EFFECTS OF FIRE FREQUENCY ON PLANT SPECIES DIVERSITY AND COMPOSITION IN QUEEN ELIZABETH NATIONAL PARK, SOUTHWESTERN UGANDA

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DECLARATION

I declare that the information contained in this dissertation is a result of my own work and has never been submitted for an award to any University or institution of higher learning. Statements from other people's work have appropriately been acknowledged.

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DEDICATION

This work is dedicated to my beloved Dad Harry Shaffy Dradiku whose dream has always been to see me attain greater heights in academics.

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LIST OF ACRONYMS

ANOVA	=	Analysis Of Variance
GMP	=	General Management Plan
IDH	=	Intermediate Disturbance Hypothesis
IVIs	=	Importance Value Indices
QENP	=	Queen Elizabeth National Park
QEPA	=	Queen Elizabeth Protected Area
RCs	=	Relative Covers
RDs	=	Relative Densities
RFs	=	Relative Frequencies
UWA	=	Uganda Wildlife Authority
PA	=	Protected Area

ABSTRACT

In savanna grasslands worldwide, periodic fires are important forces affecting plant species composition, richness, diversity and cover (Govender *et al*., 2006). It is not clear, however, how variations in fire frequency have affected these parameters in the Queen Elizabeth National Park (QENP) over time. This study investigated the effects of fire frequency on plant species diversity and composition in the QENP. Fire frequency data was derived from a burn scar map of QENP produced by the Woods Hole Research Centre. A field visit was conducted to categorize the vegetation of the study area after which the study sites were selected based on similarity in the burning frequency and nature of vegetation. The modified-Whittaker nested vegetation sampling method was then used to sample the vegetation in the study sites.

Data analysis was done using Genstat 5 Release 3.22.

Results showed that species richness of all vegetation varied quadratically with fire frequency ($y = 0.254x^2 – 4.2843x + 26.456$; $R^2= 0.9666$). Similarly, species diversity of all vegetation changed quadratically with fire frequency ($y = 0.0917x^2 – 1.3671x + 9.5838$; $R^2= 0.7464$).

Percentage cover of all vegetation also varied quadratically with ($y = -0.9784x^2 + 14.102x + 42.037$; $R^2= 0.9858$)

Plant species richness and vegetation cover were strongly correlated with fire frequency ($R^2 > 0.9$)

Fire frequency produced a change in plant species composition with herbaceous plants dominating in all the study sites. The number of shrubs and trees recorded generally decreased with fire frequency. The study among others recommended continuous monitoring of the effects of the current controlled burning program on vegetation in QENP and the need to expand the study by combining the effects of fire with those of other factors such as grazing, soils, rainfall regime and topography.

CHAPTER ONE

INTRODUCTION

1.1 Background to the study

Fire is a widely studied phenomenon in African savanna ecosystems (Wein & Edroma, 1986; Roques, O'Connor & Watkinson, 2001; Meyer *et al*., 2005; Govender, Trollope & Van Wilgen, 2006) but its effects are still not well understood. How fires affect vegetation has been and continues to be a matter of discussion (Hopkins, 1965a; Edroma, 1981a, 1981b; Stein *et al.,* 1990). The pros and cons of fire as a tool in national park management have often been disputed and even now there is probably no general agreement. Lemon (1968) summarises some of its beneficial effects, which relate to the improved palatability and nutritional value of the new grass which springs up after the burn. In many rangelands all over the world, fire has been used as an instrument in the management of range vegetation in various ways; to reduce bush cover, influence changes in species composition, to improve the quality of forage to animals, and to reduce disease-causing pests (Okello *et al.,* 2007). Fire influences changes in vegetation on its own and in conjunction with other factors like herbivory, climate, and soils (Whelan, 1995).

Vegetation fires are considered natural phenomena in many parts of the world and have shaped the landscape for millennia across entire continents (Bond & Van Wilgen, 1996). Fire is hence an important determinant of plant diversity and vegetation structure in regions where it occurs (Bond, 1997; DeBano *et al.,* 1998). As anthropogenic burning has become predominant (DeBano *et al*., 1998), the ecological impact of certain fire regimes is the subject of intense discussions. Nevertheless, relatively few studies have tried to identify the effects of certain fire regimes on vegetation (e.g. Russell-Smith *et al*., 1998; Uys *et al.,* 2004; Watson & Wardell-Johnson, 2004) and many of these studies show conflicting results because of the variety of the investigated vegetation types and fire regimes. Therefore, given the interaction between people, fire and vegetation in most parts of the world, studies considering local fire regimes on certain vegetation types are required to fully understand the ecological implications of a specific burning regime (Bond & Van Wilgen, 1996; Uys *et al*., 2004).

Fires have played an important role in shaping the plant species composition of East African grasslands (Heady & Heady, 1982). Indigenous people set fires for many reasons including the beliefs that fire brings rain, controls shrub encroachment, kills insects and facilitates hunting.

Recent studies of grasslands in southern Africa and Australia have shown only minor effects of fire frequency on species composition because of the general dominance of fire-tolerant species and the loss of disturbance-intolerant species in the flora (Morgan, 1999; Uys *et al*., 2004).

While most research regarding changes in species composition focus on the shrub-grassland interface, fires can also promote changes within grasslands. The purpose of this study was to assess the effects of fire frequency on plant species diversity and composition in Queen Elizabeth National Park (QENP).

1.2 Research problem

Jaksic-Born (2004) observed that, the QENP protected area is susceptible to fires due to the narrow and elongated shape of the park which results in a disproportionately long border at risk for man-made fires. Another problematic factor is the high human population in the park and in adjacent areas which expose the park to all kinds of (illegal) human activities – often accompanied by intentional burning. These problems are known and were already stated over 30 years ago by Eltrigham (1976).
Eltrigham investigated the fire regime in the park in 1971- 1973. He documented the frequency and extent of burns for 3 years. During this study, he evaluated a maximum percentage of 33.7% of burnt areas in the park in 1971; 1972 and 1973 gave less percentages (Eltrigham, 1976). This situation has changed significantly during the last decades. All information available including the data evaluated in 2003 and 2004 indicate

a high increase in extent and frequency of burns in QENP (Jaksic-Born, 2004). Apart from the extent of areas which are yearly affected by fire, the frequency of burns in different areas has also increased since the 1970s.

While most of the areas were burnt only once a year in 1971-1973, some only in two years (Eltrigham, 1976), at present almost all the areas in QENP are burnt atleast once a year, but many of them even twice a year.

The fire regime in the study area would not be that worrisome considering only the extent of burnt areas. Different areas are mostly burnt alternatively in the first or the second dry season of the year, with the exception of some parts in the vicinity of villages that are repeatedly burnt in one dry season, and the crater region which is burnt regularly every dry season. However, there are several problematic aspects that amplify the adverse effects of the present fire regime. The most worrisome interms of ecology is the increasing spread of fire resistant plant species ("*increasers*"). In most cases, the so called *increasers* are a direct result of frequent burning (Sabiti & Wein 1988; Edroma 1984). Fire resistant grass/plant species are significantly less palatable for grazers and browsers because of low food value. One of the consequences is a reduced carrying capacity for wildlife in affected habitats, hence, the grazing pressure by wild herbivores decreases. Subsequently, available fuel will accumulate and fires easily ignite. Every new burn (at inappropriate time, mostly early burns) will add on the spread of increasers (species that increase with fire) and suppress the development of more palatable plant species. The ecological problem of spread of increasers should not be underestimated.

It is thus crucial with regard to conservation of wildlife and natural resources, particularly in small, spatially restricted areas like QENP, that a new assessment, if not a comprehensive study of the species composition of the grasslands in QENP and the coverage by increasers in fire prone areas should accompany the efforts towards efficient fire management. Such an assessment will determine the urgency and the type of measures to be taken in different parts of the park in terms of fire management.

This study on the effects of fire frequency on plant species richness, diversity and composition in QENP formed part of the efforts to conduct such an assessment.

1.3 Objectives of the study

1.3.1 Overall objective

The overall objective of this study was to assess the effects of fire frequency on species richness, diversity and composition in QENP.

1.3.2 Specific objectives

The specific objectives of this study were therefore to:

1. assess the effects of fire frequency on species richness and diversity in QENP
2. assess the effects of fire frequency on plant species composition in QENP.
3. assess the effects of fire frequency on vegetation cover in QENP.

1.4 Scope of the study

The study was carried out in the QENP. It focused on the effects of fire frequency on plant species composition, richness and diversity, and vegetation cover in the selected areas of; the Channel track, Kasenyi Track three, PIDA and the Crater area.

1.5 Justification of the study

Fire is a major factor in the development and maintenance of most grasslands and could even have a bigger influence in combination with climate, substrate, topography, and herbivory (Jennifer *et al.*, 2001). The fire regimes in savanna ecosystems is changing from one characterized by very frequent and probably low intensity to one in which fires are relatively frequent and of potentially more severity (Uhl & Kauffman, 1990). UWA (2005) reported that, the frequency of uncontrolled fires in QENP is increasing due to especially the presence of communities living in and outside the park and fishing villages. QEPA has 11 fishing village enclaves categorized as sanctuaries. Together, these have an estimated population of 30,000 people (UWA, 2005). There are 58 parishes bordering with the park with an estimate of more than 50,000 people (UWA, 2005). These fishing villages together with the network of public roads inside the PA, contribute to the problem of uncontrolled fires. Although fire can have positive effects, the primary problem in QENP is that fires occur too frequently and cover large areas thus threatening biodiversity. Moreover not many studies have been done on the effects of fire frequency on species richness, diversity and composition in QENP hence the need for this study.

This study was expected to add to our knowledge of the ecology of fires in QENP with specific reference to the effects of fire frequency on species richness, diversity and composition in the savanna ecosystem.

The study sought to inform Park management on improvement of the fire management plans often carried out in the park.

CHAPTER TWO

LITERATURE REVIEW

2.1 Introduction

The literature reviewed in this chapter encompasses the effects of fire frequency on species diversity and composition in general, with some aspects focusing on the Queen Elizabeth National Park in particular.

2.2 Effects of fire frequency on species richness and diversity

Competitive exclusion, disturbance processes, and environmental heterogeneity are three key determinants of plant species diversity in terrestrial ecosystems (Whittaker, 1975; Connell, 1978; Huston, 1994). Competitive exclusion reduces species diversity as strong competitors first suppress lesser competitors and later drive them to local extinction.

Disturbances can reduce plant species diversity by eliminating disturbance-sensitive species, increase species diversity by opening up growing space and resources for use by colonizing species, maintain species richness by slowing or preventing competitive exclusion, and alter spatial heterogeneity in plant community composition (Huston, 1994).

Ecological theory predicts important linkages between disturbance frequency and species diversity (Grime, 1973; Connell, 1978). The intermediate disturbance hypothesis (IDH) (Connell, 1978; Huston, 1979) predicts that species diversity will be maximized at intermediate disturbance frequencies and severities that minimize species losses due to competitive exclusion and direct disturbance effects, while maintaining opportunities for establishment of new species. Although the IDH is well studied (Roxburgh *et al*., 2004), species diversity responses to fire frequency often do not follow predicted patterns.

Frequent fires and grazing by large ungulates are important processes affecting the biological diversity and heterogeneity of grassland vegetation (Collins, 1992, Fuhlendorf & Smeins, 1999).

In savannas and woodlands, plant species richness often increases with increasing fire frequency up to biennial (Tester, 1989; Brockway & Lewis, 1997) or annual (Walker & Peet, 1983) fire frequencies.

Fire frequency may also influence under-story species richness indirectly in savanna, woodland, and forest ecosystems through its influence on over-story tree density and canopy cover. Huston (1994) proposed that high fire frequencies promote high species richness in savannas by preventing competitive exclusion of grasses and forbs by woody plants. Furthermore, under-story plant community composition and diversity in savannas often varies across gradients in under-story light and soil resources associated with spatial variability in over-story tree cover (Scholes & Archer, 1997; McPherson, 1997; Brewer, 1998).

High species richness in fire-maintained savannas may also be the result of intermediate resource availability that reduces dominance and competitive exclusion by both woody plants and grasses and allows greater herb species richness (Leach & Givnish, 1999).

2.3 Effects of fire frequency on species composition

The generally accepted trend of the structural response of vegetation to fire frequency is that the more frequent the fire, the greater the herbaceous component, and the less frequent the fire, the more abundant the woody component (Booysen & Tainton, 1984). The suppression of woody species by high fire frequency is described as the 'Gulliver effect' (Bond & Van Wilgen, 1996; Higgins, Bond & Trollope, 2000), with Gullivers being stunted multi-stemmed shrubs, which could come to dominance as adults but struggle to emerge from the herbaceous layer as juveniles because of regular fire events in this layer. However, high fire frequency can also reduce the mortality of large trees because of lower fire intensity and therefore flame heights, because of lower amounts of accumulated fuel loads (Michael *et al.*, 2007).

Fires have played an important role in shaping the plant species composition of East African grasslands (Heady & Heady, 1982). Most savanna plants have become adapted to endure fire through natural selection, and this is especially true for grasses (Gillon, 1983).

In general, species that grow rapidly are better able to endure repeated burning. Fire stimulates tillering in tropical grasses, including East African species (Pratt & Gwynne, 1977; Edroma, 1984). While late dry-season fires generally increase grass production, they are known to remove younger woody species from East African savannas (Afolayan, 1978; Norton-Griffiths, 1979; Dublin, Sinclair & McGlade, 1990). Change in timing or frequency of fires can alter the response of herbaceous or woody species, thereby producing a change in species composition. While most research regarding changes in species composition focus on the shrub-grassland interface, fires can also promote changes within grasslands (Sinclair & McGlade, 1990).
The amount of protection a given grass species provides to its tiller primordial determines some of the mortality and growth differences found among different grass species in response to fire (Frost & Robertson, 1987). For example, species that exhibited a delay in tiller elevation well after the rains had begun were much less vulnerable to fire than species that elevate tillers soon after the first rain. As a result, plant species such as *Panicum maximum* Jacq. can be eliminated by ‘frequent fires, *Hyparrhenia hirta* (L.) Stapf and *Chrysopogon plumulosus* Hochst. are only moderately vulnerable, and *Themeda triandra* Forssk. can only occur in areas that receive frequent burning (Michael *et al.*, 2002).

Fires damage or kill woody plants, reduce woody plant cover, and reduce woody species frequencies (Waldrop *et al*., 1992; Peterson, 1998), and under-story woody species richness is maximized at the lowest fire frequencies. In contrast, most grasses sustain little damage from fires, so effective fire severity is low, and species richness in the grasses group is maximized at very high fire frequencies that minimize competition from woody plants (Peterson, 1998).

High fire frequencies kill (or prevent establishment of) seedlings and saplings of fire-sensitive species, prevent seedlings and saplings from growing into the over-story, and slowly kill over-story trees, thereby limiting over-story tree species composition to highly fire resistant species (Peterson & Reich, 2001).

Okello *et al*. (2007) suggested that fire may also reduce tree recruitment, although even shallow burial of seeds appears to provide considerable protection from fire in fire-adapted species.

Fire may offer a useful option to open the woody layer, but perhaps mostly in the presence of large native browsers (Mills & Fey, 2005).

The higher the fire frequency, the more difficult it is then for these woody species to emerge and to escape from the fire (Michael *et al.,* 2007). Therefore, large trees are scarce under high fire frequency.

Reduced fire frequency may also permit woody species to increase within the constraint imposed by soils. Conversely, increasing the average fire frequency throughout savanna ecosystems may have detrimental effects on those grass species that are just able to persist under the current fire regime (Michael *et al.,* 2002).

While mature trees can survive frequent fires in African savannas, utilization by elephants and other animals can damage trees and allow fires to burn exposed areas of wood (Yeaton, 1988). Tree scars tend to become larger with successive fires, and the trees eventually become structurally weakened and collapse.

Frequent, regular fires would also serve to prevent smaller trees from developing into larger ones, especially in combination with browsing (Bond & van Wilgen, 1996). Thus, increased mortality and declining recruitment as a result of a combination of frequent, regular burning and increasing numbers of elephants appear to have result in an overall decline in the number of large trees in African savannas (Bond & van Wilgen, 1996). Young plants are more badly affected by fires than mature ones (Naidu & Sribasuki, 1994).

Although fires can profoundly influence species composition in East African grasslands (Frost & Robertson, 1987), this influence is based on differential tolerance of plant species to fire and their ability to recover after a fire.

However, the effects of fire on plants could also probably be species-specific and related to the duration of fire exposure (Sabiiti & Wein, 1987; Mucunguzi & Oryem-Origa, 1996; Radford, Nicholas & Brown, 2001).

Repeated burning can have profound impacts on savanna ecosystems structure (Bond & Archibald, 2003; Nangendo *et al.*, 2005; Govender *et al*., 2006)

2.4 Fire effects on individual herbaceous plants, shrubs and tree species

Fire is a dynamic process that affects herbaceous plants especially grasses in a variety of different ways. When determining the effects of fire on individual species and ecosystems, it is important to understand the condition of the plant community and individual species existing before a fire occurs. That is, the impact of fire on an individual plant species or communities may increase if the community has been subjected to other disturbances such as drought, disease, insect infestations, overgrazing, or a combination of these factors (Brown *et al.,* 2000).

Brown *et al.,* (2000) observed that the response of individual grasses and other herbs to fire varies significantly between, and within, species and is dependant on the parameter being measured. Moreover, this response is influenced by a variety of fire parameters including intensity, severity (amount of organic matter consumed), residence time, soil heating, season of burn, and time since last fire, all of which can vary significantly among fires and within a fire (Conrad *et al.,* 1996). These variations can and will cause differences in the response of individual species and the community as a whole. In addition, numerous physical and climatic factors (fuel condition, weather, slope, and aspect) as well as biological factors (plant morphology and physiology) will influence post-fire effects on plant communities. This includes direct effects such as the ability of individual species to resist the heat of a fire (depending on age and seasonality) and the mechanisms by which they recover after fire (Conrad *et al.,* 1996).

Finally, in addition to fire parameters and individual species response, numerous external factors such as post-fire weather, post-fire animal use, and plant competition can also determine how the grassland and individual species will respond to fire. Common effects include grass mortality, increased flowering and seed production, increased and decreased productivity, increased forage quality, reduced/increased vigor and abundance of dominant species as well as numerous communal effects, including increased herb layer diversity and cover, when compared to long-term unburned communities (National Wildfire Coordinating Group, 2001).

The response of trees and shrubs to fire varies significantly between and within species, and is dependant on the parameter being measured. Moreover, this response is influenced by a variety of fire parameters including intensity, severity (e.g., amount of organic matter consumed), residence time, soil heating, season of burn, and time since last fire, all of which can vary significantly among fires and within a fire (Brown *et al.,* 2000) . These variations will cause differences in how individuals and the community as a whole respond. In addition, numerous physical and climatic factors (e.g., fuel condition, weather, slope, and aspect) as well as biological factors (plant morphology and physiology) also influence post-fire effects on plant communities. This includes direct effects such as the ability of individual species to resist the heat of a fire (depending on age and seasonality) and the mechanisms by which they recover after fire (Brown *et al.*, 2000).

In addition to fire parameters and individual species response, numerous external factors such as post-fire weather, post-fire animal use, and plant competition, can also determine how the grassland and individual species will respond to a fire. Common effects include plant mortality, increased flowering, seed production and numerous communal affects (Conrad *et al.,* 1996).

Seasonality, site characteristics such as fuel loading, geographic and climatic factors all play an integral role in determining fire intensity and severity and, when combined with plant morphology, all influence the impact of fire on shrubs (Brown *et al.,* 2000). Whether or not this impact is negative or positive is species dependant, with many species experiencing both positive and negative effects depending on fire severity. Seasonality affects fire severity by influencing the moisture content in the target plant and in the fuels surrounding the plant. In general, as moisture content of bark, leaves and twigs increases, so does the amount of heat required to raise them to ignition temperature (Brown *et al.*, 2000).

Individual tree and shrub mortality typically occurs when several plant parts are damaged. For example, crown damage combined with a significant amount of cambium and/or root damage is more likely to result in death than if only one of these components was damaged (NWCG, 2001). During intense/severe fires, tree and shrub mortality may be instantaneous. Under less severe situations, death may not occur or be delayed several years. Where death is delayed several years, it is often caused by secondary disturbances, such as infections by insects and pathogens that are able to enter the tree or shrub either due to decreased resistance or through the provision of entry points (wound sites) (Conrad *et al.*, 1996).

CHAPTER THREE

MATERIALS AND METHODS

3.1 Study area

Queen Elizabeth National Park (QENP) covers an area of 1978 Km^2, is situated astride the equator in southwestern Uganda and occupies the floor of the western arm of the East African Great Rift Valley. It is surrounded by the Rwenzori Mountains to the north, Lakes Edward and George (900 m above sea level) to the southwest and northeast, respectively, and the escarpments of the Rift Valley to the east and west (Lock, 1988). The Park has an undulating topography varying from 900 to 1345 m above sea level. The hilly crater region (192 Km^2) is located in the northern part of the Park about 6 km north of former Uganda Institute of Ecology. This consists of a series of craters formed 8,000 - 10,000 years ago in a short duration with violent explosion of volcanic activity. Numerous small streams flowing from the mountain and the eastern escarpment traverse the Park and drain into the two lakes. The lake and river systems form a unique feature providing the area with abundant water (Lock, 1988).

The climate is typically equatorial with two wet (March-June, September-November) and two dry (December-February, July-August) seasons. Temperature fluctuations are minor, ranging from a mean minimum of 16^{o}C in March to a maximum of 27^{o}C in January and the mean monthly rainfall from 22 stations varies from 32 mm to 120 mm (Lock, 1977b).

