SPATIAL AND TEMPORAL VARIATION OF THE FIRE REGIME IN MKUZI GAME RESERVE

By

CRAIG MICHAEL MULQUEENY

Submitted in fulfilment
of the academic requirements
for the degree of

MASTER OF SCIENCE
in the
School of Animal, Plant and Environmental Sciences
University of Witwatersrand

Johannesburg
2005

ABSTRACT

Fire is a key determinant of savanna dynamics, and would thus have a major influence on the vegetation dynamics of Mkuzi Game Reserve. Given this logic, it is an important and commonly used management tool in this reserve. Its main uses in the reserve are for either removing moribund material or for reducing woody plant encroachment, both of which normally entail dry season burns. As a consequence, fire often results in a green flush of vegetation that is highly favoured by grazing herbivores. A further management goal is maintaining or improving biological diversity by promoting vegetation heterogeneity. Current policy prescribes this should be achieved through point-source ignitions rather than by block-burning, which was the earlier practice.

This study explores spatial and temporal fire patterns at a landscape scale in Mkuzi Game Reserve using Geographic Information Systems (GIS). Much of our understanding of the dynamics of fire has previously been determined at a plot scale and scaling up of these insights to a landscape scale is problematic, hence this project aimed to contribute to our understanding of the dynamics of fire at a landscape scale. The study also specifically examined how the fire regime in the reserve has changed with a change in the burning philosophy and strategy, namely from block burning to the point source ignition (PSI) strategy, which began to be implemented in the mid-1980's.

Fire frequency was related to both geological type and vegetation type. The fact that geology was related to fire frequency was not surprising because the relationship between geology and vegetation in the reserve has previously been established. The varying amount of herbaceous material per vegetation type apparently influenced fire frequency. Spatial variation in fire frequency was also positively related to rainfall variation over the reserve, while the total area burnt per annum was positively related to the preceding wet season rainfall, but not for years with a high dry season rainfall. The influence of rainfall on grass production and thus fuel load explained these relationships. In addition, there was some evidence of a carry over effect of rainfall where the previous wet season rainfall together with the preceding wet season rainfall influenced total annual area burnt, but this was only significant for years when dry season rainfall was low. Contrary to an expected negative influence, dry season rainfall had no effect on the total annual area burnt. Grazer biomass had a significant limiting effect on fire frequency over the reserve (spatially), most likely due to consumption of herbaceous

material, but there was no relationship between grazer biomass and total annual area burnt (temporally). Dry season burns were significantly larger than wet season burns and can be attributed to the more favourable fuel condition during the dry season. Intense burns were also generally larger than the cooler burns, namely those rated as patchy/very patchy and clean. This was mainly attributed to a high fuel load which is critical for intense fires but also positively influences the spread of fire.

The comparison of the block burning strategy and the point source ignition (PSI) strategy showed that fire frequency was greater during the PSI burning period than during the block burning period. The total area burnt per annum was greater during the PSI burning period than during the block burning period, but individual burn sizes were not significantly different between the two strategies. Evidence showed that individual burns that occurred during the PSI period had boundaries that were more irregular than those of block burns. Fires were most common during the dry season for both burning strategies, but the proportion of the burns that occurred during the dry season was greater for the PSI burning period than for the block burning period. Evidence also showed that a much greater emphasis was put on applying dry season prescribed burns during the PSI period than during the block burning period. A greater effort was also made during the PSI period to burn firebreaks, which were only implemented during the dry season. Arson fires (started deliberately or accidentally by neighbours) were more common during the block burning period than during the PSI period, while under both burning strategies, they were more common during the dry season than the wet season. There was no distinguishable difference in the burn intensity patterns between block and PSI burning, that is, the proportions of burns in the different burn intensity classes were not significantly different between the two burning strategies. Although the contribution of the individual fire barrier types showed some change with a change from block burning to a PSI strategy, the combined contribution of natural barriers did not increase, and that of management barriers did not decrease, as would have been expected. In addition, natural and management barriers were apparently of equivalent importance during both burning strategies.

DECLARATION

I, the undersigned, hereby declare that this thesis is the result of my own original work, except where otherwise indicated and acknowledged, and that it has not been submitted for a higher degree in any other university.

C. M. MULQUEENY

DEDICATION

This work is dedicated to Maresce and Drew Mulqueeny

ACKNOWLEDGEMENTS

This work was undertaken whilst in the employ of the KwaZulu-Natal Nature Conservation Service (Ezemvelo KZNWildlife). The organisation is thanked for giving me the opportunity to undertake this work as part of my official duties, and thus allowing me the use of the organisation's equipment and resources. The many reserve management and research staff who were involved over the years in implementing fire management and collecting much of the historical data which was used in this study, are thanked for their dedication and hard work. Dr. Peter Goodman, who was the previous Ecologist in Mkuzi Game Reserve since 1975, is thanked for setting up the monitoring systems which made the long term analyses in this project possible.

I am very grateful and indebted to my supervisors, Prof. Tim O'Connor and Dr. Peter Goodman for their invaluable guidance, advice and encouragement throughout this study. I thank them also for being patient and understanding the pressure of undertaking this research in addition to my normal full-time work responsibilities.

I thank my Ezemvelo KZNWildlife supervisor, Dr. Jean Harris, for supporting and encouraging me to set aside time in my normal work schedule to undertake this research. The entire EKZNW research team in the region, especially Jeremiah Gumbi, are thanked for their continuous dedicated work which has in any way contributed to this research work, be it with regard to fire data or the other variables such as climate and large herbivore data.

Finally, I thank my wife, Maresce, who has supported me continuously and has remained a pillar of strength during a period which could only be described as challenging with the arrival of our son, Drew.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1: INTRODUCTION

1.1. BACKGROUND

Fire is a key determinant of savanna dynamics (Norton-Griffiths, 1979; Trollope, 1984a; O'Connor, 1985; Bond and Van Wilgen, 1996) and is therefore an important and commonly used management tool in savanna landscapes throughout the world, including Mkuzi Game Reserve (MGR). In wildlife management and nature conservation, traditional uses of fire are to ensure fodder flow to large herbivores by removing moribund grass material and for reducing woody plant encroachment (Edwards, P., 1984; O'Connor, 1985; Bond and Van Wilgen, 1996), both of which normally entail dry-season burns when grass is dormant (West, 1965; Trollope, 1984a). As a consequence, new growth of nutritious grass (green flush) normally follows burning (Edwards, P., 1984), and is highly favoured by grazing herbivores. In protected areas, fire has also been used to reduce the risk of accidental or arson fires (started deliberately or accidentally by neighbours) that may threaten the survival of plant species or destroy the composition or structure of a priority vegetation community (Edwards, P., 1984; Goodman, 1990; Bond and Van Wilgen, 1996; Berjak and Hearne, 2002). A more recently adopted fire management goal in a number of relatively large protected areas including MGR is that of maintaining or improving biological diversity by promoting vegetation heterogeneity (i.e. patchiness) (Trollope, 1984a; Bond and Van Wilgen, 1996; Brockett *et al.*, 2001). This use of fire is in line with the mission of KwaZulu-Natal Nature Conservation Service (Ezemvelo KZN Wildlife) which is primarily to conserve the indigenous biodiversity of KwaZulu-Natal, including landscapes, ecosystems and processes upon which they depend (Ezemvelo KZN Wildlife Policy Number 1, 1997). Current policy prescribes this should be achieved through point-source ignition (PSI) burning, or patch mosaic burning (Brockett *et al.*, 2001), rather than by block burning, which was the previous practice. Here the intention has been to manage processes (fire in this case) in a manner that closely resembles their natural or past state (Reserve Management Plan, 1986; Balfour and Howison, 2001). The implementation of the PSI burning system in MGR has therefore not been as rigid as that of the patch mosaic burning system described by, for example, Brockett *et al.* (2001), who recommend suppression of fires if they are likely to result in burning larger areas than defined by their system, whereas in MGR this is currently not the case.

The importance of understanding ecosystems and their disturbance regime, particularly in savanna, at a macro-scale has been recognised for a long time (Norton-Griffiths, 1979). The term savanna generally denotes communities or landscapes with a continuous grass layer and scattered trees, where the spatial pattern and relative abundance of grasses and woody plants are dictated by complex and dynamic interactions among climate, topography, soils, geomorphology, herbivory and fire (Scholes and Archer, 1997). Fire ecology has a reasonably long history of research, but almost all of the existing literature is concerned with a plot scale (Scott, 1984; Tainton and Mentis, 1984; Trollope, 1984a; O'Connor, 1985). Furthermore, our understanding of the spatial aspects of fire behaviour is poor (Frost and Robertson, 1985), and in general data on fire regimes in savannas is poor (Trollope, 1993, Russell-Smith *et al.*, 1997), particularly at the landscape scale (Balfour and Howison, 2001). Scaling up of the insights gained at a plot scale to a landscape scale is problematic. Hence there is an urgent need for an understanding of the dynamics of fire at a landscape scale. This study therefore aimed to contribute to this vacuum.

A long history of fire management in the reserve together with monitoring programmes for fire, vegetation, large herbivores and climate have made this research possible. Fire monitoring has involved mapping the extent of fires, a practice that was initiated in the early 1960's, and recording the fuel and environmental conditions, which was initiated some time later in the late 1970's. These conditions have been recorded as they affect the nature of the fires, particularly intensity and potential to spread (Luke and McArthur, 1978; Trollope, 1984b). Although there have been reports on large herbivores, vegetation and climate of MGR over the years, this thesis is the result of the first long-term analysis of the fire data for the reserve and will not only provide MGR managers with a greater understanding of the dynamics of fire in the reserve, but will also provide an important report-back on data that they have assisted with collecting over many years. In so doing, the importance of continuing such monitoring will hopefully also be highlighted.

1.2. OBJECTIVES OF THE STUDY

This study is concerned with fire patterns at a landscape scale. A number of factors could potentially have an effect on landscape-level fire behaviour and the manner in which landscape-level fire patterns are expressed, and these include rainfall patterns, substrate (geology and soils), vegetation, herbivory, timing of burns (season) and fire management strategy (e.g. block burning or point source ignition burning). We have very limited insight about their relative importance, particularly at a landscape level, and this study offers an opportunity to gain such insight. Current knowledge on the relationships between these variables and fire patterns, and expectations for MGR, are presented below as the rationale for their inclusion in this study, followed by a statement on the aim and objectives of the study.

In savanna fires, the main component of fuel is grass (Trollope, 1984a), the production of which is positively influenced by rainfall (Lauenroth, 1979; Dye and Spear, 1982; Sala *et al.*, 1988; Briggs and Knapp, 1995; O'Connor *et al.*, 2001). The amount of available fuel affects the intensity (Byram, 1959; Brown and Davis, 1973; Luke and McArthur, 1978; Trollope, 1984b), and in turn the spread of fire (Williams *et al.*, 1998). A positive relationship between the extent or spread of fire and fire intensity was found in Kakadu National Park, Australia (Williams e*t al.*, 1998; Gill *et al.*, 2000). In the mesic savanna of Hluhluwe-Umfolozi Park, both area burnt and frequency of fire were positively related to spatial and temporal variation in rainfall (Balfour and Howison, 2001). In the Serengeti, the extent of burning in a year was positively related to the wet-season rainfall and the frequency of fires was strongly correlated, spatially, to mean annual rainfall (Norton-Griffiths, 1979). The frequency and extent of fire in MGR is therefore expected to be directly related to spatial and temporal rainfall patterns, respectively.

A broad-scale relationship between geology and soil type, which in turn influences vegetation type and physiognomy, is well known for African savannas (Cole, 1986; Goodman, 1990; Moore and Attwell, 1999). For example, in the Transvaal Plateau Basin, the micro-climatic conditions are dependent on the stage of landscape evolution and the soils are related to superficial and bedrock geology whose deposition or exposure are in turn related to the stage in the geomorphological process, while the vegetation is influenced by the interplay of all these factors (Cole, 1986). Soil characteristics such as moisture and nutrient availability also influence grass production, which can influence the fire regime as explained previously (Dye and Spear, 1982; Walker and Langridge, 1997). Productivity is usually greater on heavier textured soils during wet years but can be greater on nutrient-poor sandy soils during drought years (Dye and Spear, 1982). The fire regime is therefore likely to vary across the different substrate (geology/soils) types, but will probably closely reflect the different vegetation types (fuel producers) occurring on them. Although an element of redundancy may exist with regard to the influence of substrate and vegetation, the relationship between geology and fire regime does provide a useful broad-scale view of fire patterns in MGR, and sets a platform for investigating finer-scale patterns.

Vegetation types differ in the amount of grass and in species composition, as well as in density of woody plants. In savanna environments, including MGR, the amount of herbaceous material is inversely related to the woody cover or density because woody plants dampen grass production through competition for resources (Walter, 1971; Pratchett, 1978; Walker *et al.*, 1981; Dye and Spear, 1982; Walker and Noy-Meir, 1982; Goodman, 1990; Scholes and Archer, 1997). Conversely, shade from trees in savanna can also improve water relations of the shaded grass plants (Belsky, 1994) and thus allow shade tolerant species to remain green well into the dry season. In both cases, the net effect of woody plants would be to inhibit spread and reduce intensity of fire. Hydromorphic grasslands within African savannas are characterised by saturated soils for part of the growing season and are maintained by a relatively shallow water table (Dean, 1967; Greenway and Vesey-Fitzgerald, 1969; Hughes, 1988; O'Connor, 2001). These habitats therefore also stay green, and thus relatively fire resistant (Luke and McArthur, 1978), well into the dry season. Furthermore, in moist environments and where fuel has high moisture content, fires are generally rare or absent (Bond and Van Wilgen, 1996). These vegetation types are therefore expected to experience fire less frequently and, when they do occur, fire will be of relatively low intensity due to high moisture content (Trollope, 1984b). With regard to varying grass species composition of different vegetation types, individual species can differ in their flammability due to varying moisture content (Trollope, 1984a), and primary productivity differs with differing species composition (Milchunas and Lauenroth, 1993; O'Connor *et al.*, 2001). Different vegetation types are therefore expected to be characterised by different fire regimes, with frequency and extent of fires expected to be positively related to the amount, and negatively related to the moisture content (physiological state), particularly during the dry season, of herbaceous material.

Herbivores, because they are area-selective (e.g. Leuthold, 1979; East, 1981), can depress the fuel load by feeding and trampling to the extent that fire is precluded from certain areas (Tainton, 1981; Bond and Van Wilgen, 1996). In addition, high grazing pressure can impact negatively on primary productivity (Illius and O'Connor, 1999; Fynn and O'Connor, 2000). Evidence from a lowveld savanna of Swaziland, southern Africa, showed that fire frequency is negatively related to grazing pressure (Roques *et al.*, 2001). Eruptions of wildebeest and buffalo in the Serengeti after the eradication of rinderpest, and consequent increase in grass consumption coincided exactly with a decrease in the extent of burning (Norton-Griffiths, 1979). It is therefore expected that the frequency of fire and the total area burnt per annum in MGR will be inversely related to the herbivore density or biomass in the reserve.

The season affects both the amount of fuel and the quality. In the dry season, there is more available fuel in terms of dry material and the fuel is usually more flammable because it has a lower moisture content as a result of generally lower relative humidity and grass being dormant (Schulze, 1972; Luke and McArthur, 1978; Trollope, 1984a; Trollope, 1984b; Lewis and Goodman, 1993). In the wet season, grass is actively growing (Trollope, 1984a) and is generally green and, in the case of MGR and other summer rainfall areas in the eastern part of South Africa, humidity levels are generally higher (fuel contains more moisture) (Schulze, 1972; Goodman, 1990; Lewis and Goodman, 1993), and thus flammability of the fuel is lower (Luke and McArthur, 1978; Trollope, 1984b). The occurrence of fire or ignition potential is significantly related to fuel moisture content, and decreases with increasing fuel moisture content (Luke and McArthur, 1978). It is therefore expected that fires will be larger and occur more frequently in the dry season than in the wet season. In the Serengeti, for example, fires only start in the dry season when the rains are over and the grass begins to dry out (Norton-Griffiths, 1979). In Kruger and Etosha National Parks, natural fires occur most frequently at the end of the dry season and just prior to the first spring rains (Siegfried, 1981; Gertenbach, 1988; Van Wilgen *et al.*, 1990).

Different fire management strategies can result in different landscape-scale fire patterns (Brockett *et al.*, 2001, Balfour and Howison, 2001). Variables that may be affected by a change in fire management strategy include fire frequency, burn size, geometric character of burns (e.g. patchiness), seasonal distribution of burns, burn intensity patterns and the relative importance of different barriers to fire spread.

The study area experienced a change, during the mid-1980's, from block burning to PSI burning (similar to patch mosaic burning) where the aim was to maintain and promote biological diversity through promoting spatial heterogeneity in fire patterns and thus vegetation (Brockett *et al.*, 2001; Bond and Van Wilgen, 1996). This requires the implementation of many small fires distributed widely over the protected area as opposed to few large fires, which is associated with a block burning strategy (Brockett *et al.*, 2001). It was therefore expected that the block burning period would have few large burns while the PSI period would have many small burns. The change in burning philosophy from block to PSI in the reserve was also accompanied by a deliberate change from burning in late winter and early spring to burning throughout the dry season (Goodman, pers. comm.) and thus it was expected that there would be differences in the seasonal distribution of fires (influences level of curing and moisture content) between the two periods, which in turn could have resulted in different burn intensity patterns (Luke and McArthur, 1978; Trollope, 1984b).

Barriers to fire spread can influence the patterns of fire occurring in a landscape (Gill *et al.*, 2000; Heyerdahl *et al.*, 2001). Barriers can be natural, such as watercourses, or man-made, such as roads and firebreaks. In general, it is expected that man-made barriers are relatively more important when a block burning strategy is used than when a PSI strategy is used. This is because the PSI strategy entails igniting fires at a point and allowing them to burn out against natural barriers, and where fires burn out against man-made barriers (e.g. roads), except firebreaks, the strategy requires that the fire should be re-ignited on the opposite side of the barrier. It was also for this reason that it was expected that the geometric character of burns (perimeter-to-area ratio) would differ between the two strategies, that is, PSI burns would be more irregularly shaped than block burns.

The aim of this study was therefore to define the relationship between spatial and temporal fire patterns, and environmental factors rainfall, vegetation, herbivory, substrate and season of burn at a landscape level in a savanna system, and to establish whether the implementation of different burning strategies has had any effect on fire patterns. Examining the effects of the two burning strategies on biological diversity was beyond the scope of this work, but if fire patterns are different then differences in vegetation heterogeneity, and consequently biological diversity, could be expected.

In order to achieve the above aim, the objectives of the research project were as follows:

1) Using Geographic Information Systems (GIS), describe the spatial and temporal variation in the fire regime of Mkuzi Game Reserve (MGR) from 1963 to 1999. Specific focuses were:

i) fire frequency in relation to spatial distribution of herbivores, rainfall, vegetation type and substrate (soil/geological type);

ii) area burnt in relation to soil/geological type, annual rainfall patterns, large herbivore biomass, fire intensity and season of burn; and

iii) the influence of barriers on fire spread.

2) Determine the difference in fire patterns between the block burning and point source ignition (PSI) burning strategies, in terms of fire frequency, burn size, geometric character of burns (perimeter-to-area ratio), burn intensity, season of burn, and relative importance of different fire barrier types.

CHAPTER 2: THEORETICAL AND CONCEPTUAL FRAMEWORK

2.1. INTRODUCTION

Spatial and temporal patterns of fire and the use of fire as a conservation management tool form the focus of this study and thus it is important to understand all aspects of fire including behaviour, factors influencing behaviour, and its use in ecosystem management. In addition, the study is concerned with savannas, in which there is a complex relationship among fire, rainfall, herbivores, both herbaceous and woody vegetation, and substrate (geology and soils). This chapter attempts to summarise relevant literature concerning the above.

2.2. FIRE MANAGEMENT

2.2.1. Introduction

Fire is a widely used tool in the management of ecosystems since it is a relatively inexpensive option. It can be used in various ways to achieve various objectives. Some common management objectives include: manipulating the fire regime to favour the quantity and quality of fodder plants for livestock production; in nature conservation and game farming, manipulating habitat structure to favour certain plant species or to improve habitat for animals; control of invasive alien plants; enhancing the water yield from catchments or the prevention of soil erosion; and the reduction of fire hazard by frequent burning to reduce fuel load (Edwards, P., 1984; Bond and Van Wilgen, 1996). The lack of fire in certain ecosystems is often desirable, for example in the fire sensitive sand forests of Maputaland, and therefore fire management involves both the use and prevention of fires.

2.2.2. Uses of Fire in Ecosystem Management

2.2.2.1. Livestock Production, Wildlife Management and Nature Conservation

Fire is used to maintain veld in a condition that ensures a high quantity (productive) and quality (palatable) of available grazing for livestock and large indigenous herbivores (Edwards, P., 1984; Bond and Van Wilgen, 1996). Both productivity and palatability of grass are related to species composition (Tainton, 1981; Snyman and Fouché, 1991; Illius and O'Connor, 1999) which can be directly affected by fire, although rainfall variability is thought to play the most important role, with fire as well as grazing being modifying rather than controlling factors (O'Connor, 1985). Plant production increases from what is conventionally termed poor condition veld through to good condition veld (Snyman and Fouché, 1991). With regard to the fire regime, the two most important parameters in terms of their effect on grass composition of mesic savannas, are the season of burning and the frequency of burning (Trollope, 1984a; Van Wyk, 1972; O'Connor, 1985). The most pronounced compositional changes have generally resulted from annual dry-season burning on heavy textured soils. In a large protected area like Kruger National Park (KNP), the mean frequency of burning in the sweetveld (low rainfall) areas has been octennial while in the sourveld (high rainfall) areas it has been triennial, with most of the fires occurring during the dry, dormant winter period, except for lightning fires that have been responsible for only 10% of burnt areas (Trollope, 1993). In KNP, where the greatest proportion of the reserve has been burnt by controlled burns, the aim of controlled burning is to remove moribund and/or unpalatable grass material and to maintain an optimum balance between grass and tree vegetation (Trollope, 1993). Furthermore, the complete lack of burning can be detrimental to veld condition and cause species compositional changes as a result of the accumulation of moribund material, with one such example being the rapid die out of *Themeda triandra* as a result of protection from fire (West, 1965; O'Connor, 1985).

Fire is used to retard woody plant encroachment (Edwards, P., 1984; Trollope, 1984a), which can impact negatively on veld condition through a reduction in overall basal cover and replacement of desirable perennial grass species by inferior biennial and annual species (Tainton, 1981). In addition to woody plant encroachment leading to a decline in grass cover (Goodman, 1990; Smit and Rethman, 1999), it also leads a decline in plant production (West, 1947; Davies, 1947; Kennan *et al.*, 1955; Plowes, 1956; Rattray, 1957; West, 1958; Dye and Spear, 1982). Therefore, the most important effects of fire on herbaceous species composition and basal cover are indirect and considered to be through the direct effects of fire on the woody component (Joubert, 1966; O'Connor, 1985). Although, in terms of direct effects, fire is a modifying rather than a controlling force on savanna grasslands, the role of fire varies over the savanna spectrum, and in high rainfall savannas the effects of fire on the woody component make it a controlling force of these grasslands (O'Connor, 1985). In addition, the effect of fire on basal cover and yield generally increases from sandy through to heavier textured soils (O'Connor, 1985).

Fire is used to promote spatial heterogeneity in vegetation so as to promote biological diversity (Brockett *et al*., 2001; Bond and Van Wilgen, 1996). The emulation of the natural or past fire regime in savanna is generally the goal in this case, as the natural fire regime is believed to have resulted in a fire mosaic of areas burnt by different types and intensities of fire occurring at various times and frequencies, all of which maintained a diversity of habitat and species (Trollope, 1984a). Patch mosaic burning or the point source ignition (PSI) method of burning has been applied in protected areas in savanna to achieve this goal (Brockett *et al*., 2001). For example, in Pilanesberg National Park, the implementation of many small fires distributed widely over the protected area (patch mosaic burning) as opposed to few large fires (as in block burning) has been prescribed to achieve spatial heterogeneity in fire patterns (Brockett *et al*., 2001). This approach is also in line with many conservation agencies' policies, including that of Ezemvelo KZN Wildlife, to follow a process-based management philosophy where natural processes are maintained, re-instated or where the former are not possible, simulated (Reserve Management Plan, 1986; KZN Wildlife Policy No.1 - Mission Statement, 1997).

The manipulation of habitat structure to favour certain plant species or improve habitat for certain animals can be achieved through the use of fire (Edwards, P., 1984). Fire is often used to increase the amount of available browse material by converting tall wooded vegetation to shorter shrublands (Tainton, 1981; Edwards, P., 1984). As a further result, protective cover for certain species of birds and large herbivores is also improved. An example where these objectives were successfully achieved is along the Pacific Coast where blacktailed deer increased several fold when forest was converted to 'brush' (Dasmann and Dasmann, 1963; Heady and Child, 1994). In South Africa, similar use of fire in savanna to optimise habitat, particularly for rare species such as black rhinoceros, has been employed (Emslie, 1999).

Fire is also sometimes used in attempts to reduce the number of ticks in a system (Edwards, P., 1984). However, burning only destroys ticks that are on the herbage at the time of the fire, and not those on the hosts and in sheltered positions. This, together with the fact that ticks have a high reproductive potential, makes the effect of fire on tick populations only temporary (Heady and Child, 1994).

Fire is one of the most effective means of limiting the spread of fire in vegetation and can be used to create fire breaks which protect vulnerable areas (Edwards, P., 1984). In protected area management, fire breaks are used to prevent the spread of fire beyond protected area boundaries, this being a legal requirement of The Forestry Act (72 of 1968) (Edwards, P., 1984), later Forest Act (122 of 1984) and more recently National Veld and Forest Fire Act (101 of 1998), for safety reasons or the prevention of damage to neighbours' property (Edwards, P., 1984), and to restrict fire within specific management blocks.

2.2.2.2. Alien Plant Control

Fire is often used to control alien invasive plants. In South African fynbos shrublands, alien species such as American and European *Pinus* species and Australian *Hakea* species are felled and left to lie before being burnt. During this time, the seeds fall to ground and are either eaten or germinate. About a year after felling, prescribed burns are applied, which effectively kill the seedlings before they mature, allowing indigenous species to reproduce normally (Van Wilgen *et. al*., 1992; Bond and Van Wilgen, 1996). In KwaZulu-Natal, a cut and burn method has been used for controlling some alien plants, but generally, fire is used as a preventative measure as part of normal veld management practices (Macdonald and Jarman, 1985). Certainly, in Mkuzi Game Reserve (MGR), areas that are prone to fire have much lower infestations of *Chromolaena odorata* as compared to the areas which are less prone to fire, for example, along drainage lines (pers. obs.).

2.2.2.3. Water Yield in Catchments

Fire has been used in many catchments to reduce biomass and thereby increase water yield. In some cases, fire has even been applied to convert woodland and forest to grasslands in order to improve water yield (Bond and Van Wilgen, 1996). Experiments in South Africa and the USA have shown that the burning of brushlands may improve water yield as opposed to the burning of grasslands which showed no detectable increase in water yield (Bosch *et. al*., 1984). It must be noted, however, that unconditional burning for maximum water yield may result in severe deterioration of the vegetation and eventually the soil mantle (Bosch *et. al*., 1984).

2.2.3. Constraints to Burning

The use of fire as a management tool is constrained by many factors including ecological factors (biological responses), temporal constraints (environmental factors), legal constraints, safety and global concerns (global atmospheric changes) (Bond and Van Wilgen, 1996).

2.2.3.1. Ecological Constraints

Various biological responses to fire limit the way in which fire can be used in various ecosystems and under various land uses. For example, winter fires in South African fynbos have detrimental effects on certain species, and therefore this fire regime cannot be applied (Van Wilgen *et. al*., 1990; Bond and Van Wilgen, 1996). In conservation, however, it is argued that variation in the fire regime is important for ensuring species coexistence (Cowling, 1987; Hobbs and Atkins, 1988; Bond and Van Wilgen, 1996). In other words, a varied fire regime should promote the maximisation of biological diversity. This is at least applicable to savannas where the natural fire regime undoubtedly resulted in a mosaic of areas burnt by different types and intensities of fire occurring at various times and frequencies, all of which promoted biological diversity (Trollope, 1984a). It has also been argued that at intermediate levels of disturbance, that is, fire at intermediate intervals, diversity should be at a maximum (Huston, 1979). In general, long intervals without disturbance will result in sufficient time for the exclusion of species by strong competitors, while very short intervals could result in elimination of all species except those most resilient to disturbance (Bond and Van Wilgen, 1996). A number of studies support the intermediate disturbance hypothesis by showing that species richness is greatest at intermediate fire intervals (Hobbs *et. al*., 1984; Le Roux, 1989; Van Wilgen and Forsyth, 1992). However, contrary results have been reported by Collins and Gibson (1990).

2.2.3.2. Other Constraints

With regard to the temporal constraints to the use of fire, there are two sorts. Firstly, conditions during a particular day or over a season may prevent actual combustion from taking place, that is, wet, cold conditions or fuel containing too much moisture such as in the growing season (Bond and Van Wilgen, 1996). Secondly, hot, dry conditions or extended drought periods will limit the application of prescribed fires, mainly for reasons of safety (Bond and Van Wilgen, 1996). Safety, therefore, is a limiting factor in the use of fire, since prescribed fires often become unmanageable and can cause damage to property and even loss of life. Concerns about safety, pollution and the ecological effects of fires, have also resulted in the formulation of a number of laws restricting the use of fire. In South Africa, the regulations controlling the use of fire are contained in the Forest Act (No. 122 of 1984), the Conservation of Agricultural Resources Act (No. 43 of 1993) and the National Veld and Forest Fire Act (No. 101 of 1998). There is also a growing concern about the role of fire in contributing to global atmospheric changes. Some atmospheric chemists have proposed major changes in land use to reduce the effects of burning on atmospheric carbon with little regard for ecological consequences (Levine, 1991; Bond and Van Wilgen, 1996). If any of these proposed changes are taken seriously and implemented, they will be an additional constraint on the management of fire-prone ecosystems.

2.3. FIRE BEHAVIOUR

2.3.1. Introduction

The term "fire behaviour" describes the release of heat energy and is dependent on fire intensity, rate of spread of the fire front, flame characteristics and other related phenomena. The study of fire behaviour therefore involves understanding the manner in which heat energy is released and the various factors influencing it (Trollope, 1984b). In order to study fire behaviour, a basic understanding of the phenomenon of combustion is also necessary (Trollope, 1984b).

2.3.2. Combustion

Combustion is described as an oxidation reaction, which requires the proper combination of heat, oxygen and fuel (Heady and Child, 1994). Variation in the rate of combustion is controlled by the balance among these three factors. Combustion has also been described as a special form of oxidation in which large amounts of heat energy are rapidly released from the fuel (Luke and McArthur, 1978). It is therefore clear from these two descriptions that combustion both requires heat and releases heat.

When heat is applied to fuel, three phases of combustion occur. Firstly, drying occurs which is accompanied by the start of decomposition, but the reaction is not self-sustaining if the heat source is withdrawn. In the second phase, decomposition is completed and is accompanied by flaming in a self-sustained reaction. Finally, when the vapours produced during the decomposition process have been consumed, the residual charcoal burns away until only mineral ash is left (Luke and McArthur, 1978).

Plant fuels are mainly composed of cellulose ($[C_6H_{10}O_5]n$). The complete combustion of a molecule of cellulose may be expressed by the following reaction:

$$C_6H_{10}O_5 + 6O_2 \rightarrow 6CO_2 + H_2O + \text{heat}$$

For this reaction to proceed, a high temperature and an ample supply of oxygen is required, while the product of this reaction is carbon dioxide, water vapour and a large amount of heat energy (Luke and McArthur, 1978).

In terms of weight, lignin, the bonding agent holding wood fibres together, is the next most important constituent after cellulose. The chemical structure of lignin is more complex than that of cellulose, and requires the application of considerably more heat than for cellulose, before it decomposes. The main products of decomposition include tars in the form of highly flammable vapours (Luke and McArthur, 1978).

In addition to cellulose and lignin, many tree and shrub species contain resins, volatile oils and waxes, which add considerably to the flammability of available fuel, especially the living foliage. Although they occur in relatively small amounts, they often ignite readily, and by heating other fuel particles, they tend to accelerate the general rate of fuel ignition, even to the extent of helping to sustain burning in damp or green fuel, which would otherwise be barely flammable (Luke and McArthur, 1978). Many of these secondary compounds are viewed as antiherbivore defences, which on one hand deter herbivores, but on the other hand, encourage fires (Bond and Van Wilgen, 1996).

2.3.3. Heat Yield and Fire Intensity

The total amount of energy contained per unit mass of fuel is referred to as the heat of combustion. Not all of this heat is released during combustion. The actual amount of heat released during a fire decreases with increasing moisture content of the fuel, and is termed the heat yield (Bond and Van Wilgen, 1996). Heat yield can be determined from heat of combustion and moisture content values using correction factors (Figure 2.1) (Byram, 1959;

Bond and Van Wilgen, 1996). Heat yield is in turn used to calculate fire intensity and this can be achieved using Byram's (1959) equation for fire-line intensity:

$$I = Hwr$$

where I is the fireline intensity (kWm^{-1}), H is the heat yield of the fuel (Jg^{-1}), w is the mass of fuel consumed (gm^{-2}), and r is the rate of spread of the fire (ms^{-1}). Heat of combustion values do not vary greatly among plant species and are probably much less important in determining flammability than mass, structure and moisture content. Furthermore, some plants contain high levels of oils, fats, waxes and terpenes, which are volatile and thus increase their flammability, and which have up to twice the heat of combustion values as other plant material and thus increase the intensity at which these plants burn (Bond and Van Wilgen, 1996).

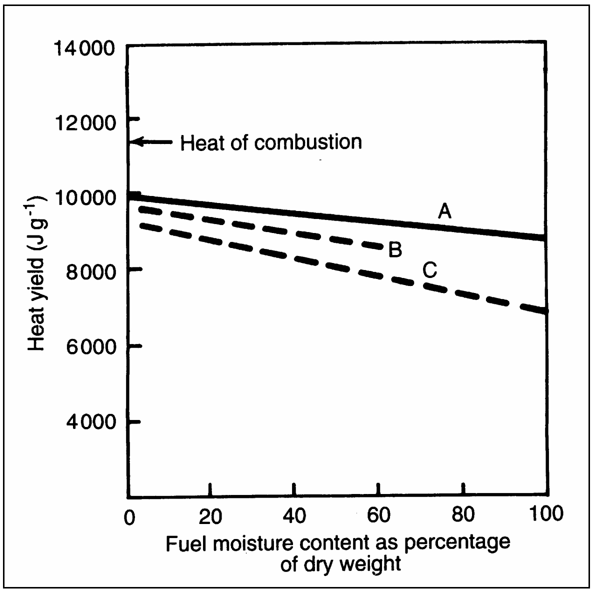


Figure 2.1: The relationship between heat of combustion, moisture content and heat yield in wildland fuel complexes (after Byram, 1959; Bond and Van Wilgen, 1996). Lines A, B, and C show estimates of heat yield for complete combustion, small fires and large fires, respectively.

2.3.4. Influence of Fuel on Fire Behaviour

2.3.4.1. Fuel Particle Size

With regard to the effects of fuel particle size, plant fuels have been classified into two broad types; fine fuels comprising all plant material with a diameter up to 6mm which burn very readily, and heavy fuels of greater than 6mm diameter, in which combustion is incomplete because of the great bulk of the material (Luke and McArthur, 1978). With fine fuels such as grass, ignition is almost instantaneous throughout a piece of grass, while heavy fuels such as dry wood will light up only on its surface. The complete combustion of fuel is fairly normal in grass fires, but rarely if ever occurs in forests, largely due to the great bulk of the fuel in forests. Studies have shown that there is greater combustion of fine fuels compared with heavy fuels in mountain shrubland and fynbos in South Africa (Le Maitre, 1981; Smith, 1982; Trollope, 1984b). In savanna fires, fine fuel in the form of grass constitutes the main fuel component (Trollope, 1984a). Furthermore, heavier material cannot readily ignite unless fine fuels are present, and when ignited add little to the flaming phase of combustion (Luke and McArthur, 1978).

2.3.4.2. Vertical Distribution of Fuel

Three broad groups of fuel can be recognised, namely, ground, surface and aerial fuels. Ground fuels include all combustible material below the loose surface litter and comprise decomposing plant material. These fuels support glowing combustion in the form of ground fires, which are difficult to ignite but are persistent once ignited. Surface fuels comprise loose surface litter, seedlings, forbs, shrublet communities and standing grass swards. These are fine fuels and can support intense surface fires. Aerial fuels comprise both live and dead combustible material located in the understorey and upper canopy of trees and shrubs. This group of fuels consists mainly of mosses, lichens, epiphytes, and branches and foliage of trees and shrubs, which can support high intensity crown fires (Brown and Davies, 1973; Trollope, 1984b). In savannas, only surface fires occur, while the tree layer is affected by the intensity of the surface fires (Bond and Van Wilgen, 1996).

2.3.4.3. Compaction of Fuel

Compaction of fuel refers to the way in which individual pieces of fuel are placed in relation to each other. Combustion is optimised when the fuel is sufficiently loosely packed to allow enough oxygen to reach the flame zone but dense enough for efficient transfer of heat to occur through radiation and convection (Luke and McArthur, 1978). Fuel spacing is especially critical in heavy fuels, which only form a minor component of fuel in savanna fires, but adequate ventilation generally occurs in the majority of fuel types (Luke and McArthur, 1978; Trollope, 1984a). Some fuels are so compacted that oxygen is not readily available, with the result that the rate of burning is slow (Luke and McArthur, 1978). This normally occurs in ground fires and is therefore not common in savannas. A good example of this situation is the slow combustion that occurs in peat beds, while a similar situation may be observed in compacted forest litter.

2.3.4.4. Fuel Moisture

Fuel moisture is a critical factor in determining the intensity of a fire since it affects ease of ignition, quantity of fuel consumed and combustion rate of different types of fuel (Trollope, 1984b). From previous discussion it was shown that fuel moisture impacts on heat yield from fuel and thus impacts on the intensity of a fire. Fuel moisture also significantly affects fire behaviour through the smothering effect of water vapour leaving the fuel and diluting the oxygen in the air immediately surrounding the fuel (Brown and Davis, 1973; Trollope, 1984b). The occurrence of fire or ignition potential is also significantly related to fuel moisture content, and decreases with increasing fuel moisture content (Luke and McArthur, 1978). Moisture content of live fuel usually varies gradually in response to seasonal and climatic changes while in dead fuel, moisture content is hygroscopic and is affected by changes in relative humidity and atmospheric temperature (Luke and McArthur, 1978; Trollope, 1984b). It has been suggested that the moisture content of grass can also vary according to the species. Grass species such as *Panicum maximum* that occur under bush clumps in savanna generally have higher moisture content than the grasses growing between bush clumps and are thus less flammable (Trollope, 1984a). Research conducted in Kruger National Park and the eastern Cape has clearly illustrated the effects of fuel moisture on fire intensity (Potgieter, 1974; Trollope 1978; Trollope, 1984b).

2.3.4.5. Fuel Load

Fuel load is a major contributor to fire intensity. Fire intensity is directly proportional to the amount of fuel available for combustion at any given rate of spread of the fire front (Brown and Davis, 1973; Trollope, 1984b). The importance of the mass of fuel consumed is also shown by Byram's (1959) equation for fire-line intensity which was discussed in section

2.3.3 on heat yield and fire intensity. Research in South Africa has shown that fuel load generally accounts for between approximately 30% and 60% of the variation in intensity among grassland fires (Trollope and Potgieter, 1983; Trollope, 1984b).

2.3.5. Influence of Climate and Weather on Fire Behaviour

2.3.5.1. Introduction

Climate is usually described in annual or seasonal terms, while weather refers to daily or even hourly atmospheric conditions. Climate usually affects the occurrence of fires through its effects on primary productivity. In some African savannas, for example, fires may only burn in years of above-average rainfall when, even after defoliation by grazing, there is a net accumulation of biomass. Fires, however, can occur under most climatic regimes, provided dry conditions occur at some time during the year (Bond and Van Wilgen, 1996). Weather, on the other hand, is an important determinant of how and when fires will burn. Its influence on fire behaviour can be both direct and indirect.

2.3.5.2. Wind

Wind directly affects the spread and therefore the intensity of fires (Brown and Davis, 1973; Luke and McArthur, 1978; Trollope, 1984b). It brings oxygen to the flames and removes carbon dioxide, while also moving hot air masses ahead of the flame close to the ground, where radiant heat dries and preheats new fuels. This makes ignition easier or even spontaneous ahead of the advancing flames (Heady and Child, 1994). However, it has been shown that although the rate of spread generally increases with increasing wind speed, in grasslands the rate of spread tends to decrease once the wind speed exceeds 50km/h (Luke and McArthur, 1978). Wind speed is also inversely related to flame height since increased wind speeds cause flames to assume a more acute angle with the ground, which partly explains why crown fires do not always occur during very windy conditions (Luke and McArthur, 1978; Trollope, 1984b) (Figure 2.2). From fire behaviour studies in South Africa, it has been concluded that at relatively low wind speeds, overall atmospheric wind conditions do not significantly affect the behaviour of surface head fires and that these fires are largely influenced by the air movements generated by the convection column (Trollope, 1984b).

