THE EFFECTS OF FIRES ON PLANTS AND WILDLIFE SPECIES DIVERSITY AND SOIL PHYSICAL AND CHEMICAL PROPERTIES AT ABERDARE RANGES, KENYA

WANGARI FAITH NJERI

BSC HONS (2011)

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF MASTER OF SCIENCE IN BIOLOGY OF CONSERVATION SCHOOL OF BIOLOGICAL SCIENCES, THE UNIVERSITY OF NAIROBI.

2016

DECLARATION

This thesis is my original work and has not been presented for a degree in any other University

Wangari Faith Njeri

Signature... Date..

This thesis has been submitted for examination with our approval as the University Supervisors

Dr. J. M. Githaiga

School of Biological Sciences

University of Nairobi

Signature... Date..

Mr. Aggrey K. Mwala
Department Of Land Resource Management & Agricultural Technology
University Of Nairobi

Signature... Date..

ACKNOWLEDGEMENT

I would like to express my sincere gratitude to Dr. Githaiga and Mr. Mwala, my supervisors, for the great support and guidance they accorded me throughout this research project, without which this research project would not have been successful. I would also like to thank Kenya Forest Service staff for allowing me to carry out my research work at the Aberdares forest, offering me accommodation at the station and for providing security personnel to guard me during my visits to the forest.

I would also like to thank Masinde Muliro University of Science and Technology, for enabling me pursue my Masters degree by awarding me a full scholarship for the two years of study.

I am very grateful to my dear mother and my entire family for the great support they offered me, both financially and morally. I say thank you very much for your support and pressure to finish my thesis.

Last and most important, I would like to thank the Almighty God, for this far that He has brought me.

DEDICATION

This work is dedicated to my dear mother Ms. Alice Wangari Mungai.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDICES

LIST OF ABBREVIATIONS

CEC - Cation Exchange Capacity

cm - Centimeters

DBH - Diameter at Breast Height

g - Grams

GDP - Gross Domestic Product

Ha - Hectares

IBA - Important Bird Area

I.V - Important Value

K - Potassium

KFS - Kenya Forest Service

KWS - Kenya Wildlife Service

LSD - Least Significant Difference

mm - Millimeter

ml - Milliliter

N - Nitrogen

OC - Organic carbon

OM - Organic Matter

P - Phosphorous

PCQ - Point Centered Quarter

P-P - Point to Plant

ppm - Parts Per Million

r.d - Relative density

r.D -Relative dominance

STMS - Scarlet Tufted Malachite Sunbird

ABSTRACT

More than 220,000 forest fires occur every year around the world resulting in more than 6 million hectares of forests burnt. Fires are a frequent phenomenon in Kenyan forests especially during the dry seasons. In the Aberdare ranges fires occur every year during the dry season as the vegetation is usually dry and there are strong winds which make it easy for any fire started spread very fast. Fire can initiate changes which affect the composition, structure, and pattern of vegetation on the landscape. These changes also affect animal habitation and utilization of the forests. It can also change the soil properties which in turn can affect the vegetation growing there.

This study was conducted to determine the effects of fires on plants, animals, birds and soil physical and chemical properties at the Aberdare Ranges forest. Data was collected on five sites that experienced fires in 2002, 2009, 2012, 2013, and 2014. In every site, data was collected from both burnt and un-burnt areas for comparison. For the woody vegetation sampling, Point Centered Quarter method was used and for herbaceous vegetation sampling, quadrat method was used. Foot count was done for animal census and point count for birds. In every site, soil was sampled at three depths from three points chosen randomly in every site and the soil samples were taken to soil laboratory for analysis.

The data showed that woody species diversity did not differ significantly between burnt and un-burnt sites for all fire episodes at all sites. This could be because most fires do not totally burn the trees but just affect the Diameter at Breast Height, height and the canopy of the trees. There was no significant difference in woody species diversity ($F_{1, 8}$ =0.001; P>0.05). However, the herbaceous vegetation in burnt sites had significantly higher species diversity than the un-burnt sites in the areas that experienced fire prior to 2014. The mean number of species in all burnt site was 34.6±5.215 and the mean number of species in all un-burnt sites was 24.8±5.215. Fire had triggered the regeneration of the herbaceous plants. There was a significant difference between the percentage cover of the burnt sites and the un-burnt sites with the burnt sites having a higher percentage cover ($F_{1,128}$=5.360; P< 0.05).

Fire has immediate negative effect on the population of animals as demonstrated on the site burnt on 2014. No single animal was found on the site one week after the fire when data was collected.

The site that was burnt in 2013 also had fewer animals compared to the un-burnt site. All the other sites which had been burnt prior to 2013 had more animals compared to the un-burnt sites. The animal diversity was directly proportional to the vegetation density caused by the vegetation regeneration due to fires. The areas with more vegetation had more animals.

Fire had negative effect on birds and all the burnt sites had fewer birds than the un-burnt sites. The site burnt in 2014 had the least number of birds. The number of birds in the sites burnt before 2014 had increased although they were still fewer than the population in the un-burnt sites.

Effects of fires were evident in the upper layer (0-15cm) of the soil for all the soil properties under study. There was no significant difference at 15-30cm and 30-45cm in soil properties between the burnt sites and un-burnt sites. Burning caused increase in pH, potassium, organic carbon and cation exchange capacity. Fire was found to decrease nitrogen and phosphorus content.

The study demonstrated that fires lead to an immediate negative effect on vegetation, wildlife and soil chemical properties. Post fire management is important on sites that have recently experienced fires in order to rehabilitate them. Since the reduced herbaceous vegetation makes it easy for the humans to access the forest, this can lead to further destruction of the forest. Authorities responsible for the management of forests should ensure that people are kept out of those sites to allow vegetation to regenerate without interference. Reforestation can also be done on the burnt sites so as to increase vegetation and habitat for the wildlife.

Key words: Animal diversity, Birds, Fire, Plant diversity, Soil properties.

CHAPTER ONE
1.0 Introduction and literature review

1.1 Introduction

Forests are indispensable sources of harvested products and a variety of services which include: provision of food, timber, fuel, genetic materials among others, regulating services such as protection of watershed and carbon storage, cultural services and supporting services (Koziowski, 2002). Forest ecosystem services have been demonstrated to be of great economic value since they provide raw materials for food, fuel and shelter (Constanza *et al.,* 1997). In forest valuation studies, service components like carbon storage or hydrological protection frequently fetch higher values than forest products. Unfortunately forests throughout the world, and especially in the tropics, are threatened by natural-induced disturbances such as wind throw, droughts, floods, disease outbreaks among others and human-induced disturbances such as fires, logging, charcoal burning, forest clearance for farming among others (Koziowski, 2002).

One phenomenon that threatens forest worldwide is forest fires. Forest fire can be defined as the free propagation of uncontrolled fire in forest ecosystems, caused by accidental, natural or intentional causes (Chuvieco, *2009*). Forest fires are a serious hazard and can cause significant damage to ecological, cultural, economic and human resources (Long *et al*., *2001*).

More than 220,000 forest fires occur every year around the world resulting in more than 6 million ha of forests burnt (Gonzalez *et al*., 2005). Many studies have been carried out on assessment and management of forest fires. Countries like Russia, the United States of America and Canada have led the way in forest fire research (Sturtevant *et al*., 2009). They have developed forest fire management systems and forest fire risk forecast systems in time and space. Kenya is also trying to adopt a fire management system through Kenya Forest Service by recording all the fire incidences and stocking every station with fire equipments although the development has been slow.

Knowledge of fire effects has attained increased importance to land managers because fire as a disturbance process is an integral part of ecosystem management. Fire can initiate changes which affect the composition, structure, and pattern of vegetation on the landscape. Disturbance is necessary to maintain a diversity of living things and processes (Botkin, 1990; Morgan *et al*., 1994). Old growth forests are valuable component of biodiversity and one way to assess the effectiveness of forest management is by assessing the proportion of old growth forest currently present in relation to that was historically present (Lesica, 1996). Anthropogenic activities create fire-prone ecosystems in the tropics through the alteration of vegetation cover by logging, burning, and development. The new ecosystems differ substantially in carbon budget, nutrient cycling, fuel and habitat characteristics (Swetnam *et al*., 1999).

As one of Kenya's five main water towers, the forests of the Aberdares play a critical role in supporting the country's economy. They are the main source of water for Nairobi and 55 percent of Kenya's electricity is generated by water flowing from the Aberdares and Mt. Kenya. The Aberdares are the main catchments for Sasumua and Ndakaini dams, which provide most of the water for Nairobi - a city of more than three million people. Nairobi accounts for 60% of Kenya GDP. The energy, water and some raw materials used to drive some economic activities in the city and its environ are derived from the Aberdare ecosystem. Good management of this ecosystem is very important for the national economy (Rhino Ark, 2011). However, fires may interfere with the ability of Aberdares to continue providing its services hence affecting the national economy.

This study was conducted in the western Aberdares at Geta forest zone. Five sites that had been burnt at different time period were identified and data was collected in both burnt sites and unburnt sites. The data was collected during the wet season and dry season. For the woody vegetation, Point Centered Quarter method was used and for herbaceous vegetation, quadrat method was used. Herbivores were counted using foot count and the birds were counted using point count. Soil samples were collected from all the sites and taken to soil laboratory for physical and chemical analysis. The soil properties analyzed include pH, nitrogen, organic matter, potassium, phosphorous, and Cation Exchange Capacity.

1.2 Literature review

Fire and shifts in fire regime are major drivers of ecosystem structure and process and they can lead to decline in species diversity. Forest fires are predicted to increase in many areas due to climate change and change in land management (Reside *et al*., 2012). Global estimates of active vegetation fires indicate that over 70% of these occur in the tropics (Haugaasen *et al.* 2003).

There is an urgent need to undertake more studies on the pattern of fire effects on ecosystem composition, structure, and functions for application in fire and ecosystem management. Understanding fire effects and underlying principles are critical to reduce the risk of uncharacteristic wildfires and for proper use of fire as an effective management tool toward management goals (Chen, 2006).

Forest fires are nearly all associated with human activities and occur mainly during the dry seasons in the dry forests and grasslands (Glogiewicz and Báez, 2001). The risk of fire increases with logging due to reduction of the upper canopy triggering development of scrub and brush undergrowth, which dries more quickly and is easier to ignite than the original understory (Cochrane and Laurance, 2008). Forest fires have both social and ecological effects. Where fires occurs naturally, they can be beneficial for fire dependent species and for invasive grasses that can be used for grazing by cattle (Myers, 2006).

Spatial and temporal interactions between humans and their environment, has resulted in the current distribution of forests in human affected landscapes. One disturbance that is strongly related to human activity is fire. The character of the fire in any forest location is determined by the amount, nature and spatial distribution of ignitable fuel in that forest (Goldammer, 1990). There has been increased dependence on forests by humans for a variety of uses leading to forest fragmentation that further exacerbates future fire events in the landscape (Cochrane, 2003).

Fire risk assessment and research on assessment methods is essential for forest fires management (Cottle, 2007). Assessment methods can be based on the probability of ignition, but this kind of studies ignore the capacity for detection and emergency rescue and damage caused by the fire and this can hinder the government decision making in forest fire prevention and management (Chou, 1992).

Fire severity is the term used to describe the ecological effects of a specific fire. It describes the magnitude of the disturbance and reflects the degree of change in ecosystem components. Fire affects both the aboveground and belowground components of the ecosystem. Severity integrates both the heat pulse above ground and the heat pulse transferred downward into the soil. It reflects the amount of heat that is released by a fire that ultimately affects resources and their functions. It can be used to describe the effects of fire on the soil and water system, ecosystem flora and fauna, the atmosphere, and society (Simard, 1991).

The overall effects of fire on ecosystems are complex, ranging from the reduction or elimination of above-ground biomass to impacts on belowground physical, chemical and microbial mediated processes. Many researchers in the tropics have focused on deforestation and static evaluations of forested areas and seldom have considered the contributing effects of landscape processes and biotic pressures such as forest fires in the loss of biodiversity (Sanchez *et al*., 1999).

1.2.1 Effects of fires on plants

Uncontrolled fire incidents results in a wide diversity of ecological effects which change forest structure and floristic composition. Fire reduce living biomass in tropical forests which can be to the same degree as logging activities. Ecosystems recover from disturbances but frequent and severe fires can offset this balance which can lead to the inability of the ecosystem to recover leading to its collapse together with the loss of their major ecological functions. (Haugaasen *et al.*, 2003; Slik *et al.*, 2008)

The death of the plant tissue is determined by the amount of heat a plant tissue receives. This heat received by a plant is thus determined by the maximum temperature reached and the duration of exposure of the plant tissue. Most plant cells die if heated to temperatures between 50 to 55 °C (Wright and Bailey, 1982).

The plant's structure determines the portion of the above ground woody plant killed. Important aerial crown characteristics include branch density, ratio of live to dead crown material, location of the base of the crown with respect to surface fuels, and total crown size (Brown and Davis, 1973).

Characteristics of the fire, existing vegetation, site conditions, and post fire weather determines the vegetative response to fire which vary from plant to plant. The effects of fire on plants can vary significantly among different fires and on different areas of the same fire. Fire behavior, fire duration, the pattern of fuel consumption, the amount of subsurface heating influence plant's injury and its recovery. The characteristics of the plant species on the site, their susceptibility to fire, and the means by which they recover after fire also influence the post fire response of a plant (Haugaasen *et al.*, 2003).

Fires lead to a large decrease in the living biomass of tropical forest which can be to the same level as logging activities in forests (Slik *et al.*, 2008). Ecosystem composition, structure and functions can be shaped by a fire through selecting fire adapted species and removing other susceptible species.

Fire can be a constructive force where it is responsible for maintaining the health of certain fire-dependent ecosystems. It can be used as an integral component of ecosystems and management due to its ecological roles in mediating and regulating ecosystems. Fire regime, vegetation type, climate, physical environments, and the scale of time and space of assessment influence the effects of a fire on an ecosystem. Studies focusing on temporal and spatial variations of fire effects through long-term experimental monitoring and modeling should be done for more knowledge (Chen, 2006).

Productivity influence the dependence of community structure upon disturbances like fire. If the fire make space and light available, it may enhance diversity more in productive plant communities where aboveground competition is high (Huston, 1994).

Response of a forest to fire can be influenced be tree species composition, as the probability of mortality is species-dependent (Balch *et al.*, 2011). Fire favor species that can tolerate heat stress and increase the abundance of pioneer species in the short term (Barlow & Peres, 2008; Cochrane & Schulze, 1999; Slik *et al.*, 2010).

Sprouting is one of the ways in which a plant recovers after a fire. Shoots originate from dormant buds located on plant parts above the ground surface or from various levels within the litter, duff, and mineral soil layers (Flinn and Wein, 1977). Fire induce germination of dormant seeds of some species which results in an abundance of seedlings of these species in the first post fire year. Perennial seedlings that mature will flourish, depending on their inherent longevity, as long as the area has their specific environmental requirements. Eventually, they may persist only as seeds.

Age determines the resistance of a tree to a fire, as the age increase, the resistance also increases. Self-pruning or removal of basal branches by surface fires enlarges the crowns and the height to the base of the live crown. Bark thickness and stem diameter also increases. A suppressed tree may develop fire resistance characteristics at a much slower rate than a vigorous tree of the same age and species resulting, in a much thinner bark (Wade, 1993).

An assemblage of many of the species that were growing on the site and represented in the seed bank at the time of the fire are usually the species composition after a fire. Vegetative regeneration is common to many species and make a major contribution after a fire disturbance (Ingersoll and Wilson, 1990)

1.2.2 Effects of fires on animals

Fires affect herbivores mainly through effects on their habitat. Fires cause short term increases in wildlife foods which contribute to increase in populations of some animals. The increase in population after a fire is moderated by the animals' ability to survive in the altered environment which is often simplified structure. The extent of change in habitat structure and species composition caused by fire affects the extent of fire effects on animal community with stand-replacement fires causing greater changes in the faunal communities of forests than grasslands fires. Stand replacement fires do change the animal community more than under storey fires in forests. Animal species are adapted to survive the pattern of fire frequency, season, size, severity, and uniformity that characterize their habitat. Habitat for many animal species declines as a result of change in fire frequency or change in fire severity from pre settlement patterns (Smith *et al.,* 2000)

Terrestrial vertebrates with poor climbing abilities and low mobility are most affected by surface fires and end up dead or injured by the fires (Peres, 1999; Peres *et al.*, 2003). Many other arboreal vertebrate species appear to succumb to smoke asphyxiation, including several primates, sloth, arboreal echimyid rodenzts and bird species (Mayer, 1989; Peres, 1999). Sub-lethal injuries of more mobile animals are also common. Animals that survive an accidental fire and smoke usually have two options: either to emigrate into nearby unburned areas if available, or to remain within the burned forest matrix. The animals that move into unburned forest always face interference competition through territorial aggression from conspecifics (Bierregaard and

Lovejoy, 1989), density-dependent reductions in fitness through exploitative competition for food, mates or other resources, and suffer from their poor familiarity with the spatio-temporal distribution of resources, all of which can be aggravated by overcrowding.

Only when the population levels in adjacent unburned forest have been reduced by game hunting is it possible to envisage a relaxation of density-dependent effects, (Peres *et al.*, 2003). Individuals remaining in the burned forest face a different set of problems. Initially, large-bodied animals may be hunted relentlessly by rural peoples desperate to compensate for losses of food crops. This is exacerbated by the lack of cover resulting from the scorched understorey and loss of midstorey and canopy foliage (Haugaasen *et al.*, 2003), and the increasing clumping of animals around remaining fruiting trees or patches of unburned forest (Lambert & Collar, 2002). Animals that escape or are unaffected by hunting may face severe food shortages, as many canopy trees abort fruit crops and shed leaves following the traumatic heat stress (Peres, 1999). In some cases, animals appear to be able to compensate by switching to alternative dietary items.

Vegetation structure, floristic composition, densities of conspecifics and heterospecifics, and microclimate influence the availability of food, risk of predation and availability of nest sites to birds (Hilden, 1965). Disturbances like forest fires have the potential to diversify avian habitats at the local, landscape and regional scale.

Prescribed fires can be used to reduce accumulations of wood on the forest ground. This can change the number of standing dead trees or snags. It can consume some existing snags and create others by killing trees. Changes in number and characteristics of snags may affect populations of cavity-nesting birds because snags provide sites for nesting, roosting, and foraging (Mannan *et al.*, 1980).

Few studies have been done regarding the effects of fires on birds. Fire can destroy nests or nest sites and interfere with breeding and territory establishment. Fire reduces population of ground and shrub nesting birds while the population of woodpeckers and species that forage on the ground increases. (Russell *et al*., 2009)

Some birds which include predators that respond immediately and take advantage of temporarily increased availability of food after a fire. The predators feed on exposed and fire killed prey.

Seed eaters are also be found in the burnt areas while feeding on seeds that have been released by the fire. The recovery avian communities are affected by the severity of fire and the amount of vegetation killed during the fire. Most communities recover rapidly following single fires regardless of fire intensity although such fires may pose a significant threat to species with a restricted distribution, limited reproductive potential, poor dispersal ability and narrow habitat requirements. Birds persisting in fragmented habitats are particularly at risk. Increases in fire frequency, impact and response are of great significance as a threatening process to avian communities (Woinarski and Recher, 1997).

1.2.3 Effects of fires on soil physical and chemical properties

Fire has well documented effects on soil (Nearly *et al*., 1999). Forest fires can affect physical and chemical properties of the underlying soils (Certini, 2005). The long term effects of fire on soils can persist for years after the fire or be permanent (DeBano *et al*., 1998). The magnitude of the effects is determined by the burn severity which consists of peak temperatures and the duration of fires. During low to moderate fires, no irreversible ecosystem change occurs, but the enhancement of hydrophobicity can render the soil less able to soak up water and become more prone to erosion (Certini, 2005).

Fires release nutrients from the biomass and improve nutrient cycling. It affects soil properties through changing soil microbial activities and water relations, and creating heterogeneous mosaics, which in turn, can further influence fire behavior and ecological processes. (Chen, 2006)

Soil properties are significantly affected by fire because of the organic matter located on the soil surface which combusts rapidly. These changes in organic matter in turn affect the physical and chemical properties of the underlying soil (John and Rundel, 1976). Fire plays a dominant role in recycling organic matter in areas where decay has been constrained either by dry or cold climates or water (DeBano *et al*., 1998). It acts as a rapid mineralizing agent that releases nutrients instantaneously as contrasted to natural decomposition processes which may require years (John and Rundel, 1976).

During a fire, heat is transferred downward into the soil and raises the temperature of the soil. The greatest increase in temperature occurs at, or near, the soil surface. Within short distances downward in the soil, however, the temperatures increases quickly diminish so that within 5 to 10 cm of the soil surface, the temperatures are scarcely above ambient temperature (Certini, 2005).

1.2.4.1 Effects of fires on pH

Soil pH is a measure of the hydrogen ion activity in the soil and is determined at specified moisture contents. Neutral soils have a pH of 7, acidic soils have a pH less than 7, and basic soils are those with a pH greater than 7. The combustion of organic matter during a fire and the subsequent release of soluble cations tend to increase pH slightly because basic cations are released during combustion and deposited on the soil surface. (Nearly *et al.,* 2005)

Low to moderate severe fires lead to an increase in soil pH (Certini, 2005). The soil pH increases after a fire as a result of organic acids denaturing and release of bases (Arocena and Opio, 2003). Increase in pH is as a result of ash accretion. The response depends on the amount of ash and buffering capacity of the soil. This rise in pH is because mineral substances are released as oxides or carbonates that usually have an alkaline reaction (Cartini, 2005). Top soil pH could increase as much as three units immediately after burning due to the production of K and Na oxides, hydroxides, and carbonates (Ulery *et al*., 1993)

1.2.4.2 Effects of fire on Nitrogen

The immediate response of soil organic nitrogen to heating is a decrement because of some loss through volatilization (Fisher and Binkley, 2000). Ammonium (NH_4^+) and nitrate (NO_3) are the inorganic forms of nitrogen that originate during the burning. Ammonium is a direct product of the combustion, while nitrate forms from ammonium some weeks or months after the fire as a result of nitrification (Covington and Sackett, 1992).

Volatilization is the chemically driven process most responsible for nitrogen losses during fire. There is a gradual increase in nitrogen loss by volatilization as temperature increases (White *et al*., 1973).

The amount of total nitrogen that is volatilized during combustion is directly proportional to the amount of organic matter destroyed (Raison *et al*., 1985). It has been estimated that almost 99 percent of the volatilized N is converted to N_2 gas (DeBell and Ralston, 1970). N_2 can be produced during organic matter decomposition without the volatilization of N compounds at lower temperatures (Grier, 1975). The N that is not completely volatilized either remains as part of the unburned fuels or it is converted to highly available NH_4-N that remains in the soil (DeBano, 1991). Nitrogen is considered the most limiting nutrient in wild land ecosystems and as such it requires special consideration when managing fire, particularly in nitrogen deficient ecosystems (Maars *et al.*, 1983).

