

**EFFECTS OF FOREST FRAGMENTATION ON FOREST COVER CHANGE, TREE
SPECIES DIVERSITY AND CARBON STOCK IN TAITA HILLS, KENYA**

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**A Thesis Submitted to the Graduate School in Partial Fulfilment for the Requirements
of the Doctor of Philosophy Degree in Natural Resources Management of Egerton
University**

EGERTON UNIVERSITY

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DECLARATION AND RECOMMENDATION

DECLARATION

I hereby declare that this thesis is my original work and that it has not been submitted for examination in any other university for the award of any degree to the best of my knowledge.

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RECOMMENDATION

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DEDICATION

This work is dedicated to my late daughter, Naomi Buyachi who passed on while I was pursuing this study.

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ABSTRACT

The fragmentation of forests into smaller isolated patches is a threat to biodiversity conservation. Fragmentation reduces the forest area and increases forest edge effect. Reduction in forested area may lead to increased local species extinctions, while effects of increased forest edge alter species interactions by increasing the degree of interaction among edge and forest interior species. Intensification of land uses surrounding forest patches resulting from population increase exacerbate the edge effects thereby having significant impact on tree species diversity and the potential of the forest fragments to sequester carbon. Comparative studies were undertaken in five main forest fragments of varying sizes in Taita Hills, Kenya to examine the effects of forest fragmentation on forest cover change, tree species diversity and above-ground carbon stock. Belt transects proportional to the forest size were established along forest edge-interior gradient, and square plots (10x10m) established systematically for assessment of the forest structure, tree species richness and diversity, edaphic factors and above-ground carbon stock. Analysis of Landsat images of 1973, 1987, 2001, 2012 and 2016 provided information on forest cover change of the fragments between 1973 and 2016. Data collected from the field was analysed using GenStat version 16.0 and CANOCO for Windows version 4.5 while Landsat images were analysed using ArcGIS version 10.2. Statistical testing for significance was performed at 95% confidence interval. The results showed that the annual rate of deforestation in Taita Hills was 0.5% and the forest fragments lost 23.2% of their cover between 1973 and 2016. The forest area lost was higher in Vuria (43%) and Chawia (32.7%) which are non-gazetted forests managed by the County government of Taita Taveta but lower in Fururu (3.2%), Mbololo (13.7%) and Ngangao (16.8%) which are gazetted forests managed by Kenya Forest Service. There was no significant edge effect on tree species diversity in Chawia ($p=0.564$), Fururu ($p=0.689$), Mbololo ($p=0.369$), Ngangao ($p=0.576$) and Vuria ($p=0.338$), although the species diversity was higher in the forest interior than in the intermediate forest and on the forest edge in all the five forest fragments. Carbon stocks were not significantly affected by the edge effect in Chawia ($p=0.704$), Fururu ($p=0.908$), Mbololo ($p=0.544$), Ngangao ($p=0.991$) and Vuria ($p=0.590$). There was a significant edge effect on the abundance and distribution of key dominant and adaptable species i.e. *Macaranga conglomerata*, *Albizia gummifera*, *Syzygium guineense*, *Xymalos monospora*, *Tabernaemontana stapfiana* and *Maesa lanceolata* ($p=0.012$). There was no significant edge effect on species richness in Chawia ($p=0.490$), Fururu ($p=0.507$), Mbololo ($p=0.303$), Ngangao ($p=0.886$) and Vuria ($p=0.090$). Large forest fragments had significantly higher species richness and diversity in all the habitats along the forest edge-interior gradient than small forest fragments ($p=0.001$) and ($p=0.001$) respectively. There was no significant effect of fragment size on carbon stock ($p=0.634$). There was significant edge effect on soil pH in Mbololo ($p=0.001$), Ngangao ($p=0.001$) and Vuria ($p=0.042$) but not in Chawia ($p=0.700$) and Fururu ($p=0.667$). There was significant edge effect on soil electrical conductivity in Fururu ($p=0.048$) but not in Chawia ($p=0.896$), Mbololo ($p=0.367$), Ngangao ($p=0.555$) and Vuria ($p=0.861$). The edge effect significantly affected soil nitrogen in Fururu ($p=0.038$) and potassium level in the soil in Mbololo ($p=0.002$) and Ngangao ($p=0.035$). The results show that forest fragmentation contributed to substantial loss in forest cover and hence reducing the capacity of the forest fragments to sequester carbon. Besides, the edge effect created by fragmentation contributed to reduction in floral species richness and diversity, and affected edaphic conditions. These results provide robust evidence for developing land use policies and appropriate management strategies for effective conservation of fragmented forest ecosystems.

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Figure 18: Changes in carbon stock of the five forest fragments in Taita Hills between 1973 and 201670

LIST OF ACRONYMS AND ABBREVIATIONS

AGB	–	Above Ground Biomass
ANOVA	–	Analysis of Variance
BAU	–	Business As Usual
CCA	–	Canonical Correspondence Analysis
CFAs	–	Community Forest Associations
DBH	–	Diameter at Breast Height
DCA	–	Detrended Correspondence Analysis
EC	–	Electrical Conductivity
ETM	–	Landsat Enhanced Thematic Mapper
FAO	–	Food and Agriculture Organization of United Nations
Gg	–	Gigagrams
GHGs	–	Green House Gases
Ha	–	Hectare
HCL	–	Hydrochloric Acid
IBT	–	Island Biogeography Theory
KEFRI	–	Kenya Forestry Research Institute
Kg	–	Kilograms
LULUCF	–	Land Use, Land Use Change and Forestry
M	–	Molar
m.a.s.l	–	Meters Above Sea Level
MCPT	–	Monte-Carlo Permutation Test
Mg C ha⁻¹	–	Mega grams of Carbon per hectare
MSS	–	Landsat Multispectral Scanner
MT	–	Metric Tons
MtCO₂eq	–	Metric Tons Carbon Dioxide Equivalent
NPK	–	Nitrogen, Phosphorus and Potassium
OC	–	Organic Carbon
OLI	–	Operation Land Imager
PFMPs	–	Participatory Forest Management Plans
pH	–	Measure of Acidity or Alkalinity
RCMRD	–	Regional Centre for Mapping Resource for Development
REDD+	–	Reducing Emissions from Deforestation and Forest Degradation

SE	–	Standard Error of Mean
TIRS	–	Thermal Infrared Sensor
TM	–	Landsat Thematic Mapper
USGS	–	United States Geological Survey
UV	–	Ultraviolet

CONVERSION TABLE

2.5 acres	–	1 Hectare
1000 Grams	–	1 Kilogram
1000 Kilograms	–	1 Megagram
1000 Megagrams	–	1 Gigagram

CHAPTER ONE

INTRODUCTION

1.1 Background information

Forest fragmentation refers to the subdivision of continuous forest blocks into clusters of small remnant patches isolated by matrices of dichotomous land-use types (Andren, 1994). Fragmentation leads to long-term effects on species diversity and composition, community dynamics and ecosystem processes, and is a major cause of biodiversity loss (Achard *et al.*, 2002; Aguirre, 2014). In Sub-Saharan Africa and other developing regions, rapid human population growth exacerbates the scale of this problem, since human settlement is attracted to areas of high species richness and endemism (Balmford *et al.*, 2001). Large-scale habitat deterioration may affect animal and plant populations in complex and diverse ways, depending on their species-specific traits and the temporal and spatial scale of the habitat changes involved (Andren, 1994). Most studies show adverse effects of fragmentation on species richness, abundance and composition in both animals and plants (Laurance *et al.*, 2002; Benítez-Malvido and Martínez-Ramos, 2003; Laurance *et al.*, 2011). To ensure successful biodiversity conservation, actions at local scale (within sites) that minimizes habitat loss and deterioration, should be combined with actions at regional scale (across landscapes) to maximize connectivity, dispersal and gene flow (Lens *et al.*, 2002).

Previous studies have shown that fragmentation decreases species number and alters community composition due to reduction in forest area, change in forest shape and increased isolation of the remaining forest fragments (Pimm, 1998). However, remnant forest fragments can still maintain high phylogenetic diversity (Arroyo-Rodriguez *et al.*, 2012; Mo *et al.*, 2013) and harbour large proportions of original species (Arroyo-Rodriguez *et al.*, 2009; Sodhi *et al.*, 2010). There is evidence that, fragmentation has virtually no effect on certain groups of tree species (Andreazzi *et al.*, 2012) with some tree species even increasing in their abundance and diversity (Crooks and Soule, 1999). Such contrasting outcomes of fragmentation are related to how much of the original forest remains after fragmentation, the size and isolation of individual fragments and the distribution of the fragments over the landscape (Laurance *et al.*, 2002; Prugh *et al.*, 2008; Pardini *et al.*, 2010).

Most studies on forest fragmentation have focused on the impacts of patch size, isolation and edge effects on forest structure, species diversity and composition (Laurance *et al.*, 2002;

Laurance *et al.*, 2011). However, forests, and especially tropical forests, show strong spatial and environmental structuring of species composition and diversity (John *et al.*, 2007; Laurance *et al.*, 2010; Garcia-Lopez *et al.*, 2012; Baldeck *et al.*, 2013), meaning that the distribution of the fragments for a large part, determine the type and diversity of tree species preserved across the landscape (Virgos, 2001). Since forest fragmentation is generally a non-random process, with people targeting areas that are easily accessible or have good soil properties for agricultural production, forest fragments are in most cases restricted to areas located on steep slopes, poor soils or with inaccessible topography (Vellend *et al.*, 2008).

The process of fragmentation has myriad impacts on the dynamics of tropical forest ecosystems (Laurance and Yensen, 1991; Laurance and Bierregaard, 1997) but its effect on plant communities has received little attention (Debinsnki and Holt, 2000; Laurance *et al.*, 2014; Benchimol and Peres, 2015). Many effects of forest fragmentation on tropical communities are poorly understood, especially its influence on diverse long-lived organisms like trees (Laurance and Bierregaard, 1997; Bierregaard *et al.*, 2001). Fragmentation increases tree mortality and turnover (Laurance *et al.*, 1998, 2000), reduces live tree biomass and increases atmospheric carbon emissions (Laurance *et al.*, 1997; Nascimento and Laurance, 2004). Additionally, in the case of some species, seedling survival are reduced in fragments because of altered micro-climatic conditions leading to poor regeneration (Benitez-Malvido, 1998; Bruna, 1999), a decline of animal seed dispersers (Silva and Tabarelli, 2000; Cordeiro and Howe, 2003), and elevated seed predation (Terborgh *et al.*, 2001).

In Kenya, the Eastern Arc Mountains in Taita Hills have suffered extensive forest loss and fragmentation due to human disturbance (Newmark, 1998) despite being rich in biodiversity (Aerts *et al.*, 2011). Although the exact pre-historic or pre-human impact on forest cover in the Eastern Arc Mountains is unknown, the existence of remnant patches of forest at varying elevations throughout these mountains suggest that nearly all Eastern Arc Mountains were forested prior to human settlement and disturbance. The presence of Iron Age sites in Taita Hills, Usambara and Pare Mountains indicates that anthropogenic activities have been altering the Eastern Arc Mountains forests for the last 2,000 years (Schmidt, 1989).

Research evidence from Kenya sheds light on how forest fragmentation affects species composition, species richness and ecosystem processes. Examples include the study by Schleuning *et al.* (2011) which evaluated the impact of forest fragmentation and selective

logging on animal-mediated ecosystem processes, species richness and community composition of animal taxa involved in the ecosystem processes such as pollination, seed dispersal, seed predation and decomposition in Kakamega forest. Forest fragmentation increased decomposition and reduced ant-bird predation, while selective logging consistently increased pollination, seed dispersal and army ant raiding (Schleuning *et al.*, 2011). Additionally, fragmentation modified species richness or community composition of five taxa of fauna namely frugivorous birds, army ants, rodents, understory bees and leaf litter fauna, whereas selective logging did not affect any component of biodiversity (Schleuning *et al.*, 2011). However, most of the studies conducted in Kenya to assess the impact of fragmentation on biodiversity have overlooked the vegetation component yet this is critical in the overall functioning of forest ecosystems. For instance, studies by Brooks *et al.* (1999), Lung (2009) and Peters *et al.* (2009) conducted in Kakamega forest to evaluate the impact of fragmentation on biodiversity only focused on birds and primates leaving out the vegetation component.

In the last decade, studies on the effects of fragmentation on plant communities have gained prominence in Kenya especially in forests that are centres of endemism for many plant and animal species. Farwig *et al.* (2006) evaluated the impact of fragmentation and small-scale disturbance on forest regeneration, species richness and total abundance of adult trees in Kakamega forest. Pellikka *et al.* (2009) assessed the changes in area and species composition of six indigenous forest fragments in the Taita Hills between 1955 and 2004. In total, 260 hectares (ha) of indigenous tropical evergreen forests were lost in Taita Hills to agriculture and bush land between 1955 and 2004 (Pellikka *et al.*, 2009). Large scale planting of exotic tree species such as *Pinus patula*, *Eucalyptus* species, *Grevillea robusta*, *Acacia mearnsii* and *Cupressus lusitanica* on barren land within the indigenous forests during the same period resulted in a balanced total forest area in six indigenous forest fragments (Omoro *et al.*, 2010). Consequently, remarkable changes have occurred in indigenous forest area and floristic species composition in the forest remnants due to fragmentation (Pellikka *et al.*, 2009).

Many authors (Brooks *et al.*, 1999; Peters *et al.*, 2009; Lung, 2009; Schleuning *et al.*, 2011) have largely focused on fauna with little attention to the vegetation component. Evidence from studies by Farwig *et al.* (2006) and Pellikka *et al.* (2009) underscores the significance of the adverse effects of fragmentation on biodiversity and forest's ability to provide ecosystem services including carbon sequestration. However, no comprehensive and systematic attempt has

been made to quantify the rate of forest cover change using satellite images since forests in Taita Hills were put under organized management through presidential decree of 1977 and relate this to the period after the Forests Act 2005 was enacted and operationalized to inform policy decisions. Moreover, the impact of increased forest edge due to fragmentation on abundance of dominant and adaptable tree species, species richness and diversity and carbon stock and the associated implications on biodiversity conservation and climate change mitigation has not been comprehensively studied.

Although fragmented forests can experience large changes in floristic composition, including an increase in disturbance-adapted plants, the details of such changes and their underlying causes are not well known. Thus, as fragmentation continues in natural forest ecosystems in Kenya including Taita Hills, conservation efforts must focus on understanding the effects of fragmentation on forest cover change, tree species diversity and carbon stock to develop strategies for conserving biological diversity. This study constitutes an in-depth assessment of the change in forest cover and fragment size to ascertain their effects on species richness and diversity, abundance of dominant and adaptable tree species and carbon stock in five forest fragments in Taita Hills namely Chawia, Fururu, Mbololo, Ngangao and Vuria.

1.2 Statement of the problem

Land use and land use change directly affect ecological integrity of natural forests and are key drivers of forest fragmentation. Past and present anthropogenic activities are responsible for the current scenario of highly fragmented forests occurring in Taita Hills. Crop farming, mining and creation of new settlements have led to clearance of forested areas resulting into large-scale fragmentation and consequently reduced ecological integrity of the forest fragments. Thus, clearance of forested areas has led to habitat reduction and increased isolation of the remaining forest patches. Despite the small size of the remaining indigenous forest fragments, they posit local ecological significance and global conservation importance, holding numerous rare and endemic plant and animal species. Moreover, deforestation resulting from fragmentation is a major source of greenhouse gases (GHGs). Kenya's total greenhouse gas emissions stood at 73 MtCO₂eq in 2010, out of which 75% are from the land use, land-use change and forestry (LULUCF) and agriculture sectors. Terrestrial forests store significant amounts of carbon and therefore, deforestation reduces the amount of carbon sequestered by these forests hindering

Kenya's ambitious mitigation contribution towards Paris Agreement of reducing its greenhouse gas emissions by 30% by 2030 relative to the business as usual (BAU) scenario of 143 MtCO₂eq. Promoting conservation and enhancement of carbon stocks through sustainable management of these terrestrial forests can significantly contribute to reducing greenhouse gas emissions and mitigating the impacts of climate change. To date, the effects of forest fragmentation on forest cover change, tree species richness and diversity, abundance of dominant and adaptable tree species and carbon sequestration capacity remains unexplored in the fragmenting mountainous forests of Taita Hills. Since the consequences of fragmentation are rapidly becoming established as a dogma in forest conservation, understanding the effects of forest fragmentation on biodiversity are essential for successful and efficient forest conservation. Therefore, up to date information is needed to accurately evaluate the effects of fragmentation on tree species richness and diversity, abundance of dominant and adaptable tree species, carbon stock and the changes in the area of forest fragments in order to develop conservation strategies for the fragile landscape and minimize forest degradation and the associated biodiversity loss.

1.3 Broad objective

To contribute towards biodiversity conservation by generating knowledge on tree species-habitat relationships in space and time for effective management of fragmented fragile forest ecosystems.