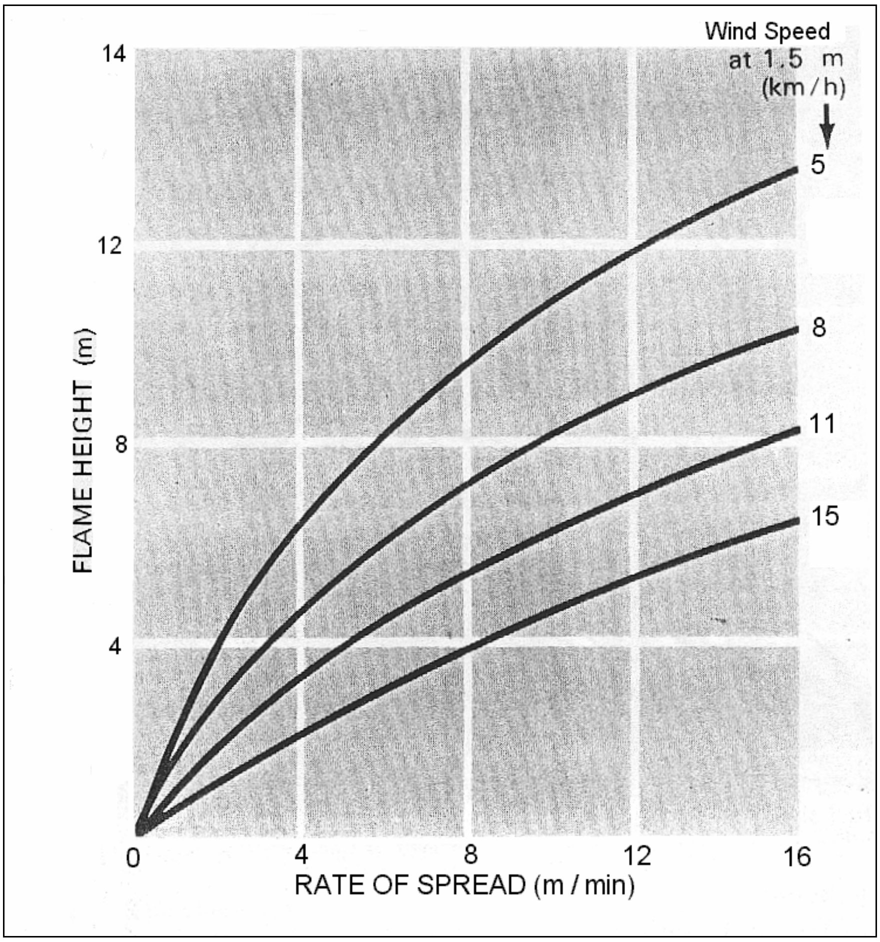


Figure 2.2: The relationship between flame height and rate of spread in dry sclerophyll eucalypt forest 10-15 m high and carrying 20 t/ha of available fuel at various wind speeds (after Luke and McArcthur, 1978).

2.3.5.3. Rainfall, Relative Humidity and Air Temperature

Rainfall, relative humidity and air temperature affect fire behaviour indirectly through their influence on the moisture content of vegetation and litter. In general, the fuels are drier and more combustible when relative humidity is low (Trollope, 1984b; Bond and Van Wilgen, 1996). Air temperature influences relative humidity and moisture losses from fuel by evaporation (Luke and McArthur, 1978; Trollope, 1984b). The relationship between air temperature, relative humidity and moisture content of cured grasslands (Figure 2.3) shows that relative humidity has more influence than temperature on moisture content (Luke and McArthur, 1978). According to Trollope (1984b), fire intensity is potentially high when relative humidity is less than or equal to 30%. An obvious and direct effect of rainfall and other forms of precipitation such as dew is that of dampening fuel, which prevents or suppresses combustion. Air temperature also has a direct effect on fire behaviour, since high temperatures result in less heat being needed to raise fuel temperature to the ignition point, and for continued combustion as a fire spreads (Brown and Davis, 1973; Trollope, 1984b; Bond and Van Wilgen, 1996).

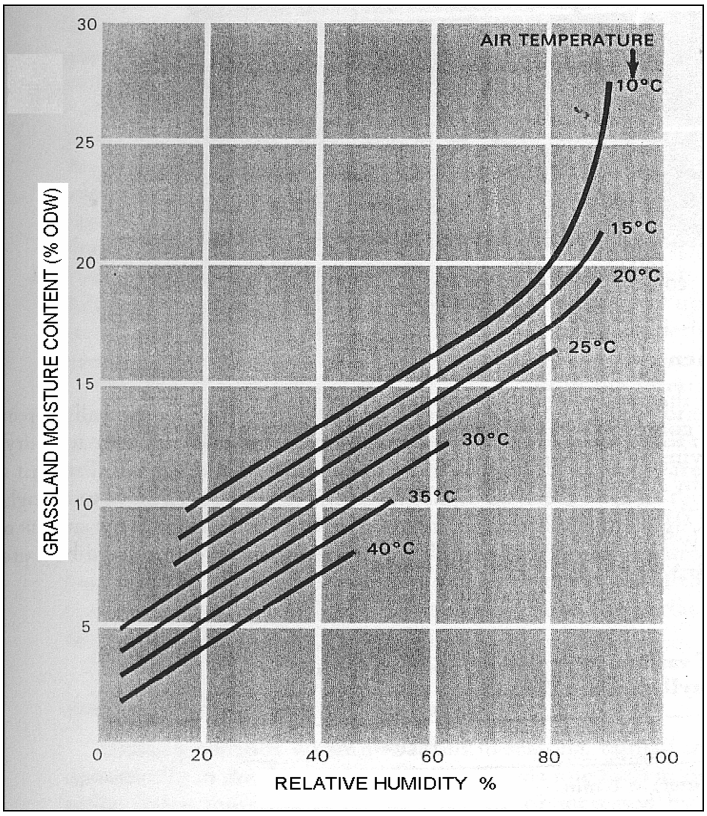


Figure 2.3: Fuel moisture content of cured standing grasslands related to relative humidity and screen temperature (after Luke and McArthur, 1978)

2.3.6. Influence of Topography on Fire Behaviour

2.3.6.1. Roughness of Land Surface

Roughness of the land surface influences weather generally, causing day-to-day variation in weather, and hence fire behaviour. In general, mountaintops tend to be cooler and moister than the lowlands by day but warmer and drier by night. Altitude also influences the growing season and fire season, for example, grasslands and shrublands at low altitudes may be subject to burning for six months or longer, while alpine grasslands a few kilometres away may not be dry enough to burn (Heady and Child, 1994).

2.3.6.2. Slope

Slope can influence the rate of spread of a fire, depending on whether the fire is moving upslope or downslope (Luke and McArthur, 1978). In the case of fires burning upslope where slopes are at least 15^0 to 20^0, flames occur at an acute angle to the ground surface, thereby increasing the degree of preheating of unburnt fuels immediately in front of the flames by means of radiation and convection (Figure 2.4). This preheating ahead of the flames, in effect, results in an increased rate of spread of fire (Luke and McArthur, 1978; Trollope, 1984b). Conversely, the rate of spread of a surface fire is decreased on a down-slope (Luke and McArthur, 1978). In experimental fires in eucalypt and grass fuels in Australia, results indicated that the rate of forward progress of a fire on level ground doubles on a 10^0 slope and increases almost fourfold travelling up a 20^0 slope (Luke and McArthur, 1978).

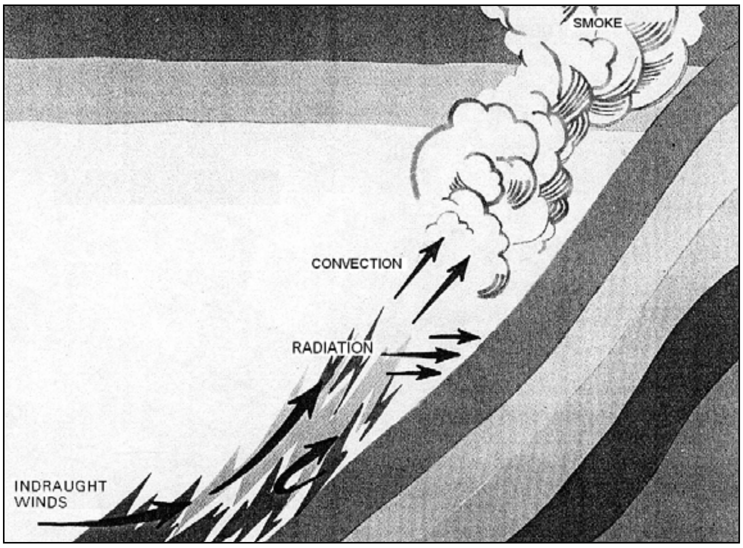


Figure 2.4: Schematic diagram of the effect of an upslope fire increasing radiation and convection heating of fuel ahead on the flame (after Luke and McArthur, 1978)

2.4. Factors affecting Fuel Load in Savannas

2.4.1. Introduction

Surface fires occur in savanna ecosystems, burning with or against the wind as head or back fires (Trollope 1984a; Bond and Van Wilgen, 1996). The most common fuel in surface fires is grass, which thus forms the main fuel component of savanna fires (Trollope 1984a, 1984b).

A number of factors influence the accumulation of grass fuel load, which in turn influences the probability of fire occurrence and fire intensity (Brown and Davis, 1973; Trollope, 1984b). These are either related to grass production or attrition of grass biomass, and include rainfall, substrate (geology and soil type), herbivory (grazing, trampling and browsing) and woody plant biomass (vegetation structure) (Figure 2.5).

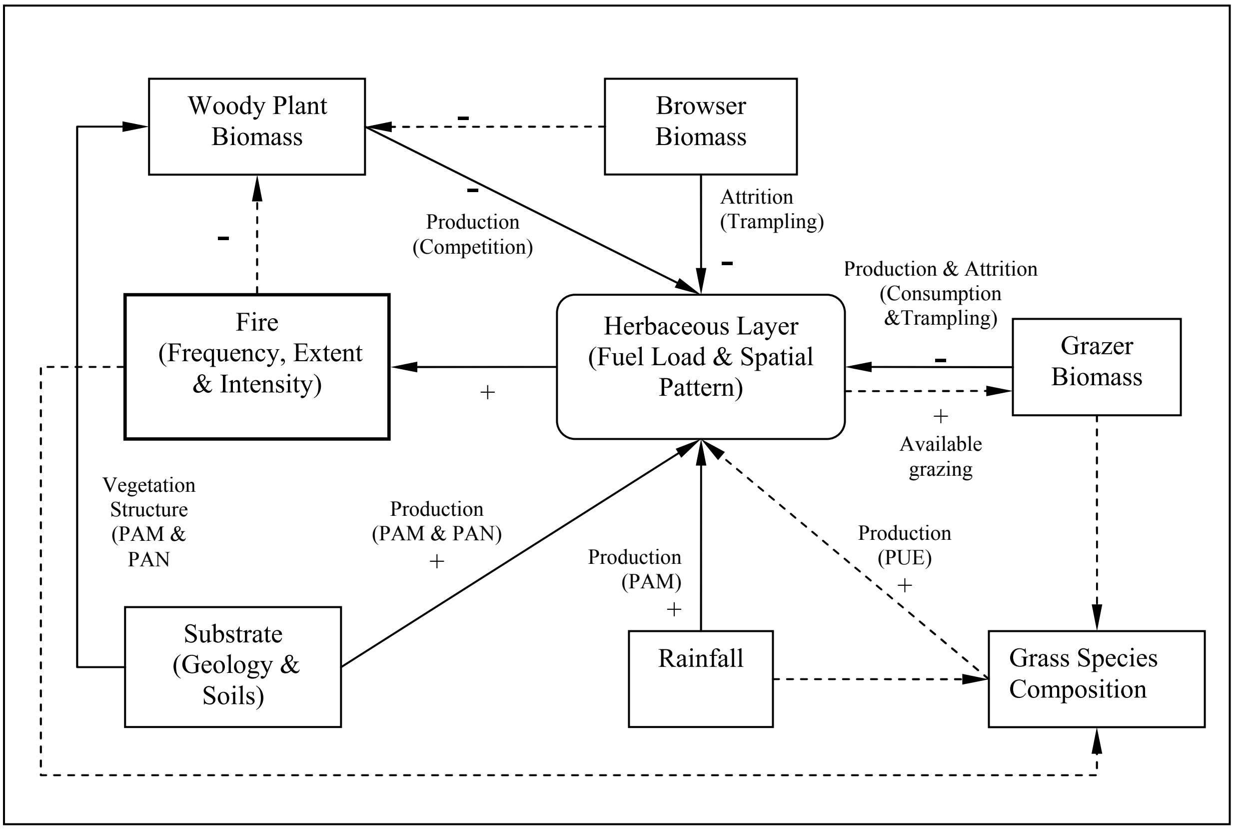


Figure 2.5: Conceptual model of factors affecting the herbaceous layer (fuel load and spatial pattern) and fire patterns (frequency, extent and intensity) in savanna. The relationships indicated by solid lines are relevant to this study while those indicated by dotted lines have not been explored in this study. A plus sign (+) indicates a positive influence and a minus sign (-) indicates a negative influence. Arrows that cross are not linked. Key to acronyms: PAM, plant available moisture; PAN, plant available nutrients; PUE, precipitation use efficiency.

2.4.2. Influence of Substrate on Vegetation and the Amount of Herbaceous Material

Geology influences soil type, which in turn influences the vegetation physiognomy and type, while different physiognomic classes and vegetation types are usually characterized by different amounts of herbaceous material. The broad-scale relationships among these factors are well known for African savannas, and indeed in MGR, geology and soils are strongly correlated with vegetation (Cole, 1986; Goodman, 1990). In the Transvaal Plateau Basin, the micro-climatic conditions are dependent on the stage of landscape evolution and the soils are related to superficial and bedrock geology whose deposition or exposure are in turn related to the stage in the geomorphological process, while the vegetation is influenced by the interplay of all these factors (Cole, 1986). Savanna structure of natural (undisturbed) plant communities has been ascribed primarily to soil moisture and soil nutrients (Bell, 1982; Tinley, 1982; Walker and Langridge, 1997; Moore and Attwell, 1999), and woody plant-grass competition for these resources (See section 2.4.5). In general, fine-grained soils favour a shrub savanna, while tree savannas are associated with coarser grained soils (Moore and Attwell, 1999). Furthermore, woody plant biomass is usually negatively related to herbaceous plant biomass because of competition for resources (See section 2.4.5.) and in MGR, vegetation physiognomic types with high woody plant cover have low herbaceous cover (Goodman, 1990). The fire regime is expected to vary according to the amount of herbaceous material and thus according to vegetation physiognomic type, and in turn should vary according to substrate because of the influence of soils and geology on vegetation.

2.4.3. **Effect** ***of Rainfall on Grass Production***

Rainfall directly influences the growth of herbaceous material in a relationship where grass yields increase linearly with increasing rainfall (Sinclair, 1979; Dye and Spear, 1982; Fynn and O'Connor, 2000; O'Connor *et al.*, 2001). The influence of precipitation on primary production has been recognised at a global (Lauenroth, 1979), regional (Sala *et al.*, 1988), landscape (Briggs and Knapp, 1995) and individual site level (Dye and Spear, 1982; O'Connor *et al.*, 2001). It has been suggested that, at least at a site level, seasonal precipitation may better predict production than annual precipitation (Hulett and Tomanek, 1969; Shiflet and Dietz, 1974; Smoliak, 1986), and there may be a carry-over effect of production on that of the succeeding year (Hanson *et al.*, 1982; O'Connor *et al.*, 2001). The effect of rainfall on grass production can be enhanced as compositional state of the veld as well as basal cover improves (O'Connor *et al.*, 2001). The influence of compositional state on grass yield is related to precipitation-use efficiency (PUE) of the species characteristic of different veld conditions. Species characteristic of a good condition in a semi-arid environment have been shown to produce more above-ground phytomass per unit rainfall than species characteristic of a poorer (medium) condition. Deteriorating composition results in production which is less in amount, efficiency and reliability from year to year (O'Connor *et al.*, 2001). A decrease in phytomass production on rangeland reduced to poor composition is a typical response across sites world-wide (Milchunas and Lauenroth, 1993). In semi-arid environments, basal cover can be a key determinant of infiltration versus runoff and hence water status of the soil, which in turn affects phytomass production. Runoff is greater in poor condition environments when basal cover is low than in good condition environments when basal cover is high (O'Connor *et al.*, 2001). In addition, basal cover is inversely related to the amount of exposed soil and thus the amount of water loss due to soil evaporation increases with deteriorating veld condition (Snyman, 1988; O'Connor *et al.*, 2001), which ultimately leads to a reduced above-ground phytomass production.

In savannas, grass species changes and resultant change in contribution to grass total yields, can take place as a result of variation in annual rainfall (Morris, 1980; Dye and Spear, 1982; O'Connor, 1985; O'Connor, 1999). More mesic species are favoured by above-average rainfall while the relatively xeric species decline (Dye and Spear, 1982). Furthermore, basal cover of grass species can decline dramatically as a result of severe drought conditions (Van Wyk, 1967), or undergo some decline even as a result of a single low rainfall season during a wetter period (Van Rooyen and Theron, 1982).

2.4.4 Effect of Soils on Grass Production

Two variables, namely plant available moisture (PAM) and plant available nutrients (PAN) are important when considering the effect of soils on grass production. PAM is influenced by soil texture. On heavier (clay) soils, infiltration of rain is low compared to that of coarse-textured soils, and the penetration of a given amount of rain occurs to a relatively shallow depth due to high water-holding capacity (Dye and Spear, 1982). Despite the high water holding capacity of heavier soils, the high clay content of these soils results in less water being available to plants, particularly when rainfall is limited, while in wetter periods PAM for a clay is higher, for the same rainfall, than PAM for a sand (Walker and Langridge, 1997). In heavier soils, there is a greater capillary movement of moisture, which means that there is greater opportunity for evaporative loss of moisture to the atmosphere, compared to sandy soils (Dye and Spear, 1982). By contrast, coarse-textured (sandy) soils are more prone to percolation loss from the upper soil layers through to the lower soil layers (Dye and Spear, 1982; Frost, 1987). Although grass biomass is positively related to PAM, the relationship is strengthened (higher coefficient of determination - r^2) when PAN is also considered in multiple regression analyses (Scholes and Walker, 1993; Walker and Langridge, 1997). The effect of soils on grass production is thus well illustrated by Dye and Spear (1982), in which the slope of the relationship between rainfall and annual grass production is strongly dependent on soil type, and is much steeper for the fertile loams and clays than for infertile sands, that is, grass production per unit rainfall is greater on heavier fertile soils than on coarse-textured infertile soils. However, during low rainfall periods grass production can be higher on sandy infertile soils than on fertile heavier textured soils (Dye and Spear, 1982), which is partly related to the fact that sands have much lower wilting points than heavier soils (Walker and Langridge, 1997).

The level of degradation (chemical and physical) in soils can affect the amount and duration of availability of plant moisture and thus primary productivity. Degraded soils have a much lower water holding capacity than non-degraded soils, and in vertisols, this can be as much as about 40% lower (Seiny-Bouker *et al.*, 1992). Runoff increases while recharge accordingly declines drastically as soil becomes degraded. For example, a case study in a northern Cameroon savanna showed that recharge was 3 to 4 times greater for a non-degraded as compared to a highly degraded vertisol. In addition, the study showed that periods of water availability to plants were 3 to 5 times greater for the non-degraded soils as compared to the highly degraded soils (Seiny-Bouker *et al.*, 1992). In a South African semi-arid savanna, declines in primary production occurred with heavy stocking rates, but only on the more sloping sites where it was likely that increased run-off and soil erosion had occurred (Fynn and O'Connor, 2000).

2.4.5. Effect of Woody Biomass on Grass Production

The fact that high bush densities decrease grass yields has been widely recognised from an early date (West, 1947; Davies, 1947; Kennan *et al.*, 1955; Plowes, 1956; Rattray, 1957; West, 1958) and was attributed to competition for water, nutrients and light (Walter, 1971; Walker and Noy-Meir, 1982), but it appears that water (soil moisture) is probably the most important factor (Pratchett, 1978; Walker *et al.*, 1981; Dye and Spear, 1982). The effect of trees on grasses does, however, range from negative to neutral to positive, and has been studied at the scale of an isolated tree to a landscape scale, but at the latter level of resolution there is often a strong negative correlation between tree density or cover and grass cover or biomass (Scholes and Archer, 1997). The negative effect of trees on grasses may result from rainfall interception, litter accumulation, shading, root competition, or a combination of these factors (Scholes and Archer, 1997). In areas where bush encroachment occurs, grass production typically declines dramatically as woody plant cover or density increases (Scholes and Archer, 1997). Where tree removal has occurred, the typical relationship between grass production and tree biomass, cover, or basal area was a negative exponential with the steepest decline in grass production resulting from the initial increments of tree cover (Walker *et al.*, 1972; Pressland, 1975; Burrows *et al.*, 1990; Scholes and Archer, 1997). Similarly, negative relationships between grass growth and tree cover were found in response to naturally occurring variations in tree density (Jameson, 1967; Scifres *et al.*, 1982; Pieper, 1990; Scholes and Archer, 1997)

The positive effect of trees on savanna grass production can be through improved fertility and structure of soil below crowns, and improved water relations of shaded plants (Belsky, 1994). A number of recent studies (Stuart-Hill *et al.,* 1987; Belsky *et al.*, 1989; Frost and McDougald, 1989; Weltzen and Coughenour, 1990) have documented that isolated trees may improve understory productivity (Belsky, 1994). This increase in productivity is localised under or near tree crowns and is found most often in the tropics and subtropics and in communities with low tree density (Stuart-Hill and Tainton, 1989), low rainfall and moderate soil fertility (Belsky, 1994). In this case, trees provide shade which reduces water stress on hot days (Walker, 1974) or act as nutrient pumps, drawing soil minerals from deep underground and depositing them on the soil surface (Bosch and Van Wyk, 1970; Stuart-Hill and Tainton, 1989). In addition, Acacia trees, which are common in African savannas, are legumes and may improve nitrogen availability to the sward (Belsky, 1994). In contrast, trees occurring in communities with high tree density, high rainfall, or extremely nutrient-poor soils display the more expected pattern of reduced understorey productivity (Burrows *et al.,* 1988), and tree removal increases herbaceous productivity (Walker *et al.,* 1986; Pieper, 1990; Belsky, 1994; Smit and Rethman, 1999). In general, where the positive effects of soil enrichment and improved water relations outweigh the negative effects of competition, then herbaceous understory productivity is enhanced, while if the effects of competition for resources outweigh the positive aspects, productivity is expected to decline (Belsky, 1994).

In MGR, woody plant cover was negatively correlated with grass cover (Goodman, 1990). Vegetation physiognomic types in the reserve with high woody plant cover have low herbaceous cover. Sand forest has an extremely poorly developed herbaceous layer, riverine forest and woodland usually has a sparse herbaceous layer, open to closed woodland of mixed bushveld and red sand bushveld has a herbaceous layer which is not well developed or generally fairly sparse, and thicket has a poorly developed grass layer (Moll, 1968, 1980; Goodman, 1990). Elsewhere, tree thinning experiments have demonstrated an increase in herbaceous cover (reduced bare ground) with a reduction in woody species density (Smit and Rethman, 1999; Smit, 2003).

2.4.6. Effects of Herbivores on Grass Production and Attrition

Grazing, which may be selective for species and plant parts (Tainton, 1981; Heady and Child, 1994), is a major influence on the vegetation biomass and potential fuel mass (Edwards, D., 1984). Herbivores affect grass, and thus the fuel load, by direct consumption or defoliation and trampling (Tainton, 1981; Heady and Child, 1994; Pickup, 1994), and indirectly through the influence of grazing on primary production (Milchunas and Lauenroth, 1993; Illius and O'Connor, 1999). High levels of herbivory in fertile savannas and grasslands, the so-called "sweet-veld" areas of low rainfall and/or unleached fertile soils, effectively reduce the occurrence of fire in many areas. In contrast, infertile savannas and grasslands, the so-called "sourveld" grasslands of moist climates and leached and low nutrient soils, are not subject to high levels of grazing, accumulate fuel and burn (Edwards, D., 1984; Bond and Van Wilgen, 1996).

Grazing pressure, measured by total grazer biomass, is negatively related to the total amount of grass available as fuel. In arid central Australia, the distribution of grazing by domestic animals was negatively related to vegetation cover (Pickup, 1994). In the Serengeti, eruptions of wildebeest and buffalo, after the eradication of rinderpest, resulted in an increase in grass consumption, which coincided exactly with a decrease in the extent of burning (Norton-Griffiths, 1979). In lowveld savanna of Swaziland, southern Africa, fire frequency was negatively related to grazing pressure, which was explained by the fact that sustained heavy grazing of grasses removed combustible herbage, which reduced the probability of ignition and restricted the spread of fire (Roques *et al.*, 2001).

Grazing can affect primary productivity, and these changes are often associated with changes in species composition of the herbaceous vegetation (Illius and O'Connor, 1999; Fynn and O'Connor, 2000). These species compositional changes are partly a result of selective grazing, which initially favours those species that are not utilised relative to those which are frequently grazed (Tainton, 1981). Persistent heavy grazing associated with high stocking rates inevitably leads to some scale of change in composition and yield (O'Connor, 1985). However, the effect of grazing on species composition is also dependent on rainfall, which independently can have a major influence on species composition (O'Connor, 1985). A number of studies have demonstrated the effect of drought combined with grazing on grass species composition (e.g., Tacheba and Mphinyane, 1993; Hodgkinson, 1995; O'Connor, 1995; Fynn and O'Connor, 2000). In general, the effect of grazing on species composition appears to be the greatest at low-rainfall sites (semi-arid savannas), which tend to experience high rainfall variability, and least in high-rainfall savannas, moderated by the influence of soil type on available soil moisture, that is, sandy soils are less prone than clay soils to drought deficits (O'Connor, 1985; Illius and O'Connor, 1999). Where the effect of grazing on species composition is extreme, primary production can be reduced by as much as 40% (Milchunas and Lauenroth, 1993; Illius and O'Connor, 1999). In a semi arid savanna in southeast Zimbabwe, in heavily stocked communal grazing land, the herbaceous layer was dominated by annual grasses and ephemerals, lacked perennial grasses, had low standing biomass and litter cover, had more bare ground, and had markedly lower herbaceous production under low rainfall, than sites under ranching with lower stocking levels (Kelly and Walker, 1976; Illius and O'Connor, 1999). In a grazing experiment in a South African semi-arid savanna, long term heavy grazing on sloping land resulted in a decline in herbaceous production, and the depletion of herbaceous biomass in an area when grazed heavily was more pronounced if grass species composition had changed as a result of drought and grazing (Fynn and O'Connor, 2000).

One of the main effects of intense utilisation of vegetation by herbivores is to reduce both aerial and basal cover, thereby reducing infiltration and hence plant production (Illius and O'Connor, 1999; O'Connor *et al.*, 2001). For example, rainfall-induced fluctuations in annual growth that were up to four times larger in sites dominated by annual grasses than in perennial-dominated vegetation, have been partly ascribed to the lower infiltration of rainwater into bare ground than where litter was present (Kelly and Walker, 1976; Illius and O'Connor, 1999).

The effects of trampling on the ecology of a grassland can be both negative and positive, and have been summarised below (Tainton, 1981; Heady and Child, 1994). Trampling can cause severe damage to leaves of the grass plant. Damage to the meristematic regions of the grass plant, caused by trampling, can also be detrimental. The amount of damage caused by trampling differs among herbaceous species, with creeping species, particularly those with rhizomes, experiencing less damage than the more upright species, while species with soft succulent leaves are more susceptible to damage. Trampling tends to affect younger grass plants more severely and can reduce grass seedling survival (Salihi and Norton, 1987; Sun, 1990). Trampling also promotes decomposition of litter and recycling of nutrients. Trampling of soils, especially when the moisture condition is about midway between wilting and field capacity, can result in compaction, which in turn leads to reduced infiltration capacity, water storage capacity, aeration and root penetration, and thus lower plant production. In *Cymbopogon-Themeda* veld of the Orange Free State, infiltration capacity was lower on all grazed veld compared to ungrazed veld (Van den Berg *et al.*, 1976; Tainton, 1981). In mopane woodland, infiltration capacity was higher under light utilisation intensities than under moderate to heavy utilisation, although in the absence of utilisation, infiltration was lower (Kelly and Walker, 1976; Tainton, 1981). The latter phenomenon is probably because of the chipping effect of animal hooves, usually on dry soil, which can reverse the effect of soil surface sealing or "capping". Loosening of the soil by trampling can also make it vulnerable to erosion by wind as well as by water if the infiltration capacity remains low or field capacity is exceeded. Reduced production on heavily stocked sloping sites has been attributed to increased run-off and soil erosion (Fynn and O'Connor, 2000). The net effect of trampling can therefore be to reduce grass biomass and thus the amount of grass available as fuel for fires.

2.5. Fire Patterns

2.5.1. Introduction

At a landscape level, fire patterns may be affected by factors such as spatial and temporal variation in rainfall and herbivore density, vegetation type, substrate (geology and soils) through its influence on vegetation and barriers to fire spread. Fire patterns are also affected by the fire management strategy employed, and thus may change with a change in strategy, as in the case of a change from a block burning to a point source ignition (patch mosaic) burning strategy in MGR in the mid-1980's. In addition, a change in fire management strategy may also result in a change in the relative importance of various fire barrier types.

2.5.2. Spatial and Temporal Patterns of Fire

Fire frequency is an important measure of disturbance history and refers to the number of times a community has been disturbed by fire over a given period of time. It can vary substantially over space within an ecosystem (Norton-Griffiths, 1979; Balfour and Howison, 2001; Heyerdahl *et al*., 2001). In savannas, the frequency of fire is influenced by the amount of herbaceous material available (Trollope, 1984a). The proportion, or size, of an area burnt is positively related to fire intensity (Williams *et al.* 1998; Gill *et al.*, 2000), which in turn is positively related to the fuel load (Brown and Davis, 1973; Trollope, 1984b). The various factors that influence the amount of available fuel have been discussed in Section 2.4.

Rainfall and soil nutrient availability, two factors which directly influence the amount of fuel (Dye and Spear, 1982; O'Connor, 1985; O'Connor *et al.*, 2001), can vary substantially over space and thus influence the spatial patterns of fire in terms of frequency and the extent of burns. This was evident in both the Serengeti and Hluhluwe-Umfolozi Park where the area burnt and fire frequency were positively related to spatial variation of rainfall (Norton-Griffiths, 1979; Balfour and Howison, 2001). In the Serengeti, the frequency of fires was strongly correlated, spatially, to mean annual rainfall (Norton-Griffiths, 1979), while in Hluhluwe-Umfolozi Park, high altitude areas burnt more frequently than low altitude areas where the mean annual rainfall was about 300mm lower (Balfour and Howison, 2001).

Geology and soils often correlate closely with vegetation and MGR is no exception (Cole, 1986; Goodman, 1990). Different vegetation types may be characterized by different types and amounts of fuel (Bond and Van Wilgen, 1996). In general, at a landscape level, woody plant biomass in a vegetation type is usually negatively related to herbaceous plant biomass because of competition for resources (Scholes and Archer, 1997). For example, in MGR, vegetation physiognomic types with high woody plant cover have low herbaceous cover (Goodman, 1990). The varying amounts of herbaceous material in the different vegetation and physiognomic types thus means that fire frequency may vary with vegetation type (Whitney, 1986; Bond and Van Wilgen, 1996). Furthermore, vegetation such as hydromorphic grasslands, which are characterised by saturated soils for part of the growing season and are maintained by a relatively shallow water table (Dean, 1967; Greenway and Vesey-Fitzgerald, 1969; Hughes, 1988; O'Connor, 2001), are not likely to experience frequent fires because of the expected high moisture content of the fuel. Fire frequency is thus expected to vary spatially over a landscape according to vegetation or physiognomic type, and in turn according to substrate because of the influence of soils and geology on vegetation.

All herbivore species are likely to show some spatial variation in population density, with higher densities in localised areas of favourable habitat (East, 1981), and thus their influence on fuel and fire frequency varies accordingly. For example, the mean density of wildebeest in the Serengeti in 1972 varied from 35 km^{-2} within the plains to 21 km^{-2} in the central parts to 17 km^{-2} in the north (Norton-Griffiths, 1979). According to Bond and Van Wilgen (1996), high levels of herbivory in fertile savannas and grasslands effectively reduce the occurrence of fire in many areas. This is related to the fact that, at a landscape level, grazing intensity is negatively related to vegetation cover (Pickup, 1994), in turn to the probability of fire occurrence and ultimately fire frequency (Roques *et al.*, 2001). The influence of herbivores on fuel load and thus fire frequency was evident in the Serengeti where fires occurred with low frequency over the period from 1963 to 1972 on the plains where the wildebeest density was highest compared with the northern stratum where the wildebeest density was lowest as estimated in 1972 (Norton-Griffiths, 1979).

Change in fire frequency has also been associated with factors such as climate change, which may occur over relatively long periods of time (Clark, 1988, 1990; Balfour and Howison, 2001). In Minnesota, using reconstructed fire histories of mixed conifer/hardwood forests and independent reconstructions of climate, it was established that fire was most frequent during the warm, dry fifteenth and sixteenth centuries and least frequent during the Little Ice Age (Clark, 1988, 1990; Bond and Van Wilgen, 1996). In Hluhluwe-Umfolozi Park, a South African mesic savanna, fire frequency was higher during wet climatic phases than during dry phases (Balfour and Howison, 2001). Similarly, it would be expected that the total extent of fires per annum would be positively related to temporal rainfall variation, and in particular rainfall of the previous summer (wet) season (Norton-Griffiths, 1979; Balfour and Howison, 2001). This was apparent in the Serengeti (Norton-Griffiths, 1979) and more recently in Hluhluwe-Umfolozi Park (Balfour and Howison, 2001).

Another important temporal factor to consider is that of season of burn, which can have a bearing on the extent of a fire, partly because relative humidity varies with season (Schulze, 1972; Lewis and Goodman, 1993) and thus also the fuel moisture content (Luke and McArthur, 1978). In MGR, for example, relative humidity is lowest during the winter months (dry season) and highest during the summer months (wet season) (Lewis and Goodman, 1993), while other areas in the eastern summer rainfall region of South Africa show similar patterns (Schulze, 1972). Fuel moisture affects ease of ignition, quantity of fuel consumed and combustion rate of different types of fuel (Trollope, 1984b). In addition, grasses are dormant during the dry season (Trollope, 1984a) and contain much less moisture than during the growing phase, that is, during the wet season (Luke and McArthur, 1978). It can therefore be expected that fires will be larger and occur more frequently during the dry season than during the wet season. In the Serengeti, for example, fires only start in the dry season when the rains are over and the grass begins to dry out (Norton-Griffiths, 1979), while in Kruger and Etosha National Parks, natural fires occur most frequently at the end of the dry season and just prior to the first spring rains (Siegfried, 1981; Gertenbach, 1988; Van Wilgen *et al.*, 1990). Over the period from 1957 to 1996 in Kruger National Park, which experiences summer rainfall, most of the area (80%) burnt in the months from June to November (Van Wilgen *et al.*, 2000). Unusually high rainfall in a dry season, however, can result in a greening (active growth) of grass and a consequent lower probability of fire occurrence. Some evidence for this is provided by San José and Medina (1976) where irrigation of natural grasslands in savanna during the dry season helped maintain a green biomass similar to that of the rainy season (Medina and Silva, 1990). In the Serengeti, dry-season rainfall had a weak negative influence on the extent of fire (Norton-Griffiths, 1979). Furthermore, research in Kruger National Park showed that the amount of green grass and preceding month's rainfall were two factors that had a significant effect on fire intensity (Potgieter, 1974; Trollope 1984b), which can affect the extent of burns (Williams *et al.*, 1998; Gill *et al.*, 2000).

Herbivore numbers or densities within a landscape often vary over time, while the loss of plant cover by grazing varies with animal numbers present (Pickup, 1995). The density of an unmanaged herbivore population will typically increase toward Ecological Carrying Capacity (ECC), often overshoot carrying capacity, and subsequently decline rapidly to a low density from which the cycle will start again (Caughley, 1976a, 1977; Sinclair, 1981). In this process, the biomass of herbivores and edible plants show an inverse relationship (Caughley, 1976b; Sinclair, 1981). Fire frequency and the extent of fires is expected to be higher during periods of low herbivore densities and low during periods of high herbivore densities because herbivores can impact on the amount of available herbaceous fuel (Norton-Griffiths, 1979; Bond and Van Wilgen, 1996). Furthermore, the effect of herbivores on the amount of herbaceous fuel and thus the fire regime is more pronounced as the aridity of the environment increases (O'Connor, 1985). In the Serengeti, a 79% increase in biomass of large grazing mammals was accompanied by an overall 48% increase in consumption, which matched exactly with a decrease in burning (Norton-Griffiths, 1979). In *Colophospermum mopane* woodland in the south-eastern lowveld of Zimbabwe, the decline in standing crop ($kg.ha^{-1}$) through the dry season after peak standing crop was reached took the general form of a negative exponential and was defined by the following equation (Walters, 2000):

$$\text{Standing crop (month x)} = 715 + 1203\,(0.826^{X})$$

This decrease with time since the attainment of peak standing crop was ascribed to increased dormancy of grass plants, removal by grazing ungulates and termites, flattening by animal movement and decreasing rainfall (Walters, 2000). Herbage cover depletion by grazing and natural decay can also be approximated by a simple exponential decay process (provided the time unit is not less than one month) (Pickup, 1994, 1995) in which:

$$H_t = H_{t-\Delta t} e^{-d\Delta t}$$

where d is the instantaneous cover depletion rate and Δt is the time increment involved.

2.5.3. Barriers to Fire Spread

The issue of fire barriers has, to my knowledge, received very little research attention. Fire barriers are features that interrupt the continuity of surface fuels, for example, rivers, pans or barren rocky slopes, and can inhibit the spread of fire and influence the patterns of fire occurring in a landscape (Gill *et al.*, 2000; Heyerdahl *et al.*, 2001). In the Blue Mountains, USA, lower fire frequency in the northern watersheds of the sampled area was partly attributed to the complex terrain where rivers, ridges, and barren rocky slopes interrupt the continuity of surface fuels and thus fire spread, in contrast to the southern watersheds that comprise more gentle topography that was more conducive to the spread of fires, which were ignited outside the sampled watersheds, into the area (Heyerdahl *et al.*, 2001). River systems in Kakadu National Park, Australia, have also been cited as potential barriers to fire spread (Gill *et al.*, 2000). In Hluhluwe-Umfolozi Park, only 2.8% of fires were larger than $50km^2$ while in Etosha National Park 36% of recorded fires were similarly sized (Balfour and Howison, 2001). This was attributed to the fact that the Hluhluwe-Umfolozi Park landscape is divided by numerous streams with riverine vegetation, which act as barriers to fire spread, as compared to the flatter, less divided landscape of Etosha National Park (Balfour and Howison, 2001). Similarly, natural features such as rivers and riverine vegetation in MGR are expected to have limited fire spread, while other vegetation types where the main fuel source, namely the herbaceous layer, is usually sparse as in the case of Sand Forest and clay thickets (Goodman, 1990), are also expected to have acted as fire barriers. Vegetated wetlands can also act as barriers to fire, possibly due to the relatively high moisture content of the vegetation, although in this study, some were found to have burned, albeit infrequently.

Human activities have also been responsible for a change in continuity of fuels in a landscape (Bond and Van Wilgen, 1996). Man-made features such as roads, tracks, paths, fence-lines and firebreaks, the latter of which are specifically created to prevent fire spread mainly across farm or reserve boundaries, interrupt the continuity of fuel and thus become fire barriers. The relative importance of man-made barriers versus natural barriers can be influenced by the fire management strategy employed. In MGR, block burning and point source ignition burning strategies have been employed and it is expected that the natural barriers such as rivers and riverine vegetation have played a greater role during the period when PSI strategy was employed, while the man-made/management barriers such as roads and fire breaks have been more important during the block burning period. This issue has, to my knowledge, been examined for the first time in this study.

The effectiveness of some landscape features as fire barriers can change under certain circumstances. For example, fire resistant vegetation types such as forest, which should act as a natural fire barrier, can become fire prone as a result of alien plant infestations such as *Chromolaena odorata*, which is highly flammable even when green (Bromilow, 1995). This effect has been witnessed in Hluhluwe-Umfolozi Park where fire burnt into a forest which was heavily infested with *Chromolaena odorata* (Van Rensburg, pers. comm.).

CHAPTER 3: THE STUDY AREA

3.1. LOCATION

The study area forms part of Mkuzi Game Reserve (latitude $27^0 33'$ to $27^0 48'$ south, longitude $32^0 06'$ to $32^0 26'$ east) which occurs in north-eastern KwaZulu-Natal, South Africa (Figure 3.1). The reserve forms part of the Greater St. Lucia Wetland Park (GSLWP). The study area of 23 651 ha is defined by the original proclaimed boundary of Mkuzi Game Reserve, and excludes Nxwala Estate (state land), the Controlled Hunting Area (CHA) and Lower Mkuze properties, which were added to the reserve in 1984, 1990 and 1990-1992, respectively (Figure 3.2).

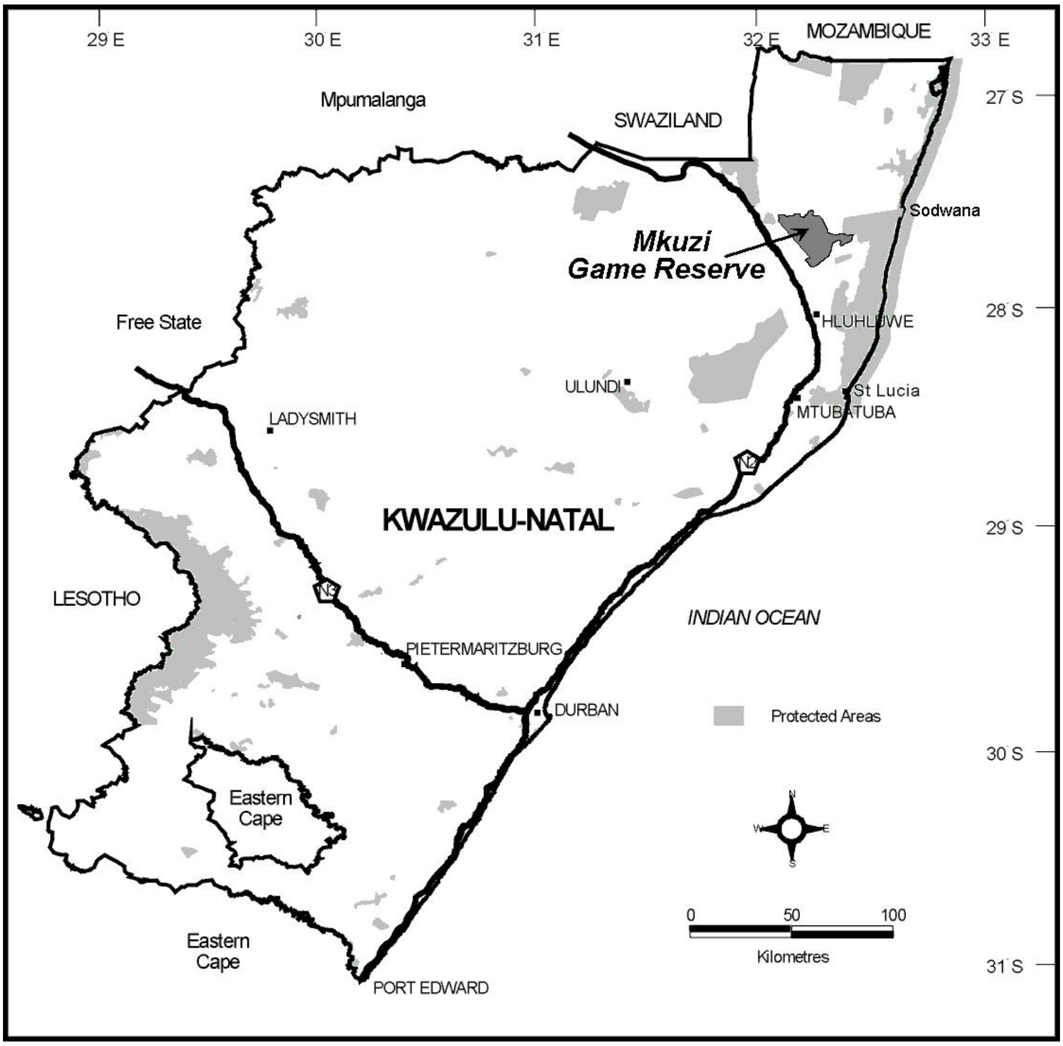


Figure 3.1: Location of Mkuzi Game Reserve in KwaZulu-Natal, South Africa. (Base map supplied by Ezemvelo KZNWildlife Cartographer, H. Snyman.)