The vegetation in the Park has been described by Lock (1977b, 1988) who distinguished four major types of vegetation. These include Maramagambo forest (tropical rain forest); woodland, dominated by species of *Acacia, Balanites* and *Albizia* interspersed with grasses; open grassland, composed of *Hyparrhenia/Themeda/Imperata* fire climax grasses dotted with *Capparis* thickets containing Euphorbia trees; and swamp in the northern portion of the Park, dominated by Cyperus with Phoenix palms. The vegetation of the Park is naturally forested and the vegetation mosaic is a product of heavy grazing, browsing and frequent fires, which maintain the open grassland habitats. The present vegetation is dominated by *Acacia sieberiana, Acacia gerrardii* and *Acacia hockii* in a matrix of tall grasses of *Hyparrhenia filipendula*, *Themeda triandra*, *Imperata cylindrica* and *Cymbopogon afronardus*. Other common woody vegetation includes *Capparis tomentosa*, *Securinega virosa*, *Indigofera arrecta* and *Solanum incanum* (Lock, 1988).

3.2 Sampling methods

3.2.1 Determination of fire frequency in Queen Elizabeth National Park (QENP)

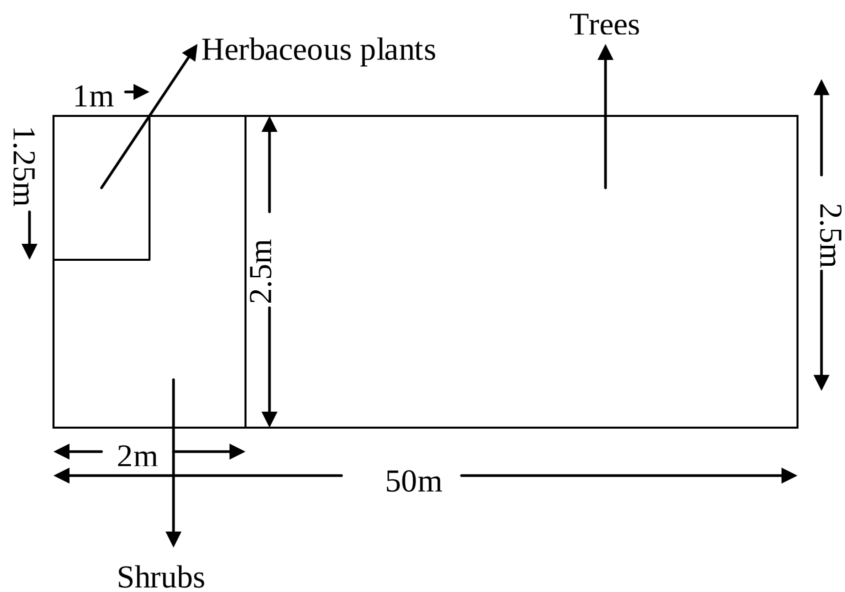
Fire frequency was determined from a map of burn images of the park produced by the Woods Hole Research Center (Appendix I) showing frequencies of burning over a period of 34 years (1973-2007). The map was carefully studied to divide the study area into strata according to burn frequencies. The frequencies of burning (number of times the areas were burnt) over the 34 year period (1973-2007) were; 1-2 times, 2-6 times, 6-10 times, and 10-14 times. The most frequently burned areas appeared red (darker) while the least frequently areas appeared yellow (lighter) on the map of burn images of the QENP.

After identifying the burn frequencies on the burn scar image of the park, a field visit was conducted to validate the vegetation types and locate the study sites based on fire frequencies and similarities in vegetation types.

3.2.2 Vegetation sampling

The modified-Whittaker nested vegetation sampling method was used to sample vegetation in the study area (Stohlgren *et al.,* 1995). Thirty big plots measuring 50m X 2.5m were laid at 100m intervals along line transects. Three transects were laid per study site and ten plots were laid per transect. The transects were 80m apart. In the big plot trees were sampled. Nested plots of 2m X 2.5m and 1m X 1.25m were then established within the big plot for sampling shrubs and herbaceous plants respectively. Hence, trees were sampled in thirty - 50m X 2.5m plots, shrubs were sampled in thirty - 2m X 2.5m plots and herbaceous plants were sampled in thirty - 1m X 1.25m plots.

Plot design



A control was not included in the study because; nearly all the areas with compararble vegetation in the park had been burnt according to the map of burn images of the park produced by the Woods Hole Research Center (Appendix I) and a field visit to the park. During the field visit and interviews with the Waden Research and Monitoring in QENP, it was observed that the permanent exclosure that had been set up by Friend in 1971 to study thicket productivity and later used by Edroma for grassland studies (Lock, 1988) had been broken and even burnt.

Most plant species were identified in the field using species identification guides by Sylvia *et al.* (2003), Lock (1977) and Lind & Tallantire (1971). Specimens of difficult plant species were collected, labeled, pressed, and dried for eventual identification through comparison with preserved specimens at the Makerere University Herbarium.

For herbaceous plants, the number of individuals of each species was counted and subsequently, the cover in the 1.25m X 1m subplot was estimated. For shrubs, only the cover of each species in the 2.5m X 2m subplot was estimated because most shrubs were in thickets and it was not possible to count their number of individuals.

For trees, the number of each species in the big (50m X 2.5m) plot was counted and then the (%) cover estimated.

All ground vegetation cover estimations were done according to the method described by Chapman (1976) where ground cover was determined by estimating the percentage area covered by vegetation in the plots.

3.3 Data analysis

Data was analyzed using Genstat 5 Release 3.22. Data sheets of the raw data were prepared in MS Excel based on the various measured vegetation parameters. These were then exported to Genstat 5 Release 3.22 using a statistical program called Start transfer after which analysis was done. Statistical tests such as regression analysis were run to determine the relationships between the measured parameters across the study sites.

Herbaceous plants were analyzed for species diversity, richness, and % cover, relative frequencies (RFs), relative covers (RCs), relative densities (RDs) and importance value indices (IVIs). Shrubs and trees were analyzed for species diversity, richness and % cover.

3.3.1 Determination of plant species diversity and richness

Plant species diversity was determined using the Shannon-Weiner diversity index according to the formula given below:

$$H' = -\sum p_i \,\text{In}\, p_i$$

Where pi is the relative cover of species i and H' is the Shannon-Weiner diversity index.

Shanon-Weiner diversity index was preferred because it is easier to compute and interprete. It can be computed from either relative cover or the number of individuals of a given species. In this study, it was computed from relative cover of a species.

Species richness (S) was the total number of plant species per plot and research site.

Variation in plant species richness and diversity between study sites was determined by using regression analysis.

3.3.2 Determination of vegetation cover

Vegetation cover was determined at three levels; for individual species (i.e. herbaceous plants, shrubs, and trees) and the combined cover of the three plant life forms (herbs, shrubs and trees) in a plot assuming uniform plot sizes. The average vegetation cover (herbs, shrubs, trees and all vegetation) in each of the study sites was further computed and the variations between them tested using regression analysis.

The following vegetation parameters for herbaceous plants were computed:

Relative species density (RD) which is the number of individuals of a given species (n_i) expressed as a proportion of the total number of individuals of all species ($\sum n$):

$RD_i = n_i/\sum n$

Relative frequency (Rf) was taken as the frequency of a given species (f_i) expressed as a proportion of the sum of frequencies of all species ($\sum f$): $Rf_i = f_i/\sum f$

Frequency of a given species i (fi) was the number of plots with species i compared with all the plots sampled.

Relative Coverage (RC_i) for species i was estimated as the coverage for that species (C_i) expressed as a proportion of the total coverage (TC) for all species:

$RC_i = C_i/TC = C_i/\sum C$, where $\sum C$ is the sum of the coverage's of all the species. From these relative measurements, importance value indices of the various species in the vegetation types were calculated and then compared (Importance Value Index (IVI) = $RD_i + Rf_i + RC_i$).

CHAPTER FOUR

RESULTS

4.1 Effect of fire frequency on species richness

4.1.1 Fire frequency vs. species richness of herbaceous plants

Table 1 shows variation in species richness by plot according to the different fire frequencies. Species richness of herbaceous plants varied greatly with fire frequency. The least frequently burnt area (1-2x) had the highest species richness (19.4) value compared to the other areas. The rest of the areas had comparable species richness values. The lowest mean species richness was 9.03 in the second most frequently burnt area.

Figure 1 shows that the variation in mean species richness of herbaceous plants with fire frequency followed a quadratic pattern with a maximum for 1-2x and a minimum for 6-10x ($y = 0.2372x^2 – 3.91x + 24.1$; $R^2= 0.9471$). If 10-14x was excluded, mean species richness of herbaceous plants decreased with fire frequency ($y = 0.0105x^2 – 0.217x + 4.3956$; $R^2= 0.3491$).

Table1: Effects of fire frequency (per 34 year period) on species richness of herbaceous plants

Species Richness (S)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	14	15	10	8
2	15	14	10	10
3	17	7	13	14
4	11	1	1	10
5	20	8	11	8
6	20	8	7	11
7	23	11	13	1
8	18	7	7	10
9	13	11	5	17
10	20	7	11	8
11	26	14	5	1
12	22	11	6	8
13	20	7	6	14
14	11	6	4	17
15	20	10	5	11
16	22	16	1	1
17	11	5	8	13
18	24	10	1	11
11	17	6	14	14
20	15	12	7	10
21	21	5	8	11
22	18	12	10	6
23	16	18	12	8
24	16	15	13	11
25	20	6	13	10
26	17	13	6	14
27	27	15	1	20
28	28	14	10	6
21	14	6	12	7
30	22	13	1	16
ΣX	582	327	271	330
$\bar{X}$	19.4	10.9	9.03	11.0
$S_{\bar{X}}$	3.79	4.34	2.83	3.45

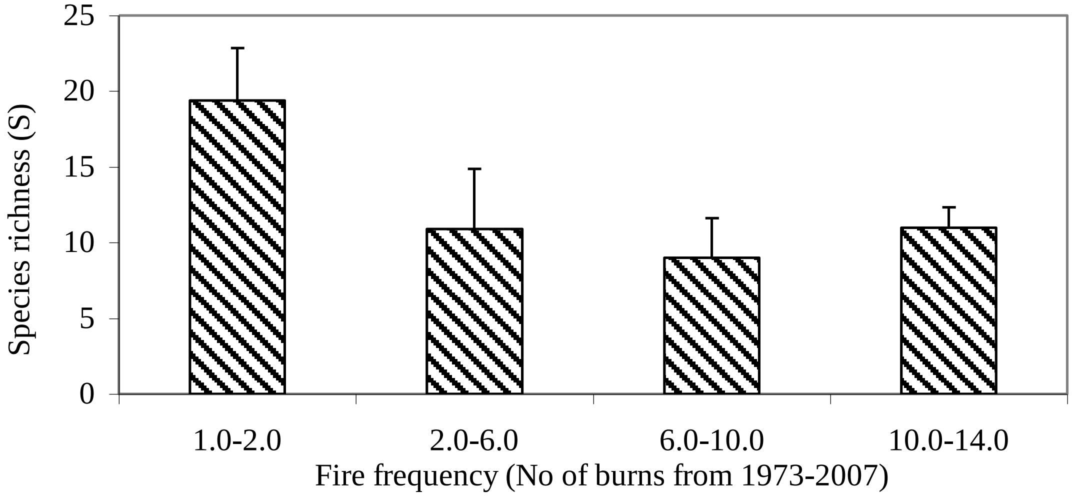


Fig 1. Distribution of mean species richness of herbaceous plants by fire frequency

$$\bar{x} = 0.237f^2 - 3.9f + 24.1 \qquad (R^2 = 0.95)$$

4.1.2 Fire frequency vs. species richness of shrubs

Table 2 shows variation in species richness for shrubs by fire frequency.

There was no much variation in species richness for shrubs by fire frequency although areas that were less frequently burnt had slightly higher values of species richness. Most plots (irrespective of fire frequency) had zero shrub species richness values. The highest value of mean species richness was 1.2 in the least frequently burnt area (1-2x). The most frequently burnt area had lower mean species richness (0.27) compared to the other areas. Figure 2 shows that the mean species richness of shrubs varied exponentially with fire frequency ($y = 1.5077e^{-0.1437x}$). If 10-14x was excluded, species richness decreased linearly with fire frequency ($y = -0.0885x + 1.2664$; $R^2 = 0.9645$).

Table 2: Effects of fire frequency (per 34 year period) on species richness of shrubs

Species Richness (S)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	0	2	0	0
2	3	1	0	0
3	0	0	0	0
4	0	0	0	0
5	0	0	4	0
6	2	0	0	0
7	2	2	2	0
8	4	0	0	0
1	0	2	0	1
10	3	0	1	0
11	1	0	1	0
12	0	0	0	0
13	0	0	0	0
14	0	0	0	1
15	5	1	0	0
16	2	1	0	0
17	2	0	0	0
18	1	2	0	0
11	0	2	1	1
20	0	0	0	0
21	2	0	0	0
22	0	2	2	0
23	0	1	0	0
24	0	2	2	0
25	0	0	1	0
26	0	2	0	2
27	1	3	0	1
28	4	3	0	0
21	0	0	0	0
30	3	0	0	2
ΣX	35	26	14	8
$\bar{X}$	1.20	0.87	0.47	0.27
$S_{\bar{X}}$	1.50	1.04	0.14	0.58

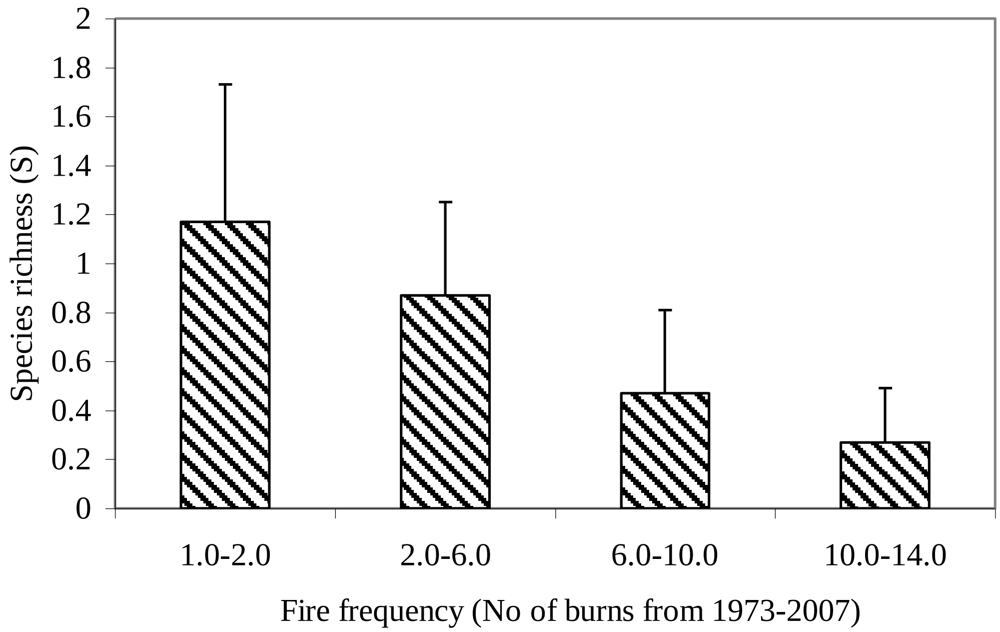


Fig 2.Distribution of mean species richness of shrubs by fire frequency

$$\overline{x} = -0.08f + 1.27 \quad (R^2 = 0.96)$$

$$\overline{x} = 1.507e^{-0.144f} \quad (R^2 = 0.9992)$$

4.1.3 Fire frequency vs. species richness of trees

Table 3 shows variations in species richness of trees by fire frequency.

Species richness for trees was generally low throughout the study sites. Most plots had zero (0) tree species richness values irrespective of fire frequency. However, species richness was slightly higher in areas with lower fire frequencies (1-2x and 2-6x) compared to those in more frequently burnt areas (6-10x and 10-14x). The highest value of species richness of trees in plots was 2 in the second least frequently burnt area (2-6x) and the least frequently burnt area (10-14x). The highest value of mean species richness of trees was 0.6 in the least frequently burnt area and the lowest value was 0.07 in the least frequently burnt area.

Figure 3 shows that the mean species richness of trees decreased exponentially with fire frequency ($y = 0.8764e^{-0.2027x}$; $R^2 = 0.9864$).

Table 3: Effect of fire frequency (per 34 year period) on species richness of trees

Species Richness (S)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	2	1	0	0
2	1	1	0	0
3	0	0	0	0
4	0	0	0	0
5	0	0	1	0
6	2	0	0	0
7	2	2	1	1
8	0	0	0	0
1	1	0	0	0
10	1	0	0	0
11	0	0	0	0
12	0	0	0	0
13	1	0	1	0
14	0	0	0	0
15	1	0	0	0
16	1	0	0	0
17	2	0	0	0
18	0	1	0	0
11	1	1	0	1
20	0	0	0	0
21	0	0	0	0
22	0	1	0	0
23	0	2	0	0
24	0	1	1	0
25	0	0	0	0
26	1	0	0	0
27	0	1	0	0
28	0	2	0	0
21	0	0	0	0
30	2	0	0	0
ΣX	18	12	6	2
$\bar{X}$	0.60	0.40	0.20	0.07
$S_{\bar{X}}$	0.77	0.67	0.48	0.25

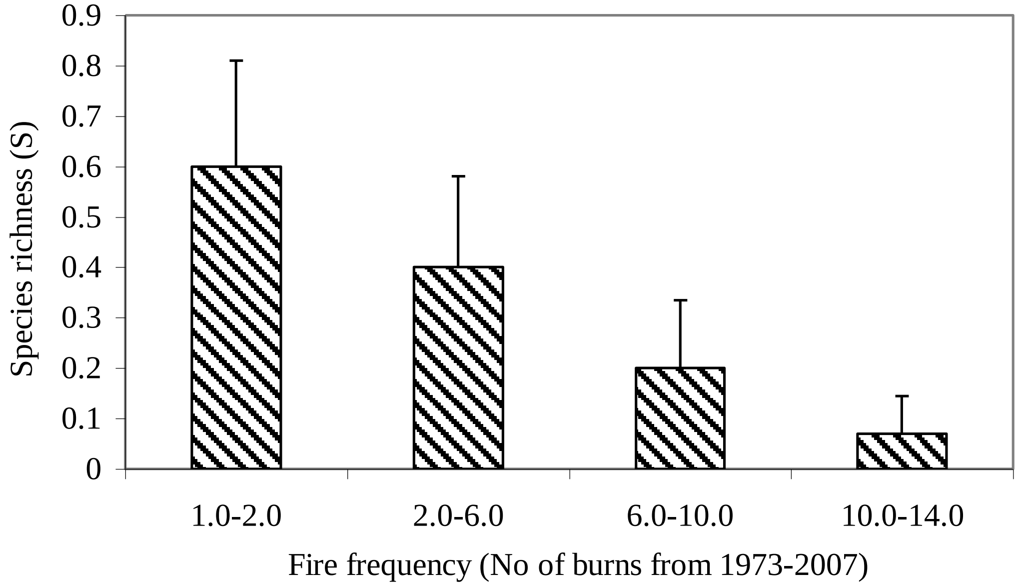


Fig 3.Distribution of mean species richness of trees by fire frequency

$$\bar{x} = 0.876e^{-0.2027f} \qquad (R^2 = 0.9864)$$

4.1.4 Fire frequency vs. species richness of all vegetation

Species richness of all vegetation varied considerably by fire frequency within and between the study sites (Table 4). The least frequently burnt area had higher values of species richness in all the plots compared to the other areas. This was followed by the second least frequently burnt area. Generally, areas with low fire frequency had higher species richness of all vegetation. The highest species richness was 32 in the plot of the least frequently burnt area and the lowest was 4 in the plot of the second most frequently burnt area. The highest value of mean species was 21.17 in the least frequently burnt area and the lowest was 9.37 in the second most frequently burnt area.

The variation in mean species richness of all vegetation with fire frequency (Figure 4) followed a quadratic pattern with a maximum for 1-2x and a minimum for 6-10x ($y = 0.254x^2 – 4.2843x + 26.456$; $R^2= 0.9666$).