1.2.4.3 Effects of fire on Organic Carbon

In forest ecosystems, carbon is more evenly distributed above ground and below ground. In general, soils with larger proportions of organic matter in the aboveground biomass and on their forest floors are more prone to disturbances including fire in their nutrient and carbon regimes than those in which most of the carbon in the ecosystem is located belowground. An aggrading forest ecosystem sequesters nutrients and carbon aboveground in both the biomass and the forest floor (Knoepp and Swank, 1994).

1.2.4.4 Effects of fire on Phosphorous

Phosphorus is probably the second most limited nutrient found in natural ecosystems. Deficiencies of phosphorous have been reported in phosphorous-fixing soils and as a result from N fertilization applications (Heilman and Gessel, 1963). Phosphorus uptake and availability to plants is complicated by the relationship between mycorrhizae and organic matter and in most cases does not involve a simple absorption from the soil solution (Trappe and Bollen, 1979).

Phosphorus is lost at a higher temperature during soil heating than nitrogen, and only about 60 percent of the total phosphorous is lost by non-particulate transfer when organic matter is totally combusted (Raison and others 1985). The combustion of organic matter leaves a relatively large amount of highly available phosphorous in the surface ash found on the soil surface immediately following fire. This highly available phosphorous, however, can be quickly immobilized if calcareous sub- stances are present in the ash and thus can become unavailable for plant growth (Nearly *et al*., 2005).

1.2.4.5 Effects of fire on Cation Exchange Capacity (CEC)

Cation exchange capacity is the sum of the exchangeable cations found on organic and inorganic soil colloids. It arises from the negatively charged particles found on clay particles and colloidal organic matter in the soil Cation exchange capacity sites are important storage places for soluble cations found in the soil. The adsorption of cations prevents the loss of these cations from the soils by leaching following fire (Nearly *et al*., 2005).

The degree of humification of organic matter affects the cation exchange capacity. The more extensive the decomposition of the organic material, the greater the exchange capacity. Soil heating during a fire can affect cation exchange capacity in two ways. The most common change is the destruction of humus compounds. The location of the humus layer at, or near, the soil surface makes it especially vulnerable to partial or total destruction during a fire because organic and humic materials start decomposing at about 100°C and are almost completely destroyed at 500 °C. In contrast, the cation exchange capacity of the clay materials is more resistant to change because heating and temperatures of 400 °C must be reached before dehydration occurs. The complete destruction of clay materials does not occur until temperatures of 700°C to 800 °C are reached. In addition, clay material is seldom located on the soil surface but instead is located at least several centimeters below the soil surface in the B-horizon where it is well protected from surface heating (DeBano *et al.*, 1998).

The leaching losses of soluble nutrients released during the fire are affected by the amount of CEC remaining after a fire. For example, the prefire CEC of sandy soils may consist mainly of exchange sites found on the humus portion of the soil. If large amounts of humus are destroyed in these sandy soils during burning, then no mechanisms are available to prevent large losses of soluble nutrients by leaching (Soto and Diaz-Fierros, 1993).

1.3 Justification

Closed-canopy forests have been destroyed over the years and their area reduced to less than two percent of the total land area in Kenya. At the same time, key economic sectors, including cash and subsistence crop production, tourism and energy generation, have increasingly relied on the environmental services provided by our shrinking forests (Lambrechts *et al*., 2003).

Every year, the Aberdare ranges experiences forest fires during the dry season which occurs from January to March. During this season, the area is usually characterized by dry vegetation and very strong winds which make the fire spread very fast once it’s started. Despite the frequent occurrences of these fires at the Aberdare Ranges forests no studies have been conducted to find out the ecological effects of these fires.

The Study on the effects of fire in the forest will help in knowing how the vegetation is affected after an occurrence of fire and after how long the vegetation regenerate. It will also help in knowing how the wildlife and birds are affected by the fires which utilizes the vegetation. The soil properties will help the forest managers have the right soil information before rehabilitating these areas in cases where these places may need to be rehabilitated.

The large body of existing literature on fire and fire ecology indicates an urgent need to synthesize the information on the pattern of fire effects on ecosystem composition, structure, and functions. Understanding fire effects and underlying principles are critical to reduce the risk of uncharacteristic wildfires and for proper use of fire as an effective management tool toward management goals. (Chen, 2006)

1.4 Objectives:

Main Objective

To determine the effect of fires at Aberdare ranges on plant species diversity, animal population and soil physical and chemical properties.

Specific objectives:

The specific objectives were to:

- Determine the effect of fire on plant species diversity at different time periods.
- Assess the effect of fire on population of herbivores and birds at different time periods.
- Determine the effects of fire on soil physical and chemical properties at different time period.

1.5 Hypotheses

1. Fires lead to a decrease in the plant species diversity.
2. Fires lead to a decrease in animal diversity.
3. Fires lead to a change in physical and chemical properties of soil.

CHAPTER TWO
2.0Materials and methods

2.1 Study area

2.1.1 Geographic location

The study was carried out at the Aberdare Range in central Kenya close to the equator. It is the third highest mountain in Kenya, with two main peaks: Oldonyo Lesatima (also known as Sattima) and Kinangop, which are at altitudes of 4,001 and 3,906 meters respectively. The Range presents a deeply dissected topography sloping gradually to the east. In contrast, the western side drops along impressive fault escarpments towards the Rift Valley (www.britannica.com retrieved 2014-06-17). The Aberdare mountain range lies between Latitude 00^{o} 00" – 01^{o} 00" South and

Longitude 36° 30" – 36° 55"East running in a NNW-SSE direction. Altitude varies from 1850 m in the lower parts to about 4000 m at the highest point. Specifically the study was done at Geta forest zone which is at the western side of the Aberdares and the biggest part of Aberdares hills. Data was also collected from Kipipiri hill which lies next to the Aberdare ranges.

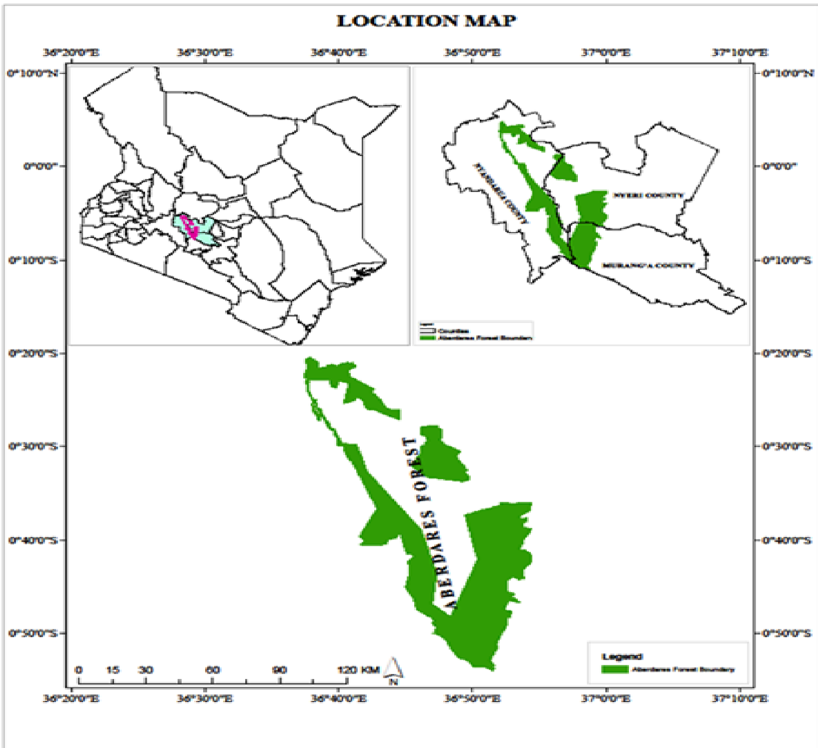

Figure 1: The location of Aberdares in Kenya.

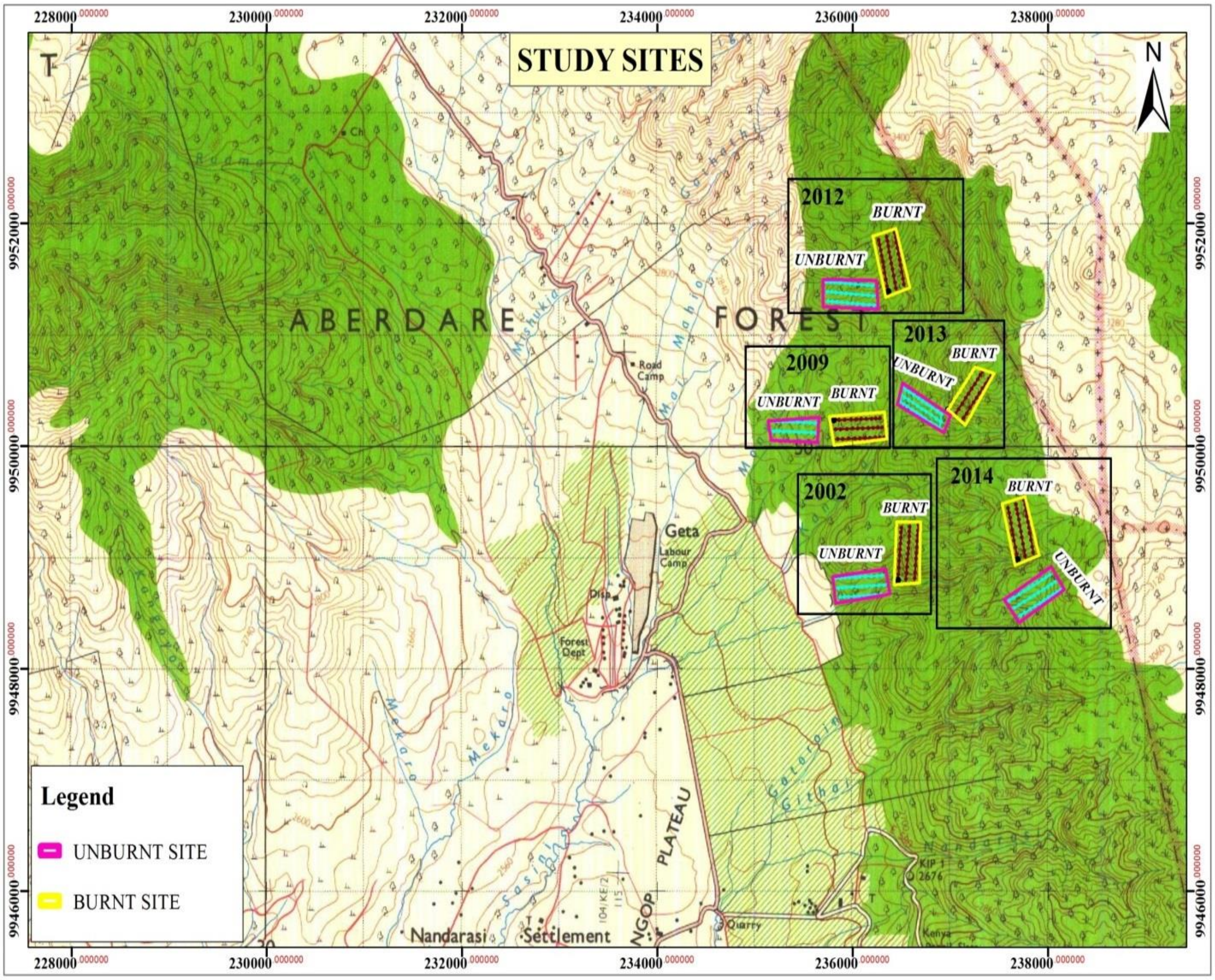

Figure 2: Study sites locations in Western Aberdares

2.1.2. Climate of Aberdare Ranges

The climate of Aberdare Ranges is largely determined by altitude. The rainfall distribution is greatly influenced by movement of inter-tropical convergence zones of air masses of southern and northern hemisphere. It is characterized by two rainy seasons: long rains from April to May, and short rains from October to November. Rainfall varies with altitude and exposure to the dominant wind from the Indian Ocean, but reaches a maximum of around 2,600 mm annually on the south-eastern slopes and drops to less than 900 mm a year on the northern and south-western lee slopes (Butynski, 1999). On the western side, rainfall reduces sharply from about 1,400 mm at the forest border to less than 700 mm in the valley of the Malewa River only 50 km from the forest boundary.

The northern end of the range has 3-4 dry months each year with the seasonal distribution showing three rainfall peaks: march-may (long rains), July –August and November. Elsewhere the rainfall distribution is bimodal with peaks in April-May and October-November and only 1-2 dry months each year. Temperature decreases with increase in altitude and rainfall also declines with altitude.

2.1.3 Vegetation in Aberdare Ranges

Various vegetation zones can be distinguished at the Aberdare Ranges, including the closed-canopy forest belt, the bamboo zone, and the sub-alpine and alpine vegetation. The forest belt covers a major part of the range. Most of the forest is gazetted as forest reserves. The key trees and shrubs species at the Aberdares are shown in the table below.

Table 1: The key vegetation species at the Aberdare ranges.

Vegatation zone	Altitude in m asl and location	Key trees and shrub species
1.Montane forest zone (a)Moist forest	1900 - 2500 / East 2100 - 2500 / South-East	*Cassipourea malosana, Ekebergia capensis, Teclea nobilis, Calodendrum capense, Podocarpus latifolius, Nuxia congesta* *Ocotea usambarensis, Macaranga kilimandscharica, Neoboutonia macrocalyx, Tabernaemontana stapfiana, Prunus Africana*
(b)Dry forest	1800 - 2400 / South-West 2400 - 3300 / West 2300 - 3200 / North, North-East	*Juniperus procera, Calodendrum capense, Teclea simplicifolia* *Juniperus procera, Olea europaea (africana), Podocarpus falcatus, Nuxia congesta*
2.Bamboo zone	2400 - 3000 / East, South-East 2700 - 3300 / West	*Arundinaria alpina with scattered trees, including Podocarpus latifolius and Nuxia congesta*
3.Hagenia-Hypericum zone	2950 - 3500 (discontinuous)	*Hagenia abyssinica, Hypericum revolutum, Rapanae melanophloeos*
4.Ericaceous zone	2900 - 3560 (discontinuous)	*Erica excelsa, Erica trimera, Erica arborea, Cliffortia nitidula, Helichrysum nandense, Stroebe kilimandscharica*

(Source; Ng'ang'a, 1990)

Some parts of the upper forest zone fall within the Aberdare National Park. The forest belt of the Aberdare Range is characterized by a high diversity of forest types, because of the wide altitudinal range (from 1,800 to 3,600 meters) and the climatic differences between the slopes. The Aberdare Range is heavily-forested. Much of the range has been protected within the Aberdare National Park since its creation in 1950(Ng'ang'a and Kamande, 1990).

Vegetation zones and species distribution are distinguished according to the different climatic zones and altitudes, mostly through variation in vegetation structure, cover and composition. A total of 778 species, sub-species and varieties of vascular plants belonging to 421 genera and 128 families, have been documented in the Aberdare. Plants endemic to Aberdare include *Lobelia deckenii ssp sattimae*, *Helichrysum gloriadei* and *Alchemilla hageniae*.

2.1.4 Animals in Aberdare Ranges

The Aberdare Range forests host a number of threatened fauna species. The Jackson mongoose (*Bdeugale jacksoni*), endemic to Kenya's montane forests and the rarely seen golden cat (*Felix aurata*) are two threatened mammals. Other large threatened mammals of international conservation interest that occur in Aberdare forests are bongo (*Tragelaphus euryceros*), giant forest hog (*Hylochoerus meinertzhageni*), black rhino (*Diceros bicornis*), elephant (*Loxodonta africana*), leopard (*Panthera pardus*) and African hunting dog (*Lycaon pictus*). In addition, the forest harbours bushbuck (*Tragelaphus scriptus*), mountain reedbuck (*Redunca fulvorufula*), waterbuck (*Kobus ellipsi prymnus*), cape buffalo (*Syncerus caffer*), suni (*Neotragus moschatus*), side-striped jackel (*Canis adustus*), eland (*Taurotragus oryx*), and varieties of duikers and bushbabies. The forests are rich in primates and the common ones include the black-and-white colobus monkey (*Colobus guereza*), sykes monkey (*Cercopithecus mitis*), vervet monkey (*Cercopithecus aethiops*) and baboons (*Papio anubis neumanni*) (Waithaka, 1994)

The Aberdare Range is internationally recognized as an Important Bird Area (IBA). The Range holds 52 of Kenya's 67 Afrotropical highland species and six of the eight restricted range species in the Kenyan montane endemic bird areas. Over 270 species of birds have been recorded in the Aberdares including the following globally threatened and restricted-range species: Sharpe's Longclaw, Abbott's Starling, Aberdare Cisticola and Jackson's Widowbird. Regionally threatened species found in the Abedares include Cape Eagle Owl, African Crowned Eagle, and

African Green Ibis. Jackson's Francolin, Hartlaub's Turaco and Bar-tailed Trogon are characteristic and spectacular birds of the Aberdare Range.

2.1.5 Soils in Aberdare Ranges

The soils are polygenetic and occur in deeply undulating topography. They have been subjected to intense leaching and have a low base saturation. They are derived from massive lava flows, deep beds of volcanic tuffs and ash showers of geological formation. Soils are deeply weathered, stone free, highly porous, free draining and support deep rooting associations of forest vegetation.

Soils on the upper eastern slopes of the Aberdare Ranges have inherent high fertility, being of basaltic origin. They are well drained, normally very deep, dark reddish brown, friable clays with a humid top- soil layer. Soils on the western boundary of the Ranges are also of medium to high inherent fertility, but are more variable and interspersed with poorer draining soils and lower fertility. The soils of the moorlands are umbric andosols. They are derived from volcanic glass. They have a high content of organic matter and are very porous.

The soils of the Northern Aberdare are rich in clay content (82.7%) and consist almost exclusively of kaolinite. Red kaolinite soils are found on slopes and dark grey, swelling montmorillonitic (black cotton) soils are found in areas of impeded drainage.

The soils of the southern area are characterized by dark surface horizons and are rich in organic matter. Their bulk density is low and includes Leptosols which are characterized by continuous coherent hard rock at very shallow depth, strong brown loams, eutrophic brown soils on volcanic ash and Gleysols which show hydromorphic properties within 50 cm of the surface and are found in valley bottoms.

2.2 Methods

The data collection was done between September 2013 and March 2014. Both the wet season (September 2013 to December 2013) and dry season (January 2014 to March 2014) were covered. Data was collected from five sites in the forest that had been burnt in 2002, 2009, 2012, 2013 and 2014. These sites were selected since they were close to each other and had similar topography and vegetation. In every site, data was collected from both burnt area and un-burnt area that acted as a control. To avoid fire effects, the un-burnt areas were separated from burnt areas by buffers of at least 100M.

2.2.1 Woody vegetation sampling

Point Centered Quarter (PCQ) sampling technique was used for the woody vegetation sampling. Three line transects of 500 m long were randomly set in each site. Sampling points along the transect lines were randomly set and a ranging pole placed at each sampling point. Each sampling point was divided into four quarters by use of a perpendicular line placed at right angles to the line transect. Individual woody plants species nearest to the point in each quarter was identified and the points to individual plant distances were measured using a tape measure (Kevin, 2007). The data that was recorded included: The nearest tree species to sampling point, point to individual plant distance measured using a tape measure, height of tree estimated measured using clinometer, diameter at breast height (DBH) measured by a Vernier calipers or a tape measure and canopy cover measured by tape measure.

2.2.2 Herbaceous Plants Sampling

Quadrat sampling method was used (Cox, 1990). Sampling was done on areas affected by fire in 2002, 2009, 2012, 2013 and 2014. Three line transects of 500 m long were randomly set in both burn and un-burnt areas at each site. Sampling points along the transect lines were randomly set. At every point, herbaceous layer species were sampled using a quadrat measuring $0.25m^2$. In every quadrat, all the species were recorded. The percentage cover of the species was determined through estimation in all the quadrats. The heights of the plants were measured using a tape measure.

2.2.3 Wildlife census

Direct foot count was done on all study sites to estimate the number of wildlife. The count was done once every week. The species name and the number of individuals were recorded.

Indirect animal count was also be done by use of animal dung as described by Barnes, (1996). The indirect technique gave an index of abundance rather than a measure of animal density (Sutherland, 1996). The length of fine dung persists may vary between habitat and between time periods, depending on weather, dung decomposition rate, fibre content and the number of dung beetles and termites. To overcome this problem, very fresh dung was marked and during the second visit, the fresh dung marked in the previous visit was used to compare the dung in the site and ignored any dung that looked more decayed. The indirect animal count gave a good indication of the study site use. The following formula was used to convert the number of dungs into the number of animals.

$$\text{Number of animals} = \frac{\text{Number of dung}}{\text{Number of days between visits x defecation rate}}$$

The species diversity was calculated using Shannon Weiner diversity index.

2.2.4 Bird count

Bird count was done once every week in the morning hours between 9.00 AM to 12.00 Noon. Point count was done along two transects in each study site. In every site, the bird count was done on the same day in both burnt and un-burnt areas. The sampling points were chosen systematically on the transect after every 100m and in every sampling point, the birds were observed for 10 minutes by use of naked eyes and 8x40 binoculars. The bird species observed were recorded. The bird species heard calling were also recorded. The distance between the observer and the birds was also estimated and recorded and the activity of the bird. Species diversity was calculated using Shannon Weiner diversity index.

2.2.5 Soil sampling

In every site, the burnt and un-burnt sites were selected such that they their terrain were as similar as possible. Three sampling points were determined randomly. Area for sampling was cleared of vegetation. Hand augers, spades and shovels were used to dig up the soil. The soil was dug up to 45cm depth. Samples were taken from 0-15cm, 15-30cm and 30-45cm in every sampling point. The soil placed in plastic-lined soil sampling bags and labeled. Soil samples were taken for further analysis at the University of Nairobi soil laboratories. The soils were air dried, sieved through 2mm screen and analyzed for physical and chemical properties.

2.2.5.1 pH

For each sample, pH was determined using distilled water at a soil to water ratio of 1:2.5. Each sample was shaken for 30 minutes using a shaker and then an electronic pH meter with a glass electrode was used to determine the pH (Peech, 1965).

2.2.5.2 Total Nitrogen

Total Nitrogen content was determined by Kjeldahl digestion method (Fleige et al., 1971). To 1 g of each soil sample, 3.5 ml of phenolic-sulphuric acid (36N) was added and left to stay for 15 minutes. 0.5g of Sodium thiosulphate was then added and the samples left to stay for another 15 minutes. 0.5g of potassium sulphate, 0.5g of selenium reaction mixture and 3.5ml of concentrated sulphuric acid were thereafter added. The samples were then digested using an electric Kjeltric digestion block. The digested samples were distilled after an addition of 40ml of 10N NaOH solution. The released nitrogen in form of NH_3 aqueous was captured by a 1% boric acid solution. The trapped NH_3 was titrated with 0.01N solution of standard sulphuric acid. The results were expressed as percentage total nitrogen. The amount of nitrogen was calculated from the stoichiometric relationship that 1 ml of 0.01N sulphuric acid used in the titration is equivalent to 0.14 mg of Nitrogen.

