on specific tree species (Thomson 1975; Okula & Sise 1986; Jachmann & Croes 199; Leuthold 1996; Whyte *et al.* 1996) out of concern that certain tree species will be lost due to overutilisation by elephants. Leuthold (1996) found that in the Tsavo National Park the original deciduous woodlands dominated by *Commiphora* spp. and *Acacia* spp. were being replaced over large areas by much more open wooded grassland. The major causes were thought to be (1) overutilisation of trees by elephants, which favored the establishment of a fairly continuous grass cover, and (2) fires that were nourished by the grass cover impeded the regrowth of woody vegetation (Leuthold 1996).

The Fire Factor

Past and present burning policies of the Kruger National Park

The inclusion of the effects and interaction of fire in this study is the general recognition that elephants and fire can have a highly significant impact on the species and structural diversity of tree and shrub vegetation in African savannas (Trollope *et al.* 1998). For a better understanding of the effects of fire on the vegetation of the Kruger National Park, Trollope *et al.* (1998) gave the following overview of the past and present burning policies as they pertain to the Park:

Indiscriminate burning was used between 1926 and 1954 to provide green grazing for wildlife. Since 1954 to 1975 a fixed triennial burning program was applied and the Park was divided into approximately 400 burning blocks generally ranging in size from 2500 to 5000 ha. The blocks were planned to be burnt every three years in spring after 50 mm rain had fallen. Major deficiencies in this burning program were identified. Many burns were scheduled but not carried out due to insufficient time; attempting only spring burns resulted in poor utilisation of grazing; and the frequent absence of rains after burning had deleterious effects on the recovery of the sward (Stocks, Van Wilgen, Trollope, McRae, Weirich & Potgieter 1993). During the application of this burning policy it became apparent that a fixed burning program was impractical to apply over the Park as a whole and did not constitute a natural fire regime. Accordingly in 1975 the program was adapted in the light of research results and field experience to permit burning during late winter, before and after the spring rains, mid-summer and during autumn. This burning was done on a rotational basis in order to provide short, palatable grazing throughout the growing season (Stocks *et al.* 1993). The frequency of burning was also changed in arid savannas to permit both triennial and quadrennial burning in order to provide different types of grazing habitat for concentrated bulk grazers (Stocks *et al.* 1993; Trollope *et al.* 1998).

A further adaptation was made to the burning program in 1980 to allow for an apparent ten-year quasi rainfall cycle where a decade of below-average rainfall is usually followed by a decade of above average rainfall. The rate of grass fuel accumulation is higher during the wet cycles resulting in a greater frequency of lightning fires, the perceived natural ignition source of vegetation fires. Consequently a variable burning frequency based on rainfall and the level of accumulation of grass fuel was introduced. It was believed that such a burning program would simulate a more natural fire regime where variable climatic conditions are a major driving force. The season of burning was also adapted to simulate fires caused by lightning which are generally limited to occur during mid-summer and high rainfall cycles. The majority of the controlled burns were consequently scheduled before and after the spring rains with mid-summer burns also being applied during above rainfall conditions (Trollope *et al.* 1998).

Finally in 1990 a further innovation was introduced to simulate point ignitions of fires which occur in a natural fire regime where lightning is the major ignition source. This involved combining the 400 burning blocks into 88 burn units, which were further grouped into 23 management units. It was believed that by burning larger areas (up to 30 000 ha) the fire front ignited around the perimeter of the burn unit would fragment into separate fires during the extended duration of the burn. These fragmented fires would then spread through the burn unit as a mosaic of different types of fires in response to change in wind direction, air temperature and relative humidity. This change was made in response to ongoing research being conducted on fire behaviour in the Park where it was recognised that the procedure where the burning blocks were ignited around the perimeter was resulting in the rangeland being burnt mainly by intense headfires. This was concluded to be preventing the recruitment of large trees as a result of the intense fires. Continuous back burns prevented species like marula from developing into large trees and, therefore, limiting the development of a parkland type of savanna in many areas. This procedure was followed until 1994 when a *Laissez Faire* burning policy was introduced, where only fires ignited by lightning were permitted to burn and all other ignition sources were controlled as far as possible (Trollope *et al.* 1998).