Table 4: Effects of fire frequency (per 34 year period) on species richness of all vegetation

Species Richness (S)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	16	18	10	8
2	11	15	10	10
3	17	7	13	14
4	11	1	1	10
5	20	8	16	8
6	24	8	7	11
7	27	23	16	10
8	22	7	7	10
1	14	13	5	18
10	24	7	12	8
11	27	14	6	1
12	22	11	6	8
13	21	7	7	14
14	11	6	4	18
15	26	11	5	11
16	25	17	1	1
17	23	5	8	13
18	25	13	1	11
11	18	1	15	16
20	15	12	7	10
21	23	5	8	11
22	18	15	12	6
23	16	21	12	8
24	16	18	16	11
25	20	6	14	10
26	18	15	6	16
27	28	11	1	21
28	32	11	10	6
21	14	6	12	7
30	27	13	1	18
ΣX	635	365	281	340
$\bar{X}$	21.17	12.17	9.37	11.33
$S_{\bar{X}}$	4.65	5.37	3.41	3.10

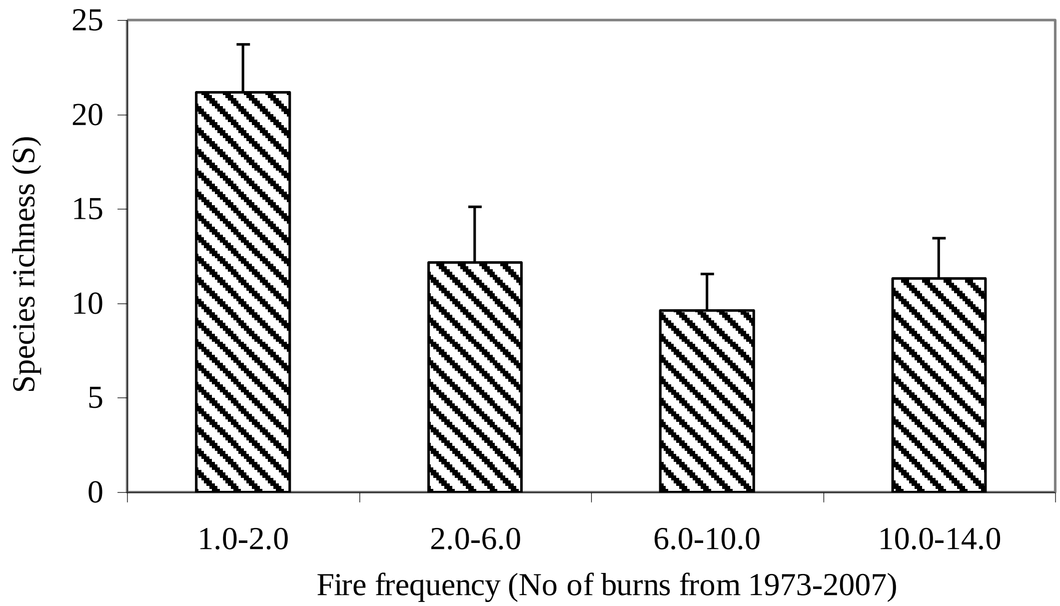


Fig 4. Distribution of mean species richness of all vegetation by fire frequency

$$\overline{x} = 0.25f^2 - 4.28f + 26.46 \quad (R^2 = 0.966)$$

4.2 Effects of fire frequency on plant species diversity

4.2.1 Fire frequency vs. species diversity of herbaceous plants

Table 5 shows variations in species diversity by plots according to fire frequency. Species diversity varied considerably by plot throughout the study sites. The least frequently burnt area showed the highest mean diversity values with the highest being 7.89. This was followed by the most frequently burnt area (6.04) while the second most frequently burnt area and the second least frequently burnt area had lower mean diversity values of 5.10 and 4.27 respectively. Figure 5 shows the patten of variation in mean species diversity of herbaceous plants with fire frequency. Species diversity for herbaceous plants followed a quadratic pattern ($y = 0.0923x^2 - 1.3539x + 9.2326$; $R^2= 0.7296$) with a maximum for 1-2x and a minimum for 2-6x.

Table 5: Effects of fire frequency (per 34 year period) on species diversity of herbaceous plants

Species Diversity (*H'*)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	7.0	6.1	5.7	4.9
2	5.2	4.0	8.4	6.4
3	7.9	4.6	7.2	6.0
4	8.3	4.6	4.0	6.6
5	7.3	5.1	4.9	4.5
6	8.1	6.2	4.5	7.9
7	6.9	4.8	5.0	5.6
8	6.2	5.8	4.7	7.3
9	8.1	4.1	4.5	6.4
10	7.8	3.8	3.9	5.0
11	9.4	4.5	4.9	6.1
12	9.5	5.3	5.2	5.8
13	10.9	6.6	2.2	3.4
14	9.8	4.8	3.8	5.4
15	7.9	6.1	4.9	4.7
16	6.3	3.5	3.7	5.6
17	6.2	1.4	3.9	7.0
18	7.6	1.9	5.7	6.9
19	7.0	2.6	5.2	5.0
20	8.6	4.4	6.0	7.0
21	7.3	2.8	6.5	7.4
22	9.2	3.2	3.8	6.2
23	9.2	4.8	7.1	6.7
24	8.2	3.0	3.7	5.5
25	8.7	4.7	4.7	5.2
26	8.8	4.3	3.5	4.4
27	7.7	5.5	5.7	6.6
28	7.3	2.2	7.6	5.9
29	7.9	1.8	5.7	8.2
30	6.4	5.5	6.5	7.7
ΣX	236.7	128.0	153.1	181.3
$\bar{X}$	7.89	4.27	5.10	6.04
$S_{\bar{X}}$	1.24	1.40	1.38	1.14

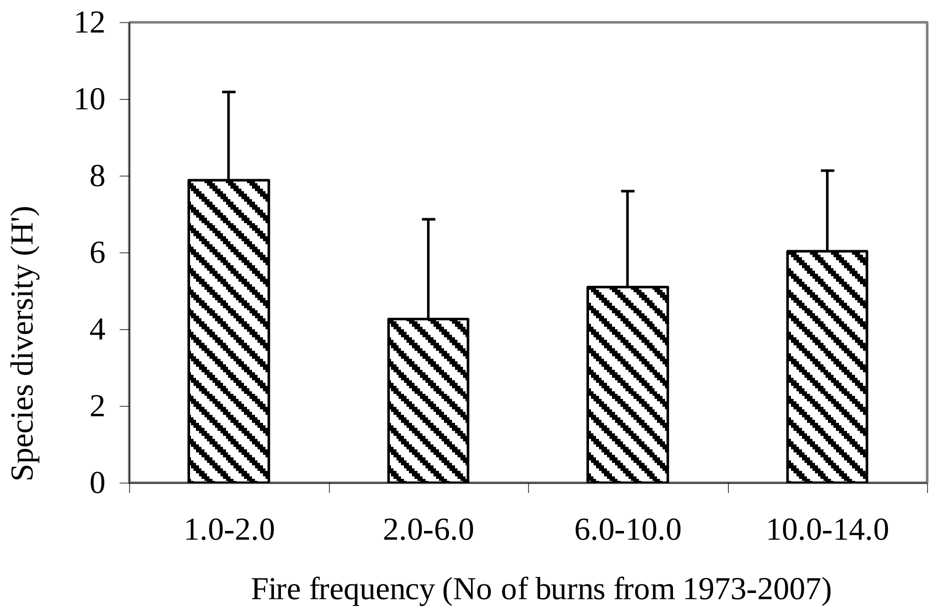


Fig 5. Distribution of mean species diversity of herbaceous plants by fire frequency

$$\overline{x} = 0.0923f^2 - 1.354f + 9.23 \qquad (R^2 = 0.7296)$$

4.2.2 Fire frequency vs. species diversity of shrubs

Table 6 shows variation in species diversity according to fire frequency for shrubs. Variation in species diversity for shrubs was not as much as that of herbaceous plants although the least frequently burnt and the second less frequently burnt areas showed fairly higher diversity values. Most plots in all the burn frequencies had zero (0) shrub diversity values. The highest mean value of species diversity for shrubs was 0.32 in the least frequently burnt area and the second highest value was 0.19 in the second least frequently burnt areas. The lowest mean value of shrub diversity was 0.02 in the most frequently burnt area. Figure 6 shows the pattern of variation of mean species diversity of shrubs with fire frequency. The mean species diversity of shrubs decreased with fire frequency following an exponential pattern ($y = 0.5107e^{-0.2682x}$; $R^2 = 0.9974$).

Table 6: Effects of fire frequency (per 34 year period) on species diversity of shrubs

Species Diversity (H')				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	0	0.7	0	0
2	1.1	0	0	0
3	0	0	0	0
4	0	0	0	0
5	0	0	1.0	0
6	0.5	0	0	0
7	0.7	0.2	0.5	0
8	0.8	0	0	0
9	0	0.2	0	0
10	0.9	0	0	0
11	0	0	0	0
12	0	0	0	0
13	0	0	0	0
14	0	0	0	0.1
15	1.3	0	0	0
16	0.6	0	0	0
17	1.3	0	0	0
18	0	0.4	0	0
19	0	0.7	0	0
20	0	0	0	0
21	0.6	0	0	0
22	0	0.5	0.1	0
23	0	0	0	0
24	0	0.7	0.3	0
25	0	0	0	0
26	0	0.3	0	0.5
27	0	0.7	0	0
28	1.1	1.3	0	0
29	0	0	0	0
30	0.8	0	0	0
ΣX	9.7	5.7	1.9	0.6
$\bar{X}$	0.32	0.19	0.06	0.02
$S_{\bar{X}}$	0.46	0.33	0.21	0.09

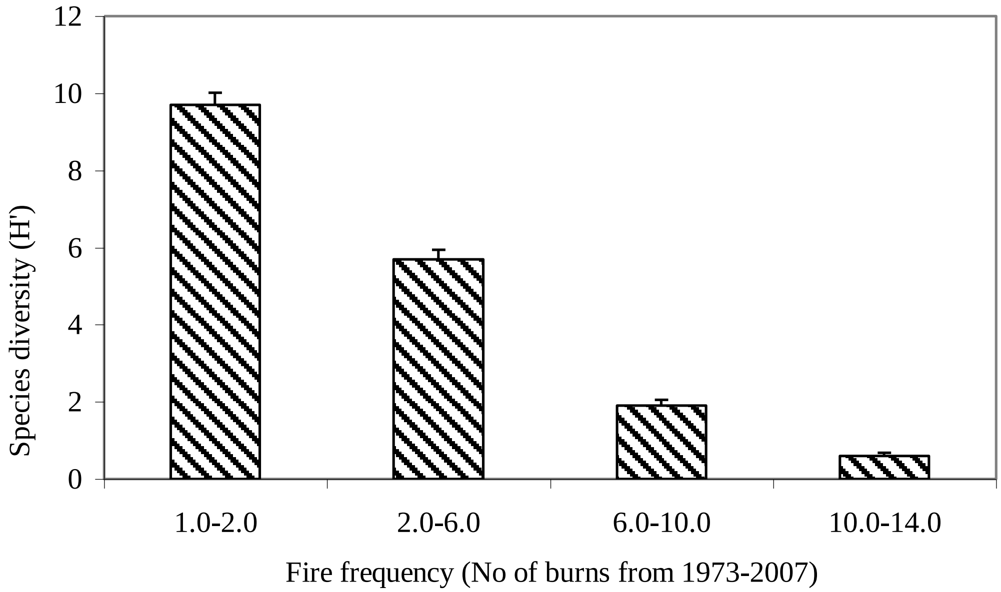


Fig 6. Distribution of mean species diversity of shrubs by fire frequency

$$\overline{x} = 0.5107e^{-0.2682f} \quad (R^2 = 0.997)$$

4.2.3 Fire frequency vs. species diversity of trees

Table 7 shows variations in species diversity by plots according fire frequency for trees. Just like in the case of shrubs, most plots (irrespective of fire frequency) had zero diversity values for trees. Areas that were less frequently burnt had slightly higher diversity values compared to areas that were more frequently burnt. The highest and second highest values of mean diversity for trees were 0.16 and 0.13 in the least frequently burnt area (1-2x) and the second least frequently burnt area (2-6x) respectively. The lowest value of mean species diversity of trees was 0.05 in the second most frequently burnt area. Figure 7 shows the pattern of variation of mean species diversity of trees with fire frequency. The mean species diversity of trees decreased with fire frequency following a quadratic pattern with a maximum for 1-2x and a minimum for 6-10x ($y = 0.0016x^2 – 0.0311x + 0.2115$; R^2= 0.9251).

Table 7: Effects of fire frequency (per 34 year period) on species diversity of trees

	Species Diversity (*H'*)			
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	0	0	0	0
2	1.1	0	0	0
3	0	0	0	0
4	0	0	0	0
5	0	0	0.7	0
6	0.7	0	0	0
7	1.1	1.1	0.7	1.0
8	0	0	0	0
9	0	0	0	0
10	0	0	0	0
11	0	0	0	0
12	0	0	0	0
13	0	0	0	0
14	0	0	0	0
15	0	0	0	0
16	0.9	0	0	0
17	0.4	0	0	0
18	0	0	0	0
19	0	1.1	0	1.0
20	0	0	0	0
21	0	0	0	0
22	0	0	0	0
23	0	0.6	0	0
24	0	0	0	0
25	0	0	0	0
26	0	0	0	0
27	0	0	0	0
28	0	1.0	0	0
29	0	0	0	0
30	0.7	0	0	0
ΣX	4.87	3.80	1.4	2.0
$\bar{X}$	0.16	0.13	0.05	0.07
$S_{\bar{X}}$	0.35	0.34	0.18	0.25

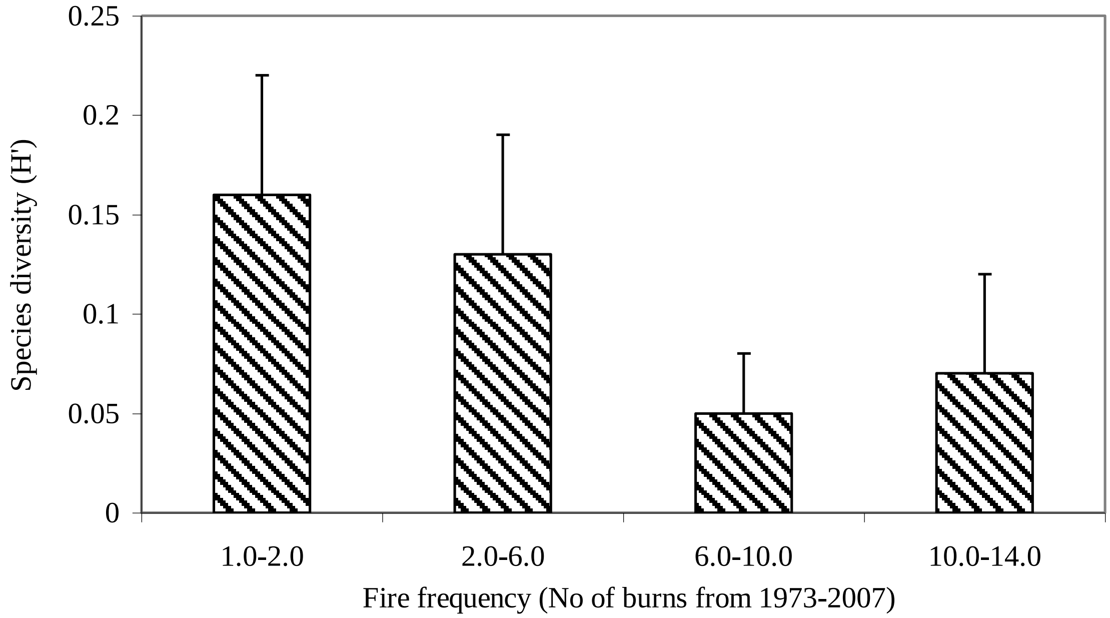


Fig 7. Distribution of mean species diversity of trees by fire frequency

$$\overline{x} = 0.0016f^2 - 0.0311f + 0.2115 \qquad (R^2 = 0.9251)$$

4.2.4 Fire frequency vs. species diversity of all vegetation

Table 8 shows fire frequency and species diversity for all vegetation types.

Fire frequency and species diversity for all vegetation (overall species diversity) varied considerably across the burn frequencies. The least frequently burnt area had a higher mean diversity value of 8.21; this was followed by the most frequently burnt area with a mean diversity value of 6.15. The lowest value of mean diversity of all vegetation was 4.58 in the second least frequently burnt area. Figure 8 shows a clear trend of mean species diversity of all vegetation types combined. It shows that mean species diversity of all vegetation changed quadratically with fire frequency ($y = 0.0917x^2 - 1.3671x + 9.5838$; $R^2 = 0.7464$).

Table 8: Effects of fire frequency (per 34 year period) on species diversity of all vegetation

Species Diversity (*H'*)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	7.5	6.6	5.7	4.9
2	6.2	4.3	8.4	6.4
3	7.9	4.6	7.2	6.0
4	8.3	4.6	4.0	6.6
5	7.3	5.1	6.1	4.5
6	8.3	6.2	4.5	7.9
7	8.2	5.2	5.8	6.5
8	6.8	5.8	4.7	7.3
9	8.1	4.7	4.5	6.6
10	8.2	3.8	4.2	5.0
11	9.7	4.5	5.2	6.1
12	9.5	5.3	5.2	5.8
13	10.9	6.6	2.5	3.4
14	9.8	4.8	3.9	5.8
15	9.0	6.3	4.9	4.7
16	7.3	3.9	3.7	5.6
17	7.7	1.4	3.9	7.0
18	7.6	2.8	5.7	6.9
19	7.1	3.7	5.5	5.7
20	8.6	4.4	6.0	7.0
21	7.8	2.8	6.5	7.4
22	9.2	4.0	4.5	6.2
23	9.2	5.1	7.1	6.7
24	8.2	3.6	4.8	5.5
25	8.7	4.7	5.3	5.2
26	8.8	4.7	3.5	5.0
27	7.9	6.5	5.7	6.8
28	7.7	4.0	7.6	5.9
29	7.9	1.8	5.7	8.2
30	7.0	5.5	6.5	7.8
ΣX	246.4	137.3	158.8	184.4
$\bar{X}$	8.21	4.58	5.29	6.15
$S_{\bar{X}}$	1.00	1.30	1.31	1.10

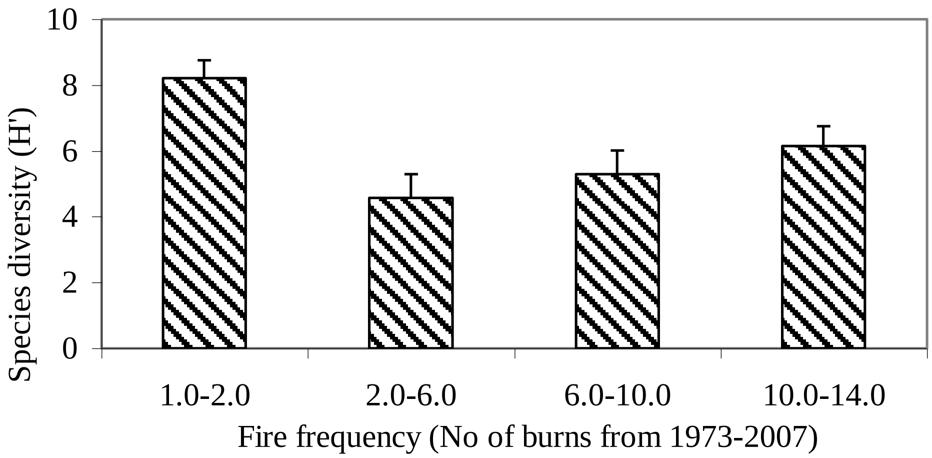


Fig 8. Distribution of species diversity of all vegetation by fire frequency

$$\bar{x} = 0.0917f^2 - 1.367f + 9.5838 \qquad (R^2 = 0.75)$$

4.3 Effect of fire frequency on vegetation cover

4.3.1 Fire frequency vs. cover of herbaceous plants

Table 9 shows variation in vegetation cover (%) for herbaceous plants by fire frequency. The highest mean cover (%) of herbaceous plants was 86.03 in the second most frequently burnt area (6-10x); this was follwed by the second least frequently burnt area with a mean cover value of 73.74. The least frequently burnt area had the lowest mean cover value of 50.07. Figure 9 shows the general trend of the mean cover (%) of herbaceous plants with fire frequency which followed a quadratic pattern with a peak at 6-10x ($y = -0.9945x^2$ 14.977x +29.811; R^2= 1).