1.3.1 Specific objectives

1. To assess the spatial changes in forest cover in Taita Hills between 1973 and 2016
2. To determine the edge effect on the forest structure, tree species diversity, abundance of dominant and adaptable tree species and above-ground carbon stock
3. To determine the effect of fragment size on species richness, diversity and above-ground carbon stock
4. To determine the edge effect on edaphic factors (moisture content, nitrogen, phosphorus, potassium, pH, electrical conductivity and organic carbon)

1.4 Hypotheses

H_{A1}: Forest cover in Taita Hills has changed between 1973 and 2016

H_{A2}: Edge effect affects the forest structure, tree species diversity, abundance of dominant and adaptable species and above-ground carbon stock

H_{A3}: Fragment size has an effect on species richness, diversity and above-ground carbon stock

H_{A4}: Forest edge soils have lower moisture, nitrogen, phosphorus, potassium, pH, electrical conductivity and organic carbon than forest interior soils

1.5 Justification of the study

Tropical forest fragmentation is under-represented in the scientific literature in comparison to temperate zone (Boyle, 2008). Current insights on the effects of forest fragmentation on forest biodiversity tend to rely heavily on results of studies undertaken in a limited number of localities in East Africa. The studies mostly undertaken in Kibale and Kakamega forests in Uganda and Kenya respectively have largely focused on fauna with little attention to the vegetation component. Few studies have explicitly examined the influence of spatial attributes of forest fragments when exploring the impacts of fragmentation on tree species in Kenya. Therefore, more research on forest fragmentation is required if we are to understand its ecological effects to inform conservation decisions. Furthermore, understanding the effects of forest fragmentation on the changes occurring in the forest patches is essential for ranking of alternative management options by their relative potential benefits to biodiversity conservation. Consequently, the findings of this study may guide forest managers, practitioners and policymakers to develop practical management approaches and appropriate policies required to prevent the collapse of important fragmented forest ecosystems that are important biodiversity hotspots. Besides, the evidence on the varying carbon sequestration capacity of the forest fragments occasioned by the process of fragmentation could help inform development of national strategies for climate change mitigation such as Reduced Emissions from Deforestation and Forest Degradation (REDD+). This is because forest ecosystem and carbon stock studies provide a super structural basis for developing appropriate policies for climate change mitigation.

1.6 Scope and limitations of the study

Studies on the influence of fragmentation on tree species richness and diversity, and above-ground carbon stock were carried out in the fragmented landscape of Taita Hills. There are currently twelve remaining indigenous forest fragments in Taita Hills. However, the study only focused on five major fragments with a rich diversity of indigenous flora and fauna under threat by fragmentation. The influence of reduction in forest area, edge effect as well as edaphic factors was determined in relation to species diversity, richness, abundance of dominant and adaptable tree species and carbon stock. Increased isolation of forest fragments and ecosystems processes such as predation, seed dispersal and pollination, and the feedback mechanisms with forest fragmentation were not covered due to resource constraints and the long period such studies require to obtain convincing results. The data collected in 2015 and 2016 along the forest edge-interior gradient provided a basis for evaluation of the effects of forest edge created by fragmentation on forests' ecological dynamics. Data collection took place one week after the end of long rain seasons to capture explicitly the nexus between edge effects and tree species diversity. Time series analysis of Landsat images was possible at different intervals between 1973 and 2016. This measure was applied due to unavailability of clear satellite images for 10-year interval. Fourteen years interval was used between 1973 and 2001, 11 years from 2001 to 2012 and 4 years for the period ranging from 2012 to 2016 and the changes in forest cover for 43 years (1973-2016) was determined.

This study had some limitations. Firstly, because of the low resolution (60m), Landsat Multispectral Scanner (MSS) images taken in 1973 were too coarse to provide accurate baseline information for subsequent monitoring of changes in forest cover over time for the five forest fragments. Besides, some areas of the forest fragments were inaccessible due to the steep slopes attributed to sharp cliffs. This affected the sampling design, which had to be slightly altered to accommodate such terrain whenever a transect fell in an area along the forest edge with a sharp cliff.

1.7 Operational definition of terms and concepts

Canonical Correspondence Analysis: This is a multivariate method to elucidate the relationships between biological assemblages of species and their environment.

Carbon sequestration: Refers to the removal of carbon from the atmosphere and storage in carbon sinks such as oceans, forests or soils through physical or biological processes, such as photosynthesis.

Core area: This refers to the total patch or landscape area that consists of interior forest outside the zone of significant edge influence (i.e. total forested area minus edge area).

Co-registration: Refers to the process of geometrically aligning two or more images so that corresponding pixels representing the same objects are integrated.

Detrended Correspondence Analysis: Refers to a simple multivariate technique for arranging species and samples along environmental gradients.

Disturbed forest areas: Areas within the forest exhibiting low density of trees with canopy cover of between 20% and 60%.

Forest cover dynamics: The process of change in the size/area, composition, structure and function of a forest that occurs at different temporal and spatial scales.

Forest edge effect: Refers to the influence of the two bordering habitats on each other resulting to changes in population or community structures that occur at the boundary of the two habitats.

Forest edge: Interface between forested and non-forested ecosystems or between two forests of contrasting composition or structure, usually 0-150m to the forest interior.

Forest fragmentation: Refers to the breaking of large, continuous, forested areas into smaller isolated patches of forest; typically roads, agricultural lands, utility corridors or other human developments separate the patches.

Forest interior: The forest area that shows no detectable edge influence, usually found in the center of the forest patch.

Intermediate forest: Refers to the forest occurring in the transition zone between the forest edge and the forest interior.

Kaolinite: This is common soil-forming clay mineral with the chemical composition $\text{Al}_2\text{Si}_2\text{O}_5(\text{OH})_4$.

Landscape: This refers to an area that is spatially heterogeneous and composed of interacting mosaic of ecosystems and encompassing population of many species (Turner *et al.*, 2002).

Monte Carlo Permutation Test: Refers to statistical test whereby the values of the environmental variables are randomly assigned to the individual samples of species composition through ordination analysis of permuted data set to calculate the value of the test statistic.

Open ground: Refers to a bare area within a forest not covered by trees or any other vegetation.

Ortho-rectification: This is the process of removing the effects of image perspective (tilt) and relief (terrain) effects to create a planimetrically correct image.

Principal Component Analysis: Statistical procedure that uses an orthogonal transformation to convert a set of observations of possibly correlated variables into a set of values of linearly uncorrelated variables called principal components.

Tree diversity: This is the variability among tree species and the associated ecological processes.

Undisturbed forest areas: Areas within a forest exhibiting high density of trees with canopy cover of more than 60%.

CHAPTER TWO

LITERATURE REVIEW

2.1 Forest fragmentation at global level

Globally, over one half of the temperate broadleaf and mixed forest biome and nearly one quarter of the tropical rainforest biome have been fragmented or removed by humans, as opposed to only 4% of the boreal forest (Wade *et al.*, 2003). According to Wade *et al.* (2003), Europe has experienced the most human-caused fragmentation while South America has had the least. Land-use change and agricultural intensification are responsible for a global decline of forest cover entailing the continuous fragmentation of forests (Green *et al.*, 2005). Activities or events responsible for forest fragmentation include road construction, logging, conversion of forests to agricultural land, or wildfire, but ultimately, the fragmenting cause is either anthropogenic or natural in origin (Cullen *et al.*, 2000; Carvalho *et al.*, 2001). Forest fragments reflect reduced habitat size and exhibit strong edge effects, which cause changes in abiotic factors, such as higher light intensity on the forest floor or altered air temperature and thereby potentially affect their habitat quality (Murcia, 1995; Laurance *et al.*, 2002). Breaking up of forests will inevitably continue over the coming years, especially in developing economies (Wade *et al.*, 2003). For instance, in Madagascar, fragmentation caused forest cover decrease of 40% from 1950 to 2000, with a reduction in core forest >1.0 km from a non-forest edge of almost 80% (Harper *et al.*, 2007).

Several studies have illustrated that forest fragmentation is one of the main drivers of biodiversity loss and consequently alters important ecological processes such as seed predation, seedling establishment and herbivory (Donoso *et al.*, 2003; Benítez-Malvido and Lemus-Albor, 2005). Nonetheless, forest fragments can contribute to landscape connectivity, provide habitat for numerous animal species and are thus important stepping-stones for maintaining ecological processes (Bodin *et al.*, 2006; Fischer and Lindenmayer, 2007). In areas that lack continuous natural forests, fragments enhance the diversity of the landscape and increase chances of species survival (Turner and Corlett, 1996).

Traditionally, habitats surrounding fragments serve as inhospitable oceans of no value to species conservation. This included a binary view of landscapes distinguishing between habitat and non-habitat (Murphy and Lovett-Doust, 2004). In fact, matrices surrounding forest fragments vary in structure and permeability for different species (Kupfer *et al.*, 2006; Brockerhoff *et al.*,

2008). High-contrast matrices differing strongly in structure from the original habitat might hinder species movement and plant-animal interactions (Prevedello and Vieira 2010; Herrera *et al.*, 2011). Alternatively, low-contrast matrices that are structurally similar to the natural habitat might facilitate species and seed dispersal and thereby contribute to the stability of human-modified fragmented landscapes (Herrera and García, 2009).

Generally, forest fragmentation involves the creation of a ‘mosaic’ landscape with remnants of natural habitat within variable matrix habitat (Farwig, 2009). Inspired by two well-established ecological theories; island biogeography and meta-population theories, it was believed that the conservation of many species could only be accomplished in large habitat patches as species richness is likely to increase with habitat area and connectivity (MacArthur and Wilson, 1967; Hanski and Ovaskainen, 2000). Many conservation scientists favour an integrated perspective of landscapes (Jules and Shahani, 2003). These scientists have highlighted the great potential of human-modified mosaic-forest landscapes comprising both small forest fragments as well as matrix habitat for biodiversity conservation and maintaining ecological processes (Daily, 2001; Jules and Shahani, 2003).

2.2 Drivers of forest fragmentation

Human disturbance is a major driver of global environmental change (Tylianakis *et al.*, 2008) and is a threat to biodiversity (Gardner *et al.*, 2009), species interactions and ecosystem processes in tropical forests (Steffan-Dewenter *et al.*, 2007; Morris, 2010). At landscape scale, forest loss and fragmentation severely affect biodiversity (Laurance *et al.*, 2002; Laurance *et al.*, 2007; Morris, 2010). Species richness declines in fragmented forests because remnants are too small for species to persist or excessively isolated for colonization by species from other remnants (Fahrig, 2003). Edge effects and spill over of species from secondary habitats can also modify the species composition in fragmented forests (Ewers and Didham, 2006).

Selective logging has been reported to either increase or decrease forest biodiversity (Hill and Hamer, 2004; Koh, 2007). In moderately logged forests, species communities can be similar to those of undisturbed forests (Edwards *et al.*, 2011), while the impacts of selective logging can strongly change the environmental conditions in tropical forests (Miller *et al.*, 2007). Although there is a consensus that human disturbance strongly affects tropical biodiversity (Morris, 2010), numerous studies show evidence of the differential effect on individual species by the same type

of human disturbance (Barlow *et al.*, 2007; Kessler *et al.*, 2009). Because of species-specific responses, the effects of human disturbance on alpha-diversity (species richness) are usually weaker than on beta-diversity (community turnover) (Barlow *et al.*, 2007; Kessler *et al.*, 2009).

2.3 Impacts of fragmentation on forest cover change

Island Biogeography theory (MacArthur and Wilson, 1963; 1967) provides a basic conceptual model for understanding habitat fragmentation. The Island Biogeography theory (IBT) has inspired much thinking about the importance of reserve size and connectivity in the maintenance of species diversity, and stimulated an avalanche of research on fragmented ecosystems (Laurance, 2008). Fragmentation reduces total area covered by forests, which may result into the extinction of some species (Murcia, 1995). Available evidence shows that approximately 2.3 million ha of humid forest is degraded annually through fragmentation, logging and/or fires (Lambin *et al.*, 2003; Mayaux *et al.*, 2005). In the sub-humid and dry tropics, annual deforestation of tropical moist deciduous and tropical dry forests as a result of fragmentation is between 2.2 and 0.7 million ha, respectively (FAO, 2001; Mayaux *et al.*, 2005). Southeast Asia is the region where forests are under the highest pressure with annual relative rate of deforestation resulting from fragmentation standing at between 0.8 and 0.9% (FAO, 2001). The annual area deforested in Latin America is large, but the annual relative rate of deforestation is lower (0.4-0.5%) and this is attributed to the vast area covered by the remaining Amazonian forests (FAO, 2001). The humid forests of Africa are receding at a similar rate to those of Latin America (0.4-0.5% per year). The global net loss of forests between 2000 and 2005 was alarmingly high, estimated to be 4.0 million ha per annum (Kelatwang and Garzuglia, 2006). Africa contributes 5.4% to the estimated loss of humid tropical forest cover (Hansen *et al.*, 2008). In East Africa, annual loss of forest area of 1.2% and 2.7% has been reported in Tanzania and Uganda respectively (Wu, 2011) while in Kenya, the rate of deforestation is 0.3% per year (Wu, 2011).

Habitat loss resulting from fragmentation has two consequences: firstly, a reduction in the range of habitat types results in fewer species; and secondly, a reduction in resources for species leads to smaller populations that are more vulnerable to extinction (Kupfer *et al.* 2006). Smaller populations lead to higher extinction rates although such effects may not be immediately manifested (Tilman *et al.* 1994). Thus, Island Biogeography Theory (IBT) postulates that species

extinction rates are higher on small islands. While (Fahrig, 1997; Haila, 1999) argued that area effect should not be considered as part of fragmentation in the strict usage of the phrase, it must be accounted for in part because the nature of isolation effects could be based on the area effect.

Reductions in forested area result in the subdivision of populations. Island Biogeography theory and meta-population models have demonstrated that patch area affects their role as target; larger patches are easier to locate. Patch size also affect the presence of animal dispersal vectors such as large mammals and some birds may require larger patches for foraging, nesting, and mating and thus may not serve as vectors in smaller forest patches. Patch size and shape might also affect wind fields (Pielke *et al.*, 1997); smaller patches with certain orientations will have less drag on wind and may have seeds more easily lofted above and away increasing dispersal.

Characteristically, types, extents, frequencies and even intensities of disturbances can be constrained by the area of remnants, and disturbances that begin in the matrix can spread to influence processes in the remnant (Malanson *et al.*, 2004). Some types of disturbances depend on area; for instance, small areas may be more subject to wind destruction (Malanson *et al.*, 2004). Specific disturbance characteristics such as fire intensity are influenced by the amount of area over which the disturbance develops (Esseen, 1994), and the disturbed area can be part of the disturbance regime. Baker (1989) has illustrated how forest fragmentation has altered the potential fire disturbance regime in Northern Minnesota. The combinations of these aspects of disturbance regime affect the dynamics of species and the resulting biodiversity (Malanson, 1984). Reductions in forest area affect potential species interactions with notable trophic effects (Malanson *et al.*, 2004).

The degree to which area-related changes occur is a function of the total amount of forest loss, the size of remaining forest remnants and the location of remnant forest in the landscape where the destruction occurs (Kupfer *et al.*, 2006). There is a correlation between the size of forest remnants and their location in a landscape where the destruction occurs and the total area of habitat destruction, although all are components of reduced forest area (Kupfer *et al.*, 2006). Even in a continuously forested landscape, spatial structure i.e. the pattern of heterogeneity or interspersions of forest types is present. Rarely are patterns of deforestation across a landscape random (Kupfer and Franklin, 2000), so the spatial selectivity of forest clearing affects the degree to which forest fragments represent the original forest. Additionally, populations in the remnants are merely a sample of the original population and may not be representative of the

pre-cut populations, not only in terms of numbers but also in terms of a range of other characteristics like genetic diversity and demographic structure (Terborgh *et al.*, 1997).

In Sub-Saharan Africa (SSA), forest loss is proceeding at an unprecedented rate of 2.8 million ha annually, particularly in areas with high biodiversity, and in Afromontane areas where the decrease in forest cover is estimated to be 3.8% or 61,000 ha annually (Eva *et al.*, 2006). Forest cover loss due to fragmentation is threatening tropical forest biodiversity in East Africa (Lung and Schaab, 2010). Kenya too has not been spared, recent time-series analysis revealed that, despite their protection status, Kakamega and Nandi forests have lost 31% of the original area between 1972 and 2001 (Lung and Schaab, 2010). In Taita Hills, indigenous evergreen forest currently covers only about 430 ha, reflecting 98% forest reduction over the last 200 years (Pellikka *et al.*, 2009).