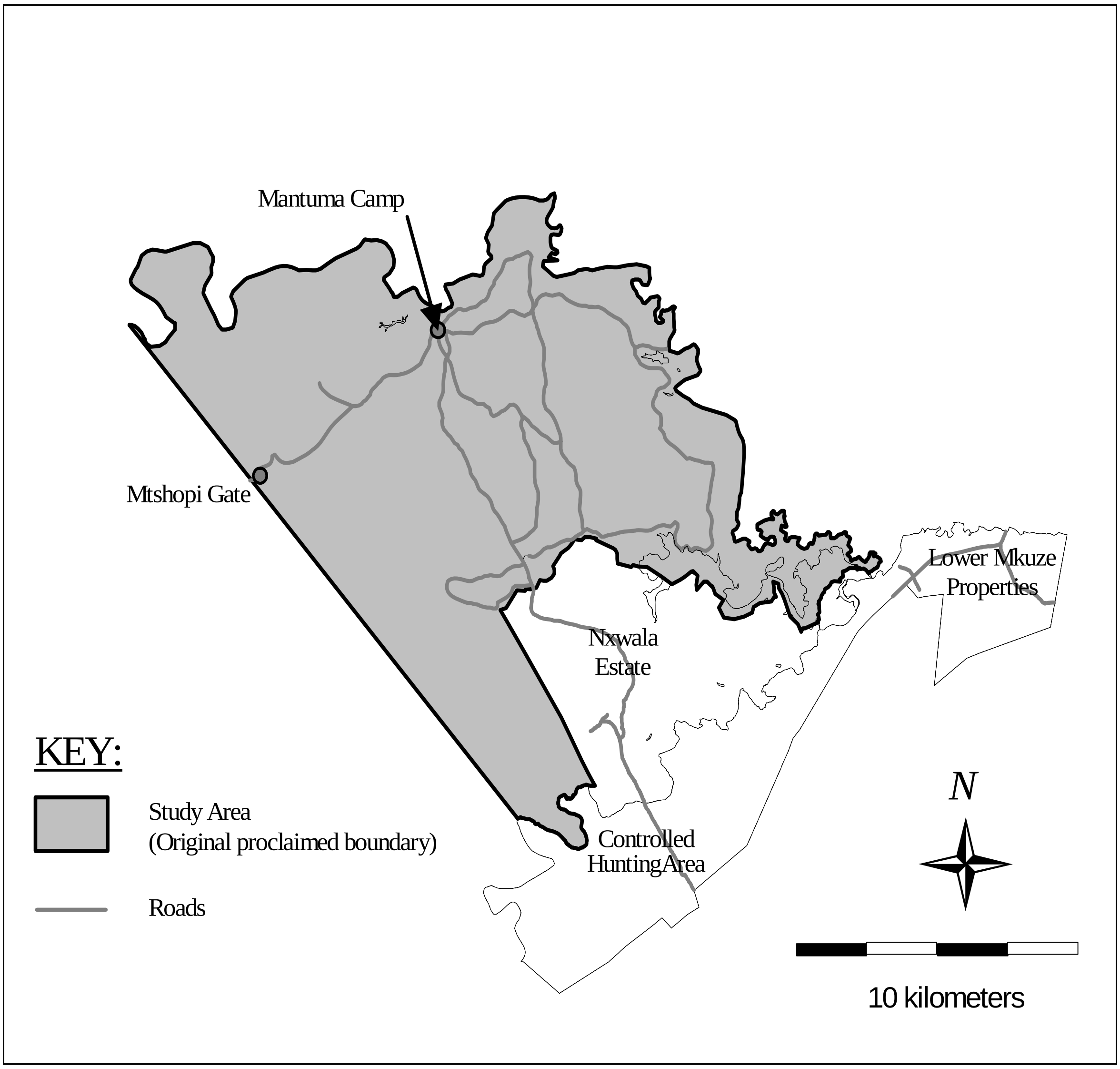


Figure 3.2: Location of the Study Area, Nxwala Estate, the Controlled Hunting Area (CHA) and the Lower Mkuze Properties in Mkuzi Game Reserve.

3.2. History of the Area

A detailed account of the history of the area is given by Goodman (1990), of which some of the more significant events and facts are summarised here. During the 19th century, the area was reported to have abundant herds of eland, kudu, buffalo, nyala, zebra, wildebeest, waterbuck, hippopotamus and impala (Drummond, 1875). By 1870 the hunting era was over due to indiscriminate shooting, habitat deterioration and increasing human population pressure. Subsequently, the remaining game populations were drastically reduced by concession hunters and farmers. Early in the 20th century, the magistrate stationed at Ubombo took a personal interest in game preservation and therefore proposed the proclamation of the reserve. As a result, an area of approximately 25 000 ha was proclaimed as Mkuzi Game Reserve in 1912. Due to continuous outbreaks of Trypanosomiasis, game eradication programmes were undertaken in 1917, when a total of 25 000 wildebeest were shot, and again over a period from 1942 to 1950, when approximately 38 500 large herbivores were killed. Extensive trapping and D.D.T. spraying campaigns, which were undertaken in the reserve and elsewhere in 1945, finally saw the elimination of tsetse fly over large areas of its former range.

In 1954, the then Natal Parks Board assumed control of the game reserve, but it was not until 1956 that people and their domestic stock were removed from the reserve. Since acquisition by the Natal Parks Board, there were two major management decisions that were of significance to the biological development of the reserve. The first was implemented between 1954 and 1958 when a permanent watering point was created at Bube Pan situated approximately in the centre of the reserve, by means of a pipe-line from Mkuze River. The second decision was implemented in 1963 when large scale animal population control programmes were initiated, after populations had been allowed to build up rapidly since becoming protected in 1954, and anxiety was expressed about deteriorating veld conditions. Since then, populations of a number of large herbivore species have been controlled through regular culling and translocations. This is discussed further in section 3.7.

In 1984, a 5 500 ha area to the south of the reserve known as Nxwala Estate, was added. This was followed in 1990 by the addition of several properties totalling approximately 4 000 ha in size, which adjoined the reserve to the south of the Msunduze River and constituted what is currently known as the Controlled Hunting Area (CHA). Between 1990 and March 1992, six properties in the Lower Mkuze area, which formed a link between Mkuzi Game Reserve and Sodwana State Forest (Ozabeni Section of GSLWP) to the east were purchased and added to the reserve making up a consolidated area of approximately 37 000 ha. In 1999, part of the GSLWP, that is the area excluding Mkuzi Game Reserve, was listed as a World Heritage Site. The Park, which comprises a total of about 325 000 ha including Mkuzi Game Reserve, was later formally proclaimed as a single legal entity under the World Heritage Convention Act (1999), Regulations 1193 of November 2000 (Draft GSLWP Integrated Management Plan, 2003).

3.3. CLIMATE

The climate of Mkuzi Game Reserve is described as warm to hot, humid sub-tropical (Schulze, 1965; Goodman, 1990). The reserve is described, according to Thornthwaite's (1948) classification, as being semi-arid with little or no moisture surplus in any season (Goodman, 1990). The prevailing winds in the area are roughly north-easterly and south-westerly, with the windiest period being during September and October (Goodman, 1981).

3.3.1 Rainfall and Evaporation

Long-term data collected from 1951 to 2001 at Mantuma weather station in the reserve show that rainfall in the area is highly seasonal, with the highest mean monthly rainfall occurring in February and the lowest mean monthly rainfall occurring in July (Figure 3.3). The mean annual rainfall (1951–2001) is 671 mm. The wet or rainy season (all months with >50 mm of precipitation (Knoch and Schulze, 1957; Goodman, 1990)) is generally from October to March, while the dry season is from April to September. Spatial variation in annual rainfall is limited, ranging from c. 800 mm in the west to c. 600 mm in the east (Goodman, 1990).

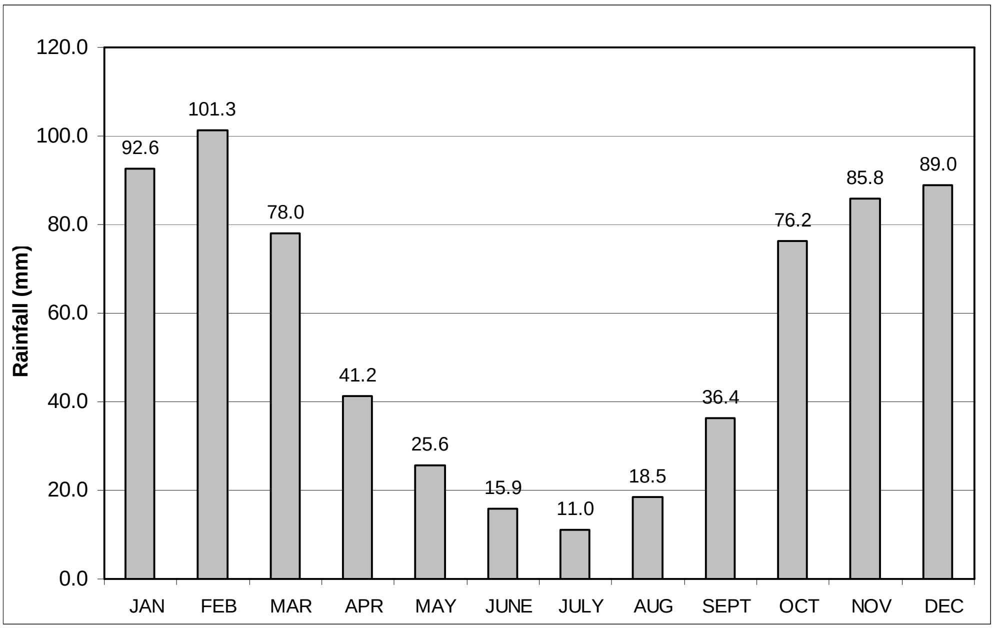


Figure 3.3: Mean monthly rainfall in Mkuzi Game Reserve as measured at Mantuma weather station for the period from 1951 to 2001.

The mean annual pan evaporation for the region is 2100 mm with a peak evaporative demand during December and January of approximately 230 mm per month (Goodman, 1990). The total annual potential evaportranspiration has been estimated to be 1 230 mm, with the peak occurring in January and the trough in June (Goodman, 1981). There is generally a moisture deficit in every month of the year, with an annual estimated deficit of 599 mm (Goodman, 1990).

3.3.2. Temperature and Relative Humidity

Temperature and relative humidity have also been determined from the Mantuma weather station data. The mean annual temperature in Mkuzi Game Reserve is 23.2 ^{0}C (Goodman, 1990). January is the hottest month during a generally hot summer, and has a mean temperature of 26.6 ^{0}C and a mean maximum temperature of 31.6 ^{0}C. Temperatures during summer can, on occasion, exceed 40 ^{0}C. The winters are relatively warm with the coldest month being July, which has a mean temperature of 18.7 ^{0}C and a mean minimum temperature of 12.7 ^{0}C.

In general, relative humidity (at midday) is highest during the summer months and lowest during the winter months. The mean humidity (1989 to 1991) at 14h00 for February was 76.6% and for June-September was 50.8% (Lewis and Goodman, 1993). For the period from 1989 to 2000, the monthly mean minimum humidity was lowest in June (29.9%) and highest in February (43.5%).

3.4. PHYSIOGRAPHY

3.4.1. Topography

Mkuzi Game Reserve is located on the western edge of the Maputaland Coastal Plain. The topography of the reserve is closely related to the geology. Topography is mainly flat or gently undulating in the eastern parts where altitude reaches a minimum of about 20 m above sea level while in the western parts, which incorporate the foothills of the southern end of the Lebombo Mountains, it is hilly and altitude reaches about 300 m above sea level. A gently undulating region underlain by Early Cretaceous sediments occurs east of the Lebombo Mountains, followed by a gently undulating but slightly elevated Quaternary dune complex, which is in turn followed by an essentially flat region underlain by Late Cretaceous sediments (Goodman, 1990). Finally, a low-lying flat alluvial floodplain of recent origin occurs adjacent to the Mkuze and Msunduze Rivers (Goodman, 1990).

3.4.2. Geology and Soils

Sixteen geological types have been recognised in Mkuzi Game Reserve and surrounding areas (Goodman, 1990). Thirteen of these geological types occur within the study area, which coincides with the original proclaimed boundary (Figure 3.4). Six of these geological types which belong to the Jurassic period (excluding the Sabi River Basalts which lie to the west of the reserve) and which are mainly volcanic in origin have been combined into a single type for mapping purposes and the purpose of this study. This study thus considers the following eight geological types: Marsh (Holocene); Young alluvium (Holocene); Old Alluvium (Pleistocene); Orange to red dune cordon sand (Pliocene); Siltstone and sandstone (Cretaceous); Glauconitic marine sandstone with shelly concretions (Cretaceous); Conglomerates, grits, sandstone and siltstone (Cretaceous) and; Rhyolite, rhyodacite, and syenite with pyroclastic material (Jurassic) (Figure 3.4). A detailed description of the geological history of the reserve has been provided by Goodman (1990).

In general, soils of the reserve correspond closely to the underlying geology and topography (Goodman, 1990). In the west of the reserve, the Lebombo Mountain volcanic rocks have given rise to lithosols of various depths. The incised valley floors of these areas are generally angular with a small area of alluvial deposition, which predominantly comprises shallow gritty clays. Early cretaceous geological deposits consisting of conglomerates, sandstone and siltstones, which have been derived from a variety of rock types, have resulted in soils of varying depths, texture and mineralogical composition. In general, soil depth and clay content increases from west to east and from upland to valley floor. Upland sites are characterised by lithic soils in the west and ferruginous soils in the east, while bottomlands are generally calcimorphic, characterised by brown to dark brown calcimorphic soils in the west and vertisols in the east. Late Cretaceous deposits occurring in the east of the reserve, consist primarily of glauconitic sandstone and have resulted in a uniformly deep soil mantle throughout the area. Calcimorphic clay to sandy clay loam soils predominate throughout this region of the reserve. Vertisols are common on this land unit and dominate in the area adjacent to Nsumu Pan, while small areas of hydromorphic gley soils are scattered throughout this region. The latter soil type is important with regard to the distribution of wet-season water sources. Quarternary sand deposits occur as two parallel dune cordons, which bisect the reserve in a roughly north-south orientation. On the older western dune, soil profiles are generally deep, red and well developed displaying advanced mineral diagenesis. On the younger, eastern dune, soil profiles are generally a poorly developed yellow to orange arenosol. Alluvial soils, by definition, occur on the geological alluvia in the reserve. In this study the alluvial deposits in the pans, reed swamps and vleis are referred to as marsh, while the old alluvium occurs in minor drainage channels originating in the Lebombo Mountains, which are no longer flooded and exhibit various amounts of soil horizon development. The young alluvium occurs mainly adjacent to the two major rivers, namely the Mkuze and the Msunduze rivers.

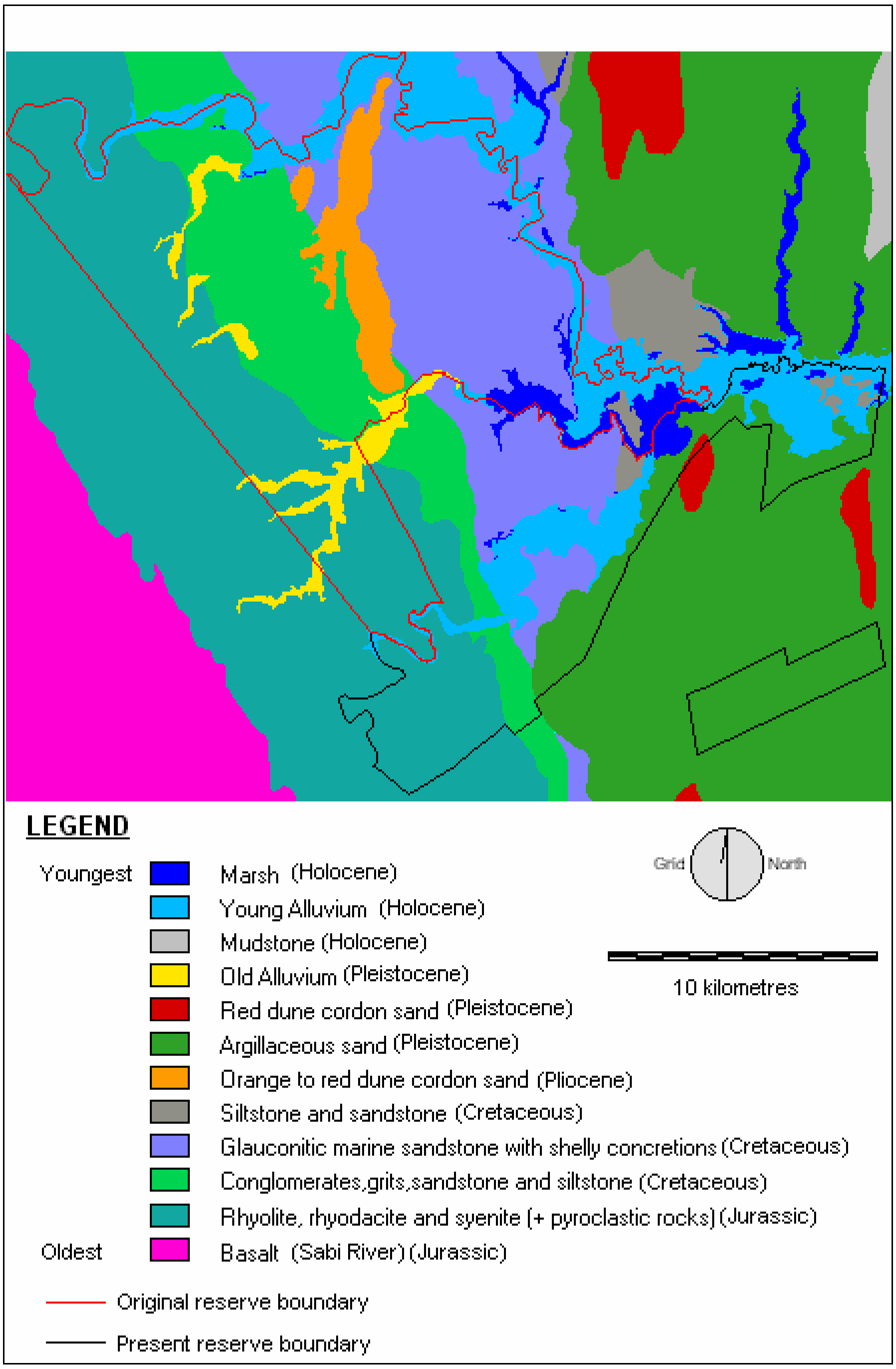


Figure 3.4: Map of geology in and surrounding Mkuzi Game Reserve. (Sources: Geological Survey, Department of Mineral and Energy Affairs. Published 1985 at 1:250 000 scale (27 ½ 32 St. Lucia) and unpublished (undated) 1: 50 000 scale geological maps, Chief Director of Surveys and Mapping, Mowbray, Cape Town. Supplemented by fieldwork (Goodman, 1990).

3.4.3. Surface Hydrology

The reserve is associated with two major rivers, namely the Mkuze River, which forms the northern and eastern boundary of the reserve, and the Msunduze River, which flows through the southern part of the reserve and forms the boundary between the Controlled Hunting Area and the rest of the reserve. Water flow in these rivers is seasonal but subsurface water is available from the bed of the Mkuze River throughout the year (Goodman, 1990). Several seasonal streams drain the eastern side of the Lebombo mountains and the reserve, flowing in an easterly direction and eventually form two major streams, namely the Nhlohlela and Nsumu streams, which drain via the Nhlohlela (c. 47 ha) and Nsumu Pans (c. 250 ha) into the Mkuze River (Goodman, 1990). A number of small (less than 0.25 ha), shallow clay floored seasonal pans occur throughout the reserve but are particularly common in the eastern half.

3.5. VEGETATION

3.5.1. General Description of Vegetation

Most of Mkuzi Game Reserves fall within Phillips (1969) bioclimatic region No.10, namely Riverine and Interior Lowland thornveld, while it contains three Acocks (1975) veld types, namely 1 – Coastal Forest and Thornveld, 10 – Lowveld and 6 – Zululand Thornveld (Goodman, 1990).

The vegetation of Mkuzi Game Reserve shows a strong correlation to geology and soils, and may be divided into nine major types, which overlap along complex environmental gradients and thus are not in all instances distinct (Goodman, 1990). They include: Dry Mountain Bushveld which is associated with lithosols on the Lebombo Mountains; Microphyllous Thorny Plains Bushveld which is associated with vertic and ferrugenous clay soils; Mixed Bushveld which occurs on soils ranging from brown, calcimorphic sandy clay loams to vertic clays and gley soils; Thicket which occurs on vertic clays and hydromorphic gley soils where topography is flat and soils poorly drained; Red Sand Bushveld which occurs on ferrugenous arenosols of Quarternary origin; Sand Forest which occurs on poorly developed yellow-orange arenosols of Quarternary origin; Riparian fringing forest and woodlands which are associated with alluvia; and Floodplain grasslands which also occur on soils of alluvial origin.

3.5.2. Physiognomic Classification of Vegetation

Goodman (1990) used the vegetation classification of Greenway (1973) to produce a physiognomic classification of vegetation in Mkuzi Game Reserve, which could be easily mapped from aerial photographs. The resultant physiognomic map was modified for this study, by combining Goodman's (1990) many detailed physiognomic classes into 7 broad classes, namely forest and closed woodland, mixed woodland, mixed bushland, mixed thicket and scrub, wooded grassland, grassland, and wetlands (Figure 3.5). A detailed description of the different physiognomic types found in the reserve is given by Goodman (1990). In addition, a summary of the relationship between these broad physiognomic types and geology in the study area is given in Appendix 1.

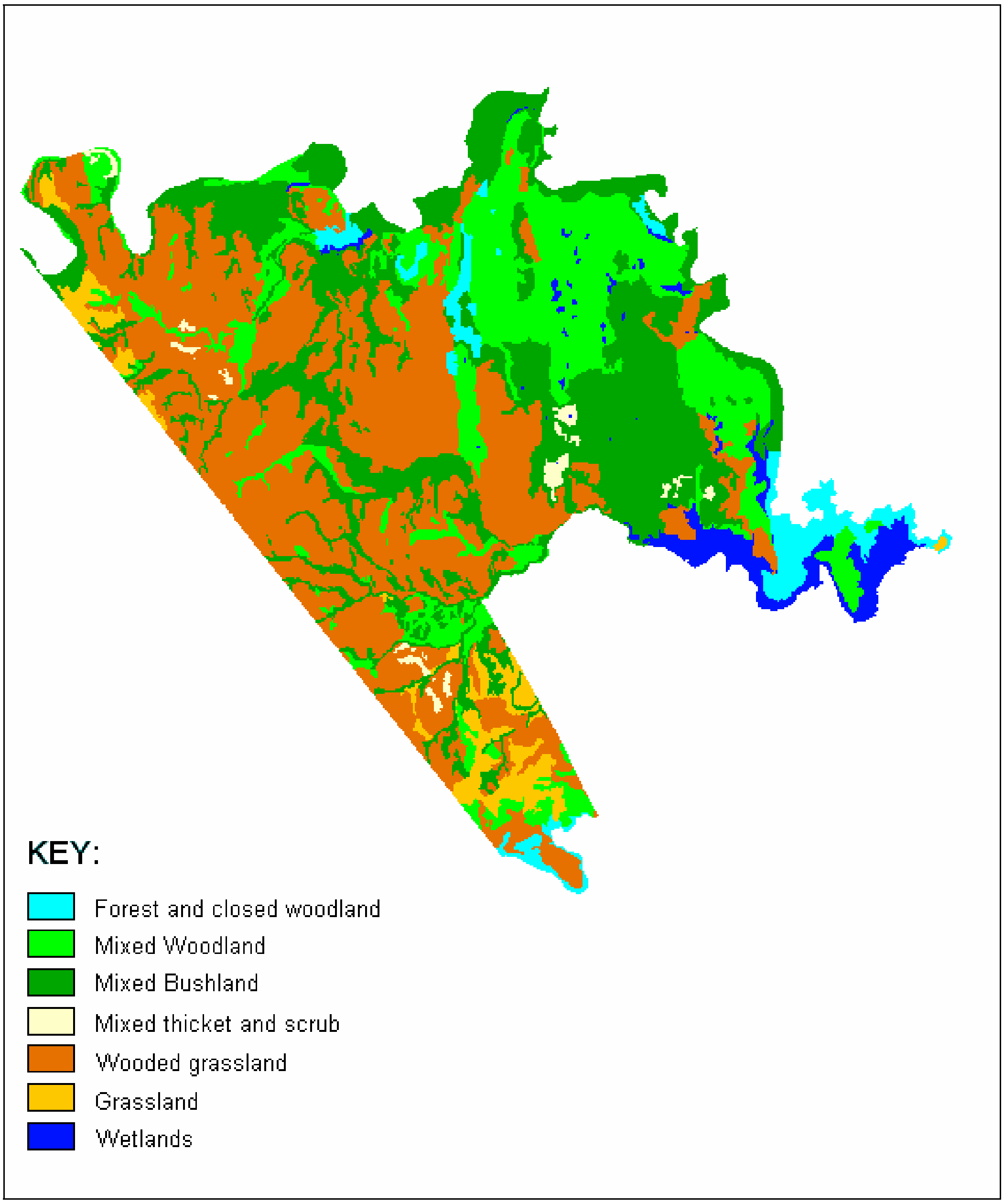


Figure 3.5: Map of the broad vegetation physiognomic classes in the study area (within the original boundary of Mkuzi Game Reserve).

3.6. FAUNA

Zoogeographically, Mkuzi Game Reserve falls into the East African Province of the Ethiopian Region (Goodman, 1990). Rautenbach *et al.* (1980) have reviewed the status of mammals in the Maputaland region while Dixon (1964) undertook such a review for the reserve. As this study is concerned mainly with the herbivores and their influence on fire (i.e. through their influence on fuel) and effect on vegetation, only the herbivores that occur in the reserve are listed here. In decreasing order of numerical importance as determined from the 1997 large herbivore line transect survey, these include impala, nyala, blue wildebeest, warthog, zebra, kudu, red duiker, suni, grey duiker, giraffe, steenbok, black rhinoceros, white rhinoceros, common reedbuck and mountain reedbuck (Table 3.1). Besides the 52 hippopotamus counted in an aerial survey in 1998, bushbuck, waterbuck, elephant and eland occur in low numbers, that is, below 50. An important large herbivore that no longer occurs in the reserve is buffalo. The greatest biomass of large herbivores consists of mixed feeders, followed by concentrate grazers, then by bulk grazers and finally by browsers.

Table 3.1: The 1997 large herbivore line transect survey results for Mkuzi Game Reserve, including the Controlled Hunting Area

Species	1997 Population Estimate
Impala	6791
Nyala	3782
Blue wildebeest	3152
Warthog	1567
Zebra	1064
Kudu	493
Red duiker	470
Suni	406
Grey duiker	370
Giraffe	163
Steenbok	157
Black rhinoceros	85
White rhinoceros	71
Common reedbuck	62
Mountain reedbuck	46

Giraffe did not naturally occur in “Zululand”, and can therefore be regarded as an alien species in the reserve, and is referred to as being introduced as opposed to being re-introduced (Goodman and Tomkinson, 1987). Similarly, white rhinoceros are considered as being introduced as there are no historical records of white rhinoceros having occurred east of the Lebombo Mountains (Goodman, pers. comm.). Eland and elephant, on the other hand, are indigenous to the area and are referred to as being re-introduced into the area (Goodman, 1990). The re-introduction of elephants has been highly successful, but the eland re-introduction appears not to have achieved the same success, most probably as a result of disease or high tick loads. The introduced species (giraffe and white rhinoceros), however, appear to be thriving.

3.7. MANAGEMENT REGIME

3.7.1. Past and Present Burning Strategy in the Reserve

Fire, as discussed in Chapter 2, is an important management tool in savannas, and thus has been used extensively in Mkuzi Game Reserve. Two strategies of burning have been applied in the reserve, namely block burning and point source ignition (PSI). The block burning strategy generally comprised late winter/spring burns, which were implemented within demarcated blocks at a frequency determined by fuel availability. It was often the aim to burn the entire block and as a result fires were often re-ignited if they burnt out prematurely or did not burn a large enough area within the block. A change in philosophy with regard to burning started around 1984/85, when there was a move away from a strict block burning strategy towards a PSI strategy (Goodman, pers. comm.). This change was in line with a shift towards a process-based management system which aimed to ensure that natural processes prevailed in the reserve over those that were obviously artificial (Mkuzi Game Reserve Management Plan, 1986; Ezemvelo KZNWildlife Policy No.1 - Mission Statement, 1997). Initially, however, blocks were still used as a guide but the PSI strategy was implemented within these blocks (Goodman, pers. comm.). The PSI system consists of igniting an area at a single point or on a very small front of 50 to 100 m, and permitting the fire to burn out against natural barriers. In effect, it is a requirement of the PSI strategy to re-ignite fires on the opposite side of man-made barriers such as roads, when fires burn out against them. It is envisaged that the PSI system should lead to a situation closely resembling the past or natural fire regime in savanna that is expected to have resulted in a fire mosaic of areas burnt by different types and intensities of fire occurring at various times and frequencies (Trollope, 1984a), all of which maintained a diversity of habitat and species. Conversely, when large areas are subject to uniform disturbance, as would be the case with a block burning strategy, spatial heterogeneity will be reduced and diversity at all scales would in all probability also be reduced (Goodman, 1990).

3.7.2. Large Herbivore Management in the Reserve

Three objectives of large herbivore management in the reserve have been stated in the Mkuzi Game Reserve Management Plan (1986; 1996). The first objective is to re-establish and maintain an indigenous animal community of genetically viable populations. Secondly, to re-instate, or where this is not possible simulate through management, those ecological processes and regulatory mechanisms that are no longer operative, for example, predation by lions. Finally, to allow the sustainable harvesting of faunal populations, where appropriate, provided that this does not conflict with the primary objective of conserving biological diversity. In addition, herbivore management in the reserve is designed to assist the attainment of a dynamic equilibrium between vegetation and large herbivores.

With regard to the first objective, a number of species have been re-introduced into the reserve with varying success. This has been briefly discussed in section 3.6. With regard to the second objective, simulated predation based on a predator/prey model has been implemented in the reserve from which annual game removals are determined. These animals are either removed live or culled, depending on the species, but such removals are always based on the principles of sustainable utilisation and maintenance of biological diversity. Furthermore, the reinstatement of ecological processes has also taken the form of the re-introduction of a key species such as elephant, which were re-introduced into the reserve in 1994.

CHAPTER 4: SPATIAL VARIATION OF THE FIRE REGIME

4.1 INTRODUCTION

Fire frequency and area burnt can vary spatially in a landscape (Norton-Griffiths, 1979; Balfour and Howison, 2001; Heyerdahl *et al*., 2001). Both the size of burns and frequency depend on the amount of available fuel (Brown and Davis, 1973; Trollope, 1984a, 1984b; Williams *et al.* 1998; Gill *et al.*, 2000), which in savanna is mainly grass (Trollope, 1984a). The amount of available herbaceous fuel is influenced by a number of factors including substrate (soils and geology), vegetation type, rainfall and herbivory.

Geology and soils often correlate closely with vegetation (Cole, 1986; Goodman, 1990), while different vegetation types may be characterized by different types and amounts of fuel (Bond and Van Wilgen, 1996). A summary of the relationship between vegetation physiognomic types and geological types in the study area is given in Appendix 1. At a landscape level, woody plant biomass in a vegetation type is usually negatively related to herbaceous plant biomass because of competition for resources (Scholes and Archer, 1997). In this study, the physiognomic classification of MGR has been used, which refers to the woody cover, but the herbaceous cover can be inferred from this because woody plant cover is negatively correlated with grass cover (Goodman, 1990; Scholes and Archer, 1997). In addition, there have been various references in the literature as to the state of the herbaceous layer in the different vegetation types in MGR. Sand forest has an extremely poorly developed herbaceous layer, riverine forest and woodland usually have a sparse herbaceous layer, open to closed woodland of mixed bushveld and red sand bushveld have a herbaceous layer which are not well developed or generally fairly sparse, and thicket has a poorly developed grass layer (Moll, 1968, 1980; Goodman, 1990). Grassland and wooded grassland, by definition, have well developed herbaceous layers.

Rainfall directly influences herbaceous production in a positive linear relationship (Sinclair, 1979; Dye and Spear, 1982; Fynn and O'Connor, 2000; O'Connor *et al.*, 2001). This influence has been recognised at various levels including at a landscape scale (Briggs and Knapp, 1995). Fire patterns are thus influenced by spatial variation in rainfall and in both the Serengeti and Hluhluwe-Umfolozi Park, the area burnt and fire frequency were positively related to spatial variation of rainfall (Norton-Griffiths, 1979; Balfour and Howison, 2001).

Herbivores have a negative impact on the amount of herbaceous material, and thus fuel, through consumption and trampling (Tainton, 1981; Heady and Child, 1994; Pickup, 1994) and their impact on primary production (Milchunas and Lauenroth, 1993; Illius and O'Connor, 1999). They ultimately have a negative impact on the occurrence and frequency of fire (Norton-Griffiths, 1979; Bond and Van Wilgen, 1996; Roques *et al.*, 2001).

Given our current understanding of the factors that influence fire patterns at a landscape scale, the following predictions were derived and examined empirically in this Chapter:

i) Fire frequency and area burnt vary spatially according to geological type, with the highest frequency occurring on substrates that support vegetation with high herbaceous cover. (A direct relationship between fire frequency and area burnt was noted and thus only the variation of fire frequency, and not of area burnt, with vegetation type was subsequently explored).

ii) Fire frequency varies spatially according to vegetation type, with the highest frequency occurring in vegetation types with highest herbaceous cover.

iii) Fire frequency is positively related to rainfall and varies spatially according to rainfall distribution.

iv) Fire frequency is negatively influenced by herbivores and varies spatially according to herbivore distribution.

4.2 METHODS

4.2.1 Collection of fire data

The fire data used in this study covered a total period of 37 years (1963 to 1999), during which a number of methods were used to map the extent of fires. The earliest fires were all mapped directly onto 1:50 000 scale maps by means of field observations. In addition to this method, later fires, namely from the mid-1990's onward, were also mapped onto 1:10 000 scale orthophotographs using amateur aerial photography (taken from a low flying fixed-wing aircraft), or mapped using Global Positioning Systems (GPS). Fire maps and orthophotographs were digitised using TOSCA (Eastman, 1998) and edited in IDRISI to produce digital fire coverages for each year. When a GPS was used, the data were downloaded from the receiver to a personal computer, where it was edited using IDRISI (Eastman, 1998) to produce a digital coverage of fire for that year.

For each fire, a range of attribute data was recorded by the reserve managers who implemented the fires. The date and cause of the fire were recorded. Fuel conditions including a subjective rating of greenness ranging from very dry to very green, an estimate of mean fuel height, an estimate of fuel density ranging from very sparse (20-30% cover) to very dense (95-100% cover) and fuel uniformity ranging from uniformly spread to very patchy, were also recorded. At the same time, environmental conditions such as general weather conditions (e.g. hot and dry) and wind conditions such as speed and direction were recorded. Finally, the results of the fires were rated to have been intense, clean, patchy or very patchy. Unfortunately, due to various reasons, it was not always possible to collect all these data, and therefore attribute records were not always complete. These attribute data, where available, were then captured and linked to the spatial data using CARTALINX (Hagan and Eastman, 1998).

4.2.2 Spatial data analyses

A raster, as opposed to a vector, GIS data model was used for the analysis of all spatial data because the type of spatial analyses undertaken, for example overlaying fire maps and regression analyses between two data layers, were well suited to a raster model. The raster-based IDRISI software was therefore chosen for use. A raster model divides the entire study area into a regular grid of cells in specific sequence, while a vector model uses discrete line segments or points to identify locations and discrete objects are formed by connecting line segments (Goodchild and Kemp, 1990). Vector coverages (layers) of fire for each year were rasterised to form Boolean images (contain only zeros and ones, with ones representing occurrence of a condition) with a 25 m x 25 m grid-cell size. This resolution was chosen since most of the data used was from 1:50 000 maps where the width of a pencil line (about 0.5 mm), drawn during mapping of fires, would generally have represented 25 m on the ground. This also applies to any other linear feature on a 1:50 000 map such as roads, tracks and streams, which are represented by a single line of about 0.5 mm thick. Another reason for using this resolution was that several other layers of raster data, for example geology, already existed at this resolution.

All statistical analyses were either performed using IDRISI, where images were analysed, or SIGMASTAT (Jandel Scientific Software, 1994) where attribute or other data were analysed. Occasionally, queries of data layers were undertaken in IDRISI to assist in interpreting observed fire patterns. This refers to the process of interrogating an image using the cursor inquiry mode and clicking on individual pixels to reveal the value, or often fire frequency in this study.

4.2.3 Creation and analysis of the fire frequency map

The 37 Boolean raster layers representing the extent of fires for each year were added together using the OVERLAY module in IDRISI. In determining the extent of each fire frequency class in the study area, the Nxwala area, CHA and the Lower link properties were first "masked out" of the image using the OVERLAY module, and the resultant image was analysed using the AREA module. These areas of land were excluded from the study (and thus all spatial analyses) because no data existed for the period prior to their inclusion in the reserve (see Chapter 3). The fire frequency image was then analysed using the HISTO module, to produce a frequency histogram of cell values that represented the total number of fires in 37 years, and to determine the mean and mode fire frequency for the entire study area. The median fire frequency was determined after the cell values for this image were exported to a separate tabular database.

4.2.4 Fire frequency and area burnt in relation to geology

A rasterized map of the reserve's geology was used to create Boolean (mask) layers for each geological type. Using the OVERLAY module, each mask layer of geology was multiplied by the fire frequency layer, to create images of fire frequency for the area defined by each geological type. Each resultant image was then analysed using the HISTO module, to produce a frequency histogram of cell values that represented the total number of fires in 37 years, and to determine the mean and mode fire frequency per geological type. The median was determined after the cell values for the above images were exported to a separate tabular database. In general, the data distributions were not symmetrical around the mean and therefore the median frequency was reported, except for bimodal distributions where the major and minor modes were reported. Other statistics were included in Appendix 2.

In determining a measure of the extent of burns (area burnt) per geological type (the area defined by a particular geological type), the average proportion of each geological type burnt per annum, was estimated. For each geological type, this was achieved by initially calculating the total accumulated area burnt over the study period (i.e. the sum of frequency x area of each of the fire frequency categories). To illustrate this using a hypothetical example, if the area defining geological type A comprised 10 ha with a fire frequency of 2, 20 ha with a fire frequency of 3 and 30 ha with a fire frequency of 4, then the total accumulated area burnt would be: (10 ha x 2) + (20 ha x 3) + (30 ha x 4) = 200 ha. The total accumulated area burnt (in each area defined by a geological type) was then averaged over the 37-year period (to give the average area burnt per annum) and finally divided by the area of the respective geological types (to represent the result as a proportion of the area defined by a particular geological type). This type of analysis was not repeated for the vegetation classes as the area results closely resembled the frequency results.

4.2.5 Fire frequency in relation to vegetation type

A map of vegetation with physiognomic classes (Goodman, 1990) was used for this analysis. The map was digitised by KZNWildlife staff who produced a vector coverage. Prior to this study, part of this spatial database was rasterised and an attribute database for this coverage was created and populated (Woods, 1997). This image, however, did not cover the entire study area of this work, and therefore both the spatial and attribute databases were updated for this study. The procedure used in this analysis was the same as that used in determining fire frequency in relation to geology. Mask layers for each broad vegetation physiognomic type (Figure 3.5) were used, together with the fire frequency image, to determine the mean and mode fire frequency per broad vegetation physiognomic type, while the median fire frequency was determined after the cell values for the respective images were exported to a separate tabular database. However, the mixed thicket and scrub physiognomic type was combined with the mixed bushland physiognomic type for a single analysis, because the former type was very small (about 220 ha in extent) and was similar to the latter type with regard to percentage shrub cover. In general, the data distributions were not symmetrical around the mean and therefore the median frequency was reported, except for bimodal distributions where the major and minor modes were reported. Other statistics were included in Appendix 2.

4.2.6 Relationship between rainfall distribution and fire frequency

Initially, a regression analysis of the mean wet-season rainfall versus fire frequency at each of the rain gauges, which fell within the study area, was undertaken. The mean wet-season rainfall was determined by summarising the long term rainfall data collected at each rain gauge while the fire frequency data was determined by querying the fire frequency data layer (Figure 4.1a). The value of fire frequency was taken from the pixel on which the rain gauge point fell, except for the Mantuma gauge, which occurs in a large camp where fire is not used in the same way as is used in managing the rest of the reserve. The fire frequency for an area immediately outside the camp, within 1km from the gauge, was used instead. The rainfall data for the Mtshopi gauge appeared to be anomalous and therefore regression analyses were undertaken with this data included as well as excluded.

In another analysis, a mean wet-season rainfall image was generated in a similar manner described by Woods (1997), but differed slightly in that a linear TREND interpolation was done as opposed to a quadratic fit used by Woods (1997). Nine rain gauges in the reserve and 1 from Makasa Nature Reserve, which is about 5km from Mkuzi Game Reserve, were used in the interpolation. A pixel-by-pixel (regression-type) analysis of the relationship between the rainfall image and the fire frequency image, over the entire study area, was then undertaken in IDRISI. A second pixel-by-pixel analysis between mean wet-season rainfall and fire frequency, where fire-resistant vegetation such as forest and closed woodland, thicket and scrub, and wetland areas were excluded, was undertaken in order to provide a clearer view of the relationship and explain the occurrence of low fire frequency in high rainfall areas. These physiognomic types were considered to be fire-resistant because they either had poorly developed herbaceous layers (Moll, 1968; Goodman, 1990), or were seasonally or permanently water-logged areas as in the case of wetlands. In addition, previous analyses in this study also showed that these vegetation types experience low frequencies of fire.