The two main causes of fires occurring in the Kruger National Park are man-induced controlled burns and unintentional fires that are caused by poachers, tourists, arsonists, accidents and lightning. The controlled burns are ignited around the perimeters of the burn blocks and are left to burn towards the centre which results in the formation of a well-developed smoke convection. This burn method is referred to as ringburning. Ringburning has the effect of increasing the intensity of the fire by drawing the fire fronts into the center of the block being burnt, therefore causing a maximum topkill of trees and shrubs (Trollope *et al.* 1998). Ringburning also leads to a disproportionately large area burning as a high-intensity headfire (an effect magnified by the fact that these fires are carried out during the day, and never at night) (Van Wilgen, Biggs & Potgieter 1998a). There is a significant difference between the intensity of controlled fires and wildfires. The latter occur as point ignitions which burn outwards towards the perimeter of the block as a mosaic of different types of fires ranging from intense headfires to less intense backfires and therefore possibly causing a lower topkill of trees and shrubs (Trollope *et al.*1998).

The Kruger National Park has a comprehensive set of fire records spanning over 5 decades and can be analysed spatially to provide information about the historic influence of fire on vegetation patterns and trends in the park. The fire records for different periods were recorded in the following three formats (Van Wilgen *et al.* 1998a):

(1) Sketch maps of the distribution of fires for each year from 1941 to 1956, available from a previous analysis of rangers' diaries. These were digitised, and overlaid on the boundaries of existing fire blocks to establish the percentage of the block that was burnt in that year. These fires were recorded at a coarse scale (1:500 000), resulting in partially burnt blocks only having a rough estimate of the percentage burned.

(2) The Board controlling the Park took a decision to institute prescribed burning in fixed blocks on a three-year cycle in 1957. Fire records of these blocks, giving the dates and causes of

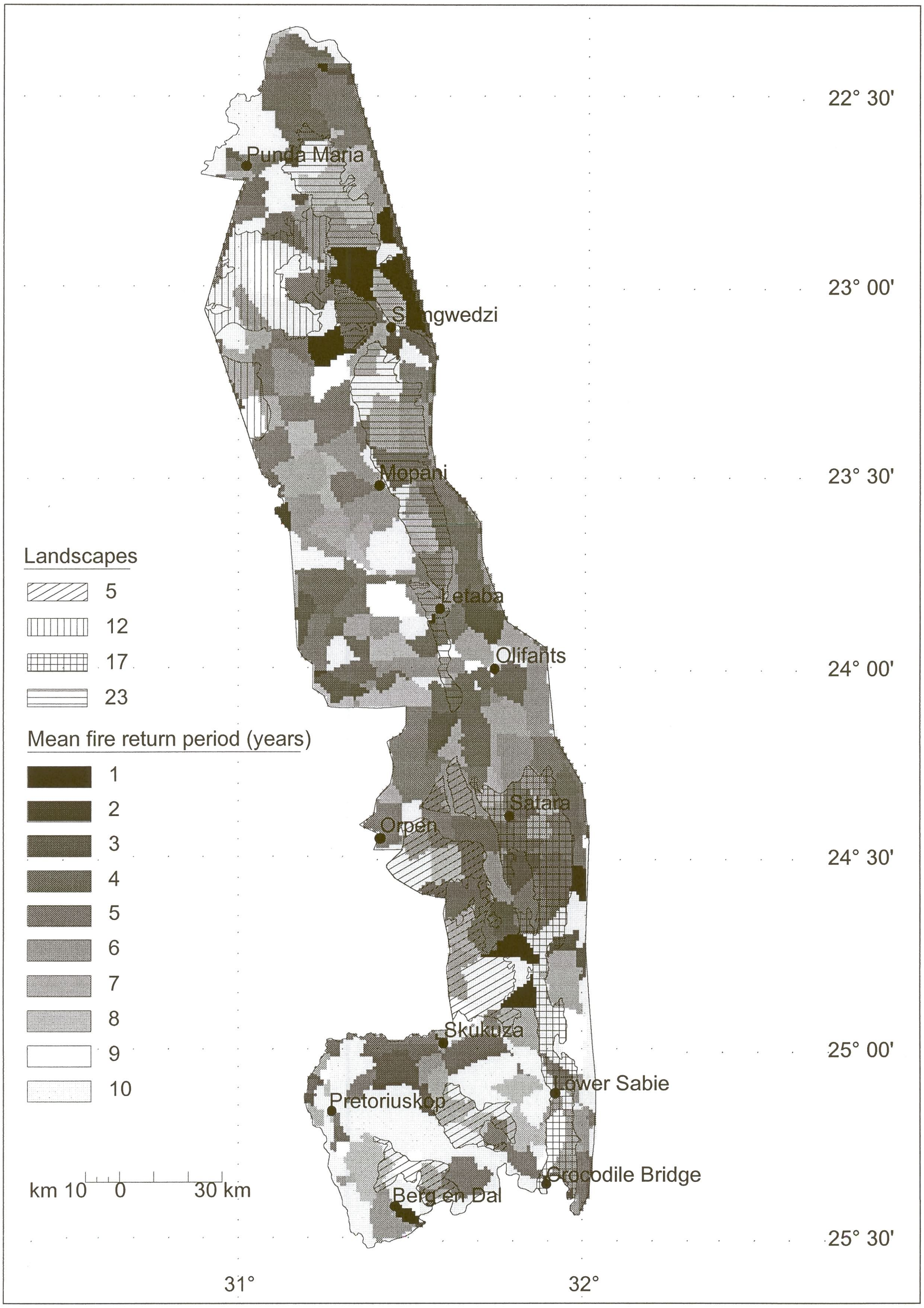
fires, and in some cases, an estimate of the percentage of the block that burned, are available for the period 1957 to 1991.These records were extracted from the management files for each block.

