EFFECT OF FIRE FREQUENCY ON THE REGENERATION OF MIOMBO WOODLAND IN NHAMBITA, MOZAMBIQUE

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ABSTRACT OF THESIS

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Fire has been part of the miombo woodlands since the Early Stone Age, about 60 000 years ago and it has been associated with the development and maintenance of tropical savannas. Fire has direct effect on vegetation by raising the soil and atmospheric temperatures, reducing organic matter and release of gaseous elements and indirect effect by modifying both the post-fire microclimate and the activity of the soil biota. Plants differ widely in their response and tolerance to fire and in their capacity to recover afterwards. Owing this, the effects of fire on the floristic and structure composition of miombo woodlands can not be seen in isolation from the influence of other biotic and abiotic factors.

This study has looked at the effect of fire frequency on the regeneration of miombo species in Nhambita Community Land in the buffer zone of the Gorongosa National Park, Mozambique. The data were collected in June 2005 from twenty randomly selected sample plots in four fire frequency treatments (unburnt, burnt 1 time, burnt two times and burnt three or more times) obtained from MODIS fire map.

The study revealed that fires are generally on increase with particular incidence in the community land where it was found 51.0% higher in relation to the Gorongosa National Park. Species number in plots which were (i) unburnt, (ii) burnt 1 time, (iii) burnt two times and (iv)burnt three or more times were 30, 34, 37 and 32 respectively, although the total number of sprouts, basal area and coppicing effectiveness declined with burning.

The study also found MODIS fire map and data every useful in the identification of fire frequency and location, although the extent could not be ascertained.

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CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATION

AGP	General Peace Accord
CBNRM	Community Based Natural Resources Management
DNA	Direcção Nac de Aguas (National Water Directorate)
ECCM	Edinburgh Centre of Carbon Management
EN	Estrada Nacional (National Road)
ER	Estrada Rural (Rural Road)
FAO	Food and Agriculture Organisation
FRELIMO	Mozambique Liberation Front
GNP	Gorongosa National Park
GPS	Global Position System
INE	Inst. Nacional de Estatistica (National Statistics Institute)
MODIS	Moderate Resolution Imaging Spectroradiometer
NASA	National Aeronautics and Space Administration
NGO	Non-Governmental Organization
NCL	Nhambita Community Land
PSP	Permanent Sample Plot
RENAMO	Mozambican National Resistance
SADC	Southern African Development Community
SE	Standard Error
WWF	World Wildlife Fund
WWF-SARPO	WWF – Southern Africa Regional Programme

1   INTRODUCTION TO THE DISSERTATION STUDY

1.1   Background to the Study

This dissertation presents the study on the "Effect of Fires Frequency on the Regeneration of Miombo Woodland in Nhambita Community, Mozambique" which was carried out in partial fulfilment of the requirements for the MSc degree in Resource Management at the University of Edinburgh.

The study was undertaken from April to August 2005 and included the collection of field data in June at the field sites located in the Nhambita community land area in the buffer zone of the Gorongosa National Park in Sofala Province, Mozambique. The fieldwork was supported by the University of Edinburgh and supervised by Dr Matthew William and Professor John Grace, both from the Institute of Atmospheric and Environmental Science, University of Edinburgh.

The study assessed the effect of fire frequency on regeneration and survival of fifty seven miombo woody plant species and explains some of the traits they use to withstand frequent fires.

1.2   Need for the Study

Miombo woodlands contribute significantly to the social and economy of rural communities throughout the entire miombo region with provision of firewood, charcoal, building material, medicinal plants and innumerable non-timber forest products. On the other hand, Governments derive substantial revenues from logging licences and concession fees for commercial exploitation of miombo timber, building material and fuel wood energy.

In Mozambique, it is estimated that miombo woodlands comprise approximately 70% of the total vegetation cover and the study area is lays within the second largest and most productive miombo mosaic after one in Cabo Delgado in the North of the country (Saket, 1999).

According to the population census of 1997 (INE, 2005)[1], about 70% of the population in Mozambique live in rural areas and of this, 40% live within the miombo zone which means that miombo woodlands contribute directly to the livelihood of about 30% of the total population. Beside this, miombo woodlands contribute 70% of the urban energy requirement in fuel wood energy (SADC, 1988; Chaposa, 2002) and support 10% of unemployed urban population who derive their income from activities in or related to the use of miombo woodlands.

Because of these values, miombo woodlands are under high pressure; land conversion, compounded with the problem of absence of appropriate management actions, weak institution capacity, increasing levels unemployment and low levels of agriculture production among others are factors undermining their sustainability.

The dynamics of miombo woodlands are essentially driven by the social, cultural, economical and ecological factors. Its capacity to regenerate from seeds and sprouting gives the woodlands a unique opportunity to be managed sustainably without compromising its floristic and structural characteristic.

Owing that (i) Fire is the most important environmental factor affecting tropical savannas dynamics (Frost and Robertson, 1977; Chidumayo, 1996) and their causes are mostly anthropogenic (Crawley, 1986), (ii) Fires in the region of the study area are common and extensive owing its proximity to

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[1] INE (National Institute of Statistics, Mozambique. 2005. Total surface area = 799 390 Km²; Total population = 19 420 036; Urban = 6 022 319; Rural = 13 397 717.

major urban centres and being traversed by good road system which favours commercial logging, charcoal burning, and shifting cultivation and (iii) Fire monitoring has been facilitated by widely available global fire products, most of which available for free on the internet.

The management of such important resource requires a holistic approach, which encompasses the interactions of social, economical, political and ecological considerations to understanding factors which govern their maintenance and productivity.

The aim of the study therefore is to tackle one part of the problem (ecological) by learning from research the effect of fire frequency on the regeneration capacity of miombo species. It is intended to assist policy makers and forest concession managers and community at large with relevant information for the management of the miombo woodlands in Mozambique.

1.3   Objectives of the Study

The overall objective of the study is to assess the effect of fire frequency on the regeneration of miombo woodlands. Drawing from the issues highlighted in the above section, the study focus on the following objectives:

▪   To explore the usefulness of MODIS fire products in locating burnt areas and their extent,

▪   To look at the effect of different burning frequency on the regeneration and survival of the miombo species

▪   To understand how the fire affects the woody plant in terms of stocking rates, species dominance and sprout effectiveness.

The project will also contribute to the understanding of the role of sprouting as a mechanism for forest regeneration, maintenance and management.

2 DESCRIPTION OF THE STUDY AREA

2.1 Introduction

The field study was conducted in the Nhambita Community Land (NCL) in the central Province of Sofala, Mozambique between the 1st and 25th of June 2005.

The choice of this area was guided by the ongoing "Miombo Community Land Use and Carbon Management – Nhambita Pilot Project" partly funded by the European Union and jointly implemented by a consortium of Envirotrade, Ltd., Edinburgh Centre for Carbon Management (ECCM) and the University of Edinburgh.

The study was also motivated by the buffer zone policy of the Gorongosa National Park Administration which seeks to implement Community Based Natural Resource Management (CBNRM) projects in the buffer zone areas of the Park as a strategy for reduction of human pressure on wildlife resources within the Park boundary by creating alternative of livelihood and improve relationships with surrounding communities.

2.2 Historical Background of Nhambita Community

The Nambita Community Land was legalised in 2003 after a claim was logged under the new Land Act which permits community to own their ancestral land and manage its resources for the benefit of the entire community based on a pre-approved management plan by the Government.

Part of the community land was dispossessed 36 years ago by the National Park Authority when the then Hunting Reserve was upgraded to the current

status of National Park in 1965. During this period, the community was comprised of small settlements (Madoda, Nhauranga, Nhamunho, Vunduzi, Bue-Maria) scattered along the Barue ridge through the western flank of the Rift Valley to its base along the northern bank of the Pungue River.

Until middle 1960s and before the colonial government adjusted the Districts administrative boundaries and incorporated traditional authorities in the formal governance scheme, this community was part of the Tambarare *Regulado*[2] whose land extended from the Pungue River northward to the slopes of the Gorongosa Mountain and from the Chitunga and Vunduzi Rivers eastward to the floodplains at the base of the Rift Valley. Owing the large size of this *Regulado*, Tambarare was ordered by the government to partition and release part of his land to form new Regulados for an effective administration. Under this order, the Regulados of Chicale (Nhambita) and Matchungire were instituted along the northern bank of Pungue River and both were given to the two son-in-laws of Tambarare.

When the Reserve was upgraded to National Park and its boundary adjusted in 1965, part of the Chicale Regulado land was acquired and its community moved to the Barue ridge. The Matchungire Regulado which was around the Chitengo camp was entirely acquired and the community transferred to the Southern bank of Pungue River in the Pungue Regulado of Nhamatanda District.

A problem still persists between the *Regulos* Pungue and Matchungire, as the transfer of the latter was not followed with land allocation, which means that he was not only dispossessed his land but also he lost his power as *Regulo*. In a recent past, Matchungire raised concern about his situation

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[2] Traditional authority under the Land Act.

and intends to return with his community in his ancestral land inside the Gorongosa National Park.

After the National Independence in 1975, the *Regulado* system was abolished but continued to function in RENAMO controlled areas during the civil war period. After the General Peace Accord (AGP) in 1992 between Frelimo and Renamo which ended the civil war, the *Regulado* system was reinstatement in 2003.

To minimise the effect of high poaching incidence inside the National Park at the time of its rehabilitation, a buffer zone strategy was established and envisaged the involvement of local communities in the management of the Park and sort to provide alternative of livelihood for the communities residing within Park's boundaries in the buffer zone by means of CBNRM programme.

Owing to its high concentration of people and easy accessibility, Nhambita community was selected and promoted to be the pilot area to start CBNRM activities. With involvement of local NGOs and the Park Administration, the land legalisation process and community project were initiated in 2001 including the current "Miombo community land use and carbon management – Nhambita pilot project".

2.3   Location of the Study Area

The study area is situated astride the south-west boundary of Gorongosa National Park between coordinates 18º 49' 30"- 19º 04' 00" South and 34º 02' 00 - 34º 17 30 East, approximately 60Km of Vila Gorongosa in the Sofala Province, Mozambique (Figure 1).
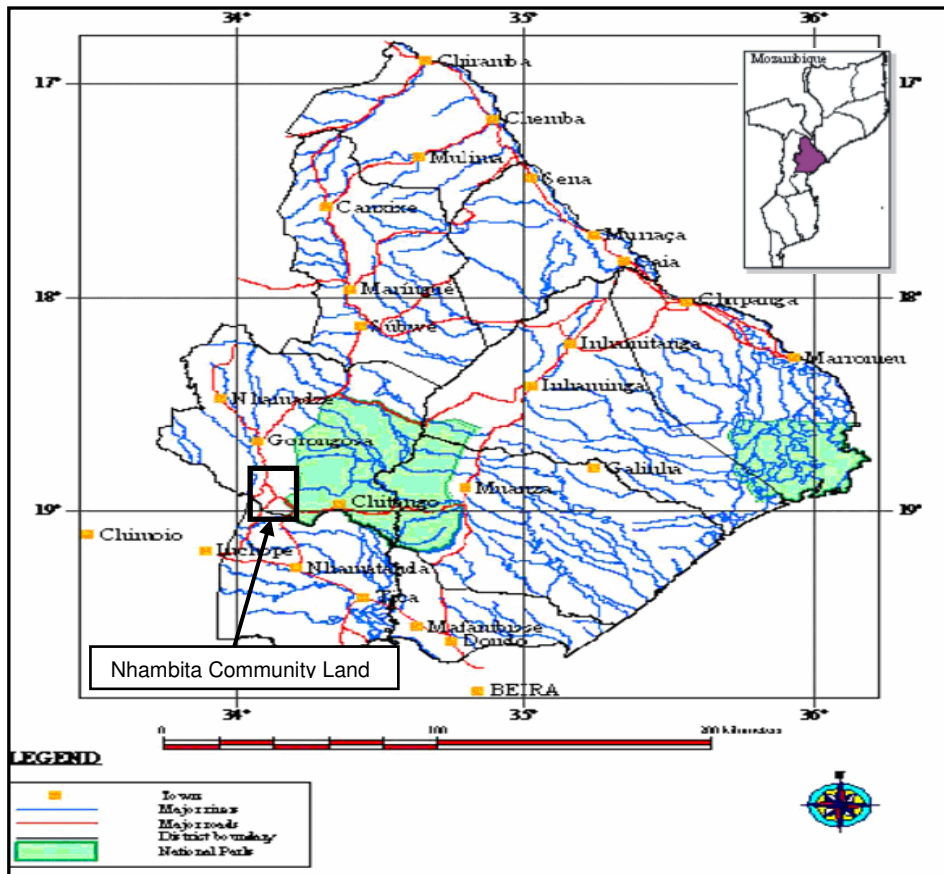
Figure 1: Location of the study area, regional drainage system and main road network.