4.2.7 Relationship between herbivore biomass and fire frequency

Herbivore density images (supplied by A. Whitley, Department of Water Affairs and Forestry, Private Bag X24, Howick, 3290) were used in conjunction with the fire frequency image determined in this study, to determine the relationship between these two variables. The herbivore density images had a 100m resolution (i.e. one grid-cell equals one hectare) and were created by using the annual large herbivore line transect survey data from the reserve to plot the distribution of herbivores in relation to the line transects. Through a number of steps in IDRISI, images of overall herbivore biomass, browser biomass and grazer biomass, which took into account the proportional allocation of graze-to-browse for each of the most abundant species in the reserve, were created. Estimates of the proportion of graze and browse per species were determined from three sources (Goodman, 1990; Skinner and Smithers, 1990; Bothma, 2002). A detailed description of the creation of the herbivore images is given as follows.

Once the distribution of each of the most abundant herbivore species was plotted in relation to the line transects, the number of animals occurring in each grid cell was converted to the animal unit (AU) equivalent (Tainton, 1981). Images representing the biomass in AU’s for each of the most abundant species in the reserve were produced in this way. An image representing the total biomass as distributed over the reserve was determined by combining the images of each species determined in the previous step. Browser and grazer images were created from the images of each species, by weighting them according to each species’ graze-to-browse ratio (Goodman, 1990; Skinner and Smithers, 1990; Bothma, 2002), and subsequently combining all the resultant grazer images together and all the resultant browser images together. The values in the images supplied by A. Whitley had been multiplied by 100 in order to make them as integers. The average biomass values shown in the results should therefore be divided by 100 and thus AU x 10^{-2} was used for the biomass units in the graphs of the results.

Three regression analyses were undertaken, using the REGRESS module in IDRISI. For these analyses, however, it was initially necessary to reduce the resolution of the fire frequency image from a grid-cell size of 25m by 25m to that of the herbivore biomass images (1 ha grid-cell). Initially, an attempt was made to determine the relationship between grazer biomass and fire frequency, and secondly between overall biomass (grazers and browsers combined) and fire frequency. These analyses included all vegetation types. No attempt was made to determine a relationship between browser biomass and fire frequency as it was felt that the browser influence on fire was not direct, as in the case of grazers, and that the analysis of the influence of overall biomass already took into account the effect of browsers. Because it was felt that the relationship between grazer biomass and fire frequency would be best shown in vegetation types with a well developed herbaceous plant layer, and where grazing and fire were likely to be among the more prevalent processes in operation, a third attempt to reveal the relationship between grazer biomass and fire frequency only included the wooded grassland and grassland areas.

An envelope or "factor ceiling" analysis (Blackburn *et al.*, 1992; Thompson *et al.*, 1996) was also undertaken to highlight further the relationship between grazer biomass and fire frequency. In brief, a regression was undertaken between the grazer biomass values and only their corresponding maximum fire frequency values, that is, a regression line was fitted through the upper edge of the point cloud on a scattergram of fire frequency versus grazer biomass (all areas included). This method of analysis is useful in highlighting relationships between two variables where one plays a limiting role, as in the case of herbivores, which can limit fire occurrence, while there are other independent variables (e.g. rainfall) operating which can affect the dependent variable (fire frequency). The data clouds in such cases are typically triangular in shape and have also been termed "triangular distributions" (Maller, 1990; Thompson *et al.*, 1996). Points below the upper edge of the cloud represent the influence of the other factors and may result in a weak correlation between the two variables concerned (Thompson *et al.*, 1996). Even with no correlation (conventional correlation analysis), the upper edge of a point cloud may still provide evidence of an ecologically important connection between the variables, and therefore a "factor ceiling" analysis (Blackburn *et al.*, 1992; Thompson *et al.*, 1996) was appropriate in this case.

4.3 RESULTS

4.3.1 Fire frequency map

The frequency of fire within the study area ranged from a minimum of zero fires in 37 years to a maximum of 21 fires in the same period (Table 4.1; Figure 4.1a, 4.1b). The median fire frequency was four fires in 37 years (one fire in 9.3 years). The frequency class covering the greatest area (3118.3 ha) was that of one fire in 37 years, while the frequency class covering the smallest area (0.3 ha) was that of 21 fires in 37 years (one fire in 1.8 years). Table 4.1 also shows that approximately 75% of the study area had eight or fewer fires in 37 years (one fire in approximately five years or fewer fires), 24.6% of the study area had between nine and 17 fires in 37 years (one fire in 4.1 years to one fire in 2.2 years) and less than 1% had 18 to 21 fires in 37 years (one fire in 2.1 years or more fires).

Table 4.1: Sizes of areas, and their proportion of the study area, for each frequency of fire in Mkuzi Game Reserve between 1963 - 1999

Frequency	Size (Hectares)	Proportion of area (%)
0	2313.5	9.782
1	3118.3	13.185
2	2763.3	11.683
3	2360.9	9.982
4	1884.5	7.968
5	1685.6	7.127
6	1241.1	5.248
7	1317.4	5.570
8	1065.5	4.505
9	1050.3	4.441
10	870.6	3.681
11	794.0	3.357
12	713.5	3.017
13	736.8	3.115
14	642.1	2.715
15	485.0	2.051
16	351.8	1.487
17	183.9	0.777
18	37.6	0.159
19	25.8	0.109
20	9.4	0.040
21	0.3	0.001
Total:	23651.2 ha	100.000 %

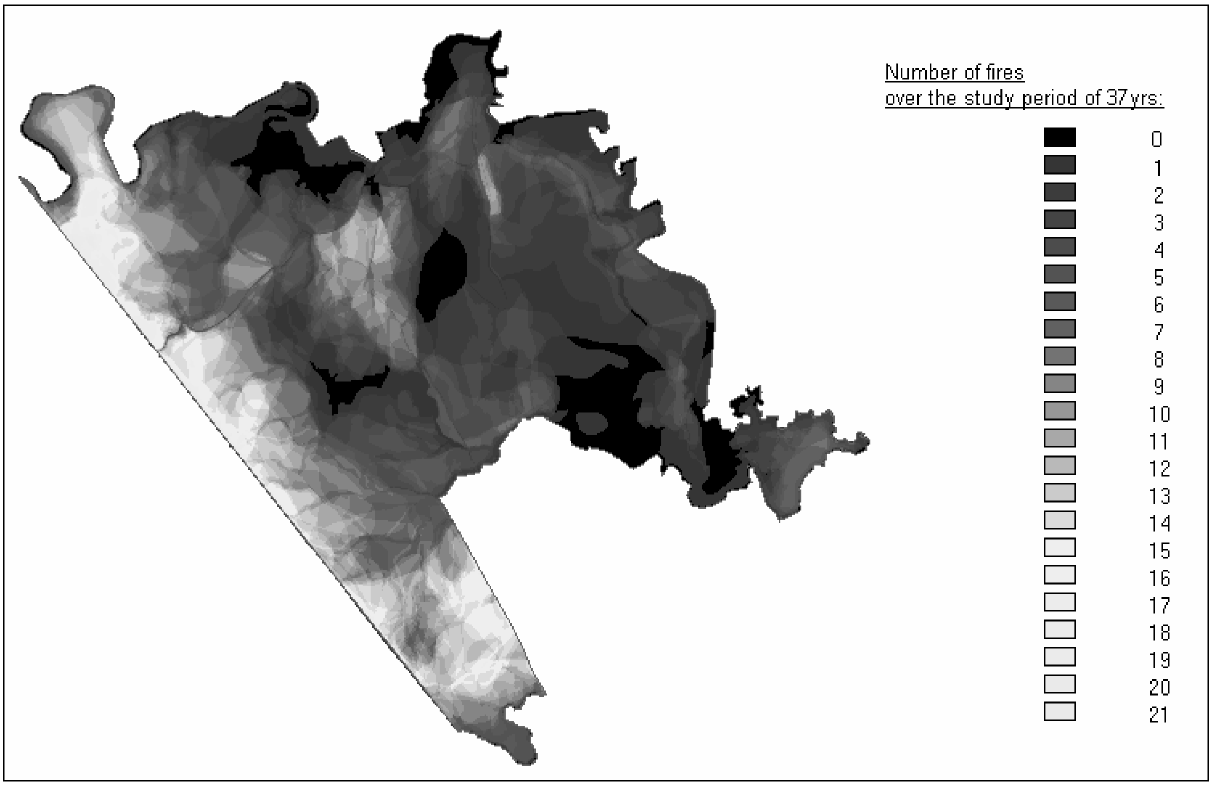


Figure 4.1a: Frequency of fire in Mkuzi Game Reserve over the period from 1963 to 1999.

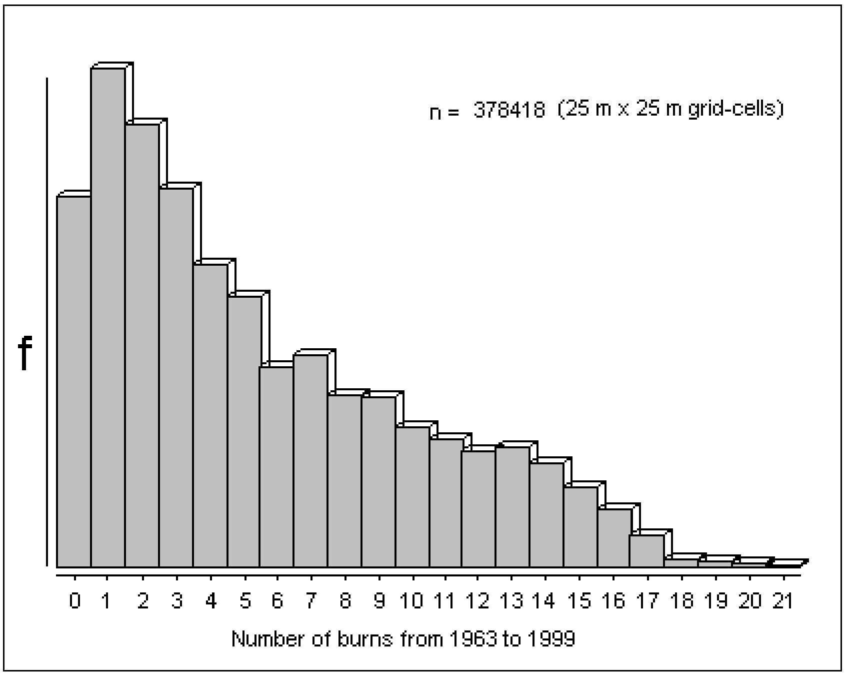


Figure 4.1b: Frequency of fire in Mkuzi Game Reserve over the period from 1963 to 1999 (histogram representation of Figure 4.1a). Summary statistics are as follows: Minimum = 0; Maximum = 21; Mean = 5.4; Mode = 1; Median = 4; Standard deviation = 4.5; d.f. = 378417.

4.3.2 Fire frequency and area burnt in relation to geology

Fire frequency and area burnt depended on geology, with the highest fire frequency occurring on substrates that supported vegetation with high herbaceous cover, while the hydrological character of the substrate also had an effect. The median fire frequency has been reported here, except where mode was appropriate, while a summary of all the relevant statistics is given in Appendix 2.

In Figure 4.2a, a bimodal pattern of fire frequency for the marsh lithology was identified. The first peak, where zero to one fires (mode of 1) have occurred in the study period of 37 years, represents the pans of Nsumu, eDisa and Nhlohlela, which are temporarily to permanently inundated with water. The second peak, where between 6 and 8 fires (mode of 7) have occurred in 37 years (one fire in 5.3 years), represents the swampy area immediately downstream from the open water of Nsumu pan. These fires have generally been what management has referred to as arson or accidental fires, with the exception of the 1976 burn, which was a controlled (prescribed) burn. This result shows that these swampy areas have during dry periods been dry enough to carry a fire.

As expected, the number of fires in the areas with the young alluvium substrate was very low (Figure 4.2b; Table 4.2). The median number of fires for this geological type is one fire in 37 years. Young alluvium is associated with riverine forest and woodland vegetation of the two major rivers, namely the Mkuze and Msunduzi rivers, where the herbaceous layer is usually sparse (Goodman, 1990). A large proportion (53%) of the young alluvium geological type is also associated with mixed bushland vegetation type, which has a high shrub cover and consequently low herbaceous cover (Goodman, 1990; Table 4.2, 4.3). It is probably this lack of herbaceous fuel that prevents the occurrence of fires in these areas.

In the areas with an old alluvium substrate, the median frequency of fire was six in 37 years (one fire every 6.2 years) (Figure 4.2c; Table 4.2). The vegetation occurring on this substrate is mainly riparian woodland and is referred to as streambed and drainage line woodland by Goodman (1990). Mixed woodland occurs on 37% of this substrate type and has a relatively high tree cover, while mixed bushland occurs on 30% of this substrate and has a relatively high shrub cover, and consequently a large proportion of the area has low herbaceous cover (Goodman, 1990; Table 4.2, 4.3). However, fire prone wooded grasslands covers about 27% of the area, thereby contributing to the relatively high (second highest for geological types) median fire frequency. From a query of the fire frequency data layer for this geological type as well as a regression analysis between the fire frequency and an altitude data layer for the same area, it was clear that higher fire frequency occurred at high altitudes, that is, in the Lebombo Mountains, while lower fire frequency occurred in the lower reaches of these drainage lines (R^2=0.42, d.f.=15245, P<0.0001, Fire frequency=(0.0515 * Altitude)-2.25). This is probably due to the lower reaches of these drainage lines being relatively moist as a result of water draining from the upper reaches. The lower reaches, being moister are thus able to support more woodland and bushland vegetation (Figure 4.3), which are relatively fire resistant. In the upper reaches of these drainage lines, however, the vegetation is characterised by a thin strip of wooded vegetation surrounded by fire-prone grasslands and wooded grasslands.

Table 4.2: Fire frequency for areas defined by the different geological types, together with the main vegetation types occurring on each geological type

Geological Type	**Area (ha)**	**% of Study Area**	**Median Number of Fires in 37 yrs.**	**Median Yrs. Between Fires**	**Main Vegetation Types with % of geological type in parentheses (See Appendix 1 for details and Table 4.3 for vegetation descriptions)**
Marsh	773	3	1*	37	Wetlands (84%)
Young Alluvium	2082	9	1	37	Mixed Bushland (53%) Forest and Closed Woodland (29%)
Old Alluvium	954	4	6	6.2	Mixed Woodland (37%) Mixed Bushland (30%) Wooded Grassland (27%)
Orange to Red Dune Cordon Sand	1679	7	3	12.3	Wooded Grassland (46%) Mixed Woodland (31%) Forest and Closed Woodland (12%) Mixed Bushland (11%)
Siltstone & Sandstone	113	<1	3	12.3	Mixed Woodland (90%)
Glauconitic Marine Sandstone	5900	25	2	18.5	Mixed Bushland (43%) Mixed Woodland (42%) Wooded Grassland (11%)
Conglomerate, Grit, Sandstone etc.	4659	20	4	9.3	Wooded Grassland (56%) Mixed Bushland (35%)
Rhyolite, Rhyodacite & Syenite	7492	32	10	3.7	Wooded Grassland (63%) Mixed Bushland (14%) Grassland (11%)
Total Study Area	23651		4	9.3	

* This statistic is not preferred as the distribution was bimodal, with a major mode = 1 and minor mode = 7.

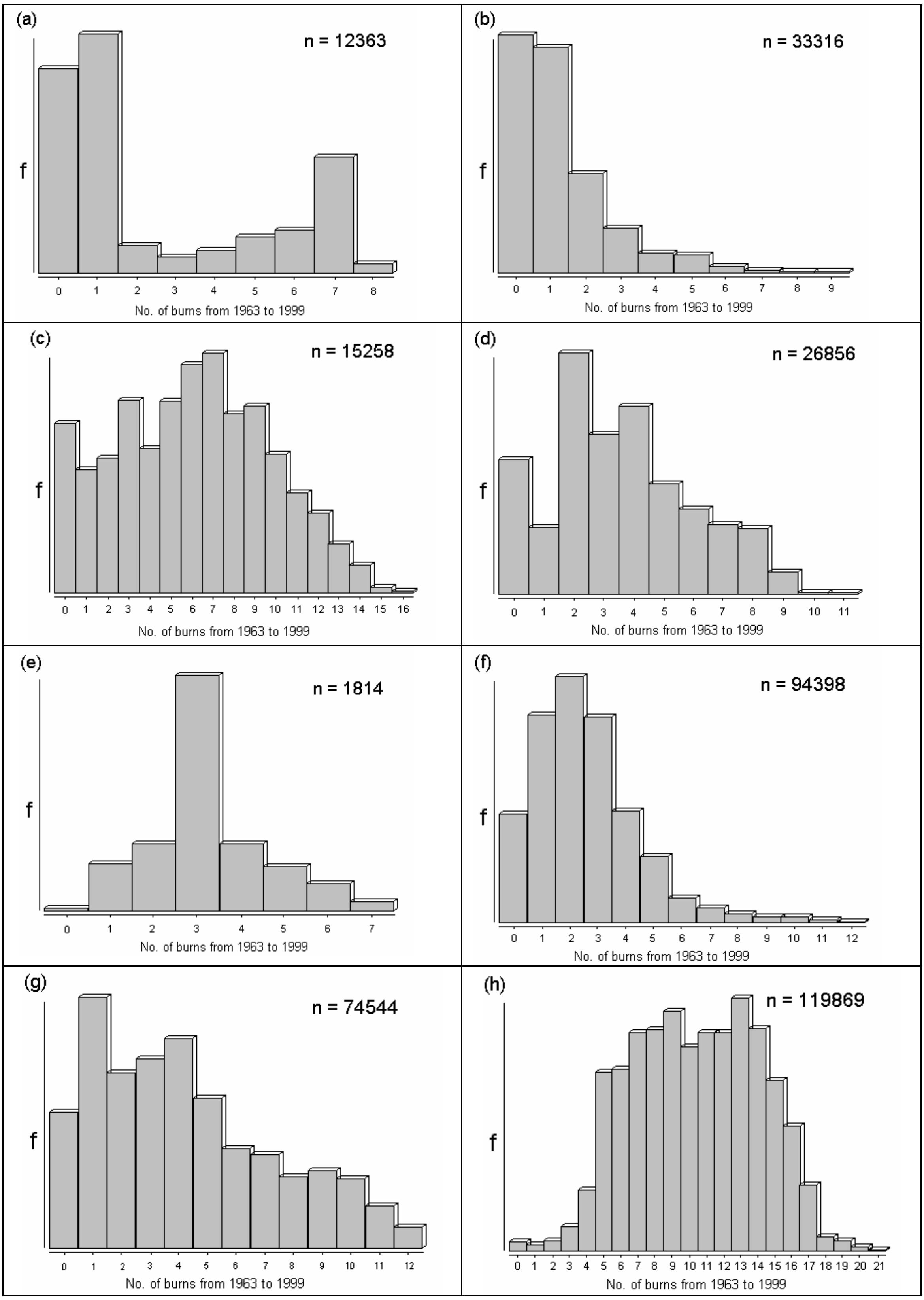


Figure 4.2: Fire frequency for areas of the different geological types, namely (a) marsh, (b) young alluvium, (c) old alluvium, (d) orange to red dune cordon sand, (e) siltstone and sandstone, (f) glauconitic marine sandstone, (g) conglomerate, grit, sandstone and siltstone, and (h) rhyolite, rhyodacite and syenite. The sample size (n) refers to the number of 25 m x 25 m pixels. Refer to Appendix 2 for summary statistics.

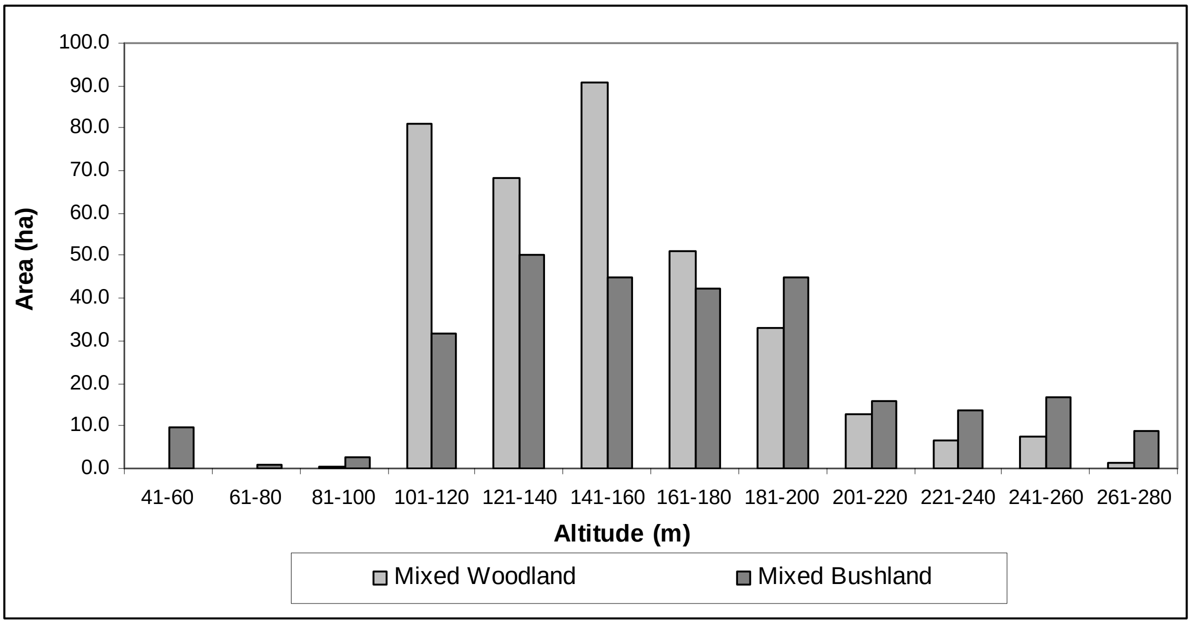


Figure 4.3: Altitudinal distribution of mixed woodland and mixed bushland in the area with a substrate of old alluvium. Note that about 100m represents the bottom end (with high woody cover) of relatively steep sloping land, while below this, the area is gently undulating.

The median frequency of fire for orange to red dune cordon sand substrate was three in 37 years (one fire every 12.3 years) (Figure 4.2d; Table 4.2). These Quaternary arenosols support vegetation described by Moll (1980) as red sand bushveld and sand forest. The herbaceous layer is generally fairly sparse in red sand bushveld and extremely poorly developed in sand forest (Goodman, 1990). This lack of herbaceous fuel is probably the main reason for such a low fire frequency in these areas. In terms of the proportion of vegetation physiognomic types (Goodman, 1990) occurring on this substrate, more than 50% consists of forest, woodland and bushland, which have relatively high woody cover and consequently low herbaceous cover (Goodman, 1990; Table 4.2, 4.3), explaining the lack of fire in the area. The peak at zero fires represents mainly the sand forest where there have been no fires during the 37 year study period. However, an anomaly appears to exist because a relatively high proportion of wooded grassland (46%) occurs on this substrate, and thus frequency should possibly have been higher (Table 4.2, 4.3). Indeed, it is likely that the higher end of the frequency range (Figure 4.2d) represents wooded grassland.

The area with a siltstone and sandstone substrate covers only 0.5% of the study area. It occurs in the extreme eastern part of the study area directly adjacent to the lower reaches of Nsumu Pan on the Mkuze River floodplain (Figure 3.4). The area is very moist and it is therefore understandable that a low fire frequency would occur. Furthermore, much of the area is vegetated by riverine (mixed) woodland (Table 4.2, 4.3), which is not prone to fire because of a sparse herbaceous fuel layer (Goodman, 1990). The median frequency of fire for this substrate is three over the 37-year period, that is, one fire every 12.3 years (Figure 4.2e; Table 4.2).

The greater part of the glauconitic marine sandstone geological type is vegetated mainly by mixed woodland and mixed bushland with less extensive patches of wooded grassland and mixed thicket and scrub (Table 4.2, 4.3). Woodland, bushland and thicket as defined by Goodman (1990) have a high woody vegetation cover, and because there is a negative correlation with grass cover, were not expected to burn frequently. Also scattered throughout this region are hydromorphic gley soils, which give rise to a number of ephemeral pans. The median frequency of fire for this geological type was two fires in 37 years (one fire in 18.5 years) (Figure 4.2f; Table 4.2). Figure 4.2f, however, does also show that there are areas within this geological type that have experienced fires up to 12 times in 37 years, that is up to one fire in approximately three years. These areas of higher fire frequency represent the wooded grasslands and parts of the mixed woodland occurring on the geological type.

The conglomerate, grit, sandstone and siltstone geological type, which has also been referred to as the Lower Cretaceous sedimentary deposits, supports a range of vegetation types including wooded grassland (56%), mixed bushland (35%) and mixed woodland (8%) (Table 4.2, 4.3). Compared to the Upper Cretaceous deposits, which occur in the eastern part of the reserve, there is a much greater proportion of wooded grassland. The median frequency of fire was therefore slightly higher for the Lower Cretaceous areas, and was estimated to be four fires in 37 years, that is, one fire in 9.3 years (Figure 4.2g; Table 4.2). As with the previous geological type, some areas had fire frequencies of up to one fire in approximately three years (Figure 4.2g).

The median frequency of fire in areas with a volcanically derived substrate of Jurassic age namely, the areas comprising rhyolite, rhyodacite and syenite (with pyroclastic rocks), was 10 fires in the 37 years, or one fire in 3.7 years (Figure 4.2h; Table 4.2). This area therefore had the greatest frequency of fire. There are two main factors that are likely the cause of this. The first and most significant reason is that much of the vegetation in the area (74%) is grassland and wooded grassland (Table 4.2, 4.3), which provide ample fuel to carry fire.

Secondly, the area is located on the boundary of the reserve and is also subject to non-prescribed fires that are started by neighbouring subsistence farmers.

The average proportion of each geological type affected by fire each year for the 37-year period is shown in Figure 4.4. This measure was found to be directly related to the estimated median frequency of fire for each geological type (Figure 4.5). For example, with young alluvium the median fire frequency was 1 fire in 37 years, which equates to approximately 0.03 fires per annum, while the average proportion of this geological type affected by fire each year also equates to 0.03 (Figure 4.4). Patterns found in this analysis were therefore the same as those found in the fire frequency analysis.

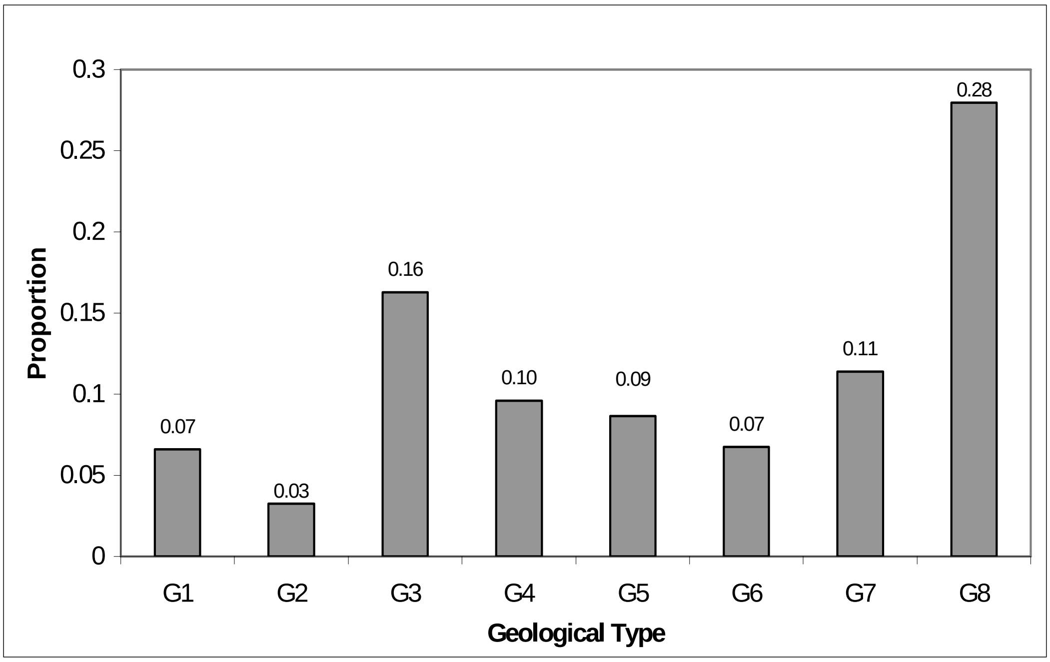


Figure 4.4: Average proportion of each geological type affected by fire each year for the 37-year period. (G1=Marsh, G2=Young alluvium, G3=Old alluvium, G4=Orange to red dune cordon sand, G5=Siltstone & sandstone, G6=Glauconitic marine sandstone, G7=Conglomerates, grits, sandstone & siltstone, and G8=Rhyolite, rhyodacite and syenite.)

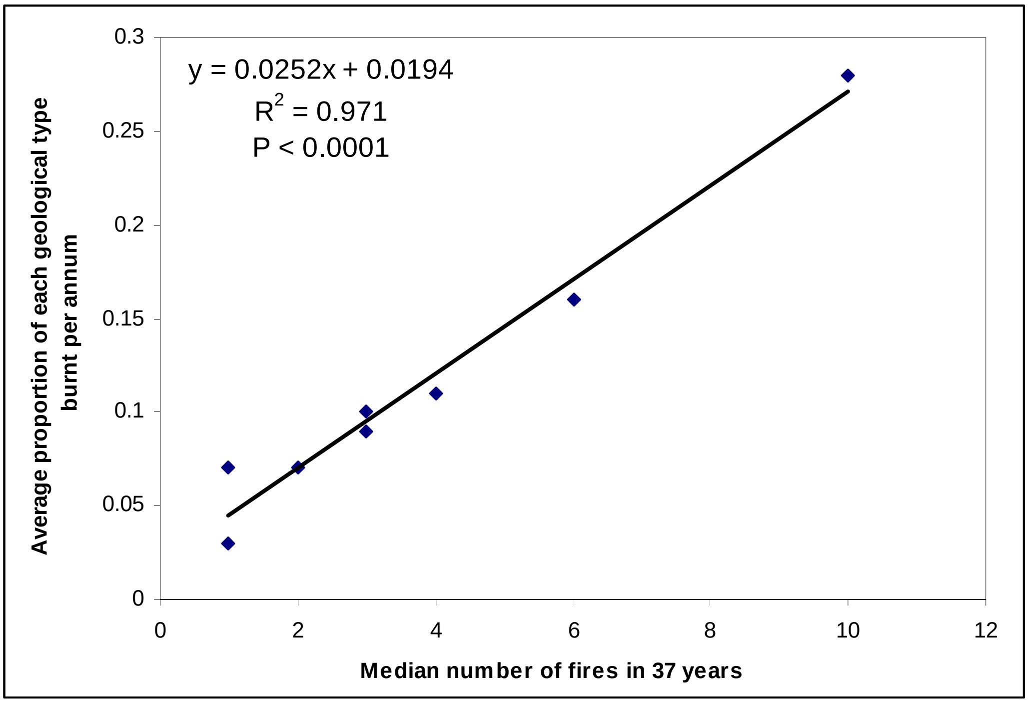


Figure 4.5: The relationship between median fire frequency and average proportion of each geological type burnt per annum.

4.3.3 Fire frequency in relation to vegetation type

Fire frequency depended on vegetation type, specifically on the amount of herbaceous material available as fuel, which is negatively correlated with woody biomass (Goodman, 1990). The physiognomic types with high woody cover and low herbaceous cover such as forest and closed woodland had very low fire frequency while those with low woody cover and high herbaceous cover such as grassland had very high fire frequency.

The median fire frequency for the forest and closed woodland physiognomic type was one fire in 37 years (Figure 4.6a; Table 4.3). Such a low fire frequency was not unexpected because this physiognomic type usually has a sparse herbaceous layer consisting of a wide variety of forbs and creeping grasses (Goodman, 1990). The largest proportion of this physiognomic type had zero fires (Figure 4.6a). By a query of the fire frequency data layer for the forest and closed woodland physiognomic type, it was apparent that the small proportion of this physiognomic type that shows higher fire frequencies mainly represents the ecotone between this physiognomic type and adjacent more fire prone physiognomic types such as wooded grassland and, to a lesser extent, mixed woodland.

The median frequency of fire for the mixed woodland physiognomic type was three fires in 37 years (one fire in 12.3 years) (Figure 4.6b; Table 4.3). This physiognomic type is defined by having between 50 and 75% tree cover and less than 35% shrub cover (Goodman, 1990). Such a density of trees would not prevent grasses from establishing as much as may be the case with bushlands and thickets (Table 4.3). However, the herbaceous layer in open to closed woodland, is not well developed or generally fairly sparse (Goodman, 1990). Fire would thus occur in these areas, although not very frequently in parts. Some smaller areas within this physiognomic type have, however, experienced fire as often as 17 times in 37 years (one fire in approximately two years) (Figure 4.6b). It was apparent, from querying the fire frequency data layer for this physiognomic type, that these areas of high frequency were mainly in the western part of the reserve where the neighboring vegetation is mainly the fire-prone wooded grasslands. The frequent occurrence of fire in the wooded grasslands, as shown later in this Chapter, probably promoted higher fire frequency in these areas of this vegetation type.

The mixed thicket and scrub physiognomic type was only about 220 ha in extent and did not produce meaningful results. The combined mixed bushland, thicket and scrub physiognomic type had a median fire frequency of two fires in 37 years (one fire in 18.5 years) (Figure 4.6c; Table 4.3). This was probably a result of the high shrub cover (more than 35%), which is negatively correlated with grass cover (Goodman, 1990; Table 4.3), and thus the potential for fire was low. Once again, by a query of the fire frequency data layer for this combined vegetation type, it was apparent that the small areas of very high fire frequency (up to 20 fires in 37 years) (Figure 4.6c) occurred mainly in the western part of the reserve in areas where this combined vegetation type abutted onto mainly fire-prone wooded grasslands.

Wooded grasslands have a low shrub cover (less than 35%) and a tree cover above 10% but always less than 50% (Goodman, 1990; Table 4.3). The relatively high grass cover in this vegetation physiognomic type is no doubt the reason for a higher frequency of fire than the previously discussed physiognomic types. The median frequency of fire for this physiognomic type was seven fires in 37 years (one fire in 5.3 years) (Figure 4.6d; Table 4.3). The broad “bell” shape of the histogram, compared to those of other physiognomic types, also shows that there is a fairly wide range of fire frequencies that cover substantial proportions of this physiognomic type. This may well be a symptom of the great extent of this particular physiognomic type, which covers about 38% of the study area.

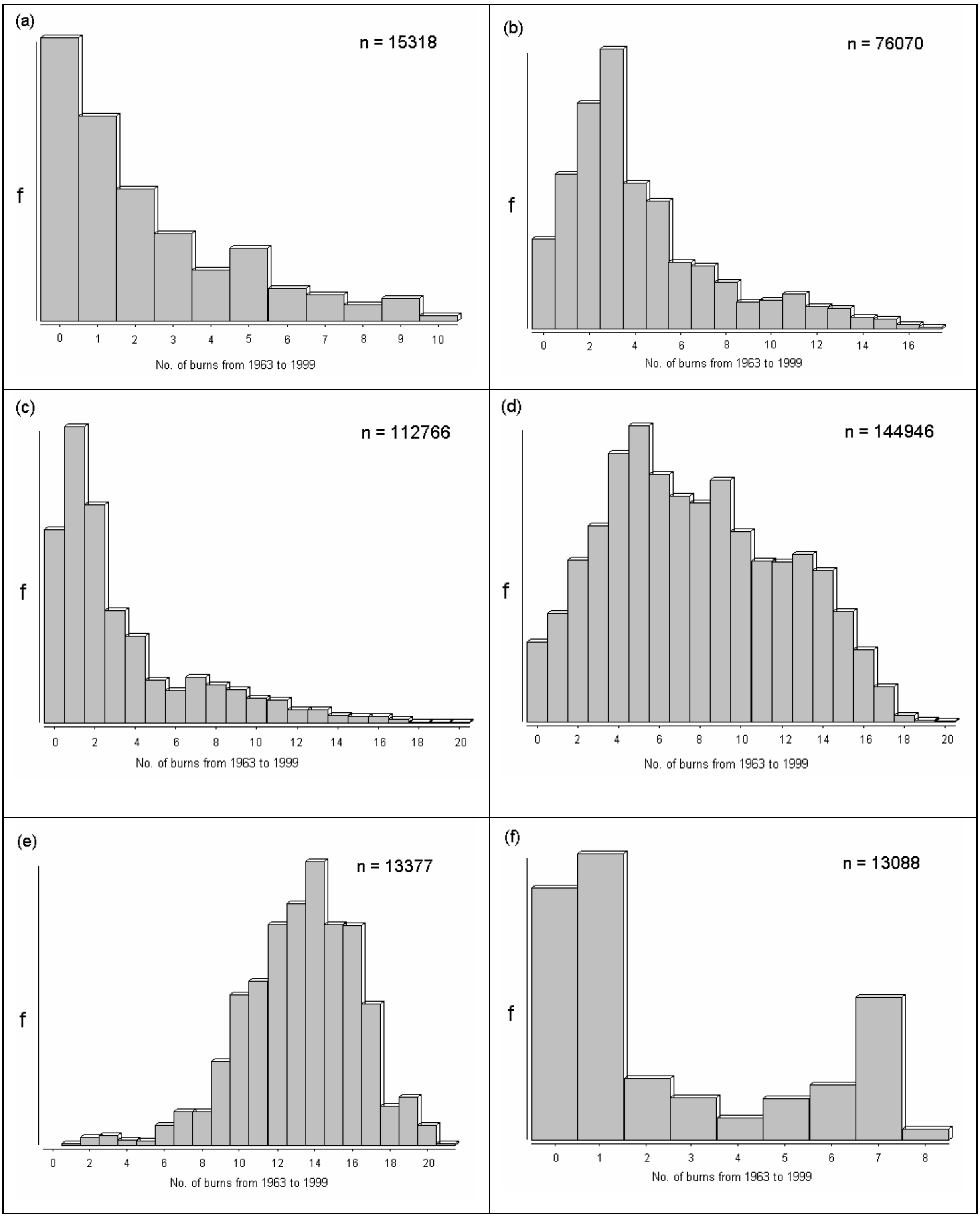


Figure 4.6: Fire frequency for the different vegetation physiognomic types, namely (a) Forest and Closed Woodland, (b) Mixed Woodland, (c) Mixed Bushland, Thicket and Scrub, (d) Wooded Grassland, (e) Grassland, and (f) Wetlands. The sample size (n) refers to the number of 25 m x 25 m pixels. Refer to Appendix 2 for summary statistics.

Table 4.3: Fire frequency for the different vegetation types with a description of woody and herbaceous biomass (late wet season)

Vegetation Type	Woody Abundance – Grass biomass is negatively correlated with woody biomass (Goodman, 1990)	Average herbaceous biomass (gm^{-2}) (after Goodman, 1990)	Area (ha)	% of Study Area	Median Number of Fires in 37 yrs.	Median Yrs. Between Fires
Forest & Closed Woodland	Tree cover closed or nearly so (> 75%)	6.0*	957	4	1	37
Mixed Bushland, Thicket & Scrub	Shrub cover > 35%; Bushland - Tree cover > 10% but < 75%; Thicket & Scrub - Tree cover < 10%	32.0**	7048	30	2	18.5
Mixed Woodland	Tree cover > 50% but < 75%; Shrub cover < 35%	59.1	4754	20	3	12.3
Wooded Grassland	Shrub cover < 35%; Tree cover > 10% but < 50%	79.5	9059	38	7	5.3
Grassland	Shrub cover < 35%; Tree cover < 10%	No data	836	4	14	2.6
Wetlands	No cover data available	No data	818	3	1***	37
Unknown Vegetation Type			178	1	-	-

*Refers to forest, closed woodland and bushland in Goodman's (1990) data.
**Refers to bushland only.
***This statistic is not preferred as the distribution was bimodal, with a major mode = 1 and minor mode = 7.

As one would expect, the frequency of fire in the grassland physiognomic type is much higher than all the other physiognomic types as it is the most prone to fire. It has previously been noted that the high grass biomass in the Lebombo foothills, the area in which the grassland physiognomic type is mainly found, facilitates the opportunity for frequent fire (Goodman, 1990). Frequent fires further discourage woody vegetation, which encourages more grass (Goodman, 1990). The median frequency of fire in this vegetation physiognomic type was 14 fires in 37 years (one fire in 2.6 years) (Figure 4.6e; Table 4.3).

The wetland physiognomic type, which includes seasonally to permanently inundated sedge, grassland and reed swamps, shows a bi-modal distribution for fire frequency. This pattern is almost identical to the histogram of fire frequency for the marsh geological type, and typifies the strong relationship between geology and vegetation in the reserve. The first peak of between zero and one fire (mode of one) in the 37-year period represents the areas which are almost permanently inundated with water, while the second peak (mode) at seven fires in 37 years (one fire in approximately five years) represents the seasonally wet areas, which dry out in the winter months and are susceptible to fires (Figure 4.6f).

A summary of the relationship between vegetation type, excluding wetlands, and fire frequency is given in Figure 4.7. A ranking of herbaceous biomass per vegetation type based on the woody and herbaceous cover descriptions (Moll, 1968, 1980; Goodman, 1990) together with late wet-season herbaceous biomass data (Goodman, 1990; Table 4.3) shows clearly that median fire frequency increases with increasing herbaceous biomass. Fire frequency in wetlands, however, is not primarily dependent on herbaceous biomass but rather on the hydrological regime.