Table 9: Effects of fire frequency (per 34 year period) on percentage cover of herbaceous plants

% Cover (herbaceous)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	49.5	84.2	83.0	61.7
2	47.8	65.7	89.2	74.8
3	68.0	94.0	81.3	86.0
4	62.8	94.0	97.2	84.5
5	67.8	88.5	60.3	84.5
6	53.8	81.7	96.5	77.0
7	41.8	40.3	63.7	79.5
8	41.5	79.5	91.8	57.0
9	51.7	45.3	82.5	41.0
10	31.7	96.7	76.7	76.5
11	36.2	95.7	85.0	72.8
12	40.2	90.5	91.3	75.2
13	55.0	91.8	86.2	56.8
14	61.7	80.5	98.7	46.7
15	29.3	80.2	98.5	85.2
16	36.5	65.8	94.3	82.7
17	44.7	83.5	98.2	65.7
18	68.5	62.7	94.0	84.3
19	60.3	63.2	62.2	32.0
20	62.3	78.7	96.7	55.7
21	52.0	60.7	98.2	59.3
22	45.3	40.3	73.3	63.2
23	50.3	57.2	97.2	65.0
24	58.5	74.7	85.0	51.0
25	53.0	95.0	82.3	53.5
26	47.2	65.5	100.0	53.8
27	43.8	65.0	66.5	69.2
28	37.0	8.3	84.5	81.5
29	60.7	90.5	71.0	66.3
30	43.2	92.5	95.5	47.0
ΣX	1502.1	2212.2	2580.8	1989.4
$\bar{X}$	50.07	73.74	86.03	66.31
$S_{\bar{X}}$	10.94	20.70	12.06	14.77

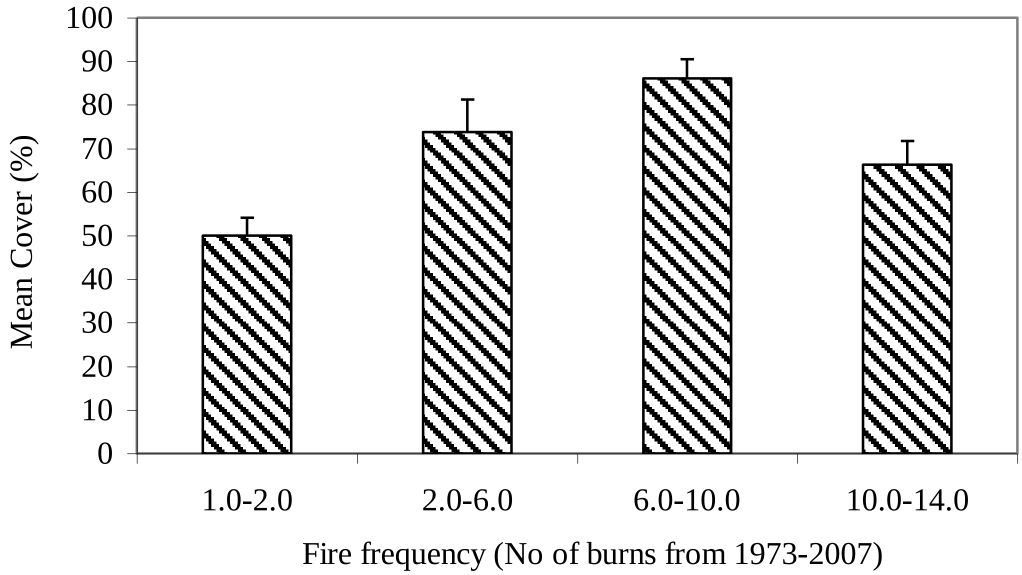


Fig 9. Fire frequency vs. mean cover (%) of herbaceous plants

$$\bar{x} = -0.995f^2 + 14.977f + 29.11 \qquad (R^2 = 0.99)$$

4.3.2 Fire frequency vs. cover of shrubs

Table 10 shows variations in % cover of shrubs with fire frequency. Percentage cover of shrubs was higher in the less frequently burnt areas (1-2x and 2-6x) with the second least frequently burnt area having higher % shrub cover values. On the other hand, areas more frequently burnt (6-10x and 10-14x) had lower shrub cover values. Most plots had zero (0) shrub cover. The highest mean shrub cover was 6.47 in the second least frequently burnt area (2-6x). The lowest mean cover of shrubs was 1.89 in the most frequently burntarea. Figure 10 shows the general trend of percentage mean cover of shrubs with fire frequency. The variation in % mean cover of shrubs with fire frequency followed a quadratic pattern with a peak at 2-6x ($y = -0.0486x^2 + 0.3211$; $R^2 = 0.7574$).

Table 10: Effects of fire frequency (per 34 year period) on % cover of shrubs

% Cover (shrubs)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	1.7	1.7	0	0
2	5.7	11.7	0	0
3	0	0	0	0
4	0	0	0	0
5	0	0	15	0
6	20	0	0	0
7	4.3	27.7	12	0
8	25.7	0	0	0
9	0	17.7	0	16.7
10	28.3	0	15	0
11	1.7	0	10	0
12	0	0	0	0
13	0	0	0	0
14	0	0	0	13.7
15	31	5	0	0
16	4	16.7	0	0
17	2.7	0	0	0
18	0	23.3	0	0
19	0	6.7	5	0
20	0	0	0	0
21	1	0	0	0
22	0	8.3	22	0
23	0	8.3	0	0
24	0	1.3	9	0
25	0	0	10	0
26	0	14.7	0	21.7
27	1.7	10.7	0	3.3
28	5.3	40.3	0	0
29	0	0	0	0
30	5	0	0	1.3
ΣX	136.4	194.1	98.0	56.7
$\bar{X}$	4.55	6.47	3.27	1.89
$S_{\bar{X}}$	8.98	10.09	6.06	5.39

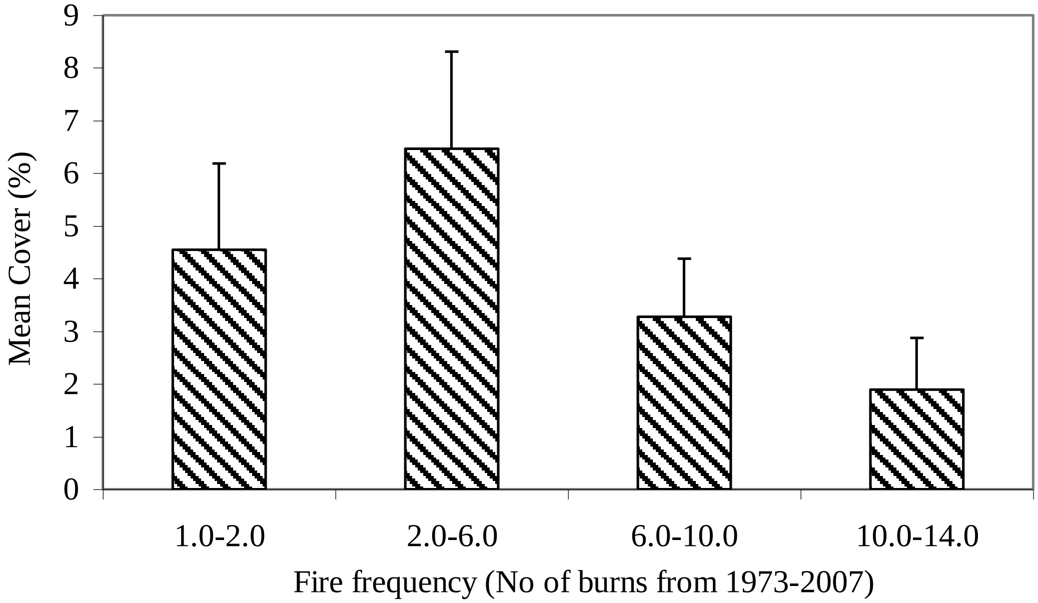


Fig 10. Fire frequency vs. mean cover (%) of shrubs

$$\overline{x} = -0.0486f^2 + 0.321f + 4.748 \qquad (R^2 = 0.76)$$

4.3.3 Fire frequency vs. cover of trees

Table 11 shows variations in % cover of trees with fire frequency. Percentage cover of trees did not vary significantly with fire frequency though the less frequently burnt areas had relatively higher % cover values for trees compared to the more frequently burnt ones. The highest mean tree cover (5.39) was observed in the least frequently burnt area (1-2x) and most plots irrespective of fire frequency had zero % tree cover. The lowest mean tree cover was 1.33 observed in the second most frequently burnt area (6-10x).
Figure 11 shows that the % mean cover of trees varied quadratically with fire frequency ($y = 0.0647x^2 – 1.196x + 7.479$; $R^2 = 0.8404$). The minimum mean % cover was at around 6-10x.

Table 11: Effects of fire frequency (per 34 year period) on % cover of trees

% Cover (trees)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	4	10	0	0
2	18	0	0	0
3	0	0	0	0
4	0	0	0	0
5	0	0	10	0
6	9	0	0	0
7	21	28	20	20
8	0	0	0	0
9	25	0	0	0
10	1	0	0	0
11	0	0	0	0
12	0	0	0	0
13	4	0	5	0
14	0	0	0	0
15	10	0	0	0
16	23	0	0	0
17	29.7	0	0	0
18	0	15	0	0
19	4	18	0	60
20	0	0	0	0
21	5	0	0	0
22	0	15	0	0
23	0	12	0	0
24	0	15	5	0
25	0	0	0	0
26	0	0	0	0
27	0	15	0	0
28	0	22	0	0
29	0	0	0	0
30	8	0	0	0
ΣX	161.7	140	40	80
$\bar{X}$	5.39	4.67	1.33	2.67
$S_{\bar{X}}$	8.80	8.27	4.14	11.43

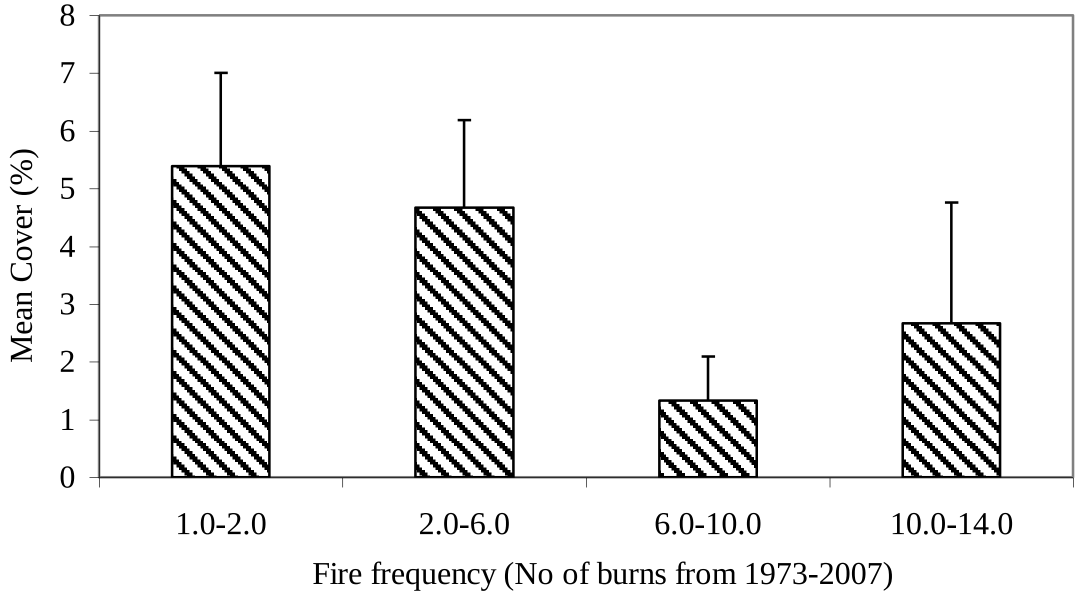


Fig 11. Fire frequency vs. mean cover (%) of trees

$$\overline{x} = 0.0647f^2 - 1.196f + 7.479 \quad (R^2 = 0.84)$$

4.3.4 Fire frequency vs. cover of all vegetation

Table 12 shows variation in vegetation cover (%) for all vegetation types with fire frequency. Vegetation cover (%) for all vegetation varied considerably with fire frequency. The second most frequently burnt area (6-10x) had the highest % mean cover (90.63) of allvegetation types followed by 84.88 in the second least frequently burnt area (2-6x). The lowest % mean cover of all vegetation types was 60.01 in the least frequently burnt area (1-2x). Relatively lower cover values for all vegetation were observed in the least frequently burnt area (1-2x) and the most frequently burnt area (10-14x) with the former having a number of plots that had cover values below 50%. The highest cover value for was 100% in the plots in the second least frequently burnt and the second least frequently burnt areas while the lowest was 37.9% in the plot in the least frequently burnt area (1-2x).

Figure 12 shows the trend of percentage mean cover of all vegetation with fire frequency. The variation in percentage mean cover of all vegetation with fire frequency followed a quadratic pattern with a peak at the 6-10x ($y = -0.9784x^2 + 14.102x + 42.037$; $R^2 = 0.9858$).

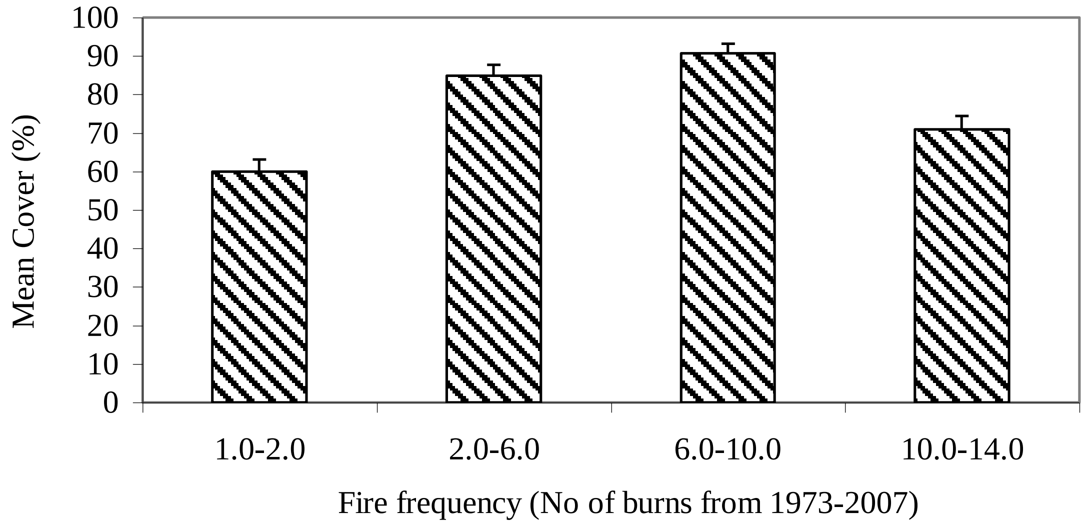


Fig 12. Fire frequency vs. mean cover (%) of all vegetation

$\overline{x} = -0.978f^2 + 14.102f + 42.037 \qquad (R^2 = 0.9858)$

Table 12: Effects of fire frequency (per 34 year period) on % cover of all vegetation

% Cover (trees)				
Plots	Burnt 1-2x	2-6x	6-10x	10-14x
1	55.2	85.9	83	61.7
2	71.5	77.4	89.2	74
3	68	94	81.3	86
4	62.8	94	97.2	84.5
5	67.8	88.5	85.3	84.5
6	82.8	81.7	96.5	77
7	67.1	96	95.7	99.5
8	67.2	79.5	91.8	57.0
9	76.7	63	82.5	57.7
10	61	96.7	91.7	76.5
11	37.9	95.7	95.0	72.8
12	40.2	90.5	91.3	75.2
13	59	91.8	91.2	56.8
14	61.7	80.5	98.7	60.4
15	70.3	85.2	98.5	85.2
16	63.5	82.5	94.3	82.7
17	77.1	83.5	98.2	65.7
18	68.5	100	94.0	84.3
19	64.3	87.9	67.2	92
20	62.3	78.7	96.7	55.7
21	58	60.7	98.2	59.3
22	45.3	63.6	95.3	63.2
23	50.3	77.5	97.2	65
24	58.5	91	99.0	51
25	53	95	92.3	53.5
26	47.2	80.2	100	75.5
27	43.8	90.7	66.5	72.5
28	42.3	70.6	84.5	81.5
29	60.7	90.5	71.0	66.3
30	56.2	92.5	95.5	48.3
ΣX	1800.2	2546.3	2718.8	2126.1
$\bar{X}$	60.01	84.88	90.63	70.87
$S_{\bar{X}}$	11.30	10.39	9.24	13.27

Effect of fire frequency on mean % cover of some species of herbaceous plants

Table 13 shows the effect of fire frequency on the mean cover (%) of some species of herbaceous plants.

Grasses generally had higher cover among the herbaceous plants that were encountered irrespective of fire frequency not withstanding the variations in % cover of herbs with fire frequency. For example, *Heteropogon contortus* (L.) Roem. & Schult. showed higher cover in almost all the study sites except in the least frequently burnt area. Its highest cover was in the second least frequently burnt area (91.3%) followed by the second most frequently burnt area (75.9%). *Sporobolus pyramidalis* P. Beauv. had considerably higher cover in the second most frequently burnt area (74.4%). Its cover in the other study sites was relatively low and comparable. *Hyparrhenia rufa* (Nees) Stapf had the highest cover in the second least frequently burnt area (53.7%) followed by the second most frequently burnt and the most frequently burnt areas with 34.8% each. *Themeda triandra* Forssk. was predominant in the second most frequently burnt area (57.8%) and had disproportionately lower cover in the other areas. *Bothriochloa insculpta* had the highest cover in the second least frequently burnt area (37.3%) followed by the most frequently burnt area and closely by the second most frequently burnt area with 28.7% and 27.6% respectively. *Cynodon dactylon (L.)* Pers. was only recorded in the second least burned area with 37.6% cover. *Microchloa kunthii* Desv. had high % cover in the most frequently burnt area (46.1) and much lower cover in the other areas. *Setaria homonyma* (Steud.) Chiov was only encountered in the area least frequently burnt (44.8%). The rest of the herbaceous plants especially non-grasses occupied much lower cover. The number of species (S) recorded in each fire frequency ranged from 61 (highest) in the least frequently burnt area to 35 (lowest) in the second most frequently burnt area. The number of species of herbaceous plants recorded tended to decrease with fire frequency.

Table 13. Effects of fire frequency (per 34 year period) on the mean cover (%) of some species of herbaceous plants

	Mean cover (% Cover)				
Species	Burnt	1-2	2-6	6-10	10-14
Aristida adscensionsis L		5.9	-	-	1.0
Bothriochloa insculpta		10.3	37.3	27.6	28.7
Chloris gayana Kunth		8.2	-	-	-
Chloris pycnothrix Trin.		25.6	-	-	1.2
Chloris roxburghiana Schult		-	0.1	-	1.9
Cynodon dactylon (L.) Pers.		-	37.6	-	-
Dactyloctenium aegyptium (L.) Willd.		0.1	-	-	-
Digitaria abyssinica (A. Rich.) Stapf		2.1	-	-	-
Dinebra retroflexa (Vahl) Panzer		3.3	0.3	0.1	-
Enteropogon macrostachys (A Rich) Benth		-	0.2	-	-
Eragrostis exasperata Peter		0.5	0.1	-	0.2
Heteropogon contortus (L.) Roem. & Schult.		4.1	91.3	75.9	62.8
Hyparrhenia rufa (Nees) Stapf		23.4	53.7	34.8	34.8
Hyperthelia dissoluta (Steud.) W.D. Clayton		-	-	0.7	-
Microchloa kunthii Desv.		16.1	12.6	8.9	46.1
Panicum infestum Peters		-	1.0	0.4	0.1
Panicum maximum Jacq.		0.1	4.9	0.1	-
Panicum repens		-	11.5	-	-
Panicum subalbidum Kunth		-	1.2	-	-
Setaria homonyma (Steud.) Chiov		44.8	-	-	-
Sporobolus festivus A. Rich		0.1	-	-	-
Sporobolus pyramidalis P. Beauv.		40.7	33.7	74.4	30.0
Themeda triandra Forssk.		-	0.1	57.8	0.2
Tragus berteronianus Schult		0.1	-	-	1.0
Urochloa panicoides Beav		-	-	-	0.1
Cyperus dubius Rottb. Subsp. Dubius		0.5	-	-	-
Cyperus teneriffae Poir		1.9	0.6	-	0.1
Cyperus articulatus L		-	2.7	-	-
Alternanthera sp.		0.1	-	-	-
Abutilon mauritianum (Jacq.) Medic.		0.3	4.2	2.9	-
Achyranthes aspera L.		0.1	1.2	5.2	4.5
Alysicarpus rugosus (Willd) DC. Subsp. Rugosus		0.1	0.8	0.3	0.6
Alysicarpus vaginalis (L.) DC		-	-	0.5	-
Anthericum subpetiolatum (Bak.) Kativu		-	-	-	0.1
Asparagus africanus Lam.		0.1	0.5	-	0.1
Asystasia gangetica (L.) T. Anderson		2.3	1.3	0.4	0.1
Asystasia mysorensis (Roth) T. Anderson		-	-	0.2	0.1
Bidens pilosa L		-	0.3	-	0.2

Blepharis maderaspatensis (L.) Roth	-	0.3	0.1	-
Boerhavia diffusa L	0.1	0.1	0.1	2.0
Caralluma dummeri N.E. Br	0.1	-	-	-
Cassia absus L	-	0.2	-	-
Cassia mimosoides L.	-	0.1	-	0.1
Cissus quadrangularis L.	4.0	12.9	4.3	7.8
Cissus rotundifolia (Forsk.) Vahl.	3.4	0.5	-	-
Commelina benghalensis L	0.9	2.1	0.3	0.1
Corbichonia decumbens (Forsk.) Exell	-	0.3	-	0.4
Corchoris trilocularis L	-	0.1	-	-
Craterostigma plantangineum Hochst	3.1	-	-	-
Crotalaria spinosa Benth.	0.3	0.2	-	1.3
Cucumis figarei Naud	-	-	-	0.1
Cyphostemma adenocayle (A. Rich.) Wild. & Drummond	0.5	1.3	1.2	0.9
Dyschoriste nagchana (Nees) Bennet	0.4	1.9	-	2.4
Euphorbia glomerifera (mill sp.) Wheeler	-	0.4	-	-
Evolvolus alsinoides (L.) L.	0.6	-	0.2	0.2
Evolvolus nummularius (L.) L.	9.2	0.8		
Fuerstia africana Th. C.E. Fries	0.2	0.1	0.1	0.9
Gomphrena celosioides Mart	0.1	-	-	-
Hibiscus aponeurus Sprague & Hutch.	0.6	0.2	-	-
Hoslundia opposita Vahl.	11.6	4.5	9.3	5.7
Hygrophila auriculata (Schumach.) Heine	-	0.5	-	-
Indigofera circinella Bak. f.	1.2	-	-	-
Indigofera vohemarensis Baill.	10.6	0.4	0.9	1.2
Jasminium eminnii Gilg	0.2	-	-	
Jasminium floribunaum Fresen.	-	0.2	-	0.4
Leucas urticifolia	0.1	-	-	0.2
Neptunia sp	-	0.4	-	-
Ocimum gratissimum L.	2.5	0.2	-	0.3
Ocimum suave Willd.	0.9	-	-	-
Pavonia patens (Andr.) Chiov.	-	-	-	0.2
Pergularia daemia (Forsk.) Chiov.	0.2	-	-	-
Phyllanthus maderaspatensis L	2.2	0.2	0.3	1.2
Polygala erioptera DC.	-	-	0.1	0.2
Polygala sphenoptera Fresen	0.1	-	-	-
Portulaca quadrifida L.	3.1	-	-	-
Psilotrichum axilliflorum Suesseng	1.2	1.3	0.7	2.6
Pupalia luppacea (L.) Juss	0.4	1.1	-	0.1
Rhynchosia minima (L.) DC. Varnuda (DC.) Kuntze	-	0.2	-	-
Rullia patula Jacq.	2.7	1.6	3.2	8.0
Sansevieria sp.	2.8	0.7	0.2	-
Schkuria pinnata (Lam.) Ktze	0.6	-	-	-
Sida ovata Forssk	2.6	0.3	0.3	1.5
Sida rhombifolia L	-	-	0.1	-
Solanum incanum L.	0.2	0.1	0.6	0.4

Talinum portulacifolium (Forssk) Schw.	-	-	-	0.2
Tephrosia pumila (Lam.) Pers	-	-	0.1	0.1
Teramnus repens Bak.f. subsp. Repens	0.9	0.7	2.0	0.1
Tribulus terrestris L.	0.6	-	-	-
Vernonia cinerea (L.) Less	0.3	-	-	-
Vigna kirkii (Bak.) Gillett	-	-	-	0.1
Vigna luteola (Jacq.) Benth	-	0.1	-	-
Zehneria pallidinerva (Harms) C. Jeffrey	-	0.1	-	-
Zornia setosa Bak f. Subsp. Obovata (Bak. f) Leon & Milne - Redh.	2.5	-	-	0.1
Wissudula amplissima (L) R.E.	0.1	-	-	-
S	61	56	35	50
ΣX	261.9	331.1	314.2	253.4
$\bar{X}$	4.2	5.9	9.0	5.1
$S_{\bar{X}}$	8.8	15.8	20.3	12.7

Effect of fire frequency on the mean % cover of shrub species

There was also a variation in the cover of shrubs across the study sites with *Capparis tomentosa* Lam., *Acalypha bipartita* Muell. Arg., and *Grewia similis* K. Schum. having higher cover across the study sites. The highest cover of *Capparis tomentosa* Lam. was 7.6% in the second least frequently burnt area followed by the least frequently burnt area (3.7%). The highest cover of *Acalypha bipartita* Muell. Arg. was 8.3% in the second least frequently burnt area and the highest cover of *Grewia similis* K. Schum. was 4.9% in the least frequently burnt area. The rest of the shrubs showed a markedly less cover with some recorded in only one or two study sites. For example, *Lantana camara* L. was only recorded in the second most frequently burnt area (crater area) while *Azima tetracantha* Lam. was recorded in the second least frequently burned area (Kasenyi Track Three). Areas less frequently burnt generally had higher shrub cover compared to those less frequently burnt. The number of shrub species recorded appears to have decreased with fire frequency with the exception (perhaps an anomaly) of the second most frequently burnt area (crater area). The least frequently burnt area and the second most frequently burnt area had the highest number of shrub species recorded (7 in each), the second least frequently burnt area had the second highest number of shrub species (6) while the most frequently burnt area had the lowest number of shrub species recorded (4). There was an apparent decline in number of shrub species recorded with fire frequency.