The area is delimited in the North by a straight line which links a point in the Vunduzi River to another point in the Nhanssenguere grassland area inside the Gorongosa National Park; to the East by another straight line linking the previous point to the confluence of the Nhacapanda and Pungue Rivers. The Pungue and Vunduzi Rivers are the southern and western boundaries respectively.

The area is crossed South-North by the national road (EN-1) and East-West by the rural road ER-418 that serves as the access to west gate of the Gorongosa National Park.

2.4   Bio-Physical Characteristics

The Nhambita community land is situated in the Barue platform and extends its boundaries westward to the flank of the Rift Valley inside the Gorongosa National Park.

Geologically the land consists of eroded surfaces of granite and basaltic gneiss complex of Precambrian times, which is heavily weathered, yielding sandy soils that are generally unsuitable for any form of intensive agriculture (Tinley, 1971, 1977).

The landform is undulating to incised with elevations changing from about 40 m on the flank of the rift valley rising towards the west, where elevations is 400 m and more.

The drainage is dense and closely spaced and assumes a typically dendritic pattern and oriented to the West, South and East. The smaller streams are seasonal and fast running and the Pungue and Vunduzi are only perennial rivers. The groundwater levels are generally very shallow and located either in the weathered regolith in valley bottoms or in fractures in the bedrock (Lynam *et al,* 2003; Tinley, 1969).

The climate is typical of the central Mozambique, sub-tropical with alternating cool-dry winters (April-October) and hot-wet summers (November-March). May-July is the coolest period and October is the hottest month.

The study area leis within the rainfall isohyets of 600 and 800 mm/yr and is generally influenced by the orographic effect of the Gorongosa Mountain. Most of the rain falls between November and March and July to September are the driest months (Figure 2).
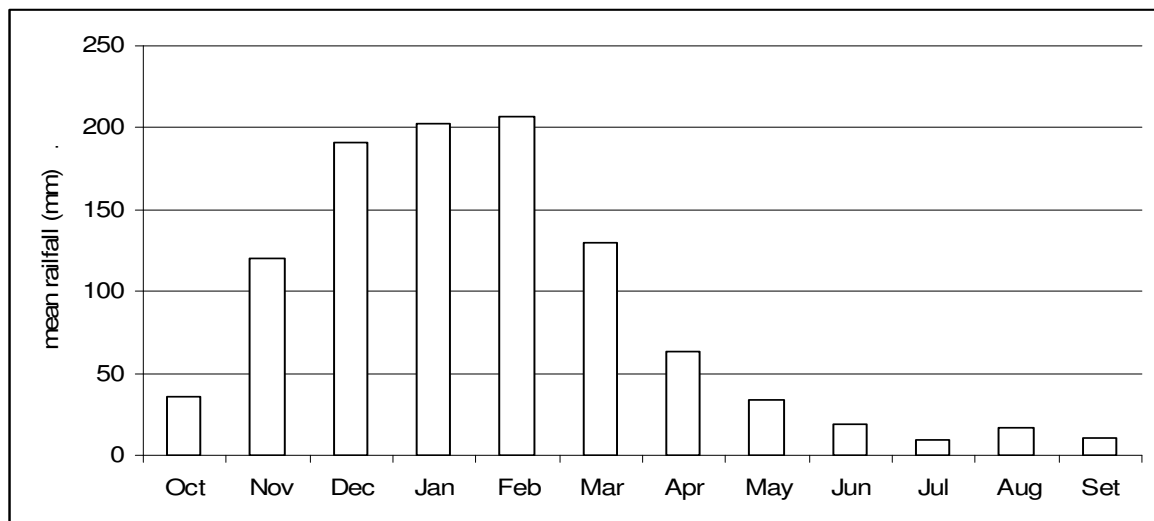
Figure 2: Mean rainfall in Gorongosa District (data from DNA, 2004)

The vegetation consists predominantly of dry miombo woodland, interspersed with evergreen thickets on the deeper alluvial sands of interfluvial crests. The dominant species are *Brachystegia boehmii, B. spiciformis, and Julbernardia globiflora*. There are some narrow patches of thick riverine forest along the longer flowing seasonal streams such as the Lupice and Nhambita Rivers.

2.5 Land Use and Management

The land use system is comprised of three major types: protected area, buffer zone and the community land. The protected area is under State ownership and management and constitutes the land contained within the boundaries of the Gorongosa National Park. It is managed for the protection of flora and fauna and for the enjoyment of the public. The buffer zone is the land immediately adjacent to the boundaries and surrounds the entire National Park area. It is co-managed by various stakeholders including government institutions, non-governmental organisations, and communities associations and to some extent with the involvement of the private sector.

The community land in or outside the buffer zone is the land legalised under the Land Act nº 19/97 and must be managed for the benefit of all members of the community organised in association. Any resource extraction must subject to a management plans approved by the Government.

Within the Nhambita community land, the main land use activities include:

- Subsistence agriculture (practised by almost all families)
- Charcoal production and firewood collection,
- Livestock rearing,
- Fishing and hunting,
- Small scale commerce.

Agriculture is the main land use and constitutes the most important source of subsistence for all families in the community. Charcoal and firewood fuel production are practiced by some members of the community and mostly by outsiders. Charcoal and firewood are sold along the EN1 and constitutes an important source of family revenue.

Human activities are central to the current dynamics of miombo ecosystems. Other than fire, the main disturbances to be considered are partial and complete clearance of woodland for cropland or charcoal production. In the region as whole, large areas of forest have been and continue to be modified by human activities which induce changes in tree density and decline in forest cover. Such changes will potentially have a wide range of long term socio-economic and environmental consequences. By affecting atmospheric chemistry and land surface properties, the impacts of such land use are likely to influence global and regional climate processes, which, in turn, could feedback to affect the patterns of productivity, resource availability, and future land use (Desanker *et al* 1995).

Land use change is often the first consequence of population and economic growth and extensive areas have already been transformed to cropland in South America and Southeast Asia. Miombo woodlands have been preserved by their inherent characteristics of low soil fertility, lack of infrastructure and the presence of diseases (Desanker *et al* 1995) but this is all changing. The acidity and low phosphorus status of the soil can be fixed with known and cost-effective agricultural techniques. Tsetse fly has been eliminated over most of the area. Regional political stability is likely to allow the infrastructure to improve. Human population growth rate in miombo region is high which means that the growing population will be fed partly by expansion of the cropped area, since there are insufficient resources for a general intensification of agriculture (Solomon *et al* 1993, Scholes *et al* 1996).

3 MIOMBO WOODLANDS AND FIRES

3.1 Introduction

Miombo woodlands are tropical savannas dominated by genera *Brachystegia, Julbernardia* and *Isoberlinia* of the family Fabaceae, subfamily Caesalpinioideae and they constitute the largest quasi contiguous block of deciduous tropical woodlands and dry forest in the world (Desanker *et al* 1995).

Miombo ecosystem directly support the livelihood of about 39 million people of rural central African countries and further 15 million people living in urban areas whose life depend on goods and services produced in miombo woodlands.

3.2 Miombo woodland Distribution

Miombo woodlands occur in southern sub-humid tropical zone from near the equator to bellow the Tropic of Capricorn, extending from Tanzania to the Democratic Republic of the Congo in the north, through Angola, Zambia in the east to Malawi, Zimbabwe and Mozambique in the south (Desanker *et al* 1995, Frost 1996).

Although its actual extent of coverage is questionable (Campbell *et al.* 1996), the estimated value of 2.8 million km² suggested by Millington et al (1994) it is generally accepted and this comprises both dry and wet miombos interspersed with dambos[3].

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[3] *Dambo or Mbuga* = broad, grassy depressions set into the landscape by subduction process induced by differential weathering and subsurface removal of material by lateral flow of groundwater. Dambos in miombo ecosystem can cover up to 40% of the landscape.

The miombo ecosystem occur in soils which are predominantly alfisols, oxisols and ultisols which are highly acidic, low in cation exchange capacity, low total exchangeable bases and low available phosphorus (Frost 1996). These soils are formed by a catenary sequence of well drained, deeply weathered soils on higher areas, a narrow zone of sandy soils along the footslopes and poorly drained vertisols in the dambos (Desanker *et al* 1995). Generally they have low levels of organic matter as a consequence of the abundant termite activities and frequent fire incidence (Cauldwell and Zieger 2000, Chidumayo, 1997, Jones 1990, Trapnell *et al*. 1976).
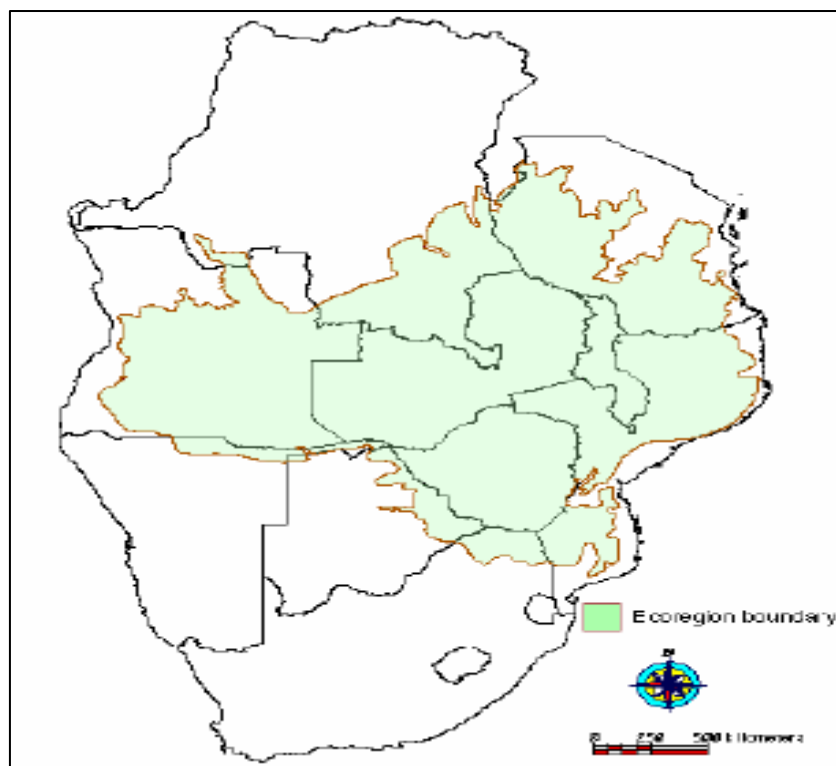

Figure 3: Miombo woodland distribution (after Timberlake, 2001; WWF-SARPO)

The mean annual precipitation ranges from 650 to 1400 mm with more than 95% falling in (November-March) summer season (Cauldwell and Zieger 2000, Desanker *et al*. 1995, Frost 1996).