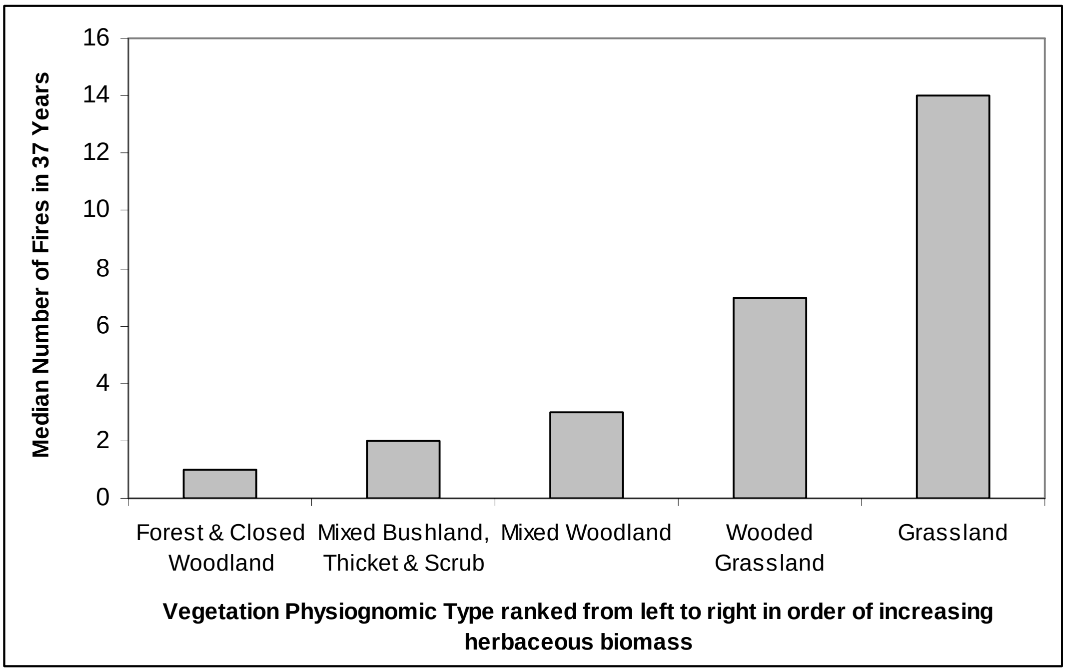


Figure 4.7: Relationship between vegetation type and fire frequency. Vegetation type is ranked, from left to right, in order of increasing herbaceous biomass.

4.3.4 Relationship between rainfall distribution and fire frequency

There was a strong positive relationship (R^2=0.666; d.f.=5; $P<0.05$) between mean wet-season rainfall, as determined from the rain gauges within the study area, and the frequency of fire (Figure 4.8a). The Mtshopi rain gauge point in Figure 4.8a appears to be an outlier, which can be explained by the fact that the actual gauge is located in a slight rain shadow (Nhlohlela River valley), which is surrounded by an area with generally higher rainfall. The interpolated wet-season rainfall image supported this showing that the general area has a mean wet-season rainfall of 547 mm, as opposed to the measured mean wet-season rainfall of 524 mm. Excluding this point from the regression analysis revealed an even stronger relationship between mean wet-season rainfall and fire frequency (R^2=0.974; d.f.=4; P<0.002) (Figure 4.8b).

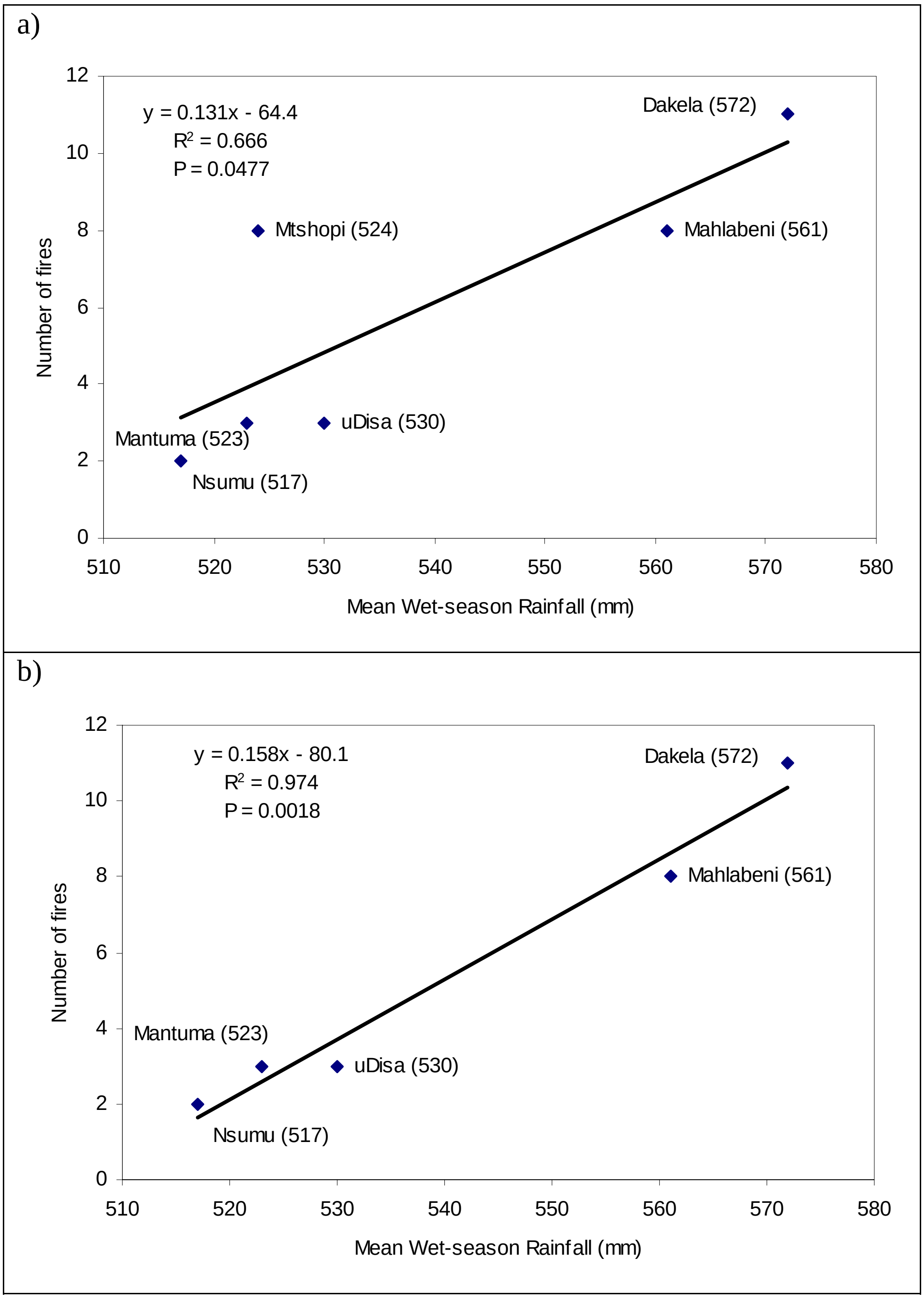


Figure 4.8: Relationship between wet-season rainfall and fire frequency (number of fires in the 37-year study period) at rainfall gauge locations, with Mtshopi rainfall data (a) included and (b) excluded.

In another analysis, where a pixel-by-pixel analysis of the relationship between an interpolated surface of mean wet-season rainfall and the fire frequency image was undertaken, a positive relationship was again noted (Figure 4.9). The part of the plot, which shows low fire frequency where there was high wet-season rainfall, can be explained by the occurrence of fire resistant vegetation types in the higher rainfall areas. This was confirmed in the second pixel-by-pixel analysis between mean wet-season rainfall and fire frequency, where fire resistant vegetation types such as forest and closed woodland, thicket and scrub, and wetland areas were excluded (Figure 4.10). In this plot, fewer points fell in the high rainfall and low fire frequency part of the graph.

The consistent positive relationship between wet-season rainfall and fire frequency observed above, is due to the positive influence of rainfall on grass production and thus fuel load (Sinclair, 1979; Dye and Spear, 1982; Fynn and O'Connor, 2000; O'Connor *et al.*, 2001), which in turn influences fire frequency (Trollope, 1984a).

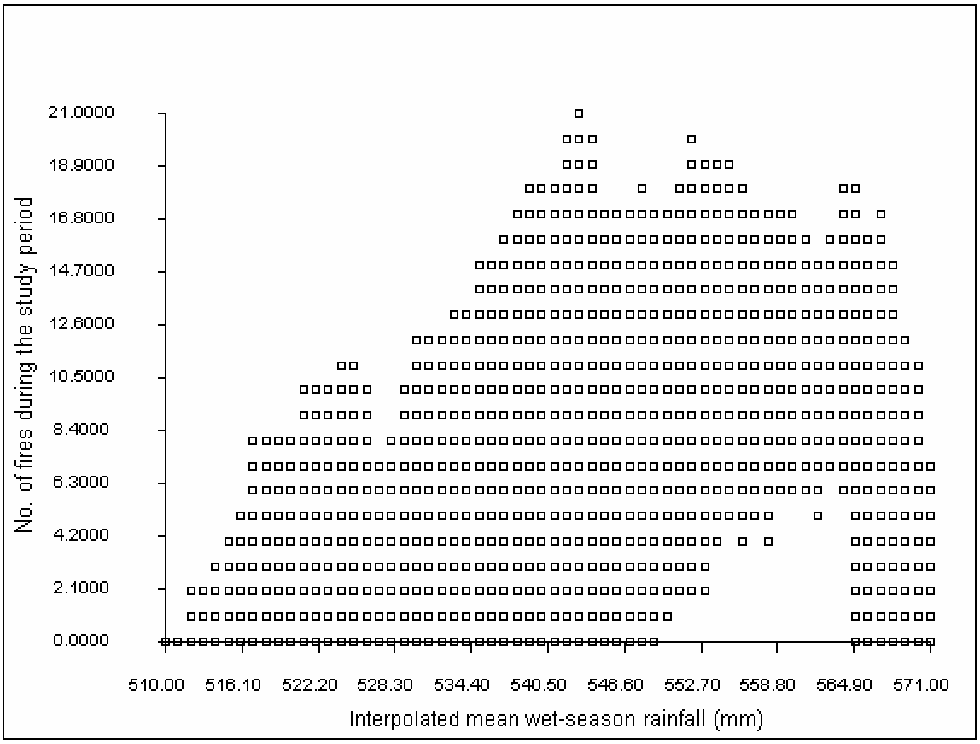


Figure 4.9: Relationship between fire frequency (number of fires during the study period of 37 years) and mean wet-season rainfall (interpolated rainfall image), for the entire study area.

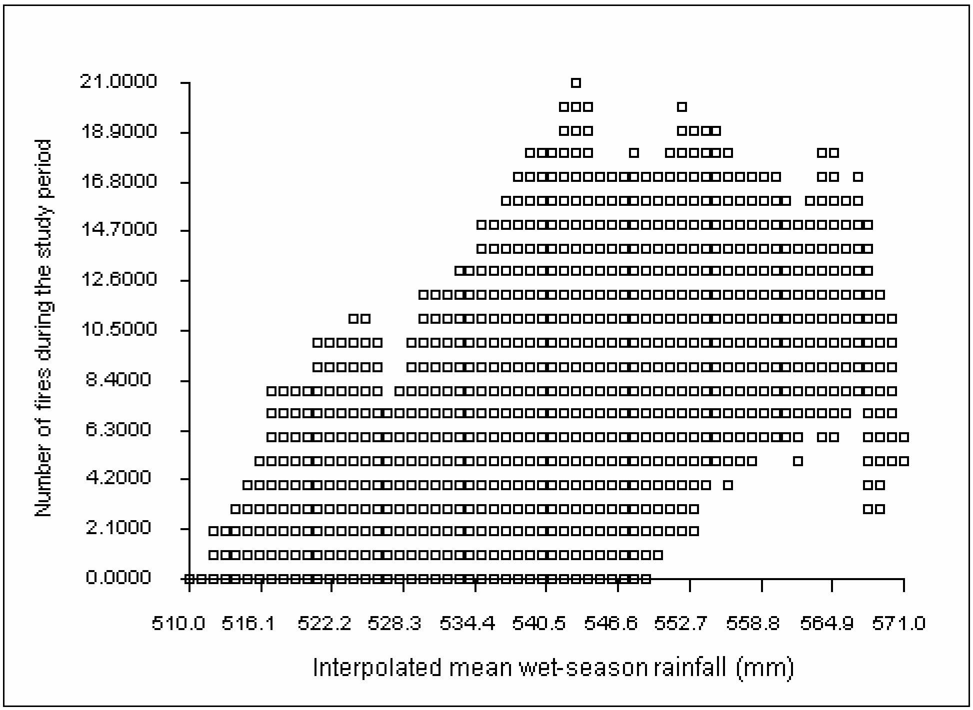


Figure 4.10: Relationship between fire frequency (number of fires during the study period of 37 years) and mean wet-season rainfall (interpolated rainfall image), where fire-resistant vegetation physiognomic types have been excluded.

4.3.5 Relationship between herbivore biomass and fire frequency

The relationship between grazer biomass and fire frequency, for an analysis using all vegetation types, was very weak and negative (R^2=0.0047; d.f.=3327; P<0.0001) (Figure 4.11). Similarly, the relationship was weak between large herbivore biomass (browsers and grazers combined) and fire frequency (R^2=0.012; d.f.=3327; P<0.0001). The coefficient of determination for the relationship between grazer biomass and fire frequency showed about a seven-fold improvement when only grassland and wooded grassland areas were included (R^2=0.036; d.f.=1713; P<0.0001) (Figure 4.12). This was mainly because areas where grazing and fire are not prominent processes were excluded from the analysis. Although the correlation coefficients were not high in the above analyses, it was evident from a visual examination of these graphs that fire frequency was at least being limited by grazer density (triangular data distribution), that is, where grazer biomass was high, fire frequency was usually low. However, where grazer biomass was low, fire frequency ranged from zero to high values, indicating that other factors were also affecting fire frequency. In an attempt to highlight this limiting relationship, a regression analysis of the upper edge of the point cloud in Figure 4.11 (envelope or factor ceiling analysis) was undertaken. This analysis defined the relationship more appropriately (R^2=0.52; d.f.=284; P<0.0001), where below a threshold of grazing pressure of about 0.5 AU ha^{-1}, fire frequency increased exponentially, while above this threshold, fire frequency was limited (Figure 4.13).

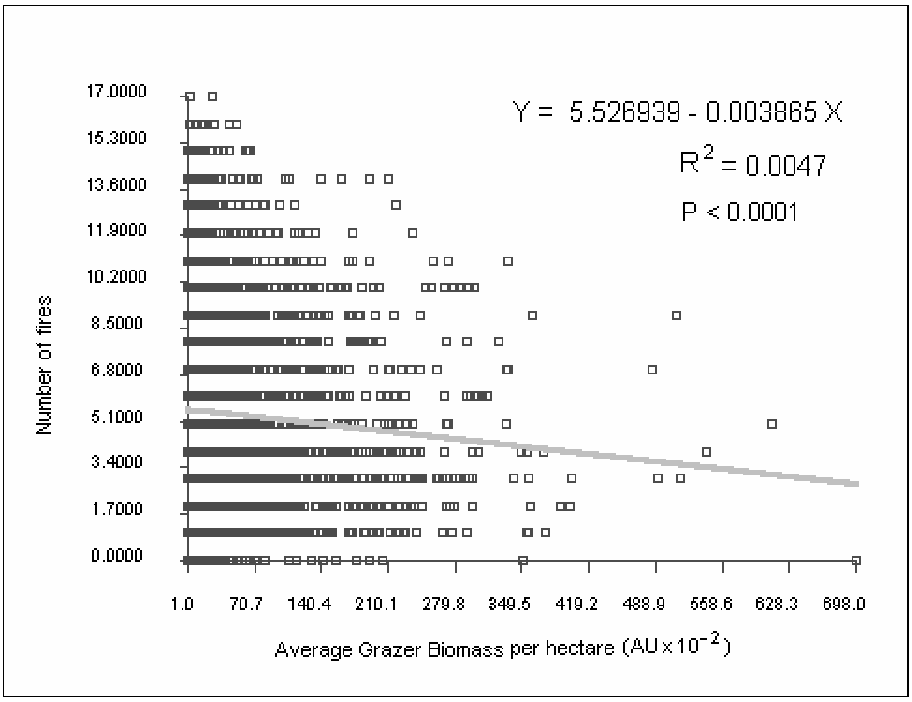


Figure 4.11: Relationship between grazer biomass (average number of AU per grid-cell) and fire frequency (number of fires in the 37-year study period), for all vegetation types.

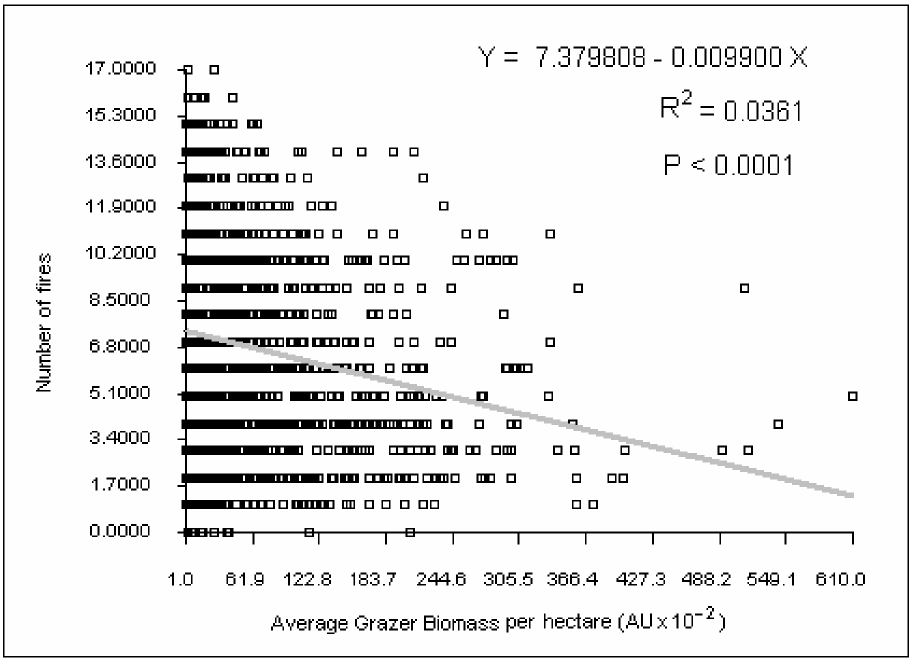


Figure 4.12: Relationship between grazer biomass (average number of AU per grid-cell) and fire frequency (number of fires in the 37-year study period), for only the grassland and wooded grassland physiognomic types.

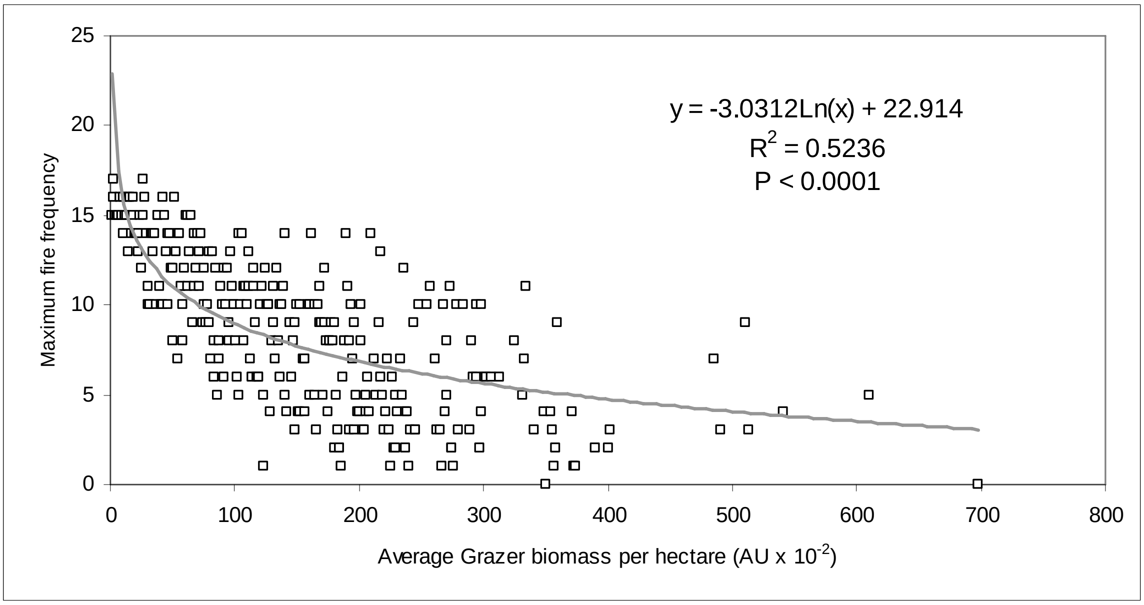


Figure 4.13: Relationship between grazer biomass and fire frequency as determined using a “factor ceiling” analysis.

4.4 DISCUSSION

4.4.1 General fire frequency patterns

In general, higher frequencies of fire occurred in the western part of the reserve as compared to the eastern part. As will be discussed in the following sections, this is related to the geology and associated vegetation, which vary from west to east in the reserve. Furthermore, a rainfall gradient decreasing from south-west to north-east has also played a significant role in creating this general fire pattern.

The term “natural fire regime” generally refers to the fire regime (season, frequency, intensity) which exists in areas that are relatively undisturbed by man and where fires are caused by factors other than man (Trollope, 1984a; Van Wilgen *et al.*, 1990), but in MGR man (Iron-age) has also been recognised as an important contributor to the fire regime in the past. Regardless of the cause of ignition, however, the frequency of burning would depend on the rate of accumulation of sufficient grass fuel to support a fire, which is affected the most by rainfall (Trollope, 1984a). The natural frequency of fire in moist savanna is thought to have ranged between annual to biennial and in arid savanna, between once every three to five years (Gertenbach and Potgieter, 1979; Trollope, 1984a), although frequencies in African savannas can be as low as once every 30 years or more, depending on rainfall and grazing pressure (Bond and Van Wilgen, 1996). It is therefore possible that a semi-arid savanna (rainfall higher than arid savanna but lower than mesic savanna), such as in Mkuzi Game Reserve, could have had a past fire frequency of between one fire in two years to about one fire in four years, depending on rainfall and degree of utilisation by herbivores. Because one of Ezemvelo KZN Wildlife's goals is to manage processes (in this case fire) in a manner that closely resembles their natural or past states (Reserve Management Plan, 1986; Balfour and Howison, 2001), it is useful to compare the recent fire regime in MGR to this estimate of what may have been the past fire frequency, but bearing in mind that rainfall, herbivory and vegetation types can affect the fire regime and may differ from those areas used as reference points for estimating the possible past fire frequency in MGR. About 25% of the study area was found to have fire frequencies in the range of one fire in two years to one fire in four years, which is also within the generally accepted range of applied fire frequency in the moist savanna areas (Trollope, 1984a). Most of the area, almost 75% of it, had fire frequencies lower than one fire in four years. This provides an interesting perspective but it should not be concluded that fire did not occur frequently enough (compared to the estimated past fire frequency) over the reserve as a whole, because it is clear from this study that rainfall, herbivory and vegetation type influence fire frequency. Indeed, the more fire prone vegetation types of wooded grassland and grassland experienced relatively high frequencies of one fire in approximately five years and one fire in approximately three years, respectively, while high rainfall areas experienced fire as frequently as one in approximately three years (e.g. Dakela area) and herbivore density above 0.5 AU ha^{-1} limited fire occurrence.

4.4.2 Spatial variation of fire patterns in relation to vegetation type and geology

There appeared to be a strong correlation between fire frequency and vegetation type, which was most probably related to, firstly, the amount of herbaceous material available for fuel and, secondly, the moisture regime. The herbaceous layer is more inflammable than the woody layer and dense woody vegetation with a low herbaceous biomass is less prone to fire than a mixed grass-woody vegetation (Edwards, D., 1984). Accordingly, when vegetation type was ranked in order of increasing herbaceous biomass, a clear pattern of increasing fire frequency was observed (Figure 4.7). The lowest frequency of fire was found in the forest and closed woodland physiognomic type, where the herbaceous layer is very sparse (Goodman, 1990; Table 4.3), while an equally low frequency occurred in wetlands most probably because of seasonal to permanent inundation by water. Within wetlands, some seasonally inundated wetlands had relatively high fire frequency, presumably as a result of drying out during the winter months, and becoming more susceptible to fire. Wooded grasslands and grasslands understandably had the highest fire frequencies as a result of an abundance of herbaceous fuel (Edwards, D., 1984; Trollope, 1984b; Figure 4.7; Table 4.3), while the woodland and bushland, thicket and scrub vegetation physiognomic types had low to intermediate fire frequency mainly as a result of fairly low amounts of herbaceous fuel, relative to the wooded grassland and grassland physiognomic types (Moll, 1968; Goodman, 1990; Figure 4.7; Table 4.3).

The analyses of fire frequency and area burnt in relation to geology showed that both were quite distinctly related to the various geological types. This was interpreted as being directly influenced by the distinct vegetation types occurring on the different geological substrates and related soil types, as identified by Goodman (1990). For example, fire frequency was the highest (an average of one fire every four years) on the volcanic rhyolite, rhyodacite and syenite geological type, which occurs in the Lebombo mountains, while fire frequency was the lowest (an average of one fire in 31 years) for areas with a young alluvium substrate. The Lebombo mountains areas with a volcanic substrate is associated mainly with grassland and wooded grassland, which together cover about 74% of the geological type (Appendix 1; Table 4.2), and which have ample grass fuel to carry fire, while on the other extreme, young alluvium is associated with riverine forest and woodland vegetation where the herbaceous layer is usually sparse (Goodman, 1990), and is therefore much less likely to carry fire. Geology therefore did not provide any more insight into the spatial variation of fire patterns than vegetation type did.

4.4.3 Relationship between rainfall distribution and fire frequency

Results consistently showed that there is a direct and positive relationship between rainfall and fire frequency in the reserve. Because grass constitutes the main fuel component of savanna fires (Trollope, 1984a), and rainfall has a positive linear effect on grass production (Sinclair, 1979; Dye and Spear, 1982; Briggs and Knapp, 1995; Fynn and O'Connor, 2000; O'Connor *et al.*, 2001) and thus fuel accumulation, it follows that the frequency of burning is positively related to rainfall (Trollope, 1984a). The results from this study are therefore consistent with our understanding of the influence of rainfall on fire frequency. Although it may be argued that most of the fires in the reserve are prescribed fires and that the frequency would depend on reserve management policy, the extent of burns (spread) are expected to be greater during higher rainfall periods and in higher rainfall areas, while the lack of fuel during dry years and in drier areas has affected management decisions to implement prescribed burns. The reserve management policy discourages burning when the goal is specifically to retard woody growth and the amount of accumulated fuel is not sufficient to achieve intense fires, or when the goal is to maintain a diverse and vigorous herbaceous sward but there is no excessive residual herbaceous material present (Reserve Management Plan, 1986). Nevertheless, if fires were ignited in areas with very low fuel load, they would not be extensive.

4.4.4 Relationship between herbivore biomass and fire frequency

A negative correlation between fire frequency and herbivore biomass, whether in the form of grazers or both grazers and browsers combined, was apparent from the regression analyses. The correlation coefficients for the two initial analyses, however, were extremely low indicating that there were other factors that influenced fire frequency, for example rainfall (shown in this study). Under normal circumstances, multiple linear regression would have been the next logical step in determining the relative influence of the different variables on fire frequency, but the different nature of the various data sets, for example the rainfall image (interpolated data) and herbivore density images (measured values for a sample set of pixels), prevented such an analysis and therefore alternative ways of identifying the effect of herbivores were employed. Furthermore, the weak correlation in the pixel-by-pixel analyses is also partly a consequence of the nature of the herbivore data, which were sampled over a very short period in the year, and therefore many pixels suitable for a herbivore would have been vacant.

What was striking about the initial plots was that the data points had a triangular distribution indicating that there was some limiting factor in operation (Maller, 1990; Thompson *et al.*, 1996). In this case, herbivores, through defoliation and trampling (Tainton, 1981; Heady and Child, 1994; Pickup, 1994), have limited the amount of herbaceous material, which in turn limited fire frequency. Defoliation directly reduces the amount of available grass fuel (Edwards, D., 1984) while trampling could lead to compaction of grass fuel and also promotes decomposition (Tainton, 1981; Heady and Child, 1994). Alternatively, the lack of herbivores would allow herbaceous vegetation to accumulate, thereby increasing the potential for fires to occur (Edwards, D., 1984; Bond and Van Wilgen, 1996). The seven-fold greater correlation obtained for the analysis which excluded areas where grazing and fire were not prominent processes, namely areas of relatively low grass biomass, further indicated that the amount of available herbaceous biomass is an important determinant of the occurrence of fire. It is therefore clear that herbivore biomass has been a limiting rather than a controlling factor, in the relationship with fire frequency. The factor ceiling regression analysis showing that about 52% of the variation in the maximum fire frequency is determined by herbivore biomass, more appropriately defines the relationship. The negative relationship between fire frequency and herbivores that was found in this study is consistent with research in lowveld savanna of Swaziland where fire frequency was negatively related to grazing pressure (Roques *et al.*, 2001). Similarly, in the Serengeti, increased grazing pressure that resulted from eruptions of wildebeest and buffalo coincided exactly with a decrease in the extent of burning (Norton-Griffiths, 1979). Furthermore, fires occurred with low frequency over the period from 1963 to 1972 on the Serengeti plains where the wildebeest density was highest compared with the northern stratum where the wildebeest density was lowest as estimated in 1972 (Norton-Griffiths, 1979).

The fact that herbivores are usually attracted to the nutritious green flush which occurs soon after fires (Edwards, P., 1984) may lead one to believe that greater herbivore biomass should occur on frequently burnt areas. However, this is a short-term phenomenon, and overall long-term habitat preferences of herbivores are more likely to influence long-term landscape-scale fire patterns.

CHAPTER 5: TEMPORAL VARIATION OF THE FIRE REGIME

5.1 INTRODUCTION

It was established in Chapter 4 that fire regimes can vary spatially at a landscape level due to factors such as rainfall, geology and soils, vegetation type and distribution of large herbivores. This chapter will attempt to show at a landscape level how the fire regime in the reserve has varied over time, that is, from year to year in relation to rainfall and herbivore biomass, and over the wet and dry seasons. In addition, the influence of fire intensity, which varies according to a number of factors including fuel load, daily weather and seasonal conditions, on the size of burns was also investigated.

It was expected that the total area burnt annually would be directly related to the amount of rainfall in the preceding wet season, as rainfall is known to positively influence the production of grass (Sinclair, 1979; Dye and Spear, 1982; Briggs and Knapp, 1995; Fynn and O'Connor, 2000; O'Connor *et al.,* 2001), the main fuel in savanna fires (Trollope, 1984a). This was evident in the Serengeti where the extent of burning in a year was positively associated with wet-season rainfall (Norton-Griffiths, 1979), and in Hluhluwe-Umfolozi Park where the area of the park that burnt in any one year was positively correlated with the rainfall of the previous summer season (Balfour and Howison, 2001). Further support for this expectation is provided by Williams *et al.*, (1998) who found that the proportion of transects burned in plots in Kakadu National Park in Australia was positively related to fire intensity (Gill *et al.*, 2000), which is in turn positively related to fuel load (Byram, 1959; Brown and Davis, 1973; Luke and McArthur, 1978; Trollope, 1984b; Bond and Van Wilgen, 1996). A greater and uniformly spread fuel load is thus expected to promote the occurrence and spread of fire, resulting in larger burns. A second factor that influences the potential of an area to burn is the amount of rainfall during the dry season. In general, it is expected that high rainfall during the dry season will promote grass growth and in turn green biomass (San José and Medina, 1976; Medina and Silva, 1990), while moisture content of herbaceous material is expected to increase, thereby reducing the potential for fire to occur. Conversely, low rainfall during the dry season allows the herbaceous grass layer to dry out (cure) sufficiently, thereby increasing the potential for fire to occur (Luke and McArthur, 1978; Bond and Van Wilgen, 1996). The effect of rainfall during the dry season was seen in the Serengeti, where dry-season rainfall had a weak negative influence on the extent of fire (Norton-Griffiths, 1979). Because the spread of fire is dependent on the amount and distribution of fuel, large herbivores that consume and trample grass (Tainton, 1981; Heady and Child, 1994) and thus impact negatively on the amount and cover of herbaceous vegetation (Pickup, 1994), were expected to influence the spread of fire negatively and consequently the size of burns. Evidence from the Serengeti showed that eruptions of wildebeest and buffalo, after the eradication of rinderpest, resulted in an increase in grass consumption, which coincided exactly with a decrease in the extent of burning (Norton-Griffiths, 1979). Therefore, during periods when high herbivore densities occurred in the reserve, the sizes of burns were expected to be low and vice versa. With regard to the effect of season on burn size, the dry season provides the best conditions for fires to occur (Luke and McArthur, 1978; Trollope, 1984a) and thus larger fires were expected during this period than during the wet season. It has been stated that provided dry conditions occurred during some time of the year, fire could occur (Bond and Van Wilgen, 1996). Fire intensity depends on the amount of fuel, the heat yield of the fuel and the rate of spread of the fire (Byram, 1959; Luke and McArthur, 1978; Bond and Van Wilgen, 1996). However, fuel moisture content, which is affected by daily weather conditions as well as seasonal changes (Luke and McArthur, 1978), is a critical factor since it affects ease of ignition, quantity of fuel consumed and combustion rate of different types of fuel (Trollope, 1984b). Therefore, fire intensity, which in this study was estimated from the type of burn (very patchy, patchy, clean, intense), was expected to be directly related to the area burnt.

5.2 METHODS

5.2.1 Area burnt annually in relation to annual rainfall patterns

Boolean raster layers representing the extent of fires for each year were analysed using the AREA module in IDRISI to determine the total area burnt in the study area for each year. Long-term rainfall records for the Mantuma weather station were used to determine the preceding wet-season (period from October to March) rainfall and the annual dry-season (period from April to September) rainfall for the area. The wet season was defined as all months with a mean precipitation of >50 mm (Knoch and Schulze, 1957; Goodman, 1990). All statistical analyses were undertaken using SIGMASTAT statistical software.

An initial analysis of annual total area burnt in relation to the preceding wet-season rainfall was undertaken by graphically representing the two sets of data in a single combination (bar and line) graph, from which a visual assessment of the relationship was done. Analyses of the relationship between these two variables were also done separately for years with wet dry seasons and years with dry dry seasons. Years with wet dry seasons and dry dry seasons were separated with the aid of a graphical representation of dry-season rainfall arranged in ascending order (Figure 5.1.). The graph revealed a clear distribution shift at about 131 mm, that is, immediately beyond the 24th highest dry-season rainfall value (130.7 mm). A cut-off point, between wet dry seasons and dry dry seasons, of 131 mm was thus used. Interestingly, the average dry-season rainfall for this period was 150.0 mm, which if used as a cut-off point would divide the data set exactly the same way as using 131 mm, because the 25th highest value was 154 mm. Linear regression analyses for each of the wet dry season and dry dry season data sets were undertaken, firstly, between the preceding wet-season rainfall and annual total area burnt, and secondly, between the annual dry-season rainfall and the residuals of the first regression. This approach was taken as it was felt that the influence of wet-season and dry-season rainfall was not independent. It was expected that fire would primarily depend on how much grass has grown, but this could have been upset by greenness, which would result from dry-season rainfall. Wet-season rainfall was the most important of the two variables because it explained the most variance for both, years of wet dry seasons and dry dry seasons.

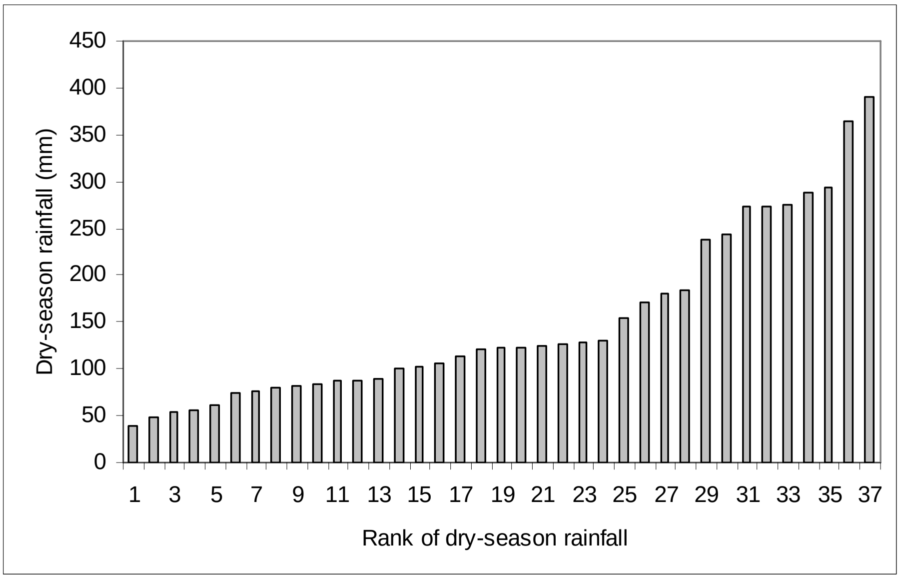


Figure 5.1: Dry-season rainfall for the 37-year study period ranked in ascending order.

It has also been shown that in addition to the effect of preceding wet-season rainfall (immediately preceding the dry season in which most burns occur) on the amount of fuel, the previous wet season (a year prior to the immediate preceding wet season) also has some influence through a carry over of fuel (Balfour and Howison, 2001) that was not completely consumed or had not totally decomposed since the previous wet season, as well as a carry over of soil moisture, which would promote phytomass production (Hanson *et al.*, 1982; Smoliak, 1986; O'Connor *et al.*, 2001). The relationship between total area burnt in a year and the preceding and previous wet-season's rainfall was investigated by means of multiple linear regression. Ideally, the rainfall in the two preceding wet seasons should have been weighted and combined as a single independent variable as their effect on the amount of fuel in the current burning season differs. There is, however, no rational way of doing this, so they were treated in this analysis as individual independent variables. The analysis was also separated, by means of the method explained above, for years of wet dry seasons and dry dry seasons in order to further illustrate the influence of dry-season rainfall on area burnt.

From preliminary investigations, two outliers were found in the data set, which were excluded from the regression analyses described above. These were data for 1972 where there was very little burnt (347 ha) despite an extremely high preceding wet-season rainfall (914.9 mm), and 1992 where an extremely large area was burnt (10251 ha) but the preceding wet-season rainfall was relatively low (395.7 mm). Both of these were thought to have been deliberate management decisions, not to burn and to implement extensive burns (58% prescribed burns; 33% firebreaks; 9% arson), respectively, and therefore fell outside of the sampling universe of this study. In the case of the 1972 outlier, it was felt that the decision not to burn was because this wetter wet season was preceded by a decade of below average wet seasons, a situation where reserve managers would not have been eager to burn, and would possibly have wanted to allow for a recovery of the herbaceous layer. The opposite scenario existed for the 1992 case, where five years of above average wet-season rainfall preceded the very dry 1992 wet season. Low rainfall in the dry season of the same year as well as a mainly moderately dense to dense fuel load (1992 fire record forms), which was possibly due to the net surplus rainfall from the two preceding wet seasons (Goodman, pers. comm.), was probably seen by management staff as a unique opportunity to implement bush control fires (more than 50% of the fires were intense). Based on rainfall patterns for the study period, no other situations similar to the two described above are likely to have occurred.

5.2.2 Area burnt annually in relation to large herbivore biomass (grazers)

The method of determining the annual total area burnt has been described above. Large herbivore biomass was determined from the line transect survey results for the reserve. Unfortunately, these data were only available for the period from 1984 to 1997, but also excluding 1991. Furthermore, reliable estimates for this period were only available for impala, blue wildebeest, zebra, nyala and warthog, and therefore only these species were used in the calculation of annual total grazing herbivore biomass. For the mixed feeders, impala and nyala, only a portion of their biomass, based on the proportion of grass in their diets, was used in the calculation of annual total grazing herbivore biomass, that is, 70% of annual total impala biomass and 40% of annual total nyala biomass was used. White rhinoceros and elephant, which can account for a substantial amount of grazer biomass in other reserves, have not been included in this analysis. In the case of white rhinoceros, no reliable estimates of abundance were available for most of the period under review in this analysis, and it is assumed that the population has not varied greatly over this period. Elephant (12 juveniles) were first introduced into the reserve in 1994 and a second introduction (a family group of 12 animals) took place in 1996. The two groups joined up soon after the second introduction and have spent most of their time in the southern half of the reserve outside of the study area, that is, about 70% of the sightings from introduction until 1997, have been outside of the study area. In addition the population was always below 30 over this period.

Initially, a visual assessment of the relationship between annual total area burnt and annual total grazing herbivore biomass was done, using a single graph showing change in these variables over time. Regression analyses were undertaken between preceding wet-season rainfall and total area burnt and between the residuals of this regression and annual total herbivore biomass, using SIGMASTAT. This approach was taken because wet-season rainfall accounts for most of the variance in annual total area burnt (established by previous analyses) through its impact on grass production, while herbivores can impact on the grass layer by consumptive utilisation and trampling.

5.2.3 Area burnt in relation to season of burn

The sample size for this analysis was limited by whether a date, accurate to a month, was recorded for a fire or not. This analysis included records of burns that occurred in the area defined by the present reserve boundary, as well as a few records from a neighbouring reserve, namely Makasa Game Reserve, so as to maximise the sample size. The total number of fire records used in this analysis was 366, which amounted to 85% of the total number of burn records available. All the burns that had dates accurate to at least the month were sorted into two data sets, namely a wet-season (fires that occurred during October to March) and a dry-season (fires that occurred during April to September) data set. The wet season was defined as all months with a mean precipitation of >50 mm (Knoch and Schulze, 1957; Goodman, 1990) (see Section 3.3.1). For reasons of data distribution, a Mann-Whitney rank sum test was used to examine the difference between the sizes of individual burns (determined using CARTALINX) of the wet- and dry-season data sets.

5.2.4 Area burnt in relation to fire intensity

Fire intensity was determined from the recorded result of each fire. The burns were rated as Intense, Clean, Patchy or Very Patchy. On the recording form created by past reserve staff, Intense was described as “intense burn with flame length > 3 m”, Clean was described as “clean but not very intense”, while Patchy and Very Patchy were not described further. It was decided for this analysis to combine the Patchy and Very Patchy classes due to the very small sample size of the latter class (n=10). This analysis included records of burns that occurred in the area defined by the present reserve boundary, as well as a few records from the neighbouring Makasa Game Reserve so as to maximise the sample size. This was necessary since an intensity rating was not recorded for a large proportion of the fires. The sample size during this analysis was thus 243 (56% of the total number of available fire records). The sizes of individual burns were compared among the intensity classes using a Kruskal-Wallis One Way Analysis of Variance on Ranks with an all pairwise multiple comparison (Dunn’s Method).

5.3 RESULTS

5.3.1 Area burnt annually in relation to annual rainfall patterns

There appears to be a general positive relationship between the annual total area burnt and the preceding wet-season’s rainfall, that is, the area burnt is large when the preceding wet-season's rainfall is high, and vice versa, although there were anomalies in some years (Figure 5.2). The regression analyses provided more convincing evidence that there is a positive relationship between these two variables for years with dry dry seasons (R^2=0.40, d.f.=21, P<0.002), but not for years with wet dry seasons (R^2=0.22, d.f.=12, P>0.1) (Figure 5.3). The was no relationship between dry-season rainfall and total area burnt (the residuals from the area versus wet-season rainfall regressions were used) for both, years with dry dry seasons (R^2=0.099, d.f.=21, P>0.15) and wet dry seasons (R^2=0.0001, d.f.=12, P>0.97) (Figure 5.3).