Table 14: Effects of fire frequency (per 34 year period) on mean cover (%) of some species of shrubs

	Mean cover (% Cover)				
Species	Burnt	1 -2	2 -6	6 -10	10 -14
Acalypha bipartita Muell. Arg.		0.2	8.3	2.0	0.3
Azima tetracantha Lam.		-	0.8	-	-
Balanites aegyptica (L.) Del.		0.1	2.3	-	-
Boscia sp.		-	-	0.4	0.1
Capparis tomentosa Lam.		3.7	7.6	0.9	3.6
Flueggea virosa (Willd.) Voigt		1.2	-	1.1	-
Grewia similis K. Schum.		4.9	0.1	0.1	1.7
Lantana camara L.		-	-	1.7	-
Maerua triphylla A. Rich.		0.2	-	0.7	-
Tarenna graveolens (S. Moore) Brem.		2.6	0.4	-	-
S		7	6	7	4
ΣX		12.9	19.5	6.9	5.7
$\bar{X}$		1.8	2.8	1.0	1.4
$S_{\bar{X}}$		1.9	3.6	0.7	1.6

Effect of fire frequency on the mean % tree cover

Tree cover in all the study sites was dominated by Euphorbia *candelabrum* Kotschy with the highest cover value of 8.8% in the area least frequently burnt followed by the second least frequently burnt area (4.7%). *Acacia gerrardii* Benth. was only recorded in the Crater area (second most frequently burnt) with a % cover of 1.5. *Acacia sieberiana* DC. and *Dichrostachys cinerea* (L.) Wight & Arn. were only recorded in the least frequently burnt area with % covers of 0.1 and 0.5 respectively. *Turraea robusta* Gürke was only recorded in the least frequently burnt and the second least frequently burnt areas where it occupied 0.2% and 0.1% respectively. The areas more frequently burned (PIDA and Crater) had slightly less tree cover than those less frequently burned (Channel Track and Kasenyi Track Three). The number of tree species recorded was highest in the least burned area (Channel Track). The highest number of tree species recorded (4) was in the least frequently burnt area while the lowest number of tree species recorded (1) was in the most frequently burnt area.

Table 15: Effects of fire frequency (per 34 year period) on the mean cover (%) of some tree species

	Mean cover (% Cover)			
Species	Burnt 1 -2	2 -6	6 -10	10 -14
Acacia gerrardii Benth.	-	-	1.5	-
Acacia sieberiana DC.	0.1	-	-	-
Euphorbia candelabrum Kotschy	8.8	4.7	0.3	3.1
Dichrostachys cinerea (L.) Wight & Arn.	0.5	-	-	-
Turraea robusta Gürke	0.2	0.1	-	-
S	4	2	2	1
ΣX	9.6	4.8	1.8	3.1
$\bar{X}$	2.4	2.4	0.9	1.6
$S_{\bar{X}}$	4.3	3.2	0.8	2.1

4.3.5 Variation in relative frequencies, relative covers, relative densities and importance value indices of herbaceous plants with fire frequency.

Relative frequencies of herbaceous plants vs. fire frequency

Table 16 shows variations in relative frequencies of herbaceous plants with fire frequencies. Among the herbaceous plants, grasses generally had higher relative frequencies than non-grasses irrespective of fire frequencies. There were however variations in relative frequencies of plants by fire frequency. *Heteropogon contortus* (L.) Roem. & Schult. had the highest relative frequency in the second most frequently burnt area (0.115) followed by the most frequently burnt area (0.091). Likewise, *Sporobolus pyramidalis* P. Beauv. had higher relative frequencies in the most frequently burnt areas with the highest being in the second most frequently burnt area (0.115). *Bothriochloa insculpta* had the highest relative frequency in the second most frequently burnt area (0.1) followed closely by the most frequently burnt areas (0.085). *Hyparrhenia rufa* (Nees) Stapf had the highest relative frequencies in the second least frequently burnt area and the second most frequently burnt area each having 0.084, this was closely followed by the most frequently burnt area (0.082). *Themeda triandra* Forssk. was only very frequent in the second most frequently burnt area (0.107). *Microchloa kunthii* Desv. was most frequent in the most frequently burnt area (0.091). These percentages show that grasses in particular had higher relative frequencies under more frequent burning.

Table 16: Fire frequency vs. relative frequencies of herbaceous plants

	Relative Frequencies (RFs)				
Species	Burnt	1-2	2-6	6-10	10-14
Aristida adscensionsis L		0.023	-	-	0.003
Bothriochloa insculpta		0.051	0.081	0.100	0.085
Chloris gayana Kunth		0.030	-	-	-
Chloris pycnothrix Trin.		0.042	-	-	0.003
Chloris roxburghiana Schult		-	0.003	-	0.012
Cynodon dactylon (L.) Pers.		-	0.019	-	-
Dactyloctenium aegyptium (L.) Willd.		0.002	-	-	-
Digitaria abyssinica (A. Rich.) Stapf		0.003	-	-	-
Dinebra retroflexa (Vahl) Panzer		0.028	0.003	0.004	-
Enteropogon macrostachys (A Rich) Benth		-	0.006	-	-

Eragrostis exasperata Peter	0.007	0.003	-	0.009
Heteropogon contortus (L.) Roem. & Schult.	0.030	0.081	0.115	0.091
Hyparrhenia rufa (Nees) Stapf	0.047	0.084	0.084	0.082
Hyperthelia dissoluta (Steud.) W.D. Clayton	-	-	0.008	-
Microchloa kunthii Desv.	0.051	0.069	0.050	0.091
Panicum infestum Peters	-	0.006	0.015	0.006
Panicum maximum Jacq.	0.002	0.034	0.011	
Panicum repens	-	0.012	-	-
Panicum subalbidum Kunth	-	0.006	-	-
Setaria homonyma (Steud.) Chiov	0.040	-	-	-
Sporobolus festivus A. Rich	0.002	-	-	-
Sporobolus pyramidalis P. Beauv.	0.052	0.044	0.115	0.067
Themeda triandra Forssk.	-	0.003	0.107	0.009
Tragus berteronianus Schult	0.005	-	-	0.027
Urochloa panicoides Beav	-	-	-	0.006
Cyperus dubius Rottb. Subsp. Dubius	0.009	-	-	
Cyperus teneriffae Poir	0.019	0.016		0.006
Cyperus articulatus L	-	0.012	-	-
Alternanthera sp.	0.002	-	-	-
Abutilon mauritianum (Jacq.) Medic.	0.003	0.016	0.031	-
Achyranthes aspera L.	0.002	0.006	0.011	0.012
Alysicarpus rugosus (Willd) DC. Subsp. Rugosus	0.005	0.034	0.019	0.018
Alysicarpus vaginalis (L.) DC	-	-	0.008	-
Anthericum subpetiolatum (Bak.) Kativu	-	-	-	0.003
Asparagus africanus Lam.	0.003	0.009	-	0.003
Asystasia gangetica (L.) T. Anderson	0.019	0.012	0.008	0.001
Asystasia mysorensis (Roth) T. Anderson	-	-	0.011	0.003
Bidens pilosa L	-	0.003	-	0.009
Blepharis maderaspatensis (L.) Roth	-	0.009	0.004	-
Boerhavia diffusa L	0.003	0.006	0.008	0.052
Caralluma dummeri N.E. Br	0.002	-	-	-
Cassia absus L	-	0.012	-	-
Cassia mimosoides L.	-	0.003	-	0.003
Cissus quadrangularis L.	0.016	0.037	0.011	0.018
Cissus rotundifolia (Forsk.) Vahl.	0.019	0.006	-	-
Commelina benghalensis L	0.017	0.028	0.011	0.003
Corbichonia decumbens (Forsk.) Exell	-	0.003	-	0.009
Corchoris trilocularis L	-	0.003	-	-
Craterostigma plantangineum Hochst	0.016	-	-	-
Crotalaria spinosa Benth.	0.016	0.016	-	0.027
Cucumis figarei Naud	-	-	-	0.003
Cyphostemma adenocayle (A. Rich.) Wild. & Drummond	0.005	0.022	0.019	0.012
Dyschoriste nagchana (Nees) Bennet	0.010	0.053	-	0.021
Euphorbia glomerifera (mill sp.) Wheeler	-	0.012	-	-
Evolvolus alsinoides (L.) L.	0.017	-	0.015	0.012
Evolvolus nummularius (L.) L.	0.031	0.009	-	-

Fuerstia africana Th. C.E. Fries	0.007	0.003	0.008	0.009
Gomphrena celosioides Mart	0.002	-	-	-
Hibiscus aponeurus Sprague & Hutch.	0.014	0.006		
Hoslundia opposita Vahl.	0.024	0.022	0.019	0.015
Hygrophila auriculata (Schumach.) Heine	-	0.006	-	-
Indigofera circinella Bak. f.	0.021	-	-	-
Indigofera vohemarensis Baill.	0.051	0.012	0.038	0.037
Jasminium eminnii Gilg	0.002	-	-	-
Jasminium floribunaum Fresen.	-	0.006	-	0.003
Leucas urticifolia	0.002	-	-	0.003
Neptunia sp	-	0.009	-	-
Ocimum gratissimum L.	0.009	0.003	-	0.006
Ocimum suave Willd.	0.009	-	-	-
Pavonia patens (Andr.) Chiov.	-	-	-	0.003
Pergularia daemia (Forsk.) Chiov.	0.003	-	-	-
Phyllanthus maderaspatensis L	0.037	0.006	0.015	0.049
Polygala erioptera DC.	-	-	0.008	0.009
Polygala sphenoptera Fresen	0.003	-	-	-
Portulaca quadrifida L.	0.010	-	-	-
Psilotrichum axilliflorum Suesseng	0.009	0.016	0.008	0.015
Pupalia luppacea (L.) Juss	0.005	0.006		0.003
Rhynchosia minima (L.) DC. Varnuda (DC.) Kuntze	-	0.003	-	-
Rullia patula Jacq.	0.030	0.059	0.031	0.085
Sansevieria sp.	0.003	0.009	0.004	-
Schkuria pinnata (Lam.) Ktze	0.017	-	-	-
Sida ovata Forssk	0.037	0.012	0.008	0.012
Sida rhombifolia L	-	-	0.004	-
Solanum incanum L.	0.002	0.003	0.031	0.006
Talinum portulacifolium (Forssk) Schw.	-	-	-	0.003
Tephrosia pumila (Lam.) Pers			0.004	0.003
Teramnus repens Bak.f. subsp. Repens	0.014	0.019	0.057	0.003
Tribulus terrestris L.	0.010	-	-	-
Vernonia cinerea (L.) Less	0.003	-	-	-
Vigna kirkii (Bak.) Gillett	-	-	-	0.003
Vigna luteola (Jacq.) Benth	-	0.003	-	-
Zehneria pallidinerva (Harms) C. Jeffrey	-	0.009	-	-
Zornia setosa Bak f. Subsp. Obovata (Bak. f) Leon & Milne - Redh.	0.045	-	-	0.021
Wissudula amplissima (L) R.E.	0.002	-	-	-
S	61	56	35	50
ΣX	1.000	1.000	1.000	1.000
$\bar{X}$	0.011	0.011	0.011	0.011
$S_{\bar{X}}$	0.015	0.018	0.025	0.022

Relative covers of herbaceous plants vs. fire frequency

Table 17 shows variations in relative covers of herbaceous plants with fire frequency. Like in the case of relative frequencies, relative covers of herbaceous plants were higher for grasses and lower for the other species of herbaceous plants. *Heteropogon contortus* (L.) Roem. & Schult. had the highest relative cover in the second least frequently burnt area (0.276) followed closely by the most frequently burnt area (0.249). *Sporobolus pyramidalis* P. Beauv. had the highest relative cover in the second most frequently burnt area (0.237). *Bothriochloa insculpta* had the highest relative cover in the most frequently burnt area (0.114) closely followed by the second least frequently burnt area (0.113). *Hyparrhenia rufa* (Nees) Stapf had the highest relative cover in the second least frequently burnt area (0.162) followed by the most frequently burnt area (0.138). *Themeda triandra* Forssk. only had relative cover in the second most frequently burnt area (0.184). *Microchloa kunthii* Desv. had the highest relative cover in the most frequently burnt area (0.183).

Table 17: Fire frequency vs. relative covers of herbaceous plants

	Relative Covers (RCs)			
Species	Burnt 1-2	2-6	6-10	10-14
Aristida adscensionsis L	0.023	-	-	0.004
Bothriochloa insculpta	0.039	0.113	0.088	0.114
Chloris gayana Kunth	0.031	-	-	-
Chloris pycnothrix Trin.	0.098			0.005
Chloris roxburghiana Schult	-	0.001	-	0.008
Cynodon dactylon (L.) Pers.	-	0.113	-	-
Dactyloctenium aegyptium (L.) Willd.	0.001	-	-	-
Digitaria abyssinica (A. Rich.) Stapf	0.008	-	-	-
Dinebra retroflexa (Vahl) Panzer	0.013	0.001	0.001	-
Enteropogon macrostachys (A Rich) Benth	-	0.001	-	-
Eragrostis exasperata Peter	0.002	0.001	-	0.001
Heteropogon contortus (L.) Roem. & Schult.	0.016	0.276	0.241	0.249
Hyparrhenia rufa (Nees) Stapf	0.089	0.162	0.111	0.138
Hyperthelia dissoluta (Steud.) W.D. Clayton	-	-	0.002	-
Microchloa kunthii Desv.	0.061	0.038	0.028	0.183
Panicum infestum Peters	-	0.003	0.001	0.001
Panicum maximum Jacq.	0.001	0.015	0.001	-

Panicum repens	-	0.035	-	-
Panicum subalbidum Kunth	-	0.004	-	-
Setaria homonyma (Steud.) Chiov	0.171	-	-	-
Sporobolus festivus A. Rich	0.001	-	-	-
Sporobolus pyramidalis P. Beauv.	0.155	0.102	0.237	0.119
Themeda triandra Forssk.	-	0.001	0.184	0.001
Tragus berteronianus Schult	0.001	-	-	0.004
Urochloa panicoides Beav	-	-	-	0.001
Cyperus dubius Rottb. Subsp. Dubius	0.002	-	-	-
Cyperus teneriffae Poir	0.007	0.002		0.001
Cyperus articulatus L	-	0.008	-	-
Alternanthera sp.	0.001	-	-	-
Abutilon mauritianum (Jacq.) Medic.	0.001	0.013	0.009	-
Achyranthes aspera L.	0.001	0.004	0.017	0.018
Alysicarpus rugosus (Willd) DC. Subsp. Rugosus	0.001	0.002	0.001	0.002
Alysicarpus vaginalis (L.) DC	-	-	0.002	-
Anthericum subpetiolatum (Bak.) Kativu	-	-	-	0.001
Asparagus africanus Lam.	0.001	0.002	-	0.001
Asystasia gangetica (L.) T. Anderson	0.009	0.004	0.001	0.001
Asystasia mysorensis (Roth) T. Anderson	-	-	0.001	0.001
Bidens pilosa L	-	0.001	-	0.001
Blepharis maderaspatensis (L.) Roth	-	0.001	0.001	-
Boerhavia diffusa L	0.001	0.001	0.001	0.008
Caralluma dummeri N.E. Br	0.001	-	-	-
Cassia absus L	-	0.001	-	-
Cassia mimosoides L.	-	0.001	-	0.001
Cissus quadrangularis L.	0.015	0.039	0.014	0.031
Cissus rotundifolia (Forsk.) Vahl.	0.013	0.002	-	-
Commelina benghalensis L	0.003	0.006	0.001	0.001
Corbichonia decumbens (Forsk.) Exell	-	0.001	-	0.002
Corchoris trilocularis L	-	0.001	-	-
Craterostigma plantangineum Hochst	0.012	-	-	-
Crotalaria spinosa Benth.	0.001	0.001	-	0.005
Cucumis figarei Naud	-	-	-	0.001
Cyphostemma adenocayle (A. Rich.) Wild. & Drummond	0.002	0.004	0.004	0.004
Dyschoriste nagchana (Nees) Bennet	0.002	0.006	-	0.010
Euphorbia glomerifera (mill sp.) Wheeler	-	0.001	-	-
Evolvolus alsinoides (L.) L.	0.002	-	0.001	0.001
Evolvolus nummularius (L.) L.	0.035	0.002	-	-
Fuerstia africana Th. C.E. Fries	0.001	0.001	0.001	0.004
Gomphrena celosioides Mart	0.001	-	-	-
Hibiscus aponeurus Sprague & Hutch.	0.002	0.001		
Hoslundia opposita Vahl.	0.044	0.014	0.030	0.023
Hygrophila auriculata (Schumach.) Heine	-	0.002	-	-
Indigofera circinella Bak. f.	0.005	-	-	-
Indigofera vohemarensis Baill.	0.040	0.001	0.003	0.005

Jasminium eminnii Gilg	0.001	-	-	-
Jasminium floribunaum Fresen.	-	0.001	-	0.002
Leucas urticifolia	0.001	-	-	0.001
Neptunia sp	-	0.001	-	-
Ocimum gratissimum L.	0.010	0.001	-	0.001
Ocimum suave Willd.	0.003	-	-	-
Pavonia patens (Andr.) Chiov.	-	-	-	0.001
Pergularia daemia (Forsk.) Chiov.	0.001	-	-	-
Phyllanthus maderaspatensis L	0.008	0.001	0.001	0.005
Polygala erioptera DC.	-	-	0.001	0.001
Polygala sphenoptera Fresen	0.001	-	-	-
Portulaca quadrifida L.	0.012	-	-	-
Psilotrichum axilliflorum Suesseng	0.005	0.004	0.002	0.010
Pupalia luppacea (L.) Juss	0.002	0.003		0.001
Rhynchosia minima (L.) DC. Varnuda (DC.) Kuntze	-	0.001	-	-
Rullia patula Jacq.	0.010	0.005	0.010	0.032
Sansevieria sp.	0.011	0.002	0.001	-
Schkuria pinnata (Lam.) Ktze	0.002	-	-	-
Sida ovata Forssk	0.010	0.001	0.001	0.006
Sida rhombifolia L	-	-	0.001	-
Solanum incanum L.	0.001	0.001	0.002	0.002
Talinum portulacifolium (Forssk) Schw.	-	-	-	0.001
Tephrosia pumila (Lam.) Pers			0.001	0.001
Teramnus repens Bak.f. subsp. Repens	0.003	0.002	0.006	0.001
Tribulus terrestris L.	0.002	-	-	-
Vernonia cinerea (L.) Less	0.001	-	-	-
Vigna kirkii (Bak.) Gillett	-	-	-	0.001
Vigna luteola (Jacq.) Benth	-	0.001	-	-
Zehneria pallidinerva (Harms) C. Jeffrey	-	0.001	-	-
Zornia setosa Bak f. Subsp. Obovata (Bak. f) Leon & Milne - Redh.	0.010	-	-	0.001
Wissudula amplissima (L) R.E.	0.001	-	-	-
S	61	56	35	50
ΣX	1.008	1.007	1.005	1.009
$\bar{X}$	0.011	0.011	0.011	0.011
$S_{\bar{X}}$	0.028	0.038	0.041	0.038

Relative densities of herbaceous plants vs. fire frequency

Table 18 shows variations in relative densities of herbaceous plants with fire frequency. As in the case of relative frequencies and relative covers, relative densities of herbaceous plants were higher for grasses and lower for the other species of herbaceous plants. *Heteropogon contortus* (L.) Roem. & Schult. for example, tended to have the highest relative density in the most frequently burnt area (0.38) followed closely by the second least frequently burnt area (0.36). *Sporobolus pyramidalis* P. Beauv. apparently had the highest relative density in the second most frequently burnt area (0.23) followed by the least frequently burnt area (0.18). *Bothriochloa insculpta* tended to have the highest relative density in the second most frequently burnt area (0.09); this was followed closely by the second least frequently burnt area (0.07). *Hyparrhenia rufa* (Nees) Stapf had apparently the highest relative density in the second least frequently burnt area (0.20) followed by the second most frequently burnt area (0.07). *Microchloa kunthii* Desv. had the highest relative density in the most frequently burnt area (0.37). *Themeda triandra* Forssk. had a relatively high relative density in the second most frequently burnt area (0.13) and had much lower relative densities in the other areas where it occured.