In Mozambique, miombo woodlands constitute the dominant vegetation type occurring north of the Limpopo River and covers approximately 70% of the total forest cover (Malleux, 1981;Saket *et al* 1995). The Nhambita community lays within the second most productive miombo woodland mosaic in the country after the mosaic along the Lugenda and Rovuma Rivers in the Northern Province of Cabo Delgado (Saket, 1994).

During the national forest inventory work, Saket (1999) listed a number of issues facing the management of forest in Mozambique which are summarised bellow:

- Uncontrolled burning of the forest affects 40% of the country annually. The north western and central parts of the country are the most affected regions with about 70% of these areas burnt annually,

- Forest management is constrained by inadequate infrastructure and material and human resources,

- Fires and land use practices significantly contribute to the loss of standing timber very much needed by the national economy,

- Increasing pressure on forest resources around urban areas and along the main road corridors,

- Loss of species diversity and fragmentation of the forest communities.

3.3   Structure and Floristic Composition

Mature miombo woodland typically comprises an upper canopy layer made of umbrella-shaped dominant tree species of 10-20 m high with a scattered layer of sub-canopy trees; a discontinuous understorey of broad-leafed shrubs and saplings and a sparse but continuous herbaceous layer of grasses, forbs and sedges (Campbell *et al*, 1996; Desanker *et al*, 1995).

The structure and species composition of miombo woodlands are distinguished by the characteristics of the dominant species and these

tend to follow the rainfall gradient, from the drier fringes of the miombo region to the wetter core area (Desanker *et al*, 1995) and as result miombo woodlands are divided into dry and wet types (White,1983 cited by Frost, 1996). Dry miombo occur in the region receiving less than 1 000 mm annual rainfall and extend from Zimbabwe, central Tanzania and southern areas of Mozambique, Malawi and Zambia. They are characterised by having trees with canopy height of less than 15 m and the vegetation is floristically impoverished and many of the dominant canopy tree species of the wet miombo are absent or locally in occurrence (Campbell *et al*, 1996; Desanker *et al*, 1995). Where the species are present, dominant species usually include *Brachystegia spiciformis, B. boehmii* and *Julbernardia globiflora*.

In contrast wet miombos cover the region receiving more than 1 000 mm rainfall annually and extend from the eastern Angola, northern Zambia, south of Tanzania and central Malawi (Campbell et al, 1996; Desanker et al, 1995; Chidumayo, 1997). They develop in deeper and moister soils which results in higher canopy height (> 15 m) and floristically richer including all of the characteristic miombo species with *Brachystegia floribunda, B, glaberrima, B. longifolia, B. wangermeeana, Julbernardia paniculata, Isoberlinia angolensis* widely distributed (Desanker *et al*, 1995; Frost 1996).

Within this broad division, miombo woodlands display local variation in structure and floristic composition mainly influenced by edaphic and climatic conditions, fire regime, herbivory and past and present land use (Desanker *et al*, 1995).

The miombo woodlands have an estimated 8 500 species of higher plants of which, 54% are endemic to the region and 4% are tree species. Zambia is considered to be the centre of endemism for *Brachystegia* and has the highest diversity of tree species (Rodger et al, cited by Frost, 1996).

3.4   Reproduction and Establishment

Most of the miombo species are capable of reproducing sexually and asexually. Sexual reproduction is affected by annual variation in seed production despite it occurs throughout the year with a peak towards the end of the dry season (Chidumayo, 1993). Seed production is also often affected by lack of flowering or flower abortion such as the case of *Julbernardia paniculata* and some members of the *Brachystegia* genera which are susceptible to insect infection particularly in old-growth miombo (Clauss, 1992 cited in Chidumayo 1997).

Seed dispersal is generally limited and varies with type of fruit and dispersal agents. For wind-dispersed (anemochorous) species, the longest distance found was 28 – 103 m, followed by those with an explosive pods (autochorous) species with 10 – 20 m and the shortest distance (5-10) was among the animal-dispersed (zoochorous) species Chidumayo, 1997; Tinley, 1977; Chidumayo and Frost, 1996). Seed production is not always good indicators of seed crop available for germination and their limited dispersal capacity which implies that complete clearing of miombo woodland is likely to affect the floristic composition (Chidumayo and Frost, 1996). Also pre- and post-dispersal seed predation account for 86% and 33% respectively (Chidumayo, 1997).

According to Chidumayo (1997), Ernst (1998) and Trapnell (1959), tree seedling development is mostly impaired by biotic and abiotic factors and there is greater inter-specific variation in the survival rate of miombo tree seedlings, especially during the establishment phase and mortality can be up to 70%.

Majority of miombo seeds germinate immediately after dispersal as long as there is sufficient moisture on the soil (Chidumayo and Frost, 1996), Seedling

mortality is mainly caused by drought, water stress, insectivory, herbivory and fire (Chidumayo, 1997).

Shoot growth among established seedlings is slow and it appears that they use this time to accumulate a relatively large belowground biomass which constitutes the perennating organ that regenerates new shoots following repeated shoot die-back during the dry season or fire event (Cauldwell and Zieger, 2000).

Asexual reproduction is mainly by vegetative regeneration and it is the most effective form of reproduction among miombo species as almost all species produce root sucker and possesses epicormic buds which allow them to coppice once the aboveground parts have been removed or damaged (Chidumayo, 1997; Chidumayo and Frost, 1996).

Root sucker are produced either by the extended surface roots of mature trees or seedling subjected to successive damage which accumulate substantial amount of reserve in the root system. Coppicing of harvested tree is affected by the height at which the tree is cut; for example *Brachystegia spiciformis* and J*ulbernardia globiflora,* when are cut close to the ground (< 5 cm) produce less coppice growth that plants cut higher (Grundy, 1995).

During the establishment period the numbers of coppicing shoots decreases due to inter-shoot competition and only shoots that have attained superior size contribute to the next generation (Chidumayo, 1997). Fire has an additional effect on this natural thinning process particularly at the late dry season fires owing to their intensity and timing in relation to plant phenology.

In this context, regeneration by coppicing makes miombo species remarkably resilient to disturbance and constitutes the most effective form

of persistence (Boaler and Sciwale 1966; Grundy 1995; Frost, 1996) rather depending on seed production.

3.5   Role of Sprouting

The importance of resprouting in forest regeneration lay on the fact that sprouts grow faster than established seedlings; they are capable of quickly reoccupying their own gaps and therefore minimising the effect of the disturbance and minimising the population turnover often caused in plants that relay on seeds (Bond and Midgley, 2001).

Sprouting varies among species, among life history stages of species and among disturbances of differing severity. To be able to resprout after damage, a plant needs to have a surviving meristems and sufficient stored reserve to support regrowth (Grime 1988).

The majority of seedlings of miombo tree experience a prolonged period of successive shoot die-back during their development phase and this are caused by water stress or fire during the dry season (Chidumayo 1997, Trapnell 1959). Die-back does not necessarily result in death if the root can survive and produce new shoot the following growing season (Chidumayo 1997).

Luoga et al (2004) studying regeneration by coppicing of miombo woodland in Tanzania found that over 80% of the species resprouted after their aboveground parts have been harvested or damaged. On the other hand, most miombo species produce large amount of shoots but owing the inter-shoot competition and probably the effect of fires, the number of coppice shoots decreases over time and eventually only the dominant shoots contribute to the next generation of regrowth (Chidumayo, 1988).

The long term effect of sprouting of established trees is the reduction of recruitment opportunities in both space and time and consequently reducing forest diversity and species richness and therefore affecting the forest dynamics by favouring self-replacement after stem death (Bond and Midgley, 2001). Non-sprouters on the other hand, are more vulnerable to recruitment failure after severe disturbance and more vulnerable to the problem of small population size such as inbreeding, loss of pollinators and dispersers (Bond, 1994).

In the case of most miombo species which combine both embryonic and vegetative reproduction are often in better condition to maintain their population by withstand disturbance and environmental fluctuation.

3.6   Survival Trait and Fire Tolerance Classification

Miombo species have specialised reproductive organs which allows them to store food reserves and these include (i) lignotubers; large woody swelling of the stem that occurs below the soil surface and act as an organ of storage nutrient reserves, (ii) epicormic buds, dormant buds present below the bark where they are protected from both fire and insect attack and (iii) root sucker, the extension of the surface root system which allows the development of new shoots away from the may plant (Grime, 2001).

In such plants species, the importance of persistence by sprouting outweighs the recruitment by seedlings; however allocation of resources to storage has a cost trade off against growth or reproduction (Masaka *et al*, 2004). Studies in Mediterranean vegetations concluded that sprouters generally have fewer seeds, smaller seedbank, slower growth and maturation rate and almost always have fewer seedlings and poor seedling survival than non-sprouter (Bond et al 1996).

Survival characteristics of miombo species include the thickness of the bark, late leaf flush and high moisture content and the production of enormous quantities of sprouts either from root suckers of epicormic buds, (Chidumayo, 1996).

Fire tolerance level of miombo species have been devised by Trapnell (1959) and Lawton (1978). Basically both classification are based on fire seasonality and physiognomic characteristics of the species and Lawton (1978) included some ecological parameters such as competition and species association to differentiate species tolerance levels.

Trapnell (1959) classification includes: (i) Fire-tolerant species which are those capable to survive regular late dry season fires as adults, saplings and regrowth. They include some canopy and understorey trees and shrubs. (ii) Semi-tolerant species are relatively unaffected by early dry season fires but are reduced somewhat under late dry season fires, (iii) Fire-tender species are those that decline under regular burning and increase under complete protection. Most of the dominant canopy species belong to this group.

On the other hand Lawton (1978) proposed 5 groups: (i) Group 1 or "*chipya*" species comprising species which can survive intense late dry-season fires but which are intolerant of shade and therefore depend on regular fires to maintain an open woody canopy. (ii) Group 2 species which are moderately fire-resistant, but cannot establish or persist in tall grassland which is subject to intense dry-season fires. When mature, these species form a low dense canopy beneath which grass production is reduced. These conditions are presumed to favour the establishment and growth to maturity of fire-tender species. (iii) Group 3 includes most of the dominant *Brachystegia, Julbernardia* and *Isoberlinia* species characteristic of mature miombo woodland. Although these trees can invade the *Uapaca*-dominated communities, they cannot invade or persist under chipya, (iv) Group 4 comprises species which are intolerant of fire. Many of these are

species characteristic of the evergreen and semi-deciduous forest patches found alongside wet miombo woodland. (v) Group 5 is made up of a suite of ubiquitous species which persist throughout.

Within any one stand, however, there is considerable overlap in the occurrence of these species groups (Lawton, 1978; Kikula 1986) and according to (Frost 1996) the discreteness of this group may be questionable.

3.7   Fires in Miombo Woodlands

The occurrence of fires in the tropics is regular and frequent and they have been associated with the dynamics of tropical savannas (Budowski, 1956; Clayton, 1961). Although there are widespread tropical savannas that have been influenced by frequent fires, particularly in the higher rainfall zones, fire is only one of a number of interacting factors affecting savannas dynamics (Huntley and Walker, 1982; Frost and Robertson 1977). Much of the current knowledge of fire and its effects on savanna structure and functioning has come from general observation supplemented by information derived from a limited number of experiments and it does not allow the effects of fires to be predicted or exclusively attributed to vegetation dynamics (Frost and Robertson, 1977).