In Figure 5.3c, a single point (792mm, 50ha) representing 1984 data was identified as a possible outlier. Much of the rainfall (42%) in the 1983/1984 wet season was contributed by Cyclone Demoina where 335mm fell in five days (283mm in three days). At the same time, the reserve would have had low ground cover because the preceding five years' wet-season rainfall totals were below average. Much of the Cyclone Demoina rainfall is thus likely to have been lost through runoff and therefore would not have had an effect on grass growth and in turn not have contributed to fuel accumulation. Indeed, if this point is removed from the regression between preceding wet-season rainfall and area burnt, for years of wet dry seasons, a positive relationship is revealed (R^2=0.507, d.f.=11, P<0.01). Dry-season rainfall, however, still shows no relationship with total annual area burnt after the 1984 data are removed (R^2=0.006, d.f.=11, P>0.8).

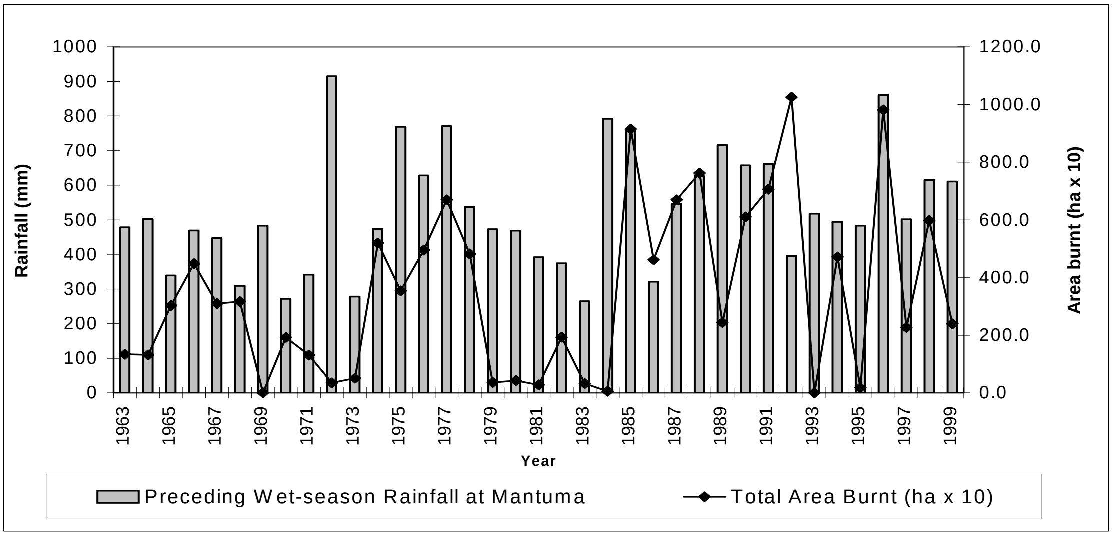


Figure 5.2: Annual total area burnt in relation to the preceding wet-season's rainfall. Note that the values of area burnt were scaled down (i.e. divided by 10) for display purposes.

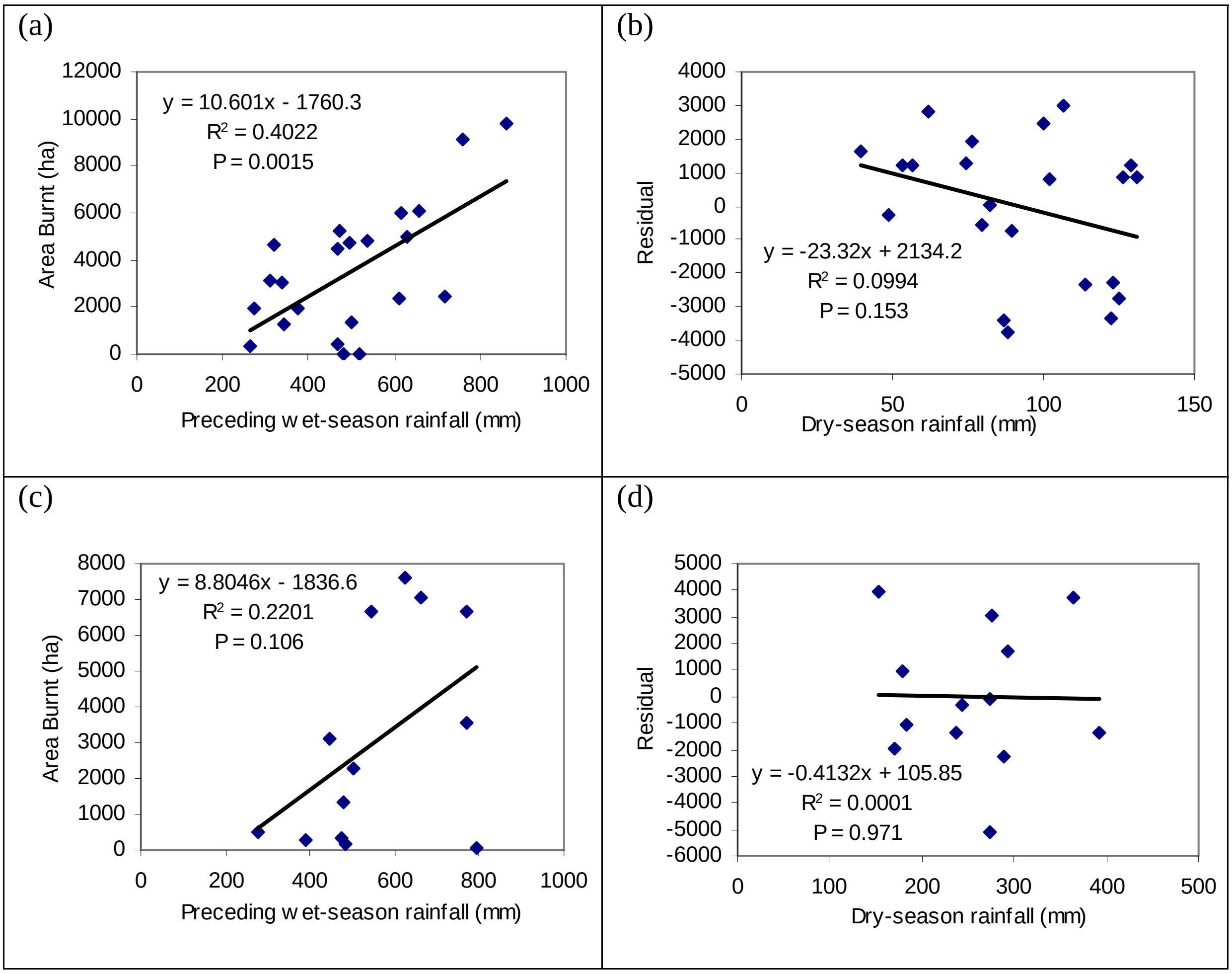
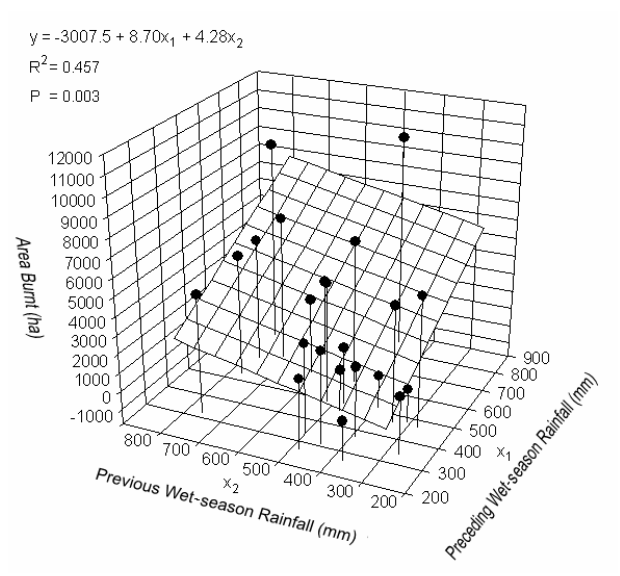


Figure 5.3: Relationship between the annual total area burnt and the preceding wet-season's rainfall for years with (a) dry dry seasons and (c) wet dry seasons, and between annual total area burnt (using residuals from the relationship with wet-season rainfall) and dry-season rainfall for years with (b) dry dry seasons and (d) wet dry seasons.

The relationship between total area burnt, and the preceding and previous wet-seasons rainfall for years of dry dry seasons was positive and significant (R^2=0.46, d.f.=21, P<0.004), while for years of wet dry seasons the relationship was not significant (R^2=0.269, d.f.=12, P>0.2) (Figure 5.4). In comparison to the effect of only the preceding wet-season rainfall (Figure 5.3), the previous wet-season rainfall accounted for a further 6% of the variance for years of dry dry seasons, and a further 5% of the variance for years of wet dry seasons. Some evidence for a possible negative influence of dry-season rainfall on area burnt was shown by a lower variance being explained by preceding and previous wet-seasons' rainfall for years of wet dry seasons (27%) as compared to years of dry dry seasons (46%) (Figure 5.4).

(a)



(b)

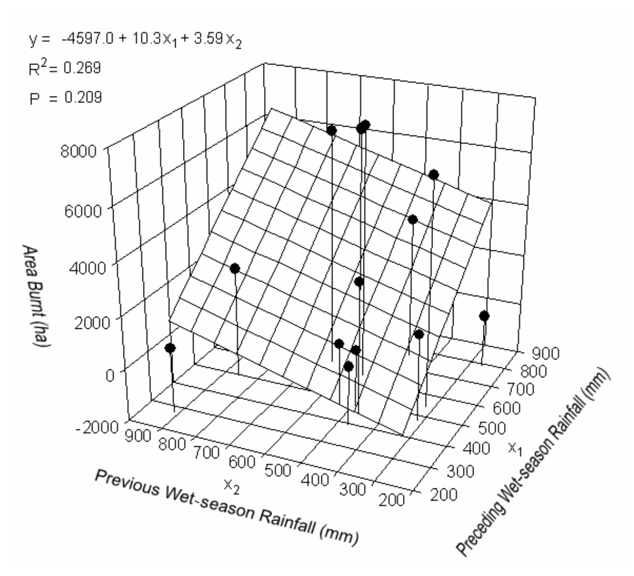


Figure 5.4: Relationship between the annual total area burnt and the preceding (immediately before dry season) and previous (1 year before preceding wet season) wet-seasons' rainfall for years with (a) dry dry seasons and (b) wet dry seasons.

5.3.2 Area burnt annually in relation to large herbivore biomass (grazers)

The annual total grazer biomass appears to have increased steadily from 1984 to 1997, but the annual total area burnt appears to have varied over this period, showing no definite relationship to annual total grazer biomass (Figure 5.5). It appears, however, that as biomass has increased since 1984, the area burnt in the reserve has become more variable, ranging from very large areas to very small areas, as can be noted for the period from 1992 onward. This phenomenon, however, may not necessarily be a result of increased herbivore biomass, but due to other factors such as extreme weather conditions. The regression analyses also showed no relationship (R^2=0.08, d.f.=12, P>0.3) between the annual total grazer biomass and annual total area burnt (residual of wet-season rainfall and annual total area burnt) (Figure 5.6).

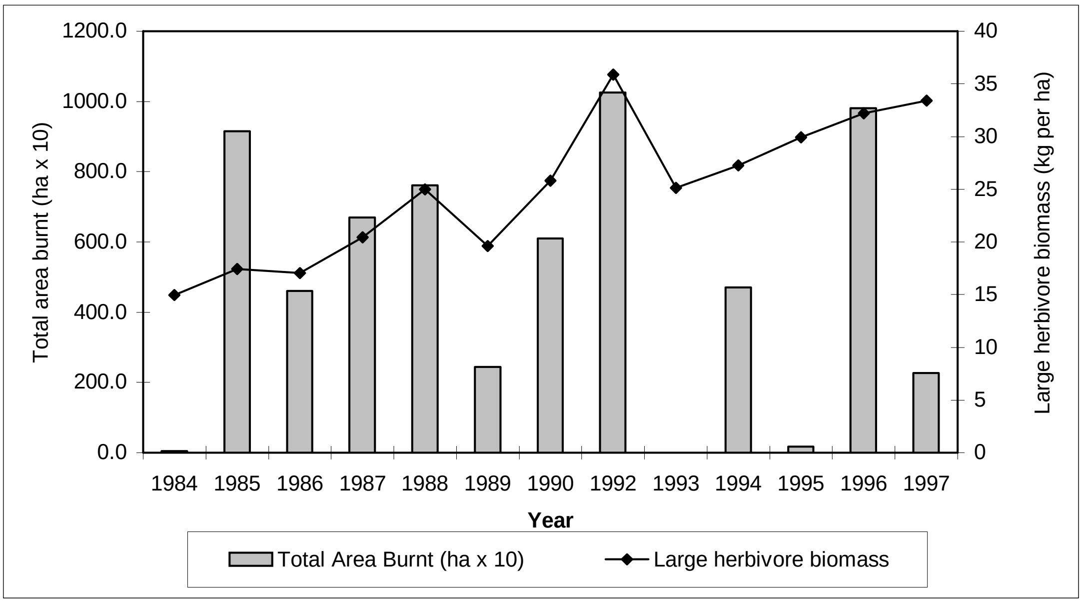


Figure 5.5: Total area burnt per year in relation to large herbivore (grazer) biomass, for the period from 1984 to 1997.

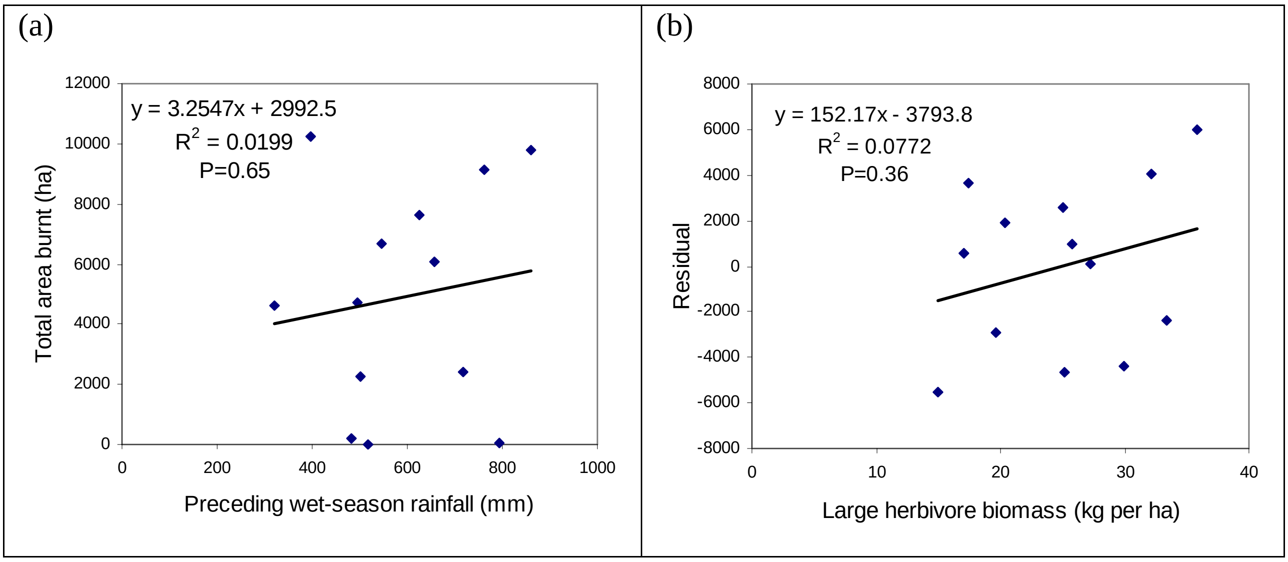


Figure 5.6: Relationship between (a) preceding wet-season rainfall and the annual total area burnt, and (b) the annual total grazer biomass and total area burnt, using the residual of the first regression. Note that the relationships shown here are only for the period from 1984 to 1997.

5.3.3 Area burnt in relation to season of burn

The Mann-Whitney rank sum test showed that the size of individual burns was greater for the dry season (median=194.1ha; n=299) than for the wet season (median=89.4ha; n=67) ($P<0.04$) (Figure 5.7). As would be expected, this result shows that the conditions during the dry season are more conducive to the spread of fire. In a typical dry season, much of the grass biomass is dead (dormant phase) (Trollope, 1984a) and has little moisture content because of generally drier climatic conditions (low rainfall and low humidity) (Schulze, 1972; Luke and McArthur, 1978; Lewis and Goodman, 1993). Conversely, during the wet season the grass is in a growing phase and is generally green with a relatively high moisture content. Humidity is also relatively high and even dead plant material can have a higher moisture content than during the dry season. In addition, the frequent occurrence of rain may keep dead potential fuel moist for extended periods. These factors therefore prevent the occurrence or spread of fires during the wet season.

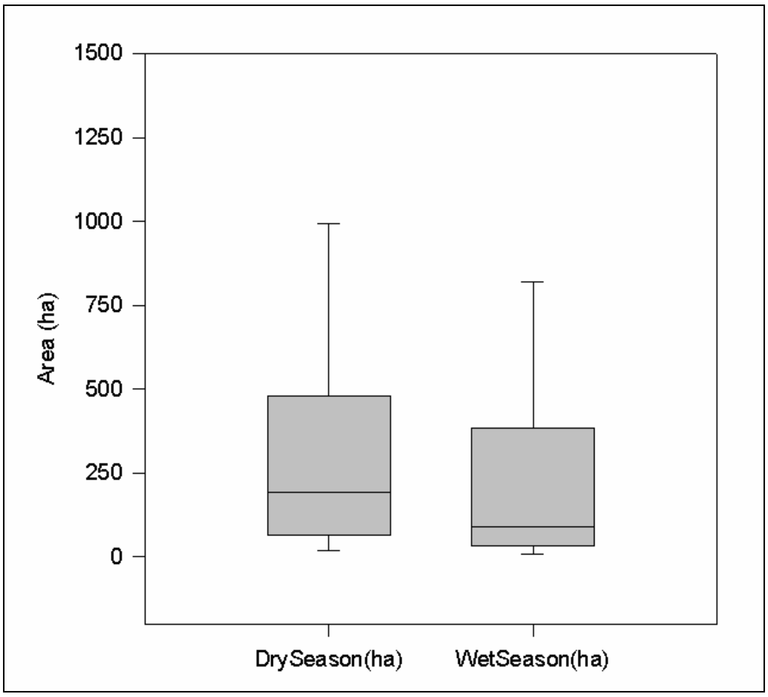


Figure 5.7: Comparison of individual burn sizes for the dry season and wet season.

5.3.4 Area burnt in relation to fire intensity

The overall comparison of the three intensity classes showed that they differed significantly (H=15.2; d.f.=2; $P<0.0005$) (Figure 5.8). The pairwise comparison revealed that the size of individual burns that were rated as Intense (median=341.2 ha) was significantly larger ($P<0.05$) than the Patchy/Very Patchy burns (median=171.4 ha) and the Clean burns (median=144.3 ha), but the latter two classes did not differ significantly. This indicates that intense fires generally spread over a larger area than “cooler” or less intense fires. Because fire intensity is dependent on the amount of available fuel (Luke and McArthur, 1978; Bond and Van Wilgen, 1996), a high load of fine fuel (grass) must have been present and, provided it was uniformly spread, would thus have promoted the spread of fire over large areas.

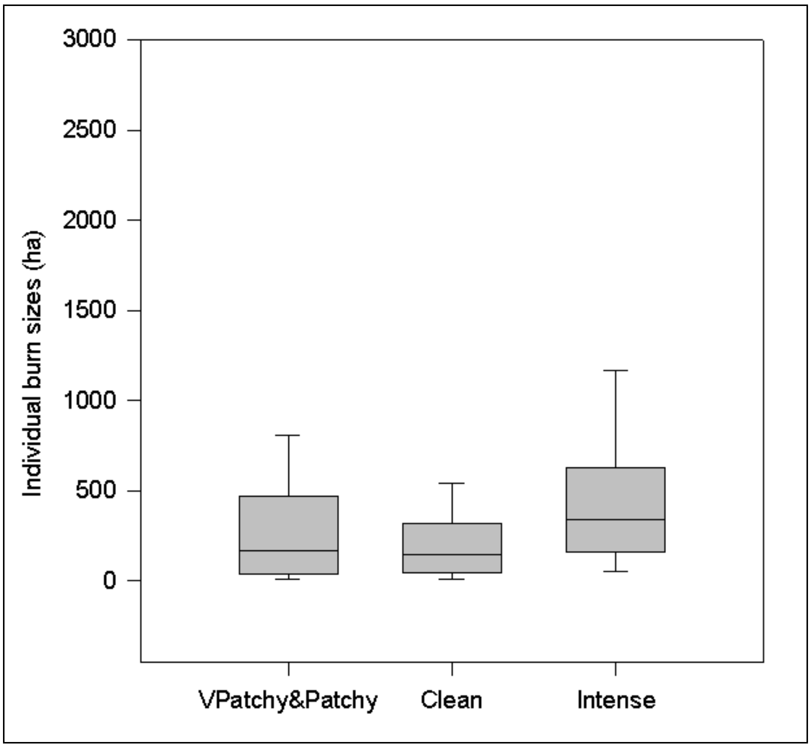


Figure 5.8: Relationship between burn intensity (based on burn result) and individual burn size.

5.4 DISCUSSION

5.4.1 Influence of fuel load on temporal patterns of fire

Rainfall undoubtedly had an influence on the extent over which fire can occur. In this case, there was a positive relationship between the preceding wet-season's rainfall and the annual total area burnt. This is because rainfall directly influences the growth of herbaceous material (Dye and Spear, 1982; Briggs and Knapp, 1995; Fynn and O'Connor, 2000; O'Connor *et al.*, 2001), which forms the main component of fuel in savanna (Trollope, 1984a). High rainfall in the wet season therefore leads to a higher herbaceous fuel load, and provided the fuel is allowed to dry out sufficiently (cure) during the dry season, the probability of the occurrence of fire is increased, and ultimately the total annual area burnt is expected to be higher. The relationship between the extent of burns and annual rainfall patterns found in this study is consistent with those found in the Serengeti (Norton-Griffiths,

1979) and Hluhluwe-Umfolozi Park (Balfour and Howison, 2001). A carry over effect of rainfall from one year to the next on phytomass was also supported by findings of this study. The preceding wet-season rainfall, together with the previous wet-season rainfall (one year prior to the immediate preceding wet season) affected the extent of burns significantly, but only for years when the dry-season rainfall was low. Although preceding wet-season rainfall was by far more important than the previous wet-season rainfall in influencing fuel load and extent of burns, their combined effect was more important than that of the preceding wet season alone, that is, the amount of variation explained by rainfall increased by about one seventh when previous wet season was also considered. These findings were in line with fire patterns observed in Hluhluwe-Umfolozi Park where the area burnt in any one year was more strongly and positively correlated with the mean rainfall for the previous two summers than that of one summer season, which was explained by a carry over of fuel from one year to the next (Balfour and Howison, 2001). However, this effect could also be due to a carry over of soil moisture from one year to the next, which would promote phytomass production (Hanson *et al.*, 1982; Smoliak, 1986; O'Connor *et al.*, 2001).

Contrary to the expected negative effect of herbivores on fuel load and thus extent of burns, the results revealed no relationship. Generally, it would be expected that as grazing pressure increased, the amount of available herbaceous fuel would be reduced (Tainton, 1981; Heady and Child, 1994; Milchunas and Lauenroth, 1993; Illius and O'Connor, 1999), and thus the total area burnt in a year would also be reduced (Norton-Griffiths, 1979). The following factors may be responsible for these anomalous results. Firstly and probably most significantly, the sample size, which was limited by the number of years of available large herbivore census data, was small (n=13). In addition to this, it is possible that herbivores limit (rather than control) the extent of fire, as was established for fire frequency in Chapter 4, and if such a relationship does exist, it would be difficult to identify if the sample size was small. Normal correlation analysis is not well suited for such relationships and therefore some type of factor-ceiling analysis may be appropriate (Thomson *et al.*, 1996), where a large sample size would most likely be required to distinguish the characteristic triangular data distribution (Maller, 1990; Thompson *et al.*, 1996). Further monitoring of these two variables will therefore be required before it can be determined whether herbivores do indeed impact on or limit the extent of burns in MGR. Secondly, it is possible that although the numbers of large herbivores have increased over the period from 1984 to 1997, they still remained below a level at which they could exert an influence on the total herbaceous cover over the reserve. The application of predator simulation in MGR may have contributed to this by preventing eruptions of game populations, as it is known that predators are able to regulate herbivore populations (Sinclair, 1981).

The amount of available fuel and the rate of spread determines fire intensity (Byram, 1959; Brown and Davis, 1973; Luke and McArthur, 1978; Trollope, 1984b; Bond and Van Wilgen, 1996) and thus it would be reasonable to expect that the size of burns would be positively related to intensity, which is supported by Williams *et al.* (1998) findings on the proportions of transects burned in plots in Kakadu National Park, Australia (Gill *et al.*, 2000). As expected, the more intense fires spread over a larger area than the less intense fires, but the trend was not distinguishable between the Patchy/Very Patchy and Clean burns. It is not clear why the size of burns for the latter two intensity classes did not differ. A possible weakness of this investigation, however, was that of the subjective nature of the burn intensity rating used, where observers' opinions may have differed with regard to what constituted the various intensity ratings. The method, although considered adequate for this investigation, could be improved in future by developing a better description of what characterises the different types of burns, as well as by using aerial survey methods or satellite photography to obtain a quantitative result, which could be used to check (or calibrate) field ratings. These options would, however, have to be considered in relation to limited resources and the contribution that such information would make to improving management of the reserve.

5.4.2 Influence of fuel condition on temporal patterns of fire

The sizes of individual burns were, as expected, significantly larger for the dry season than for the wet season. This is in line with the current understanding of the influence of climate on fire behaviour through its influence on fuel conditions. During the dry season, the grass is dormant (Trollope, 1984a) and cures (senescence takes place), containing much less moisture than during the growing phase, and this would favour the occurrence and thus the spread of fire (Luke and McArthur, 1978). Furthermore, fuels are drier and more combustible when relative humidity is low (Trollope, 1984b; Bond and Van Wilgen, 1996), as in the case of the dry season in MGR (Lewis and Goodman, 1993). Wind also directly affects spread of fire (Luke and McArther, 1978; Trollope, 1984b), and may have been partly responsible for the occurrence of larger fires in the dry season, particularly because September, the last month of the dry season, is one of the windiest months in the reserve (Goodman, 1981). A possible weakness of this analysis, however, was that fires were not allocated to season by checking the date of the fire against the first spring rainfall (>15 mm) which would have initiated growth. Instead, long-term rainfall data patterns were used to define fixed sets of months which represented the wet (October to March) and dry (April to September) seasons in MGR (Knoch and Schulze, 1957; Goodman, 1990), and fires were allocated to a season based on the month in which they occurred. Nevertheless, the significant results are testimony to the validity of the method that was used.

The herbaceous fuel condition can also be affected by rainfall during the dry season. High dry season rainfall can lead to an increased moisture content and greening of the herbaceous layer (San José and Medina, 1976; Medina and Silva, 1990), which in turn should reduce the probability of the occurrence of fire and impede the spread of fire because of reduced fire intensity (Luke and McArthur, 1978; Bond and Van Wilgen, 1996). Ultimately, it would be expected that the extent of fire or area burnt would be reduced (Norton-Griffiths, 1979; Williams *et al.*, 1998; Gill *et al.*, 2000). Contrary to this expectation, rainfall during the dry season did not impact on area burnt in this case, which may have been due to an overwhelming influence of wet-season rainfall.

CHAPTER 6: DIFFERENCE IN FIRE PATTERNS BETWEEN BLOCK BURNING AND A POINT SOURCE IGNITION STRATEGY

6.1 INTRODUCTION

A change in philosophy with regard to burning in Mkuzi Game Reserve (MGR) started around 1984/85, when there was a move away from a strict block burning strategy towards a Point Source Ignition (PSI) strategy (Goodman, pers. comm.). This change was in line with the process-based management philosophy adopted by the then Natal Parks Board (MGR Management Plan, 1986; KZNWildlife Policy No.1 - Mission Statement, 1997). It was envisaged that the PSI system would lead to a situation closely resembling the past or natural fire regime in savanna, which is believed to have resulted in a fire mosaic of areas burnt by different types and intensities of fire occurring at various times and frequencies, all of which maintained a diversity of habitat and species (Trollope, 1984a). The primary goal of the PSI strategy (also known as patch mosaic burning) is therefore to produce fire heterogeneity in the landscape (Brockett *et al*., 2001). This represents an important departure from block burning, which tends to create a relatively regular mosaic of post-fire ages (Brockett *et al*., 2001), that is, a more homogenous fire regime. It was therefore expected that there would be a difference in fire patterns between the block burning (applied from 1963 to 1984) and PSI burning (applied from 1985 to 1999) strategies, and that these differences should be evident in factors such as frequency, size, geometric character, season and intensity of burns, and the importance of different barriers to fire spread.

The change in burning philosophy from block to PSI in the reserve was also accompanied by a deliberate change from burning in early spring to burning throughout the dry season (Goodman, pers. comm.). In the early 1980's, Hluhluwe-Umfolozi Park, also made a shift to winter burns, particularly to combat bush encroachment (Wills, 1987). It was therefore expected that a greater proportion of burns would have occurred during the dry season of the PSI period as compared to the block burning period.

The goal of creating spatial heterogeneity in fire patterns, as has been shown in other reserves such as Pilanesberg National Park, requires the implementation of many small fires distributed widely over the protected area as opposed to few large fires, which is associated with a block burning strategy (Brockett *et al*., 2001). Similarly, the expectation in this study was that the block burning period would have few large burns while the PSI period would have many small burns.

The difference in the geometric character of burns (perimeter-to-area ratio) between block burns and PSI burns has to my knowledge never been examined before. The expectation, however, was that PSI burns would be more irregularly shaped than block burns since the PSI philosophy advocated that fires be left to burn out by themselves against natural barriers, as opposed to only burning in a strictly defined block, which was often bounded on one or more sides by a regularly shaped man-made feature such as a road or vehicle track.

With regard to features that prevent the spread of fire (barriers), the reserve has an extensive drainage system which consists of numerous easterly flowing seasonal streams (Goodman, 1990), while it also has an extensive road and management track network which existed as early as 1959 (Map of MGR dated 1959, originally drawn by L.C. Denyer) although some alignments may have changed. Rivers and riverine vegetation, as well as roads and tracks were thus expected to have a significant influence on fire spread. Although the difference in importance of various barrier types between the block burning and PSI strategies has, to my knowledge, never been examined before, it was expected that the natural barriers such as rivers and riverine vegetation would play a greater role during the PSI period while the man-made/management barriers such as roads would be less important during the PSI period, as the PSI strategy requires that only natural barriers prevent fire spread. One of the ways of achieving this, as advocated by reserve scientific advisory staff, was that in cases where the natural spread of fires was stopped by unnatural features, fires were to be reignited on the opposite side of the barrier (Goodman, pers. comm.).

The objective of this chapter was therefore to determine whether, and to what extent, the following fire-related variables differed between the block burning and PSI burning strategies:

i) Fire frequency.
ii) Total area burnt per annum and the size of individual burns.
iii) Geometric character of burns in the form of the perimeter-to-area ratio.
iv) Seasonal distribution of burns.
v) Burn intensity patterns.

vi) The relative importance of various barrier types, with particular emphasis on natural versus management or man-made barriers.

6.2 METHODS

In general, the approach of this investigation involved sorting the data into two sets, namely one data set representing the block burning period (1963 to 1984) and the other representing the PSI burning period (1985 to 1999). The different variables were then compared between the two periods. Statistical analyses were undertaken using SIGMASTAT and spatial analyses were undertaken using IDRISI or CARTALINX, except where indicated otherwise.

6.2.1 Difference in fire frequency between the different burning strategies

Fire frequency images were created for the 22-year block burning period and the 15-year PSI period, using the OVERLAY module in IDRISI. Because these two images represented fire frequency for different periods, they were standardised to represent the number of fires per decade. These images were then analysed using the HISTO module in IDRISI to produce histograms of fire frequency (number per decade) and to determine the mean and mode fire frequency for each of the periods. The median fire frequency for each of the burning strategies was determined after the cell values of the fire frequency images were exported to a tabular database.

6.2.2 Difference in burn size between the different burning strategies

For reasons of data distribution, the difference in total area burnt per annum, as well as in individual burn size, between the two fire regimes was examined using Mann-Whitney rank sum tests. Where the medians were not different, a Chi-square test was used to compare the data distributions. The comparison of individual fires excluded those that overlapped the study area boundary or occurred outside of it. These fires were mainly from the period after 1984 when the Nxwala state land and the other properties were incorporated into the reserve.

6.2.3 Difference in the geometric character of burns (perimeter-to-area ratio) between different burning strategies

The ratio of perimeter (km) to area (ha) was determined for individual burns. Only data for fires that occurred within the original reserve boundary were used in this analysis, while those that overlapped the original boundary were excluded. For reasons of data distribution, a comparison of the ratio values for the block burning and PSI periods was undertaken by means of a Mann-Whitney rank sum test. The regression coefficients for the relationship between area and perimeter of individual burns were also compared between the two periods using a t-test as described by Zar (1984).

6.2.4 Difference in seasonal distribution of burns between different burning strategies

For this analysis, records of fires that affected the original reserve were used, that is, those that occurred within, as well as those that overlapped the original reserve boundary. Furthermore, only those records with a date and cause of fire could be used. The total number of records used was thus 297, with 184 for the PSI period and 113 for the block burning period. These data were then sorted as per cause and season (wet or dry season), and represented graphically, as proportions of the total number of fires in each of the respective periods. The wet season was defined as all months with a mean precipitation of >50 mm (Knoch and Schulze, 1957; Goodman, 1990) (see Section 3.3.1) and therefore refers to the period from October to March, while the dry season is the period from April to September. Chi-square or Fisher Exact Tests were undertaken, where appropriate, to compare the difference in occurrence of wet and dry season burns over the block and PSI periods, for each type of burn.

6.2.5 Difference in burn intensity patterns between different burning strategies

A description of the fire intensity categories and the method of recording fire intensity data have been given in Chapter 5. The fire intensity rating, which was unfortunately not always recorded, provided an indication of the relative intensity of a fire. All available recorded data for the mapped fires were captured in CARTALINX. All fires for which an intensity rating had been recorded, and which occurred entirely or partially within the original reserve boundary, were considered in this analysis. It was established that the intensities of about 34% of these fires were likely to have been affected by winter rainfall (above dry season mean of 150mm in that year), but it was not possible to exclude these due to a limited sample size. The number of each type of burn and their percentages were compared between the block and PSI periods. A chi-square test was used to determine whether burning strategy influenced intensity, while z-tests were used to determine whether there was a significant difference in burn intensity patterns between the two burning strategies.

6.2.6 Difference in importance of various barrier types between different burning strategies

A random sample of 30 individual fires for each of the block burning and PSI periods were selected from fires that occurred inside the original reserve. This sample represents approximately 18% of such fires.

An a priori formulation of potential barrier types was made. These included: (i) Game survey transects (cleared paths); (ii) Roads and tracks; (iii) Fire-breaks and fence-lines; (iv) Rivers and riverine vegetation; (v) Other vegetation that does not usually carry a fire (e.g. Sand Forest); (vi) Wetlands; and (vii) Other fires. In addition, "unknown" was used for undetermined sections of fire boundaries, which did not include any of the recognised categories. For the boundary of each fire, the length of each section of the different barrier types was calculated as a proportion of the total perimeter of the fire. Proportions were used in order to accommodate for the differences in fire sizes.

For each barrier variable, the differences between block and PSI (each n=30) periods were examined with Fisher exact tests for frequency of occurrence, with Mann-Whitney rank sum tests for proportion of perimeter (including zero values), and, depending on data properties, with a Mann-Whitney rank sum or a t-test using arcsine-transformed data for proportion of perimeter (excluding zero values).

6.3 RESULTS

6.3.1 Difference in fire frequency between the different burning strategies

Fire frequency during block burning was generally low with much of the area having no fires per decade or in the 22-year period (Figure 6.1a). The median number of fires during this period was 0.5 per decade (one fire in 20 years). Only a small proportion of the study area experienced frequencies of one fire in approximately two years, as can be seen by the maximum number of fires per decade (4.5) (Figure 6.1a). The median number of fires per decade for the PSI period was 2.0, that is, a frequency of one in five years (Figure 6.1b). Some areas, although relatively small, experienced frequencies of almost one per annum during this period, as can be seen by the maximum of eight per decade (Figure 6.1b). There has thus been a clear increase in fire frequency from the block burning period to the PSI period. The range of frequency was wider for the PSI period (zero to eight fires per decade) as compared to the block burning period (zero to 4.5 fires per decade). This was not surprising as fire patterns are expected to be more heterogeneous for a PSI burning strategy than for a block burning strategy (Brockett *et al*., 2001).

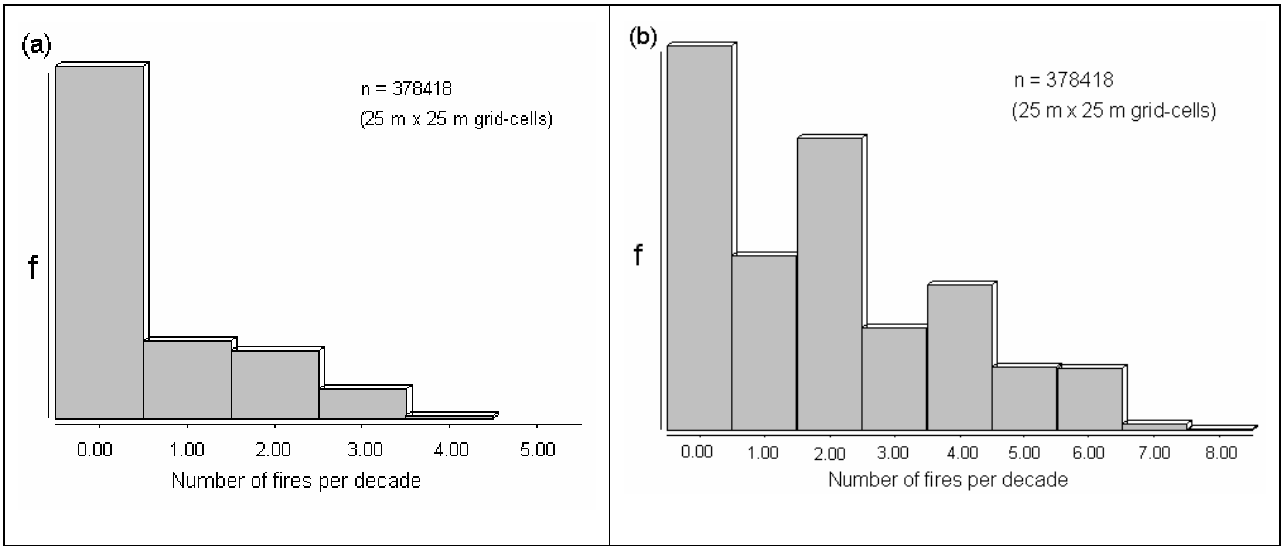


Figure 6.1: Comparison of fire frequency (number per decade) between (a) the block burning period (1963-1984) and (b) the point source ignition (PSI) period (1985 to 1999). Summary statistics for (a) are: Minimum = 0; Maximum = 4.5, Mean = 0.9; Mode = 0; Median = 0.5 Standard deviation = 1.0; d.f. = 378417; while for (b) are: Minimum = 0; Maximum = 8; Mean = 2.2; Mode = 0; Median = 2.0; Standard deviation = 1.8; d.f. = 378417.

6.3.2 Difference in burn size between the different burning strategies

The total area burnt per annum was greater for PSI strategy (median=5976 ha; n=15) than for block-burning strategy (median=1629 ha; n=22) ($P<0.008$) (Figure 6.2). This was in line with the fire frequency results. The median sizes of individual burns, 166 and 155 ha for block-burning (n=126) and PSI burning (n=211) respectively, were however, not significantly different ($P>0.13$) (Figure 6.3). In addition, the individual burn size distributions for the two periods were also not significantly different (χ^2=6.16; d.f.=9; $P>0.7$). This was unexpected as patch mosaic burning should result in numerous smaller burns as compared to block burning where fewer larger burns should result (Brockett *et al.*, 2001).

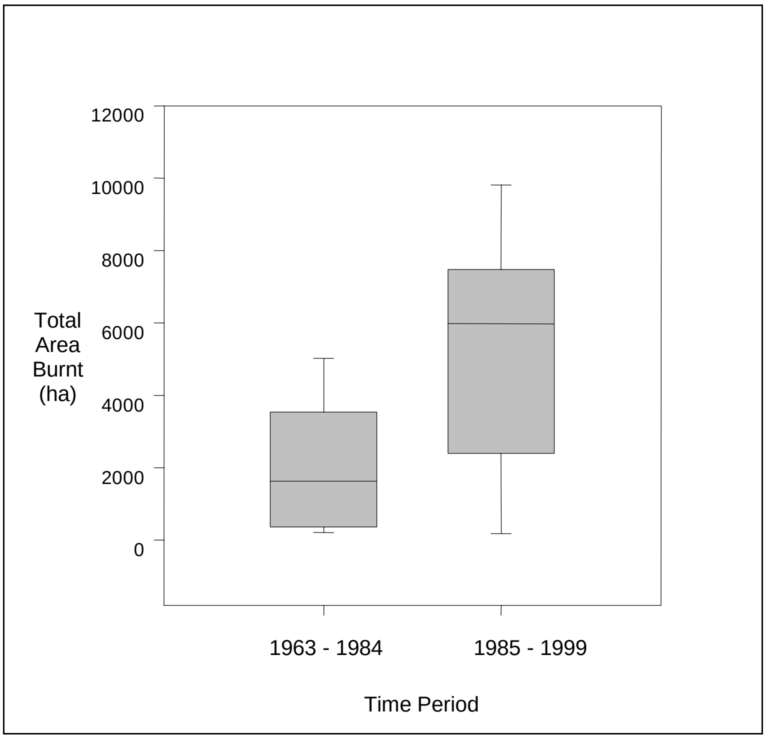


Figure 6.2: Comparison of total area burnt per annum between the block burning period (1963 to 1984) and the point source ignition period (1985 to 1999).