Table 18: Fire frequency vs. relative densities of herbaceous plants

	Relative Densities (RDs)			
Species	Burnt 1-2	2-6	6-10	10-14
Aristida adscensionsis L	0.030	-	-	0.001
Bothriochloa insculpta	0.040	0.070	0.090	0.060
Chloris gayana Kunth	0.020	-	-	-
Chloris pycnothrix Trin.	0.120			0.001
Chloris roxburghiana Schult	-	0.001	-	0.001
Cynodon dactylon (L.) Pers.	-	0.070	-	-
Dactyloctenium aegyptium (L.) Willd.	0.001	-	-	-
Digitaria abyssinica (A. Rich.) Stapf	0.001	-	-	-
Dinebra retroflexa (Vahl) Panzer	0.010	0.001	0.001	-
Enteropogon macrostachys (A Rich) Benth	-	0.001	-	-
Eragrostis exasperata Peter	0.001	0.001	-	0.001
Heteropogon contortus (L.) Roem. & Schult.	0.020	0.360	0.300	0.380
Hyparrhenia rufa (Nees) Stapf	0.040	0.200	0.070	0.060
Hyperthelia dissoluta (Steud.) W.D. Clayton	-	-	0.001	-
Microchloa kunthii Desv.	0.180	0.120	0.060	0.370

Panicum infestum Peters	-	0.001	0.001	0.001
Panicum maximum Jacq.	0.001	0.010	0.001	-
Panicum repens	-	0.010	-	-
Panicum subalbidum Kunth	-	0.001	-	-
Setaria homonyma (Steud.) Chiov	0.110	-	-	-
Sporobolus festivus A. Rich	0.001	-	-	-
Sporobolus pyramidalis P. Beauv.	0.180	0.030	0.230	0.040
Themeda triandra Forssk.	-	0.001	0.130	0.001
Tragus berteronianus Schult	0.001	-	-	0.010
Urochloa panicoides Beav	-	-	-	0.001
Cyperus dubius Rottb. Subsp. Dubius	0.001	-	-	-
Cyperus teneriffae Poir	0.001	0.001	-	0.001
Cyperus articulatus L	-	0.030	-	-
Alternanthera sp.	0.001	-	-	-
Abutilon mauritianum (Jacq.) Medic.	0.001	0.001	0.010	-
Achyranthes aspera L.	0.001	0.001	0.001	0.001
Alysicarpus rugosus (Willd) DC. Subsp. Rugosus	0.001	0.010	0.001	0.001
Alysicarpus vaginalis (L.) DC	-	-	0.001	-
Anthericum subpetiolatum (Bak.) Kativu	-	-	-	0.001
Asparagus africanus Lam.	0.001	0.001	-	0.001
Asystasia gangetica (L.) T. Anderson	0.001	0.001	0.001	0.001
Asystasia mysorensis (Roth) T. Anderson	-	-	0.001	0.001
Bidens pilosa L	-	0.001	-	0.001
Blepharis maderaspatensis (L.) Roth	-	0.001	0.001	-
Boerhavia diffusa L	0.001	0.001	0.001	0.100
Caralluma dummeri N.E. Br	0.001	-	-	-
Cassia absus L	-	0.001	-	-
Cassia mimosoides L.	-	0.001	-	0.001
Cissus quadrangularis L.	0.001	0.001	0.001	0.001
Cissus rotundifolia (Forsk.) Vahl.	0.001	0.001	-	-
Commelina benghalensis L	0.001	0.010	0.001	0.001
Corbichonia decumbens (Forsk.) Exell	-	0.001	-	0.001
Corchoris trilocularis L	-	0.001	-	-
Craterostigma plantangineum Hochst	0.030	-	-	-
Crotalaria spinosa Benth.	0.001	0.001	-	0.001
Cucumis figarei Naud	-	-	-	0.001
Cyphostemma adenocayle (A. Rich.) Wild. & Drummond	0.001	0.001	0.001	0.001
Dyschoriste nagchana (Nees) Bennet	0.100	0.020	-	0.000
Euphorbia glomerifera (mill sp.) Wheeler	-	0.001	-	-
Evolvolus alsinoides (L.) L.	0.001	-	0.001	0.001
Evolvolus nummularius (L.) L.	0.040	0.010	-	-
Fuerstia africana Th. C.E. Fries	0.001	0.001	0.001	0.001
Gomphrena celosioides Mart	0.001	-	-	-
Hibiscus aponeurus Sprague & Hutch.	0.001	0.001	-	-
Hoslundia opposita Vahl.	0.001	0.001	0.001	0.001
Hygrophila auriculata (Schumach.) Heine	-	0.010	-	-

Indigofera circinella Bak. f.	0.010	-	-	-
Indigofera vohemarensis Baill.	0.050	0.001	0.010	0.010
Jasminium eminnii Gilg	0.001	-	-	-
Jasminium floribunaum Fresen.	-	0.001	-	0.001
Leucas urticifolia	0.001	-	-	0.001
Neptunia sp	-	0.001	-	-
Ocimum gratissimum L.	0.001	0.001	-	0.001
Ocimum suave Willd.	0.001	-	-	-
Pavonia patens (Andr.) Chiov.	-	-	-	0.001
Pergularia daemia (Forsk.) Chiov.	0.001	-	-	-
Phyllanthus maderaspatensis L	0.020	0.001	0.001	0.010
Polygala erioptera DC.	-	-	0.001	0.001
Polygala sphenoptera Fresen	0.001	-	-	-
Portulaca quadrifida L.	0.010	-	-	-
Psilotrichum axilliflorum Suesseng	0.001	0.001	0.001	0.001
Pupalia luppacea (L.) Juss	0.001	0.001	-	0.001
Rhynchosia minima (L.) DC. Varnuda (DC.) Kuntze	-	0.001	-	-
Rullia patula Jacq.	0.010	0.020	0.040	0.040
Sansevieria sp.	0.001	0.001	0.001	-
Schkuria pinnata (Lam.) Ktze	0.001	-	-	-
Sida ovata Forssk	0.010	0.001	0.001	0.001
Sida rhombifolia L	-	-	0.001	-
Solanum incanum L.	0.001	0.001	0.001	0.001
Talinum portulacifolium (Forssk) Schw.	-	-	-	0.001
Tephrosia pumila (Lam.) Pers	-	-	0.001	0.001
Teramnus repens Bak.f. subsp. Repens	0.001	0.001	0.020	0.001
Tribulus terrestris L.	0.001	-	-	-
Vernonia cinerea (L.) Less	0.001	-	-	-
Vigna kirkii (Bak.) Gillett	-	-	-	0.001
Vigna luteola (Jacq.) Benth	-	0.001	-	-
Zehneria pallidinerva (Harms) C. Jeffrey	-	0.001	-	-
Zornia setosa Bak f. Subsp. Obovata (Bak. f) Leon & Milne - Redh.	0.020	-	-	0.001
Wissudula amplissima (L) R.E.	0.001	-	-	-
S	61	56	35	50
ΣX	1.091	1.021	0.985	1.120
$\overline{X}$	0.012	0.011	0.010	0.012
$S_{\overline{X}}$	0.033	0.045	0.043	0.056

Importance value indices (IVIs) of herbaceous plants vs. fire frequency

Table 19 shows importance value indices (IVIs) of herbaceous plants by fire frequency. Importance value indices of herbaceous plants varied considerably with fire frequency. IVIs were much higher for grasses than the other species herbaceous plants. *Heteropogon contortus* (L.) Roem. & Schult. for example tended to have the highest IVI (0.72) in the most frequently burnt area followed closely by the second least frequently burnt area (0.717). *Sporobolus pyramidalis* P. Beauv. had apparently the highest IVI in the second most frequently burnt area (0.582) followed by the least frequently burnt area (0.388). *Bothriochloa insculpta* appeared to have the highest IVI in the second most frequently burnt area (0.277) followed closely by the most frequently burnt and second least frequently burnt areas (0.259). *Hyparrhenia rufa* (Nees) Stapf was observed to have the highest IVI in the second least frequently burnt area (0.446) followed by the most frequently burnt area (0.28). *Microchloa kunthii* Desv. tended to have a much higher IVI in the most frequently burnt area (0.644) and lower IVIs in the other areas. *Themeda triandra* Forssk. tended to have a relatively high IVI in the second most frequently burnt area (0.421). *Setaria homonyma* (Steud.) Chiov tended to be more important under low fire frequency with an IVI of 0.321. In the least frequently burnt area, non-grasses tended to have lower IVIs compared to grasses but there were also variations in their IVIs with fire frequency. For example, *Indigofera vohemarensis* Baill. had a relatively higher IVI in the least frequently burnt area (0.141) compared to the other areas each with an IVI of 0.051 for the this species (*Indigofera vohemarensis* Baill.). *Dyschoriste nagchana* (Nees) Bennet also tended to have a relatively higher IVI in the least frequently burnt area (0.112) compared to the other areas where it occured. Most non-grasses tended to have relatively higher IVIs in the least frequently burnt area.

Table 19: Fire frequency vs. importance value indices of herbaceous plants

Species	Importance Value Indices (IVIs)			
	Burnt 1-2	2-6	6-10	10-14
Aristida adscensionsis L	0.075	-	-	0.008
Bothriochloa insculpta	0.130	0.264	0.277	0.259
Chloris gayana Kunth	0.081	-	-	-
Chloris pycnothrix Trin.	0.260	-	-	0.009
Chloris roxburghiana Schult	-	0.005	-	0.021
Cynodon dactylon (L.) Pers.	-	0.202	-	-
Dactyloctenium aegyptium (L.) Willd.	0.004	-	-	-
Digitaria abyssinica (A. Rich.) Stapf	0.013	-	-	-
Dinebra retroflexa (Vahl) Panzer	0.050	0.005	0.006	-
Enteropogon macrostachys (A Rich) Benth	-	0.008	-	-
Eragrostis exasperata Peter	0.010	0.005	-	0.010
Heteropogon contortus (L.) Roem. & Schult.	0.065	0.717	0.656	0.720
Hyparrhenia rufa (Nees) Stapf	0.176	0.446	0.265	0.280
Hyperthelia dissoluta (Steud.) W.D. Clayton	-	-	0.011	-
Microchloa kunthii Desv.	0.292	0.227	0.138	0.644
Panicum infestum Peters	-	0.010	0.018	0.008
Panicum maximum Jacq.	0.004	0.059	0.013	-
Panicum repens	-	0.057	-	-
Panicum subalbidum Kunth	-	0.011	-	-
Setaria homonyma (Steud.) Chiov	0.321	-	-	-
Sporobolus festivus A. Rich	0.004	-	-	-
Sporobolus pyramidalis P. Beauv.	0.388	0.175	0.582	0.226
Themeda triandra Forssk.	-	0.005	0.421	0.011
Tragus berteronianus Schult	0.007	-	-	0.041
Urochloa panicoides Beav	-	-	-	0.008
Cyperus dubius Rottb. Subsp. Dubius	0.012	-	-	-
Cyperus teneriffae Poir	0.027	0.018	-	0.008
Cyperus articulatus L	-	0.051	-	-
Alternanthera sp.	0.004	-	-	-
Abutilon mauritianum (Jacq.) Medic.	0.006	0.029	0.050	-
Achyranthes aspera L.	0.004	0.011	0.029	0.031
Alysicarpus rugosus (Willd) DC. Subsp. Rugosus	0.007	0.047	0.021	0.022
Alysicarpus vaginalis (L.) DC	-	-	0.010	-
Anthericum subpetiolatum (Bak.) Kativu	-	-	-	0.005
Asparagus africanus Lam.	0.005	0.012	-	0.005
Asystasia gangetica (L.) T. Anderson	0.029	0.017	0.010	0.002
Asystasia mysorensis (Roth) T. Anderson	-	-	0.013	0.005
Bidens pilosa L	-	0.005	-	0.011
Blepharis maderaspatensis (L.) Roth	-	0.011	0.006	-
Boerhavia diffusa L	0.005	0.008	0.010	0.160

Caralluma dummeri N.E. Br	0.004	-	-	-
Cassia absus L	-	0.014	-	-
Cassia mimosoides L.	-	0.005	-	0.005
Cissus quadrangularis L.	0.032	0.077	0.026	0.050
Cissus rotundifolia (Forsk.) Vahl.	0.033	0.009	-	-
Commelina benghalensis L	0.022	0.044	0.013	0.005
Corbichonia decumbens (Forsk.) Exell	-	0.005	-	0.012
Corchoris trilocularis L	-	0.005	-	-
Craterostigma plantangineum Hochst	0.058	-	-	-
Crotalaria spinosa Benth.	0.018	0.017	-	0.034
Cucumis figarei Naud	-	-	-	0.005
Cyphostemma adenocayle (A. Rich.) Wild. & Drummond	0.008	0.027	0.024	0.017
Dyschoriste nagchana (Nees) Bennet	0.112	0.079	-	0.032
Euphorbia glomerifera (mill sp.) Wheeler	-	0.015	-	-
Evolvolus alsinoides (L.) L.	0.021	-	0.017	0.014
Evolvolus nummularius (L.) L.	0.106	0.022	-	-
Fuerstia africana Th. C.E. Fries	0.009	0.005	0.010	0.014
Gomphrena celosioides Mart	0.004	-	-	-
Hibiscus aponeurus Sprague & Hutch.	0.017	0.008	-	-
Hoslundia opposita Vahl.	0.070	0.036	0.050	0.039
Hygrophila auriculata (Schumach.) Heine	-	0.018	-	-
Indigofera circinella Bak. f.	0.035	-	-	-
Indigofera vohemarensis Baill.	0.141	0.015	0.051	0.051
Jasminium eminnii Gilg	0.004	-	-	-
Jasminium floribunaum Fresen.	-	0.008	-	0.006
Leucas urticifolia	0.004	-	-	0.005
Neptunia sp	-	0.012	-	-
Ocimum gratissimum L.	0.019	0.005	-	0.008
Ocimum suave Willd.	0.013	-	-	-
Pavonia patens (Andr.) Chiov.	-	-	-	0.005
Pergularia daemia (Forsk.) Chiov.	0.005	-	-	-
Phyllanthus maderaspatensis L	0.065	0.008	0.017	0.064
Polygala erioptera DC.	-	-	0.010	0.011
Polygala sphenoptera Fresen	0.005	-	-	-
Portulaca quadrifida L.	0.032	-	-	-
Psilotrichum axilliflorum Suesseng	0.014	0.021	0.011	0.027
Pupalia luppacea (L.) Juss	0.008	0.011	-	0.005
Rhynchosia minima (L.) DC. Varnuda (DC.) Kuntze	-	0.005	-	-
Rullia patula Jacq.	0.050	0.084	0.081	0.157
Sansevieria sp.	0.015	0.012	0.005	-
Schkuria pinnata (Lam.) Ktze	0.021	-	-	-
Sida ovata Forssk	0.057	0.014	0.010	0.019
Sida rhombifolia L	-	-	0.006	-
Solanum incanum L.	0.004	0.005	0.034	0.009
Talinum portulacifolium (Forssk) Schw.	-	-	-	0.005
Tephrosia pumila (Lam.) Pers			0.006	0.005

Teramnus repens Bak.f. subsp. Repens	0.018	0.022	0.084	0.005
Tribulus terrestris L.	0.014	-	-	-
Vernonia cinerea (L.) Less	0.006	-	-	-
Vigna kirkii (Bak.) Gillett	-	-	-	0.005
Vigna luteola (Jacq.) Benth	-	0.005	-	-
Zehneria pallidinerva (Harms) C. Jeffrey	-	0.011	-	-
Zornia setosa Bak f. Subsp. Obovata (Bak. f) Leon & Milne - Redh.	0.075	-	-	0.023
Wissudula amplissima (L) R.E.	0.004	-	-	-
S	61	56	35	50
ΣX	3.099	3.028	2.990	3.129
$\bar{X}$	0.033	0.032	0.032	0.033
$S_{\bar{X}}$	0.069	0.096	0.106	0.109

4.4 Effects of fire frequency on plant species composition

A total of 109 species were recorded across the study sites; 94 species of herbaceous plants, 10 species of shrubs and 5 species of trees (Table 20). Herbaceous plants dominated in all the study sites. The least burnt area had the highest number tree species. The most frequently burned area had the lowest number of shrubs.

The number of species recorded per site ranged from 44 in the second most frequently burnt area to 72 in the least frequently burnt area. The total number of species recorded generally decreased with fire frequency (Table 20).

Table 20: Plant life forms and number of species recorded at each study site

Life form	Number of species recorded: Burnt 1 -2	2 -6	6 -10	10 -14	Total species by life form
Herbaceous plants	61	56	35	50	94
Shrubs	07	06	07	04	10
Trees	04	02	02	02	05
Total no. of species recorded	72	64	44	56	109

CHAPTER FIVE

DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

5.1 Effects of fire frequency on species richness

Fire constitutes disturbance in any ecosystem, but should be recognized as a common natural event in grasslands, savannas, and in many temperate forests and tropical savannas (Mooney 1981 & Nuzzo 1986). The results of this study showed that species richness was highest for lowest fire frequency (1-2 burns in 34 years) and it decreased exponentially with fire frequency.

David & Peter (2008) conducted a study on how fire frequency and tree canopy structure influence plant species diversity in a forest-grassland ecotone and reported a general increase in species richness with fire frequency but noted that grass species richness tended to decrease at fire frequencies greater than five fires per decade.

Similarly, in this study, mean species richness of herbaceous plants (including grasses) decreased with fire frequency. Since the fires in QENP were both management/prescribed fires and unprescribed (those started by unknown factors), areas with higher fire frequencies in the park must have experienced some of these fires so frequently/at short intervals that they did not have enough time for fire intolerant species to recover/regenerate thereby eliminating some of them and reducing species richness in these areas.

Kucera and Koelling (1964) reported that burning every other year in Missouri prairie controlled invasion of woody species, thereby affecting overall species richness. It seems likely that the relation between diversity and frequency of burns is primarily dependent upon the type of community. In the case of the QENP, the vegetation community consisted of herbaceous plants, shrubs and trees and the decrease in overall species richness with fire frequency (Figure 4) could have been due to the decreasing species richness of especially the herbaceous plants with fire frequency (Figure 1) much morethan decreases in species richness of shrubs (Figure 2) and trees (Figure 3) with fire frequency.

As noted by Kucera & Koelling (1964), more frequent burning in the QENP must have also controlled the invasion of woody species (shrubs and trees) in sites most frequently burned in the park thus resulting in reduction in overall species richness of shrubs and trees. Species richness of shrubs and trees (woody species) were highest in sites with less fire frequency and declined with increasing fire frequency (Figures 2 and 3 respectively). This is consistent with the findings of Veen *et al.*, (2008) who observed that, species richness of woody species decreases with fire frequency.
Maximum species richness in annually burnt areas corroborated with findings of several studies (Lewis & Harshbarger 1976; Walker & Peet 1983).

Tree and shrub dominance increases on many savanna sites in the absence of fire (Nuzzo 1986; Mapesa *et al.,* 1996; Faber-Langendoen & Davis, 1995). Fires kill aboveground stems of most shrubs and tree saplings, thereby reducing their ability to preemptively intercept light and competitively exclude herbaceous plants via shading (Faber-Langendoen & Davis, 1995). This partly explains the decreasing species richness of shrubs and trees (woody species) with fire frequency of. David & Peter (2008) also reported that woody plant species richness was highest in plots protected from fire and declined with increasing fire frequency.

Peterson *et al*., (2007) noted that high fire frequencies appear sufficient to limit woody plant cover and to minimize herb species richness. Higher fire frequencies reduce woody plant species richness without producing equivalent increases in species richness of herbaceous plants (Peterson *et al*., 2007). He further argued that fires also modify and constrain over-story tree cover and diversity by periodically killing over-story trees, limiting growth of saplings into the over-story canopy, and eliminating regeneration of shade-tolerant, but fire intolerant, tree species. In the case of the QENP, the change in species richness with fire frequency followed a quadratic trend with a maximum for 1-2x and a minimum for 6-10x. The decrease in species richness of shrubs and trees (woody species) did not produce a corresponding increase in species richness of herbaceous plants thus resulting in decrease in overall species richness. Although this study did not particularly focus on the species richness of over-story trees, it is important that such categorization be done in future studies for purposes of comparison with under-story species richness. This is because the effects of tree canopy are bound to cause differences in species richness.