The likelihood of occurrence of fires is determined by the prevailing weather condition, the flammability of both living and dead plant material, the volume of fuel at the surface and above ground, and the proximity of igniters (Crawley, 1986).

Forest fires reduce plant biomass and litter, thereby altering the energy, nutrient and water fluxes between the soil, plants and atmosphere. These changes in turn may affect the long term nutrient status and productivity of

the vegetation (Frost, 1996). Fires also kill individual organisms, damage or destroy unprotected living tissues, modify growth and reproductive rates, change the availability and use of resources and alter competitive and other relationships between organisms.

The effects of these impacts depend largely on the recent history of a site, the physiological and developmental state of an organism at the time of burning and the occurrence of future events such as rainfall, drought or herbivory (Frost and Robertson, 1977).

The long term impact of frequent fires may result in changes to the productivity and population structure of a species, the composition of communities and, ultimately the probability and characteristics of future fires.

3.7.1 Fire Characteristics

Fire regime of an area includes the type and intensity, seasonality of occurrence and periodicity and these vary considerably across the range of savanna types. The majority of savanna fires are surface fires, burning through the herbaceous layer with flame heights generally low with the mean flame-length of head and back fires in a savanna grassland averaging 2.8 m and 0.8 m respectively (Trollope *et al*, 2002).

The rate of spread of savanna fires is highly variable and is influenced by the differences in wind speed, topography and the amount and moisture content of the fuel load (Trollope, 1978). Head fires, burning with the wind, move faster than fires burning against the wind, and this affects both the temperature and duration of fire at a point and thereby fire intensity (Trollope, 1978).

Fire intensity also depends on the amount and type of fuel, its moisture content, and prevailing climatic conditions, principally air temperature and relative humidity. In savannas, fires are fuelled largely by grass and fire intensities vary considerably between seasons, landscape units and vegetation types. Communities in which there is a high biomass of grass, such as in valley bottoms or on floodplains, generally experience the most intense and uniform fires. In contrast, in woodlands and shrublands, where there is generally a lower grass biomass and more uneven distribution of fuel, fires tend to be less intense and burn more patchily (Frost and Robertson 1977).

The rate of heat energy release in a particular fire event does not provide sufficient information on the duration of the heat pulse at different points in the system and yet the survival of plants and their propagules are influenced by both the magnitude and duration of the heat pulse, the amounts of nutrient released and volatilized, and changes in soil properties (Frost and Robertson 1977; Trollope, 1978).

3.7.2 Causes of Fires

In tropical savannas of Africa, intentional burning has been practised for at least 50 000 years and have changed little from those of the ancestors (Rose-Innes 1971, Chidumayo 1997). People cause fires in the preparation of land for cultivation, clear bush and undergrowth to improve visibility around settlements and foot paths, keeping away dangerous animals, clear roadside areas before or after grass slashing in road maintenance operations, management of grazing land for both livestock and wildlife, charcoal burning, honey collection and hunting. Fires also originated accidentally by transients in abandoned kitchens left alight in overnight resting camps along pathways linking distant settlements.

Cultural and tradition believes of the many inhabitants of miombo region discourage bush burning until crops have been harvested because of the danger fire pose to crops, while the first rains usually marks the end of the burning season (Chidumayo, 1997). In the study area fire are still set for other purposes such as hunting and improvement of grazing areas and often cause great losses in properties and human life.

Man made fires are by far more frequent than natural fires and natural fires are mostly caused by lightning particularly at the beginning of the rain season (West, 1972).

3.7.3 Effect of Fires Frequency on Vegetation

One of the characteristics of the fire regime is its frequency or return interval. At regional scale, fire returning interval is estimated to be about 3 years but according to Frost (1996) this depend on fuel accumulation rate and on the proximity to potential source of ignition which can reduce the returning intervals and be more variable. In Zambia average returning interval of 1.6 years was estimated at four closely situated sites over a four year period (Chidumayo 1998).

Using satellite imageries, Taquidir (1996) reported an average of 39.6% of Mozambique is burnt every year and the northwest and the central regions comprising 70% of the country vegetation cover are the most affected with 73.6% burnt annually.

In parts of Mozambique most bush fires start as early as in March each year. Their intensity and frequency increases as the vegetation moisture content drops to a low by late August to October and they often continue until the first rains in November or December (Campos Andrade, 1951; Saket 1999).

According to Saket (1999), fire pattern in Mozambique seem to have not changed much over the past twenty years. This finding is questionable given that during the period between 1976 and 1992, extensive areas were unoccupied due to the civil war one would expect to be less fire as opposed to the period after the war when many of the war displaced people returned to the areas of origin and started firming. On the other hand, owing the migration of people from urban to rural areas and occupying areas for agriculture, charcoal production and timber exploitation, would result in an increase of fire frequency.

Analysis of current MODIS fire maps revealed that many sites of Mozambique, including the study area, burn more than one time every year particularly. The complete absence of fire in miombo woodlands is rare and probably limited to dense miombo forests with an evergreen understorey and little grass (Frost and Robertson 1977).

Fire frequency determines the length of time that a plant has to recover before the next fire event. According to Frost and Robertson (1977) there are three conditions affecting the rate of plant recovery after fire event: (i) the extent of damage sustained by the plant, (ii) the method of regeneration and (iii) the favourableness of the post fire environment for establishment and growth, and herbivory extent and intensity. The slower the rate of recovery, the more likely it is that the structure and composition will be altered in event of frequent burning.

Although there is variation in the seasonal timing of fires and in the interval between them at a given site, dry season fires in the understorey are common and frequent and are most destructive (Trapnell 1959; Kikula 1986).

Fire behaviour, timing, intensity and frequency of occurrence all vary independently of each other and affect both the environment and the

vegetation direct and indirect. The effect depends largely on the recent history of a site, the physiological and developmental state of the vegetation, the physical and chemical properties of the fuel available (Trollope, 1978). The prevailing weather conditions, soil moisture, topography, proximity of an ignition factor and the occurrence of future events (rainfall, drought or herbivory), are other factors influencing the occurrence of fires (Cauldwell and Zieger 2000).

The effect of frequency on the vegetation is mostly related to its impact on the soil, water, atmosphere and their interconnectedness. By reducing biomass and litter, fires alter the energy, nutrient and water fluxes between the soil, plants and atmosphere. These changes in turn may affect the long term nutrient status and productivity of the system and consequently the productivity and population structure of a species, the composition of communities and, ultimately, the probability and characteristics of future fires (Frost and Robertson 1977).

Soil is the media from which the plants get the mechanical support and derive their entire requirement for establishment, growth and reproduction. The affect of fires on the physical and the chemical properties of the soils, adversely affect the soil bulk density and porosity by the impact of the reduction in plant and litter cover, changes in microclimate, increase in the rate of organic mater mineralization and change in soil faunal activities (Trapnell et al., 1976; Webber, 1979; Brookman-Amissah et al., 1980; Kadeba, 1982; Hoffmann *et al 2003*).

Soil moisture balance is also affected by the reduction in plant and litter cover and subsequently exposure after fire which leads to increasing in evaporation. The rate of infiltration can be slowed down by frequent fires, largely as a result of change in soil surface structure and the effect of distillation and condensation of aliphatic hydrocarbons from litter and soil

organic matter during fires which leads to the formation of water repellent surface even on sandy soils (Cass et al., 1984).

Soil acts as an effective insulator during fire; the soil temperatures decrease very rapidly with increasing depth and because of this, soil organisms and subterranean parts of plants (including perennating buds of many plants) can survive surface fires (Frost and Robertson, 1977).

Fires modify growth and reproductive rates, change the availability and use of resources and alter competitive and other relationships between organisms (Frost and Robertson, 1977). Frequent fires reduce woody plant densities, change of the floristic and structure composition of vegetation, primarily by killing or suppressing individuals in the smaller size classes (Trapnell, 1965; Hopkins, 1965; Brookman-Amissah *et al*, 1980).

3.7.4 Effect of Fires Intensity on Vegetation

Fire intensity is a function of fuel type, fuel load, moisture content, and atmospheric conditions at the time of fire and therefore it is linked to both the seasonality and frequency of burning. The longer the interval between fires, the greater is the fuel load accumulated and the more intense the fire will be. Fire intensity exerts differential effects on the survival of plants and their propagules, and stimulates the germination of seeds of different species to various degrees (West, 1972; Trollope *et al,* 2002).

The impact of fire intensity on biomass fuel depends on the time of burning, which is related both to the moisture content and the amount of fuel. Late dry-season fires are more intense and destructive than fires occurring in the early dry season when fuel moisture contents are relatively high and ambient temperature and wind speed are low. These early dry season fires are less destructive and tend to create a mosaic type of differentiated

vegetation in the landscape and contribute to the survival of wildlife species which find refuge in the unburnt patches (Frost, 1984).

The time of burning interacts with plant phonology and post-fire weather conditions to affect plant survival and reproduction. Intensity also variously affects the physical and biological environments of each species, making conditions sometimes more, sometimes less, suitable for establishment and growth. The late dry season is a period of nutrient translocation, woody plant density is lowest and has rapid growth; fire at this time is considered to be very damaging (West, 1972; Kennan, 1972).

In moist savannas, frequent late dry season fires destroy young trees and shrubs, or their aboveground parts, so preventing the development of taller, more fire resistant size classes. Although woody plant growth may be further inhibited by browsing ungulates, fire alone is sufficient to promote a lower woody plant density (Kennan, 1972, Trollope et al, 1998, Cumming, 1982).

3.7.5 Effect of Fire on Regeneration of Woody Species

Dry season fires are more destructive not only due to its intensity but also this period coincides with the time that woody plants higher moisture content, hence higher thermal conductivity and rapid transfer of heat to the interior of the plant tissues (Frost and Robertson 1977). Consequently, the proportion of woody plants in the community declines as fire regimes become progressively hotter.

Many woody plants produce new leaves before the start of the rainy season which means that part of the food reserves have been used up rendering the plant weak for the next growing season. Owing that, new leaves are also more susceptible to damage by fire, burning at the start of

the wet season forces the plant to draw on already depleted reserves in order to replace those consumed in the fire (West, 1972; Kennan, 1972).

Fire tends to favour those woody species which have protected meristems and bellow-ground reproductive organs and seeds that can survive fire and in which heat triggers germination (West, 1972; Everhan and Brokan, 1996).

Mortality rates of mature trees and saplings are higher under late dry-season burning than under early dry-season burning; the lowest is observed under complete protection.