2.2.5.3 Organic Carbon

The organic carbon was determined by the Walkley Black wet oxidation method (Black, 1965). The soil samples were crushed and passed through 0.6 mm sieve. 10 ml of dichromate solution was added into 1g of soil sample. 20ml sulphuric acid was added into the solution and mixed gently and the mixture allowed to stand for 30 minutes. It was then diluted to 200ml with deionized water. 10 ml. phosphoric acid, 0.2g ammonium fluoride, and 10 drops diphenylamine indicator were added.

It was titrated using a standard solution of ferrous sulphate. This procedure was repeated for all the soil samples. The organic carbon content was calculated using the equation below:

$$\%C = \frac{\text{Me of } K_2Cr_2O_7 - \text{Me of } Fe_2\text{SO4}SO_4 * 0.39}{\text{weight of soil}} * 100$$

Where Me = milliequivalents (normality * mls of solution)

0.39 = correction factor

2.2.5.4 Potassium

Potassium was extracted from air-dried soil samples by shaking the sample with 0.5M ammonium acetate acid solution for 30 minutes. This effectively displaces the potentially available potassium ions. The potassium content of the filtered extract was then determined using a Jenway PFP7 Flame Photometer.

2.2.5.5 Phosphorous

Phosphorus content was determined spectrophotometrically at an acidity of 0.20M H_2SO_4 by reacting with ammonium molybdate using ascorbic acid as a reductant in the presence of antimony (Mehlich, 1984).

Air dried soil measuring 2g was passed through < 2.0 mm into a 50 ml glass Erlenmeyer flask. 20.0 ml of Mehlich 3 extracting solution was added. The extraction flask was placed on mechanical shaker for five minutes. The suspension was filtered immediately and the extract collected in 40ml plastic vials.

2.2.5.6 Cation Exchange Capacity

The Total Cation Exchange Capacity was determined by the ammonium acetate extraction method (Black, 1965). The soil samples were equilibrated with 1N ammonium acetate of an adjusted P^H of 7.0. The equilibrated soils were then washed using four 50ml portions of ethanol, or until no NH_4^+ ions were detected in the supernatant liquid after centrifuging as tested by Nessler's reagent.

The soil samples were then distilled using the Kjeldahl distilling unit after an addition of magnesium oxide. 200ml of the distillate was collected over 2% boric acid indicator solution. The distillate was then titrated to endpoint with 0.1N standard HCl solution. 1 ml of 0.1N HCl used in titration is equivalent to 1 milliequivalent per 100g of soil for an original soil sample site of 10g. The results were expressed in milliequivalents per 100g of soil.

2.3 Data analysis

Species diversity of the vegetation and wildlife was calculated using Shannon Weiner diversity index:

$$H' = -\Sigma Pi \times Log Pi$$

$$\text{Where } Pi = \frac{ni}{N}$$

Where: *ni* is number of individuals of species *i*

N is total number of species

The data was analyzed using SPSS version 20 and Microsoft Excel spreadsheet statistical packages.

For the woody species diversity, T-test was used to determine whether there was a difference between the burnt sites and unburnt sites. Univariate Analysis of Variance was used to determine whether there were differences in the DBHs, heights, and canopies between the burnt and unburnt sites. ANOVA was also used to determine whether there were differences in DBHs, heights, and canopies among the five burnt sites.

For the herbaceous vegetation, T-test was used to determine whether there was a difference in species diversity between the burnt sites and unburnt sites. The percentage cover of the herbaceous vegetation was first transformed using arcsin transformation before it was analyzed. ANOVA was then used to determine the differences in cover, number of individual plants and heights between the burnt sites and unburnt sites and among the five burnt sites. The possible sources of variances where there were more than two groups were determined using Post Hoc method. The individual means were separated by the LSD method.

T-test was also used to determine whether there was a difference in wildlife species diversity.

For the soil properties, ANOVA was used to determine if there was any difference between the burnt and unburnt sites. ANOVA was also used to determine differences among the three depth levels. It was also used to determine differences in the soil properties among the five burnt sites.

CHAPTER THREE
3.0 Results

3.1Woody Species diversity
The data demonstrates that fires did not have effect on the species diversity of the woody plants as the calculated species diversity index did not show any big difference between the species diversity in the burnt sites and the un-burnt sites.

Table 2: The species diversity (H') of the woody vegetation in burnt and un-burnt sites.

YEAR	**BURNT (**H')	**UN-BURNT (**H')
2002	1.786	1.778
2009	2.341	2.362
2012	1.894	1.848
2013	2.018	2.114
2014	2.185	2.098

There was no significant difference in the species diversity of the woody vegetation between the burnt sites and un-burnt sites ($t_{0.05\ (1),\ 4}$=0.033288; P>0.05). The mean species diversity of all burnt sites was 2.045 ± 0.102 and that of all un-burnt sites was 2.040 ± 0.102.

3.2 Woody species community structure analysis
A total of 7 different species were counted belonging to 7 different families. The woody species found in all the sites are shown in the table 3. All the woody species found in each site and their parameters are shown in appendix 1.

Table 3: The woody species found in all the sites and the families they belong to.

Family	Species	Burnt Sites	Un-burnt sites
Bignoniaceae	*Markhamia lutea*	101	94
Canellaceae	*Warbugia ugandensis*	113	95
Cupressaceae	*Juniperous procera*	62	85
Lauraceae	*Ocotea usambarensis*	95	86
Oleaceae	*Olea Africana*	105	97
Podocarpaceae	*Podocarpus falcatus*	91	100
Stilbaceae	*Nuxia congesta*	36	39

3.2.1 DBH

The mean DBH of all the woody species at the burnt sites were less than those at the un-burnt sites. The mean DBH of the burnt sites increased with the years after burning with the one of 2014 being the smallest at 0.34±0.1m and that of 2009 being the highest at 0.60±0.2m. Although time had caused recovery of the woody species in burnt sites, there was no burnt site whose DBH had recovered to be equal or more than that of the un-burnt site. The DBH of all species are shown in appendix 1.

There was significant difference in the DBH of all species between all burnt sites and all un-burnt sites with the species in the un-burnt sites having a higher mean DBH of 0.63 ± 0.026m and those in the burnt sites having a mean DBH of 0.49 ± 0.026m ($F_{1, 64}$ =14.388; $P<0.05$) . There was also a significant difference in the DBH between all the five burnt sites ($F_{4, 28}$ =3.644; $P<0.05$) with the maximum mean difference being between 2002 and 2014 of 0.240 ± 0.08m and the minimum mean difference being between 2002 and 2009 of 0.02± 0.08m.

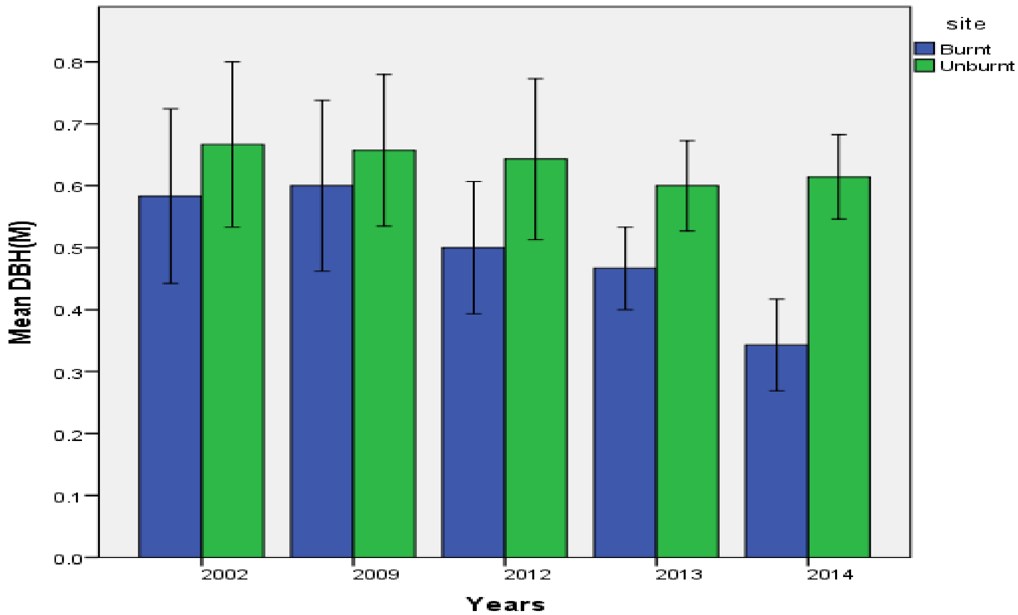

Figure 3: The mean DBH of the woody species in all the sites

3.2.2 Height

The heights of the woody species in the burnt sites were less than those in the un-burnt sites. The mean height of the burnt sites increased with the years after burning with the one of 2014 being the shortest at 21±5.2m and that of 2002 being the tallest at 24±5.1m. There is no burnt site whose the height had recovered to be equal or more than that of the un-burnt site. The heights of all woody species are shown in appendix 1.

There was no significant difference in the heights between all burnt sites and all un-burnt sites ($F_{1, 64}$ =3.037; P>0.05). There was also no significant difference in heights between all the burnt sites at different years ($F_{4, 28}$ =0.869; P>0.05).

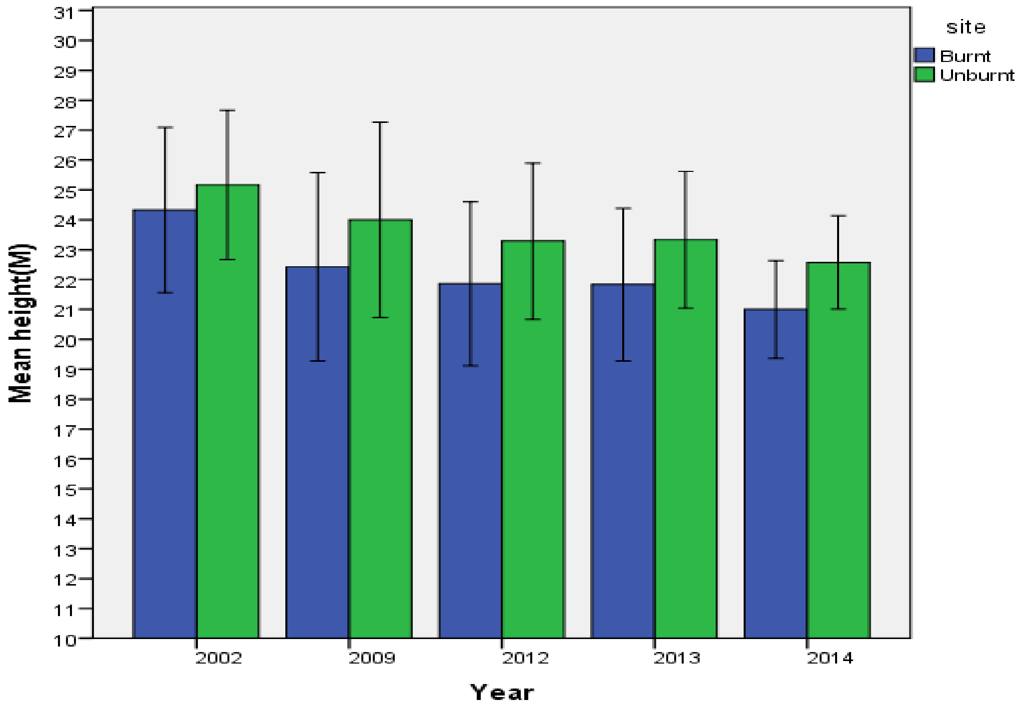

Figure 4: The mean height of the trees in all the burnt and unburnt sites

3.2.3 Canopy

The mean canopies of all the burnt sites were smaller than the un-burnt sites. The mean canopies of the burnt sites increased with the years after burning with the one of 2014 being the smallest at 1.5±0.4m and that of 2002 being the highest at 4.7±1.0m. There is no burnt site whose the canopy had recovered to be equal or more than that of the un-burnt site. The values of the canopy of all species are shown in appendix 1.

There was significant difference in the canopy between all burnt sites and un-burnt sites ($F_{1, 64}$ =12.467; P<0.05). There was also a significant difference in the canopy between all the five burnt sites ($F_{4, 28}$ =17.418; P<0.05) with the maximum mean difference being between 2002 and 2014 of 3.157 ± 0.44m and the minimum mean difference being at 2002 and 2009 of 0.371± 0.44m.

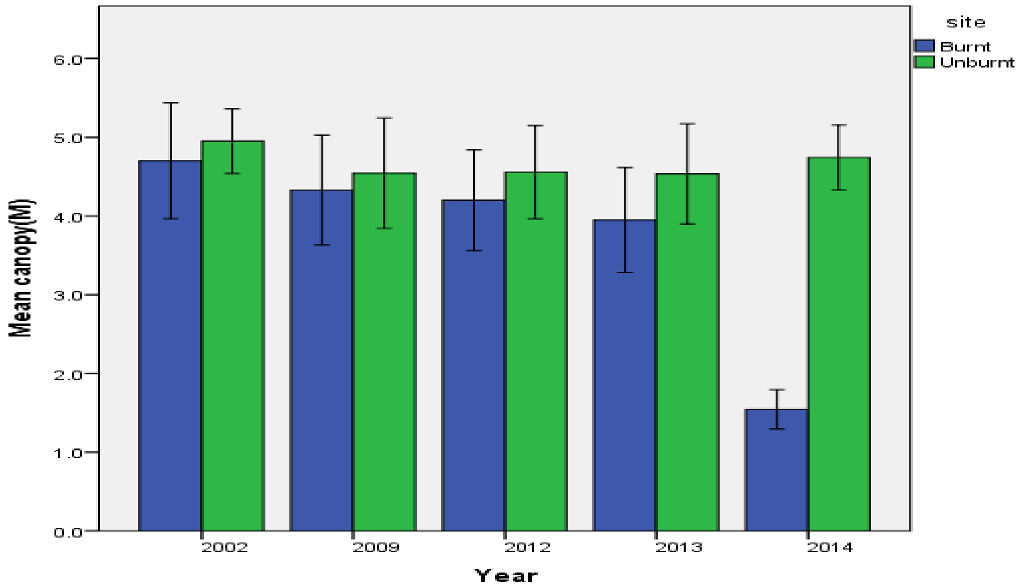

Figure 5: The mean canopy of trees in all the sites

3.3 Herbaceous species diversity

Fire markedly reduced species diversity of the herbaceous plants immediately after it occurred. However, the species diversity quickly recovered within one year and got higher than the un-burnt sites as plants regenerated after the fires. The site that had been burnt less than a year ago i.e. on 2014, is the only one that had lesser species diversity than the un-burnt site. Within a year, the herbaceous vegetation had already recovered and was more in the sites that had been burnt than in those that had not been burnt. The species diversity of the herbaceous vegetation of all sites is shown in table 4.

Table 4: The species diversity (H') of the herbaceous vegetation in burnt and un-burnt sites.

YEAR	**BURNT** (H')	**UN-BURNT(** H')
2002	2.441	1.682
2009	2.469	1.716
2012	2.369	1.837
2013	2.298	1.603
2014	0.950	1.737

3.4 Herbaceous species structural analysis

In all the burnt sites, a total of 39 species were counted while in all un-burnt sites, a total of 22 species were counted. There were 17 species that were only found in the burnt sites. All the species found in the un-burnt sites were also found in the burnt sites.

The percentage cover, number of individuals, height, density and frequency of all the species in all sites are shown in appendix 3. The list of all the herbaceous species found in both burnt and un-burnt sites is given in appendix 2.

The following is a list of the herbaceous plant species that were found in the burnt sites and not found in the un-burnt sites:

Table 5: The herbaceous species found in the burnt sites and not found in the un-burnt sites and the families they belong to.

Family	**Species**
Apiaceae	*Ferula communis*
Apocynaceae	*Gomphocarpus stenophyllus*
Asteracea	*Cardius keniensis*
	Crepis aurea
	Dichrocephala chrysanthemifolia
	Gnaphilium purpureum
Fabacea	*Crotalaria paniculata*
	Rhynchosia minima
	Trifolium cryptopodium
	Trifolium tembense
	Adenocarous manii
Lamiaceae	*Leucas venulosa*
Poaceae	*Oplismenus compositus*
Ranunculaceae	*Delphinium macrocentron*
Rhamnaceae	*Rhamnus prinoides*
Rosaceae	*Alchemilla gracilipes*
Rubiaceae	*Pentas lanceolata*

3.4.1 Cover

The site burnt on 2104 was the only one that had a lower mean percentage cover compared to its un-burnt sites. All the other burnt sites had more percentage cover than their un-burnt sites as the herbaceous vegetation in those burnt sites had already regenerated and exceeded the un-burnt sites. It had taken only one year for the herbaceous vegetation to regenerate its cover increase. The percentage cover of all the species found in every site is shown in appendix 3.

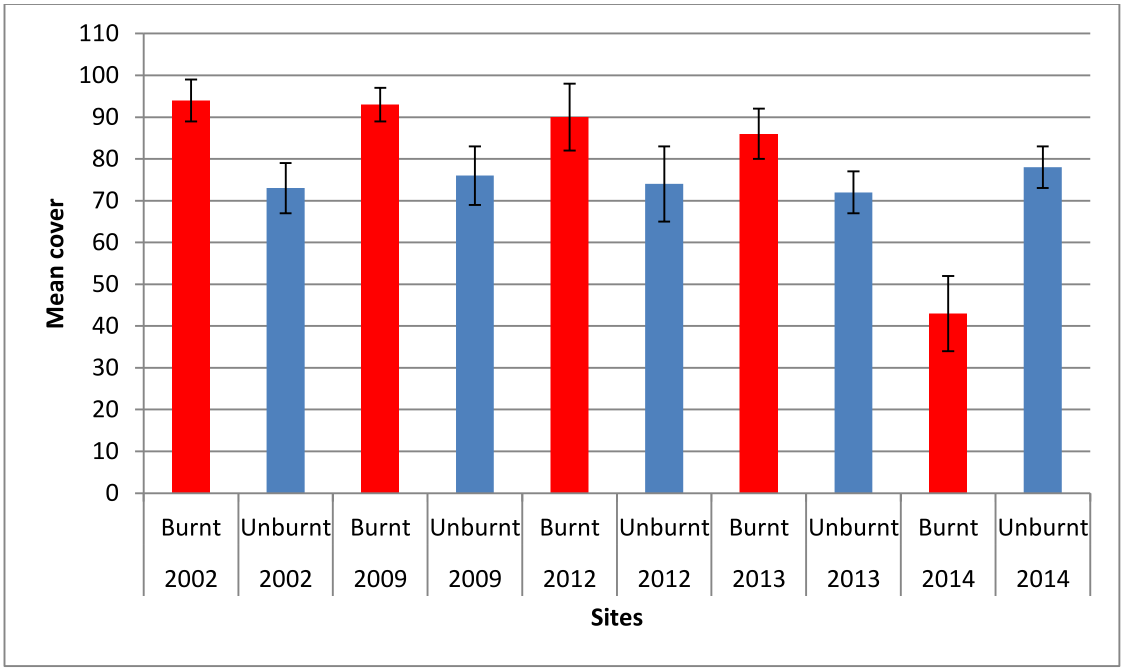

Figure 6: The mean percentage cover of the herbaceous vegetation in all sites.

After Arcsin transformation of the percentage cover, there was a significant difference between the percentage cover of the burnt sites and the un-burnt sites with the burnt sites having a higher mean cover 8.39±0.94% than the un-burnt sites which had a mean cover of 5.47±0.84 %

($F_{1, 128}$=5.360; P< 0.05).

There was also a significant difference in cover between all the burnt sites at the different years ($F_{4, 73}$=8.309; P< 0.05). The largest mean difference was between year 2014 and 2002 which was 20.23 ± 4.09% and the smallest mean difference was between year 2009 and 2002 which was 1.42 ± 2.13%

3.4.2 Number of individual plants

The site burnt in 2014 was the only one that had a lower number of herbaceous plants compared to the un-burnt sites. All the other burnt sites had a higher number of herbaceous plants than the un-burnt sites as fire led to regeneration in a year's time leading to a higher number of individuals in the burnt sites. The total number of every species is shown in appendix 3.

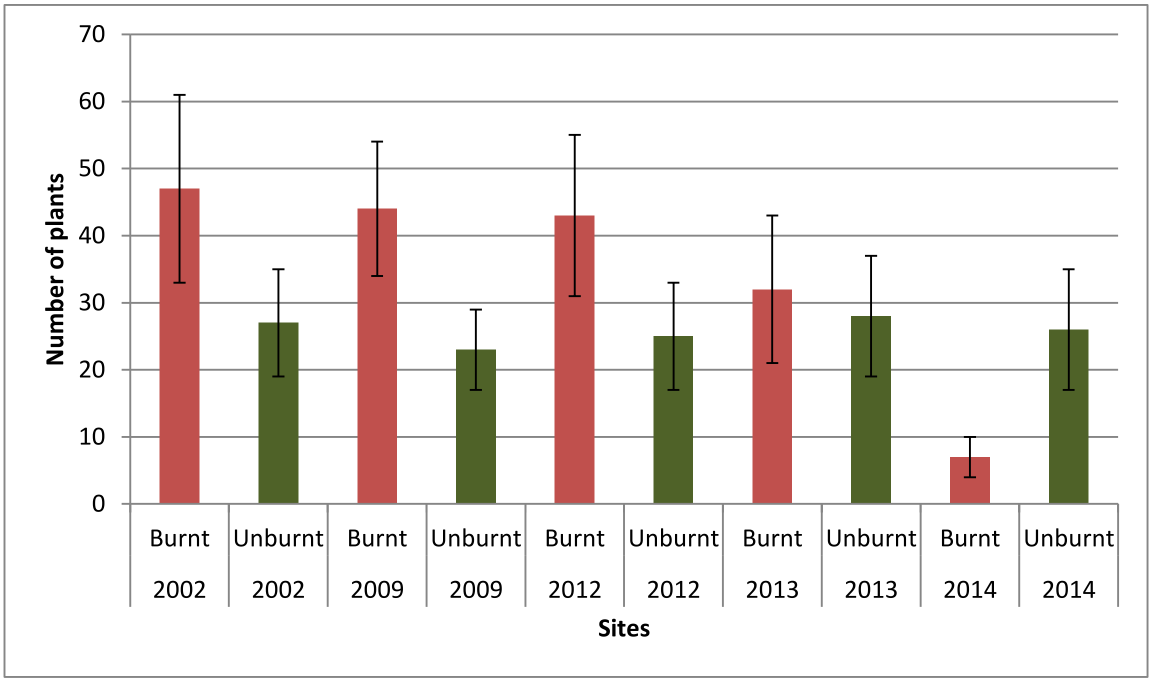

Figure 7: The mean number of plants found in all sites**.**

There was a significant difference between the number of individuals in the burnt sites and the un-burnt sites with the burnt sites having a higher mean number of 58.42±8.37 plants than the un-burnt sites which had a mean number of 35.83±7.53 plants ($F_{1,128}$=8.454; P< 0.05) .