(3) As from 1992, all prescribed burns were stopped. Natural (lightning) fires were allowed to burn in line with a natural fire policy, although other fires did also occur. These fires were mapped and added to the database by allocating a date, percentage burned, and cause of each fire to the existing fire management blocks. Between 1957 and 1988, numerous changes and adjustments were made to the boundaries of the blocks. Van Wilgen, Biggs, Regan, Mare (1998b) re-allocated the fire records to the existing blocks, and estimated the mean fire return period throughout the Kruger National Park (Figure 2).

The effect of fire on the vegetation of the Kruger National Park

The effect of fire on vegetation depends upon the combined effects of the different components of the fire regime, being the type and intensity of fire and the season and frequency of burning (Bailey 1988; Trollope *et al.* 1998). Reviewing literature on the role of fire in savanna development, Trollope (1980) concluded that fire favours the development and maintenance of a predominantly grassland vegetation by destroying the juvenile trees and shrubs and preventing the progression of more mature plants to a taller fire-resistant stage. Generally herbaceous plants are favoured over woody plants because of the location of the perennating buds at or below ground level (Bailey 1988). Bailey (1995) further stated that the effect of fire on vegetation is contradictory depending on ecosystems and circumstances. Trollope & Potgieter (1985) recognised the need to obtain more information on the effect of fire behaviour and in particular fire intensity, on the flora of the Kruger National Park.

The Kruger National Park had different fire regimes for different periods in the Park's history, where protection against fire was followed by prescribed burning and then by a "natural" fire policy in 1994. Fires covering 16.79 million ha occurred between 1941 and 1996. The mind shift, away from rigid prescribed burning on a fixed dominantly triennial cycle has been in response to concerns after Trollope *et al.* (1995) concluded that a dominance of grass species characteristic of poorly managed pastures and overgrazing was a result of excessively frequent burning. Van Wilgen *et al.* (1998a) expressed concern over putative trends in woody vegetation



5=Mixed *Combretum/Terminalia sericea* woodland, 12= *Colophospermum mopane/Acacia nigrescens* savanna, 17= *Sclerocarya birrea/Acacia nigrescens* savanna, 23= *Colophospermum mopane* shrubveld

Figure 2. Mean fire return periods in the Kruger National Park for the period 1958 to 1996 (Source: Van Wilgen *et al.* 1998b).

structure. It would appear from examining early aerial photographs that large areas have been homogenized, possibly due to the rigid application of a triennial burn policy. Tree densities have declined dramatically for some species, and this decline may be due to the "unnatural" fire regime in combination with other factors, such as browsing by ever-increasing numbers of elephants (Van Wilgen *et al.* 1998a).