According to Frost and Robertson (1977) the effect of fire on tree regrowth is strongly influenced by the stem diameter and height (Figure 4).
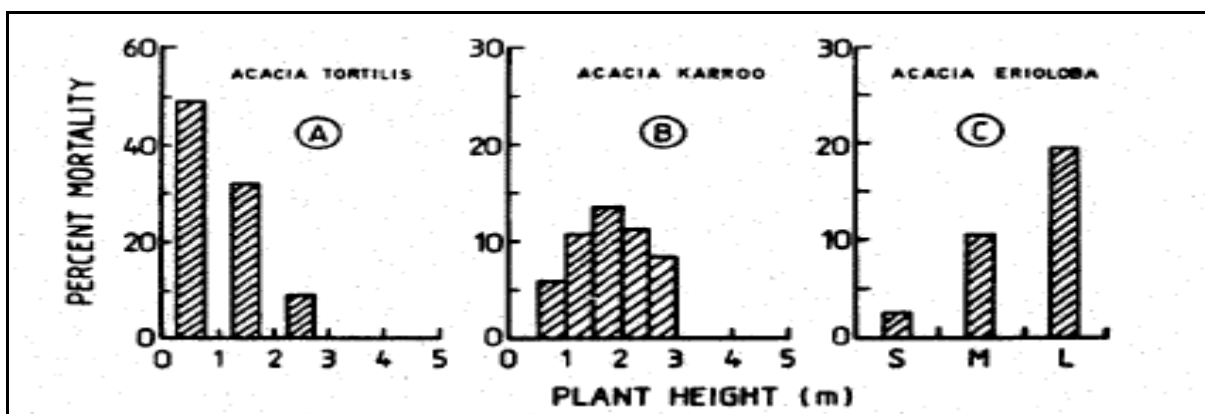
Figure 4: Percent mortality caused by fire in various savanna woody plants. **(A)** *Acacia tortilis;* (**B**) *Acacia karoo*; (**C**). *Acacia erioloba* (Frost and Robertson 1977)

Seedling and saplings are generally at the greatest risk, but in some species, mortality appears to be highest in intermediate height classes, e.g. *Acacia karoo*. In contrast, in *Acacia erioloba*, mortality of mature trees was more than 7 times higher than that of saplings (Figure 4)

Generally protection from fires results in an increase in tree density and decrease in grass production (Trapnell, 1959; Frost, 1984). In dry savannas, fires seldom occur frequently enough to limit the density of woody plants, though when they occur, often after prolonged period of above-average rainfall, mature woody plants may experience considerable mortality (Younes, 1986)

3.8   Fire Detention and Monitoring

Fire monitoring globally has been made easier with the advancement of remote sensing and internet technologies and the good will of many institutions to share their products. Several fire monitoring products and database are available on web sites for downloading often free of charge.

This research made use of MODIS (Rapid Response System) Web fire mapper (NASA, 2004a) a product of the collaboration between the University of Maryland and the NASA Goddard Space Flight Centre designed to provide rapid access to MODIS fire data globally. MODIS instrument on board NASA's Aqua and Terra satellites detect of active fire from observations made twice daily using a contextual algorithm that exploits the strong emission of mid-infrared radiation from fires relative to the background temperature to account for variations in the surface temperature and solar reflection. The algorithm examines each pixel of the MODIS swath (approximately 2330km x 10km) and assigns to each one a class (missing data, cloud, water, non-fire, fire, or unknown). For each active fire detected, detailed information of its geographical location, brightness, date and time of acquisition and the level of confidence is provided.

One of the advantages of this product in relation to other MODIS products (MODIS Thermal Anomalies and MODIS Burned Area) (NASA, 2004b) lays on

the fact that locations of active fires and respective data are provided near real time which makes very useful to resource managers act upon any situation. The disadvantage is that the data is only available as jpegs or ArcView shape files which renders its further manipulation more difficult. The datasets can be obtained from http://maps.geog.umd.edu/default.asp

3.8.1 Limitations

Each fire detection represent the centre of a 1 km pixel flagged as containing one or more actively burning fires within that pixel. In any given scene the minimum detectable fire size is a function of scan angle, biome, sun position, land surface temperature, cloud cover, amount of smoke and wind direction. The precise value will vary slightly with these conditions but generally, in many biomes the minimum flaming (~800-1000K) fire size typically detectable at 50% probability with MODIS is on the order of 100m² (NASA, 2004a).

Because each fire event is fitted in a 1 km pixel independently of its size, makes the exact location of the fire incident difficult in the field unless the fires are directly monitored and mapped in the field. Small hot fire late in the dry season can be detected and been shown in 1 km pixel which makes searching for the exact position in 1 km² grid difficult.

Plotting the coordinates of fire events of the past four years without setting an area of influence, as it was done in the present work (Figure 4), they tend to show little or no overlapping between the fires events.

MODIS fire information can be very powerful fire monitor tool where ground truthing can be carried out immediately after or during the fire incidence.

4   RESEARCH DESIGN AND METHODOLOGY

4.1   Introduction

Research design is the most important stage in any scientific work. It provides the framework for the establishment of sample units, data collection, analysis and the outputs. To make the most of a research work, the design must be *a prior* involving careful thinking around what is intended to be achieved, in other words the precision, accuracy and generality. Poor design may result in inconsistent results and ineffective use of resources.

4.2   Research Hypothesis and Objectives of the Study

The effect of fires on tropical savannas has been extensively studied (Trapnell, 1959, 1976; West, 1972; Frost and Robertson, 1977; Webber, 1979; Kikula 1986) but much still needed to be learned about its effects and how these interact with other ecological processes which influence the dynamics of savannas.

The present research looks at the interaction of different fire frequency on some characteristics of the miombo woodland in Mozambique assuming that among other factors, fires play the most important role in shaping the structural and floristic composition of miombo woodlands.

The overall research hypothesis is: "Different fire frequencies have no significant effect on the regeneration of miombo woodland".

Regeneration is assumed to encompass species diversity, number of sprout/stem or regrowth, species dominance and relative importance of individual species. Owing the limitation in the used of MODIS data as

mentioned above, it is also assumed that the identification and location of the fire treatments are correct and represents the reality on the ground.

The specific objectives of the study are:

- To assess and compare the sprout stocking rate, species dominance and coppicing effectiveness between the four fire treatments.
- To produce a list of miombo species that survive the effect of fires that can be used for forest management.
- To evaluate the applicability of MODIS web fires map in locating fire events.

4.3   Research Design and Treatments

To evaluate the effect of fire frequency on the regeneration, four burning treatments representing fire frequencies were chosen as (i) unburnt, (ii) burnt one time, (iii) burnt at least two times, and (iv) burnt more than three times from 2001.

Similar studies have previously been conducted but under controlled experimental condition whereby blocks in the forest were selected and subjected to different treatments (Trapnell, 1959, Chidumayo, 1988, Kennard *et al* 2002). The present study, due to time availability, the selection and location of the treatment was based on freely available MODIS data and fire maps of the study area.

Fire frequency data of the study area from 2001 to 2004 were obtained from MODIS (Rapid Response System) Web fire mapper (NASA, 2004a) and imported to an Excel spreadsheet (Annex 1) and plotted on map (Figure 5).

To assess the level of fire recurrence (overlap) the original coordinates were rounded up to the nearest decimal point and sorted using Excel PivotTable and then grouped according to the number of time a fire has occurred on

the same site (unburnt, burnt 1 time, burnt 2 times and burnt 3 or more time) as shown in Annex 2.

The following procedure was used to locate the plots: A number was assigned to individual coordinate of fire point per group of fire frequency and using random number table, five numbers in each group was picked up to represent the central location of the 20 replicates sample plots, five in each of the four treatments. Given that some of the selected sample plots were in the vicinity of the already established PSP, adjustments of 20-50 m was made so that the sample plot was within the PSP as shown in Table 1.
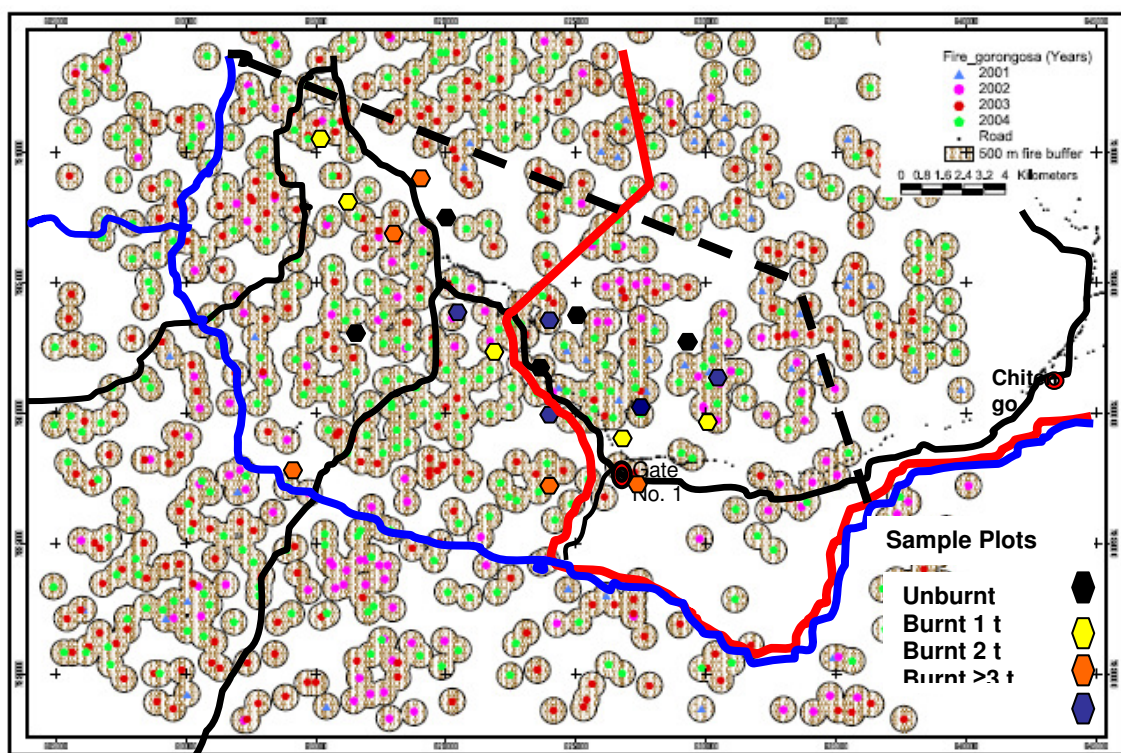

Figure 5: Fire pattern in the study area. The symbols show the year as recorded by MODIS. The cycle around each symbol is a 500 m area of influence of the fire. The approximate locations of sample plots are also shown by the hexagonal symbols.

The coordinates of the 20 sample plots were entered in a hand-held Garmen II plus GPS which was then used to track the points. At each point

circular plots of fixed area of 78.53 m² (r = 5 m) were demarcated using a tape measure.

Circular plots were used because of their relative advantage for having a single dimension (the plot radius) to define the perimeter. They also have advantage of having smallest area to perimeter ratio of any shape, which reduces the number of borderline trees. Fewer borderline trees reduce the number of measurement errors (Mussa et al. 2002). This plot size is considered to be adequate for understorey and regeneration sampling (Mussa et al. 2002, MacDicken, 1997).

Table 1: Sample Plots location in the study areas.

Fire Treatment	Plot ID	Location (WGS 84)	
		Latitude	Longitude
Unburnt	PO4 *	18º58′33.3′′	34º06′08.3′′
Unburnt	PO6	18º57′48.9′′	34º10′03.9′′
Unburnt	PM1	18º59′03.4′′	34º11′39.1′′
Unburnt	PM2	18º55′33.6′′	34º07′52.2′′
Unburnt	PM6	18º58′26.2′′	34º10′26.5′′
Burnt 1 time	PO2*	18º53′30.3′′	34º07′13.8′′
Burnt 1 time	PO3*	18º56′32.0′′	34º08′04.9′′
Burnt 1 time	PO7*	18º59′43.6′′	34º12′02.3′′
Burnt 1 time	P12*	18º54′48.7′′	34º05′00.6′′
Burnt 1 time	P13*	19º00′56.3′′	34º13′11.0′′
Burnt 2 times	PO1*	18º51′50.8′′	34º05′59.0′′
Burnt 2 times	P10*	18º52′47.5′′	34º05′34.7′′
Burnt 2 times	P14*	18º59′57.7′′	34º14′00.1′′
Burnt 2 times	PM3	18º57′12.9′′	34º09′27.9′′
Burnt 2 times	PM5	19º00′19.6′′	34º11′58.6′′
Burnt ≥ 3 times	PO5*	18º56′18.2′′	34º08′23.9′′
Burnt ≥ 3 times	PO8*	18º58′17.2′′	34º14′08.8′′
Burnt ≥ 3 times	P11*	18º54′47.2′′	34º06′27.3′′
Burnt ≥ 3 times	P15	18º58′33.9′′	34º12′39.6′′
Burnt ≥ 3 times	PM4	18º52′23.0′′	34º10′29.3′′

(*) = inside of the existing Permanent Sample Plots.