There was also a significant difference in number of individuals between all the burnt sites at the different years ($F_{4,\ 73}$=3.101; P< 0.05). The maximum mean difference was between year 2014 and 2002 which was 58.23 ± 20.97 plants and the minimum mean difference was between year 2009 and 2002 which was 16.48 ± 18.34 plants.

3.4.3 Height

The site burnt in 2014 was the only one that had a smaller mean height of herbaceous plants compared to the un-burnt sites. All the other burnt sites had a bigger mean height of herbaceous plants than the un-burnt sites. The height of every species is shown in appendix 3.

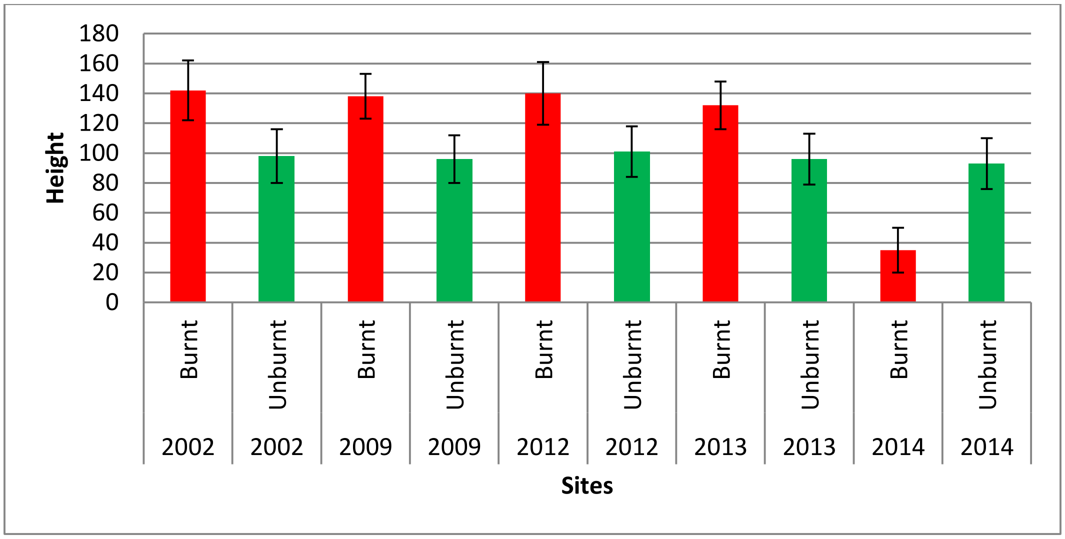

Figure 8: The mean height (cm) of the species found in all sites.

There was a significant difference between the height of the herbaceous plants in the burnt sites and the un-burnt sites with the burnt sites having a higher mean height of 81.56±5.05cm than the un-burnt sites which had a mean number of 61.85±4.47cm ($F_{1,128}$=8.536; $P< 0.05$).

There was also a significant difference in height of the plants between all the burnt sites at the different years ($F_{4, 73}$=5.972; $P< 0.05$). The maximum mean difference was between year 2014 and 2002 which was 41.34 ± 14.19cm and the minimum mean difference was between year 2009 and 2002 which was 10.74 ± 13.04cm

3.5 Animals

A total of 9 different species from 5 families were counted in all the sites. A list of all the animals found in all the sites is found in appendix 4:

There were no animals that were counted in the site that was burnt in 2014. Only the sites that had been burnt on 2014 and 2013 had fewer animals than their corresponding un-burnt sites. All the other burnt sites had more animals compared to their corresponding un-burnt sites. The site that was burnt on 2012 had the highest number of animals which was132. Amongst the un-burnt sites, 2014 had the highest number 86.

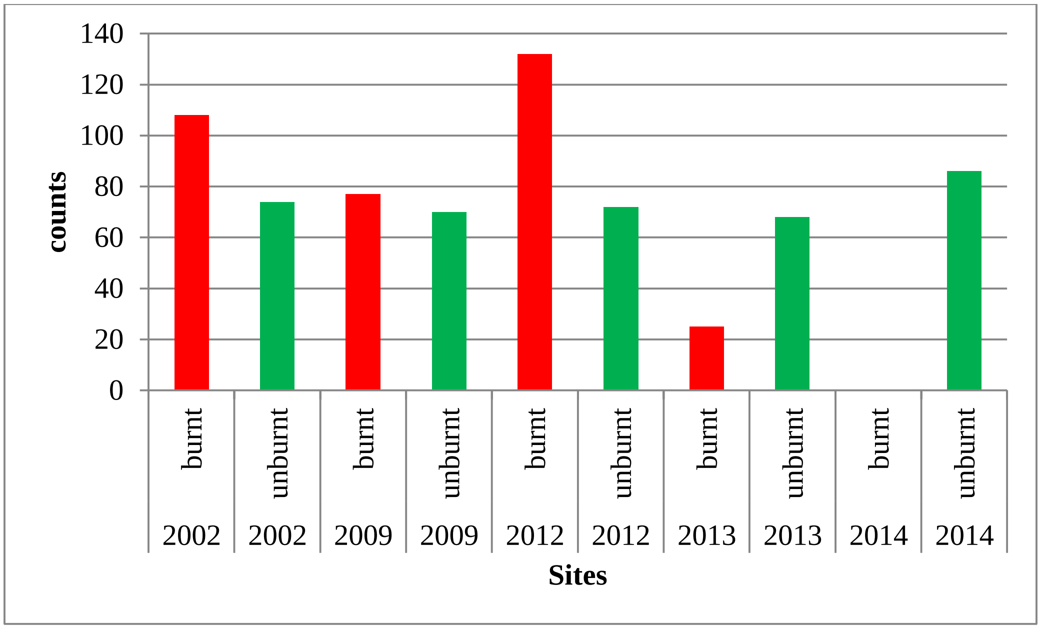

Figure 9: The total number of animals counted in all the burnt and un-burnt sites**.**

3.5.1Mammals species diversity

The species diversity of the animals counted in all the sites is shown in table 7. There was no animal that was observed on the site that had recently been burnt (less than 2 weeks after the fire-2014). In the site that had been burnt one year ago, the species diversity of the animals was less than the un-burnt site. In all the other sites that had been burnt more than one year ago, the species diversity of the animals was more than in their corresponding un-burnt sites.

Table 6: The species diversity (H') of animals in burnt and un-burnt sites.

YEAR	**BURNT**(H')	**UN-BURNT**(H')
2002	1.6771	1.5924
2009	1.6315	1.5997
2012	1.9407	1.7144
2013	1.2613	1.7926
2014	0	1.8623

3.6 Birds

A total of 34 species were counted in all sites belonging to 13 different families. The number of birds counted in every site is found in appendix 6. A list of the birds counted is found in appendix 5.

All the un-burnt sites had a higher number of birds than the un-burnt sites.

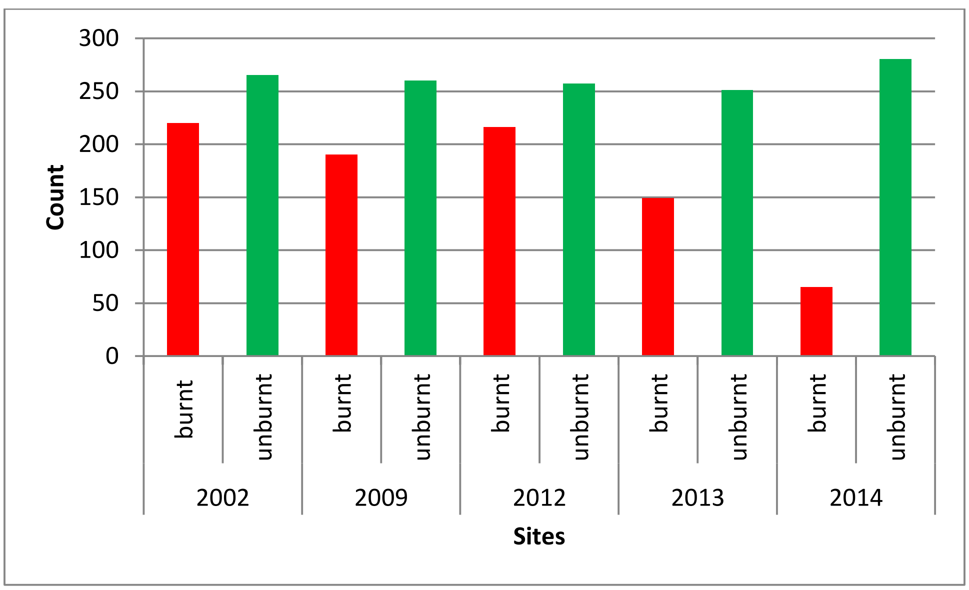

Figure 10: The total count of birds in all burnt and un-burnt sites.

3.6.1 Birds species diversity

The species diversity of the birds in all the sites is shown in table 8. The diversity was higher in all sites that had been burnt more than a year ago than in the un-burnt sites. Only the sites that had been burnt recently (less than a year ago) that had less species diversity than the un-burnt sites. This is because there was more vegetation in the burnt sites that provided habitat and food for the birds.

Table 7: The species diversity (H') of birds in burnt and un-burnt sites.

SITE	**BURNT**(H')	**UN-BURNT**(H')
2002	2.7352	2.6989
2009	2.8815	2.8291
2012	2.7871	2.6314
2013	2.1923	2.2248
2014	2.2036	2.6927

3.7 Soil properties

3.7.1 pH

All the pH values are shown in the appendix 7. The mean pH values of soils from both the burnt sites and un-burnt sites at the 3 levels of depth are shown in figure 10. The data showed that in all the sites, the soils were acidic as their pH ranged from 3.4 in the 2014 un-burnt to 6.5 in the site that was burnt in 2014. At 0-15 cm deep, the mean pH of the burnt sites was higher in the sites burnt on 2014, 2013, 2012 and 2009. The site that was burnt in 2002 was the only that had lower pH compared to the un-burnt site. The mean pH at 0-15 cm of the burnt sites was 5.28±0.68 and that of the un-burnt sites was 4.26±0.57. There was a significant difference in soil pH at 0-15cm depth between burnt sites and un-burnt sites ($t_{0.05\,(1),\,14}=3.619$; $P<0.05$).

At 15-30cm, the mean pH of all the burnt sites was higher than the un-burnt sites. The mean pH of the burnt sites at this depth was 4.99±0.28 and that of the un-burnt sites was 4.68±0.27. The pH at the depth of 15-30cm had a significant difference between the burnt sites and un-burnt sites ($t_{0.05\,(1),\,14}$=3.961; P<0.05).

At 30-45cm, the mean pH of the burnt sites was higher than un-burnt sites apart from the sites burnt on 2002 and 2009. The mean pH of the burnt sites was 4.85±0.27 while that of the un-burnt sites was 4.82±0.32. The pH at the depth of 30-45cm did not have a significant difference between the burnt and the un-burnt sites ($t_{0.05\,(1),\,14}$=0.349; P>0.05).

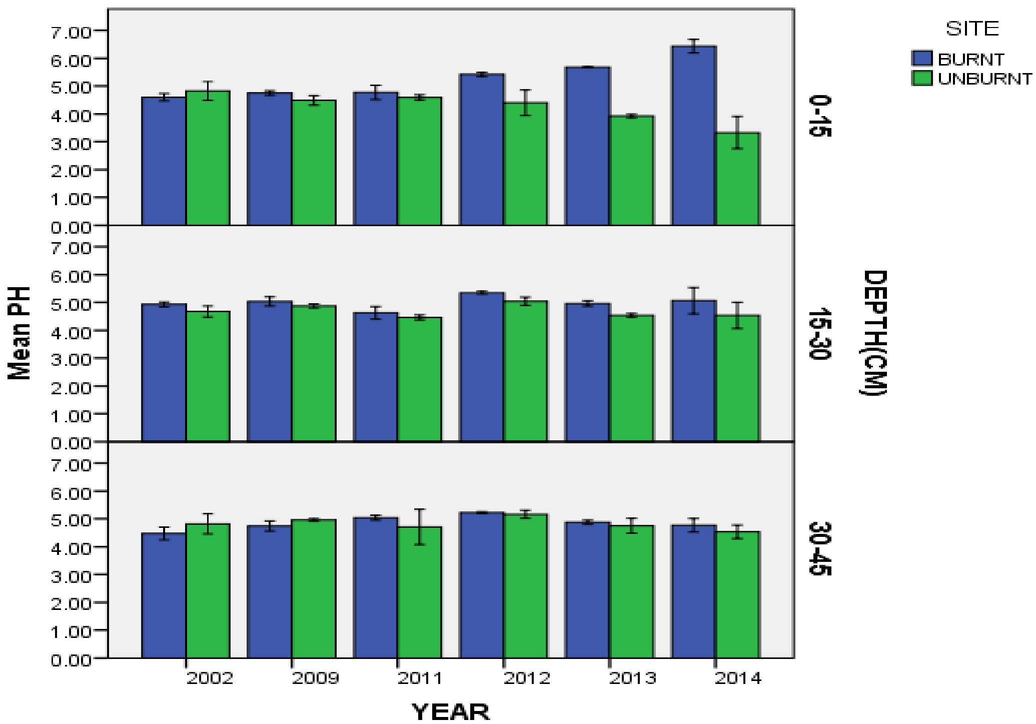

Figure 11: The pH of soils from both the burnt sites and un-burnt sites at the 3 levels of depth.

3.7.2 Nitrogen

Data for percent nitrogen is shown in Figure 11. The data shows that at 0-15 cm deep, the amount of nitrogen in the soils from the burnt sites was lower than in soils from un-burnt sites in the sites burnt on 2014, 2013 and 2012. However sites that were burnt in 2009 and 2002, the soils had slightly higher nitrogen content compared to the un-burnt site. The mean nitrogen at 0-15 cm of the burnt sites was 0.68±0.19% and that of the un-burnt sites was 1.10±0.66%. There was a significant difference in soil nitrogen at 0-15cm depth between burnt sites and un-burnt sites ($F_{1, 34}$=5.353; P<0.05). The data for all the nitrogen values is shown in appendix 7

At 15-30cm, the mean nitrogen of all the burnt sites was lower than the un-burnt sites apart from the site burnt in 2014 where the mean nitrogen was higher in burnt site than in the un-burnt site. The mean nitrogen of the burnt sites at this depth was 0.58±0.21% and that of the un-burnt sites was 0.70±0.17%. At this depth, there was no significant difference in nitrogen between the burnt sites and un-burnt site ($F_{1, 34}$=2.576; P>0.05).

At 30-45cm, the mean nitrogen of the burnt sites was lower than un-burnt sites apart from the sites burnt on 2014. The mean nitrogen of the burnt sites was 0.45±0.26% while that of the un-burnt sites was 0.67±0.26%. The nitrogen at the depth of 30-45cm had a significant difference between the burnt and the un-burnt sites ($F_{1, 34}$=29.55; P<0.05).

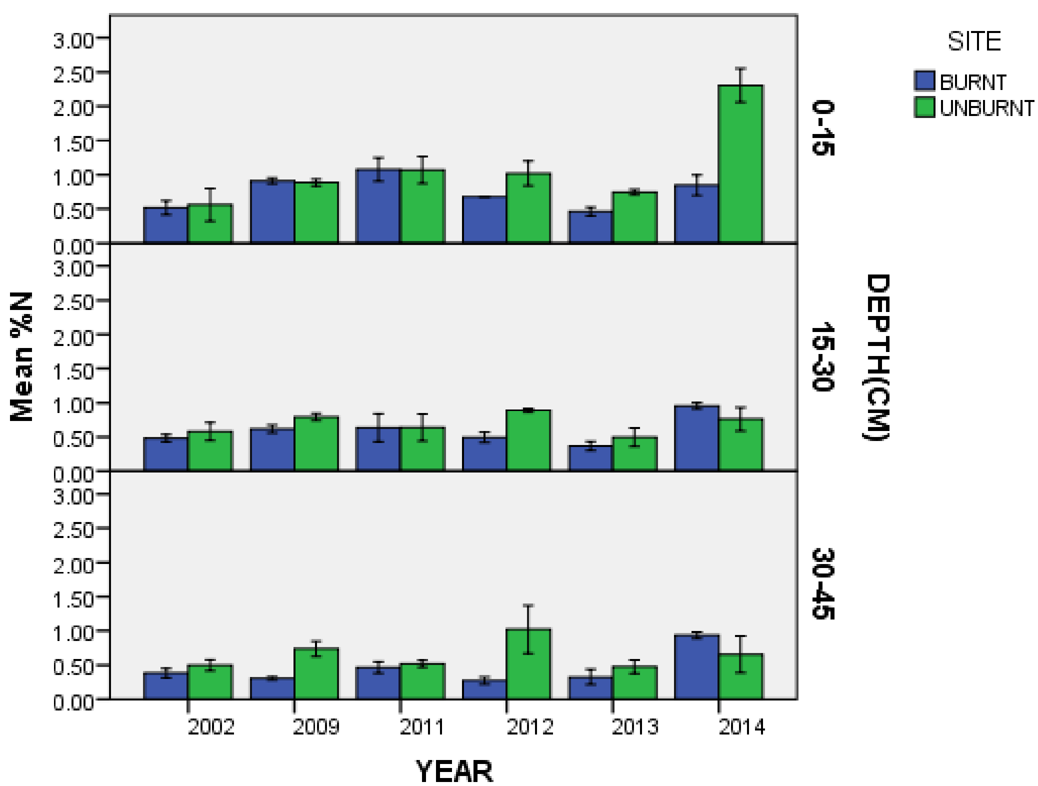

Figure 12: The nitrogen (%) of soils from all burnt and un-burnt sites at the 3 depth levels.

Univariate tests of nitrogen between depths showed that there was a significant difference between the three levels of depths ($F_{2, 51}$=7.819; P<0.05). After post hoc tests, the major difference was between 0-15cm and 30-45cm where the mean difference was 0.30% and P<0.05. Between 0-15cm and 15-30cm, the mean difference was 0.16 and P<0.05meaning there was a significant difference. Between 15-30cm and 30-45cm, the mean difference was 0.14 and P>0.05 meaning there was no significant difference.

There was a significant difference in the nitrogen between the sites burnt at different years ($F_{4, 40}$ =15.736; P<0.05). The maximum mean difference was between 2002 and 2014 which was 0.45±0.07 and the minimum mean difference was between 2002 and 2012 which was 0.02±0.07.

3.7.3 Organic Carbon

The mean values of the organic carbon in the soils from all the sites are shown in figure 12. The data shows that at 0-15 cm deep, the mean organic carbon of the burnt sites was higher in the sites burnt on 2014, 2013 and 2012 than in their un-burnt sites. The sites that were burnt in 2009 and 2002 had a lower mean organic carbon compared to the un-burnt site. The mean organic carbon at 0-15 cm of the burnt sites was 7.05±0.93 and that of the un-burnt sites was 6.07±1.59. At the depth of 0-15cm, there was a significant difference between the amount of organic carbon in the burnt sites and un-burnt sites ($F_{1, 34}$=30.085; P<0.05). All the values of organic carbon are in appendix 7.

At 15-30cm, the mean organic carbon of the sites burnt on 2014, 2013 and 2012 was higher than the un-burnt sites. The sites burnt on 2009 and 2002 had a lower mean organic carbon than in the un-burnt sites. The mean organic carbon of the burnt sites at this depth was 6.07±1.26% and that of the un-burnt sites was 5.60±1.52%. At this depth there was no significant difference in the amount of organic carbon between the burnt sites and the un-burnt sites ($F_{1, 34}$=0.343; P>0.05).

At 30-45cm, the mean organic carbon of the sites burnt on 2014 and 2012 was higher than the un-burnt sites. The sites burnt on 2009, 2002 and 2013 had a lower mean organic carbon than in the un-burnt sites. The mean organic carbon of the burnt sites was 5.19±1.94% while that of the un-burnt sites was 5.27±1.31%. The organic carbon at the depth of 30-45cm had no significant difference in the amount of organic carbon between the burnt sites and the un-burnt sites ($F_{1, 34}$=0.141; P>0.05).

Univariate tests of organic carbon between depths showed that there was a significant difference between the three levels of depths ($F_{2, 51}$=6.374; P<0.05). After post hoc tests, the major difference was between 0-15cm and 30-45cm where the mean difference was 1.64 and P<0.05. Between 0-15cm and 15-30cm, the mean difference was 0.93% and P<0.05meaning there was also a significant difference. Between 15-30cm and 30-45cm, the mean difference was 0.70% and P>0. 05 meaning there was no significant difference.

There was a significant difference in the organic carbon between the sites burnt at different years ($F_{4, 40}$ =3.151; P<0.05). The maximum mean difference was between 2002 and 2014 which was 1.76±0.69% and the minimum mean difference was between 2002 and 2013 which was 0.54±0.69%.

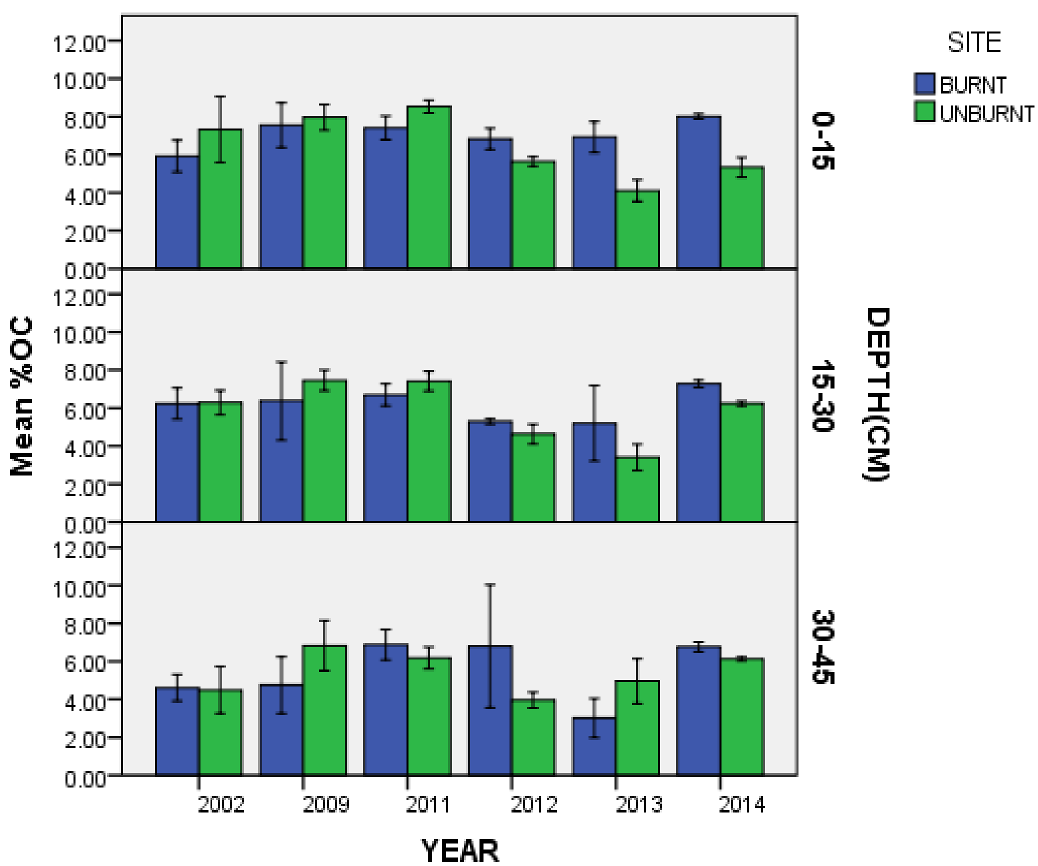

Figure 13: The Organic Carbon of soils from all burnt and un-burnt sites at different depths.