The degradation and fragmentation of the forests in Taita Hills started before the colonial era (Hildebrandt, 1877) whereby forested slopes were cleared for agriculture. Moreover, the construction of Kenya-Uganda railway between 1898 and 1924 caused large-scale reduction in the forest cover. Since 1960's, forest cover loss in Taita Hills has been very substantial, Vuria (99%), Sagala (95%), Chawia (85%), Ngangao (50%) and below 50% for Mbololo (Beentje, 1988). In response to the decreasing forest cover, exotic tree species such as *Cupressus lusitanica*, *Eucalyptus saligna*, *Pinus elliottii*, *Pinus caribaea* and *Pinus patula* were introduced by the Forest Department between 1950s and 1970s; and *Maesopsis eminii* and *Grevillea robusta* in the late 1970s and early 1980s (Pellikka *et al.*, 2009). The coordinated management of indigenous forests in Taita Hills began in 1977 following Presidential decree banning cutting of trees in the indigenous forests without a license (Beentje, 1988). This was followed by the posting of the first District Forest Officer (DFO) to Taita Taveta County (formerly Taita Taveta District) in 1982. Prior to the posting, forest management in the County was administered from Provincial Forest Office in Mombasa.

After 1982, the management of forests in Taita Hills by the Forest Department entailed enforcement, through policing and punitive actions, of laws to prevent illegal activities (Himberg *et al.*, 2009). Such management approach led to widespread conflicts between the local communities and the Forest Department as more forest fragments were being gazetted amid the rising population and dwindling forest cover (Himberg *et al.*, 2009; KFS, 2016). To allay the rising discontent and conflicts and arrest the run-away deforestation, a new legislation (The

Forest Act 2005) was enacted to allow communities to participate in forest management (Republic of Kenya, 2005; Omoro, 2012). Consequently, Community Forest Associations (CFAs) were formed, registered and hence permitted to participate in the management of state or local forests in Taita Hills. This ushered in a new era of forest management in Taita Hills as Participatory Forest Management Plans (PFMPs) were developed to ensure sustainable management of the forest fragments to conserve biodiversity, improve local livelihoods and enhance provision of ecosystem services (KFS, 2016).

Several studies (Beentje, 1988; Wilder *et al.*, 1998; Pellikka *et al.*, 2009) have presented different estimates of the area covered by each of the forest fragment in Taita Hills. This makes it difficult for forest managers and policy makers to develop appropriate policies to conserve the forests. Moreover, information on the systematic changes in forest cover that have occurred over time since well-coordinated conservation of indigenous forests began in 1977 by presidential decree banning the cutting of indigenous forests without a license is lacking. This study endeavoured to provide information on the systematic changes in forest cover from the time the presidential ban came into effect and relate this to the period after the Forests Act of 2005 was enacted and operationalized to evaluate the effectiveness of well-coordinated forest management in reducing deforestation. The lack of long-term data on forest cover change has been an impediment to sustainable forest management and formulation of appropriate policies for enhancing forest conservation.

2.4 Forest fragmentation and the edge effects

Even though forest edges reflect sharp boundaries between forested and non-forested systems, the structure and composition of the adjacent habitats may vary greatly, resulting in significant differences in edge characteristics and complexity (Matlack, 1993). Forest fragmentation increases susceptibility of forest remnants to edge effects. Microclimatic changes such as incoming solar radiation, temperature, wind speed and evapotranspiration differentiate forest edges (Ranney *et al.*, 1981). The microclimate at the forest edge adjacent to cleared land differs from that of the forest interior in attributes such as incident light, humidity, ground and air temperature, and wind speed. These physical changes affect biological processes such as litter decomposition and nutrient cycling, and the forest structure, composition of vegetation and ecological function occurring along forest edges exposed to non-forested habitats following

fragmentation (Bennett and Saunders, 2010). Several studies have documented the depth to which various micro-climatic edge effects may be observed (Baker and Dillon, 2000), but it is important to recognize that edges represent gradients of multiple physical factors (Cadenasso *et al.*, 1997) rather than isolated forest communities. Although reduced forest area influences edge effects through the ratio of edge to interior habitat, these edge effects are a function of spatial configuration independent of habitat amount (Cadenasso *et al.*, 1997).

Changes in abiotic conditions along edges typically lead to biotic responses. Tree species growing on the edge are usually those adapted to edge microclimates and are often species found in the early stages of forest succession in a given region (Kupfer and Malanson, 1993). The responses of forest interior species to conditions that develop along the newly created forest edge vary, some species are advantaged and increase in abundance (Bennett and Saunders, 2010) while others are unable to survive in the newly created conditions and hence decline becoming locally extinct. Species' abundances in fragments will differ from those in intact and continuous forest, with some declining and others becoming hyper-abundant (Laurance, 2002).

Consequently, for edge sensitive species that become locally extinct, the amount of habitat lost to fragmentation is greater than that of simple conversion of the habitat into new land use. Mortality and damage rates of large trees (>10 cm in DBH) have been found to be sharply elevated within 100m of forest margins due to increased wind turbulence and micro-climatic changes near forest edges (Laurance, 1997a). In the case of closed canopy forests, light is strongly limiting to small trees and hence, many species are adapted for growing rapidly in tree fall gaps (Clark, 1990).

According to Medley (1997), enhancement of invasibility by exotic species is a function of habitat destruction since many invasive plant species are prolific seed producers that thrive in higher light conditions and have widely dispersed seeds. These traits make them more likely to establish and thrive along forest edges, after which they may be able to play an increasing role in vegetation dynamics within remnant interiors (Burke and Nol, 1998). The edge effects alter species interactions by increasing the degree of interaction among edge and interior species (Fagan *et al.*, 1999; Gustafson *et al.*, 2002).

Edge-interior relationships for a forest fragment are often quantified using metrics such as core-area models, which provide an approximation of total edge and core habitat based on a measure of edge penetration depths and forest geometry (Laurance, 1991). The intensity of edge

effects in forest fragments and the distance over which they act varies between processes and between ecosystems. In tropical forests in the Brazilian Amazon, changes in soil moisture content, vapour pressure deficit, and the number of tree fall gaps extend about 50m into the forest, whereas the elevated tree mortality extend 200m or more from the forest edge (Laurance, 1997a; Laurance, 2008). Generally, changes at forest edges are detrimental to conservation values because they modify formerly intact habitats (Laurance, 2008). Previous findings (Gascon *et al.*, 2000; Laurance, 2008) suggest that edge effects and area-related extinctions will rapidly degrade smaller fragments (<100 ha), which are predominant in anthropogenic landscapes. Laurance *et al.* (1998; 2000) have unequivocally demonstrated the negative impacts of habitat fragmentation on the abundance of Neotropical trees in remnants of Amazonian rainforest. The major source of these impacts is edge effects; the diverse physical and biotic changes associated with the abrupt, artificial margins of forest fragments (Bierregaard *et al.*, 2001). Little information exists on the behaviour of fragmented ecosystems in terms of forests dynamics, plant community composition, emergence of invasive species and carbon storage potential attributed to edge-related changes (Laurance *et al.*, 2007). This is also true for Kenya where studies on the impacts of fragmentation on biodiversity dynamics are scarce.

2.5 Forest fragmentation and regeneration patterns

One phenomenon that has received little attention is tree regeneration in fragmented forests (Viana *et al.*, 1997). Regeneration patterns are important because they determine the floristic composition of the remnant patches. According to Janzen (1983), fragments of tropical dry forests in Costa Rica were prone to invasions by weedy generalists' plant species from the surrounding modified matrix that progressively altered the floristic composition of the remnants. Fragments in some tropical regions have suffered chronic disturbance by winds and other factors and hence may experience a total shift towards successional trees, lianas and vines adapted to recurring disturbance (Laurance, 1991; 1997a). There is evidence that tree regeneration rate is higher in forest fragments than in continuous forests (Laurance *et al.*, 1998). The regeneration rates increase with the age of the fragment and are markedly elevated near forest edge (Laurance *et al.*, 1998). Hence, successional trees increase in older and smaller fragments and in forest areas near the edges while old growth trees declines in older fragments (Laurance *et al.*, 1998).

Successional trees increase in fragments because recurring disturbances provide light and micro-climatic conditions that favour their germination and growth (Clark, 1990).

Leaf litter has been found to accumulate near edges (Carvalho and Vasconcelos, 1999) because drought-stressed trees shed leaves and possibly, because drier edge conditions slow litter decomposition (Didham, 1998). Accumulating litter may negatively affect seed germination (Bruna, 1999) and seedling survival (Scariot, 2001) and makes forest edges vulnerable to surface fires during droughts (Cochrane *et al.*, 1999). Cordeiro *et al.* (2009) reported that *Leptonychia usambarensis* exhibit poor regeneration and there were fewer juveniles in small fragments than continuous forest hence emphasizing on how fragmentation affects the regeneration of certain tree indigenous species.

2.6 Floral species richness response to fragmentation

In Amazonian forest, species richness in habitat fragments is positively correlated with fragment size (Laurance *et al.*, 2002). Smaller patches have low species richness compared to large forest fragments (Laurance *et al.*, 2002). Habitat area and increasing habitat heterogeneity are the most important factors of plant diversity (Krauss *et al.*, 2004). Species richness of recruited trees, lianas, herbs, and palm seedlings are lower in forest fragments than in continuous forests, with a 1.0 ha fragment having the poorest species richness (Benítez-Malvido and Martínez-Ramos, 2003). In contrast, studies in Kakamega forest have shown that fragmentation does not reduce species richness (Schleuning *et al.*, 2011). The relatively large size of the forest fragments (>40.0 ha) in Kakamega forest mitigate effects of forest fragmentation on species richness (Schleuning *et al.*, 2011). There is evidence that structurally rich farmlands surrounding natural forests alleviates the effects of forest fragmentation on species richness as observed by Schleuning *et al.* (2011).

Evidence from studies carried out in Mexico indicates that effects of fragmentation are dependent on the vegetation type and that these are not strongly related to the species richness and diversity in a micro-scale of patches of one hectare (Aguirre-Gutiérrez, 2014). Cagnolo *et al.* (2006) demonstrated that the effect of area was most pronounced for rare species suggesting that large forests are necessary to preserve species with low local or regional abundance. Forest transformation into smaller remnants may lead to an impoverishment of plant communities (Cardoso da Silva and Tabarelli, 2000). Subsets of remnant species defined by ecological traits

such as rarity, origin and pollination are susceptible to transformation effects (Cagnolo *et al.*, 2006). There is evidence of higher species richness in the interior of the forest than at the edge of the woodlands for native plants (Cagnolo *et al.*, 2006).

Farwig *et al.* (2006) postulated that either fragmentation or small-scale disturbance has an impact on adult tree species richness or total tree abundance. Fragmentation and especially small-scale disturbance significantly reduced seedling species richness, particularly of late-successional species (Farwig *et al.*, 2006). Harcourt and Doherty (2005) also established that African studies on forest fragmentation showed weaker relations between forest size and species richness, but they suggest this was probably due to either confounding effects of isolation distance or too few studies.

Many studies in Africa on forest fragmentation have involved the Kibale forest complex in Western Uganda (Skorupa, 1986; Chapman and Onderdonk, 1998). Studies on forest fragmentation on biodiversity are rapidly gaining prominence in East Africa especially in forests that are biodiversity hotspots. For instance, Ojoi *et al.* (2014) examined the effect of forest fragmentation on tree species abundance and diversity in the Eastern Arc Mountains of Tanzania, and the results showed that individual tree species responded differently to patch area dynamics, habitat status and soil conditions. Generally, the abundance of dominant species like *Maytenus undata*, *Zenkerella capparidacea* and *Oxyanthus speciosus* decreased with a reduction in patch area (Ojoi *et al.*, 2014). In Taita Hills, the Afromontane forests have been severely reduced and fragmented into small patches in the last 200 years, and have suffered varying levels of local small-scale disturbance through encroachment for agricultural land (Loven and Wasser, 1993). An examination of the relationship between species richness and size of forest fragments in Taita Hills is urgently required to understand the rate of species loss due to fragmentation. This information is necessary in the formulation of appropriate conservation measures to reduce forest degradation.

2.7 Nexus between forest fragmentation, species composition and forest structure

The influence of the adjacent non-forest environment on forest structure and species composition at created forest edges is now widely recognized (Harper *et al.* 2005). Alterations of spatial structure of forest fragments are well known to result in changes to community composition within the remnant patches (Saunders *et al.*, 1991; Laurance, 1997b; Laurance *et al.*,

1998; Drinnan, 2005) and to alter ecological processes such as nutrient cycling and predator-prey relationships (Bennett, 2003). The ecological consequences of forest fragmentation reflect the temporal and spatial configuration dynamics of the fragments within the landscape (Forman and Godron, 1986; Drinnan, 2005). Recent studies have revealed the importance of considering spatial configuration when assessing the impacts of fragmentation on herbaceous plants (Petit *et al.*, 2004), but few studies have explicitly considered the effect of spatial attributes when modelling the impacts of fragmentation on woody species (Tabarelli *et al.*, 1999; Metzger, 2000).

Patch size is the most important attribute influencing different measures of species composition, stand structure and anthropogenic disturbances in the temperate forests (Echeverría *et al.*, 2007). There is a relationship between abundance of tree and shrub species associated with interior edge of habitats and variation in patch size (Echeverría *et al.*, 2007). Basal area, a measure of forest structure, significantly declined with decreasing patch size, suggesting that fragmentation affects successional processes in the remnant forests (Echeverría *et al.*, 2007). Findings by Echeverría *et al.* (2007) demonstrated that the progressive fragmentation by logging and clearance is associated with dramatic changes in the structure and composition of the temperate forests.

In comparison with the large numbers of studies undertaken in Brazilian forests (Tabarelli *et al.*, 1999; Laurance *et al.*, 2006), lowland tropics (Fukamachi *et al.*, 1996) and temperate forests (Echeverría *et al.*, 2007), few researchers have explored the influences of fragmentation on species composition and forest structure in tropical forests of Africa. Although some hypotheses have been advanced on the processes influencing forest structure and composition in tropical forests (Hill and Curran, 2001; Laurance *et al.*, 2006), little is known about the impacts of fragmentation on the floristic composition and stand structure of forest communities (Hobbs and Yates, 2003; Harper *et al.*, 2005). In addition, the central theme of most studies on spatial attributes has revolved around singletree species even in highly diverse ecosystems similar to Eastern Arc Mountains of Taita Hills (Henríquez, 2004).

2.8 Forest fragmentation and its impact on carbon sequestration

Tropical forests play an important role in global carbon cycle. It is estimated that about 60% of the world's total forest carbon in living biomass (298 billion metric tons of carbon) is

stored in tropical forests (FAO, 2010) but every year since 2000 to 2010 over 90,000 km² of this forest type was cleared, representing 70% of global forest loss (FAO, 2010). Between 2000 and 2010, approximately 1.5 billion metric tons of annual global carbon emission was due to forest loss, although increments are not expected in the near future (Friedlingstein *et al.*, 2010). Consequently, deforestation has been identified as a key driver of the current climate change (Shukla *et al.*, 1990), raising the need for research on the drivers of carbon cycling and cross-forest differences in terms of carbon storage. In this context, tropical evergreen forests store much more living carbon per unit area than seasonal ones, most of it stored in the above-ground biomass (Poorter *et al.*, 2015; Sullivan *et al.*, 2017) but human disturbances are likely to alter forest potential for carbon storage in ways that are not completely understood yet (Wright, 2010). In addition to deforestation, other human-related disturbances may disrupt the carbon-storage services provided by tropical forests, such as biomass collapse due to edge effects (Laurance *et al.*, 1997; Dantas de Paula *et al.*, 2011), particularly when human disturbances convert intact forest landscapes into small fragments (Tabarelli *et al.*, 2004; Aide and Grau, 2004; Wright, 2005). In human-modified landscapes, remaining forest cover consists preferentially of edge-affected habitats (Santos *et al.*, 2008). The edge-affected habitats become progressively impoverished in terms of large-seeded (Santos *et al.*, 2008), understorey and shade-tolerant (Tabarelli *et al.*, 1999), vertebrate-dispersed and vertebrate-pollinated tree species (Girão *et al.*, 2007; Lopes *et al.*, 2009) heavy-wooded (Michalski *et al.*, 2007), outbreeding species (Girão *et al.*, 2007), supra-annual (Tabarelli *et al.*, 2010) and large-tree species (Oliveira *et al.*, 2008).

Large tree species usually represent only about 10% of the total tropical species richness, but can have a disproportional influence on forest structure and ecosystem functioning (Peres, 2000; Vieira *et al.*, 2004). Emergent trees store a large portion of above-ground biomass, contributing decisively to other ecological services such as nutrient cycling (Vieira *et al.*, 2004), water catchment, soil erosion control (Myers, 1997) biodiversity retention (Camargo *et al.*, 1994; Oliveira *et al.*, 2008) and provision of forest products (Thiollay, 2003). Unfortunately, emergent tree species are susceptible to wind turbulence and physiological stress in edge-affected habitats, which may result in increased mortality (Laurance *et al.*, 2000; D'Angelo *et al.*, 2004), and reduced recruitment of trees of large stems from saplings (Vieira *et al.*, 2004). Additionally, selective logging can also reduce populations of emergent tree species in fragmented landscapes (Pinard and Putz, 1996).