4.4   Data Collection

In each sample plot, measurements of basal diameter and height of all sprouts with basal diameter ≤40 mm were taken using callipers and tape measures. For dead or damaged sprouts only basal diameter were measure, and plants with diameter >40 mm were identified and counted.

No attempt was made to differentiate seedling from sprouts as often was not possible to discriminate between shoots from seeds and those from root sucker without destroying the plant.

Species identification in the field was facilitated by the knowledge of the field assistants Mr. Santos John and Mr. Cesar Citaube who new most of the plants by their local names. Later, using a list of local and Portuguese names their botanical names confirmed. Unidentified species in the field were collected and later identified using family key from the field guide books of Trees of Southern Africa (Palgrave, 2002; Van Wyk et all, 1997). All the species nomenclature follows Palgrave (2002).

4.5   Data Analysis

Analysis of variance (ANOVA) test was performed to assess the level of significance in differences between the averages of: (i) number of species, (ii) number of sprouts, (iii) basal area and (iv) number of sprout/species between the four treatments. All statistical tests were performed using Excel software.

The number of fire incidence per year between the Nhambita Community Land and Gorongosa National Park was calculated as the number of fires occurred in each year in proportion to the respective surface areas (Annex 4).

The number of species per sample plot and per treatment were counted and plotted on histogram and ANOVA performed (Annex 5).

The mean number of sprouts per treatment was calculated by dividing the total number of sprout by the number of stems/regrowth in each treatment and respective standard errors calculated. The sprout density was calculated by dividing the number of sprouts by the size of the sample plots (78.5 m²) (Annex 6). The totals of each treatment result were the summation of the five replicate.

Species dominance was calculated from the basal area (the surface area in m^2 ha^{-1} that is covered by wood). Basal area of each species was calculated from basal diameters using circumference surface formulae ($\pi * r^2$). The mean basal area of each plot was calculated as summation of individual sprout basal area divided by the number of sprout in the treatment and respective standard error calculated (Annex 7).

Coppicing effectiveness was calculated as the mean number of sprouts per plant. The mean number of sprouts per plant was calculated by averaging the number of sprouts per species across the five plots in each treatment. The sum of all specie's average gave the mean sprout/treatment. Coppicing effectiveness of all species was expressed as the mean number of sprouts per plant ± standard error (Annex 8).

Mortality of species which occurred across all the treatment was calculated as the proportion of dead sprouts that remained on the regrowth or stem.

Species richness or importance value (I.V.) is defined as the sum of the relative density, relative frequency and relative dominance (Curtis, 1959 cited by Mueller-Dombois and Ellenberg, 1974; Kershaw, 1980). Relative sprout density was calculated by dividing the number of sprouts of a species by the total number of sprouts of all species in each treatment and

multiplied by 100. The relative frequency of a species was calculated by dividing the frequency of a species by the sum frequency of all species in each treatment multiplied by 100. The relative dominance was calculated by dividing the dominance of a species by the dominance of all species in the treatment and multiplied by 100. The sum of the three parameters defines the Importance value of a particular species in each treatment.

5   RESULTS

In general fire incidence in the region are on increase and comparison between the two most important land use in the study area shows that fire incidences were high in the Nhambita Community Land (NCL) compared with those occurring inside the boundaries of the National Park (Figure 6).
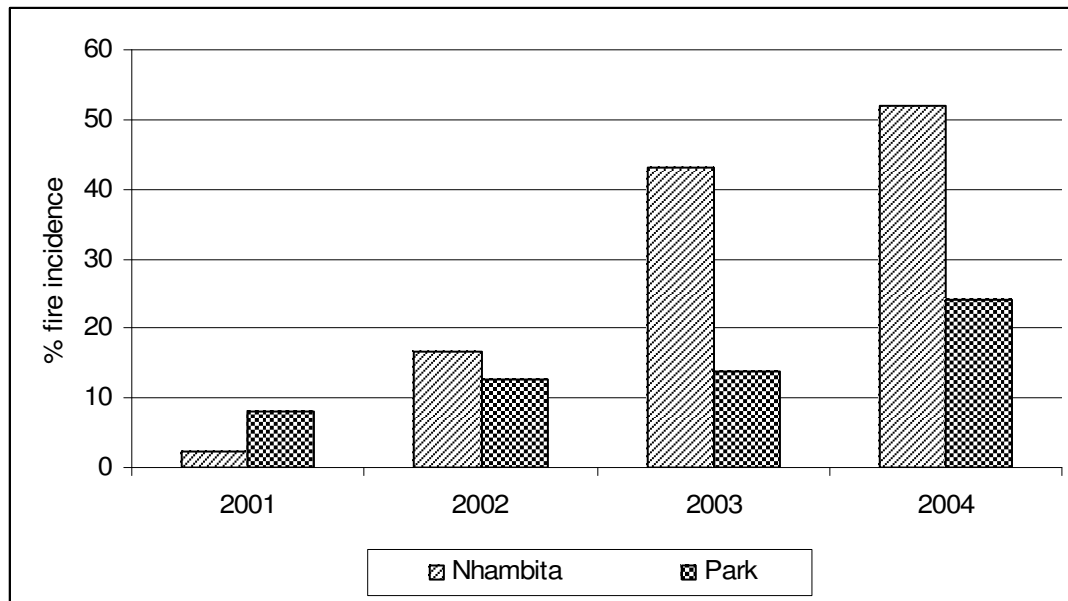

Figure 6: Annual fire incidence in Nhambita Community Land and Gorongosa National Park

In the past four years fires increased by 46.0% and 40.0% in the two land uses respectively (Table 2). In the past, fire incidence was much higher in the Park but the rate of incidence slowed down from 2001 and 2002. In NCL, fires are on increase at alarming rate since 2001.

Table 2: Fire incidences and percentage increase in Nhambita Community Land and Gorongosa National Park

| Year | Nhambita Community Land | | Gorongosa National Park | |
	% fire incidence	% increase	% fire incidence	% increase
2001	2.23	1.96	8.13	13.89
2002	16.76	14.71	12.60	21.53
2003	43.02	37.75	13.82	23.61
2004	51.96	45.59	23.98	40.97

The number of species increased 18%, 10% from the unburnt plots to the plots which were burnt twice respectively but decreased of 13% between the plots burnt twice and plots burnt more than three times (Figure 7). The difference between treatments were not significant ($F_{3, 130}=2.448$, $P>0.05$).
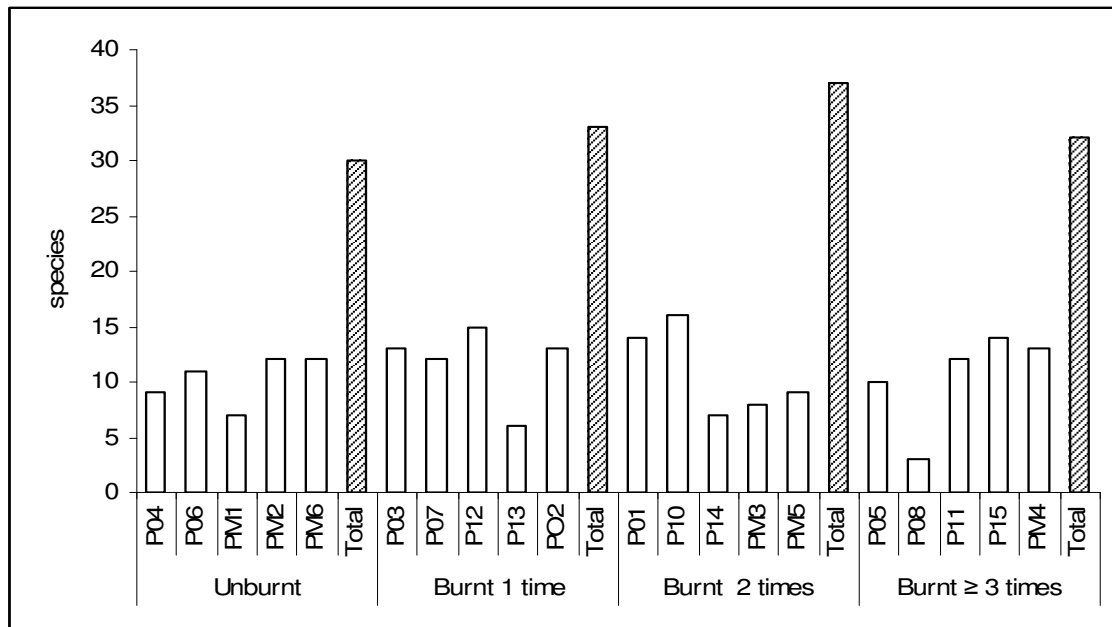

Figure 7 Effect of fire frequency on the number of species. Hatched bars indicate the total number of species in each treatment.

Despite the variation in the mean number of sprouts/plot and standard errors (SE), the treatment means shows a declining trend from the unburnt to the most burnt plots. The decline in mean sprouts per treatment were 47.5%, 15.0% and 25.0% from unburnt (73.40±26.25) (mean ± standard error) to the plots burnt three or more times (24.5±6.39) respectively (Figure 8-A). Similar results were observed with density (sprout/m⁻²) of sprouts (Figure 8-B). Although the sprout density declined steadily as fire frequency increased, the difference was not greatly significant ($F_{3, 129} = 2.32$, $P>0.05$). Between the unburnt and area burnt once plots there was remarkable

decreased of about 47.5% before levelling down in the subsequent fire treatment (9% and 10% decrease) respectively (Figure 8-B).
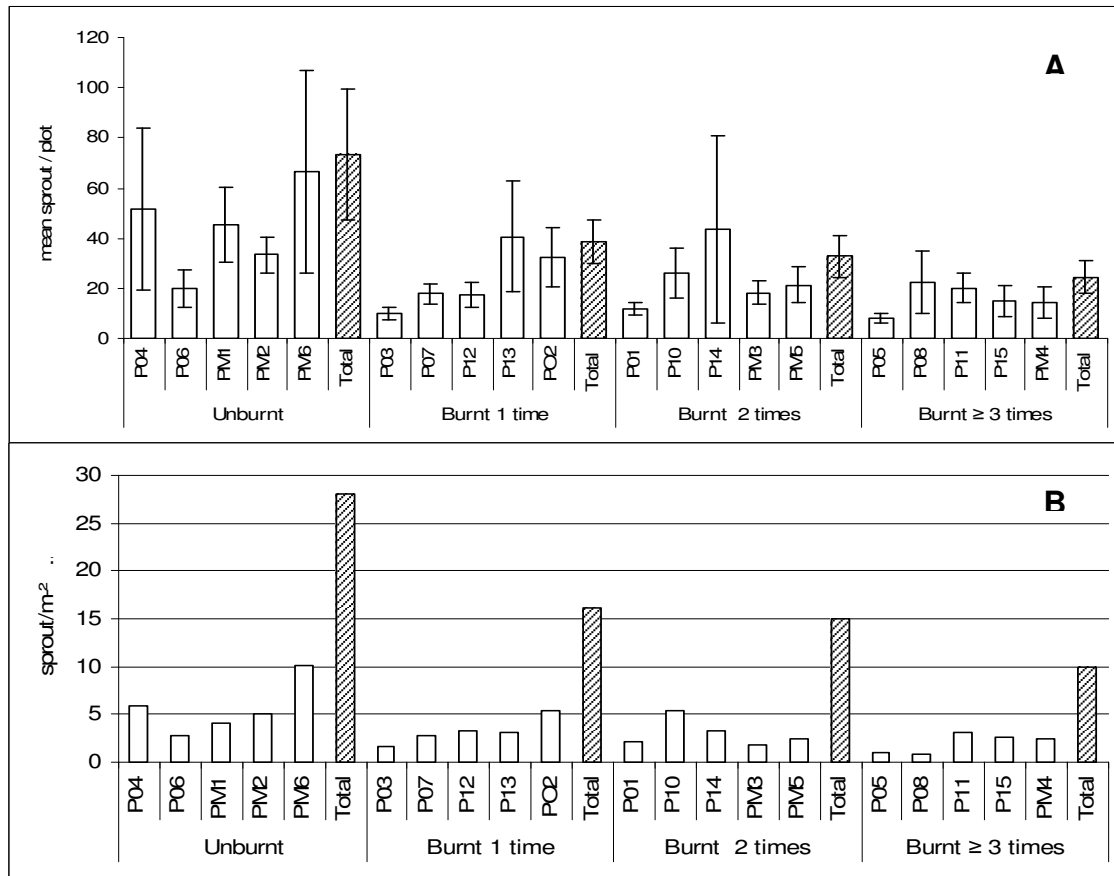

Figure 8: Effect of fire frequency on the number of species (A) and sprout density (B). The hatched bars indicate the treatment totals and the limits shown on the bars (A) represent the standard errors.