3.7.4 Potassium

The mean values of the potassium in the soils from all the sites are shown in figure 13. The data shows that at 0-15 cm deep, the mean potassium was higher in all the burnt sites than in their un-burnt sites apart from the site burnt in 2009. The mean potassium at 0-15 cm of the burnt sites was 2.68±1.78and that of the un-burnt sites was 1.14±0.61. There was a significant difference between the amount of potassium in the burnt sites and un-burnt sites at this depth ($F_{1, 34}$=13.320; P<0.05). All the values of potassium in the soil are shown in appendix 7.

At 15-30cm, the mean potassium of all the burnt sites was higher than the un-burnt sites. The mean potassium of the burnt sites at this depth was 1.58±0.98 Cmol/kg and that of the un-burnt sites was 1.21±0.81 Cmol/kg. At 15-30cm, there was no significant difference in the amount of potassium between the burnt sites and the un-burnt sites ($F_{1, 34}$=0.766; P>0.05).

At 30-45cm, the mean potassium of the sites burnt on 2014, 2012 and 2002 was higher than the un-burnt sites. The sites burnt on 2013 and 2009 had lower mean potassium than in the un-burnt sites. The mean potassium of the burnt sites was 1.31±0.97 Cmol/kg while that of the un-burnt sites was 1.03±0.61 Cmol/kg. At 30-45cm, there was also no significant difference in the amount of potassium between the burnt sites and the un-burnt sites ($F_{1, 34}$=1.659; P>0.05).

Univariate tests of potassium between depths showed that there was a significant difference between the three levels of depths ($F_{2, 51}$=5.951; P<0.05). After post hoc tests, the major difference was between 0-15cm and 30-45cm where the mean difference was 1.32 Cmol/kg and P<0.05. Between 0-15cm and 15-30cm, the mean difference was 1.04and P<0.05meaning there was also a significant difference. Between 15-30cm and 30-45cm, the mean difference was 0.28 and P>0.05 meaning there was no significant difference.

There was a significant difference in the potassium between the sites burnt at different years ($F_{4, 40}$ =15.697; P<0.05). The maximum mean difference was between 2002 and 2014 which was 2.71±0.43 Cmol/kg and the minimum mean difference was between 2002 and 2009 which was 0.46±0.43 Cmol/kg.

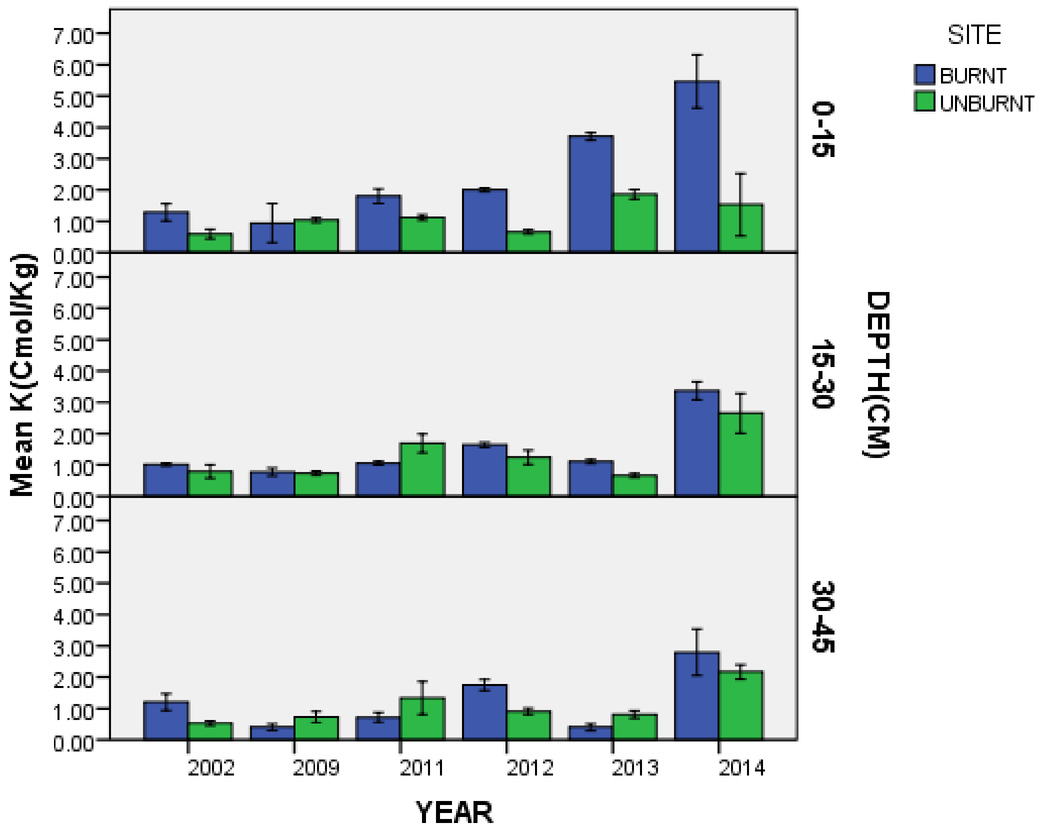

Figure 14: The Potassium of soils from all the burnt and un-burnt sites at different depths.

3.7.5 Phosphorous

The mean values of the phosphorous in the soils from all the sites are shown in figure 14. The data shows that at 0-15 cm deep, the mean phosphorous was lower in all the burnt sites than in their un-burnt sites apart from the site burnt in 2009. The mean phosphorous at 0-15 cm of the burnt sites was 8.26±2.49 ppm and that of the un-burnt sites was 13.32±3.83ppm. There was a significant difference in the amount of phosphorous between the burnt sites and un-burnt sites at this depth ($F_{1, 34}$=15.211; P<0.05). All the values of phosphorous in the soil are shown in appendix 7.

At 15-30cm, the mean phosphorous was lower in all the burnt sites than in their un-burnt sites apart from the site burnt in 2009. The mean phosphorous at 15-30 cm of the burnt sites was 7.59±3.86 and that of the un-burnt sites was 10.11±2.89 ppm. There was no significant difference in the amount of phosphorous between the burnt sites and the un-burnt sites ($F_{1, 34}$=3.55; P>0.05).

At 30-45cm, the mean phosphorous of the sites burnt on 2014, 2012 and 2002 was lower than the un-burnt sites. The sites burnt on 2013 and 2009 had higher mean phosphorous than in the un-burnt sites. The mean phosphorous of the burnt sites was 8.49±3.46ppm while that of the un-burnt sites was 9.01±1.72 ppm. At 30-45cm, there was also no significant difference in the amount of phosphorous between the burnt sites and the un-burnt sites ($F_{1, 34}$=0.011, P>0.05).

Univariate tests of phosphorous between depths showed that there was no significant difference between the three levels of depths ($F_{2, 51}$=0.208; P> 0.05).

There was a significant difference in the phosphorous between the sites burnt at different years ($F_{4, 40}$ =39.477; P<0.05). The maximum mean difference was between 2002 and 2014 which was 5.93±0.73ppm and the minimum mean difference was between 2002 and 2009 which was 0.14±0.73 ppm.

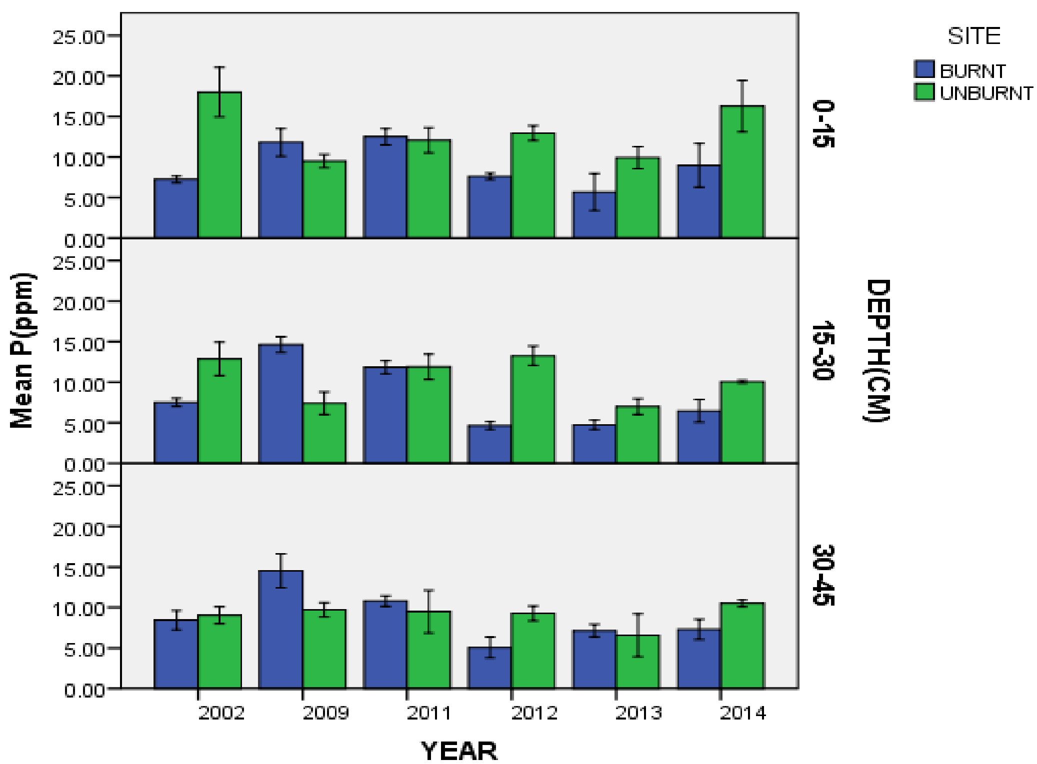

Figure 15: The phosphorous of soils from all the burnt and un-burnt sites at different depths.

3.7.6 Cation Exchange Capacity (CEC)

The mean values of the CEC in the soils from all the sites are shown in figure 15. The data shows that at 0-15 cm deep, the mean CEC was higher in all the burnt sites than in their un-burnt sites apart from the site burnt in 2002. The mean CEC at 0-15 cm of the burnt sites was 23.09±3.78 mol/kg and that of the un-burnt sites was 20.52±3.64 mol/kg. There was no significant difference in CEC between the burnt sites and un-burnt sites at this depth ($F_{1, 34}$=2.204; P>0.05). All the values of CEC in the soil are shown in appendix 7.

At 15-30cm, the mean CEC was higher in all the burnt sites than in their un-burnt sites apart from the site burnt in 2002. The mean CEC at 15-30 cm of the burnt sites was 20.16±2.78 mol/kg and that of the un-burnt sites was 18.34±3.41 mol/kg. The CEC at the depth of 15-30cm did not have a significant difference between the burnt sites and un-burnt sites ($F_{1, 34}$=1.523; P>0.05).

At 30-45cm, the mean CEC of the sites burnt on 2014, 2012 and 2002 was higher than the un-burnt sites. The sites burnt on 2013 and 2009 had lower mean CEC than in the un-burnt sites. The mean CEC of the burnt sites was 18.91±3.90 mol/kg while that of the un-burnt sites was 18.28±3.39 mol/kg. The CEC at the depth of 30-45cm did not have a significant difference between the burnt and the un-burnt sites ($F_{1, 34}$=0.675; P>0.05).

Univariate tests of CEC between the three levels of depths showed that there was a significant difference in the CEC between depths ($F_{2, 51}$ =5.648; P<0.05). After post hoc tests, the major difference was between 0-15cm and 30-45cm where the mean difference was 3.88 and P<0.05.There was also a significant difference between 0-15cm and 15-30cm, the mean difference was 2.691 and P=0.027. There was no significant difference between 15-30cm and 30-45cm where the mean difference was 1.19 mol/kg and P>0.05.

There was a significant difference in the CEC between the sites burnt at different years ($F_{4, 40}$ =6.983; P<0.05). The maximum mean difference was between 2002 and 2014 which was 5.29±1.46 mol/kg and the minimum mean difference was between 2002 and 2012 which was 0.09±1.46 mol/kg.

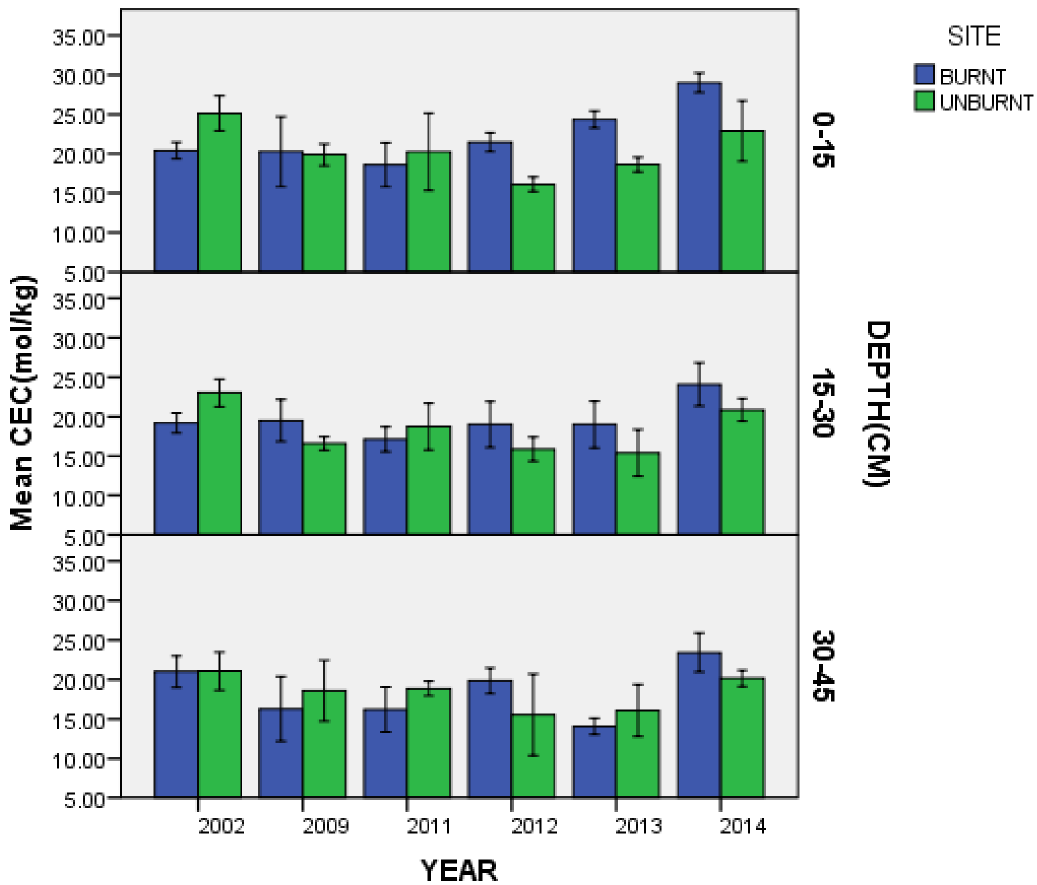

Figure 16: The CEC of soils from all the burnt and un-burnt sites at different depths.

CHAPTER FOUR
4.0Discussions

4.1 The effect of fire on plant species diversity

4.1.1 Woody vegetation

The results indicated that fires did not have major negative effect on the species diversity of the woody species. This may be because most fires experienced at Aberdare Ranges are usually ground fires that do not totally burn down the woody trees but the barks and sometimes small canopies of the woody trees are burnt. Consequently there was no difference in the species diversity of the woody species between the burnt sites and the un-burnt sites. Very small woody plants and seedlings are the only ones usually burnt down by the fires occurring at the Aberdares. The large trees found in Aberdares are resilient to ground fires and are not much affected by the fires like the way herbaceous vegetation is usually affected.

The results agreed with the studies that indicate that the effects of fire on woody vegetation depend on the frequency and the intensity of the fire (lehmann *et al*, 2014). Intensity could be low at Aberdares such that they do not have much effect on woody species diversity.

However, there was significant difference in DBH between the woody species in the burnt sites and the un-burnt sites with the un-burnt sites having a higher DBH than that of the burnt sites. All the five burnt sites had a lower mean DBH than their corresponding un-burnt sites meaning that DBH takes long to increase. DBH of a tree is important in estimating the amount of timber volume in a tree and also in estimating the age of a tree since diameter increment is the only constant non reversible feature of a tree growth (White, 1998). This means that the fires at Aberdares led to decrease in the timber volume of the trees and also could lead to a wrong estimation of the age of the trees at Aberdares.

The height and the canopies of the trees were also smaller in all the burnt sites than the un-burnt sites. There was no any burnt site that the height and canopy had recovered. This means that the fires at Aberdares had negative effect on both height and canopy.

The negative effect of fire on DBH, height and canopy of trees affects the carbon storage ability of tree since according to Waring and Pitman, 1985, carbon is allocated preferentially to the new leaves and roots and then to storage and stem diameter growth. This means that the fires at

Aberdares contribute to global warming, not only through the carbon dioxide that is emitted to the atmosphere during the fires, but also by reducing the carbon storage.

4.1.2 Herbaceous plants

The data showed that fire causes decrease in the herbaceous plant species diversity but one year later, burnt sites had totally regenerated the species diversity of the burnt areas were diverse than the un-burnt areas. The species diversity of the herbaceous plants in the site that was burnt in 2014 was less compared to the un-burnt site of the same year. The fire burnt almost all herbaceous plants and only a few species which include *Micromera imbricata*, *Oplisnemus compositus* and *Erica arborea* which had been partially burnt were found. Fire tends to favor species that can tolerate heat stress. Regeneration had not yet occurred in this site and this indicated that it takes about one year for regeneration to occur.

Immediately after a fire, the percentage of bare ground is increased which makes the land prone to erosion depending on its topography and weather. The percentage of the bare ground of the site that had been burnt most recently (2014) was 57%.

In all the other sites other than the one that was burnt in 2014, the species diversity of the herbaceous plants on the burnt sites was higher than on the un-burnt sites. This was an indication that fire triggered regeneration of herbaceous plants. It took less than one year for the burnt areas to regenerate as the site that had been burnt one year ago had totally recovered and the species diversity was higher than the un-burnt site.

The statistical analysis showed that there was a significant difference ($p<0.05$) in the vegetation cover between the burnt sites and the un-burnt sites. There was also significant difference ($p<0.05$) in the height of the vegetation between the burnt sites and the un-burnt sites. In the sites that had recently been burnt, the height of the vegetation was shorter than the height of the vegetation in their corresponding un-burnt sites. In the sites where vegetation had already recovered, the height of the vegetation was even more than the un-burnt sites. Fires triggered the regrowth of the herbaceous vegetation and after one year, the vegetation was taller in the burnt areas other than the site burnt in 2014.

Fires also had the same effect on the number of individuals as the cover and height. The sites that had already recovered had a higher number of individuals compared to the un-burnt areas. Only the sites that were recently burnt had fewer numbers of individuals in the burnt sites than in the un-burnt sites.

This was an indication that fire has a positive long term effect to the herbaceous vegetation. It triggers regeneration of pioneer species whose seeds could be lying dormant in the soil or on the soil surface.

4.2 Effects of fires on animals

4.2.1 Effects of fires on mammals

Fire had negative immediate effect to the animals. The burnt sites that had not yet regenerated had fewer animals compared to the un-burnt sites. No single animal was found in the site that was burnt on 2014 as the data was collected one week after the fire. Animals had emigrated out of that area as a result of the fire since there was no vegetation that could support the herbivores that support the rest of the food chain. No dead bodies of animals were found at the site meaning that no animal succumbed to the fire.

The site that was burnt on 2013 also had fewer animals compared to the un-burnt site of the same year. This is because the vegetation cover of that burnt site was less than the un-burnt site and animals would prefer an area with higher vegetation cover where there are more resources. Individuals remaining in the burned forest face a different set of problems. In all the other sites which had been burnt more than one year ago, there were more animals compared to the un-burnt sites. This was as a result of more vegetation which had regenerated after the fires. The areas with more vegetation had more animals.

This was an indication that fires have an immediate negative effect on the wildlife, which in turn can affect the tourism of the country considering that wildlife is one of major attraction to tourists. Fire lead to emigration of the wildlife from the burnt areas to the un-burnt areas which will lead to completion of resources in the un-burnt areas leading to exclusion of the weaker wildlife. This ends up in reduction of wildlife population.

4.2.2 Effects of fires on birds

Fire had negative effect on birds. All the burnt sites had fewer birds than the un-burnt sites. The site burnt in 2014 had the least number of birds. The number of the birds in the other sites burnt in 2013, 2012, 2009 and 2002 had increased although they were still fewer than the un-burnt sites. This could be as a result of reduced canopy of the trees in all the burnt sites as vegetation structure and floristic composition influence the availability of food, risk of predation and availability of nest sites.

Aberdares is an Important Bird Areas where many visitors go for bird watching. Fires can lead to negative impacts to the bird watching as a result of decline in bird's diversity which can have a negative effect to the country's economy.

4.3 Effects of fire on soil properties

The results showed that the fire affected mostly the upper layer; 0-15cm. very few effects were observed at 15-30cm and no adverse effects were obtained at 30-45cm depth. This probably because most fires that occur at Aberdares are not very severe and cannot exert effects beyond 15cm deep as the effects of fires depend on the fire severity which is determined by the intensity and the duration of fire.

4.3.1 pH

Data indicated that there was significant difference in soil pH ($p<0.05$) at 0-15cm and 15-30cm. There was no significant difference at 30-45cm. pH was higher in the burnt sites that had recently been burnt i.e. sites burnt less than two years ago compared to the un-burnt sites. After two years, there was no much difference in pH between burnt sites and un-burnt sites. This indicated that fire resulted into an increase in pH and it took almost two years for the pH to revert to its initial levels.

The increase in pH could be as a result of ash accretion. According to Cartini (2005) the response depends on the amount of ash and buffering capacity of the soil. This rise in P^H is because mineral substances are released as oxides or carbonates that usually have an alkaline reaction.

4.3.2 Nitrogen

The data showed that soil nitrogen was significantly lower in the recently burnt sites than in the un-burnt sites in 0-15cm depth. Though the soils in the burnt areas had higher nitrogen within

15-30cm depth, the concentrations were not significantly different. Sites that had experienced fire before 2013 had similar soil nitrogen content between burnt and un-burnt sites for all depths. There was no difference in soil nitrogen concentration between burn and un-burnt areas at 30-45cm for all fire episodes.