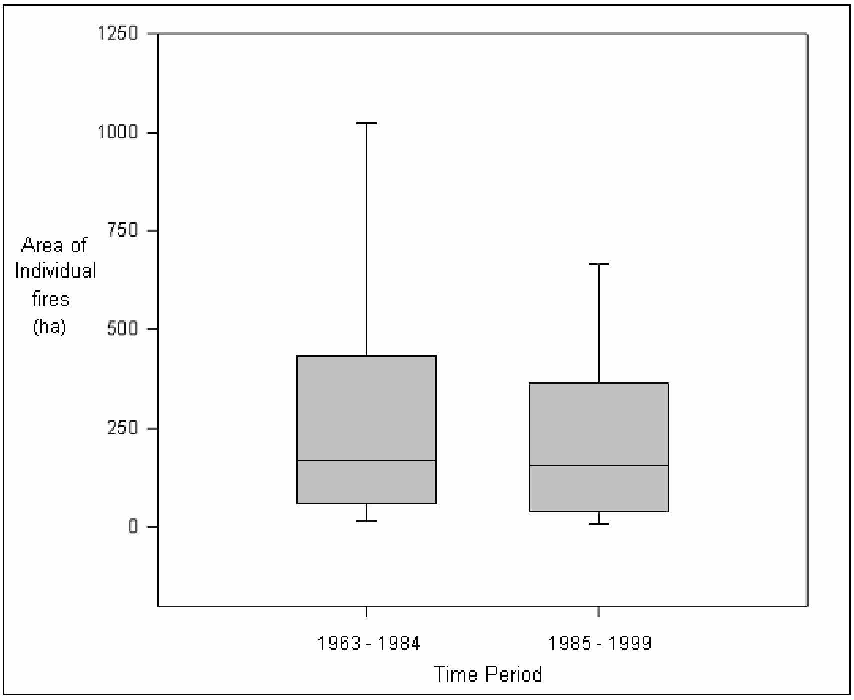


Figure 6.3: Comparison of individual burn sizes between the block burning period (1963 to 1984) and the point source ignition period (1985 to 1999).

6.3.3 Difference in geometric character of burns (perimeter-to-area ratio) between different burning strategies

Burns had a greater perimeter-to-area ratio during the PSI period than during the block burning period ($P<0.03$; Mann-Whitney rank sum test), with median values of 0.0480 km/ha (n=211) and 0.0415 km/ha (n=126), respectively (Figure 6.4). The sizes of burns can affect the edge to area ratio, but in a previous analysis, it was shown that individual burn size did not differ significantly between the two periods. This therefore indicates that PSI burns probably had boundaries that were more irregular than those of block burns, which is consistent with the PSI strategy. A larger perimeter versus area regression coefficient for the PSI period (β=0.0169) (R^2=0.790; d.f.=210; $P<0.0001$) as compared to the block burning period (β=0.0123) (R^2=0.348; d.f.=125; $P<0.0001$), provided further evidence for this (t=-2.2996; d.f.=333; $P<0.05$) (Figure 6.5).

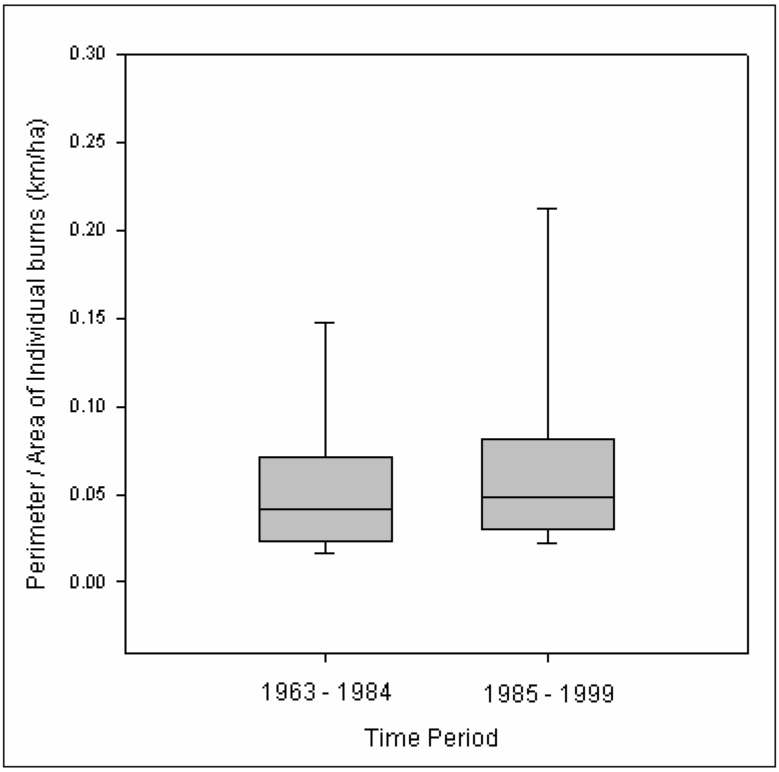


Figure 6.4: Comparison of individual burn perimeter (km) to area (ha) ratio, between the block burning period (1963 to 1984) and the PSI period (1985 to 1999).

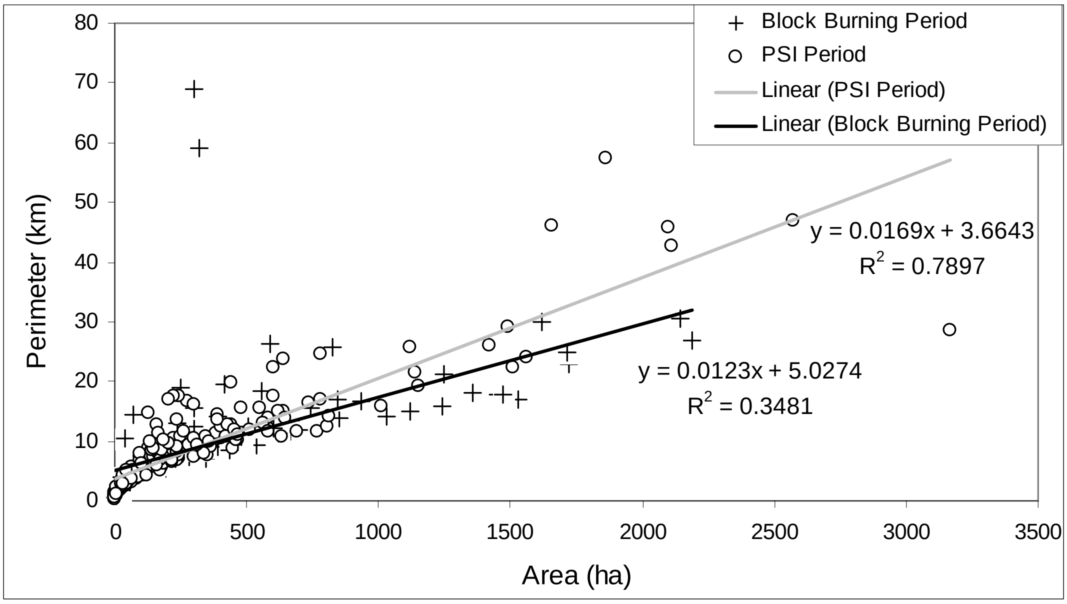


Figure 6.5: Relationship between area and perimeter of individual burns during the block burning and PSI periods.

6.3.4 Difference in seasonal distribution of burns between different burning strategies

For both block burning and PSI periods, fires were more common during the dry season than during the wet season, but as expected PSI dry season burns were more common than block dry season burns (χ^2=41.6; d.f.=1; P<0.0001). During the PSI period, 91.8% of the burns occurred during the dry season while during the block burning period, 60.2% of the burns occurred during the dry season (Figure 6.6). This overall pattern was also true for the most abundant fire types, namely prescribed burns and arson burns.

A much greater emphasis was put on applying dry season prescribed burns during the PSI period than during the block burning period (χ^2=27.9; d.f.=1; P<0.0001). Dry season prescribed burns increased from 22.1% of the burns during the block burning period to 69.0% of the burns during the PSI period, while prescribed burns that were implemented during the wet season (defined by long term rainfall patterns) decreased from 17.7% of the burns during the block burning period to only 6.5% of the burns during the PSI period (Figure 6.6).

Firebreaks were burnt only during the dry season for both the block burning and PSI periods, but made up only 0.9% of the total burns during the block burning period and later increased to 12.5% during the PSI period (Figure 6.6). A greater effort was thus made during the PSI period to burn firebreaks. Although this may appear to contradict the PSI philosophy, these firebreaks were created mainly along the boundaries of the reserve, the purpose of which was probably partly to prevent arson fires from entering the reserve. The lower proportion of arson fires recorded during the PSI period (10.9%) as compared to the block burning period (54.0%) supports this argument. In addition, the Forest Act (Act 122 of 1984) would have prompted managers to be more vigilant about containing prescribed fires within the reserve.

Arson fires were more common during the block burning period than during the PSI period for both the wet and dry seasons, while under both burning strategies, they were more common during the dry season than the wet season (Fisher Exact Test; $P<0.05$). During the block burning period, dry season arson burns made up 35.4% of the burns and wet season arson burns made up 18.6% of the burns, while during the PSI period, dry season arson burns made up 9.8% of the burns and wet season arson burns made up 1.1% of the burns (Figure 6.6).

Lightning fires, which are independent of burning strategy used, occurred during the wet season only, and a total of only three such fires were recorded for the entire study period, all of which were less than 50 ha in extent. Records for the two smallest of these fires (21 ha and 2 ha) indicated that they occurred during rain, one of which was recorded to have been extinguished by the rain soon after it started. Interestingly, the fuel condition ranged from slightly green in the largest (49 ha), to very green in the smallest (2 ha).

(a) (b)

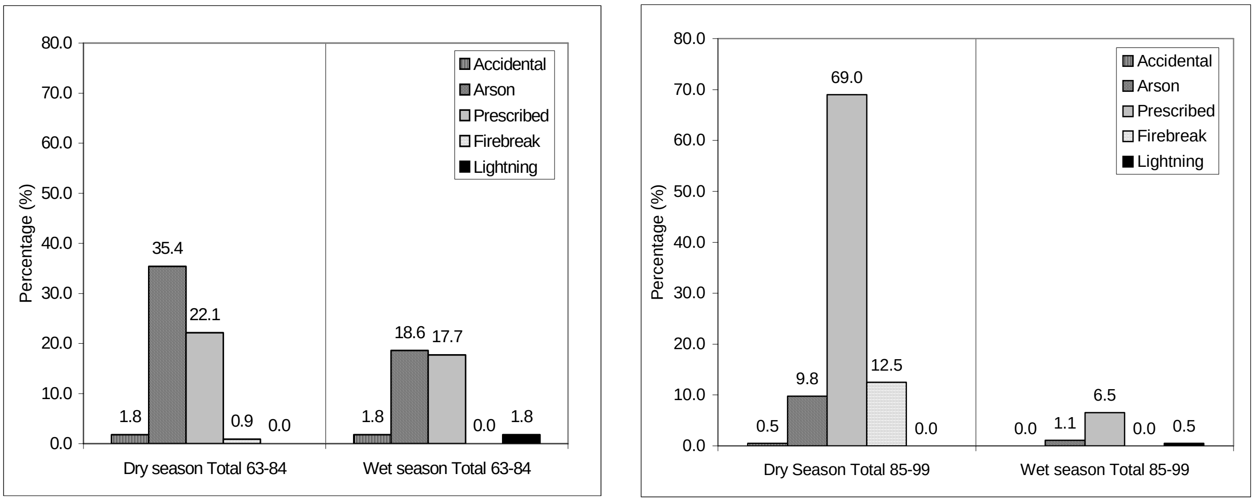


Figure 6.6: Comparison of fires during the (a) Block Burning and (b) PSI periods, according to season and cause.

6.3.5 Difference in burn intensity patterns between different burning strategies

For the 22-year period during which block burning was practised, intensity ratings were recorded for only 32 (23.4%) of the total of 137 fires that affected the study area. For the 15-year PSI period, 155 (69.5%) of the total of 223 recorded fires that affected the study area, had a recorded intensity rating.

There was no association between burning strategy and fire intensity, that is, burning strategy did not influence the intensity of burns (χ^2=4.97; d.f.=3; P>0.17). The proportion of intense burns was not significantly different between the PSI and block burning periods (z=1.71; P>0.08) and so were the clean burns (z=1.45; P>0.14), the patchy burns (z=0.0454; P>0.96), and the very patchy burns (z=0.126; P>0.9) (Table 6.1). The more intense fires (rated as intense and clean) were more common for both burning strategies, making up more than 60% of the burns in either the block or PSI burning period. The lack of difference in burn intensity patterns between the two strategies is possibly a result of burning to achieve the other fire management goals that did not change over the two periods, for example, combating bush encroachment, which requires more intense burns. This would also explain why the more intense burns were more common for both burning strategies.

Table 6.1: The numbers and percentages of types of burns achieved for the block burning period and the PSI period

Period	**Intense**	**Clean**	**Patchy**	**Very Patchy**	**Total**
Block Burning: 63 - 84	3	17	10	2	32
Percentage	9.38	53.13	31.25	6.25	100
PSI: 85 – 99	39	58	52	6	155
Percentage	25.16	37.42	33.55	3.87	100

6.3.6. Difference in importance of various barrier types between different burning strategies

Barriers were identified for 66% of the total accumulated perimeter (does not refer to the average proportion) of the 60 sampled fires. Other than unknown, the rank order of importance of barrier types for block burning was rivers and riverine vegetation, firebreaks and fencelines, and roads and tracks, whilst other vegetation, wetlands, other fires, and transects each contributed less than five percent and together contributed only eight percent (Table 6.2). For the PSI period, the rank order of importance was roads and tracks, rivers and riverine vegetation, and other vegetation, whilst transects, firebreaks and fencelines, wetlands, and other fires each contributed less than five percent and together contributed 11% (Table 6.2).

The contribution to barriers, with a change from block to PSI burning, increased for transects, roads and tracks, and other vegetation, decreased for rivers and riverine vegetation, and remained unchanged for fire breaks and fencelines, wetlands, and other fires (Table 6.2). The increased contribution of transects with a change to PSI burning was expected because, based on available records held at the reserve, these were only established in 1978. The increased contribution of roads and tracks (approximately 2.5 times greater during the PSI period based on average proportion data) and decreased contribution of rivers and riverine vegetation (approximately half that of the block burning period based on average proportion data) was unexpected because it was in contradiction of PSI management philosophy, whereas the increased contribution of other vegetation was in accordance. However, neither the combined contribution of management barriers (transects, road and tracks, firebreaks and fencelines) nor of natural barriers (rivers and riverine vegetation, wetlands, other vegetation, other fires) changed ($P>0.3$; t-test and Mann-Whitney rank sum test, respectively) from block burning to PSI burning. The proportional contribution of natural barriers (0.375) was marginally greater than that of management barriers (0.246) ($t=1.98$, d.f.=58, $P<0.053$) for block burning, but this was not evident for the PSI period (proportions 0.337 and 0.299, respectively; $P>0.4$).

The increased contribution by roads was possibly because fires were not always re-ignited on the opposite side of roads according to the PSI strategy (Llewellyn, pers. comm.). Keet (pers. comm.) estimated that, while he worked in the reserve, in at least about 40% of the cases where fires should have been re-ignited on the opposite side of a road, they were not, mainly for practical reasons. In addition, over time some roads have been widened, gravelled or tarred in places and this could have increased the effectiveness of these features as fire barriers. The increased contribution by other vegetation could possibly be due to bush encroachment in the reserve (Goodman, 1977; Bell and Van Staden, 1993). Although the decreased contribution of rivers and riverine vegetation is difficult to explain, one possible explanation is that increasing densities and distribution in the riverine areas of the invasive alien plant *Chromolaena odorata* (cover density was less than 1% in 1986 and between 1% and 50% in places in 1991) (Campbell, 1991), which is highly flammable even when green (Bromilow, 1995), has reduced the effectiveness of riverine vegetation as a barrier to fire. It is possible that this could have occurred at least in the drier upper reaches of drainage lines, but this effect can only be confirmed with future monitoring. Indeed, in Hluhluwe-Umfolozi Park, fire burnt into a forest that was heavily infested with *Chromolaena odorata* (Van Rensburg, pers. comm.) and which would otherwise have been resistant to fire.

Table 6.2: Relative importance of different fire barriers during the block burning and PSI periods

	Management barriers			Natural barriers				
Period	Transects /Paths	Roads /Tracks	Firebreak /Fenceline	Rivers /Riverine Vegetation	Other Vegetation	Wetlands	Other Fires	Unknown
Average Proportion								
Block Burning	0	0.10	0.15	0.30	0.02	0.02	0.04	0.38
PSI	0.01	0.25	0.04	0.16	0.12	0.03	0.03	0.36
Frequency of Occurrence								
Block Burning	0	13	13	27	10	4	4	-
PSI	5	22	6	16	18	7	4	-
Fisher exact P-value	0.0261	0.0352	0.0946	0.0034	0.0692	0.5062	1	-
d.f.	1	1	1	1	1	1	1	
Median proportion								
Block Burning	0	0	0	23.17	0	0	0	-
Block Burning-N	30	30	30	30	30	30	30	-
PSI	0	23.67	0	6.7	5.64	0	0	-
PSI-N	30	30	30	30	30	30	30	
Mann-Whitney P-value	0.268	0.0046	0.0671	0.015	0.00898	0.518	0.994	-
Mean Proportions (Arcsine transformed data)								
t-value	-	-1.86	1.61	-	-	0.231	0.343	-
t-test P-value	-	0.0719	0.1263	-	-	0.8221	0.7434	-
d.f.	-	33	17	-	-	9	6	-
Median Proportions (Arcsine transformed data)								
Block Burning	-	-	-	0.558	0.226	-	-	-
Block Burning-N				27	10	-	-	-
PSI	-	-	-	0.558	0.346	-	-	-
PSI-N				16	18	-	-	-
Mann-Whitney P-value	-	-	-	0.99	0.00467	-	-	-

6.4 DISCUSSION

6.4.1 Difference in fire frequency patterns between the different burning strategies

Fire frequency was about four times higher after the change from block burning to the PSI method after 1984, with a change in median fire frequency from approximately one fire in 20 years to one in five years. This increase is partly attributed to PSI fires not being limited to a specific block but allowed to spread naturally, burning any area that had sufficient fuel. It is also possible that increasing awareness over time by reserve managers of the importance of fire in maintaining savanna ecosystems could have directly resulted in more frequent use of fire in the latter years of the period under review. Indeed, data from this study show that about 75% of burns during the PSI period were prescribed while during the block burning period, only about 40% were prescribed. An important goal of PSI burning is to produce a fire regime that varies across the landscape (Brockett *et al*., 2001), and this would include fire frequency. The wider range of fire frequency in the PSI burning period is indicative of a more heterogeneous fire regime and is thus in line with the objectives of the PSI burning strategy.

6.4.2 Difference in physical characteristics of burns between the different burning strategies

A larger area was burnt annually during the PSI period than during the block burning period, while individual burns were of similar size for the two periods. The trend in total area burnt was consistent with the higher fire frequency experienced during the PSI period, and further highlights the greater effort put into burning during the PSI period. However, the lack of difference in individual burn sizes was unexpected as the PSI strategy generally aims to achieve many smaller burns as opposed to a few large burns. This was most probably related to the seasonal distribution of fires during the PSI period where, although the dry season was favoured for prescribed burns, records showed that most of these were ignited during mid to late dry season (July to September). At this stage in the dry season, grass fuel would have been well cured and thus more likely to have supported large fires. Conversely, smaller fires and a fine scale mosaic could have been achieved if some fires were ignited early in the dry season when fuels were not fully cured (Brockett *et al*., 2001).

Unlike the size of individual burns, the general shape of the burns differed between the two burning strategies. The PSI burns had more irregularly shaped boundaries as shown by the generally higher perimeter-to-area ratios. This was consistent with the PSI strategy where fires are left to burn naturally, with as little disruption as possible from linear man-made features such as roads and management tracks. Because of the more irregularly shaped PSI burn boundaries, a mosaic of areas with different fire disturbance histories (frequency, intensity, etc.), at least at a coarse scale, would more likely be achieved under the PSI burning regime than under the block burning regime in the reserve. Such a mosaic, albeit coarse, should favour biodiversity (Trollope, 1984a; Goodman, 1990).

The percentage of records from the PSI period for which an intensity rating had been recorded, as compared to that of the block burning period, clearly showed that there was a greater record-keeping effort in the latter years of the study period. Nevertheless, the subset of data used shows that the intensity patterns did not differ significantly between the two burning strategies, and also suggests that "hot" fires (those rated as intense or clean) have always been more common than "cool" fires (those rated as patchy or very patchy) during the entire study period. Although it was recognised that growing conditions at the time of burns would have influenced intensity, it was not possible to account for this due to limited detail on fires, particularly for the block burning period. It should be noted that there are a number of specific objectives of using fire in savanna areas, including MGR, that are not necessarily dependent on the overall burning strategy used, and which may affect fire patterns. Combating woody plant encroachment (Edwards, P., 1984; Trollope, 1984a; Bond and Van Wilgen, 1996) is probably the main example, which was applicable during both the block and PSI burning periods in MGR and would have required relatively intense fires (Trollope, 1984a). It is burning for this purpose that is likely to have had an influence on the intensity patterns observed in this study.

6.4.3 Difference in seasonal distribution of burns between different burning strategies

Although the two periods were similar in that a greater proportion of the fires occurred during the dry seasons, they differed in that during the PSI period, the proportion of fires was more skewed towards the dry season, as compared to that of the block burning period. This was not unexpected because with a change to the PSI strategy, burning throughout the dry season was encouraged, a practice which would have more closely resembled the historical (Iron-age) man-induced fire regime (Goodman, pers. comm.). As can be seen with the prescribed burns and firebreaks during the PSI period, there appears to have been a more concerted effort by reserve staff to burn during the dry season, avoiding burning later than the end of

September (MGR Management Plan, 1986). This was also in line with the understanding that burning while the grass layer is dormant prevents adverse impacts on grass yield and species composition (West, 1965; Van Wyk, 1972; Trollope, 1984a; O'Connor, 1985), and that the end of the dry season (late winter/early spring) is probably the best time to burn to combat bush encroachment (West, 1965; Trollope, 1984a), as fire intensity would be maximised. Furthermore, during the block burning period, fires would generally have been implemented after >15 mm of rain was received in 24 hours (first spring rain) and this constraint may have prevented burning until after September, thus accounting for the relatively high proportion (17.7%) of prescribed burns occurring during the wet season, which is defined here by long-term rainfall patterns.

With regard to the non-prescribed fires, more arson fires have been set during the dry seasons, many of which were ignited outside the reserve and spread into it. These fires are started presumably to stimulate green growth for cattle outside of the reserve, during this time when the amount of good grazing is scarce. Some reserve managers, however, have felt that these fires have been started as a ploy to attract game closer to the boundary where they can be poached more easily. The apparent greater effort to burn firebreaks, coupled with the more frequent use of prescribed burns during the PSI period may have contributed to the lower proportional incidence of arson in the reserve during this period. Lightning fires, on the other hand, are completely uncontrolled and only occur during the wet season when storms are common. Such fires have not been recorded often in the study area and when they did occur, were very small due to being influenced by rain. Similarly in Hluhluwe-Umfolozi Park, lighting fires have been shown to be unimportant (Berry and McDonald, 1979; Edwards, D., 1984; Van Wilgen *et al.*, 1990).

6.4.4 Difference in importance of various barrier types between different burning strategies

In a medium-sized conservation area (<400km^2) such as MGR, natural and management barriers were apparently of equivalent importance. The contribution of natural barriers did not increase, and that of management barriers did not decrease, with a change from block burning to a PSI strategy, as would have been expected. To my knowledge this is the first insight of relative importance of natural versus man-made or management barriers in curtailing spread of fire. This interpretation could be reviewed were it to materialise that the unknown category belonged predominantly to one or the other of these barrier types.

The importance of roads as barriers during the PSI period indicates that the PSI strategy is functioning within artificial blocks partly delineated by roads. There appears to have been a failure to ensure a continuation of fire across roads in accordance with management philosophy, which can be rectified by re-igniting fires on the opposite side of roads when they burn out against them. This is particularly important currently as roads may have become more effective barriers as a result of widening and surfacing over time. Fencelines and firebreaks, which were management barriers of minor importance during the PSI period, are legal requirements that cannot be manipulated.

The management goal of maintaining natural processes, of which fire is an important component, is clearly being compromised as expectations of a change from block burning to PSI burning are not being met in terms of barriers. This also raises the question of the effectiveness of the PSI strategy for other objectives, which to my knowledge, have never been critically assessed. There is therefore a need to look at the biodiversity implications of a change from block burning to a PSI strategy.

CHAPTER 7: GENERAL DISCUSSION AND CONCLUSIONS

7.1 INTRODUCTION

This study comprises three main sections, namely analyses of spatial fire patterns and temporal fire patterns, and a comparison of fire patterns between a block burning and point source ignition burning strategy in Mkuzi Game Reserve (MGR). In this chapter, the value of using Geographic Information Systems (GIS) for analysing large spatial data sets as used in this study is discussed; the observed spatial and temporal patterns are discussed in relation to our current understanding of fire behaviour; the observed differences in fire patterns between the two burning strategies employed in the reserve and the possible ecological implications of these patterns are discussed; and the most pertinent management recommendations are presented.

7.2 VALUE OF METHODS

The use of Geographic Information Systems (GIS) for natural resource management has become increasingly popular as it is a useful tool for capturing, storing, manipulating, displaying and publishing geographically referenced data (Maguire *et al.*, 1991; Woods, 1997). It is thus particularly useful for landscape-scale investigations. There are a number of examples where GIS has been used for natural resource management (Kok *et al.*, 1995; Looijen *et al.*, 1995; Mulqueeny, 1995; Mallawaarachchi *et al.*, 1996; Woods, 1997) and some specifically relating to fire regimes (Gill *et al.*, 2000; Balfour and Howison, 2001; Grau, 2001). GIS was the ideal tool for this work as it provided a means to manage and analyse large volumes of spatial data relatively efficiently. The process of incorporating all the required data layers into the GIS, which included digitising the historical fire maps together with capturing the associated attribute data, was the most time consuming component of the work. The subsequent manipulation and analysis of this large data set was made easier by GIS, a process which would by any other means have been arduous, if at all possible. The results from these spatial analyses were also enhanced by the use of traditional statistical procedures, for example, the comparison of individual burn sizes between the two burning strategies, where the sizes were calculated using the GIS and compared by means of a Mann-Whitney rank sum test using a statistics computer software package. A similar approach of using GIS together with traditional statistical methods was followed by Balfour and Howison (2001).

Most of the specific methods applied in this work were formulated while undertaking the project. Although some of the methods used are similar to those of Balfour and Howison (2001), as far as I am aware, several of the analyses undertaken in this work have not been done before at a landscape scale using GIS. Examples of these analyses include the pixel by pixel regression analyses of fire frequency versus herbivore density, that of fire frequency versus rainfall, and in particular, the various comparisons between the block and PSI burning strategies. Valuable landscape-level research with some similar aspects to this study has been achieved without using GIS (Norton-Griffiths, 1979) but the sample size and resolution of the investigation was limited as compared to that of a GIS-based investigation. This project has therefore not only illustrated a novel way of analysing spatial and temporal patterns of fire, but has also further illustrated the versatility of GIS.

As useful as GIS may be, it should be noted that it is merely a tool and that output from GIS analyses are only as good as the data that was fed into the system as well as the techniques used to analyse the data. Often in error, both laymen and skilled GIS users place too much confidence in GIS information merely because of its digital nature, without critically analysing the data quality. GIS data quality can vary in terms of positional accuracy, attribute accuracy and completeness (important data excluded). It is, however, very difficult, if not impossible, to get rid of error completely in spatial data as representations are usually not a perfect replica of something as complex as the earth (Maguire *et al.*, 1991). The challenge is thus to minimise error in GIS data so that resultant products are of high quality and can be used with confidence.

7.3 Spatial and Temporal Variation of the Fire Regime

Fire frequency varied spatially in relation to vegetation type (and associated substrate), rainfall distribution and herbivore distribution. There appeared to be a general gradient from high fire frequency in the west of the reserve to lower fire frequency in the east, which corresponded with roughly east-west geological and associated vegetation gradients in the reserve (see Figures 3.4 and 3.5). Fire frequency was positively related to rainfall, which also displayed a general gradient of decreasing from south-west to north-east, while herbivores tended to limit the occurrence of fire. Temporal fire patterns showed that annual total area burnt was positively related to preceding and previous wet season rainfall but dry season rainfall had no effect. There was no relationship between annual total area burnt and annual total grazer biomass. Dry season (individual) burns were larger than wet season burns. Fire intensity had an effect on the size of individual burns with those rated as "intense" being larger than the less intense burns. The amount of herbaceous material available as fuel and the condition of the fuel were the key factors in controlling fire patterns and thus the way in which the different variables affected the fuel load and its condition is what determined the spatial and temporal variation of the fire regime.

Geology and vegetation type are related in MGR (Goodman, 1990) while the amount of herbaceous material varies per vegetation type (Moll, 1968, 1980; Goodman, 1990). Herbaceous material forms the main component of fuel in savanna fires, which are mainly surface fires (Trollope, 1984a), and thus it was no surprise that fire frequency varied according to both vegetation and geological type. Geological type, however, did not provide any more insight into the spatial variation of fire patterns than vegetation type did because of their association. Those vegetation types that had well developed herbaceous layers had higher fire frequency than those with poorly developed herbaceous layers. In addition, the soil moisture regime on the floodplain and wetland areas likely affected the plant moisture content and consequently fire frequency there was relatively low.

Spatial and temporal fire patterns were directly related to wet season rainfall. Fire frequency was positively correlated with rainfall variation across the study area, while the total annual area burnt was directly correlated with the preceding wet season's rainfall, albeit only significantly for years with low dry season rainfall. This can be attributed to the direct positive influence of rainfall on herbaceous production (Dye and Spear, 1982; Briggs and Knapp, 1995; O'Connor *et al.*, 2001) and thus the amount of available grass fuel. In addition, the previous wet season rainfall, together with the preceding wet season rainfall, was positively correlated with total annual area burnt. Although this relationship was only significant for years when dry season rainfall was low, it did nonetheless indicate that there was possibly a carry over effect of fuel from one wet season to the next. This was in line with recent findings in Hluhluwe-Umfolozi Park where preceding and previous summer season rainfall had a significant effect on total area burnt in any one year, which was explained as a carry-over effect of fuel from one year to the next (Balfour and Howison, 2001). It is also possible that there could be a carry-over of plant available moisture from one year to the next, which would affect grass production (Hanson *et al.*, 1982; Smoliak, 1986; O'Connor *et al.*, 2001) and thus fuel load in the year following a good rainfall year. The carry over effect is expected to differ between different soil types because texture affects infiltration, water-holding capacity, the water-loss mechanism (percolation or capillary movement) and the amount of moisture available to plants (Dye and Spear, 1982; Frost, 1987; Walker and Langridge, 1997). Furthermore, recharge and water holding capacity are greater on non-degraded soils than on degraded soils (Seiny-Bouker, Floret and Pontanier, 1992). The carry-over effect was not investigated separately for the different soil types in this study but does warrant some attention in future. Although dry season rainfall was expected to negatively influence total annual area burnt (Norton-Griffiths, 1979) because higher than normal dry season rainfall results in greening of the grass (less combustible) due to active growth (San José and Medina, 1976; Medina and Silva, 1990), dry season rainfall had no influence on the extent of burning in MGR. In general, the conditions during the dry season are most conducive to fires in that grass fuel is dormant (Trollope, 1984a) and dry, containing much less moisture than during the growing phase (Luke and McArthur, 1978). In addition, low humidity favours the occurrence of fire (Bond and Van Wilgen, 1996), while wind affects the spread of fire (Luke and McArther, 1978; Trollope, 1984b). Both low humidity and windy conditions are more common during the dry season in the reserve than during the wet season (Goodman, 1981; Lewis and Goodman, 1993), and therefore the finding that fires during the dry season were significantly larger than wet season fires, was consistent with our current understanding of fire behaviour.

The amount of grass fuel can be reduced by grazing herbivores through direct defoliation and trampling (Tainton, 1981; Heady and Child, 1994) and indirectly through impact on primary production (Milchunas and Lauenroth, 1993; Illius and O'Connor, 1999), and in this study the negative relationship between fire frequency and herbivore density over the reserve provides further evidence for this. It was, however, beyond the scope of this work to determine to what extent each of the direct and indirect factors influenced the reduction in fuel load. In essence, it was found that grazers had a limiting effect on fire frequency over the study area. This relationship, however, was not evident for the temporal analysis and a possible reason for this was the small sample size which was limited by the number of large herbivore population surveys conducted over the study period. The limiting effect of herbivores on fire occurrence can be illustrated simply by the following reasoning. In areas with low preceding wet season rainfall, grass production will be low, and even if herbivore density is low, the probability of fire occurring is expected to be low because of a lack of fuel. In areas with higher preceding wet season rainfall, the grass production is expected to be high, but if herbivore density and consequently utilisation is high, then the probability of fire occurring will again be low because of a lack of fuel. Rainfall (wet season rainfall) is thus the main controlling factor while herbivore density is a limiting factor.

Fire intensity was positively related to the extent of burns, which was shown by the significantly larger size of fires that were rated as intense compared to all other fires, although differences in burn size could not be distinguished between the two lower intensity classes used, namely clean and patchy/very patchy. This trend was consistent with evidence from Kakadu National Park in Australia where the proportion of transects burned in plots increased with increasing fire intensity (Williams *et al.*, 1998; Gill *et al.*, 2000). This pattern was also consistent with this study's findings on the influence of rainfall and season on the extent of burns, and our understanding of the influence of rainfall and season on fire intensity. Rainfall directly influences the growth of herbaceous material (Dye and Spear, 1982; Briggs and Knapp, 1995; Fynn and O'Connor, 2000; O'Connor *et al.,* 2001) and thus the amount of available fuel, which together with the rate of spread determines fire intensity (Byram, 1959; Brown and Davis, 1973; Luke and McArthur, 1978; Trollope, 1984b; Bond and Van Wilgen, 1996). With regard to the season of burn, grass is dormant and cures (senescence takes place) during the dry season, containing much less moisture than during the growing phase (Trollope, 1984a; Luke and McArthur, 1978); this lower fuel moisture would favour more intense fires (Brown and Davis, 1973; Trollope, 1984b). The greater total extent of burns which resulted from years with high preceding wet season rainfall and the larger individual burns that occurred during the dry season were therefore in line with the larger burns that occurred as a result of more intense fires. Considering the relationship between fire intensity and the extent of burns, and that the goal of combating bush encroachment requires intense fires that ensure a greater topkill of bush (Trollope, 1984a) while that of maximising vegetation heterogeneity requires many small fires (Brockett *et al.,* 2001), it may appear that a conflict between the fire management goals exists. However, this is not necessarily the case because smaller intense fires are possible, particularly if some fires are ignited early in the dry season when the grass is not fully cured and the potential for these fires to be large is low. These early fires serve to "break up" the fuel and in this way, the spread of late dry season fires, which are likely to be intense, would be hindered thus limiting their size (Brockett *et al.*, 2001).

7.4 Block Burning vs. Point Source Ignition Strategy

The PSI burning strategy aims to create a fire mosaic of areas burnt by different types and intensities of fire occurring at various times and frequencies, which should in turn, maintain a diversity of habitat and species. This situation, according to Trollope (1984a), closely resembles the natural fire regime in savanna. In order to create fire-derived mosaics and maintain spatial heterogeneity through prescribed burning it is important to vary the fire parameters (frequency, seasonality, intensity and type of fire) spatially and temporally across the landscape and this should mimic the prehistoric and historic fire patterns (Brockett *et al.,* 2001). In this study, the comparison of fire patterns between the two burning strategies shows that the objectives of point source ignition (PSI) burning, at least in terms of the desired fire regime, have only partly been achieved.

Fire has been applied much more frequently (median frequency of one fire in five years) during the PSI burning period, and this is also reflected in the much larger total area burnt per annum than during the block burning period. This can possibly be attributed to the nature of PSI burning which allows fires to spread and burn out naturally against natural barriers without being restricted to certain areas, and also possibly to an increasing awareness amongst conservationists of the value of fire in savanna ecosystem management. The results from this study certainly showed that there was a greater effort during the PSI burning period to implement prescribed burns in the reserve. Prescribed burns made up about 75% of the burns during the PSI burning period and only about 40% during the block burning period, while firebreaks also increased from about 1% to about 13% of the burns with a change from block burning to PSI burning. The increased percentage of firebreaks showed that a much greater effort was made to control the spread of fire across the reserve boundaries than during the block burning period, and this was probably partly due to the legal controls that were put in place by the Forest Act (Act 122 of 1984). This increased effort was obviously successful because the proportion of arson fires, which mostly occurred during the dry season, was lower during the PSI burning period than the block burning period.

Contrary to an expected decrease in burn size with a change from block to PSI burning, the implementation of a PSI strategy apparently made no difference to the size of individual burns (median burn size of *c.*160 ha). This could possibly be seen as a shortcoming in terms of achieving a fine-scale fire mosaic, particularly when comparing the median size of burns in MGR to other areas where patch mosaic burning has been implemented such as Pilanesberg National Park (median burn size *c.* 50 ha) and Kakadu National Park in Australia (median burn size *c.* 60 ha) (Russell-Smith *et al.*, 1997; Brockett *et al.*, 2001). However, the situation in MGR appears quite favourable when compared to Hluhluwe-Umfolozi Park where a PSI strategy was also implemented from the mid-1980's and where the median size of individual burns for a wet phase (1984-1991) was 410 ha and a dry phase (1992-1995) was 320 ha (Balfour and Howison, 2001). It should be noted, however, that the difference in size of burns can also be partly attributed to the difference in the nature of the landscape among different areas. An area with more natural barriers, for example a landscape dissected by numerous streams, rivers and associated riverine vegetation, will probably tend to have smaller fires than one which is less divided. It is therefore difficult to judge, in terms of the size of fires, whether the implementation of PSI burning in MGR has been more or less successful than in other areas. Considering MGR on its own, however, the lack of difference in size of burns between the two burning strategies indicates that PSI burning did not contribute to creating a finer fire mosaic than that which was achieved using block burning.

Although the size of individual burns between the two strategies was not different, there was a significant difference in the shape of burns, as indicated by the perimeter-to-area ratio. This was in line with PSI strategy objectives as it was felt that more irregularly shaped burns that occurred during the PSI burning period should promote spatial heterogeneity of fire patterns at a landscape scale. The more irregularly shaped burns were attributed to the practice of allowing fires to burn out against natural barriers, which are generally not as regularly shaped as management barriers such as roads and tracks.

The intensity patterns did not differ significantly between the two burning strategies, and also suggested that "hot" fires (those rated as intense or clean) have always been more common than "cool" fires (those rated as patchy or very patchy) during the entire study period. The specific goal of combating bush encroachment, which was applicable during both the block and PSI burning periods, has possibly contributed to this bias towards more intense fires. This pattern is also reflected in the seasonal distribution of fires over the entire study period, where a greater proportion of fires occurred during the dry season, when fires are expected to be more intense because of the dormant dry condition of the grass fuel load (Luke and McArthur, 1978; Trollope, 1984a). Dry season burns were, however, more common during the PSI burning period than during the block burning period, a difference which was not consistent with the lack of a significant difference in the intensity patterns between the two burning strategies. This suggests that other factors, in addition to season of burn, have influenced the fire intensity patterns. The lack of difference in intensity patterns does not necessarily indicate a failure to achieve the PSI strategy objectives. However, the strategy aims to achieve burns with varying intensities and because the general intensity patterns show a relatively uneven spread of fires across the intensity range, it is suggested that this possibly represents a shortcoming. It is not known what the spread of fire intensities should be in an area where PSI or patch mosaic burning is practiced, but a more even spread does seem more in line with the goal of achieving a mosaic of areas burnt under different conditions.

The seasonal distribution of burns during the PSI burning period was more in line with the principles of sound veld management practices, that is, burning while the grass is dormant (West, 1965; Van Wyk, 1972; Trollope, 1984a; O'Connor, 1985), than the seasonal distribution of burns during the block burning period. During the PSI period, more than 90% of all the burns occurred during the dry season while during the block burning period, about 60% of all burns occurred during the dry season. Furthermore, during the PSI period, most of the prescribed fires were ignited during mid to late dry season (July to September). By the middle of the dry season, the grass should be in an advanced stage of curing and would therefore support more intense fires (Luke and McArthur, 1978; Trollope, 1984a) and likely result in larger burns (Williams *et al.*, 1998). This is a possible cause of the PSI burns remaining as large as the block burns. It has been suggested that in order to achieve smaller burns and maintain a fine-scale mosaic, some fires should be ignited early in the dry season (from April) so as to break up the fuel (Brockett *et al.*, 2001). These early dry season burns do not become large because the grass fuel is not fully cured, and by breaking up the fuel, prevent large late season fires (Brockett *et al.*, 2001).

With regard to fire barriers, the PSI strategy objectives have not been achieved as there was no change in importance of natural and management barriers. The contribution of natural barriers did not increase, and that of management barriers did not decrease, with a change from block burning to a PSI strategy, as would have been expected. Furthermore, the great importance of roads and tracks as barriers during the PSI burning period indicated that the PSI strategy was functioning within artificial blocks partly delineated by roads, and represents a failure to implement the strategy properly, particularly with regard to re-ignition of fires on the opposite side of roads when they have burnt out against them.

It is clear from the above assessment that despite a change from block burning to PSI burning, the spatial patterns that would promote heterogeneity in the landscape were not being fully achieved, although there has been a vast improvement since the change in burning strategy. In essence, by not achieving the desired fire-derived mosaic, the overall goal of conserving biological diversity is most likely being compromised. This, however, can only be determined for certain through an actual comparison of the state of biological diversity under the two burning strategies described here, and also rating biological diversity of these against a benchmark area where a suitable fire-derived patch mosaic has been achieved. As such a study would be an enormous undertaking it is felt that the fire patterns are a reasonable indicator of whether heterogeneity across the landscape has been achieved. Indeed, it has been shown that fire patterns, in certain cases, can be used as a surrogate for biotic diversity (Parr, 1999; Parr and Brockett, 1999).

7.5 Recommendations and Implications for Management

The management recommendations given here are based on the fire management objectives for the reserve, namely, to apply fire in a manner that will maintain or enhance spatial heterogeneity, to ensure a fodder flow to support large mammal populations, to retard woody plant growth where appropriate, and to reduce the risk of accidental or so-called arson fire that would threaten the survival of plant species or destroy the composition or structure of a priority vegetation community (MGR Management Plan, 1986, 1996). In addition, recommendations also take into account that a guiding principle in the management of this reserve has been to allow natural processes to prevail over those that are obviously artificial (MGR Management Plan, 1986, 1996).