The overall sprout basal area (m^{-2}/ha^{-1}) as a measure of species dominance decreased steadily with increase in fire frequency (Figure 9) and the difference between the treatments was highly significant ($F_{3, 128} = 4.173$, $P<0.05$). Species dominance was 35.0% higher in unburnt plots (0.005196±0.001695) in relation to plots that were burnt once (0.003385±0.000828) and decreased subsequently as fire frequency increased.
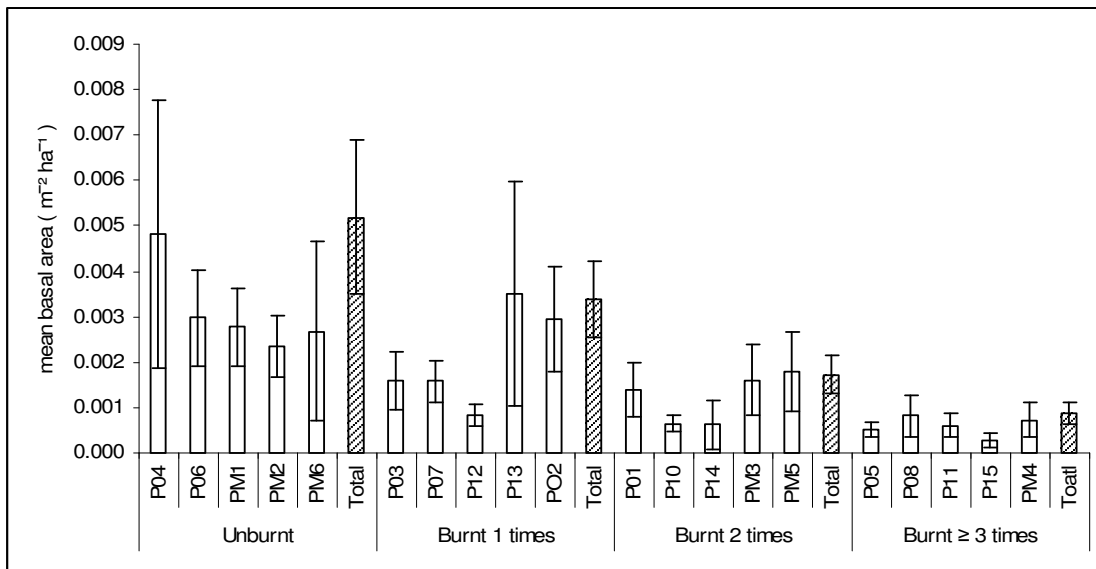
Figure 9: Effect of fire frequency on species dominance (basal area). Hatched bars represent totals and limits shown indicate the standard errors.

The numbers of sprouts produced by each plant or root sucker (coppicing effectiveness) showed a declining trend (Figure 10).
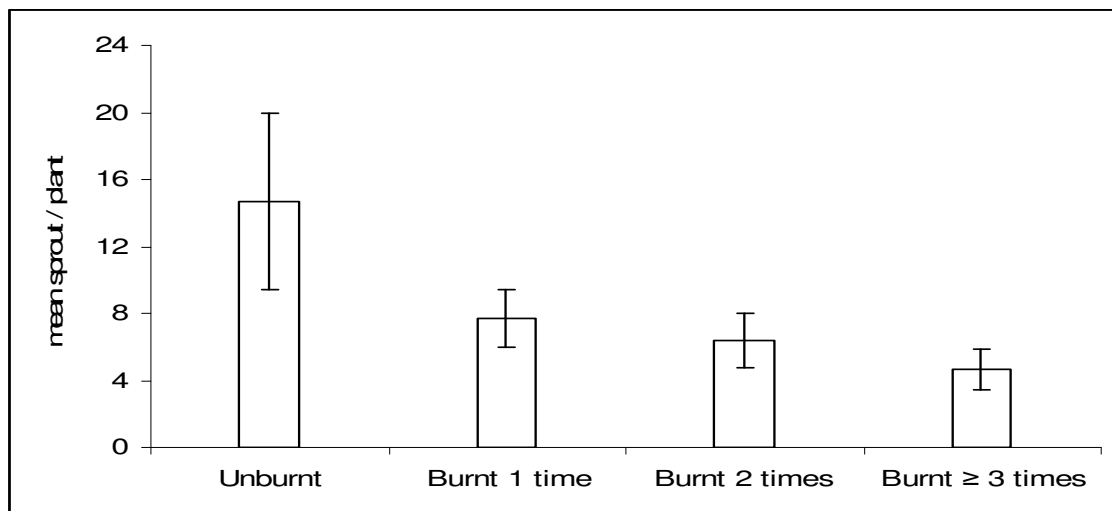

Figure 10: Average number of sprouts/plant. The limits shown represent the standard errors.

The difference between the unburnt (14.68 ± 5.25) and burnt one time plots (7.70 ± 1.74) was 47.5% while the decline in the subsequent fire frequency was only 17% for the plots burnt two time (6.37 ± 1.62) and 27% to plots

burnt more the three time (4.62 ± 1.22). The overall difference between treatments was not very significant ($F_{3, 130} = 2.44$, $P > 0.05$)

The effect of fire frequency on coppicing effectiveness of selected individual species is shown in Figure 11. Most species produced large numbers of sprouts in fire protected environment and declined with fire frequency.
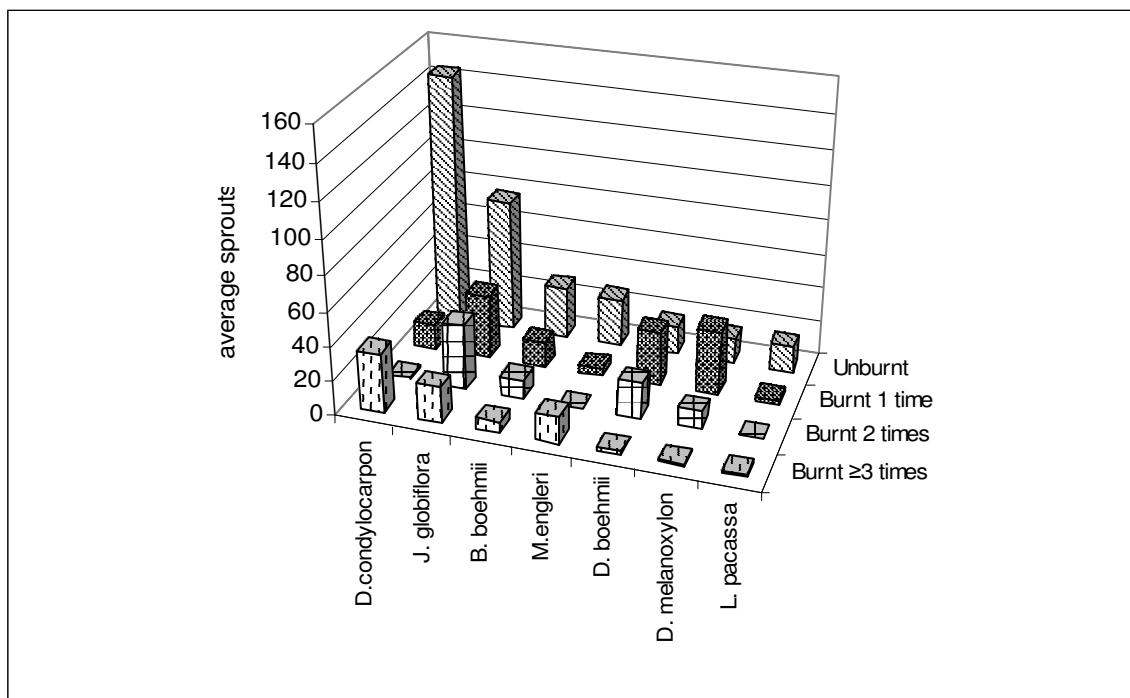

Figure 11: Average number of sprouts in selected species as influenced by fire frequency.

The number of sprouts produced by *Diplorhynchus condylocarpon* and *Julbernardia globiflora* for example declined considerably with fire frequency. In contrast *Lonchocarpus pacassa* was very sensitive to fire and most of the other species showed similar trend (Table 3)

Table 3 Coppicing effectiveness of selected species (mean ± standard error). The numbers in brackets represent the sample size.

Species	Unburnt	Burnt 1 time	Burnt 2 times	Burnt ≥ 3 times
B. boehmii	29.4±12.49(147)	14.2±10.31(71)	11±9.13(55)	6.6±4.4(33)
D. boehmii	17.2±17.20(86)	31.6±18.87(158)	20.8±9.44(104)	1.75±1.17(7)
D. melanoxylon	15.2±15.20(76)	37.4±25.24(187)	10.2±6.25(51)	1±0.7(5)
D. condylocarpon	145.2±88.81(726)	14.4±8.19(72)	1	34.4±14.9(172)
J. globiflora	76.8±57.33(384)	37.2±27.13(186)	37.6±32.72(188)	20.8±16.25(104)
L. pacassa	15.2±15.20(76)	2.8±2.80(14)	0	0
M. engleri	27.8±25.37(139)	4.6±2.44(23)	0	15.2±15.20(76)
P. rotundifolius	17.2±11.43(86)	7.6±7.60(38)	2.2±2.2(11)	2.6±1.94(13)
S. birrea	5.2±2.85(26)	2±2(10)	7.6±4.7(38)	0