Conversion of tropical forests into human-modified landscapes can thus reduce the abundance of large trees, profoundly affecting the ecological services provided by tropical forest ecosystems. Habitat fragmentation and the subsequent establishment of edge-affected habitats (forest edges and fragments) drastically limit forest capacity for carbon storage across human-modified landscapes since the loss of carbon due to reduced abundance of large trees is not compensated by either canopy or understorey species (Dantas de Paula *et al.*, 2011). This is particularly true if canopy and understorey tree species are unable to benefit from the loss of large tree populations to compensate for edge-related carbon loss (Dantas de Paula *et al.*, 2011). Information is lacking on the patterns of carbon retention and distribution across human-modified forest landscapes in tropical Africa including Kenya. This study investigated the spatial patterns of carbon stock distribution in highly fragmented forest patches. The study also tested whether there is a general reduction in the proportion of large trees, or a greater chance of them to fall over closer to the edge or in the interior section of the forest fragments.

2.9 Soils characteristics, forest fragmentation and species diversity

According to Baimas-George (2012), deforestation has a strong negative effect on soil fertility and health. Deforestation generates forest edges which amplify disturbance within the forest and alter the ecosystem functioning. Such edge effects include greater aridity, lower soil moisture levels, dehydration, and damaging winds (Laurance *et al.*, 2000). Typically, tree mortality allows solar radiation and winds to penetrate further into the forest, reaching previously unexposed areas (Laurance *et al.*, 1998). The increased solar radiation and wind can cause soil drought, lowering the amount of available water for vegetation (Camargo and Kapos, 1995).

The major differences in species composition may be a function of changes in soil nutrients and the proximity to the forest edge (Aerts *et al.*, 2006). Soil nutrient health drastically changes at the forest border due to edge effects. According to Didham (1998), there is increased litter decomposition rate in exterior forest floor soils. Increased litter decomposition on the forest edge increases the rates of net nitrogen mineralization leading to more available, mobile nitrogen. However, after disturbance, plants are unable to take up the mineralized nitrogen efficiently leading to nitrogen loss due to leaching (Aerts and Chapin, 2000). Studies have shown that forest edge soils have significantly lower percentage nitrogen and carbon than forest interior soils (Toledo-Aceves and García-Oliva, 2007).

Furthermore, diminished phosphorus level has been reported in soils on the forest edge indicating that available phosphorus is inversely related to nearness to edge of the forest. Reduced phosphorus on the forest edge has been attributed to increased occlusion and subsequent erosion (Toledo-Aceves and García-Oliva, 2007). Thus, it has been concluded that the forest floor exterior soils would have lower nutrient levels than interior forest floor soils. The effects of forest edge on soil moisture and nutrient levels in fragmented tropical Afromontane forests have not been comprehensively studied. The ‘edge effect’ on soil physical and chemical properties (soil moisture, nitrogen, phosphorus, potassium, pH, electrical conductivity and carbon) was evaluated along the gradient from the forest edge to the forest interior and related to species richness, diversity and distribution.

2.10 Research gaps

Previous studies indicate that forest fragmentation causes important changes in the dynamics of forests, especially within 100-150m of habitat edges (Laurance *et al.*, 1998). Core-area model has predicted that edge effects increases rapidly in the order of importance once fragments fall below 400 ha in area, depending on the fragment shape (Laurance *et al.*, 1998). All forest fragments in Taita Hills fall below 400 ha in area and therefore likely to be heavily affected by increased forest edge created because of forest fragmentation. Although a number of studies in East Africa have dealt with the influence of forest fragmentation on plant community dynamics (Farwig *et al.*, 2006; Pellikka *et al.*, 2009 and Ojoyi *et al.*, 2014), the focus has been on the effect of fragment area/size on tree species in relation to species composition, diversity, species richness and regeneration potential.

Studies on the edge effect on species richness, diversity, abundance of dominant and adaptable tree species and carbon stock dynamics in mountainous forests are extremely limited. However, there is evidence that edge effect stands out as the most important and influential factor causing biodiversity loss (Laurance *et al.*, 1998). In fact, a review of literature on forest fragmentation in Kenya clearly indicates that the understanding of the effects of fragmentation on forest cover change, tree diversity and above-ground carbon stocks is poor and only limited to the differences in the forest structure and species composition of different forest patches occurring in fragmented landscapes. There is little information available on the interactions of species due to altered disturbance occasioned by fragmentation and yet forest managers and

policy makers require this information for effective management of the fragmented forest ecosystems. In addition, detecting forest cover change is necessary to understand the intensity and the extent of fragmentation with the aim of developing management strategies for sustainable conservation of fragmented forests rich in biodiversity. This study endeavoured to bridge these existing knowledge gaps by generating information required to improve the management of fragmented forests that are rich in biodiversity.

2.11 Theoretical framework

Island Biogeography Theory (IBT), meta-population models and source-sink dynamics have identified three effects of forest fragmentation leading to the loss of biodiversity (Zuidema *et al.*, 1996; Jacquet *et al.*, 2017). These effects focus primarily on the effects of pattern on processes in the forest remnants and include; reduction in forest area, increased isolation of forest fragments and creation of forest edge (Figure 1). Each of these effects influences a range of population, community and ecosystem processes that affect biodiversity.

The area of the forest cleared and the resulting patch sizes may constrain the types, extents, frequencies and even intensities of disturbances. Some types of disturbances depend directly on area; for example, small forest fragment in non-forested landscapes may be susceptible to wind destruction than comparably sized areas in continuous forest (Esseen, 1994). Wind can affect the dynamics of species due to increased tree fall and hence resulting into biodiversity loss (Malanson, 1984).

The degree to which area-related changes occur is a function of the total amount of forest loss, the size of remaining forest remnants and location of destruction in a landscape. Pattern of deforestation across a landscape is not random (Thiollay and Meyburg, 1988; Kupfer and Franklin, 2000), so the spatial selectivity of forest clearing affects the degree to which forest fragments represent the original forest.

Forest fragmentation usually results into increased susceptibility of forest fragments to edge effects. Consequently, changes in microclimate, forest structure, biotic composition and ecological functioning occur along forest edges exposed to non-forested habitats. The differentiation of the edges is primarily due to microclimatic changes such as incoming solar radiation, temperature, wind speed and evapotranspiration (Ranney *et al.*, 1981). Vicissitudes in abiotic conditions along edges automatically triggers biotic responses. Edge species are adapted

to edge microclimate and are often species found in the early stages of forest succession in a given region (Kupfer and Malanson, 1993). The responses of forest interior species to conditions that develop along the newly created edge differ, but some species are unable to survive in the newly created micro-climatic conditions on the forest edge. The enhancement of alien species invasions may also occur due to habitat destruction (Medley, 1997). This is because many invasive tree species are prolific seed producers that thrive in higher light conditions and have widely dispersed seeds. These traits make them more likely to establish and thrive along forest edges, after which they may be able to play an increasing role in vegetation dynamics within fragment interiors (Burke and Nol, 1998).

Moreover, edge effects alter species interactions by increasing the degree of interaction among edge and interior species (Fagan *et al.*, 1999; Gustafson *et al.*, 2002). The synthesis of empirical results concerning the effects of forest edge on population dynamics occur in the literature (Gustafson *et al.*, 2002). There is also evidence of contemporary theoretical developments outlining how species interactions and the dynamics of the communities can be changed (Fagan *et al.*, 1999). The theory that habitat fragmentation causes alterations in forest composition leading to reduced tree diversity and capacity to sequester carbon needs substantiation. This study focused on the effects of reduction in forest area and creation of forest edge on forest cover dynamics and ecosystems processes. The forest cover dynamics studied included species diversity, species richness and abundance of the key dominant and adaptable tree species while the ecosystem process studied was carbon sequestration potential.

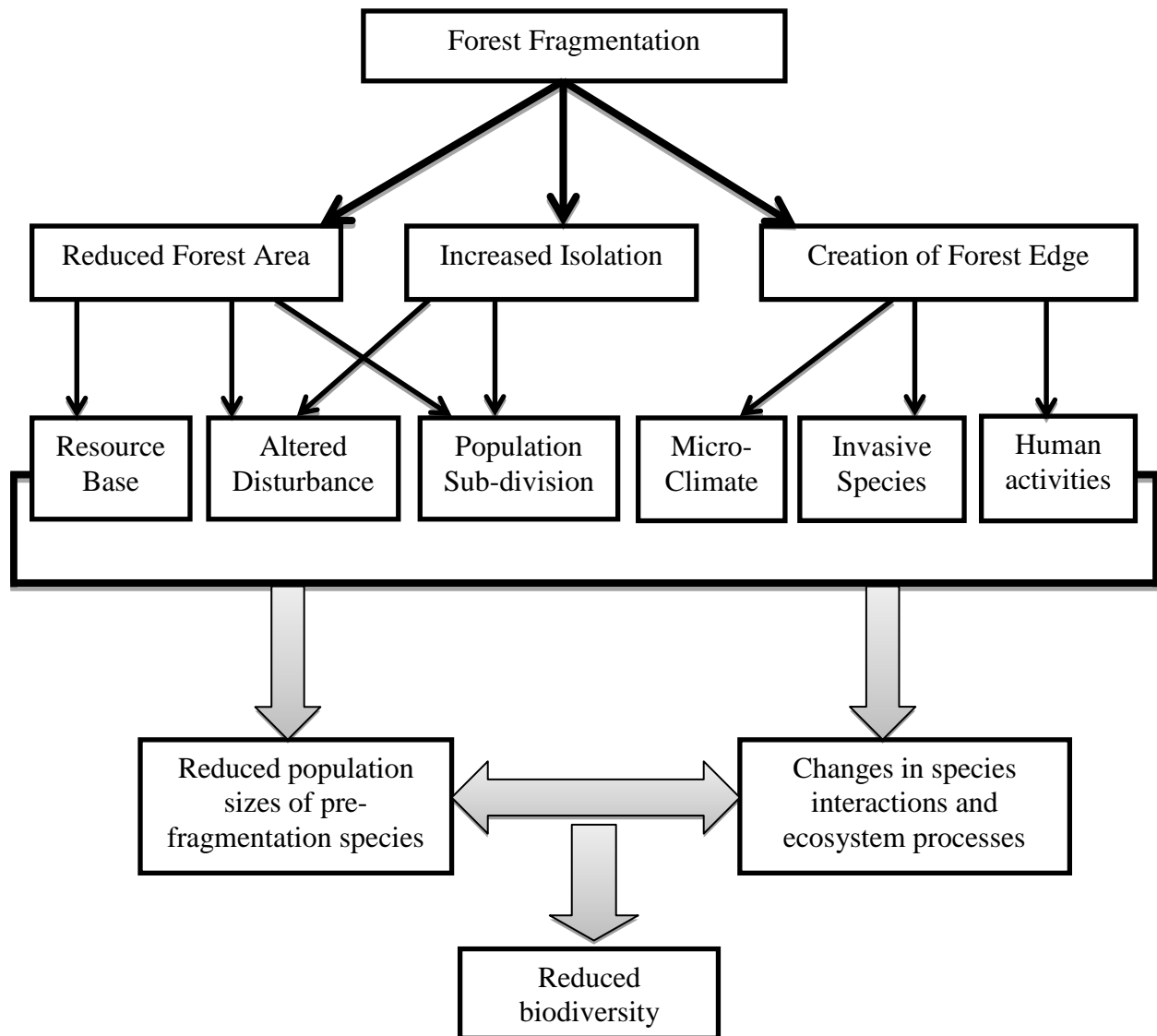


Figure 1: Theoretical framework (Source: Kupfer *et al.*, 2006)

2.12 Conceptual framework

The hypothesis that fragmentation causes forest degradation and damages the habitat spurring secondary succession was tested in this study. The process of secondary succession is more manifested on the forest edge in a fragmented forest ecosystem. This is because increased edge effect attributed to fragmentation affects the forest structure and influences the dynamics of tree species abundance and diversity, species richness and above-ground carbon stock in fragmented forest patches (Figure 2). The effect of forest edge on the forest structure, tree species diversity, richness, abundance of dominant and adaptable tree species and above-ground carbon stock was measured as a function of changes in edaphic factors along the forest edge-interior gradient.

In addition to the forest structure, tree species richness and diversity, edaphic factors such as moisture content, nitrogen, phosphorous, potassium, pH, electrical conductivity and organic carbon are indicators of forest ecosystem health. Consequently, the relative importance of soil physical and chemical characteristics was evaluated to provide an explanation of the differences in tree species diversity, richness, abundance of dominant and adaptable tree species and above-ground carbon stock in relation to distance from the forest edge to the interior for spatially differentiated forest fragments of varying sizes ranging from 1.0 to 200.0 ha. The determination of the annual rate of reduction in the area under the forest fragments to test space-time substitution effects established the magnitude of forest cover loss due to fragmentation. Spatial patterns of above-ground carbon stock distribution in highly fragmented forest patches were also investigated by testing whether there is a general reduction in the proportion of large trees, or a greater chance of them to fall over closer to the edge or in the interior section of the forest fragments. This helped to explain why above-ground carbon stock per unit area varied between small and large forest fragments along the forest edge-interior gradient.

Fragment size, distance from the forest edge, edaphic factors and the rate of change in forest cover/area were the independent variables while forest degradation was the dependent variable. Forest structure, species diversity, species richness, abundance of dominant and adaptable tree species and above-ground carbon stock were the indicators of forest degradation, which was the dependent variable in this study. The policy intervention was the intervening variable influencing the relationship between the independent variables and the dependent variable and hence producing interaction effect.

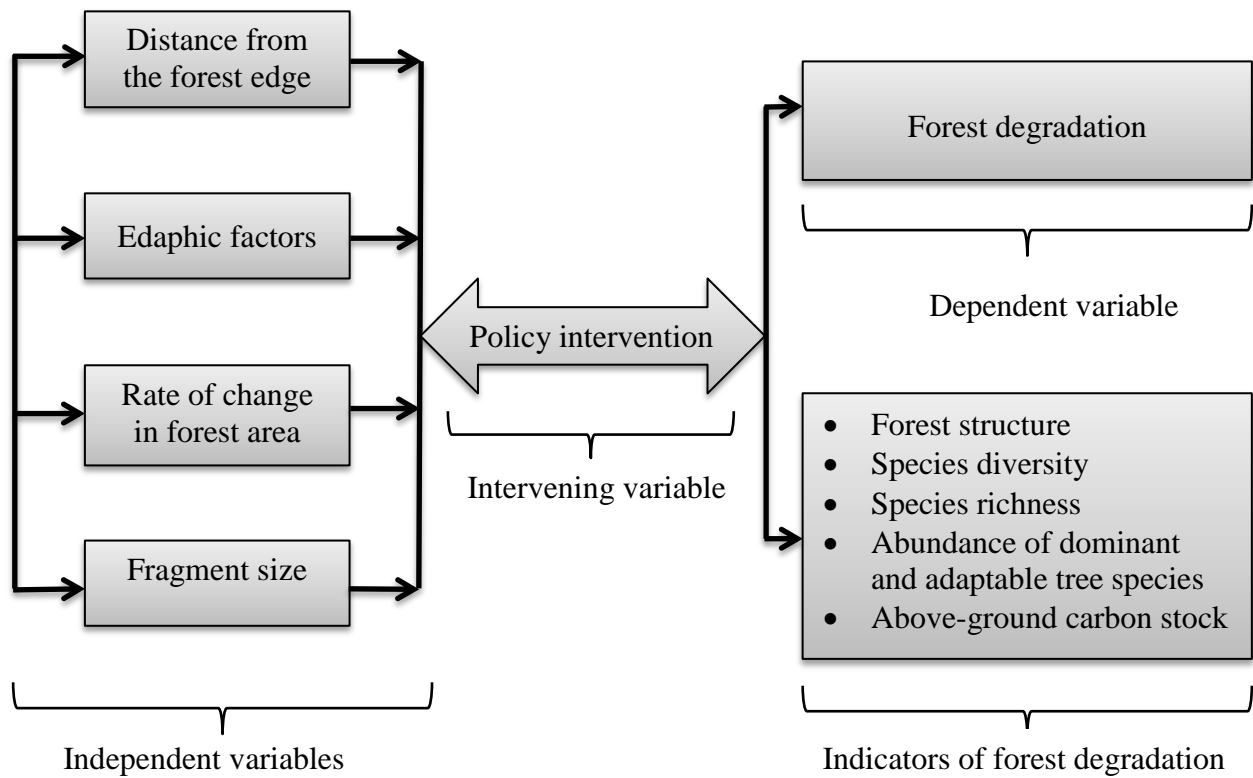


Figure 2: Conceptual framework

CHAPTER THREE

RESEARCH METHODOLOGY

3.1 Description of the study area

The study took place in Taita Hills, Taita Taveta County (Figure 3). The identified study site has a high degree of endemism and conservation value because it is part of the Eastern Arc Mountains (Myers *et al.*, 2000; Burgess *et al.*, 2007; Hall *et al.*, 2009). Besides, the existing forest fragments in Taita Hills have been isolated from one another, embedded in agricultural landscape for over 100 years, and hence ranked among the most threatened biodiversity hotspots globally (Newmark, 1998; Pellikka *et al.*, 2009). Currently, indigenous evergreen forests cover about 430 ha, reflecting about 98% forest reduction over the last 200 years (Adriaensen *et al.*, 2003). Despite the small size and fragmented nature of the remaining indigenous evergreen forests with over 90% loss during the last 200 years (Pellikka *et al.*, 2009), Taita Hills continues to boast an outstanding diversity of flora and fauna and very high level of endemism among vertebrates, invertebrates and plants and hence offered an exceptional site to undertake this particular study.