The decrease in nitrogen in the recently burnt sites is as a result of nitrogen volatilization caused by fire. Nitrogen has low temperature thresholds and is easily volatilized. Significant losses of nitrogen during the fires could adversely affect long-term site productivity in Aberdare forest ecosystem, particularly if nitrogen replenishment mechanisms are not provided for during post fire management. Nitrogen is considered the most limiting nutrient in wild land ecosystems and as such it requires special consideration when managing fire, particularly in nitrogen-deficient ecosystems (Maars *et al*., 1983)

4.3.3 Organic carbon

The organic carbon was high in the burnt sires than in the un-burnt sites and this was more evident at 0-15cm where there was significant difference ($p<0.05$). There was no significant difference in organic carbon at 15-30cm and 15-30cm ($P>0.05$). This is a result of rapid decomposition of organic matter on the soil during burning releasing organic carbon to the underlying soil (Certini, 2005).

The findings were similar to that of a study by Bird and others, 2000 which was carried out on tropical savanna sites in Africa. Low frequency burning resulted in increase in soil carbon of about 10%

The effect of organic carbon by a fire is determined by the severity of a fire. Fire severity affects the amount of organic matter that is lost. On a study carried out by Groeschl and others, 1990 they found out that areas burned by a low-severity fire, the forest floor Oi and Oe layers were completely combusted, but the Oa layer remained. High-severity burning also consumed the Oa layer. Of the 10.1 tons/acre (22.6 Mg/ha) of Carbon present in the forest floor in the unburned areas, no Carbon remained in the high-severity burned areas compared to 9.3 tons/ acre (20.8 Mg/ha) Carbon that was left on the burned areas at low severities.

4.3.4 Potassium

The result showed that potassium was mostly affected at 0-15cm as there was significant difference ($p<0.05$) between the burnt sites and the un-burnt sites. The differences were mainly on the sites that had been burnt recently but the sites that were burnt more than two years ago didn't have much difference between burnt and un-burnt sites. At 15-30cm and 30-45cm, there was no significant difference ($p>0.05$) between burnt and un-burnt sites.

Increase in potassium in the burnt sites could be as a result of rapid decomposition of organic matter during burning due to increased temperatures releasing nutrients to the soil. Organic matter acts as the primary reservoir for several nutrients and, therefore, is the source for most of potassium (Certini, 2005).

4.3.5 Phosphorous

Phosphorous at 0-15cm was significantly lower ($p<0.05$) in the burnt sites than in the un-burnt sites. At 15-30cm and 30-45cm, there was no significant difference between the burnt and un-burnt sites. The differences were also observed in the sites that had been burnt less than two years ago but in the sites burnt more than two years ago didn't have much difference between the burnt and un-burnt sites.

Just like nitrogen, phosphorous has low temperature thresholds and is easily volatilized. This led to the decrease in the amount of phosphorous in the burnt sites.

4.3.6 Cation Exchange Capacity

There was no significant difference in Cation Exchange Capacity ($p>0.05$) at all layers between burnt sites and un-burnt sites. As much as fire can make water soluble cations available for plant uptake, light burns like the ones experienced at Aberdares do not affect the exchange system.

CHAPTER FIVE
5.0Conclusion and recommendation

5.1 conclusions
Fires occur every year at Aberdares during the dry season mainly due to arsonist when they want to ferry poles from the forest, accidents during honey harvesting as the harvesters use fire to drive bees away, charcoal burning and clearing of the forest to create farmlands. The fire spreads fast as the vegetation is usually dry and there are strong winds which make the fire spread fast. These fires affect the vegetation, wildlife, birds and soil nutrients of the underlying soil.

The fires occurring at Aberdares do not affect the woody tress species diversity as only the DBH, height and canopy of trees are affected. The trees are not usually fully burnt. The fires lead to decrease in DBH, height and canopy which take long to recover as even the site that was burnt more than ten years ago had not yet recovered. In all the burnt sites, the DBH, height and canopy of the burnt sites was lower than the un-burnt sites. This has a negative effect to the forest as it reduces the amount of carbon storage.

Fires have an immediate negative effect on the herbaceous vegetation but then leads to a positive effect after the vegetation regenerates. For the site that had been burnt recently (2014), the fire led to a negative effect on the herbaceous vegetation and the vegetation cover was reduced leading to an increase in bare ground which is prone to erosion. It takes one year for the herbaceous vegetation to regenerate. The sites that were burnt on 2013, 2012, 2009 and 2002 had more herbaceous vegetation and a higher cover than their un-burnt sites. The cover, height and the number of plants increases with time with the site burnt on 2002 having the highest and that burnt on 2014 having the lowest.

Animals are also affected by the fires. No animal was counted in the site that was burnt on 2014. Decrease in vegetation led to reduction in the wildlife population and diversity. Wildlife population and diversity tends to increase again after the vegetation regenerates. The sites burnt on 2013, 2012, 2009 and 2002 had more wildlife than the un-burnt sites since vegetation had regenerated. Loss of vegetation after a fire leads to emigration of wildlife to the un-burnt sites. The un-burnt site of 2014 had the highest number of animals which could have emigrated from the burnt site.

Birds are negatively affected by fires. All the sites burnt had fewer birds than the un-burnt sites. This is as a result of reduced canopy of the trees in the burnt forests. Immediately after a fire, the number of birds utilising that area reduces drastically as a result of reduced amount of food and destruction of their nests.

Fires also have an effect on the soil properties investigated: pH, nitrogen, potassium, phosphorous, organic carbon and cation exchange capacity. It leads to an increase in pH, potassium, organic carbon and cation exchange capacity due to rapid decomposition of organic matter and leads to reduction of nitrogen and phosphorous since these two elements are easily volatilized at low temperature threshold.

Despite the immediate negative effects of fires on species diversity and some soil properties, prescribed burning can be adopted by ecosystem managers where controlled fires can be used to burn an ecosystem so as to increase the herbaceous vegetation. Burning leads to an increase in herbaceous vegetation after a period of about one year.

5.2 Recommendations

The study showed that fires lead to an immediate negative effect on vegetation, wildlife and soil's chemical properties hence post fire management is important on sites that have been recently burnt so as to rehabilitate them. Since the reduced herbaceous vegetation makes it easy for the humans to access the forest, this can lead to further destruction of the forest and the forest guards should ensure that people are kept out of those sites so that the vegetation can regenerate without interference. Reforestation can also be done on the burnt sites so as to increase vegetation and habitat for the wildlife.

One way of minimizing the occurrences of these fires is by increasing public awareness about fire ecology and potential damages of uncontrolled forest fires. Depending on the ecological environment, there is a need to distinguish between good and damaging fires. Most local people living around the Aberdares depend on the forest for their livelihood and some of them end up logging *Juniperous procera* for selling. This is a major cause of fires at Aberdares as they burn the forest so that they may get a way of ferrying the logs out of the forest as the guards put off the fire. The local people can be provided with other means of earning a livelihood so as to prevent then from burning the forests.

Forest managers can apply prescribed burning to forest or grassland ecosystems where there would increase the herbaceous vegetation cover as the results showed that fires can lead to increase in the cover after sometime. This would require control measures to avoid the fire getting out of control.

Although little can be done to control organic matter loss during fires, every opportunity must be taken to revegetate the site so that organic litter can again be restored on the site as quickly as possible. This should be done on sites that have recently been burnt as they are the ones with low vegetation cover.

Nitrogen is one of the important nutrients likely to be lost during a fire due to volatilization during combustion of organic matter. Consideration should be given to its replenishment when planning post fire management. Nitrogen fixing capacity of some forest soils can be provided by microorganisms responsible for decaying wood on the surface and in the soil profile; thus, management of woody residues in a burnt site may be an important dimension of nitrogen management.

Other recommendations include:

- Increase participation of stakeholders especially the local people in the prevention and management of forest fires.
- KFS should improve documentation and recording of fire events to improve the understanding of forest fire and to research strategies for fire management.
- Increase law enforcement. Many countries have legislation that governs the use of fire, mainly for agriculture. The laws define techniques for fire prevention, institutional arrangements to obtain fire permits, as well as penalties, but the level of enforcement is generally low, and the implementation of some laws is controversial.
- Encourage interagency collaboration. Increased collaboration among responsible authorities such as KFS, KWS, NGOs and the community living around the forests would increase the effectiveness of fire control and prevention. In order to facilitate the cooperation, standardized protocols and operational procedures should be developed.

- Establish clear and transparent institutional responsibilities for government agencies to define who is responsible for fire prevention, monitoring, and suppression.

REFERENCES

Aerial survey of the destruction of the Aberdare Range forests. Report by UNEP, KWS, Rhino Ark, KFWG. April 2003.

Amiro, B. D., B. J. Stocks, M. E. Alexander, M. D. Flannigan, and B. M. Wotton. 2001. Fire, climate, carbon and fuel management in the Canadian boreal forest. Int. J. Wildland Fire 10:405–413.

Ark, R. 2011. Environmental, Social and Economic Assessment of the fencing of the Aberdare Conservation Area. *UNEP, Nairobi, Kenya*.

Arocena JM, Opio C .2003. Prescribed fire-induced changes in properties of sub-boreal forest soils. Geoderma 113:1–16

Balch, J. K., Nepstad, D. C., Brando, P. M., Curran, L. M., Portela, O., De carvalho, O. & Lefebvre, P. 2008. Negative fire feedback in a transitional forest of southeastern Amazonia. *Global Change Biology* 14:2276–2287.

Balch, J. K., Nepstad, D. C., Curran, L. M., Brando, P. M., Portela, O., Guilherme, P., ... & de Carvalho, O. 2011. Size, species, and fire behavior predict tree and liana mortality from experimental burns in the Brazilian Amazon. *Forest Ecology and Management*, *261*(1), 68-77.

Barlow, J., & Peres, C. A. 2008. Fire-mediated dieback and compositional cascade in an Amazonian forest. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *363*(1498), 1787-1794.

Bierregaard, R. O. & Lovejoy, T. E. 1989. Effects of forest fragmentation on Amazonian understorey bird communities. *Acta Amazon.* **19**, 215–241.

Black, C. A. 1965. Methods Of Soil Analysis Part1And2.

Blake, J. G. 2005. Effects of prescribed burning on distribution and abundance of birds in a closed-canopy oak-dominated forest, Missouri, USA. Biological Conservation 121:519–531.

Botkin, D. B. 1990. *Discordant harmonies: a new ecology for the twenty-first century*. Oxford University Press.

Bowman, D. M., Balch, J. K., Artaxo, P., Bond, W. J., Carlson, J. M., Cochrane, M. A., ... & Pyne, S. J. 2009. Fire in the Earth system. *science*, *324*(5926), 481-484.

Brando, P. M., Nepstad, D. C., Balch, J. K., Bolker, B., Christman, M. C., Coe, M., & Putz, F. E. 2012. Fire-induced tree mortality in a neotropical forest: the roles of bark traits, tree size, wood density and fire behavior. *Global Change Biology*, *18*(2), 630-641.

Brooks, M. L., C. M. D'Antonio, D. M. Richardson, J. B. Grace, J. E. Keeley, J. M. Ditomaso, R. J. Hobbs, M. Pellant. 2004. Effects of invasive alien plants on fire regimes. BioScience 54:677–688.

Brown, Arthur A.; Davis, Kenneth P. 1973. Forest fire: Control and use. New York: McGraw-Hill Book Company. 686 p.

Butynski, T. 1999. Aberdares National Park and Aberdares Forest Reserves Wildlife Fence Placement Study and Recommendations. Africa Biodiversity Conservation Programme. Zoo Atlanta.

Certini, G. 2005. Effects of fire on properties of forest soils: a review. *Oecologia*, *143*(1), 1-10.

Chen Z. 2006. Effects of fire on major forest ecosystem processes.

Costanza, R, R. d'Arge, R de Groot, S.Farber, M.Grasso, B.Hannon, K.Limburg, S.Naeem, R.O'Neill, J.Paruelo, R.Raskin, P.Sutton and M van den Belt, 1997. The value of the world's ecosystem services and natural capital. Nature 387:253-260.

Chou, YH. 1992. Management of wildfire with GIS. *Int J Geogr Inf Syst.*, 6(2): 123–140.

Chuvieco, E. 2009. *Earth observation of wild land fires in Mediterranean ecosystems*, 1st, Dordrecht, The Netherlands: Springer.

Cochrane, M. A. 2003. Fire science for rainforests. Nature 421: 913–919.

Cochrane, M. A. and M. D. Schultz. 1999. Fire as a recurrent event in tropical forests of the eastern Amazon: effects of forest structure, biomass, and species composition. Biotropica 31:2–16.

Cochrane, M. A. and W. F. Laurance. 2008. Synergisms among fire, land use, and climate change in the Amazon. Ambio 37:522–527.

Costanza, R., & Folke, C. 1997. Valuing ecosystem services with efficiency, fairness, and sustainability as goals. *Nature's services: societal dependence on natural ecosystems. Island Press, Washington, DC*, 49-70.

Cottle, P. 2007. Insuring Southeast Asian commercial forests: fire risk analysis and the potential for use of data in risk pricing and reduction of forest fire risk. *Mitig Adapt Strat Glob Change.*, 12: 181–201.

Covington WW, Sackett SS .1992. Soil mineral nitrogen changes following prescribed burning in ponderosa pine. For Ecol Manage 54:175–191

Cox, G.W., 1990. Laboratory Manual of General Ecology, 6th edition. WM C. Brown, Publishers, Dubuque, Iowa

Cumming, S. G. 2001. Forest type and wildfire in the Alberta boreal mixed wood: what do fires burn. Ecol. Appl. 11:97–110.

D'Antonio, C. M. and P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Ann. Rev. Ecol. System. 23:63–87.

DeBano, L.F.; Neary, D.G.; Ffolliott, P.F. 1998. Fire's effects on ecosystems. New York: John Wiley & Sons, Inc. 333 p.

DeBell, D. S., & Ralston, C. W. 1970. Release of nitrogen by burning light forest fuels. *Soil Science Society of America Journal*, *34*(6), 936-938.

Dwyer, E., J. M. Gregoire, and J. P. Malingreau. 1998. A global analysis of vegetation fires using satellite images: spatial and temporal dynamics. Ambio 27:175–181.

Dwyer, E., S. Pinnock, J. M. Gregoire, and J. M. C. Pereira. 2000. Global spatial and temporal distribution of vegetation fire as determined from satellite observations. Int. J. Remote Sens 21:1289–1302.

Finch, D. M., J. L. Ganey, W. Yong, R. T. Kimball, and R. Sallabanks. 1997. Effects and interactions of fire, logging, and grazing. Pages 103–136 *in* W. M. Block and D. M Finch, editors. Songbird ecology in Southwestern ponderosa pine forests: a literature review. U.S. Forest Service General Technical Report RMRS-292, Fort Collins, Colorado, USA.

Fisher, R. F., & Binkley, D. 2000. Ecology and management of forest soils. John Willey and Sons. *Inc. New York, USA.*

Flinn, M. A., & Wein, R. W. 1977. Depth of underground plant organs and theoretical survival during fire. *Canadian Journal of Botany*, *55*(19), 2550-2554.

Flora da Silva Ramos Vieira Martins, Haron Abrahim Magalhães Xaud, João Roberto dos Santos and Lênio Soares Galvão.2012. Effects of fire on aboveground forest biomass in the northern Brazilian Amazon. Journal of Tropical Ecology, 28, pp 591-601

Gerwing, J. J. 2002. Degradation of forests through logging and fire in the eastern Brazilian Amazon. *Forest ecology and management*, *157*(1), 131-141.

Glogiewicz, J. and J. Báez. 2001. Wildland Fire Dynamics in Puerto Rico: A Report on Incidence, Cause and Danger, with Emphasis on the Wildland-Urban Interface Unpublished report. San Juan, Puerto Rico. 8 pp., 3 appendices

Goldammer, J. G. 1990. Fire in tropical biota: ecosystem processes and global challenges. Springer-Verlag, Berlin.

Gonzalez, JR, Palahi, M and Pukkala, T. 2005. Integrating fire risk considerations in forest management planning in Spain – a landscape level perspective. *Landsc Ecol.*, 20: 957–970.

Grier, C. C. 1975. Wildfire effects on nutrient distribution and leaching in a coniferous ecosystem. *Canadian Journal of Forest Research*, *5*(4), 599-607.

Groeschl, D. A., Johnson, J. E., & Smith, D. W. 1990. Forest soil characteristics following wildfire in the Shenandoah National Park, Virginia. In *Fire and Environment: Ecological and Cultural Perspectives: Proceedings of an International Symposium. Edited by SC Nodvin and TA Waldrop. USDA For. Ser. Gen. Tech. Rep. SE-69* (pp. 129-137).

Hauer, F.R.; Spencer, C.N.1998 Phosphorus and nitrogen dynamics in streams associated with wildfire: A study of immediate and longterm effects. *Int. J. Wildland Fire 8*, 183–198.

Haugaasen, T., Barlow, J. & Peres, C. A. 2003. Surfacewildfires in central Amazonia: short-term impact on forest structure and carbon loss. *Forest Ecology andManagement* 179:321–331.

Heilman, P. E., & Gessel, S. P. 1963. The effect of nitrogen fertilization on the concentration and weight of nitrogen, phosphorus, and potassium in Douglas-fir trees. *Soil Science Society of America Journal*, *27*(1), 102-105.

Hildén, O. 1965. Habitat selection in birds: a review. In *Annales Zoologici Fennici* (pp. 53-75). Societas Zoologica Botanica Fennica Vanamo.

Huston, M. A. 1994. Biological diversity. Cambridge Uni- versity Press, Cambridge, UK.

Ingersoll, Cheryl A.; Wilson, Mark V. 1990. Buried propagules in an old-growth forest and their response to experimental distur- bances. Canadian Journal of Botany. 68: 1156-1162.

John, T. V. S., & Rundel, P. W. 1976. The role of fire as a mineralizing agent in a Sierran coniferous forest. *Oecologia*, *25*(1), 35-45.

Jones, N. 2005. Wildland fires, management and restoration (in Barbados). In: Wildfire Management and Restoration: Proceedings of the Twelfth Meeting of Caribbean Foresters in Puerto Rico. Weaver, P. L. and K. A. González, editors. USDA Forest Service. Río Piedras, Puerto Rico. pp. 3–5.

Kasischke, E. S., N. L. Christensen Jr., and B. J. Stock. 1995. Fire, global warming, and the carbon balance of boreal forests. Ecol. Appl. 5:437–451.

Kevin. M. 2007. Quantitative Analysis by the Point-Centered Quarter Method, Department of

Mathematics and Computer Science Hobart and William Smith Colleges Geneva, NY 1445

Knoepp, J. D., & Swank, W. T. 1994. Long-term soil chemistry changes in aggrading forest ecosystems. *Soil Science Society of America Journal*, *58*(2), 325-331.

Kotliar, N. B., S. Hejl, R. L. Hutto, V. A. Saab, C. P. Melcher, and M. E. McFadzen. 2002. Effects of fire and post-fire salvage logging on avian communities in conifer-dominated forests of the western United States. Studies in Avian Biology 25:49–64.

Kozlowski, T. T. 2002. Physiological ecology of natural regeneration of harvested and disturbed forest stands: implications for forest management. *Forest ecology and management*, *158*(1), 195-221.

Lambert, F. V. & Collar, N. J. 2002 The future for Sundaic lowland forest birds: long-term effects of commercial logging and fragmentation. *Forktail* **18**, 127–146.

Lambrechts, C., Woodley, B., Church, C., & Gachanja, M. 2003. Aerial survey of the destruction of the Aberdare Range forests. *Division of Early Warning and Assessment, UNEP*.

Lehmann, C. E., Anderson, T. M., Sankaran, M., Higgins, S. I., Archibald, S., Hoffmann, W. A., ... & Bond, W. J. 2014. Savanna vegetation-fire-climate relationships differ among continents. *Science*, *343*(6170), 548-552.

Lesica, P. 1996. Using fire history models to estimate proportions of old growth forest in northwest Montana, USA. Biological Conservation 77:33–39.

Long, DG, Morgan, P, Hardy, CC, Swetnam, TW and Rollins, MG. 2001. Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. *Int J Wildland Fire.*, 10(4): 329–342.

Lovejoy, T. 1989: The Third World's environment: a global dilemma. EPA Journal.15(4): 42.

Maars, R.H.; Roberts, R.D.; Skeffinton, R.A.; Bradshaw, A.D. 1983. Nitrogen in the development of ecosystems. In: Lee, J.A.; McNeill, S.; Rorison, I.H., eds. Nitrogen as an ecological factor. Oxford, England: Blackwell Science Publishing: 131-137.

Mannan, R. W., E. C. Meslow, And H. M. Wight. 1980. Use of snags by birds in Douglas-fir forests, western Oregon. J. Wildl. Manage. 44:787-797.

Martin, T.E. 1998. Are habitat preferences of coexisting species under selection and adaptive? Ecology 79:656-70

Mayer, J.H., 1989: Socioeconomic aspects of the forest fire 1982/83 and the relation of local communities towards forestry and forest management. FR Report NO.9

Mehlich, A. 1984. Mehlich-3 soil test extractant: a modification of Mehlich-2 extractant. Commun. Soil Sci. Plant Anal. 15(12): 1409-1416.

Miller, A. C. 2007. Fire History of Caribbean Pine (Pinus caribaea var. bahamensis [Griseb.] W.H. Barret & Golfari) Forests on Abaco Island, the Bahamas MS research paper,. Department of Geography, University of Tennessee. Knoxville, Tennessee. 92 pp.

Morgan, P., Aplet, G. H., Haufler, J. B., Humphries, H. C., Moore, M. M., & Wilson, W. D. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. *Journal of Sustainable forestry*, *2*(1-2), 87-111.

Myers, R., D. Wade, and C. Bergh. 2004. Fire Management Assessment of the Caribbean Pine (Pinus caribea) Forest Ecosystems on Andros and Abaco Islands, Bahamas. Global Fire Initiative Publication 2004–1 The Nature Conservancy. Arlington, Virginia. 21 pp.

Myers, R. L. 2006. Living with Fire—Sustaining Ecosystems & Livelihoods through Integrated Fire Management The Nature Conservancy Global Fire Initiative. Tallahassee, Florida. 36 pp

Neary, D.G.; Klopatek, C.C.; DeBano, L.F.; Ffolliott, P.F.1999. Fire effects on belowground sustainability: A review and synthesis. *For. Ecol. Manag.*, *122*, 51–71.

Neary, Daniel G.; Ryan, Kevin C.; DeBano, Leonard F., eds. 2005. (revised 2008). Wildland fire in ecosystems: effects of fire on soils and water. Gen. Tech. Rep. RMRS-GTR-42-vol.4.

Ng'ang'a, E. M., Kamande, L. M. 1990: The Vegetation of the Aberdares Mountain Ranges. Department of Resource Surveys and Remote Sensing, Ministry of Planning and National Development. Nairobi, Kenya.

Peech, M. 1965. Hydrogen-ion activity. *Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties*, (methodsofsoilanb), 914-926.