In an experimental fire applied in arid savanna areas of the Kruger National Park, surface headfires caused higher topkill of stems and branches compared to backfires (Trollope *et al.* 1995). Trollope (1983) and Trollope *et al.* (1998) found a significant correlation between fire intensity and topkill. Trollope & Potgieter (1985) classified fires into categories according to fire intensity (Table 1). Fire intensity is a measure of energy released in fires, which varies with fuel moisture content, wind and slope conditions under which the fires burn. Fire type will also affect fire intensity. Fire types include headfires (fires burning with the wind or upslope), backfires (fires burning against the wind or downslope) as well as ground fires (fires burning in the organic layers of the soil), surface fires (fires in the lower vegetation strata) and crown fires (fires in the canopies of trees) (Trollope 1983). The most common types of fires in savanna areas as found in the Kruger National Park are surface fires that burn either as head or backfires. Trollope (1983) found that a hot fire of approximately 2500 $kJ.s^{-1}.m^{-1}$ was necessary to cause a significant topkill of bush to a height of 2 m. Results of this study (Trollope 1983) also indicated that the topkill of bush does not increase appreciably when fire intensities are greater than 2500 $kJ.s^{-1}.m^{-1}$. Shrubs and seedlings in the < 0.5 m height class however, suffered a significant topkill of stems and branches, irrespective of the fire intensity (Trollope 1983).

Research in the Kruger National Park showed that the bush became more resistant to fire as the height of the trees and shrubs increased (Figure 3) (Trollope *et al.* 1998). The topkill of bush being 1 m in height is as high as 84.6% opposed to a topkill of 16.3% for bush of 5 m. These results indicate that generally the woody vegetation of the Kruger National Park is not significantly affected by fire alone when the trees and shrubs are taller than 3 m. Woody vegetation of the Kruger National Park is not sensitive to the season of burn (Trollope *et al.* 1995), but the frequency of burning, however, has a significant effect on the physiognomy of tree and shrub communities in savanna areas (Trollope 1983).

Table 1.

Fire categories according to fire intensity (Trollope *et al.* 1995).

Fire Intensity ($kJ.s^{-1}.m^{-1}$)	Description
<500	Very cool fire
501-1000	Cool fire
1001-2000	Moderately hot fire
2002-3000	Hot fire
>3000	Extremely hot fire

The visual difference in woody phytomass between the experimental control plots that have been protected from fire for 42 years and the annual, biennial and triennial burning treatments show a dramatic decline in the woody phytomass with an increase in the frequency of burning. Trollope *et al.* (1998) found the main effect of fire on the woody vegetation in the Kruger National Park is to cause a topkill of stems and branches, forcing the plants to coppice from the collar region of the stem. Fire in the Kruger National Park seems to have an effect mainly on the structure of the woody vegetation and not on the species diversity (Enslin, Potgieter, Biggs & Biggs 2000).

Bush surveys conducted inside and outside three elephant exclosures in the Kruger National Park that had been subjected to controlled burning for extended periods of time, led to the following conclusions on the effect of fire and the interaction of elephants and fire on the vegetation of the Kruger National Park. (Trollope *et al.* 1998):

- Neither fire nor the interaction of elephants and fire have a significant effect on the overall density of bush
- The interaction of elephants and fire causes a significantly marked reduction in the phytomass of bush in areas with clay soils irrespective of rainfall
- Elephants are having a significant impact on taller trees (>3 m) outside the exclosures. In all cases there was a higher proportion of trees inside than outside the exclosures in the >3 m height class.

Objectives

The most basic management objective of the Kruger National Park is to conserve all the species, which constitute the park's ecosystem, as well as the ecological processes that sustain these species. Implicit in this, is the fact that no single species should be allowed to threaten the existence of any other species of plant or animal. The Parks Board's policy on controlling elephant numbers as well as implementing a fire management plan rests on this objective of conservation (Hall-Martin 1992a). The current elephant population and fire regime in the Kruger National Park may both be seen as artifacts of man's interference in the system, and the loss of a species such as the marula tree from the ecosystem would clearly constitute a failure to achieve this objective.