3.1.1 Location and size

The Taita Hills forests (3°25'S, 3°820'E) are located in Southeast Kenya, 25 km West of Voi town in Taita Taveta County (Figure 3). The County covers a total surface area of 17, 084 km². It borders Tana River, Kitui and Makueni Counties to the North, Kwale and Kilifi Counties to the East, Kajiado County to the Northwest and the Republic of Tanzania to the South and Southwest. Taita Hills rise abruptly from the semi-arid Tsavo plains at 600-700 m.a.s.l. to a series of ridges, reaching 2208 m.a.s.l. at Vuria peak. The plains isolate the Hills from other mountains and highland blocks, the closest being Mount Kasigau (50 km to the Southeast), North Pare (70 km Southwest), Mount Kilimanjaro (110 km West) and the West Usambara mountains (120 km South) (BirdLife International, 2007).

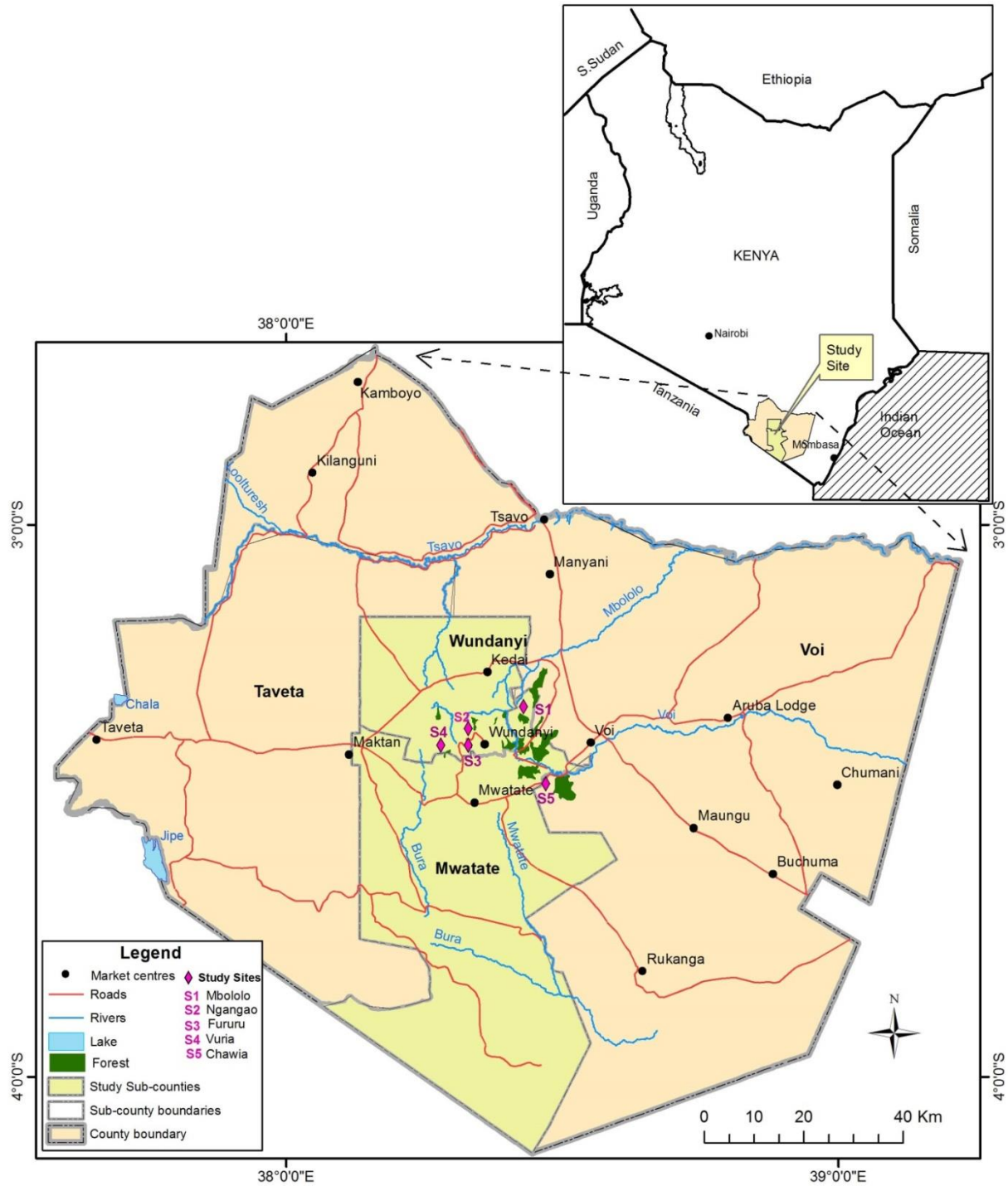


Figure 3: Location of the study sites in Taita Taveta County (Source: RCMRD, 2005)

3.1.2 Study sites

The study involved five forest fragments in Taita Hills exhibiting a wide range of conditions to address critical knowledge gaps on the effects of forest fragmentation on forest structure, tree species richness and diversity and above-ground carbon stock. These forest fragments are Mbololo, Ngangao, Chawia, Fururu and Vuria. The forest fragments are in close proximity to a densely populated agricultural landscape. While Mbololo forest is less disturbed and difficult to access, Ngangao is moderately disturbed, easily accessible and close to Wundanyi town. On the other hand, Chawia forest is heavily disturbed because it is surrounded by intensively cultivated farmlands on the North, and borders highly populated Mwatate and Ng'ambwa towns to the East and West respectively. Fururu and Vuria are also heavily disturbed and easily accessible from all directions. Nevertheless, Vuria is very steep making it difficult to access the top of the forest. The common feature in all the selected forest fragments is the existence of small portions of plantations of exotic species mainly of Pinus, Cypress and Eucalyptus.

3.1.3 Climate, geology and soils

Taita Hills are located within the Inter-tropical Convergence Zone. The Köppen climate classification for Taita Hills is tropical savanna climate (Köppen, 1936). The rainfall pattern is bimodal. The long rain season occurs from March to May with a short rain season between November and December, but the mist and cloud precipitation is a year-round phenomenon. The annual average rainfall is 1132mm in Mgange at 1768 m.a.s.l. and 587mm in Voi town at 560 m.a.s.l. The yearly maximum rainfall could reach 2000 and 200mm for Mgange and Voi respectively. Owing to the orographic rainfall pattern, Southeastern slopes of the hills receive more precipitation than Northwestern slopes. Evergreen forest fragments are restricted to areas receiving over 900mm of annual precipitation, being above 1400m altitude at the Southeastern slopes and above 1700m altitude at the Northwestern slopes (Jaetzold and Schmidt, 1983). Therefore, the five forest fragments studied occur between 1604 and 2200 m.a.s.l and exhibit similarity in climatic conditions as demonstrated by the resemblance in agricultural crops grown in the matrices separating the fragments.

The average temperature in Taita Taveta County is 23°C with variations between 18°C and 24.6°C with the hills experiencing lower temperatures of 18.2°C compared to the lower

zones with an average temperature of 24.6°C (Republic of Kenya, 2013). The relative humidity ranges from 79 to 83 percent (Republic of Kenya, 2013).

The soils are predominantly Cambisols originating from weathered gneiss and are often gravely to sandy-loamy and shallow (Sombroek *et al.*, 1982). The soils are well drained and moderately fertile. Majority of Cambisols are in a transitional stage of development from a young soil to a mature one (Sombroek *et al.*, 1982). On the steep slopes and transitional zones, the dominant soil types are regosols, which are shallow soils, have high permeability and low water holding capacity (Sombroek *et al.*, 1982). Luvisols, Acrisols and Arenosols soils that are of moderate to low in fertility, characterize the drier foothills bordering the hills. In valley bottoms, alluvial soils (fluvisols) are present. The variations in edaphic factors in the forest fragments along the forest edge-interior gradient were assessed and related to tree species richness and diversity.

3.1.4 Human settlement and economic activities

The population of Wundanyi and Mwatate Sub-Counties where the forest fragments are located is 127,534 people accounting for 44.8% of the total Taita Taveta County population (Republic of Kenya, 2009). Majority of the population lives in Wundanyi, Mwambirwa and Tausa, which are high altitude areas whose altitude ranges between 1000 and 2000 m.a.s.l. These are high potential agricultural areas reflecting past and present high population growth rates, leading to increased land fragmentation and land shortage. Foothills and lowlands are drier and hence sparsely populated.

Several economic activities occur in the areas surrounding the fragmented forests. Crop farming and livestock keeping are the most common economic activities due to favourable climatic and edaphic conditions. The farmlands surrounding the hills are intensively cultivated with maize, beans, tomatoes, peas, cassava, mangoes, cabbages, bananas and sweet potatoes. The extensive agriculture and grazing on the foothills and plains surrounding the hills coupled with the high population pressure has caused dynamic changes in the land use patterns, leading to serious forest degradation and deforestation. Tourism also contributes to the economy of the local population (Falcetto, 2012). Some of the tourist attraction sites include Tsavo West and Tsavo East National Parks that have rich biodiversity of wild animals and birds. Other economic activities include mining, formal employment and small-scale businesses.

3.2 Research design

Manipulative experimental approaches are exceedingly difficult to apply in studies to assess biological consequences of forest fragmentation because of difficulties related to control and replication (Mcgarigal and Cushman, 2002). Only a few independent variables can be directly controlled (Mcgarigal and Cushman, 2002). This lack of control of independent variables, coupled with the problem of obtaining independent replication, makes it difficult to obtain unbiased and reproducible results from field experiments (Diamond, 1986). Besides, manipulation of natural landscapes is laborious and costly (Debinski and Holt, 2000). Given the complexities and logistical difficulties associated with manipulative experiments, they are often not practical or feasible for studying species dynamics in complex and fragmented ecosystems (Debinski and Holt, 2000; Mcgarigal and Cushman, 2002). In such circumstances, comparative mensurative experiments provide an alternative means of gaining reliable knowledge if properly designed (Mcgarigal and Cushman, 2002).

Consequently, comparative studies were undertaken to evaluate the effects of fragmentation on tree species richness and diversity, abundance of dominant and adaptable tree species and carbon sequestration potential in five forest fragments that have undergone different levels of disturbance and vary considerably in area/size (Omoró *et al.*, 2010). The approximate area of the forest fragments are: Mbololo (200.0 ha), Ngangao (120.0 ha), Chawia (86.0 ha), Fururu (5.0 ha) and Vuria (1.0 ha). Mbololo is the largest continuous forest exhibiting low level of disturbance while Ngangao, the second largest fragment has undergone medium disturbance (intermediate levels of disturbance). The level of disturbance in Chawia is high because it is surrounded by intensively cultivated farmlands and highly populated Mwatate and Ng'ambwa towns. Fururu and Vuria exhibit extremely high level of disturbance (Beentje, 1988; Pellikka *et al.*, 2009). The identified forests have been fragmented for a long time providing the opportunity for the fragments dynamics to equilibrate and this reduced the risk of reaching erroneous conclusions usually associated with short-term biological responses (Wilder *et al.*, 1998; Adriaensen *et al.*, 2003; Pellikka *et al.*, 2009). Besides, the forest fragments have similar physical environment, are less than 21 km apart and used to have similar plant communities' characteristics of the landscape. All the fragments selected for the study are in close proximity to human settlements and are spatially independent as they are located at least 4 km apart from one another.

The effect of fragment size that was greater (large fragments) or smaller than 40 ha (small fragments) on species richness, diversity and abundance of dominant and adaptable tree species and above-ground carbon stock was determined. Mbololo, Ngangao and Chawia represented replicates of large forest fragments whereas Fururu and Vuria represented replicates of small forest fragments. The effect of large and small forest fragments along the forest edge-interior gradient were evaluated for forest edge-fragment size associated effects on species richness and diversity, abundance of key dominant and adaptable species and carbon stock.

3.2.1 Sampling strategy

Belt transects of different dimensions proportionate to the fragment size were established for data collection (Ibarra-Macias, 2009). The size and number of belt transects varied from one fragment to the other depending on the size of the forest fragment as described in Table 1. To normalize the differences within the forests, transects were laid up slope from the edge of the forest to the forest interior and data on tree species richness, diversity and abundance of dominant and adaptable tree species, carbon stock and edaphic factors were collected along the gradient in plots established within the transects.

Table 1: Sampling frame and sampling intensity. The size and the number of transects are proportional to the fragment size

Forest Fragment	Area (ha)	Transect Dimensions (m)		Number of Transects	Number of plots
		Length	Width		
Mbololo	200.0	150.0	40.0	9	27
Ngangao	120.0	120.0	40.0	5	15
Chawia	86.0	90.0	40.0	4	12
Fururu	5.0	60.0	40.0	3	9
Vuria	1.0	30.0	40.0	3	9

3.3 Data collection methods

Forest structure, tree species richness and diversity and above-ground carbon stock data was obtained through intensive field assessment of the five forest fragments using standard protocol (Kindt and Coe, 2005) while forest cover change data was obtained by analyzing

twenty-five Landsat imageries of 1973, 1987, 2001, 2012 and 2016 using methods described by Young *et al.* (2017). The methods are described in details in the subsequent sub-sections.

3.3.1 Assessment of forest cover change using Landsat images

Landsat satellite images namely Multispectral Scanner, Thematic Mapper, Enhanced Thematic Mapper, and Operation Land Imager and Thermal Infrared Sensor (MSS, TM, ETM+ and OLI and TIRS) taken in 1973, 1987, 2001, 2012 and 2016 were used because they are readily available, cheap and covered the intended period of study. The United States Geological Survey (USGS) website (<https://www.usgs.gov>) for polygon overlays provided the images used in this study. The best images were selected based on cloud cover (<10%) and periods of interest and analysed to give in-depth insights on the trends in forest cover change due to fragmentation. The rate of forest cover change was also analysed to determine the periodic percentage loss. In addition, space-time substitution effect was analysed to ascertain which forest patch had fragmented faster. In addition, the satellite images were analysed for undisturbed and disturbed sites as well as open grounds created within the forest due to natural phenomena and anthropogenic activities. The beginning of the study of forest cover change coincides with the year 1973, which marks the phase when forest management in Taita Hills was not well coordinated. It was until 1977 that well-coordinated forest management started following a Presidential decree banning logging in the forests without a permit (Beentje, 1988).

Landsat MSS images used had four spectral bands, spatial resolution of 60m and a swath of 165x165m. Landsat TM images consisted of seven spectral bands, resolution of 30m and scene size (swath) of 185x185m. Landsat ETM+ images had eight spectral bands with a spatial resolution of 30m for bands 1-7 and 15m for band 8, which is a panchromatic band. The swath for ETM+ was 185x185m. Landsat 8 Operational Land Imager (OLI) and Thermal Infrared Sensor (TIRS) images comprised of nine spectral bands with a spatial resolution of 30m for bands 1-7 and 9. The resolution for band 8 (panchromatic) was 15m. The OLI and TIRS images had a swath of 185x185m.

3.3.1.1 Pre-analysis of the images

The Landsat images were georeferenced using permanent features detectable on the ground (Griffiths *et al.*, 2014). Landsat MSS was resampled to 30 meter resolution (Studley and

Weber, 2011) to conform to other images and account for the differences in spatial and spectral resolutions when creating forest cover change maps. The comparison of the images from different sensors and dates for detection of changes in forest cover was made possible through precise co-registration and orthorectification. Earlier images for each scene were co-registered onto the appropriate recent image for correction of image shifts (Muturi *et al.*, 2010; Song *et al.*, 2015). The orthorectification corrected for the pixel displacement caused by the topographical variations at the off-nadir viewing and made the image orthographic to ensure that every pixel was in its correct location regardless of elevation and viewing direction, and reduced sensor system and terrain related errors by transforming the image to a particular projection and reference system (Young *et al.*, 2017).

3.3.1.2 Analysis of the images

The images were processed in IDRISI Image Processing Software (Clark Labs, Clark University, Canada) to produce false color composites using 4, 3 and 2 band combination for red, green and blue for display and visual analysis (Griffiths *et al.*, 2012; 2013). The composited images were enhanced for clear vision to distinguish the forests features for interpretation. Supervised image classification using ArcGIS 10.2 Software (ESRI, Redlands, CA, USA) categorized the forests into different disturbance classes (disturbed, undisturbed and open ground) for logical and statistical analysis to determine time-series changes in forest cover between 1973 and 2016. Post classification change detection method was applied to map forest cover change between 1973 and 2016.