Fires damage or kill woody plants, reduce woody plant cover, and reduce woody species frequencies (Axelrod & Irving, 1978; Waldrop *et al*., 1992; Peterson, 1998). In contrast, most grasses sustain little damage from fires, so effective fire severity is low, and species richness in the grasses group is maximized at very high fire frequencies that minimize competition from woody plants. The ability of most grasses to sustain little damage from fires probably accounts for the apparent later increase in species of herbaceous plants as the frequency of burning kept on increasing from the second most frequently burnt area (6-10 burns) to the most frequently burnt area (10-14 times) in QENP. The highest fire frequency of 10-14 burns in the QENP might not have been high enough to maximize species richness of herbaceous plants but did indicate an increase in species richness as fire frequency kept on increasing.

Steinauer *et al*., (2001) and Augustine *et al*., (2001) noted that activities of large herbivores, such as selective grazing, nutrient deposition (i.e. urine and dung patches) and soil disturbance (i.e. trampling and wallowing) increase species richness, presumably by producing heterogeneity in resource availability and altering species composition and community structure (Steinauer *et al*., 2001; Augustine *et al*. 2001). In the case of the QENP, R^2 for species richness was strongly related to fire frequency ($R^2 > 0.9$). Therefore, fire frequency explains almost all the variation in species in the QENP.

Studies by Coppedge & Shaw (1998) also showed that fire and grazing can interact at landscape scales through a series of positive and negative feedbacks, because fire influences grazing patterns and grazing, in turn, modifies the effect of fire by altering the accumulation and distribution of fire fuels (i.e. litter). Such feedbacks create a spatially and temporarily variable mosaic of vegetation structure (Fuhlendorf & Engle, 2001, 2004).

Although high fire frequencies seem to have promoted low species richness in this study, fire frequency effects may vary with local site factors like soil fertility, topography, and species pools that influence vegetation productivity, competitive interactions, and fire resiliency.

If maximum species richness is a major objective in protected area conservation efforts, then burning regimes must be carefully studied in the short, intermediate and long-term in order to find one which supports/enhances high species richness and diversity.

5.2 Effects of fire frequency on species diversity

Species diversity of herbaceous plants, trees and all vegetation changed quadratically with fire frequency while that of shrubs changed exponentially with fire frequency.

Species diversity for shrubs and trees were strongly related to fire frequency ($R^2 > 0.9$). Therefore, fire frequency explains almost all the variation in species richness of the woody species in the QENP.

Disturbances can reduce plant species diversity by eliminating disturbance-sensitive species, increase species diversity by opening up growing space and resources for use by colonizing species, maintain species richness by slowing or preventing competitive exclusion, and alter spatial heterogeneity in plant community composition (Huston, 1994). Borrowing from Huston's observation, the general increase in species diversity of herbaceous plants could have been due to the availability of growing space and resources following burning, which eventually led to increase in species diversity of all vegetation with fire frequency. On the other hand, decrease in species diversity of woody species was probably due to the elimination of fire sensitive species as noted by Huston (1994). However, in the case of the QENP, decrease in the diversity of woody speies with fire frequency did not produce a corresponding decrease in the species diversity of all vegetation which instead increased fire frequency.

In savannas and woodlands, plant species diversity often increases with increasing fire frequency up to biennial (Tester, 1989; Brockway & Lewis, 1997) or annual (Walker &

Peet, 1983) fire frequencies. Although this study did not specify biennial and annual fire frequencies, the increase in species diversity of all vegetation with fire frequency was constant.

Fire frequency may also influence under-story species diversity indirectly in savanna, woodland, and forest ecosystems through its influence on over-story tree density and canopy cover. Huston (1994) proposed that high fire frequencies promote high species diversity in savannas by preventing competitive exclusion of grasses and forbs by woody plants. This may have partly accounted for the increase in species diversity of herbaceous plants and eventually of all vegetation in the QENP.

High species diversity in fire-maintained savannas may also be the result of intermediate resource availability that reduces dominance and competitive exclusion by both woody plants and grasses and allows greater herb species diversity (Leach & Givnish 1999).

5.3 Effects of fire frequency on vegetation cover

Percentage cover of vegetation in the QENP generally changed quadratically with fire frequency. The % covers of herbaceous plants and of all vegetation were strongly correlated to fire frequency ($R^2 > 0.9$). Therefore, fire frequency explains almost all the variation in species richness of herbaceous plants and of all vegetation in the QENP.

In general, species that grow rapidly are better able to endure repeated burning. Fire stimulates tillering in tropical grasses, including East African species (Pratt & Gwynne, 1977; Edroma, 1984). While late dry-season fires generally increase grass production, they are known to remove younger woody species from East African savannas (Afolayan, 1978; Norton-Griffiths, 1979; Dublin, Sinclair & McGlade, 1990).

The increase in mean cover of herbaceous plants with fire frequency could be partly attributed to grasses. A notable example was the dominance of *Themeda triandra*, a fire resistant species in the Crater area (second most frequently burnt area) thus reinforcing the argument that frequent burning favors mainly grasses. This is also consistent with the findings of Michael *et al*., (2007) who observed that, grasses are the most important floral element of most savannas, generally comprising a major proportion of the herbaceous vegetative cover. Fire frequency, soil moisture, and site characteristics all influence the effect of fire on grasses (Michael *et al*., 2007).

The amount of protection a given grass species provides to its tiller primordial determines some of the mortality and growth differences found among different grass species in response to fire (Frost & Robertson, 1987). For example, Michael *et al*., (2002) observed that, species that exhibit a delay in tiller elevation well after the rains have begun are much less vulnerable to fire than species that elevate tillers soon after the first rain. As a result, species such as *Panicum maximum* Jacq. can be eliminated by ‘frequent fires, while *Themeda triandra* Forssk. can only occur in areas that receive frequent burning (Michael *et al.*, 2002).

Percentage cover of shrubs and trees decreased with fire frequency. This implies that in this study, the cover of woody species (shrubs and trees) was generally greater in areas less frequently burnt. This is consistent with the findings of Booysen & Tainton (1984) who postulated that the generally accepted trend of the structural response of vegetation to fire frequency is that the more frequent the fire, the greater the herbaceous component, and the less frequent the fire, the more abundant the woody component. Waldrop *et al*., (1992) & Peterson (1998) also observed that fires damage or kill woody plants, reduce woody plant cover, and reduce woody species frequencies.

5.4 Effects of fire frequency on species composition

We encountered 109 vascular plant species. Out of this total, 94 were herbaceous plant species, 10 were shrubs while 05 were tree species (Table 20).
All the study sites were therefore dominated by herbaceous plants. The number of shrubs and trees (woody species) generally decreased with fire frequency (Table 20).

My study is consistent with the findings of Michael *et al* (2007) who observed that, at high fire frequency a significantly lower number of woody species were found. The data therefore imply a negative impact of high fire frequency on the occurrence of woody species in QENP.

Okello *et al*., (2007) argued that fire may not only reduce densities of living trees, but also reduce their ability to recruit via seeds after a fire. However, the effects of fire on seeds are probably species-specific and related to the duration of fire exposure (Sabiiti & Wein, 1987; Mucunguzi & Oryem-Origa, 1996; Radford *et al.* 2001).

Repeated burning can have profound impacts on savanna structure (Bond & Archibald, 2003; Nangendo *et al*., 2005; Govender *et al*., 2006). Okello *et al.,* (2007) also argued that even a single fire had the potential to reverse woody encroachment in savanna ecosystems when large herbivores are present. He further suggested that fire may also reduce recruitment, although even shallow burial appears to provide considerable protection from fire in this fire-adapted species.

Fire may offer a useful option to open the woody layer, but perhaps mostly in the presence of large native browsers (Mills & Fey, 2005)

The higher the fire frequency, the more difficult it is then for these woody species to emerge and to escape from the fire (Michael *et al.*, 2007). Therefore, large trees are scarce under high fire frequency and young trees very rare or non-existent.

In addition to these findings of a negative impact of high fire frequency on the occurrence of larger trees, significantly fewer woody species numbers were recorded at high fire frequency (Michael *et al.,* 2007).

Reduced fire frequency may also permit woody species to increase. Conversely, increasing the average fire frequency throughout grasslands may have detrimental effects on those grass species that are just able to persist under the current fire regime (Michael *et al.*, 2002).

However, fires may not be the only reason why woody species were uncommon. Grazing, infiltration properties of soils, water availability and churning of soils could also explain low abundance of woody species (Frost *et al.,* 1986; Miller & Donahue, 1990; Hillel, 1998).

Schloeder (1999) found that species distributions throughout the savanna grasslands were primarily driven by climate, geomorphology and substrate.

Michael *et al*., (2002) argued that fire frequency was not the driving force behind the distribution pattern of species in the perennial grasslands of south-west Ethiopia *per se.* Rather, fire plays a role in the distribution of species in these grasslands in the sense that species that can tolerate the current fire regime are able to persist. Although the pattern in species composition does not appear to be driven by fire, increasing or decreasing the frequency of fires could potentially have marked effects on these grasslands (Michael *et al*., 2002).

Owen-Smith (1988) noted that, in African savannas, mega-herbivores, and especially elephants, have a disproportionate effect on the structure of woody vegetation, although other species (Yeaton, 1988) also play an important role. Many studies have implicated elephants in observed declines in woody vegetation (Pellew, 1983). Conversely, increases in the density of woody vegetation have been linked to declining numbers of elephants in many areas in Africa (Lock, 1993; Leuthold, 1996).

Fire and elephants, have played an important role in shaping savanna ecosystems in Africa for many hundreds of thousands of years (Laws, 1970; Dublin, Sinclair & McGlade, 1990).

Viljoen (1988) observed that the numbers of trees in the Kruger National Park had declined over the past 50 years, putatively because of frequent fire and large numbers of elephant and that the vegetation had become structurally more homogeneous due to regular prescribed burning at fixed intervals over large areas (van Wilgen *et al*., 1998a,b).

Michael *et al*., (2002) argued that, timing of fires can have an effect on species composition. Timing of fires can potentially influence individual grass species. A determination of which species would be affected would require additional research, with attention paid to specific-responses to rainfall events (i.e. quick or delayed growth after rainfall) (Michael *et al*., 2002).

It appears that, regardless of timing or intensity, fire frequency has a direct impact on the composition of the species in QENP and favors species that are adapted to fire.

5.5 Conclusions

The following conclusions can be drawn from this study.

Fire frequency was strongly correlated with plant species richness and vegetation cover in QENP.

Fire frequency considerably affected plant species diversity in QENP.

Fire frequency affected plant species composition in QENP. The number of woody species (shrubs and trees) generally decreased with fire frequency.

5.6 Recommendations

An investigation of the possible changes in the vegetation of QENP and the factors influencing such changes besides fire and grazing is necessary in order to arrive at conclusive results which can influence policy formulation on the management of the park.

There is need to reinstate the permanent plots that were established in QENP for studying the effects of fire and grazing on various vegetation parameters in the park or establish new plots for the same.

Before any controlled burning schemes are implemented in these areas, it would be prudent to construct fire-breaks. Many of the existing tracks would serve the purpose if they were widened and regularly treated to prevent the encroachment of grass.

Continuous monitoring of the effects of the current controlled burning programme on vegetation is necessary. This is important because, if the effects are largely detrimental, it will be necessary to change the current fire regime.

The need to expand the study by combining the effects of fire with those of other factors such as grazing, soils, rainfall regimes and topography among others cannot be over emphasised.

It is necessary to identify sections of the park that need special attention in a controlled burning scheme as well as those that are at a greater risk of interference from unwanted fires in order to plan effective monitoring in such sections.

Most wildlife management areas in Africa have a shortage of resources. Operating a fire management programme is both costly and time-consuming. One way of solving the problem of the need for an effective fire programme and the limited resources available for operating is to adopt a land-use practice which ensures an optimal fire regime in the area.

REFERENCES

Afolayan, T.A. (1978). Savanna burning in Kainji Lake National Park, Nigeria. *E.Afr.Wildl. J.* **16**, 245-255.

Augustine, D. J. and Frank, D. A. (2001). Effects of migratory grazers on spatial heterogeneity of soil nitrogen properties in a grassland ecosystem. *Ecology* **82**, 3149-3162.

Axelrod, A.N. & Irving, F.D. (1978). Some effects of prescribed fire at Cedar Creek Natural History Area. J Minnesota *Acad Sci* **44**, 9–11.

Bond, W. & Van Wilgen, B. (1996). Fire and Plants. Population and Community Biology Series **14**, Chapman & Hall, London.

Bond, W. (1997). Fire. In: Vegetation of Southern Africa (Eds R. Cowling, D. Richardson and S. Pierce). Cambridge University Press, Cambridge.

Bond, W.J. & Archibald, S. (2003). Confronting complexity: fire policy choices in South African savanna parks. *Int. J. Wildland Fire* **12,** 381–389.

Booysen, P. & Tainton, N. (1984). Ecological Effects of Fire in South African Ecosystems. *Ecological Studies* **48**, Springer-Verlag, Berlin.

Brewer, J.S. (1998). Patterns of plant species richness in a wet slash-pine (Pinus elliottii) savanna. *Bull Torrey Bot Club* **125**, 216–224.

Brockway, D.G, Lewis, C.E. (1997). Long-term effects of dormant-season prescribed fire on plant community diversity, structure, and productivity in a longleaf pine wiregrass ecosystem. *Forest Ecol Manage* **96**, 167–183.

Brown, J.K. and Smith, J.K. (2000). Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-vol. 2. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 257 p.

Carlo, R. L., Paolo, V. & Mapesa, M. (1996). Effects of 20 years of grazing exclusion in an area of the Queen Elizabeth National Park, Uganda. *Afr. J. Ecol.* **34**, 333-341.

Chapman, G.P. (1976). Desertified grasslands: Their biology and management. 1^{st} edt.

Sandiego, London, Newyork.

Collins, S. L. (1992). Fire frequency and community heterogeneity in tallgrass prairie vegetation. *Ecology* **73**, 2001-2006.

Connell, J.H. (1978). Diversity in tropical rain forests and coral reefs. *Science* **199**, 1302-1310.

Conrad, C.E and Charles E. Poulton. (1996). Effect of a Wild fire on Idaho Fescue and Bluebunch Wheatgrass. Journal of Range Management (19): 138-141.

Coppedge, B. R. & Shaw, J. H. (1998). Bison grazing patterns on seasonally burned tallgrass prairie. *J. Range Manage*. **51**, 258-264.

David, W. P. & Peter, B. (2008). Fire frequency and tree canopy structure influence plant species diversity in a forest-grassland ecotone. *Plant Ecol* **194**, 5–16.

DeBano, L., Neary, D. & Folliott, P. (1998). *Fire's Effects on Ecosystems.*Wiley & Sons. Dissertation. University of Minnesota, Saint Paul, Minnesota.

Dublin, H.T., Sinclair, A.R.E. & McGlade, J. (1990). Elephants and fire as causes of multiple stable states in the Serengeti-Mara woodlands. *J. Anim. Ecol.* **59**, 1147-1164.

Edroma, E.L. (1981a). The role of grazing in maintaining high species composition in *Imperata* grassland in Rwenzori National Park, Uganda. *Afr .J. Ecol.* **19,** 215-223.

Edroma, E.L. (1981b). Some effect of grazing on the productivity of grassland in Rwenzori National Park, Uganda. *Afr.J.Ecol.* **19,** 313-326.

Edroma, E.L. (1984). Effects of burning and grazing on productivity and number of plants in Queen Elizabeth National Park, Uganda. *Afr. J. Ecol.* **22**, 165-174.

Eltrigham, S.K. (1976). The frequency and extent of uncontrolled grass fires in the Rwenzori National Park, Uganda. E. *Afr. Wildl .J.* **14,** 215-222.

Faber-Langendoen D. & Davis, M.A. (1995). Effects of fire frequency on tree canopy cover at Allison Savanna, eastcentral Minnesota, USA. *Nat Areas J* **15**, 319–328

Frost, P.G.H. & Robertson, F. (1987). The ecological effects of fire in savannas. In: *Determinants of Tropical Savannas* (Ed. B.H. Walker). IRL Press, Oxford.

Fuhlendorf, S. D. and Smeins, F. E. (1999). Scaling effects of grazing in a semi-arid grassland. *J. Veg. Sci.* **10**, 731-738.

Gillon, D. (1983). The fire problem in tropical savannas. In: *Tropical Savannas* (Ed. F. Bourliere). Elsevier, Amsterdam.

Govender, N., Trollope, W.S.W. & Van Wilgen, B.W. (2006). The effect of fire season, fire frequency, rainfall and management on fire intensity in savanna vegetation in South Africa. *J. Appl. Ecol.* **43,** 748–758.

Grime, J.P. (1973). Control of species density in herbaceous vegetation. *J Ecol* **1**:151–167.

Heady, H.F. & Heady, E.B. (1982). Fire on rangeland. In: *Range and Wildlife Management in the Tropics* (Ed. W.A.J. Payne). Longman, London.

Higgins, S., Bond, W. & Trollope, W. (2000). Fire, resprouting and variability: a recipe for grass-tree coexistence in savanna. *J. Ecol.* **88**, 213–229.

Hillel, D. (1998). Nature and behavior of clay. In: *Environmental Soil Physics* (Ed. D. Hillel). Academic Press, San Diego.

Hopkins, B. (1965a). Observation on the savanna burning in the Olokemeji forest reserve, Nigeria. *J. appl. Ecol.* **2,** 367-381.

Horn, H.S. (1974). The ecology of secondary succession. *Ann. Rev. Ecol. Syst.* **5**, 25-37.

Huston, M.A. (1994). Biological diversity: the coexistence of species on changing landscapes. Cambridge University Press, Cambridge.

Jaksic-Born, C. (2004). Approach to the ecological context and the anthropogenic factors related to the fire regime in QENP.

Jennifer, S., Daniel, K. & Barbara, D. (2001). Short-term effects of fire frequency on vegetation composition and biomass in mixed prairie in south-western Manitoba. *Plant Ecology* **155**, 157-167.

John, R. T. (1989). Effects of fire frequency on oak savanna in east-central Minnesota. *Bulletin of the Torrey Botanical Club* **116**, 134-144.

Kucera, C.L. & Koelling, M. (1964). The influence of fire on composition of central Missouri prairie. *Amer. Midl. Nat.* **72**, 142-147.

Laws, R.M. (1970). Elephants as agents of habitat and landscape change in East Africa. *Oikos* **21,** 1–15.

Leach, M.K. & Givnish, T.J. (1999). Gradients in the composition, structure, and diversity of remnant oak savannas in southern Wisconsin. *Ecol Monogr* **69**,353–374.

Lemon, P.C. (1968). Fire and wildlife grazing on an African plateau. *Proc. Tall Timbers Fire Ecol. Conf.* **8,** 71-88.

Leuthold, W. (1996). Recovery of woody vegetation in Tsavo National Park, Kenya. 1970–94. *Afr. J. Ecol.* **34,** 101–112.

Lewis, C.E. & Harshbarger, T.J. (1976). Shrub and herbaceous vegetation after 20 years of prescribed burning in the South Carolina coastal plain. *J Range Manage* **29**, 13–18.

Lind, E.M. & Tallantire, A.C. (1971). *Some Common Flowering Plants of Uganda.* Oxford University Press, Nairobi.

Lock, J.M. (1977). The vegetation of the Rwenzori National Park, Uganda. *Bot. Jb. Syst.* **98,** 372-448.

Lock, J.M. (1977b). Preliminary results from fire and Elephant exclusion plots in Kabalega National Park, Uganda. *East African Wildlife Journal* **15**, 229-232.

Lock, J.M. (1988). Vegetation Studies in the Queen Elizabeth National Park, Uganda. *Uganda Institute of Ecology,* Uganda.

Lock, J.M. (1993). Vegetation change in Queen Elizabeth National Park, Uganda: 1970-88. *Afr. J. Ecol,* **31**, 106-117.

McPherson, G.R. (1997). Ecology and management of North American Savannas. *University of Arizona Press*, Tucson, Arizona.

Meyer, K.M., Ward, D., Moustakas, A. & Wiegand, K. (2005). Big is not better: small Acacia mellifera shrubs are more vital after fire. *Afr. J. Ecol.* **43,** 131–136.

Michael, H., Jan, S., Budzanani, T. & Michael, M.H. (2007). The relevance of fire frequency for the floodplain vegetation of the Okavango Delta, Botswana. *Afr. J. Ecol*, **46**, 350–358.

Michael, J. J. & Catherine, A. S. (2002). Fire frequency and species associations in perennial grasslands of south-west Ethiopia. *Afr. J. Ecol.,* **40,** 1-9.

Miller, R.W. & Donahue, R.L. (1990). Soil water. In: *Soils: an Introduction to Soils and Plant Growth* (Eds. R.W. Miller, R.L. Donahue and J.U. Miller). Prentice Hall, Englewood Cliffs, NJ.

Mills, A.J. & Fey, M.V. (2005). Interactive response of herbivores, soils and vegetation to annual burning in a South African savanna. *Austral Ecol.* **30,** 435–444.

Mooney, H.A. (1981). Fire regimes and ecosystem properties. USDA, *Forest Serv., Gen. Tech. Rep*. WO-26. 594 p.