The effort to implement prescribed burns must be maintained as observed over the PSI burning period. Although the occurrence of fire will be greatly dictated by annual rainfall patterns, all opportunities to burn should be seized, provided the timing and conditions are appropriate for achieving the desired objectives. The non-prescribed burns such as the so-called arson fires that are often ignited outside of the reserve and spread into the reserve, should continue to be considered as acceptable provided they occur during the dry season and the time since the last burn in the affected area is at least two years (biennial burning is acceptable) (MGR Management Plan, 1996). Although the reserve management plan has not considered early dry season burns as being acceptable, based on recommendations that are discussed elsewhere in this chapter to achieve smaller burns, it is suggested that these be considered as acceptable as they are likely to help in breaking up the fuel load and thereby reducing the probability of large late season burns.

Because the amount of fuel is affected by the vegetation type, preceding wet season rainfall and the grazing herbivore densities, these factors must be taken into account when the annual burning plan is being formulated. The net result of the influence of these factors will be the available fuel observed in the veld, and therefore a field inspection should be undertaken by reserve staff early in the dry season to assess the amount and condition of the potential fuel. The veld condition evaluation exercise which is normally undertaken in April each year will also complement such field inspections. It is, however, suggested that a formal fuel load assessment be incorporated as part of the veld evaluation exercise. The disc pasture meter method of estimating grass fuel load is recommended (Bransby and Tainton, 1977; Trollope and Potgieter, 1986). A calibration exercise as described by Trollope and Potgieter (1986) will initially be required but once the relationship between the disc height and fuel load has been established, annual assessments of fuel load will be relatively quick.

With regard to the way in which the PSI burning strategy is currently being implemented, two key areas of improvement have been identified. Firstly, it is recommended that some early dry season burns are ignited, which should result in an overall reduced median size of burns and thus maintain a finer scale mosaic (Brockett *et al*., 2001). There is therefore a need to start burning as early as April (these fires will be small as the grass is not fully cured), so as to break up the fuel load and prevent late season burns being large. This will also influence fire intensity patterns by increasing the proportion of low intensity fires. This approach will, however, not preclude the use of “hot” fires later in the dry season to combat bush encroachment in specific areas. The approach departs slightly from the current situation where fires are ignited and allowed to spread. Because the intention is to create small fires, a greater degree of manipulation is required through spreading the ignitions over the entire dry season and through controlling the number of burns. As this study has shown that the total area burnt in a year is directly related to preceding wet season rainfall, this relationship could be used as a guide in estimating the proportion of the reserve that should be burnt and thus the number of small burns required. Alternatively, an assessment of grass biomass at the end of the rainy season can be used to determine the proportion of the area to be burnt (Van Wilgen *et al.*, 2000; Brockett *et al.*, 2001). Secondly, it is recommended that a more concerted effort be made to re-ignite fires on the opposite side of artificial barriers such as roads and tracks when fires have burnt out against them. Furthermore, the initial ignitions for prescribed burns should not be directly alongside roads as the lower intensity in the initial build-up phase of the fire (Luke and McArthur, 1978) eventually results in almost a hedge of woody vegetation developing alongside roads (pers. obs.).

While undertaking this work, a number of possible future avenues of research have been identified which would either enhance results from this study or fill some of the gaps in our understanding of spatial and temporal variation in fire patterns. These include:

i. an investigation of the relationship between herbaceous species composition and fire frequency and intensity so as to illustrate the difference in flammability, as well as production, between different species;
ii. an investigation of whether alien invasive plants such as *Chromolaena odorata* have changed the effectiveness of natural fire barriers such as riverine vegetation; and
iii. an assessment of the carry-over effect of fuel (phytomass already produced) from one year to the next and of phytomass production (transfer of soil water and potential to produce phytomass) for different soil types.

Furthermore, all the study of landscape-level fire behaviour for conservation areas will be of limited value until we address the question of how block burning and PSI or patch mosaic burning strategies actually affect herbaceous and woody species composition and biological diversity.

Continuous monitoring of fires since 1963 has been critical for the undertaking of this investigation. Overall, the monitoring systems in the reserve are fairly well developed, and data have been recorded and managed well. The systems require little improvement except for the following suggestions which are aimed at improving data quality and management effectiveness:

i. A clearer description of what constitutes the different types of burns (in terms of intensity rating) needs to be devised. In addition, different methods of surveying a completed burn, for example, using a fixed-wing aircraft to obtain a better view of the entire burn before rating the intensity of the burn, or possibly using satellite imagery and GIS to obtain a quantitative result, should be considered. The method of using Landsat data to determine burn severity as employed in Kruger National Park (Landmann, 2003), could be considered. The use of satellite data would also improve the accuracy of burn mapping and area estimates.

ii. A system of monitoring fuel load (quantitative assessment) at the end of the rainy season, which will assist reserve staff in developing the annual prescribed burning plans, should be implemented. The fuel load assessment, using the disc pasture meter method (Trollope and Potgieter, 1986), should be done as part of the annual veld condition evaluation exercise.

REFERENCES:

Acocks, J. P. H., 1975. *Veld Types of South Africa*. Memoirs of The Botanical Survey of South Africa, No. 40. Botanical Research Institute, Department of Agricultural Technical Services, South Africa.

Balfour, D. A. and Howison, O. E., 2001. Spatial and temporal variation in a mesic savanna fire regime: responses to variation in annual rainfall. *African Journal of Range and Forage Science*, 19, 43-51.

Bell, R. H. V., 1982. The effect of soil nutrient availability on community structure in African ecosystems. In: Huntly, B. J. and Walker B. H. (eds.), *Ecology of Tropical Savannas*. Springer-Verlag, Berlin, Germany, pp. 193-216.

Bell, W. E. and Van Staden, J., 1993. Seed structure and germination of *Dichrostachys cinerea*. *South African Journal of Botany*, 59, 9-13.

Belsky, A. J., 1994. Influences of trees on savanna productivity: Tests of shade, nutrients, and tree-grass competition. *Ecology*, 75, 922-932.

Belsky, A. J., Amundson, R. G., Duxbury, J. M., Riha, S. J., Ali, A. R. and Mwonga, S. M., 1989. The effects of trees on their physical, chemical and biological environments in a semi-arid savanna in Kenya. *Journal of Applied Ecology*, 26, 1005-1024.

Berjak, S. G. and Hearne, J. W., 2002. An improved cellular automaton model for simulating fire in a spatially heterogeneous Savanna system. *Ecological Modelling*, 148, 133-151.

Berry, A. and McDonald, I. A. W., 1979. Fire regime characteristics in the Hluhluwe-Corridor-Umfolozi Game Reserve Complex in Zululand. Area description and an analysis of causal factors and seasonal incidence of fire in the central complex with particular reference to the period 1955 to 1978. Unpublished report.

Blackburn, T. M., Lawton, J. H. and Perry, J. N., 1992. A method of estimating the slope of upper bounds of plots of body size and abundance in natural animal assemblages. *Oikos* 65, 107 - 112.

Bond, W. J. and Van Wilgen, B. W., 1996. *Fire and plants*. Population and community biology series 14, Chapman and Hall, London, 263pp.

Bosch, J. M., Schulze, R. E. and Kruger, F. J., 1984. The effect of fire on water yield. In: de V. Booysen, P. and Tainton, N. M. (eds.), *Ecological effects of fire in South African ecosystems*. Ecological Studies 48, Springer-Verlag, Berlin, pp. 327-348.

Bosch, O. H. J. and Van Wyk, J. J. P., 1970. The influence of bushveld trees on the productivity of Panicum maximum: a preliminary report. *Proceedings of the Grassland Society of Southern Africa*, 5, 69-74.

Bothma, J. du P. (ed.), 2002. *Game Ranch Management*. Van Schaik Publishers, Pretoria.

Bransby, D. I. and Tainton, N. M., 1977. The disc pasture meter: possible applications in grazing management. *Proceedings of the Grassland Society of Southern Africa,* 12, 115-118.

Briggs, J. M. and Knapp, A. K., 1995. Interannual variability in primary production in tallgrass prairie: climate , soil moisture, topographic position, and fire as determinants of aboveground biomass. *American Journal of Botany*, 82, 1024-1030.

Brockett, B. H., Biggs, H. C. and Van Wilgen, B. W., 2001. A patch mosaic burning system for conservation areas in southern African savannas. *International Journal of Wildland Fire,* 10, 169-183.

Bromilow, C., 1995. *Problem plants of South Africa*. Briza Publications, Arcadia.

Brown, A. A. and Davis, K. P., 1973. *Forest fire: control and use*. McGraw-Hill, New York.

Burrows, W. H., Scanlan, J. C. and Anderson, E. R., 1988. Plant ecological relations in open forest, woodlands and shrublands. In: Burrows, W. H., Scanlan, J. C. and Rutherford, M. T. (eds.), *Native Pastures in Queensland: the resources and their management*. Queensland Department of Primary Industries Information Series Q187023, Queensland Government Printer, Brisbane, Australia, pp. 72-90.

Burrows, W. H., Carter, J. O., Scanlan, J. C. and Anderson, E. R., 1990. Management of savannas for livestock production in north-east Australia: contrasts across the tree-grass continuum. *Journal of Biogeography*, 17, 503-512.

Byram, G. M., 1959. Combustion of forest fuels. In: K.P.Davis (ed.), *Forest Fire: Control and Use.* McGraw-Hill, New York, pp. 155-182.

Campbell, J. A. E., 1991. Survey of the invasive status of *Chromolaena odorata* in Mkuzi Game Reserve, with special reference to control measures and seed rain dynamics. Unpublished report, Cape Technikon. 40pp.

Caughley, G., 1976a. Wildlife management and the dynamics of ungulate populations. In: Coaker, T. H. (ed.), *Applied Biology*. Academic Press, New York, pp 183-246.

Caughley, G., 1976b. Plant-herbivore systems. In: May, R. M. (ed.), *Theoretical Ecology*. Saunders, Philadelphia, Pennsylvania.

Caughley, G., 1977. *Analysis of vertebrate populations*. John Wiley and Sons, London. 234pp.

Clark, J. S., 1988. Effects of climate change on fire regime in north-western Minnesota. *Nature*, 334, 233-235.

Clark, J. S., 1990. Fire and climate change during the last 750yr in north-western Minnesota. *Ecological Monographs*, 60, 135-159.

Cole, M. M., 1986. *The Savannas: Biogeography and Geobotany*. Academic Press, London.

Collins, S. L. and Gibson, D. J., 1990. Effects of fire on community structure in tall-grass and mixed grass prairie. In: Collins, S. L. and Wallace, L. L. (eds.), *Fire in North American Tallgrass Prairies*. University of Oklahoma Press, Norman, pp. 81-98.

Cowling, R. M., 1987. Fire and its role in coexistence and speciation in Gondwana shrublands. *South African Journal of Science*, 83, 106-112.

Dasmann, W. P. and Dasmann, R. F., 1963. Abundance and scarcity in California deer. *California Fish and Game,* 49, 4-15.

Davies, W., 1947. The grassland problem in Southern Rhodesia. *Rhodesia Agricultural Journal*, 44, 697-708.

Dean, G. J. W., 1967. Grasslands of the Rukwa valley. *Journal of Applied Ecology*, 4, 45-67.

Dixon, J. E. W., 1964. Preliminary notes on the mammal fauna of the Mkuzi Game Reserve. *Lammergeyer*, 3, 40-58.

Drummond, W. H., 1875. The large game and natural history of South and south-east Africa. Edmonston and Douglas, Edinburgh.

Dye, P. J. and Spear, P. T., 1982. The effect of bush clearing and rainfall variability on grass yield and composition in south-west Zimbabwe. *Zimbabwe Journal of Agricultural Research,* 20, 103-117.

East, R., 1981. Species-area curves and populations of large mammals in African savanna reserves. *Biological Conservation*, 21, 111-126.

Eastman, J. R., 1998. The Idrisi Project. Clark University, Massachusetts, USA.

Edwards, D., 1984. Fire Regimes in the Biomes of South Africa. In: de V. Booysen, P. and Tainton, N. M. (eds.), *Ecological effects of fire in South African ecosystems*. Ecological Studies 48, Springer-Verlag, Berlin, pp. 19 - 37.

Edwards, P. J., 1984. The use of Fire as a Management Tool. In: de V. Booysen, P. and Tainton, N. M. (eds.), *Ecological effects of fire in South African ecosystems*. Ecological Studies 48, Springer-Verlag, Berlin, pp. 349 - 362.

Emslie, R. H., 1999. The feeding ecology of the black rhinoceros (*Diceros bicornis minor*) in Hluhluwe-Umfolozi Park, with special reference to the probable causes of the Hluhluwe population crash. Unpublished PhD. thesis, University of Stellenbosch.

Frost, P. G. H., 1987. The regional landscape: Nylsvley in perspective. *South African National Scientific Programmes Report* No.133.

Frost, P. G. H. and Robertson, F., 1985. The ecological effects of fire in savannas. In: Walker, B. H., 1985 (ed.), *Determinants of Tropical Savannas: Presentations made by savanna researchers at a workshop in Harare, Zimbabwe, December 1985.* The International Union of Biological Sciences Monograph Series, 3, 93-140.

Frost, W. E. and McDougald, N. K., 1989. Tree canopy effects on herbaceous production of annual rangeland during drought. *Journal of Range Management*, 42, 281-283.

Fynn, R.W.S. and O'Connor, T. G., 2000. Effect of stocking rate and rainfall on rangeland dynamics and cattle performance in a semi-arid savanna, South Africa. *Journal of Applied Ecology*, 37, 491-507.

Gertenbach, W. P. D., 1988. Fire as management tool in Kruger National Park. *Fire Protection*, Dec 1988, 6-10.

Gertenbach, W. P. D. and Potgieter, A. L. F., 1979. Veldbrandnavorsing in die struikmopanieveld van die Krugerwildtuin. *Koedoe*, 22, 1-28.

Gill, A. M., Ryan, P. G., Moore, P. H. R. and Gibson, M., 2000. Fire regimes of World Heritage Kakadu National Park, Australia. *Austral Ecology*, 25, 616-625.

Goodchild, M. F. and Kemp, K. K., 1990. Introduction to GIS: NCGIA Core Curriculum. National Center for Geographic Information and Analysis, University of California, Santa Barbara.

Goodman, P. S., 1977. Soil types prone to bush encroachment. In: *Proceedings of Veld and Bush Management for Beef and Game Production*, pp. 61-71. Hlabisa Soil Conservation Committee.

Goodman, P. S., 1981. A preliminary summary of the climatic features of Mkuzi Game Reserve. Unpublished N.P.B. report, Pietermaritzburg.

Goodman, P. S., 1990. Soil, vegetation and large herbivore relations in Mkuzi Game Reserve, Natal. Unpublished PhD. thesis, University of the Witwatersrand, Johannesburg.

Goodman, P. S. and Tomkinson, A. J., 1987. The past distribution of giraffe in Zululand and its implications for reserve management. *South African Journal of Wildlife Research*, 17, 28-32.

Grau, H. R., 2001. Regional-scale spatial patterns of fire in relation to rainfall gradients in sub-tropical mountains, NW Argentina. *Global Ecology and Biogeography*, 10, 133-146.

Greenway, P. J., 1973. A classification of vegetation of East Africa. *Kirkia*, 9, 1 – 68.

Greenway, P. J. and Vesey-Fitzgerald, D. F., 1969. The vegetation of Lake Manyara National Park. *Journal of Ecology*, 57, 127-149.

Hagan, J. E. and Eastman, J. R., 1998. Cartalinx, The Spatial Data Builder (v 1.0). Clark University, Massachusetts, USA.

Hanson, C. J., Wight, J. R., Smith, J. P. and Smoliak, S., 1982. Use of historical yield data to forecast range herbage production. *Journal of Range Management*, 35, 614-616.

Heady, H. F. and R. D. Child, 1994. *Rangeland ecology and management*. Westview Press, Inc., USA, 519pp.

Heyerdahl, E. K., Brubaker, L. B. and Agee, J. K., 2001. Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. *Ecology*, 82, 660-678.

Hobbs, R. J. and Atkins, L., 1988. Spatial variability of experimental fires in south-wets Western Australia. *Australian Journal of Ecology*, 13, 295-299.

Hobbs, R. J., Mallik, A. U. and Gimingham, C. H., 1984. Studies on fire in Scottish heathland communities III. Vital attributes of the species. *Journal of Ecology*, 72, 963-976.

Hodgkinson, K., 1995. A model for perennial grass mortality under grazing. In: West, N. E. (ed.), *Proceedings of the Fifth International Rangeland Congress*. Society for Range Management, Denver, Colorado, USA, pp.240-241.

Hughes, F. M. R., 1988. The ecology of African floodplain forest in semi-arid and arid zones: a review. *Journal of Biogeography*, 15, 127-140.

Hulett, G. K. and Tomanek, G. W., 1969. Forage production on a clay upland site in western Kansas. *Journal of Range Management*, 22, 270-276.

Huston, M., 1979. A general hypothesis of species diversity. *American Naturalist*, 113, 81-101.

Illius, A. W. and O'Connor, T. G., 1999. On the relevance of nonequilibrium concepts to arid and semiarid grazing systems. *Ecological Applications*, 9, 798-813.

Jameson, D. A., 1967. The relationship of tree overstory and herbaceous understory vegetation. *Journal of Range Management*, 20, 247-249.

Jandel Scientific Software, 1994. SigmaStat Statistical Software for Windows. San Rafael, California, USA.

Joubert, J. G. V., 1966. Die invloed van verskillende behandelings op die verhouding van swaarthaak tot gras in die Damaralandse doringboomsavanne in Suid-Wes Afrika. *Annals of the University of Stellenbosch*, 41, 435-463.

Kelly, R. D. and Walker, B. H., 1976. The effects of different forms of land use on the ecology of a semi-arid region in south-eastern Rhodesia. *Journal of Ecology*, 64, 553-576.

Kennan, T. C. D., Staples, R. R., and West, O., 1955. Veld management in Southern Rhodesia. *Rhodesia Agricultural Journal*, 52, 4-21.

Knoch, K. and Schulze, A, 1957. Niederschlag, und schwule in Afrika. *World atlas of Epidemis Diseases*, Volume 2. Heidelberger Akademie der Wissens chaften Heidelberg.

Kok, K., Clavaux, M.B.W., Heerebout, W.M. and Bronsveld, K., 1995. Land degradation and land cover change detection using low-resolution satellite images and the CORINE database: a case study in Spain. *ITC (International Training Centre for Aerial Survey) Journal*, 3, 217-228.

Landmann, T., 2003. Characterizing sub-pixel Landsat ETM+ fire severity on experimental fires in the Kruger National Park, South Africa. *South African Journal of Science*, 99, 357-360.

Lauenroth, W. K., 1979. Grassland primary production: North American grasslands in perspective. In: N. R. French, ed. *Perspectives in Grassland Ecology*. Ecological Studies, 32, pp. 3-24. Springer-Verlag, New York.

Le Maitre, D. C., 1981. Kogelberg season of burn trial II. The experimental fires. Report 81-02. Directorate of Forestry and Environmental Conservation, Jonkershoek Forestry Research Station.

Le Roux, C. J. C., 1989. Die invloed van brand en maai op onbeweide langrasveld in Natal. (The effect of burning and mowing on ungrazed tall grassveld of Natal.) *Journal of the Grassland Society of Southern Africa*, 6, 59-64.

Leuthold, W., 1979. The lesser kudu, *Tragelaphus imberbis* (Blyth, 1869). Ecology and behaviour of an African antelope. *Saugetierkundliche Mitteilungen*, 27, 1-75.

Levine, J. S., 1991. *Global Biomass Burning: Atmospheric, Climate and Biospheric Implications*, MIT Press, Cambridge, MA.

Lewis, F. and Goodman, P.S., 1993. Mkuzi Game Reserve climatic summary for the period ending May 1992. Unpublished N.P.B. report, Pietermaritzburg.

Looijen, J., Pelesikoti, N. and Staljanssens, M., 1995. ICOMIS: a spatial multi-objective decision support system for coastal resource management. *ITC (International Training Centre for Aerial Survey) Journal*, 3, 202-216.

Luke, R. H. and McArthur, A. G., 1978. *Bushfires in Australia*. Australian Government Publishing Service, Canberra, 359pp.

Macdonald, I. A. W. and Jarman, M. L., 1985. Invasive alien plants in the terrestrial ecosystems of Natal, South Africa. *South African National Scientific Programmes Report*, No. 118.

Maguire, D.J., Goodchild, M.F. and Rhind, D.W. (eds.), 1991. *Geographical Information Systems: Principles and Applications*. Longman, Harlow, UK.

Mallawaarachchi, T., Walker, P.A., Young, M.D., Smyth, R.E. and Lynch, H.S., 1996. GIS-based Integrated Modelling Systems for Natural Resource Management. *Agricultural Systems*, 50, 169-189.

Maller, R. A., 1990. Some aspects of a mixture model for estimating the boundary of a set of data. *Journal du Conseil International pour l' Exploration de la Mer*, 46, 140 - 147.

Medina, E. and Silva, J. F., 1990. Savannas of northern South America: a steady state regulated by water-fire interactions on a background of low nutrient availability. *Journal of Biogeography*, 17, 403-413.

Milchunas, D. G. and Lauenroth, W. K., 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs*, 63, 327-366.

Moll, E. J., 1968. Some notes on the vegetation of Mkuzi Game Reserve, Natal. *The Lammergeyer*, 8, 25-30.

Moll, E. J., 1980. Terrestrial Plant Ecology. In: Bruton, M. N. and Cooper, K. H. (eds.), *Studies on the Ecology of Maputaland*. Rhodes University and The Wildlife Society of Southern Africa, pp.52-68.

Moore, A. E. and Attwell, C. A. M., 1999. Geological controls on the distribution of woody vegetation in the central Kalahari, Botswana. *South African Journal of Geology*, 102, 350-362.

Morris, P. P. J., 1980. Die na-effek van boomuitdunning op die samestelling van die grasbedekking van 'n verteenwoordigende plantgemeenskap van die soetbosveld. DSc. thesis, University of Potchefstroom.

Mulqueeny, C. M., 1995. Enhancing the ecological status of wetlands in the afforested Mpate river catchment. Unpublished BSc (Hons) Thesis. University of Natal, Durban.

Norton-Griffiths, M., 1979. The influence of grazing, browsing, and fire on the vegetation dynamics of the Serengeti. In: Sinclair, A. R. E. and Norton-Griffiths, M. (eds.), *Serengeti: Dynamics of an Ecosystem*. The University of Chicago Press, Chicago, pp. 310-352.

O'Connor, T. G., 1985. A synthesis of field experiments concerning the grass layer in the savanna regions of southern Africa. *South African National Scientific Programmes Report* No. 114., 126pp.

O'Connor, T. G., 1995. Transformation of a savanna grassland by drought and grazing. *African Journal of Range and Forage Science*, 12, 53-60.

O'Connor, T. G., 1999. Impact of sustained drought on a semi-arid *Colophospermum mopane* savanna. *African Journal of Range and Forage Science*, 15, 83-91.

O'Connor, T. G., 2001. Effect of small catchment dams on downstream vegetation of a seasonal river in semi-arid African savanna. *Journal of Applied Ecology*, 38, 1314-1325.

O'Connor, T. G., Haines, L. M. and Snyman, H. A., 2001. Influence of precipitation and species composition on phytomass of a semi-arid African grassland. *Journal of Ecology*, 89, 850-860.

Parr, C. L., 1999. Testing patch-mosaic burning in South Africa: the ant perspective. M.Sc. Thesis, Percy FitzPatrick Institute, University of Cape Town, Cape Town.

Parr, C. L. and Brockett, B. H., 1999. Patch-mosaic burning: a new paradigm for savanna fire management in protected areas? *Koedoe*, 42, 117-130.

Phillips, J., 1969. *The agricultural and related development of the Tugela Basin and its influent surroundings: a study in subtropical Africa.* Planning Report Vol. 19. Natal Town & Regional Planning Commission, Pietermaritzburg. 1969 (Map) and 1973 (Text).

Pickup, G., 1994. Modelling patterns of defoliation by grazing animals in rangelands. *Journal of Applied Ecology*, 31, 231-246.

Pickup, G., 1995. A simple model for predicting herbage production from rainfall in rangelands and its calibration using remotely-sensed data. *Journal of Arid Environments*, 30, 227-245.

Pieper, R. D., 1990. Overstory-understory relations in pinyon-juniper woodlands in New Mexico. *Journal of Range Management*, 43, 413-415.

Plowes, D. C. H., 1956. Bush or grass: which do we want? *Rhodesia Agricultural Journal*, 53, 589-595.

Potgieter, A., 1974. Temperatures of veld fires in the Kruger National Park as influenced by burning frequency, season and weather conditions. Unpublished report. University of Zimbabwe, Salisbury.

Pratchett, D., 1978. Effects of bush clearing on grasslands in Botswana. *Proceedings of the1st International Rangeland Congress, Denver, Colorado*, pp. 667-670.

Pressland, A. J., 1975. Productivity and management of mulga in south-western Queensland in relation to tree structure and density. *Australian Journal of Botany*, 23, 965-976.

Rattray, J. M., 1957. The grasses and grass associations of Southern Rhodesia. *Rhodesia Agricultural Journal*, 54, 197-234.

Rautenbach, I. L., Skinner, J. D. and Nel, J. A. J., 1980. The past and present status of mammals of Maputaland. In: Bruton, M. N. and Cooper, K. H. (eds.), *Studies on the Ecology of Maputaland*. Rhodes University and the Wildlife Society of Southern Africa.

Roques, K. G., O'Connor, T. G. and Watkinson, A. R., 2001. Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. *Journal of Applied Ecology*, 38, 268-280.

Russell-Smith, J., Ryan, P. G. and Durieu, R., 1997. A LANDSAT MSS-derived fire history of Kakadu National Park, monsoonal northern Australia, 1980-94: seasonal extent, frequency and patchiness. *Journal of Applied Ecology,* 34, 748-766.

Sala, O. E., Parton, W. J., Joyce, L. A. and Lauenroth, W. K., 1988. Primary production of the central grassland region of the United States. *Ecology*, 69, 40-45.

Salihi, D. O. and Norton, B. E., 1987. Survival of perennial grass seedlings under intensive grazing in semi-arid rangelands. *Journal of Applied Ecology*, 24, 145-151.

San José, J. J. and Medina, E., 1976. Organic matter production in the *Trachypogon* savanna at Calabozo, Venezuela. *Tropical Ecology*, 17, 113-124.

Scholes, R. J. and Archer, S. R., 1997. Tree-grass interactions in savannas. *Annual Review of Ecology and Systematics*, 28, 517-544.

Scholes, R. J. and Walker, B. H., 1993. *An African savanna*. Cambridge University Press, Cambridge.

Schulze, B. R., 1965. *Climate of South Africa* Part 8, General Survey. South African Weather Bureau. Government Printer, Pretoria.

Schulze, B. R., 1972. South Africa. In: Griffiths, J. F. (ed.), *Climates of Africa: World Survey of Climatology*, Volume 10. Elsevier Publishing Company, Amsterdam-London-New York, pp501-586.

Scifres, C. J., Mutz, J. L., Whitson, R. E. and Drawe, D. L., 1982. Interrelationships of huisache canopy cover with range forage on the coastal prairie. *Journal of Range Management*, 35, 558-562.

Scott, J.D., 1984. An Historical Review of Research and Fire in South Africa. In: de V. Booysen, P. and Tainton, N. M. (eds.), *Ecological effects of fire in South African ecosystems*. Ecological Studies 48, Springer-Verlag, Berlin, pp. 53-65.

Seiny-Bouker, L., Floret, C., and Pontanier, R., 1992. Degredation of savanna soils and reduction of water available for the vegetation: the case of northern Cameroon Vertisols. *Canadian Journal of Soil Science*, 72, 481-488.

Shiflet, T. N. and Dietz, H. E., 1974. Relationship between precipitation and annual rangeland herbage production in southeastern Kansas. *Journal of Range Management*, 27, 272-276.

Siegfried, W. R., 1981. The incidence of veld-fire in the Etosha National Park, 1970-1979. *Madoqua*, 12, 225-230.

Sinclair, A. R. E., 1979. The eruptions of the ruminants. In: Sinclair, A. R. E. and Norton-Griffiths, M. (eds.), *Serengeti: Dynamics of an Ecosystem*. The University of Chicago Press, Chicago, pp. 82-103.

Sinclair, A. R. E., 1981. Environmental carrying capacity and the evidence for overabundance. In: Jewell, P. A., Holt, S. and Hart, D. (eds.), Problems in management of locally abundant wild mammals. Academic Press, New York, pp. 247-257.

Skinner, J. D. and Smithers, R. H. N., 1990. *The mammals of the southern African subregion*. University of Pretoria, Pretoria.

Smit, G. N., 2003. The importance of *Salvadora australis* in relation to tree thinning in preserving herbaceous plants in a semi-arid Colophospermum mopane savanna. *Journal of Arid Environments*, 55, 483-501.

Smit, G. N. and Rethman, N. F. G., 1999. The influence of tree thinning on the establishment of herbaceous plants in a semi-arid savanna of southern Africa. *African Journal of Range and Forage Science*, 16, 9-18.

Smith, F. R., 1982. Responses of four shrub species to timing and behaviour of fire in the Natal Drakenseberg. M.Sc. Thesis, University of Natal.

Smoliak, S., 1986. Influence of climatic conditions on production of *Stipa-Bouteloua* prairie over a 50-year period. *Journal of Range Management*, 39, 100-103.

Snyman, H. A., 1988. Bepaling van waterverbruiksdoeltreffendheid van veld in die sentrale Oranje-Vrystaat vanaf evapotranspirasiemetings. *Water (South Africa)*, 14, 153-158.

Snyman, H. A. and Fouché, H. J., 1991. Production and water-use efficiency of semi-arid grasslands of South Africa as affected by veld condition and rainfall. *Water (South Africa)*, 17, 263-267.

Stuart-Hill, G. C. and Tainton, N. M., 1989. The competitive interaction between *Acacia karroo* and the herbaceous layer and how this is influenced by defoliation. *Journal of Applied Ecology*, 26, 285-298.

Stuart-Hill, G. C., Tainton, N. M. and Barnard, H. J., 1987. The influence of an *Acacia karroo* tree on grass production in its vicinity. *Journal of the Grassland Society of Southern Africa*, 4, 83-88.

Sun, D., 1990. Effect of plant age on tolerance of two grasses to simulated trampling. *Australian Journal of Ecology*, 16, 183-188.

Tacheba, G. and Mphinyane, W. N., 1993. Dry matter yield of herbaceous rangeland plants and livemass gain of Tswana steers in eastern Botswana in response to stocking rate and phosphorus supplementation. *African Journal of Range and Forage Science*, 10, 103-107.

Tainton, N. M. (ed.), 1981. *Veld and Pasture Management in South Africa*. Shuter and Shooter, Pietermaritzburg, 481pp.

Tainton, N. M. and Mentis, M. T., 1984. Fire in Grassland. In: de V. Booysen, P. and Tainton, N. M. (eds.), *Ecological effects of fire in South African ecosystems*. Ecological Studies 48, Springer-Verlag, Berlin, pp.117-147

Thomson, J. D., Weiblen, G., Thomson, B. A., Alfaro, S. and Legendre, P., 1996. Untangling multiple factors in spatial distributions: lilies, gophers, and rocks. *Ecology*, 77, 1698-1715.

Thornthwaite, C. W., 1948. An approach towards a rational classification of climate. *Geographical Review*, 38, 55-94.

Tinley, K. L., 1982. The influence of soil moisture balance on ecosystem patterns in southern Africa. In: Huntly, B. J. and Walker B. H. (eds.), *Ecology of Tropical Savannas*. Springer-Verlag, Berlin, Germany, pp. 175-192.

Trollope, W.S.W., 1978. Fire behaviour - a preliminary study. *Proceedings of the Grassland Society of Southern Africa*, 13, 123-128.

Trollope, W.S.W., 1984a. Fire in savanna. In: de V. Booysen, P. and Tainton, N. M. (eds.), *Ecological effects of fire in South African ecosystems*. Ecological Studies 48, Springer-Verlag, Berlin, pp. 149-175.

Trollope, W.S.W., 1984b. Fire Behaviour. In: de V. Booysen, P. and Tainton, N. M. (eds.), *Ecological effects of fire in South African ecosystems*. Ecological Studies 48, Springer-Verlag, Berlin, pp. 199-217.

Trollope, W.S.W., 1993. Fire regime of the Kruger National Park for the period 1990-1992. *Koedoe,* 36, 45-52.

Trollope, W. S. W. and Potgieter, A. L. F., 1983. Characteristics of fire behaviour, in the Kruger National Park Research Report, University of Fort Hare, Alice.

Trollope, W. S. W. and Potgieter, A. L. F., 1986. Estimating grass fuel loads with a disc pasture meter in the Kruger National Park. *Journal of the Grassland Society of Southern Africa,* 3, 148-152.

Van den Berg, J. J., Roberts, B. R. and Vorster, I. F., 1976. Die uitwerking van seisoen-beweiding op die infiltrasievermoë van gronde in ’n *Cymbopogon-Themeda* veld. *Proceedings of the Grassland Society of Southern Africa*, 11, 91-95.

Van Rooyen, N. and Theron, G. K., 1982. n’ Kwantitatiewe analise van die kruidstratum van die *Eragrostis pallens-Burkea africana* boomsavanne op die Nylsvlei natuurreservaat. *South African Journal of Science,* 78, 116-121.

Van Wilgen, B. W. and Forsyth, G. G., 1992. Regeneration strategies in fynbos plants and their influence on the stability of community boundaries after fire. In: Van Wilgen, B. W., Richardson, D. M., Kruger, F. J. and Van Hensbergen, F. J. (eds.), *Fire in South African Mountain Fynbos*. Ecological Studies 93, Springer-Verlag, Berlin, pp. 81-107.

Van Wilgen, B. W., Everson, C. S., and Trollope, W. S. W., 1990. Fire management in Southern Africa: some examples of current objectives, practices, and problems. In: Goldammer, J. G. (ed.), *Fire and Tropical Biota*, Ecological Studies 84, Springer-Verlag, Berlin, pp. 179-215.

Van Wilgen, B. W., Bond, W. J., and Richardson, D. M., 1992. Ecosystem management. In: Cowling, R. M. (ed.), *The Ecology of Fynbos: Nutrients, Fire and Diversity*. Oxford University Press, Cape Town, pp. 345-371.

Van Wilgen, B. W., Biggs, H. C., O’Regan, S. P. and Maré, N., 2000. A fire history of the savanna ecosystems in the Kruger National Park, South Africa between 1941 and 1996. *South African Journal of Science,* 96, 167-178.

Van Wyk, J. J. P., 1967. Ekologiese studies in verband met die agteruitgang en rehabilitasie van die grasbedekking in die laer Krokodilriviervallei noordwes van Thabazimbi. *South African Medical Journal*, 41, 1248-1253.

Van Wyk, P., 1972. Veld burning in the Kruger National Park, an interim report on some aspects of research. In: *Fire in Africa. Proceedings of the Tall Timbers Fire Ecology Conference*, 11, 9-31.

Walker, B. H., 1974. Ecological considerations in the management of semi-arid ecosystems in south-central Africa. *Proceedings of the International Congress of Ecology*, 1, 124-129.

Walker, B. H. and Langridge, J. L., 1997. Predicting savanna vegetation structure on the basis of plant available moisture (PAM) and plant available nutrients (PAN): a case study from Australia. *Journal of Biogeography*, 24, 813-825.

Walker, B. H. and Noy-Meir, I., 1982. Aspects of the stability and resilience of savanna ecosystem. In: Huntley, B. J. and Walker, B. H. (eds.), *Ecology of tropical savannas*. Springer-Verlag, New York, New York, USA, pp. 556-590.

Walker, B. H., Ludwig, D., Holling, C. S. and Peterman, R. M., 1981. Stability of semi-arid savanna grazing systems. *Journal of Ecology*, 69, 473-498.

Walker J., Moore, R. M. and Robertson, J. A., 1972. Herbage response to tree and shrub thinning in *Eucalyptus populnea* shrub woodlands. *Australian Journal of Agricultural Research*, 23, 405-410.

Walker, J., Robertson, J. A., Penridge, L. A., and Sharpe, P. J. H., 1986. Herbage response to tree thining in a *Eucalyptus creba* woodland. *Australian Journal of Ecology*, 11, 135-140.

Walter, H., 1971. *Ecology of tropical and subtropical vegetation*. Oliver and Boyd, Edinburgh, Scotland.

Walters, M. J., 2000. Effect of season and type of fire on *Colophospermum mopane* woodland in the south-eastern lowveld of Zimbabwe. Unpublished MSc. thesis. University of Natal, Pietermaritzburg.

Weltzen, J. R. and Coughenour, M. B., 1990. Savanna tree influence on understory vegetation and soil nutrients in northwestern Kenya. *Journal of Vegetation Science*, 1, 325-332.

West, O., 1947. Thorn bush encroachment in relation to the management of veld grazing. *Rhodesia Agricultural Journal*, 44, 488-497.

West, O., 1958. Bush encroachment, veld burning and grazing management. *Rhodesia Agricultural Journal*, 55, 407-425.

West, O., 1965. Fire in vegetation and its use in pasture management with special reference to tropical and subtropical Africa. Mimeographed Publication No.1. Commonwealth Bureau of Pastures and Field Crops, Hurley, Berkshire, pp1-53.

Whitney, G. G., 1986. Relation of Michigan's pre-settlement pine forests to substrate and disturbance history. *Ecology*, 67, 1548-59.

Williams, R. J., Gill, A. M. and Moore, P. H. R., 1998. Seasonal changes in fire behaviour in a tropical savanna in northern Australia. *International Journal of Wildland Fire*, 8, 227-239.

Wills, A. J., 1987. Proposed agenda for the subcommittee discussing the implications of implementing a winter burning strategy. Unpublished Natal Parks Board document. 5pp.

Woods, D. B., 1997. Siting artificial water points in Mkuzi Game Reserve using GIS. Unpublished MSc. thesis. University of Natal, Pietermaritzburg.

Zar, J. H., 1984. Biostatistical Analysis: Second Edition. Prentice Hall, Englewood Cliffs, New Jersey, USA. 718pp.

PERSONAL COMMUNICATIONS:

Goodman, P. S., KwaZulu-Natal Nature Conservation Service: Assistant Director – Biodiversity Division. (Previously Regional Ecologist based at Mkuzi Game Reserve from 1975 to 1997).

Keet, G., Conservation Manager, Opathe Game Reserve (Previously officer in charge of Mkuzi Game Reserve from 1997 to 1999). P.O. Box 1209, Ulundi, 3838.

Llewellyn, J., Conservation Manager, Weenen Game Reserve. (Previously second in charge of Mkuzi Game Reserve from 1995 to 1998). P. O. Box 122, Weenen, 3325.

Van Rensburg, S., Regional Ecologist, South Zululand (including Hluhluwe-Umfolozi Park). Hluhluwe Research Centre, P. O. Box 515, Hluhluwe, 3960.

Whitley, A., GIS Manager, Working for Water Programme, Department of Water Affairs and Forestry, (Initiated a research project on the distribution of herbivores in Mkuzi Game Reserve). Private Bag X24, Howick, 3290.

APPENDIX 1: Summary of the relationship between vegetation physiognomic types and geological types

Vegetation: / Geology:	Forest & Closed Woodland (ha)	Row %	Mixed Woodland (ha)	Row %	Mixed Bushland (ha)	Row %	Mixed Thicket & Scrub (ha)	Row %	Wooded Grassland (ha)	Row %	Grassland (ha)	Row %	Wetlands (ha)	Row %	Unknown Vegetation Type (ha)	Row %	Total (ha)
Marsh (ha)	32	4	43	6	11	1	0	0	26	3	0	0	646	84	15	2	773
Column %	3		1		0		0		0		0		79		8		
Young Alluvium (ha)	598	29	146	7	1108	53	3	0	26	1	9	0	63	3	128	6	2082
Column %	62		3		16		1		0		1		8		72		
Old Alluvium (ha)	14	1	353	37	282	30	20	2	254	27	25	3	3	0	2	0	954
Column %	1		7		4		9		3		3		0		1		
Orange to Red Dune Cordon Sand (ha)	199	12	528	31	184	11	0	0	768	46	0	0	0	0	0	0	1679
Column %	21		11		3		0		8		0		0		0		
Siltstone & Sandstone (ha)	1	1	103	90	0	0	0	0	0	0	0	0	9	8	1	1	113
Column %	0		2		0		0		0		0		1		1		
Glauconitic Marine Sandstone (ha)	3	0	2467	42	2545	43	111	2	673	11	0	0	94	2	8	0	5900
Column %	0		52		37		50		7		0		11		4		
Conglomerate, Grit, Sandstone etc. (ha)	14	0	390	8	1626	35	0	0	2626	56	0	0	3	0	0	0	4659
Column %	1		8		24		0		29		0		0		0		
Rhyolite, Rhyodacite & Syenite (ha)	96	1	725	10	1069	14	89	1	4687	63	802	11	0	0	25	0	7492
Column %	10		15		16		40		52		96		0		14		
Total (ha)	957		4754		6825		223		9059		836		818		178		23651

137

APPENDIX 2: Summary statistics for the relationship between fire frequency (over 37 years) and geological type, and between fire frequency and vegetation physiognomic type

Geological Type:	**Min.**	**Max.**	**Median**	**Mode**	**Mean**	**SD**	**d.f.**	**Sample Size***
(a) Marsh	0	8	1	1 & 7	2.5	2.71	12362	12363
(b) Young Alluvium	0	9	1	0	1.2	1.41	33315	33316
(c) Old Alluvium	0	16	6	7	6.0	3.61	15257	15258
(d) Orange to red dune cordon sand	0	11	3	2	3.5	2.36	26855	26856
(e) Siltstone and sandstone	0	7	3	3	3.2	1.32	1813	1814
(f) Glauconitic marine sandstone	0	12	2	2	2.5	1.83	94397	94398
(g) Conglomerate, grit, sandstone and siltstone	0	12	4	1	4.2	3.13	74543	74544
(h) Rhyolite, rhyodacite and syenite	0	21	10	13	10.3	3.69	119868	119869

Vegetation Physiognomic Type:	**Min.**	**Max.**	**Median**	**Mode**	**Mean**	**SD**	**d.f.**	**Sample Size***
(a) Forest and Closed Woodland	0	10	1	0	2.2	2.40	15317	15318
(b) Mixed Woodland	0	17	3	3	4.2	3.32	76069	76070
(c) Mixed Bushland, Thicket and Scrub	0	20	2	1	3.3	3.53	112765	112766
(d) Wooded Grassland	0	20	7	5	7.7	4.30	144945	144946
(e) Grassland	1	21	14	14	13.3	3.04	13376	13377
(f) Wetlands	0	8	1	1 & 7	2.4	2.65	13087	13088

* Sample size refers to the number of 25 m x 25 m pixels/grid-cells.