Peres, C. A. 1999. Ground fires as agents of mortality in a central Amazonian forest. *J. Trop. Ecol.* **15**, 535–541.

Peres, C. A., Barlow, J. & Haugaasen, T. 2003. Vertebrate responses to surface fires in a central Amazonian forest. *Oryx* **37**, 97–109.

Raison RJ, Khanna PK, Woods PV .1985. Mechanisms of element transfer to the atmosphere during vegetation fires. Can J For Res 15:132–140

Reich, P. B., D. W. Peterson, D. A. Wedin, and K. Wrage. 2001. Fire and vegetation effects on productivity and nitrogen cycling across a forest-grassland continuum. Ecology 82:1703–1719.

Reiners, W. A., A. F. Bouwman, W. F. J. Parsons, and M. Keller. 1994. Tropical rainforest conversion to pasture: changes in vegetation and soil properties. Ecol. Appl. 4:363–377.

Reside, A. E., VanDerWal, J., Kutt, A., Watson, I. and Williams, S. 2012. Fire regime shifts affect bird species distributions. Diversity and Distributions, 18: 213–225.

R. H. Waring and G. B. Pitman. 1985. Modifying Lodgepole Pine Stands to Change Susceptibility to Mountain Pine Beetle Attack. Ecology 66:889–897

Rossiter, N. A., S. A. Setterfield, M. M. Douglas, and L. B. Hutley. 2003. Testing the grass-fire cycle: alien grass invasion in the tropical savannas of Northern Australia. Divers. Distrib 9:169–176.

Russell, R. E., Royle, J. A., Saab, V. A., Lehmkuhl, J. F., Block, W. M., & Sauer, J. R. 2009. Modeling the effects of environmental disturbance on wildlife communities: avian responses to prescribed fire. *Ecological Applications*, *19*(5), 1253-1263.

Saab, V., W. Block, R. Russell, J. Lehmkuhl, L. Bate, and R. White. 2007. Birds and burns in the interior West: descriptions, habitats, and in Western forests. U.S. Forest Service General Technical Report, PNW-712, Portland, Oregon, USA.

Sanchez-Azofeifa, G. A., C. Quesada-Mateo, P. Gonzales-Quesada, S. Dayanandan, and K. S. Bawa. 1999. Protected areas and conservation of biodiversity in the tropics. Conservation Biology 13:407–411.

Simard, A.J. 1991. Fire severity, changing scales, and how things hang together. International Journal of Wildland Fire. 1: 23–34.

Slik, J. F., Bernard, C. S., Van Beek, M., Breman, F. C., & Eichhorn, K. A. 2008. Tree diversity, composition, forest structure and aboveground biomass dynamics after single and repeated fire in a Bornean rain forest. *Oecologia*, *158*(3), 579-588.

Slik, J. W. F., Aiba, S. I., Brearley, F. Q., Cannon, C. H., Forshed, O., Kitayama, K., ... & van Valkenburg, J. L. 2010. Environmental correlates of tree biomass, basal area, wood specific gravity and stem density gradients in Borneo's tropical forests. *Global Ecology and Biogeography*, *19*(1), 50-60.

Smith, J. K., Lyon, L. J., Huff, M. H., Hooper, R. G., Telfer, E. S., & Schreiner, D. S. 2000. Wildland fire in ecosystems. Effects of fire on fauna. *General Technical Report-Rocky Mountain Research Station, USDA Forest Service*, (RMRS-GTR-42).

Soto B, Diaz-Fierros F .1993. Interactions between plant ash leachates and soil. Int J Wildland Fire 3:207–216

St. John, T.V.; Rundel, P.W. 1976. The role of fire as a mineralizing agent in the Sierran coniferous forest. Oecologia. 25: 35–45.

Sturtevant, B. R., Scheller, R. M., Miranda, B. R., Shinneman, D., & Syphard, A. (2009). Simulating dynamic and mixed-severity fire regimes: a process-based fire extension for Landis-II. *Ecological Modelling*, *220*(23), 3380-3393.

Sutherland, W. J. 1996. *From individual behaviour to population ecology* (Vol. 11). Oxford University Press.

Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9:1189–1206.

Trappe, J. M., & Bollen, W. B. 1979. *Forest soil biology*. Washington State University, Cooperative Extension Service.

Ulery AL, Graham RC, Amrhein C .1993. Wood-ash composition and soil pH following intense burning. Soil Sci 156:358–364

Uhl, C. and J. B. Kauffman. 1990. Deforestation, fire susceptibility, and the potential tree responses to fire in the Eastern Amazon. Ecology 7:437–449.

Wade, D. D. (1993). Thinning young loblolly pine stands with fire. *International Journal of Wildland Fire*, *3*(3), 169-178.

Waithaka J. M. 1994. Monitoring human-elephant conflict through remotely located stations.Pachyderm, 27, pp66 –68.

Waring, R. H., & Pitman, G. B. 1985. Modifying lodgepole pine stands to change susceptibility to mountain pine beetle attack. *Ecology*, 889-897.

White, E.M.; W.W. Thompson, F.R. 1973. Gartner Heat effects on nutrient release from soils under ponderosa pine. Journal of Range Management, 26, pp. 22–24

White, j. 1998. Estimating the age of large and veteran trees in Britain. Forestry Commision Information Note 12. Surrey.

Wilson, C. W., R. E. Master, and G. A. Bukenhofer. 1995. Breeding bird response to pine-grassland community restoration for red-cockaded woodpeckers. Journal of Wildlife Management 59:56–67.

Woinarski, JCZ and Recher, HF.1997. Impact and Response: A Review of the Effects of Fire on the Australian Avifauna. Pacific Conservation Biology, Vol. 3, No. 3: 183-205.

Wright, Henry A.; Bailey, Arthur W. 1982. Fire ecology United States and southern Canada. New York: John Wiley & Sons. 501 p.

APPENDICES

Appendix 1: The structural analysis of the woody vegetation of all the sites.

Study site	Species	Mean P-P distance(M) (X±s.d)	Mean DBH(M) (X±s.d)	Mean height(M) (X±s.d)	Mean canopy(M) (X±s.d)	Number of individuals	r.d(%)	density	Dominance	r.D(%)	frequency	r.f(%)	I.V
2002 Burnt	*Olea Africana*	6.8± 1.9	0.6± 0.2	28± 4.6	5.4± 1.1	21	17.5	26.16	9.365	16.35	0.5	17.43	51.28
	Podocarpus falcatus	7.3± 2.8	0.9± 0.2	26± 4.9	5.1± 0.8	18	15	22.43	14.265	24.9	0.37	12.89	52.79
	Markhamia lutea	8±4.8	0.4± 0.1	24± 5.1	4± 1.1	18	15	22.43	7.088	12.37	0.57	19.86	47.23
	Ocotea usambarensis	9.4± 4.2	0.5± 0.2	19± 5.1	3.4± 0.8	20	16.7	24.97	8.889	15.51	0.43	14.98	47.19
	Juniperus procera	9.3± 3.7	0.5± 0.2	27± 5.3	5.8± 1.0	18	15	22.43	8.232	14.37	0.43	14.98	44.35
	Warbugia ugandensis	8.3± 3.9	0.6± 0.2	22± 5.2	4.5± 0.9	25	20.8	31.1	9.454	16.5	0.57	19.86	57.16
2002 Un-burnt	*Olea Africana*	5.6± 1.7	0.5± 0.1	26± 5.1	5.2± 0.8	19	15.8	23.63	8.632	13.42	0.42	15	44.22
	Podocarpus falcatus	7.2± 2.9	0.7± 0.2	29± 4.6	5.4± 1.1	22	18.4	27.36	10.485	16.3	0.36	12.86	47.56
	Markhamia lutea	6.9± 2.1	0.6± 0.2	25± 4.8	4.6± 0.9	20	16.7	24.97	9.625	15	0.5	17.86	49.56
	Ocotea usambarensis	8.4± 3.6	0.8± 0.1	20± 5.1	4.1± 1.1	18	15	22.43	12.432	19.33	0.46	16.43	50.76
	Juniperus procera	6.4± 1.8	0.5± 0.1	27± 5.3	5.3± 1.2	22	18.3	27.36	8.466	13.17	0.54	19.29	50.76
	Warbugia ugandensis	8.7± 4.3	0.9± 0.2	24± 5.1	5.1± 0.8	19	15.8	23.63	14.658	22.78	0.52	18.56	57.14

Study site	Species	distance (M) (X±s.d)	DBH(M) (X±s.d)	Mean height(M) (X±s.d)	Mean canopy(M) (X±s.d)	Number of individuals	r.d(%)	density	Dominance	r.D(%)	frequency	r.f(%)	I.V
2009 Burnt	*Olea Africana*	4.7± 1.8	0.7± 0.2	30± 4.5	5.3± 1.1	18	15	22.43	11.282	16.2	0.46	16.14	47.34
	Nuxia congesta	10.2± 3.6	0.3± 0.1	18± 5.1	3.1± 0.7	7	5.8	8.72	6.462	9.3	0.2	7.01	22.11
	Ocotea usambarensis	8.2± 4.6	0.5± 0.2	20± 5.3	3.2± 0.8	19	15.8	23.67	8.654	12.5	0.51	17.89	46.19
	Podocarpus falcatus	4.3± 1.5	0.8± 0.2	24± 5.1	4.8± 0.9	17	14.3	21.18	11.897	17.2	0.48	16.84	48.34
	Markhamia lutea	7.5± 2.2	0.5± 0.1	25± 4.7	4.2± 1.2	18	15	22.43	8.756	12.7	0.38	13.33	41.03
	Juniperus procera	6.8± 1.9	0.6± 0.2	20± 4.9	5.4± 1.1	16	13.3	19.94	9.613	13.8	0.29	10.18	37.28
	Warbugia ugandensis	8.9± 4.8	0.8± 0.2	20± 5.1	4.3± 0.8	25	20.8	31.1	12.527	18.1	0.53	18.7	57.6
2009 Un-burnt	*Olea Africana*	4.5± 1.8	0.8± 0.2	32± 4.4	5.6± 1.2	19	15.8	23.32	12.362	16.6	0.48	15.9	48.3
	Nuxia congesta	9.7± 3.7	0.4± 0.1	19± 5.1	3.3± 0.8	8	6.7	9.53	8.538	11.5	0.31	10.3	28.5
	Ocotea usambarensis	8.3± 4.6	0.5± 0.1	20± 5.2	3.4± 0.8	17	14.2	21.18	8.892	11.9	0.48	15.9	42
	Podocarpus falcatus	4.5± 1.9	0.8± 0.2	23± 5.1	4.8± 0.9	18	15	22.43	11.953	16.1	0.57	18.9	50
	Markhamia lutea	7.2± 2.8	0.6± 0.2	26± 4.9	4.5± 0.9	15	12.5	18.72	9.625	12.9	0.44	14.6	40
	Juniperus procera	7.1± 2.1	0.7± 0.2	25± 5.1	5.6± 1.1	17	14.2	21.18	11.348	15.2	0.32	10.6	40
	Warbugia ugandensis	8.2± 4.3	0.8± 0.2	23± 5.2	4.6± 0.9	26	21.6	32.35	11.726	15.8	0.41	13.8	51.2

Study site	Species	Mean P-P distance (M) (X±s.d)	Mean DBH (M) (X±s.d)	Mean height (M) (X±s.d)	Mean canopy (M) (X±s.d)	Number of individuals	r.d(%)	density	Dominance	r.D(%)	frequency	r.f(%)	I.V
2012 Burnt	*Olea Africana*	5.2± 2.2	0.6± 0.2	28± 4.6	5.2± 0.8	22	18.4	27.36	9.765	18.3	0.52	16.1	52.8
	Nuxia congesta	10.1± 3.6	0.2± 0.1	20± 5.1	3.2± 0.7	10	8.3	12.41	5.264	9.8	0.32	9.9	28
	Ocotea usambarensis	8.6± 4.5	0.5± 0.1	18± 5.1	3.1± 0.9	16	13.4	19.94	8.912	16.8	0.53	16.5	46.7
	Podocarpus falcatus	4.5± 2.4	0.6± 0.2	25± 4.9	4.6± 0.9	18	15.1	22.43	9.536	17.8	0.48	14.9	47.8
	Markhamia lutea	7.3± 2.8	0.5± 0.2	23± 5.1	4.3± 1.2	21	17.6	26.16	5.264	9.8	0.53	16.5	43.9
	Juniperus procera	8.8± 4.6	0.5± 0.2	20± 5.2	5.1± 0.8	13	10.5	15.61	5.314	9.9	0.38	11.8	32.2
	Warbugia ugandensis	8.5± 4.2	0.6± 0.2	19± 5.4	3.9± 0.8	20	16.7	24.97	9.428	17.6	0.46	14.3	48.6
2012 Un-burnt	*Olea Africana*	4.8± 2.1	0.8± 0.2	30± 4.5	5.4± 1.1	20	16.7	24.98	11.875	15.9	0.53	16.2	48.8
	Nuxia congesta	8.9± 3.4	0.3± 01	22± 5.2	3.5± 0.8	12	10	14.87	6.534	8.8	0.33	10.1	28.9
	Ocotea usambarensis	8.4± 3.2	0.6± 0.2	20± 5.3	3.6± 0.8	18	15	22.43	9.852	13.2	0.53	16.2	44.4
	Podocarpus falcatus	4.2± 1.7	0.8± 0.3	25± 4.9	4.8± 0.9	20	16.7	24.98	11.875	15.9	0.46	14.1	46.7
	Markhamia lutea	6.9± 2.6	0.6± 0.2	24± 5.1	4.5± 0.9	22	18.3	27.35	9.785	13.1	0.52	15.9	47.3
	Juniperus procera	8.5± 3.8	0.7± 0.2	21± 5.2	5.5± 1.1	16	13.3	19.94	12.324	16.5	0.4	12.2	42
	Warbugia ugandensis	8.4± 3.1	0.7± 0.3	21± 5.2	4.6± 0.8	12	10	14.87	12.324	16.6	0.5	15.3	41.9

Study site	Species	Mean P-P distance(M) (X±s.d)	DBH(M) (X±s.d)	Mean height(M) (X±s.d)	Mean canopy(M) (X±s.d)	Number of individuals	r.d(%)	density	Dominance	r.D(%)	frequency	r.f(%)	I.V
2013 Burnt	*Olea Africana*	5.2± 2.1	0.4± 0.1	26± 5.3	5± 0.7	22	18.6	27.45	8.538	16.1	0.54	18.8	53.5
	Podocarpus falcatus	6.1± 2.8	0.5± 0.2	25± 4.9	3.6± 0.8	21	17.9	26.36	8.912	16.7	0.36	12.5	47.1
	Markhamia lutea	7.5± 2.9	0.4± 0.1	22± 5.2	4.1± 1.1	23	19.8	29.21	8.537	16	0.57	19.9	55.7
	Ocotea usambarensis	8.1± 3.5	0.5± 0.2	20± 5.3	2.8± 0.7	19	16	23.52	8.924	16.7	0.43	15	47.7
	Juniperus procera	8.7± 3.6	0.4± 0.2	20± 5.3	4.7± 0.9	10	7.7	11.41	8.641	16.2	0.45	15.7	39.6
	Warbugia ugandensis	6.7± 1.9	0.6± 0.2	18± 5.4	3.5± 0.8	25	20	29.42	9.785	18.3	0.52	18.1	56.4
2013 Un-burnt	*Olea Africana*	4.5± 1.8	0.6± 0.2	27± 5.3	5.5± 1.1	21	17.2	26.54	9.854	15.9	0.55	19.1	52.2
	Podocarpus falcatus	5.8± 2.6	0.5± 0.2	26± 5.2	4.2± 0.9	20	16.5	25.42	8.924	14.4	0.37	12.8	43.7
	Markhamia lutea	6.4± 1.8	0.5± 0.2	24± 5.1	4.8± 1	22	17.9	27.63	8.924	14.4	0.58	20.1	52.4
	Ocotea usambarensis	7.3± 2.8	0.7± 0.3	22± 5.2	3.4± 0.8	18	14.5	22.41	12.324	19.8	0.41	14.3	48.6
	Juniperus procera	7.8± 2.6	0.6± 0.2	21± 5.2	5.2± 0.9	16	15.3	23.61	9.785	15.7	0.42	14.6	45.6
	Warbugia ugandensis	6.1± 2.4	0.7± 0.2	20± 4.8	4.1± 1.1	23	18.5	28.53	12.324	19.8	0.55	19.1	57.4

Study site	Species	Mean P-P distance (M) (X±s.d)	Mean DBH (M) (X±s.d)	Mean height (M) (X±s.d)	Mean canopy (M) (X±s.d)	Number of individuals	r.d(%)	density	Dominance	r.D(%)	frequency	r.f(%)	I.V
2014 Burnt	*Olea Africana*	5.8± 2.2	0.5± 0.2	24± 5.1	2.1± 0.6	22	17.6	27.63	8.924	18.3	0.54	19.6	55.5
	Nuxia congesta	6.2± 1.8	0.4± 0.1	22± 5.2	1.6± 0.5	19	14.9	23.42	7.521	15.4	0.48	17.4	47.7
	Ocotea usambarensis	6.5± 1.6	0.4± 0.1	21± 5.2	1.8± 0.4	21	16.9	26.54	7.523	15.4	0.36	13	45.3
	Podocarpus falcatus	6.1± 2.2	0.3± 0.1	23± 5.1	1.4± 0.3	17	15.4	24.15	6.425	13.2	0.43	15.6	44.2
	Markhamia lutea	6.6± 2.1	0.3± 0.1	20± 5.2	1.3± 0.3	20	16.1	25.31	6.425	13.2	0.45	16.3	45.6
	Juniperus procera	6.8± 1.9	0.2± 0.1	18± 5.2	1.1± 0.2	5	4.1	6.33	5.438	11.3	0.18	6.5	21.9
	Warbugia ugandensis	7.1± 2.8	0.3± 0.1	19± 5.3	1.5± 0.3	16	15	23.62	6.425	13.2	0.32	11.6	39.8
2014 Un-burnt	*Olea Africana*	4.2± 1.6	0.7± 0.2	25± 4.9	5.2± 0.8	18	15	22.43	12.324	16.5	0.56	20.7	52.2
	Nuxia congesta	5.1± 3.2	0.6± 0.2	24± 5.1	4.6± 0.9	19	15.6	23.42	9.785	13.1	0.45	16.7	45.4
	Ocotea usambarensis	5.4± 2.6	0.7± 0.3	22± 5.2	5.1± 0.8	15	12.5	18.72	12.335	16.5	0.32	11.9	40.9
	Podocarpus falcatus	4.8± 2.1	0.5± 0.1	25± 4.9	4.8± 0.9	20	16.9	25.31	8.924	12	0.36	13.3	42.2
	Markhamia lutea	4.7± 2.2	0.6± 0.2	21± 5.2	3.9± 1.1	16	13.3	19.94	9.841	13.2	0.38	14.1	40.6
	Juniperus procera	5.3± 2.6	0.7± 0.2	20± 5.2	4.2± 1.1	14	11.7	17.54	12.324	16.5	0.32	11.9	40.1
	Warbugia ugandensis	6.4± 2.1	0.5± 0.2	21± 5.1	5.4± 1.2	18	15	22.67	8.972	12.2	0.31	11.4	38.6

Appendix 2: A list of all the herbaceous species found in all the sites and the families that they belong to.

Family	**Species**
Amaranthaceae	*Achyranthus aspera*
Apiaceae	*Caucalis incognita*
	Ferula communis
	Heracleum abyssinicum
Apocynaceae	*Gomphocarpus stenophyllus*
Asteraceae	*Berkheya spekeana*
	Cardius keniensis
	Crepis aurea
	Dichrocephala chrysanthemifolia
	Gnaphilium purpureum
	Helichrysum odoratissmum
	Vernomia altissima
Crassulaceae	*Kalanchoe millottii*
Dennstaedtiaceae	*Pterridium aquilinum*
Ericaceae	*Erica arborea*
Euphorbiaceae	*Acalypha padifolia*
Fabaceae	*Crotalaria paniculata*
	Indigofera arrecta
	Kotschya recurvifolia
	Psoralea foliosa
	Rhynchosia minima
	Trifolium cryptopodium
	Trifolium tembense
	Adenocarous manii
Geraniaceae	*Geranium occelatum*
Hypericaceae	*Hypericum lanceolatum*
	Hypericum revolutum
Lamiaceae	*Leucas venulosa*
	Micromeria imbricate
Peraceae	*Cluria robusta*
Poaceae	*Enneapogon branchystachys*
	Oplismenus compositus
	Panicum virgatum
Primulaceae	*Myrsine Africana*
Ranunculaceae	*Delphinium macrocentron*

Rhamnaceae	*Rhamnus prinoides*
Rosaceae	*Alchemilla gracilipes*
Rubiaceae	*Pentas lanceolata*
Thymelaeaceae	*Lasiosiphon glaucus*

Appendix 3: The structural analysis of herbaceous vegetation of all sites

Study site	Species	Mean% cover± s.d	Total No of individual	Mean Height(cm)± s.d	Density	r.d (%)	frequency	r.f (%)
2002 Burnt	*Pteridium aquilinum*	5.50±0.27	20	131.11±11.35	2.22	1.12	0.30	6.45
	Hypericum lanceolati	11.75±1.05	104	163.50±28.32	10.40	5.22	0.33	7.10
	Achyranthus aspera	17.43±1.86	345	99.22±10.54	19.16	9.62	0.60	12.90
	Myrsine africana	13.76±1.74	181	101.13±20.57	11.31	5.68	0.53	11.40
	Heracleum abyssinicum	4.13±0.69	22	59.60±8.76	4.40	2.21	0.17	3.66
	Berkheya spekeanna	3.07±0.71	31	64.00±8.59	6.20	3.11	0.17	3.66
	Geranium ocellatum	6.37±0.41	29	77.13±9.26	3.22	1.62	0.30	6.45
	Cluria robusta	2.00±0.36	38	58.25±10.32	9.50	4.77	0.13	2.80
	Acalypha padifolia	2.00±0.45	26	104.80±10.65	5.20	2.61	0.17	3.66
	Vernomia altissima	1.40±0.72	10	177.00±15.89	2.50	1.26	0.13	2.80
	Micromeria imbricate	3.38±1.02	21	86.67±10.35	5.25	2.64	0.13	2.80
	Panicum virgatum	6.06±0.57	109	80.20±17.34	21.8	10.95	0.17	3.66
	Helichysum odoratissinum	1.13±0.65	23	75.00±11.54	7.67	3.85	0.10	2.15
	Indigofera arrecta	2.73±0.51	26	69.00±9.67	6.50	3.27	0.13	2.80
	Oldenlardic fastigicts	1.86±0.76	26	66.25±9.82	6.50	3.27	0.13	2.80
	Anailema leocaule	1.03±0.00	15	59.00±10.44	15.00	7.53	0.03	0.65
	Enneapogon branchystachyes	1.00±0.00	20	91.50±12.87	10.00	5.02	0.07	1.51
	Kotschya recurvifolia	4.80±0.03	83	93.50±15.64	20.75	10.42	0.13	2.80