Morgan, J. (1999). Defining grassland fire events and the response of perennial plants to annual fire in temperate grasslands of south-eastern Australia. *Plant Ecol.,* **144**, 127–144.

Mucunguzi, P. & Oryem-origa, H. (1996). Effects of heat and fire on the germination of Acacia *sieberiana* DC and *Acacia gerrardii* Benth in Uganda. *J. Trop. Ecol.* **12,** 1–10.

Nangendo, G., Stein, A., Ter Steege, H. & Bongers, F. (2005). Changes in woody plant composition of three vegetation types exposed to a similar fire regime for over 46 years. *For. Ecol. Manage.* **217,** 351–364.

National Wildfire Coordinating Group. (2001). Fire Effects Guide. Available online:

http://www.nwcg.gov/pms/RxFire/FEG.pdf. 313 p.

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

Nuzzo, V.A. (1986). Extent and status of Midwest oak savanna: Presettlement and 1985. *Natural Areas J*. **6**, 6-36.

Okello, B. D., Young, T. P., Riginos ,C., Kelly, D. & O'Connor, T. G. (2007). Short-term survival and long-term mortality of Acacia drepanolobium after a controlled burn. *Afr. J. Ecol.,* **46,** 395–401.

Owen-Smith, R.N. (1988). *Megaherbivores: the Influence of Very Large Body Size on Ecology.* Cambridge University Press, Cambridge.

Pellew, R.A.P. (1983). The impacts of elephant, giraffe and fire upon the *Acacia tortilis* woodlands of the Serengeti. *Afr. J. Ecol.* **21,** 41–74.

Peterson, D.W. (1998). *Fire effects on oak savanna and woodland vegetation in Minnesota*.

Peterson, D.W. & Reich, P.B. (2001). Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. *Ecol Appl* **11**, 914–927.

Pratt, D.J. & Gwynne, M.D. (1977). *Rangeland Management and Ecology in East Africa.* Hodder & Stoughton, London.

Radford, I..J., Nicholas, D.M. & Brown, J.R. (2001). Impact of prescribed burning on Acacia nilotica seed banks and seedlings in the Astrebla grasslands of northern Australia. *J. Arid Environ.* **49,** 795–807.

Roques, K. G., O'connor, T.G. & Watkinson, A.R.R. (2001). Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. *J. Appl. Ecol.* **38,** 268–280.

Roxburgh, S.H., Shea, K. & Wilson, J.B. (2004). The intermediate disturbance hypothesis: patch dynamics and mechanisms of species coexistence. *Ecology* **85**, 359–371.

Russell-Smith, J., Ryan, P., Klessa, D., Waight, G. & Harwood, R. (1998). Fire regimes, fire-sensitive vegetation and fire management of the sandstone Arnhem Plateau, monsoonal northern Australia. *J. Appl. Ecol.* **35**, 829–846.

Sabiiti, E.N. & Wein, R.W. (1987). Fire and acacia seeds: a hypothesis of colonization success. *J. Ecol.* **75,** 937–946.

Schloerder, C.A. (1999). *Investigation of Savanna Vegetation determinant Relationships using Spatial Statistics.* PhD Dissertation, Utah State University, Logan.

Scholes, R.J. & Archer, S.R. (1997). Tree-grass interactions in savannas. *Ann Rev Ecol Syst* **28**, 517–544.

Shafi, M.I. & Yarranton, G.A. (1973). Diversity, floristic richness, and species evenness during a secondary (post-fire) succession. *Ecology* **54**, 897-902.

Shesh, K. K. (2004). Effects of Forest Fire Protection on Plant Diversity in a Tropical Deciduous Dipterocarp-Oak Forest, Thailand. *Proceedings of the Second International Symposium on Fire Economics, Planning, and Policy: A Global View.*

Stein, R.M., Per, W. &. Kapela, E.B. (1990). The influence of man-made fires on large herbivores in lake Birungi area in northern Tanzania. *Afr. J. Ecol,* **28**, 35-43.

Steinauer, E. M. & Collins, S. L. (2001). Feedback loops in ecological hierarchies following urine deposition in tallgrass prairie. *Ecology* **82**, 1319-1329.

Stohlgren, T.J., Falkner, M.B. & Schell, L.D. (1995). A Modified-Whittaker nested vegetation sampling method. *Vegetatio* **117**, 113-121.

Sylvia, P., Mary, N & Kåre, A.L. (2003). 115 Ugandan Grasses. Department of Botany, Makerere University.

Tester, J.R. (1989). Effects of fire frequency on oak savanna in east-central Minnesota. *Bull Torrey Bot Club* **116**,134–144.

UWA (2005). *General management plant for Queen Elizabeth Protected Area*. QEPA, Uganda.

Uys, G., Bond, W. & Everson, T. (2004). The effect of different fire regimes on plant diversity in southern African grasslands. *Biol. Conserv*. **118**, 489–499.

Van Wilgen, B.W., Biggs, H.C. & Potgieter, A.L.F. (1998). Fire management and research in the Kruger National Park, with suggestion on the detection of thresholds of potential concern. *Koedoe* **41,** 69–87.

Veen (Ciska), G. F., John, M. B., Melinda, D. S. & Scott, L.C. (2008). Influence of grazing and fire frequency on small-scale plant community structure and resource variability in native tallgrass prairie. *Oikos* **117**, 859-866.

Viljoen, A.J. (1988). Long-term changes in the tree component of the vegetation in the Kruger National Park. In: *Long-Term Data Series Relating to Southern Africa's Renewable Natural Resources* (Eds I. A. W. MacDonald and R. J. M. Crawford). South African National Scientific Programmes Report 157, FRD, CSIR, Pretoria.

Waldrop, T.A., White, D.L. & Jones, S.M. (1992). Fire regimes for pine-grassland communities in the southeastern United States. *Forest Ecol Manage* **47**, 195–210.

Walker, J. & Peet, R.K. (1983). Composition and species diversity of pine-wiregrass savannas of the Green Swamp, North Carolina. *Vegetatio* **55**, 163-179.

Watson, P. & Wardell-Johnson, G. (2004). Fire frequency and time-since-fire effects on the open-forest and woodland of Girraween National Park, south-east Queensland, Australia. *Aust. J. Ecol.* **29**, 225–236.

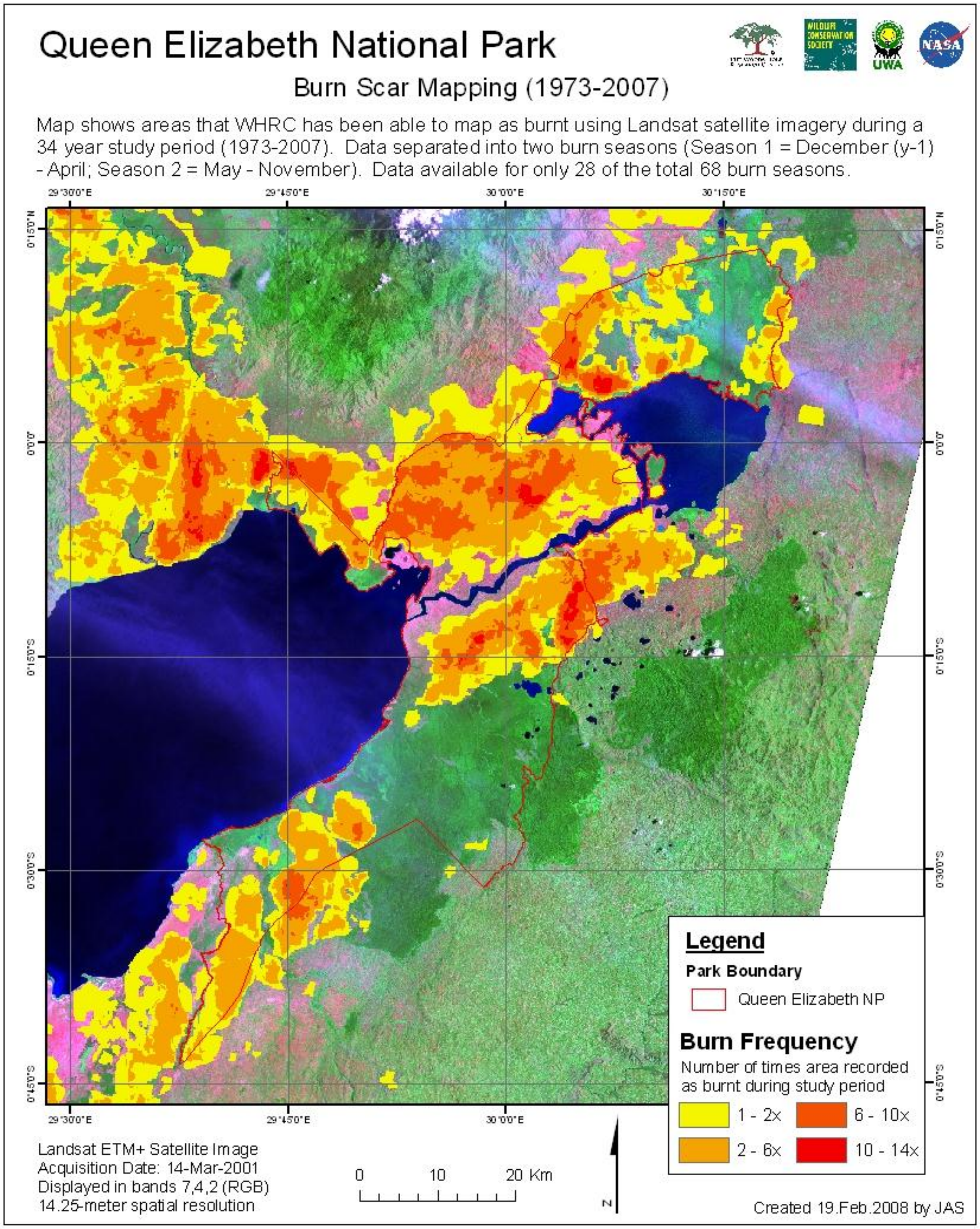
Wein, R.W. & Edroma, E.L. (1986). *Fire in Tropical Savannas with Emphasis on East Africa*: A Bibliography. International Development Research Centre Canada, Ottawa, ON.

Whelan, R.J. (1995). *The Ecology of Fire*. Cambridge University Press, Cambridge, U.K.

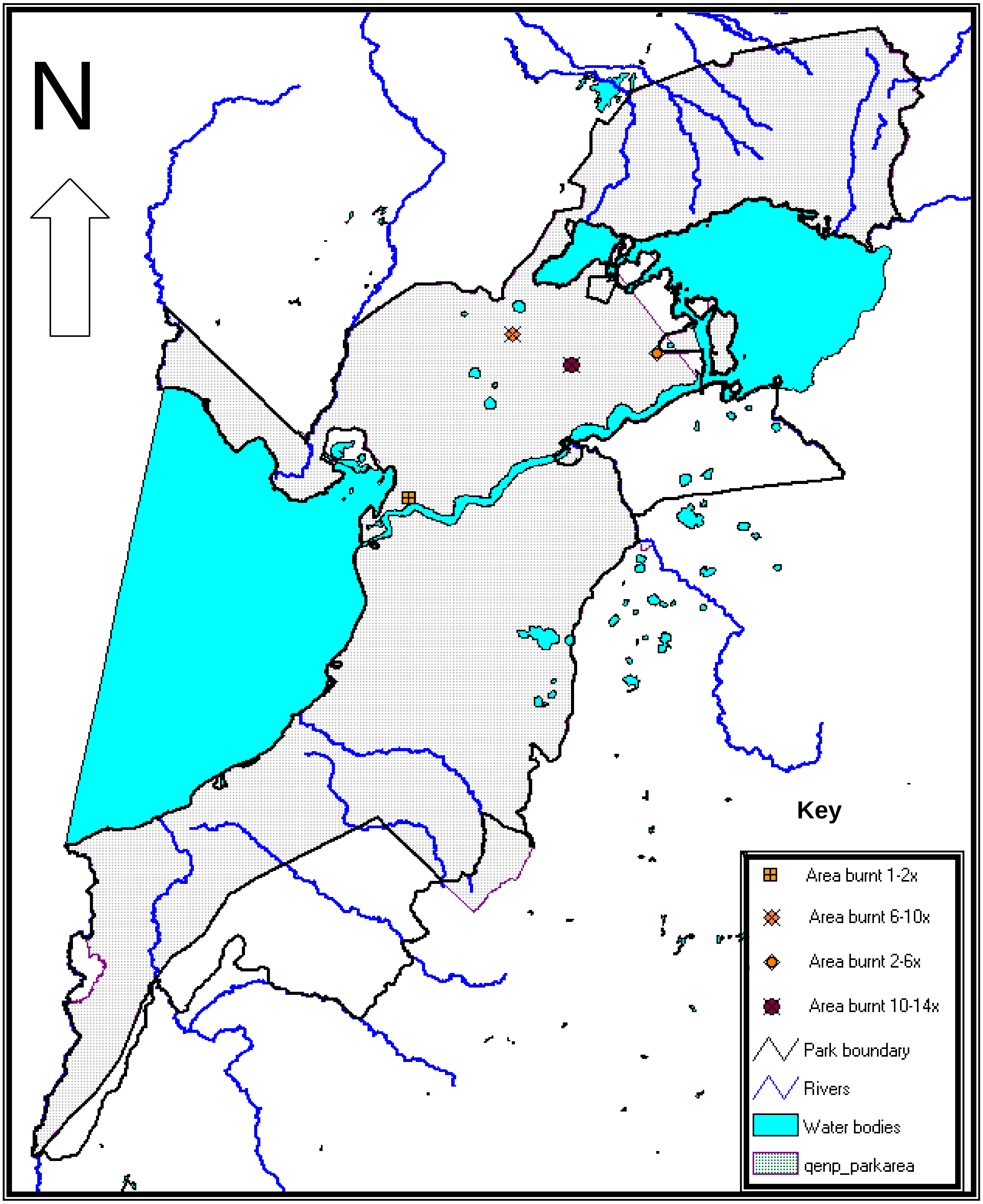
Whittaker, R.H. (1975). *Communities and ecosystems*, 2nd edn. McMillan Publishing Company, New York.

Yeaton, R.I. (1988). Porcupines, fires and the dynamics of the tree layer of the *Burkea africana* savanna. *J. Ecol.* **76,** 1017–1029.

APPENDIX I. BURN SCAR MAP OF THE QENP



APPENDIX II. MAP OF QENP SHOWING THE STUDY SITES



APPENDIX III. DATA SHEET: TABULATION OF RAW DATA FOR SAMPLING HERBACEOUS PLANTS IN QENP

Date………………… Observers…………………………………………………………..

Habitat/stratum.………………………… Locality………………………………………..

Plot identification No...…………………………. Plot size…………………………….....

Sub plot (Quadrat) size……………………………

Treatment………………………………………….

Coordinates………………………………………………………………………………

Sub plot (Quadrat) No:	**Species:**	**No. of individuals**	**% Cover**	**Notes**

APPENDIX IV. DATA SHEET: TABULATION OF RAW DATA FOR SAMPLING SHRUBS IN QENP

Date………………… Observers…………………………………………………………..

Habitat/stratum………………………… Locality………………………………………..

Plot identification No..…………………………. Plot size…………………………….....

Sub plot (Quadrat) size……………………………

Treatment………………………………………….

Coordinates………………………………………………………………………………

Sub plot (Quadrat) No:	**Species**	**% Cover**	**Notes**

APPENDIX V. DATA SHEET: TABULATION OF RAW DATA FOR SAMPLING TREES IN QENP

Date………………… Observers…………………………………………………………..

Habitat/stratum………………………… Locality………………………………………..

Plot size……………………………...............................

Treatment……………………………………………….

Coordinates………………………………………………………………………………

Plot No	**Species**	**% Cover**	**Notes**

APPENDIX VI: LIST OF HERBACEOUS PLANTS AT THE STUDY SITES IN QENP

S/NO	SPECIES
1	*Aristida adscensionsis* L
2	*Bothriochloa insculpta*
3	*Chloris gayana* Kunth
4	*Chloris pycnothrix* Trin.
5	*Chloris roxburghiana* Schult
6	*Cynodon dactylon (L.)* Pers.
7	*Dactyloctenium aegyptium* (L.) Willd.
8	*Digitaria abyssinica* (A. Rich.) Stapf
1	*Dinebra retroflexa* (Vahl) Panzer
10	*Enteropogon macrostachys* (A Rich) Benth
11	*Eragrostis exasperata* Peter
12	*Heteropogon contortus* (L.) Roem. & Schult.
13	*Hyparrhenia rufa* (Nees) Stapf
14	*Hyperthelia dissoluta* (Steud.) W.D. Clayton
15	*Microchloa kunthii* Desv.
16	*Panicum infestum* Peters
17	*Panicum maximum* Jacq.
18	*Panicum repens*
11	*Panicum subalbidum* Kunth
20	*Setaria homonyma* (Steud.) Chiov
21	*Sporobolus festivus* A. Rich
22	*Sporobolus pyramidalis* P. Beauv.
23	*Themeda triandra* Forssk.
24	*Tragus berteronianus* Schult
25	*Urochloa panicoides* Beav
26	*Cyperus dubius* Rottb. Subsp. Dubius
27	*Cyperus teneriffae* Poir
28	*Cyperus articulatus* L
21	*Alternanthera* sp.
30	*Abutilon mauritianum* (Jacq.) Medic.
31	*Achyranthes aspera* L.
32	*Alysicarpus rugosus* (Willd) DC. Subsp. Rugosus
33	*Alysicarpus vaginalis* (L.) DC
34	*Anthericum subpetiolatum* (Bak.) Kativu
35	*Asparagus africanus* Lam.
36	*Asystasia gangetica* (L.) T. Anderson
37	*Asystasia mysorensis* (Roth) T. Anderson
38	*Bidens pilosa* L
31	*Blepharis maderaspatensis* (L.) Roth
40	*Boerhavia diffusa* L

41 *Caralluma dummeri* N.E. Br
42 *Cassia absus* L
43 *Cassia mimosoides* L.
44 *Cissus quadrangularis* L.
45 *Cissus rotundifolia* (Forsk.) Vahl.
46 *Commelina benghalensis* L
47 *Corbichonia decumbens* (Forsk.) Exell
48 *Corchoris trilocularis* L
41 *Craterostigma plantangineum* Hochst
50 *Crotalaria spinosa* Benth.
51 *Cucumis figarei* Naud
52 *Cyphostemma adenocayle* (A. Rich.) Wild. & Drummond
53 *Dyschoriste nagchana* (Nees) Bennet
54 *Euphorbia glomerifera* (mill sp.) Wheeler
55 *Evolvolus alsinoides* (L.) L.
56 *Evolvolus nummularius* (L.) L.
57 *Fuerstia africana* Th. C.E. Fries
58 *Gomphrena celosioides* Mart
51 *Hibiscus aponeurus* Sprague & Hutch.
60 *Hoslundia opposita* Vahl.
61 *Hygrophila auriculata* (Schumach.) Heine
62 *Indigofera circinella* Bak. f.
63 *Indigofera vohemarensis* Baill.
64 *Jasminium eminnii* Gilg
65 *Jasminium floribunaum* Fresen.
66 *Leucas urticifolia*
67 *Neptunia* sp
68 *Ocimum gratissimum* L.
61 *Ocimum suave* Willd.
70 *Pavonia patens* (Andr.) Chiov.
71 *Pergularia daemia* (Forsk.) Chiov.
72 *Phyllanthus maderaspatensis* L
73 *Polygala erioptera* DC.
74 *Polygala sphenoptera* Fresen
75 *Portulaca quadrifida* L.
76 *Psilotrichum axilliflorum* Suesseng
77 *Pupalia luppacea* (L.) Juss
78 *Rhynchosia minima* (L.) DC. Varnuda (DC.) Kuntze
71 *Rullia patula* Jacq.
80 *Sansevieria* sp.
81 *Schkuria pinnata* (Lam.) Ktze
82 *Sida ovata* Forssk
83 *Sida rhombifolia* L
84 *Solanum incanum* L.
85 *Talinum portulacifolium* (Forssk) Schw.
86 *Tephrosia pumila* (Lam.) Pers

87	*Teramnus repens* Bak.f. subsp. Repens
88	*Tribulus terrestris* L.
89	*Vernonia cinerea* (L.) Less
90	*Vigna kirkii* (Bak.) Gillett
91	*Vigna luteola* (Jacq.) Benth
92	*Zehneria pallidinerva* (Harms) C. Jeffrey
93	*Zornia setosa* Bak f. Subsp. Obovata (Bak. f) Leon & Milne - Redh.
94	*Wissudula amplissima* (L) R.E.

APPENDIX VII: LIST OF SHRUBS AT THE STUDY SITES IN QENP

S/NO	SPECIES
1	*Acalypha bipartita* Muell. Arg.
2	*Azima tetracantha* Lam.
3	*Balanites aegyptica* (L.) Del.
4	*Boscia* sp.
5	*Capparis tomentosa* Lam.
6	*Flueggea virosa* (Willd.) Voigt
7	*Grewia similis* K. Schum.
8	*Lantana camara* L.
1	*Maerua triphylla* A. Rich.
10	*Tarenna graveolens* (S. Moore) Brem.

APPENDIX VIII: LIST OF TREE SPECIES AT THE STUDY SITES IN QENP

S/NO	SPECIES
1	*Acacia gerrardii* Benth.
2	*Acacia sieberiana* DC.
3	*Euphorbia candelabrum* Kotschy
4	*Dichrostachys cinerea* (L.) Wight & Arn.
5	*Turraea robusta* Gürke