From the 2016 maps, three points were randomly selected from each of the disturbance classes (disturbed, undisturbed and open ground), and their coordinates entered in hand held Global Positioning System (GPS). In total, 45 ground truthing points were used for accuracy assessment to determine the overall individual forests' accuracies. The selected points were tracked in the field, and the land cover compared to that in the maps for the same location. The number of ground truthing points whose land cover in the map corresponded to that on the ground was expressed as a percentage to establish the reliability of the created maps.

3.3.2 Assessment of forest structure, species richness, diversity and abundance

The assessment of tree species richness, diversity and abundance of key dominant and adaptable tree species was undertaken in plots established within the belt transects. Within each belt transect, plots of 10.0x10.0m were systematically established and replicated three times in a stratum at an interval of between 10.0 and 50.0m along the forest edge-interior gradient depending on the size of the forest fragment. All the trees within the 10.0x10.0m plots were identified by a plant taxonomist and recorded. The height of the trees identified within 10.0x10.0m plots was measured using a laser rangefinder. Only trees with a diameter at breast height (DBH) greater than 2.5m were measured for their DBH using a diameter tape.

Moreover, six dominant and adaptable tree species were investigated along the gradient from the forest edge to the forest interior to establish their population changes, frequency and distribution. The selected species are indicator species sensitive to environmental changes and quickly respond to adverse ecosystem changes (Omoro *et al.*, 2010; Aerts *et al.*, 2011). The shift in equilibrium in the occurrence and distribution of these species can give an early warning that the habitat is under stress and hence are used as a proxy to diagnose the health of natural forest ecosystems (Madoffe *et al.*, 2005; Rogers *et al.*, 2008; Aerts *et al.*, 2011). The six tree species are *Macaranga conglomerata*, *Albizia gummifera*, *Syzygium guineense*, *Xymalos monospora*, *Tabernaemontana stapfiana* and *Maesa lanceolata*. These species are associated with different levels of forest disturbance and hence are exceptional indicators of the changes occurring along the gradient from the forest edge to the forest interior (Omoro *et al.*, 2010). While *Macaranga conglomerata* and *Albizia gummifera* are pioneer species, *Syzygium guineense* and *Xymalos monospora* are associated with low level of disturbance. *Tabernaemontana stapfiana* and *Maesa lanceolata* are secondary native species associated with heavy disturbance (Omoro *et al.*, 2010; Aerts *et al.*, 2011).

The measure of diversity used was Shannon-Wiener diversity index (Shannon and Wiener, 1963; Krebs, 1999):

s

$$H = -\sum (P_i * \ln P_i)$$

$i=1$

Where,

- H = Shannon Weiner diversity index
 P_i = Fraction of the entire population made up of species i
 S = Numbers of species encountered

3.3.3 Quantification of above-ground carbon stock in trees

The same plots and transects used for assessment of the forest structure, trees species richness and diversity were utilized for carbon stock quantification. Twenty-seven plots in Mbololo, 15 (Ngangao), 12 (Chawia) and 9 each in Fururu and Vuria were used in the assessment of above-ground carbon stock. To estimate carbon stock, three variables i.e. DBH, tree height and wood density were used. Consequently, within each 10.0x10.0m plot, one tree with DBH>10.0cm was randomly picked for each of the representative species and cored to obtain wood samples for density (specific wood gravity) determination in the laboratory using displacement method. The determined default wood density (gcm^{-3}) for each of the tree species together with height and DBH were parameters used in improved allometric models developed by Chave *et al.* (2014) to calculate the above-ground carbon stock. The improved allometric models (Chave *et al.*, 2014) are the product of a global database of directly harvested trees at 58 sites, spanning a wide range of climatic conditions and vegetation types in the tropics including Tanzania, Zambia, Angola and South Africa. The degree of uncertainty associated with previously published Pan-tropical above-ground biomass allometries where none of the destructively sampled trees used to develop biomass models was from Africa (Chave *et al.*, 2005) was eliminated in the new improved models (Chave *et al.*, 2014). The new allometric models provide accurate carbon stock estimates across forest types and bio-climatic conditions. The allometric equation used for estimating carbon stock was given by:

$$AGB_{est} = 0.0673 X (\rho D^2 H)^{0.976}$$

Where,

- D = Diameter at breast height (cm)
 H = Tree height (m)
 ρ = Wood specific gravity (gcm^{-3})

3.3.3.1 Determination of wood density

Weighing of green wood cores was done at collection points using a weighing balance and the weight in grams recorded. The green wood core was then taken to the laboratory, and placed into an oven and dried to constant weight at 105°C. The final constant weight (oven dry mass) was recorded. The water displacement method was used to determine the volume of wood core by immersing it into a beaker of water loaded on a top-loading electronic weighing balance. The volume of the displaced water obtained by pressing the wood core with the aid of a needle was recorded (Williamson and Wiemann, 2010). Thus, the green volume of the wood core before drying and the dry volume after drying were determined and the specific gravity (SG) of wood also referred to as wood density calculated using the following formula:

$$\text{Oven dry SG} = (\text{Oven dry mass} / \text{Oven dry volume}) / \rho_{\text{water}}$$

Whereby:

SG = Specific gravity

Oven dry mass = Mass after drying at (101 - 105 °C)

ρ_{water} = Density of water which is 1.000 gcm⁻³ at 4.4 °C

3.3.4 Soil sampling and analysis

Soil sampling was conducted in June 2015 and 2016, one week after the last measurable rain event and so the surface soils had high levels of moisture. Three replicate soil samples were randomly collected at two depths of 0-15cm and 15-30cm within 10x10m plots using soil auger. The soil samples were composited to obtain a representative sample for the plot, put into zip lock plastic bags and transported to the KEFRI Soil Science Laboratory for analysis to determine the levels of moisture, nitrogen, phosphorus, potassium, pH, electrical conductivity and organic carbon. In total, 432 soil samples were collected from the five forest fragments. Mbololo, the largest forest fragment had the highest number of soil samples (162) followed by Ngangao and Chawia which had 90 and 72 samples respectively. Fururu and Vuria had 54 soil samples each.

Prior to the analysis, the soil samples were air-dried and sieved using 2.0 mm sieve. Phosphorus and nitrogen were determined using Olsen and Kjeldahl methods respectively (Okalebo *et al.*, 2002). Measurements of potassium involved complete oxidation of soil samples using Kjeldahl procedures followed by photometric analysis using photoelectric flame photometer (Okalebo *et al.*, 2002). Organic carbon was determined using Walkley-Black method through

complete oxidation (Walkley and Black, 1934). Determination of soil pH and electrical conductivity (EC) involved the use of soil pH meter and electrical conductivity meter respectively as prescribed by Anderson and Ingram (1993) and Okalebo *et al.* (2002). Soil moisture content was determined using oven-dry method (Okalebo *et al.*, 2002). Detailed laboratory procedures on soil analysis for the moisture content, pH, electrical conductivity, phosphorus, nitrogen and potassium are elaborated in sub-sections 3.3.4.1, 3.3.4.2, 3.3.4.3, 3.3.4.4 and 3.3.4.5 respectively. The results of soil moisture and nutrients analysis provided explanation of the differences in species diversity, richness, abundance of dominant and adaptable tree species and carbon stock in relation to distance from the forest edge and ascertain the effect of forest edge on forest health and biodiversity.

3.3.4.1 Determination of soil moisture content

Cored soil samples collected from the field were analysed for moisture content using oven-dry method. The soil fresh weight and core sampler (W_1) was measured and recorded. The samples were then oven dried at 105°C for 24 hours and allowed to cool in a desiccator for 30 minutes. The soil dry weight and core sampler (W_2) was then measured. The core samplers were emptied by removing the soil and their weight measured (W_3). The percentage moisture content in the soil was calculated using the following formula:

$$\text{Soil moisture (\%)} = \frac{W_1 - W_2}{W_2 - W_3} \times 100$$

Whereby:

- W_1 = Weight of core + fresh soil
- W_2 = Weight of core + dry soil
- W_3 = Weight of core

3.3.4.2 Determination of soil pH and electrical conductivity

Twenty grams (20 g) soil sample was weighed into 100 ml plastic bottles and 50 ml distilled water added. The contents were shaken for 30 minutes, and allowed to stand for 30 minutes. The pH was measured using a pH meter (Model 691) and electrical conductivity measured with conductivity meter (Model TOA Cm-20S).

3.3.4.3 Determination of phosphorus in the soil

Air-dried soil weighing 2.5 g was mixed with 50 ml of 0.5 M sodium bicarbonate solution of pH 8.5 and shaken for 30 minutes. The suspension was filtered through Whatman filter paper number 42 and an aliquot of the filtrate (10 ml) pipetted into a 50 ml volumetric flask. To this aliquot, 5 ml of 0.8 M boric acid was added and, finally, 10 ml of ascorbic acid added as the reducing agent. The contents were then diluted to the 50 ml mark of the volumetric flask using distilled water, mixed well and let to stand for 1 hour to allow full colour development and the reading taken using UV spectrophotometer (Model UV Spectronic 21) to determine the available phosphorus in the soil in parts per million (ppm).

3.3.4.4 Determination of nitrogen in the soil

Finely ground air-dried soil weighing 0.3 g was transferred to clean labelled digestion tubes. A digestion mixture of 4.4 ml was prepared by dissolving 0.42 g of selenium (Se) powder and 14 g of lithium sulphate in 350 ml of 30% hydrogen peroxide and mixed well. Concentrated sulphuric acid (420 ml) was slowly added while cooling in an ice bath. Then, 4.4 ml of the mixture was transferred to the digestion tubes with samples and placed in a block digester (Skalar Block Digester System, Model SA 5640). The digestion tubes were heated at 360°C for 2 hours until the solution was clear. The contents were allowed to cool. After cooling, 25 ml of distilled water was added and contents transferred into a 50 ml volumetric flask and topped up to the 50 ml mark with distilled water. The contents were allowed to settle and a clear solution taken from the top of the tube for nitrogen analysis. An aliquot of 10 ml of the solution was transferred to the distillation tubes and 10 ml of 40% sodium hydroxide added for distillation. The extract was steamed immediately into 5 ml of 1% boric acid and 4 drops of mixed indicator added. Distillation was continued for 2 minutes from the time the indicator turned green. The distillate was removed and titrated with 0.1 M HCl until the colour changed from green to grey and finally to definite pink. The percentage nitrogen content in soil was calculated using the formula:

$$\text{Nitrogen content in the soil (\%)} = \text{Corrected ml of N/140HCl} \times 0.1 / \text{Weight of sample}$$

Whereby:

$$\text{Corrected ml of N/140HCl} = \text{Burette reading} - \text{The ml of N/140HCl required for the blank}$$

3.3.4.5 Determination of potassium in the soil

Air-dried soil (5 g) was weighed into 250 ml clean plastic bottle and 100 ml of 1.0 M neutral ammonium acetate solution added to extract maximum cations that occupy exchange sites on the soil surface and the contents shaken for 30 minutes using a mechanical shaker. The contents were filtered using Whatman No. 42 filter paper. 5 ml of the filtrate was pipetted into 50 ml volumetric flask and 1 ml of 26.8% Lanthanum chloride solution added and the contents made up to 50 ml mark using ammonium acetate solution. Standards containing potassium at concentrations 0.0, 2.5, 5.0, 7.5 and 10 ppm potassium were prepared similarly to fall within the measurable range of the calibrated flame photometer. The flame emission intensities were measured at 766 nm using flame photometer model Corning M 410 to determine potassium in the soil in parts per million (ppm).

3.3.4.6 Determination of organic carbon in the soil

Air-dried ground soil was passed through 0.5 mm sieve. 1 g of sieved soil was weighed into a 500 ml conical flask and 10 ml of 1N potassium dichromate solution added. The contents were shaken gently to mix. 20 ml of concentrated sulphuric acid was added to the mixed solution and the flask swirled 3 times. The contents were allowed to stand for 30 minutes, and 200 ml of distilled water, 1 ml of diphenylamine indicator and 0.5 g sodium fluoride added. The solution was back titrated with ferrous ammonium sulfate solution until the colour changed to brilliant green. The volume of the ferrous ammonium sulfate solution consumed was noted. A blank titration (without soil) was also carried out in a similar way and the volume of the ferrous ammonium sulfate solution consumed noted. Since this method has 77% recovery rate of organic carbon, a correction factor of 100/77 (1.3) was applied in calculating organic carbon content in the soil. Organic carbon content in the soil was calculated using the formula:

$$\text{Organic carbon (\%)} = (B-S) \times N_{FAS} \times 0.003 \times 1.3 \times 100/W$$

Whereby:

- B = Volume of ferrous ammonium sulfate consumed for blank titration in ml
- S = Volume of ferrous ammonium sulfate consumed for sample in ml
- N = Normality of ferrous ammonium sulfate from blank titration
- W = Dry mass of soil sample (g)

3.4 Data analysis

The field data was analysed using GenStat version 16.0 and CANOCO for Windows version 4.5 while Landsat images were analysed using ArcGIS version 10.2 Software (ESRI, Redlands, CA, USA). Testing for normality using Levene's test was necessary to decide on the appropriate statistical tests. Square root transformation of count data on the number of tree species per plot for each observation was performed to correct the problem of non-normality before parametric analysis. The continuous and transformed discrete data were subjected to One-way Analysis of Variance (ANOVA) to separate the means that were significantly different in relation to stand density, DBH, height, wood specific gravity, above-ground carbon stock, species richness and edaphic factors (moisture content, nitrogen, phosphorus, potassium, pH, electrical conductivity and organic carbon) within and among the five forest fragments along the forest edge-interior gradient. Least significant differences in means were separated using Tukey's post hoc test with a 5% probability significance threshold. Tree species diversity was calculated using Shannon-Wiener diversity index (H') and the values obtained subjected to One-way ANOVA to test the significant differences in species diversity along the forest edge-interior gradient within and among the forest fragments. The F-test isolated significant effects associated with the independent variables at 95% confidence interval.

Mean values and standard error of mean for stand density, DBH, height, wood specific gravity, above-ground carbon stock and edaphic factors (moisture content, nitrogen, phosphorus, potassium, pH, electrical conductivity and organic carbon) were estimated using parametric models.

To evaluate the relationship between soil physical and chemical characteristics along the forest edge-interior gradient, Spearman's rank correlation analysis was performed for the following variables; moisture content, nitrogen, phosphorus, potassium, pH, electrical conductivity, organic carbon and distance from the forest edge.

Canonical Correspondence Analysis (CCA) produced relationships between different tree species and the edaphic factors as well as species association along the forest edge-interior gradient. To avoid rare tree species distorting the results, only tree species that occurred in at least 6% of the plots were included in the analysis. Tree species classification involved two categories: most abundant and less abundant. Most abundant tree species were those species that occurred in at least 11% of the sampled plots while less abundant tree species were those that

occurred in at least 6% but less than 11% of the sampled plots. Prior to undertaking CCA, the length of the gradient, which is an estimate of species heterogeneity, was determined using Detrended Correspondence Analysis (DCA). When the length of the gradient from DCA is less than three standard deviations (SD), linear models such as Principal Component Analysis (PCA) and Redundancy Analysis (RDA) are appropriate. When the length of the gradient is greater than four SD, unimodal methods i.e. Correspondence Analysis (CA), Canonical Correspondence Analysis (CCA) and Detrended Correspondence Analysis (DCA) are appropriate. When the length of the gradient lies between three and four SD, either of the ordination methods that explain better can be used (Leps and Smilauer, 2003). Besides, the constrained or unconstrained nature of the ordination determines the ordination technique to be used (Ter Braak and Smilauer, 2002).

Exploration of the response of tree species to edaphic factors and distance from the forest edge involved application of DCA taking into account all the species that occurred in the sampled plots in the five forest fragments. Preliminary DCA test revealed that the tree species abundances exhibited unimodal responses to the edaphic factors and distance from the forest edge with gradient length of five standard deviations (SD). Since the gradient length was greater than four SD, CCA was the most appropriate method for analysis. The effect of edaphic factors and distance from the forest edge on the distribution and association of six dominant and adaptable tree species, most abundant and less abundant tree species involved the application of CCA with automatic forward selection using 1000 permutations. The determination of the influence of six dominant and adaptable tree species on the distribution of rare/less-abundant tree species also used CCA. Significance testing at the 95% probability level made use of Monte-Carlo Permutation Test (MCPT).

Finally, the temporal-spatial change in forest cover was analysed by the real change, percentage change and by trajectory analysis of satellite imageries i.e. by studying the changes occurring from one forest class to another (undisturbed forest, disturbed forest and open ground). Tables and graphs were used to illustrate the synthesized data obtained from the analysis of satellite imageries. A summary of statistical methods used to analyse the data is provided in Table 2.