Study site	Species	Mean% cover± s.d	Total No of individual	Mean Height(cm)± s.d	Density	r.d (%)	frequency	r.f (%)
2002 Burnt	*Kalachoe millotii*	2.83±0.24	69	82.00±14.58	23.00	11.55	0.10	2.15
	Hypericum revolutum	2.53±0.37	34	70.50±8.96	8.50	4.27	0.13	2.80
	Bare ground	5.24±0.26	0	0	0.00	0.00	0.70	15.05
2009 Burnt	*Pteridium aquilinum*	17.23±2.47	79	135.00±21.57	3.16	3.39	0.83	17.40
	Hypericum lanceolati	9.40±1.68	164	80.06±20.14	9.64	10.35	0.57	11.95
	Achyranthus aspera	36.10±3.57	859	91.50±20.28	35.79	38.42	0.80	16.77
	Myrsine africana	12.63±1.87	224	66.25±15.86	11.20	12.02	0.67	14.05
	Cluria robusta	1.47±0.31	16	12.20±2.85	3.20	3.43	0.17	3.56
	Acalypha padifolia	4.67±0.59	17	86.11±14.35	1.89	2.03	0.30	6.29
	Vernomia altissima	4.13±0.44	30	179.16±19.24	5.00	5.37	0.20	4.19
	Kalachoe millotii	2.97±0.32	104	57.43±8.97	14.86	15.95	0.23	4.82
	Crotalaria paniculata	5.90±0.51	101	46.42±10.82	8.42	9.04	0.40	8.39
	Bare ground	5.50±0.57	0	0	0.00	0.00	0.60	12.58
2012 Burnt	*Pteridium aquilinum*	4.60±0.61	20	119.78±23.61	2.22	1.52	0.30	5.58
	Myrsine africana	0.90±0.17	9	40.50±8.56	2.25	1.54	0.13	2.42
	Heracleum abyssinicum	6.80±1.26	73	31.65±10.24	6.08	4.17	0.40	7.43
	Berkheya spekeanna	0.47±0.09	4	46.00±9.73	2.00	1.37	0.07	1.30

Study site	Species	Mean% cover± s.d	Total No of individual	Mean Height(cm)± s.d	Density	r.d (%)	frequency	r.f (%)
2012 Burnt	*Geranium ocellatum*	2.73±0.27	32	34.14±7.67	4.57	3.13	0.23	4.28
	Micromeria imbricate	25.97±2.89	344	36.16±12.34	13.76	9.44	0.83	15.43
	Indigofera arrecta	2.53±0.14	18	34.50±8.49	3.00	2.06	0.20	3.72
	Oldenlardic fastigicts	6.10±0.47	66	36.54±10.21	6.00	4.11	0.37	6.88
	Anailema leocaule	1.97±0.07	31	27.00±8.97	15.50	10.63	0.07	1.30
	Enneapogon branchystachyes	3.00±0.19	22	37.50±8.17	5.50	3.77	0.13	2.42
	Kotschya recurvifolia	0.33±0.00	8	40.00±7.19	8.00	5.49	0.03	0.56
	Kalachoe millotii	0.77±0.04	10	43.50±6.81	5.00	3.43	0.07	1.30
	Hypericum revolutum	0.17±0.00	2	26.00±1.54	2.00	1.37	0.03	0.56
	Trifillium cryptopodium	0.27±0.00	3	19.00±3.17	3.00	2.06	0.03	0.56
	Rhamnus prihoides	2.63±0.31	27	41.75±7.64	6.75	4.63	0.13	2.42
	Peritaschistis borrusiaca	1.13±0.06	9	34.67±7.61	3.00	2.06	0.10	1.86
	Caucalis incognita	0.15±0.00	1	32.00±0.00	1.00	0.69	0.03	0.56
	Dichrocephala chysanthamifolia	2.10±0.09	17	35.50±8.61	4.25	2.91	0.13	2.42
	Erica arborea	5.67±0.43	22	109.55±20.14	2.00	1.37	0.37	6.88
	Ferula communis	0.77±0.00	10	35.00±8.63	10.00	6.86	0.03	0.56
	Leucas venulosa	1.40±0.05	18	33.67±9.47	6.00	4.11	0.10	1.86
	Gnaphalium purpureum	0.97±0.03	15	48.00±10.25	7.50	5.14	0.07	1.30
	Cardius keniensis	1.77±0.08	21	32.25±8.56	5.25	3.60	0.13	2.42

Study site	Species	Mean% cover± s.d	Total No of individual	Mean Height(cm)± s.d	Density	r.d (%)	frequency	r.f (%)
2012 Burnt	*Hebelishrahia dentate*	0.13±0.00	6	30.00±7.27	6.00	4.11	0.03	0.56
	Adenocarpus manii	0.57±0.00	8	33.50±8.91	8.00	5.49	0.03	0.56
	Gomophorcapus stenophyllus	0.37±0.05	3	39.50±5.46	1.50	1.03	0.07	1.30
	Crepis aurea	0.43±0.04	3	33.00±4.89	1.50	1.03	0.07	1.30
	Psoralea foliosa	0.47±0.00	2	52.00±8.95	2.00	1.37	0.03	0.56
	Delphinium macrocera	1.10±0.06	11	38.20±7.29	2.20	1.51	0.17	3.16
	Bare ground	23.73±4.62	0	0	0.00	0.00	1.00	18.59
2013 Burnt	*Pteridium aquilinum*	20.47±3.87	79	110.20±16.35	3.16	2.32	0.83	14.93
	Hypericum lanceolati	3.17±0.29	27	28.50±5.87	4.50	3.31	0.20	3.60
	Achyranthus aspera	5.37±1.24	135	15.92±4.12	10.38	7.63	0.43	7.73
	Myrsine africana	15.27±2.74	188	22.86±7.36	8.95	6.58	0.70	12.59
	Acalypha padifolia	5.30±1.83	178	27.29±8.92	12.71	9.34	0.47	8.45
	Trifillium cryptopodium	1.27±0.04	8	19.50±5.64	4.00	2.94	0.07	1.26
	Erica arborea	9.76±1.52	91	119.35±20.15	5.35	3.93	0.57	10.25
	Cardius keniensis	0.40±0.00	1	124.00±18.48	1.00	0.73	0.03	0.54
	Pentas lanceolere	6.37±1.36	118	47.29±9.63	16.86	12.39	0.23	4.14
	Alchemilla gracilipes	5.37±1.45	62	23.38±7.28	7.75	5.69	0.27	4.86
	Trifolium tembese	10.30±2.01	249	15.17±5.26	20.75	15.25	0.40	7.19
	Rhynchosis minima	6.63±1.68	187	18.23±6.58	14.38	10.56	0.43	7.73
	Bare ground	10.32±2.14	0	0	0.00	0.00	0.93	16.73

Study site	Species	Mean% cover± s.d	Total No of individual	Mean Height(cm)± s.d	Density	r.d (%)	frequency	r.f (%)
2014 Burnt	*Micromeria imbricate*	12.6±1.56	204	44.20±8.94	13.6	6.35	0.50	17.67
	Erica arborea	15.14±1.45	48	119.78±21.36	2.67	1.25	0.60	21.20
	Oplisnemus compositus	16.13±1.83	221	15.29±4.29	10.05	4.69	0.73	25.80
	Bare ground	56.13±4.28	0	0	0.00	0.00	1.00	35.34
2002 Un-burnt	*Pteridium aquilinum*	7.78±2.43	28	122.54±14.62	2.15	1.00	0.43	8.88
	Hypericum lanceolati	8.41±1.59	48	194.43±18.59	6.85	3.20	0.23	4.75
	Achyranthus aspera	15.25±2.26	257	102.20±16.52	17.13	7.99	0.50	10.33
	Myrsine africana	9.79±1.25	112	104.00±17.85	9.33	4.35	0.40	8.26
	Heracleum abyssinicum	7.81±2.14	46	60.10±12.36	4.60	2.15	0.33	6.82
	Berkheya spekeanna	4.11±0.54	52	64.86±12.79	7.43	3.47	0.23	4.75
	Geranium ocellatum	10.00±1.06	41	95.70±13.25	3.15	1.47	0.43	8.88
	Cluria robusta	1.61±0.02	27	57.67±8.51	9.00	4.20	0.10	2.07
	Acalypha padifolia	2.66±0.13	34	100.29±11.34	4.86	2.27	0.23	4.75
	Vernomia altissima	2.00±0.21	12	175.00±18.37	2.00	0.93	0.20	4.13
	Micromeria imbricate	5.23±0.57	48	80.38±12.54	6.00	2.80	0.27	5.58
	Panicum virgatum	5.41±0.61	82	63.00±8.39	20.5	9.57	0.13	2.69
	helichysum odoratissimum	1.51±0.06	33	77.00±8.72	11.00	5.13	0.10	2.07
	Indigoffera arrecta	2.18±0.09	19	67.50±5.23	4.75	2.22	0.13	2.69
	Oldenlardic fastigicts	12.00±0.34	19	69.50±5.18	4.75	2.22	0.13	2.69
	Anailema leocaule	3.11±0.26	45	76.67±8.21	15.00	7.00	0.10	2.07

Study site	Species	Mean% cover± s.d	Total No of individual	Mean Height(cm)± s.d	Density	r.d (%)	frequency	r.f (%)
2002 Un-burnt	*Enneapogon branchystacheys*	1.50±0.00	10	90.00±8.74	10.00	4.67	0.03	0.62
	Kotschya recurvifolia	1.40±0.00	21	116.00±14.63	21.00	9.80	0.03	0.62
	Kalachoe milloti	2.83±0.31	69	82.33±9.52	23.00	10.73	0.10	2.07
	Hypericum revolutum	0.80±0.04	11	70.00±6.59	5.50	2.57	0.07	1.45
	Bare ground	5.51±1.56	0	0	0.00	0.00	0.67	13.84
2009 Un-burnt	*Hypericum lanceolati*	17.87±2.31	259	100.78±14.21	14.39	18.12	0.60	8.22
	Achyranthus aspera	25.63±3.69	403	104.40±13.56	20.15	25.37	0.67	9.18
	Myrsine africana	14.73±2.54	209	99.59±10.43	12.29	15.47	0.57	7.81
	Acalypha padifolia	17.50±2.36	256	102.53±10.76	15.06	18.96	0.57	7.81
	Kalachoe milloti	17.97±3.12	298	93.47±9.58	17.53	22.07	0.57	7.81
	Bare ground	6.30±1.34	0	0	0.00	0.00	0.67	9.18
2012 Un-burnt	*Pteridium aquilinum*	0.40±0.00	1	117.00±0.00	1.00	15.48	0.03	0.98
	Heracleum abyssinicum	1.27±0.08	17	48.67±5.32	5.67	6.44	0.10	3.28
	Micromeria imbricate	5.44±1.62	68	46.70±6.21	6.80	6.18	0.33	10.82
	Panicum virgatum	8.65±1.84	159	41.44±8.52	17.67	5.48	0.30	9.84
	helichysum odoratissimum	1.54±0.06	17	52.00±4.28	5.67	6.88	0.10	3.28
	Enneapogon branchystacheys	0.33±0.00	4	56.00±3.65	4.00	7.41	0.03	0.98
	Kotschya recurvifolia	7.67±1.22	104	53.80±9.62	10.40	7.12	0.33	10.82

Study site	Species	Mean% cover± s.d	Total No of individual	Mean Height(cm)± s.d	Density	r.d (%)	frequency	r.f (%)
2012 Un-burnt	*Hypericum revolutum*	19.03±1.41	277	54.90±10.89	27.70	7.26	0.33	10.82
	Caucaus incognita	0.17±0.00	2	40.00±3.65	2.00	5.29	0.03	0.98
	Erica arborea	34.00±2.14	68	113.18±15.39	4.00	14.97	0.57	18.69
	Psoralea foliosa	9.00±0.62	30	83.80±7.52	6.00	11.09	0.17	5.57
	Lasiosiphon glaucus	3.00±0.08	20	48.33±4.44	6.67	6.39	0.10	3.28
	Bare ground	9.50±1.54	0	0	0.00	0.00	0.63	20.66
2013 Un-burnt	*Pteridium aquilinum*	21.28±3.21	64	106.38±14.28	2.67	4.89	0.80	16.91
	Hypericum lanceolati	8.57±1.47	56	159.23±15.95	4.31	7.89	0.43	9.09
	Achyranthus aspera	13.06±1.87	287	77.94±10.24	16.88	30.89	0.57	12.05
	Myrsine Africana	9.06±0.78	114	78.00±8.26	6.71	12.28	0.57	12.05
	Cluria robusta	5.43±0.62	90	48.88±5.17	11.25	20.59	0.27	5.71
	Acalypha padifolia	9.20±1.26	83	77.38±7.12	6.38	11.68	0.43	9.09
	Clotararia paniculata	4.90±0.07	11	166.00±12.95	2.75	5.03	0.13	2.75
	Erica arborea	17.57±1.69	59	131.81±10.25	3.68	6.75	0.53	11.21
	Bare ground	10.93±4.25	0	0	0.00	0.00	1.00	21.14
2014 Un-burnt	*Myrsine Africana*	6.13±.064	92	61.00±6.18	15.33	13.04	0.20	6.01
	Micromeria imbricate	9.47±0.87	52	52.89±5.84	5.78	4.92	0.30	9.01
	Panicum virgatum	4.70±0.35	155	38.67±5.36	17.22	14.65	0.30	9.01
	helichysum odoratissimum	9.57±0.00	4	38.00±3.25	4.00	3.40	0.03	0.90
	Indigoffera arrecta	0.33±0.05	48	62.20±5.78	9.60	8.17	0.17	5.11

Study site	Species	Mean% cover± s.d	Total No of individual	Mean Height(cm)± s.d	Density	r.d (%)	frequency	r.f (%)
2014 Un-burnt	*Oldenlardic fastigicts*	3.82±0.09	28	62.33±4.21	9.33	7.94	0.10	3.00
	Enneapogon branchystacheys	1.30±0.07	58	72.50±7.12	14.50	12.34	0.13	3.90
	Kotschya recurvifolia	3.53±0.59	194	63.23±7.45	14.92	12.69	0.43	12.91
	Hypericum revolutum	13.08±1.24	228	36.79±5.12	16.29	13.86	0.47	14.11
	Caucaus incognita	15.50±0.00	2	27.00±2.56	2.00	1.70	0.03	0.90
	Erica arborea	0.17±0.08	64	108.79±10.28	4.57	3.89	0.47	14.11
	Psoralea foliosa	31.93±0.00	4	54.00±3.69	4.00	3.40	0.03	0.90
	Bare ground	0.47±0.07	0	0	0.00	0.00	0.67	20.12

Appendix 4: A list of animal species found in all sites and the families that they belong to.

Family	**species**
Bovidae	Bush buck (*Tragelaphus scriptus*)
	Duiker (*Cephalophus maxwellii*)
	eland (*Taurotragus oryx*)
	Water buck (*Kobus ellipsi prymnus*)
Cercopithecidae	Colobus monkey (*Colobus guereza*)
	Sykes monkey (*Cercopithecus mitis*)
Elephantidae	Elephants (*Loxodanta africana*)
Herpestidae	Jackson mongoose (*Bdeugale jacksoni*)
Suidae	Giant forest hog (*Hylochoenus meinertzhageni*)

Appendix 5: A list of bird species counted in all sites and the families that they belong to.

Family	**Species**
Accipitridae	African goshawk (*Accipiter tachiro*)
	Augur buzzard (*Buteo augur*)
Acrocephalidae	Mountain yellow wabler (*Iduna similis*)
	Icterine warbler (*Hippolais icterina*)
Apodidae	African black swift (*Apus barbatus*)
Cisticolidae	Singing cisticola (*Cisticola contans*)
	Pectoral patch cisticola (*Cisticola brunnescens*)
	Aberdare cisticola (*Cisticola aberdare*)
	Hunter cisticola (*Cisticola hunteri*)
	Puff bellied warbler (*Phyllolais pulchella*)
Estrildidae	Black headed waxbill (*Estrilda atricapilla*)
Hirundinidae	Swallow hirundo (*Hirundo rustica*)
Locustellidae	Grasshoper warbler (*locustella naevia*)
Nectariniidae	Scarlet Tufted Malachite Sunbird (*Nectarinia johnstoni)*
	White bellied sunbird (*Cinnyris talatala*)
	Scarlet chested sunbird (*Chalcomitra senegalensis*)
	Bronze sunbird (*Nectarinia kilimensis*)
	Olive sunbird (*Cyanomitra olivacea*)
	Collared sunbird (*Hedydipna collaris*)
	Golden winged sunbird (*Drepanorhynchus reichenowi*)
	Tiny sunbird (*Cinnyris minullus*)
	Purple brested sunbird (*Nectarinia purpureiventris*)
	Shining sunbird (*Cinnyriss habessinicus*)
Phylloscopidae	Willow warbler (*Phylloscopus trochilus*)
	Wood warbler (*Phylloscopus sibilatrix*)
Picidae	Brown backed woodpecker(*Dendropicos obsoletus*)
Platysteiridae	Forest batis (*Batis mixta*)
Ploceidae	Yellow crowned bishop (*Eupectes afer*)
	Black necked weaver (*Ploceus nigricollis*)
	Fox's weaver (*Ploceus spekeoides*)
	Red billed quelea (*Quelea quelea*)
Pycnonotidae	Icterine greenbul (*Phyllastrephus icterinus*)
	Red tailed leaf love (*Phyllastrephus scandens*)

Appendix 6: The list of all birds counted in all the sites.

Years	2002		2009		2012		2013		2014	
species	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt
African goshawk	10	6	0	8	12	0	0	0	2	0
Forest batis	11	6	2	7	0	0	0	0	0	0
Singing cisticola	42	19	58	32	18	37	44	74	0	17
Pectoral patch cisticola	13	19	8	20	0	7	0	4	0	7
S.T.M.S	32	53	16	34	10	18	30	45	6	20
Shining sunbird	19	12	20	17	16	14	15	25	0	6
Aberdare cisticola	40	42	4	26	37	33	3	20	12	41
Black headed waxbill	3	0	0	3	0	3	0	0	1	4
Icetrine greenbill	10	0	0	0	0	0	1	0	0	0
Leaf love	1	0	0	0	0	0	1	0	1	0
Mountain yellow wabler	21	23	12	14	10	3	8	12	6	10
Augur buzzard	13	6	0	4	7	9	2	0	10	15

Years	2002		2009		2012		2013		2014	
species	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt
Hunter cisticola	4	4	0	0	0	8	0	0	0	11
Yellow crowned bishop	6	10	0	6	10	7	0	4	0	8
Willow wabler	35	44	12	15	13	15	15	22	1	9
swallow hirundo	24	26	1	17	10	53	12	13	12	28
White bellied sunbird	3	10	0	3	6	3	2	5	0	0
Scarlet chested sunbird	2	4	0	0	8	0	0	0	2	5
Grasshoper wabler	1	0	0	0	13	0	0	0	0	3
Bronze sunbird	2	0	11	8	0	0	0	0	0	0
Puff bellied wabler	7	4	7	8	9	4	2	3	9	9
Olive sunbird	1	0	11	7	0	0	0	0	0	0
Collared sunbird	3	3	1	4	14	3	4	4	0	0
Brown backed woodpecker	4	6	2	0	8	3	4	5	0	0

Years	2002		2009		2012		2013		2014	
species	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt	burnt	un-burnt
African black swift	4	6	0	12	0	13	0	0	1	13
Icetrine wabler	0	4	0	8	8	11	0	0	0	9
Black necked weaver	0	10	2	0	5	3	2	5	0	0
Fox weaver	0	4	3	4	0	0	2	0	2	4
Golden winged sunbird	0	0	2	3	0	0	0	0	0	0
Tiny sunbird	0	0	1	0	0	3	2	10	0	0
Red billed quelea	0	0	0	0	2	0	0	0	0	0
Wood wabler	0	0	0	0	0	4	0	0	0	4
Purple brested sunbird	0	0	0	0	0	3	0	0	0	0

Appendix 7: The mean values of different soil properties at different depths.

STUDY SITE	DEPTH (CM)	PH	%N	%OC	K(Cmol/Kg)	P(ppm)	CEC (mol/kg)
2002 BURNT	0-15	4.6	0.52	5.93	1.29	7.24	20.4
	15-30	4.93	0.48	6.23	1	9.16	17.53
	30-45	5.48	0.39	4.61	1.21	8.43	21
2002 UN-BURNT	0-15	5.44	0.56	7.34	0.6	18	25.13
	15-30	5.7	0.58	6.95	0.79	12.87	23.13
	30-45	4.82	0.5	4.82	0.53	9.03	21.07
2009 BURNT	0-15	5.04	0.91	7.56	0.94	11.81	20.27
	15-30	5.03	0.62	6.37	0.77	15.64	19.5
	30-45	4.73	0.31	4.75	0.4	4.51	16.27
2009 UN-BURNT	0-15	5.33	0.89	7.97	1.04	6.93	19.86
	15-30	4.87	0.79	7.45	0.73	6.4	16.6
	30-45	4.96	0.74	6.82	0.73	8.17	18.6
2011 BURNT	0-15	5.72	1.08	7.41	1.12	61.7	18.6
	15-30	5.04	0.63	7.03	1.05	79.26	13.73
	30-45	5.03	0.43	6.88	0.72	78.33	16.2
2011 UN-BURNT	0-15	6.26	1.07	8.87	2.2	71.92	20.23
	15-30	5.89	0.64	5.08	1.93	23.13	21.27
	30-45	6.06	0.59	1.85	1.34	8.81	18.87
2012 BURNT	0-15	5.41	1.02	7.4	2	12.94	23.13
	15-30	5.48	0.89	7.34	1.64	13.26	15.87
	30-45	5.52	1.02	6.8	1.75	5.07	19.84
2012 UN-BURNT	0-15	5.8	0.68	6.42	0.67	7.6	16.13
	15-30	6.04	0.49	4.95	1.24	4.64	19
	30-45	5.15	0.27	3.96	0.91	9.26	15.54

STUDY SITE	DEPTH (CM)	pH	%N	%OC	K(Cmol/ Kg)	P(ppm)	CEC (mol/kg)
2013 BURNT	0-15	5.69	0.46	4.11	1.71	5.68	20.28
	15-30	4.53	0.37	3.41	0.66	6.06	17.33
	30-45	4.87	0.33	3.01	0.4	7.14	14.07
2013 UN-BURNT	0-15	4.92	0.68	6.25	1.86	8.91	24.33
	15-30	4.95	0.46	5.2	1.1	6.97	19
	30-45	4.75	0.48	4.96	0.81	6.56	16.07
2014 BURNT	0-15	5.43	1.31	7.97	3.57	8.3	28.33
	15-30	5.07	0.96	7.27	3.04	6.47	24.08
	30-45	4.77	0.9	6.77	2.79	6.33	22.74
2014 UN-BURNT	0-15	4.33	0.97	6.34	2.2	34.9	23.21
	15-30	4.53	0.76	5.25	1.32	34.4	20.85
	30-45	4.53	0.66	3.85	1.42	15.27	17.79