The sprout mortality of some selected species (Figure 12) showed a general declining trend as fire frequency increased. *Holarrhena pubescens* had a marked decline of 23% between the unburnt and burnt twice plots despite an increase of 8% in plots burnt three or more time. On the other hand *Pterocarpus rotundifolius* was the least sensitive species or more fire-tolerant species. *Dalbergia boehmii* showed an unexpected increase from plots burnt once to burnt twice before a decrease of about 25% in plots burnt three or more time.
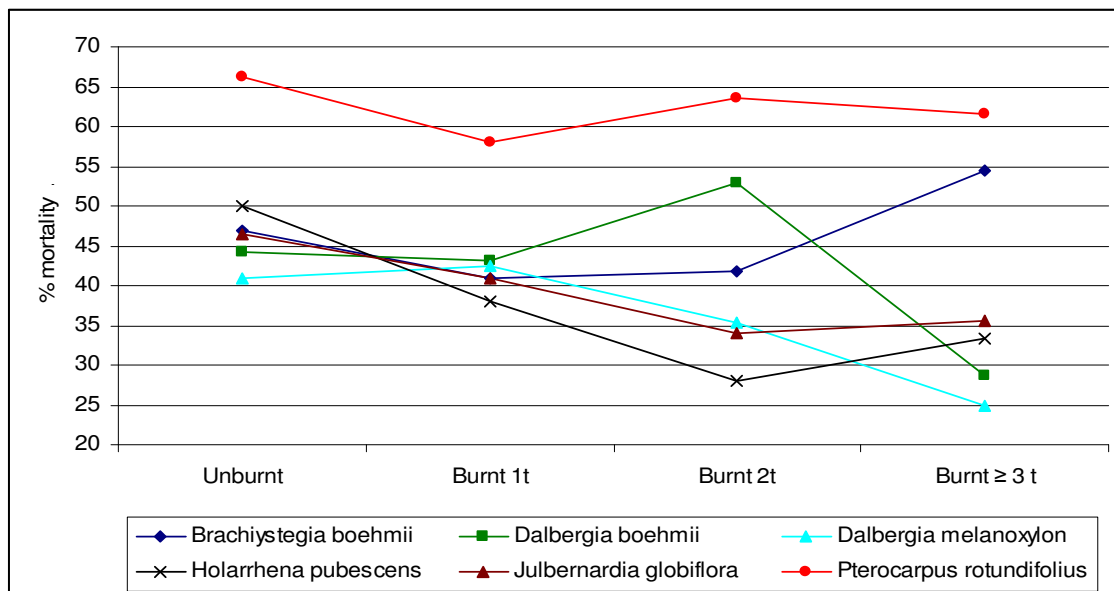
Figure 12: Percentage sprout mortality of selected species

The diversity index calculated as the Importance Value (I.V.) of the species shows a greater variability among species and between treatments (Table 4). In unburnt plots, *Dalbergia melanoxylon* had the highest Importance Value owing its high relative frequency and high relative density. Although moderately frequent, *Holarrhena pubescens* had lower dominance. In plots that were burnt once *Holarrhena pubescens* had the highest I.V. due to its higher relative frequency. Species G and *Dalbergia melanoxylon* were most relevant in plots burnt twice and three times due to their relative dominance and relative frequency respectively.

Table 4 : Importance Value (IV) of selected species.

Species	Relative Density	Relative Frequency	Relative Dominance	Importance Value (IV)	IV Rank
Unburnt					
Dalbergia melanoxylon	0.61	13.36	0.43	14.40	1
Holarrhena pubescens	0.32	7.07	0.16	7.55	2
Stereospermum kunthianum	0.01	0.31	2.52	2.85	3
Bauhinia petersiana	0.12	2.70	0.00	2.83	4
Burnt 1 time					

Holarrhena pubescens	0.27	3.42	0.92	4.61	1
Acacia sp3	0.00	0.00	4.48	4.48	2
Dalbergia boehmii	0.27	3.44	0.61	4.32	3
Crossopteryx febrifuga	0.23	2.91	0.00	3.14	4
Burnt 2 times					
Species G	0.01	0.17	6.80	6.98	1
Xeroderris stuhlmannii	0.01	0.13	5.59	5.73	2
Julbernardia globiflora	0.36	4.27	0.19	4.83	3
Holarrhena pubescens	0.29	3.46	0.05	3.80	4
Burnt ≥ 3 times					
Dalbergia melanoxylon	0.40	3.16	0.20	3.77	1
Holarrhena pubescens	0.24	1.91	0.30	2.46	2
Brachystegia boehmii	0.14	1.09	0.86	2.09	3
Millettia stuhlmannii	0.18	1.40	0.35	1.93	4

Holarrhena pubescens was most important in plots burnt one time and decreased it importance in all other treatments. *Stereospermum kunthianum* was important in unburnt plots and showed very low importance in other treatments.

6   DISCUSSION

Contrary to belief that fire pattern in Mozambique did not change in the last 20 year (Saket, 1999), fire incidences are on increase year after year. In the study area fire incidence are on increase at an alarming rate with areas out side the conservation area mostly affected. Owing the increasing effectiveness in the management of the Gorongosa National Park, fire incidences rate has reduced (Table 2). In community land in the buffer zone of the Park fires are still abundant and the trend is to increase as more people occupy these areas. The traditional slash-and-burn cultivation, charcoal burning and population increase are some of the causes.

The increasing number of species in frequently burnt area happens in the presence of fire tolerant species. As discussed earlier, these species are capable of surviving regular late dry season fires as adults, saplings or

regrowth and are intolerant to shade (Trapnell, 1959, Lawton, 1978). Species such as *Acacia polyanthus, Cleistochlamy kirkii, Combretum molle, Lecaniodiscus fraxinifolius* were among additional species that occurred in the most burnt plots and did not occur in fire protected plots.

Figure 13: Dense grass understorey in unburnt plot.

The Lower number of species in fire protected plots was consistent with the findings in Kafubu coppice plots in Zambia (Chidumayo, 1988) and was attributed mainly to the loss of large number of understorey species in the coppice due to competition for light. Dense grass due to low herbivory in the study area may have contributed to the results by preventing the regeneration on non-shade tolerant species (Figure 10).

The results were inconsistent with the findings in Ghana where in a 27 years old coppice protected from fire had higher species density and diversity than (i) before felling and (ii) in an early burnt plot (Trapnell, 1959, Brookma-Amissah et al 1980).

In contrast to the increase in species diversity as fire frequency increased, the number of sprouts decreased significantly between the unburnt and burnt plots. In unburnt plots, species produced 47.5% more sprouts than in

burnt once plots and most species had highest chance to survive to the next generation (Figure 11). Although at slower rate, the decrease in the number of sprout continued between the subsequent treatments.

The results support the evidence that fire frequency reduce woody plant density by killing or suppressing individuals in smaller size classes (Trapnell, 1959, Brookma-Amissah et al. 1980).

Stumps and root suckers produce large number of shoots or sprouts (Figure 14) but during the establishment period but this number decrease due to inter-shoot competition and with age of the regrowth (Chidumayo, 1988; Frost and Robertson, 1977; Luoga *et al*, 2002) and fires may play additional stress by killing those individuals that have not attained sufficient height or bark thickness to sustain the heat.

Species dominance was higher in the unburnt plots reflecting the sprout density and coppicing effectiveness of the species.

Figure 14: Large number of sprouts coppicing from root sucker. (*Diplorhynchus condylocarpon*)

Apart from the physical reduction sprout density, frequent fires have an effect on perennating organs and root food reserves which can cause die-back of shoots as result of depletion root reserve of parent plats due to systematic and continuous effect of fire (Figure 15), (Chidumayo, 1898, Frost and Robertson 1977, Kennard et al 2002). The increased grass production, leads to higher dry season fuel loads, more frequent and intense fires which cause further suppression of woody plants (Desanker *et al* 1995).

Species dominance was generally very low compared with results found in Zambia where in fire protected plots of 9 and 13 years old the coppice basal area was between 6.19 - 0.72 m² ha^{-1} (Chidumayo, 1988). In Ghana, similar work found a basal area of 0.51 m² ha^{-1} which is still higher than the current results (Brookman-amissah *et ai.* 1980). Shorter fire return interval, low soil fertility and low rainfall can be the contributing factors.

Figure 15: Effect of successive burning on *Julbernardia globiflora*. The red arrows indicate fire scares of dead sprouts.

Coppicing effectiveness per treatment also declined with fire frequency and the results were much higher compared with those found in Zambia (Chidumayo, 1988). The overall average coppicing effectiveness was 8.8 sprout/plant compared with 5.5 sprouts/plant found in Zambia. Miombo coppicing effectiveness is affected by the age of the woodland and fire, according to Chidumayo (1988) may either slow sprouting by deflecting back the production process or accelerate by producing an equal or larger number of replacement sprouts (Figure 16).

Figure 16 : Resprouting in a recently burnt site in the study area.

Species response to fires varies and depend not only the physiological and developmental stage of the plant but also on the time fires occur and events in the interval between fires (drought, floods herbivory) that influence the fuel load and consequently the intensity of the fire (Frost and Robertson 1977). *Diplorhynchus condylocarpon* and followed by *Julbernardia globiflora* have the highest coppicing effectiveness (Figure 10). Similarly *Diplorhynchus condylocarpon* was most fire tolerant species and was found to be dominant in unburnt area (0.0405±0.01) but also doing very well in the most frequently burnt areas (Figure 9).

In terms of their contribution to the diversity, species shift positions according to their levels of fire tolerance and this if reflected in their dominance, density and frequency. *Dalbergia melanoxylon* despite of its thin back, appear to do reasonably well in both environment of fires. In unburnt plots it had the highest relative frequency and relative density so it

did in the most frequently burnt plots. *Dalbergia melanoxylon* is not a good sprouter but relays on its high seed production and fire play an important role in triggering germination. It is also considered to be an invasive in drought stricken areas and areas subject to frequent fires.

Figure 17 Bark of *Diplorhynchus condylocarpon* stem provides in fire-tolerance capacity.

Holarrhena pubescens follows the same pattern and is most dominant in frequently burnt plots although it appears to be a cosmopolitan species. *Diplorhynchus condylocarpon* which does not appear in the list of the first four species (Table 3) is regarded as very adapted to fire environment. This capacity is attributed to its characteristic tick bark (Figure 17) and deep meristems which can withstand frequent fire and can survive ¾ of its stem damaged.

7   CONCLUSIONS

Fire is one of the most important factors influencing the global vegetation pattern and it has been associated with the development and maintenance of the floristic and structural composition tropical savannas. Even though considerable practical knowledge is available on its application in savanna management, much still needs to be learnt about its different effects and how these interact with other ecological processes to influence savanna dynamics.

This study found that fire frequency was high and on increase in the community and still significant inside the National Park.

Many of the species could tolerate burning, and many of them seemed to thrive with some burning. Examples of fire tolerant specie are *Dalbergia melanoxylon, Acacia polyacantha, Cleistochlamys kirkii, Crossopteryx febrifuga, Pterocarpus rotundifolius, and Julbernardia globiflora.* Examples of fire intolerant species are *Bauhinia petersiana, Pseudolachnostylis maprouneifolia, Millettia stuhlmannii* and *Terminalia stenostachya.*

Species number found in plots which were (i) unburnt, (ii) burnt one time, (iii) burnt two times and (iv) burnt three times and more were 30, 34, 37, 32 respectively, although the total number of sprouts and total basal area declined with burning.

Sprout stocking rate, dominance and coppicing effectiveness of most species showed a declining trend as fire frequency increased. Sprout stocking rate decreased by 47.5% between unburnt and burnt once plots. Similarly basal area decreased 35.0%, the number of sprouts produced per plant decreased 47.5% between the unburnt and burnt once plots.

The use of remote sensing information has permitted to identify fire recurrence in the study area but there was some uncertainty with regards to it precision as most of the plots seemed to be burnt regularly. In this respect, future work should be preceded by an in-depth GIS analysis and interpretation and areas of concerns should be annually monitored by ground truthing to be able to validate the extent and frequency of fires.

With regards to the effect of fires, more consideration needs to be given to the possible contingency of effects, not only on the intensity, timing and frequency of fire, but also on the state of the organisms at the time, as well as on subsequent interactions with rainfall, drought, and other fires.

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