The main objectives of this study were therefore:

I. To determine the status and the population structure of the marula (*Sclerocarya birrea* (A. Rich.) Hochst. subp. *caffra* (Sond.) Kokwaro (Kokwaro & Gillet 1980)) in four major landscapes (two on granite and two on basalt) of the Kruger National Park.

II. To determine the impact of the African elephant (*Loxodonta africana*, Blumenbach) on marula (*Sclerocarya birrea* (A.Rich.) Hochst. subsp. *caffra* (Sond.) (Kokwaro) (Kokwaro & Gillet 1980)) trees in the Kruger National Park.

III. To determine the effect of different fire treatments on the population structure and density of the marula (*Sclerocarya birrea* (A. Rich.) Hochst. subsp. *caffra* (Sond.) Kokwaro (Kokwaro & Gillet 1980)) in the Kruger National Park.

Hypotheses

To achieve the objectives, the following hypothesis have been compiled:

1. The population structure of the marula across the different landscapes and hence sub strata is not homogenous, but can be correlated to the elephant densities in the landscapes.
2. The extent of elephant impact to the marula population is high, but not homogeneous across the different landscapes.

3. Different fire treatments have different effects on the population structure and density of the marula.
4. Fire influences the morphology of the marula, enhancing a multi-stemmed morphology.
5. Fire and herbivory is preventing the young marula individuals (<2 m) from developing into adult trees, while elephants are preventing the establishment of a mature marula structure.

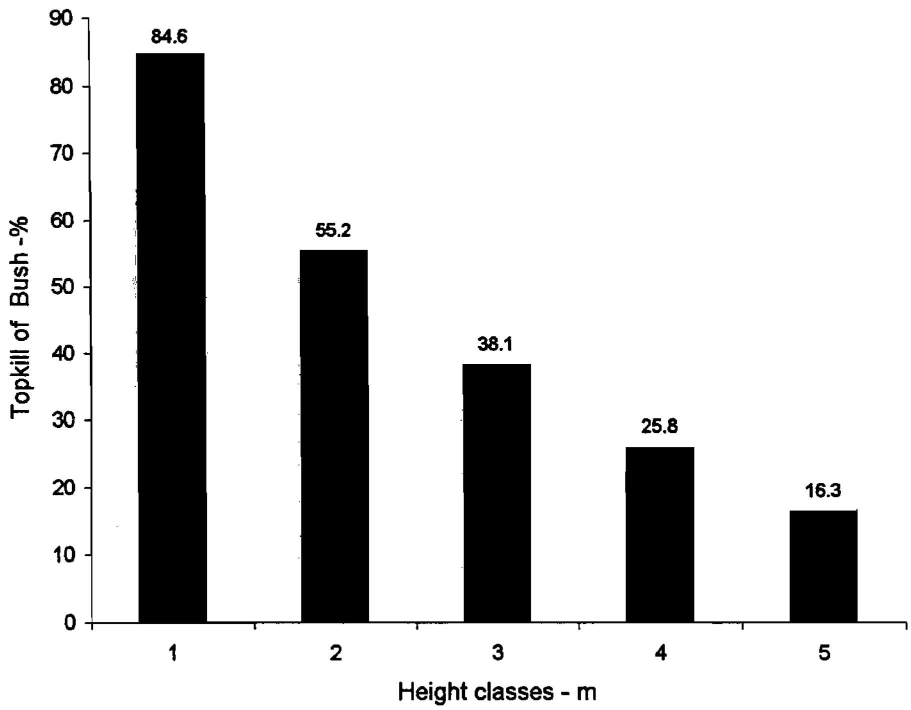


Figure 3. Effect of height on topkill of bush for different height classes subjected to a fire intensity of 3000 kJ. $s^{-1}.m^{-1}$ in the Kruger National Park (Trollope *et al.* 1998).

Assumptions

To test the hypotheses, the following assumptions were made:

1. The population structure derived from an elephant exclosure represents the normal population structure for the marula in the rest of the Park should there have been no elephant impact.

2. All uprooted marula trees and main stem breakage of trees with a circumference >10 cm is ascribed to elephant impact.
3. Branch breakage does not impact on the marula population structure.
4. Impact on marula seedlings is mainly ascribed to herbivores other than elephants.
5. Fire does not have a significant impact on the structure of marula trees >2 m.

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