Table 2: Summary of methods used for data analysis

Hypothesis	Variables of interest	Analysis
The forest cover in Taita Hills has changed for the last 43 years between 1973 and 2016	Forest cover or area change/rate of forest loss	Change detection analysis using satellite images
Forest edge affects the forest structure, tree species diversity, abundance of dominant and adaptable tree species and carbon stock	Stand density, DBH, height, tree species diversity, abundance of dominant and adaptable tree species and carbon stock	One-way ANOVA, summary statistics and Canonical Correspondence Analysis
Fragment size has an effect on tree species richness, diversity and carbon stock	Tree species richness, diversity and carbon stock	One-way ANOVA
Forest edge soils have lower moisture content and nutrients levels than forest interior soils	Soil moisture, soil nutrients, species richness and diversity	One-way ANOVA, Spearman rank correlation and Canonical Correspondence Analysis

CHAPTER FOUR

RESULTS

4.1 Introduction

The results in this chapter address forest cover change in the last 43 years, from 1973 to 2016. Besides, information is presented on the variations of tree species richness, diversity, abundance of dominant and adaptable tree species and above-ground carbon stock in the five forest fragments in Taita Hills along the forest edge-interior gradient. The highlights of this chapter include species richness, diversity and above-ground carbon stock as affected by the size of the forest fragment. Additionally, changes in edaphic factors i.e. moisture content, nitrogen, phosphorus, potassium, pH, electrical conductivity and organic carbon that occur along the forest edge-interior gradient hence altering the ecosystem health are elucidated. Finally, the relationships between species distribution and edaphic factors as well as species association along the forest edge-interior gradient are elucidated.

4.2 Forest cover change

Based on the overall classification accuracy of 84.4%, the five forest fragments that form a significant part of Taita Hills landscape lost 23.2% between 1973 and 2016 representing a deforestation rate of 0.5% per annum (Table 3). The annual rate of deforestation was highest in Vuria (1.0%) and lowest in Fururu (0.07%). Vuria lost the largest proportion of its forest cover (43.0%) followed by Chawia (32.7%), Ngangao (16.8%), Mbololo (13.7%) and Fururu (3.2%).

Table 3: Forest cover change of five forest fragments in Taita Hills between 1973 and 2016 using Landsat imageries

Changes in forest cover between 1973 and 2016				
Forest	Area in 1973 (ha)	Area in 2016 (ha)	Loss (ha)	Loss (%)
Chawia	165.6	111.4	54.2	32.7
Fururu	56.9	55.1	1.8	3.2
Mbololo	316.4	273.0	43.5	13.7
Ngangao	252.0	209.6	42.4	16.8
Vuria	211.7	120.7	91.0	43.0
Total	1,002.6	769.8	232.9	23.2

4.2.1 Trends in forest cover (area) change between 1973 and 2016

The greatest loss of the forested area in all the five forest fragments occurred between 1973 and 1987 (Figure 4; Table 4). During this period, Chawia lost 33.0% of its forest cover, Fururu (8.7%), Mbololo (13.2%), Ngangao (10.5%) and Vuria (31.6%) (Figure 5; Figure 6). The period after 1987 witnessed reduced loss in forest cover in all the five forest fragments that ranged from 0.3% to 16.7%. Moreover, it is worth noting that the forest cover increased in some of the forest fragments between 1987 and 2016. The forest cover increased by 3.3% in Chawia between 2012 and 2016. Correspondingly, the forested cover in Fururu increased by 9.2% (2001 - 2012) and 0.8% (2012 - 2016). In the case of Mbololo, an increase of 0.2% in the forest cover occurred between 2001 and 2012 while in Vuria, the forest cover increased by 4.0% from 2012 to 2016. However, Ngangao experienced continuous decline in forest cover over the years studied.

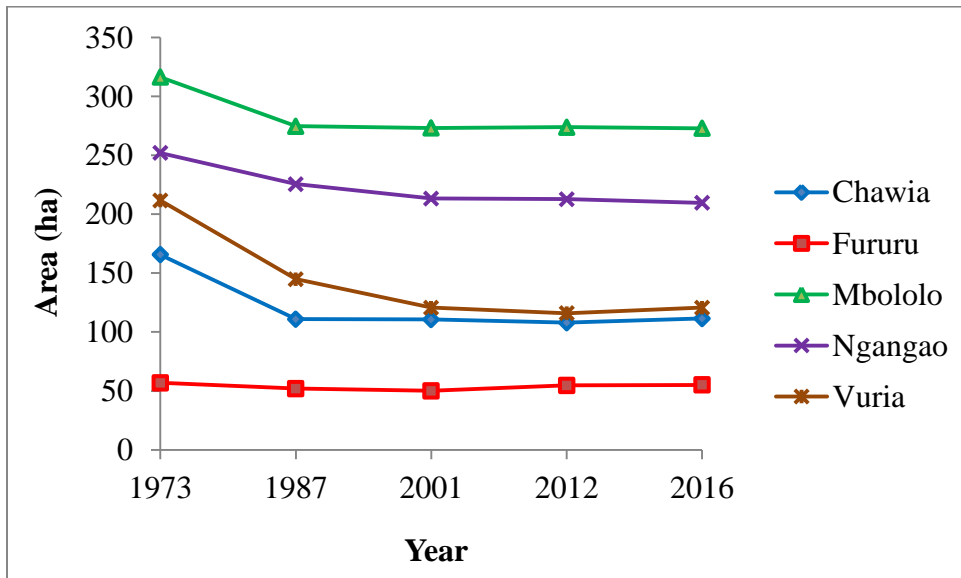


Figure 4: Trends in forest cover change of five forest fragments in Taita Hills between 1973 and 2016 using Landsat imageries

Table 4: Comparison of forest cover change of the five forest fragments in Taita Hills between 1973 and 2016 using Landsat imageries; (+) = Positive change, (-) = Negative change and (0) = No change

Forest fragment	Year	Area (ha)	Change (ha)	Change (%)	Type of change
Chawia	1973	165.60	0	0	0
	1987	110.97	54.63	33.0	-
	2001	110.61	0.36	0.3	-
	2012	107.88	2.73	2.5	-
	2016	111.42	3.54	3.3	+
Fururu	1973	56.88	0	0	0
	1987	51.93	4.95	8.7	-
	2001	50.04	1.89	3.6	-
	2012	54.66	4.62	9.2	+
	2016	55.08	0.42	0.8	+
Mbololo	1973	316.44	0	0	0
	1987	274.68	41.76	13.2	-
	2001	273.24	1.44	0.5	-
	2012	273.87	0.63	0.2	+
	2016	272.97	0.9	0.3	-
Ngangao	1973	252.0	0	0	0
	1987	225.54	26.46	10.5	-
	2001	213.39	12.15	5.4	-
	2012	212.76	0.63	0.3	-
	2016	209.61	3.15	1.5	-
Vuria	1973	211.68	0	0	0
	1987	144.90	66.78	31.6	-
	2001	120.69	24.21	16.7	-
	2012	116.01	4.68	3.9	-
	2016	120.69	4.68	4.0	+

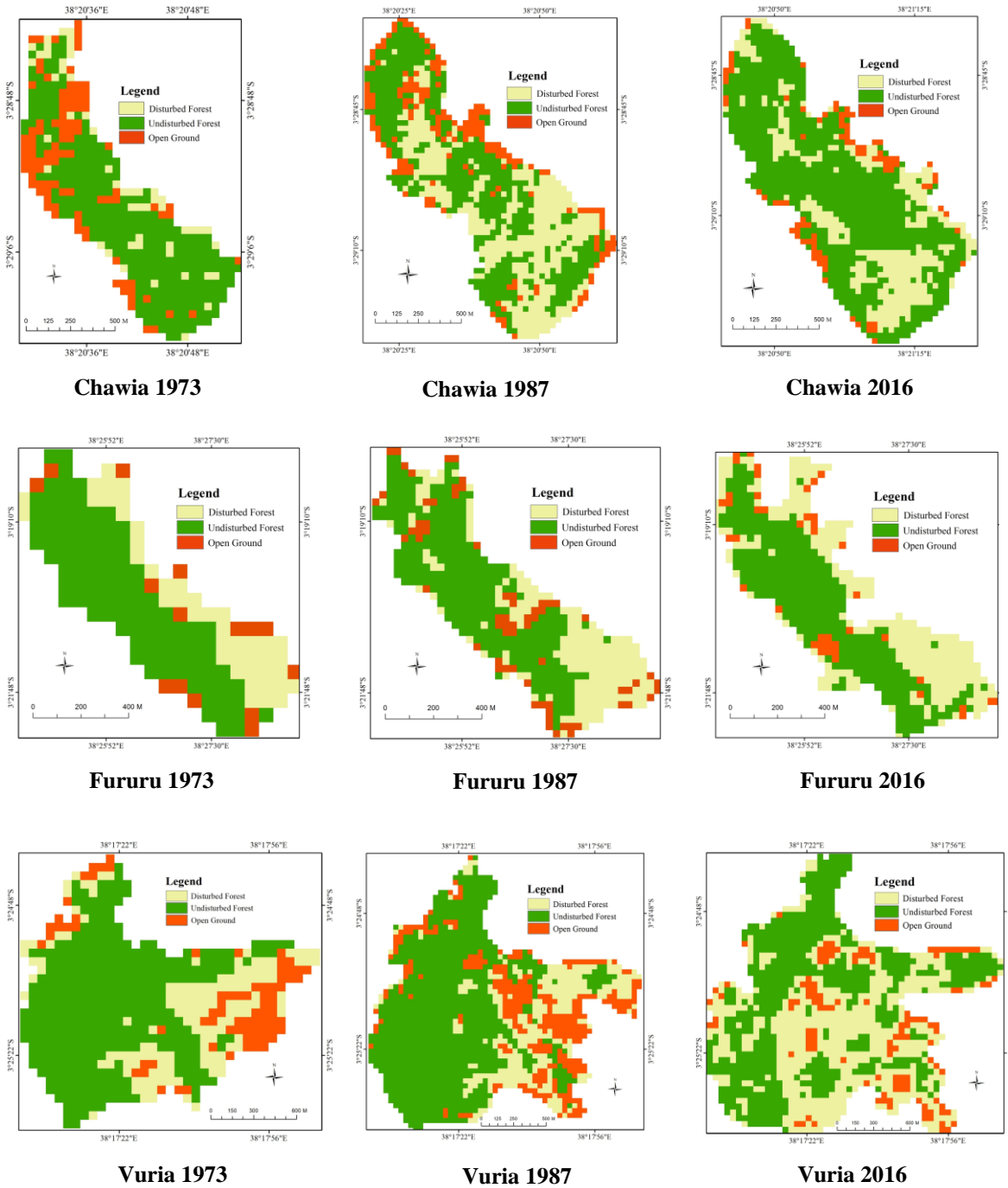
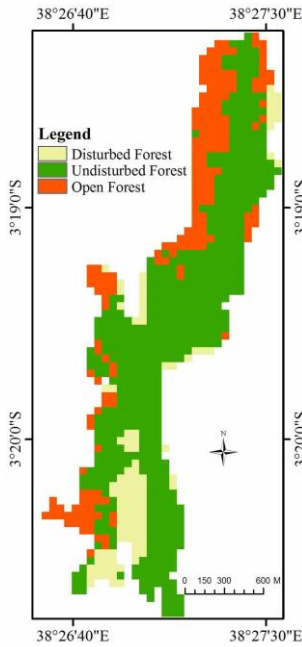
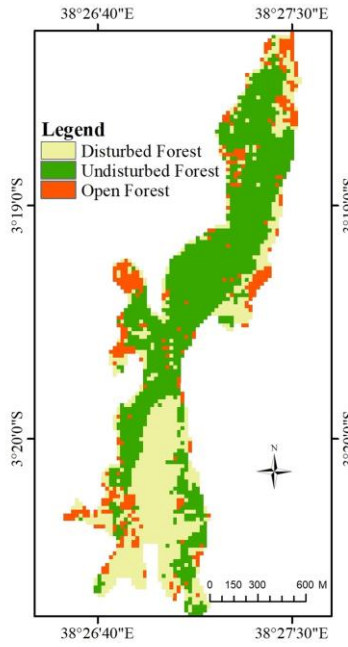


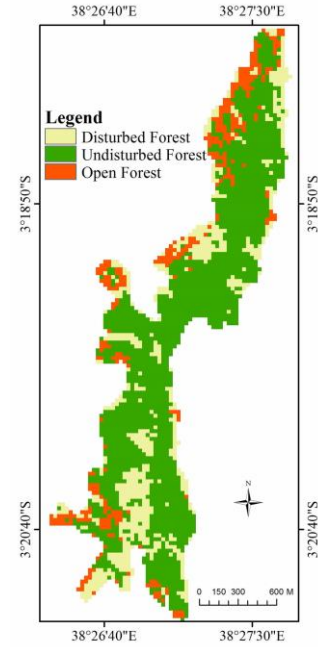
Figure 5: Forest cover change between 1973 and 2016 in Chawia, Fururu and Vuria forest fragments in Taita Hills using Landsat imageries



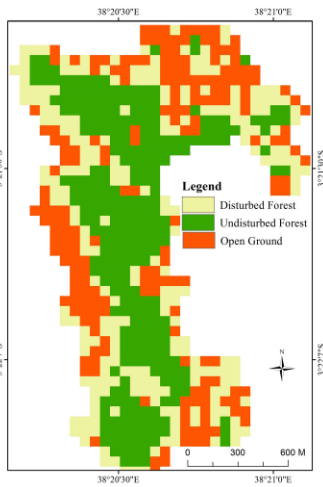
Mbololo 1973



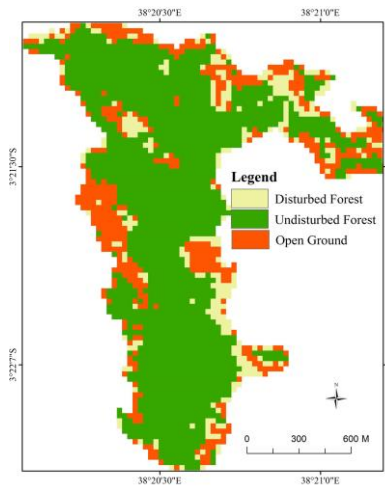
Mbololo 1987



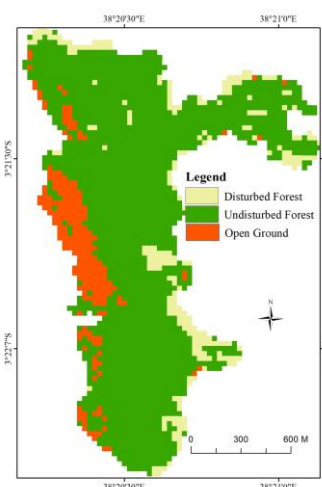
Mbololo 2016



Ngangao 1973



Ngangao 1987



Ngangao 2016

Figure 6: Forest cover change between 1973 and 2016 in Mbololo and Ngangao forest fragments in Taita Hills using Landsat imageries

4.2.2 Trends in forest cover change for undisturbed, disturbed and open ground areas

The trend in forest cover change spanning the period between 1973 and 2016 was analysed for undisturbed, disturbed and open ground for the five forest fragments using Landsat imageries. The undisturbed forest area in Chawia reduced from 111.24 ha in 1973 to 45.99 ha in 1987 before increasing to 78.39 ha in 2001. It decreased further to 70.41 ha in 2012 before slightly increasing to 73.89 ha in 2016. Conversely, the disturbed forest area in Chawia increased from 24.12 ha in 1973 to 39.06 ha in 1987 before considerably reducing to 19.98 ha in 2001. However, from 2001 to 2012 the disturbed forest area increased from 19.98 ha to 30.75 ha before marginally reducing to 29.61 ha in 2016. The open ground area declined significantly from 30.24 ha in 1973 to 6.72 ha in 2012 followed by a slight increase to 7.92 ha in 2016 (Figure 7).

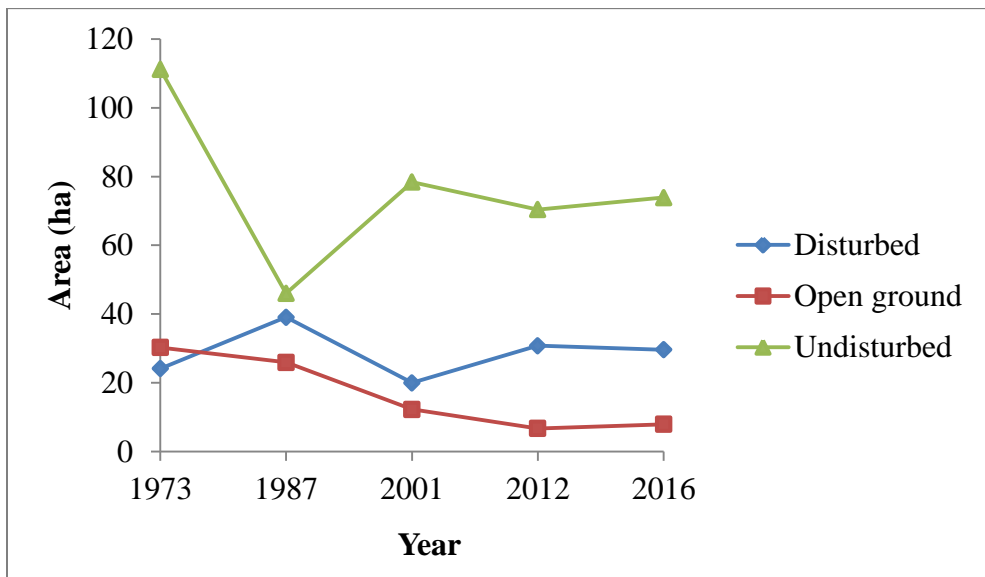


Figure 7: Forest cover change for disturbed, undisturbed and open ground areas for Chawia forest fragment using Landsat imageries

In Fururu, undisturbed forest area decreased from 35.64 ha in 1973 to 28.35 ha in 1987. The undisturbed forest area further decreased to 24.93 ha in 2001 before increasing to 33.45 ha in 2012 and decreasing again to 28.62 ha in 2016. The disturbed forest area went up marginally from 15.84 ha in 1973 to 17.37 ha in 1987 but reduced to 15.93 ha in 2001. The disturbed forest area in Fururu was 16.27 ha in 2012 but increased to 22.59 ha in 2016. The open ground area increased from 5.4 ha in 1973 to 9.18 ha in 2001 before decreasing to 3.87 ha in 2016 (Figure 8).

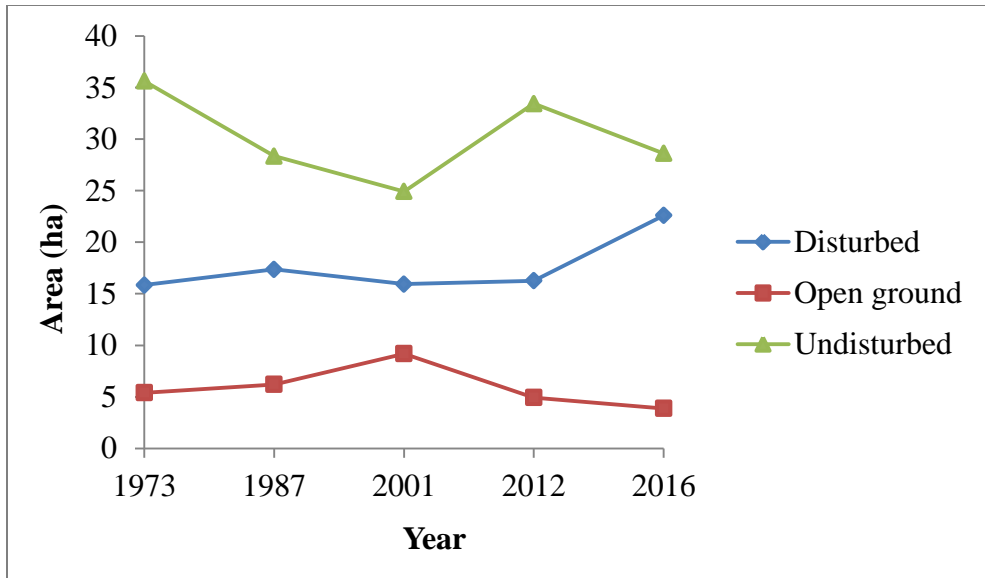


Figure 8: Forest cover change for disturbed, undisturbed and open ground areas for Fururu forest fragment using Landsat imageries

The undisturbed forest area in Mbololo reduced from 208.44 ha in 1973 to 143.28 ha in 1987. Between 2001 and 2012, the undisturbed forest area increased from 143.19 ha to 172.98 ha. The undisturbed forest area in Mbololo in 2016 was 178.83 ha, a slight increment of 5.85 ha compared to 2012. From 1973 to 1987, disturbed forest area increased from 38.16 ha to 97.56 ha. The disturbed forest area then decreased to 72.09 ha in 2001 and 56.43 ha in 2012 before slightly increasing to 61.83 ha in 2016. The open ground area decreased from 69.84 ha in 1973 to 33.84 ha in 1987 before increasing to 57.96 ha in 2001. Between 2012 and 2016, the open ground area in Mbololo decreased from 44.46 ha to 32.31 ha (Figure 9).

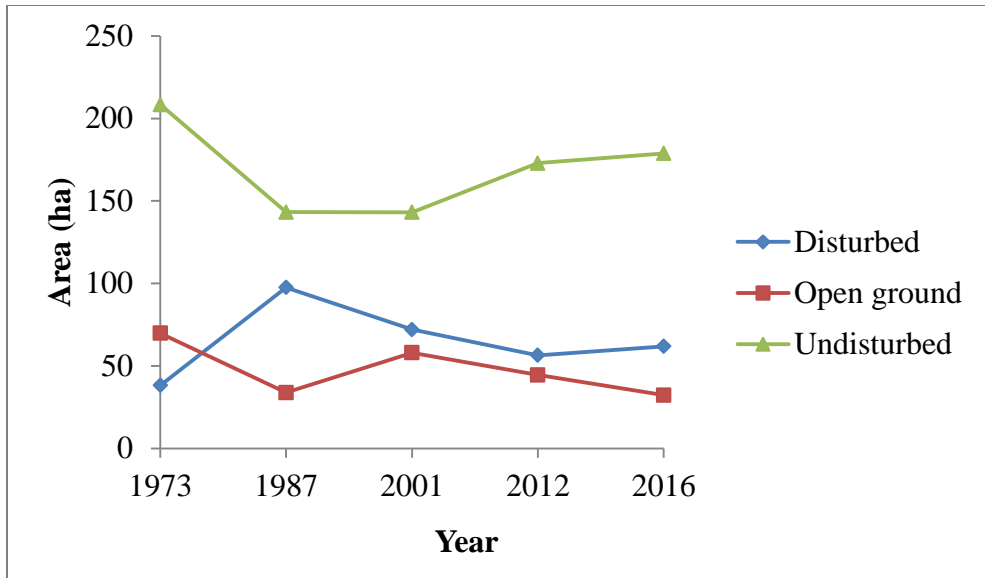


Figure 9: Forest cover change for disturbed, undisturbed and open ground areas for Mbololo forest fragment using Landsat imageries

Ngangao exhibited a different trend in forest cover change for undisturbed forest area compared to Chawia, Fururu and Mbololo. From 1973 to 1987, the undisturbed forest area increased from 142.92 ha to 154.53 ha before plummeting to 151.92 ha in 2001, 147.24 ha in 2012 and lastly 144.36 ha in 2016. The disturbed forest area decreased from 37.44 ha in 1973 to 26.73 ha in 1987 before increasing to 29.09 ha in 2001. Between 2001 and 2012, the disturbed forest area increased from 29.09 ha to 48.6 ha before reducing to 39.51 ha in 2016. A decreasing linear trend in the forest area under open ground was evident from 1973 to 2012. Accordingly, the open ground area declined from 71.64 ha in 1973 to 16.92 ha in 2012 before increasing to 25.74 ha in 2016 (Figure 10).

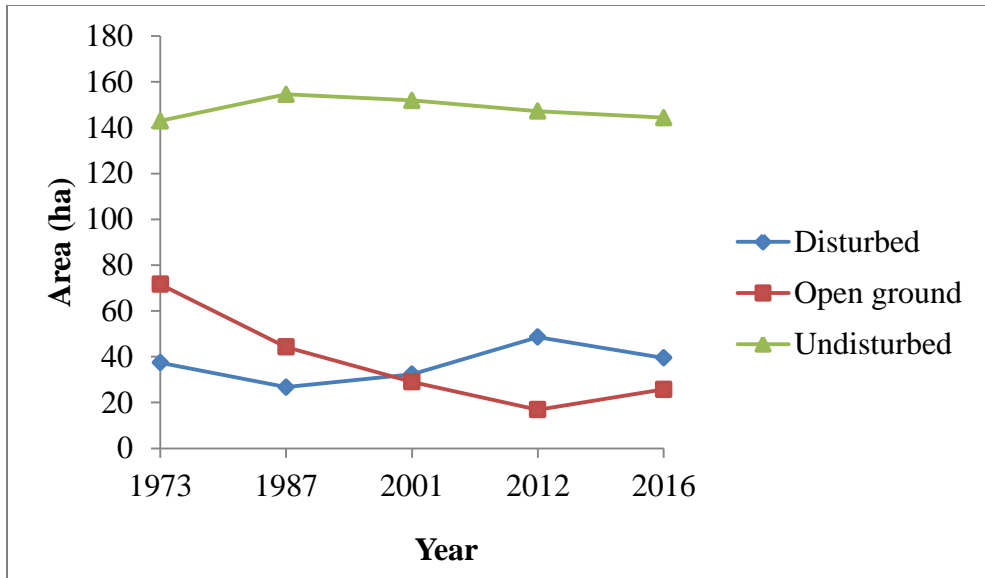


Figure 10: Forest cover change for disturbed, undisturbed and open ground areas for Ngangao forest fragment using Landsat imageries

The undisturbed forest area in Vuria decreased from 133.2 ha in 1973 to 87.75 ha in 1987. Undisturbed forest area further decreased to 80.28 ha and 78.12 ha in 2001 and 2012 respectively. By 2016, the undisturbed forest area in Vuria had dropped considerably to 56.34 ha. The trend for the disturbed forest area was almost similar to Ngangao. The disturbed forest area substantially decreased from 50.76 ha in 1973 to 28.44 ha in 1987 before slightly increasing to 33.57 ha in 2001. Between 2001 and 2012, the disturbed forest area reduced from 33.57 ha to 26.64 ha before increasing to 52.83 ha in 2016. In the case of open ground, the area increased marginally from 27.72 ha in 1973 to 28.44 ha in 1987 and then significantly plummeted to 6.84 ha in 2001. The area under open ground increased from 6.84 ha in 2001 to 11.25 ha in 2012 followed by negligible increase of 0.27 ha to 11.52 ha in 2016 (Figure 11).

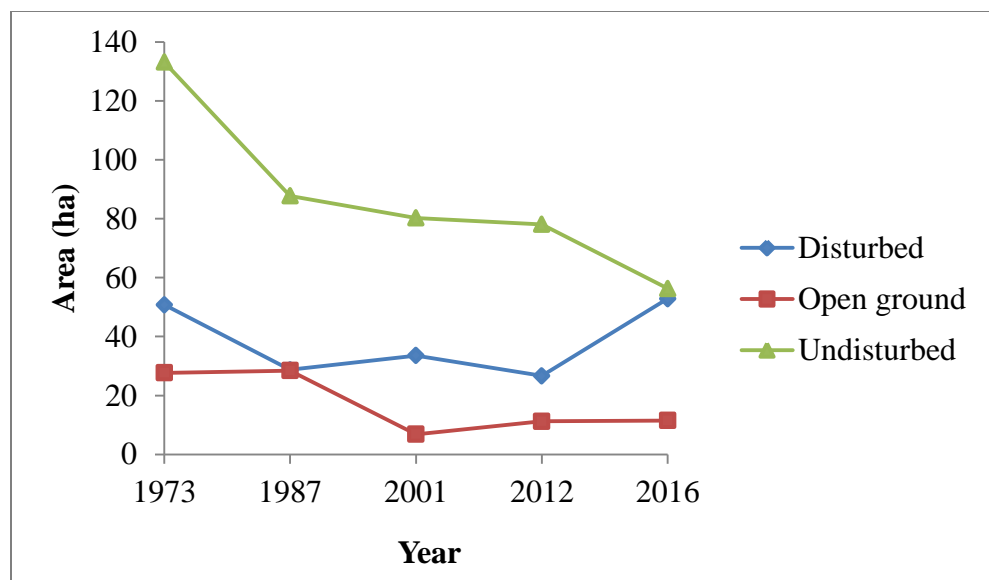


Figure 11: Forest cover change for disturbed, undisturbed and open ground areas for Vuria forest fragment using Landsat imageries

4.3 Effect of forest edge on tree species diversity

Overall species diversity varied significantly among the forest fragments ($F_{(1, 4)} = 13.26$; $p=0.001$). Ngangao had the highest species diversity of 2.86 ± 0.09 , followed by Mbololo (2.84 ± 0.08), Fururu (2.07 ± 0.19), Chawia (1.79 ± 0.29) and Vuria (1.68 ± 0.22). Considering the individual forest fragments along the forest edge-interior gradient, the Shannon-Wiener diversity index in Chawia for the forest interior was higher (2.25 ± 0.32) than forest edge (1.60 ± 0.65) and intermediate forest (1.52 ± 0.51) although the means were not significantly different ($F_{(1, 2)} = 0.61$; $p=0.564$) (Table 5). In Fururu, the Shannon-Wiener diversity index did not vary significantly along the forest edge-interior gradient ($F_{(1, 2)} = 0.4$; $p=0.689$) even though the forest interior had slightly higher diversity (2.30 ± 0.47) than the forest edge (2.12 ± 0.42) and the intermediate forest (1.84 ± 0.05). Similarly, there was no significant variation in Shannon-Wiener diversity index in Mbololo from the edge to the interior of the forest ($F_{(1, 2)} = 1.04$; $p=0.369$). However, the interior forest habitat of Mbololo had slightly higher species diversity (2.99 ± 0.11) than the forest edge (2.79 ± 0.11) and intermediate forest (2.74 ± 0.16). Ngangao had Shannon-Wiener diversity indices which were comparable with Mbololo although they were not significantly different along the forest edge-interior gradient ($F_{(1, 2)} = 0.58$; $p=0.576$). Shannon-

Wiener diversity index in Vuria did not exhibit significant variations along the gradient ($F_{(1,2)} = 1.31$; $p=0.338$) although it was lower at the forest edge (1.20 ± 0.32) than 1.99 ± 0.29 in the intermediate forest and 1.87 ± 0.48 in the forest interior.

Table 5: Comparison of mean Shannon-Wiener diversity index (H') along forest edge-interior gradient in the five forest fragments of Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts along the rows are not significantly different at $p\leq 0.05$ level

Forest fragment	Shannon-Wiener diversity index (H')				LSD
	Forest edge	Intermediate forest	Forest interior	p value	
Chawia	1.60 ± 0.65^a	1.52 ± 0.51^a	2.25 ± 0.32^a	0.564	1.646
Fururu	2.12 ± 0.42^a	1.84 ± 0.05^a	2.30 ± 0.47^a	0.689	1.259
Mbololo	2.79 ± 0.11^a	2.74 ± 0.16^a	2.99 ± 0.11^a	0.369	0.385
Ngangao	2.72 ± 0.17^a	2.98 ± 0.10^a	2.88 ± 0.22^a	0.564	0.520
Vuria	1.20 ± 0.32^a	1.99 ± 0.29^a	1.87 ± 0.48^a	0.338	1.297

4.4 Effect of forest edge on the abundance of dominant and adaptable tree species

The six common and dominant species studied in all the five forest fragments to determine the changes in their occurrence along the forest edge-interior gradient as influenced by soil physical and chemical properties as well as the distance from the forest edge (hereafter-referred to as environmental variables) exemplified salient patterns. These species include *Macaranga conglomerata*, *Albizia gummifera*, *Syzygium guineense*, *Xymalos monospora*, *Tabernaemontana stapfiana* and *Maesa lanceolata*. Canonical Correspondence Analysis (CCA) model for the relationship between the six dominant species and the environmental variables (MCPT; 1000 Permutations) was significant for the distance from the forest edge ($F_{(1000)} = 2.76$; $p=0.012$), electrical conductivity ($F_{(1000)} = 2.53$; $p=0.028$), potassium ($F_{(1000)} = 2.39$; $p=0.031$) and soil moisture ($F_{(1000)} = 2.46$; $p=0.037$). However, soil pH ($F_{(1000)} = 2.15$; $p=0.068$), nitrogen ($F_{(1000)} = 0.41$; $p=0.838$), phosphorus ($F_{(1000)} = 1.50$; $p=0.187$) and organic carbon ($F_{(1000)} = 0.22$; $p=0.947$) were not significant in the model.

Albizia gummifera had large abundance (higher probability of occurrence) within the forests' areas whose soils were saline and had high organic carbon content (Figure 12). However, there was a negative association for *A. gummifera* with soil moisture. Phosphorus and nitrogen had a large positive correlation with *M. conglomerata*. *X. monospora* was more abundant in areas within the forests that had high concentration of potassium in soil. Moreover, soil moisture had a positive relationship with *X. monospora*. *T. stapfiana* had large positive correlation with soil pH and the distance from the forest edge. *Syzygium guineense* had a positive relationship with organic carbon and electrical conductivity. However, the abundance of *M. lanceolata* exhibited no correlation with the environmental variables.

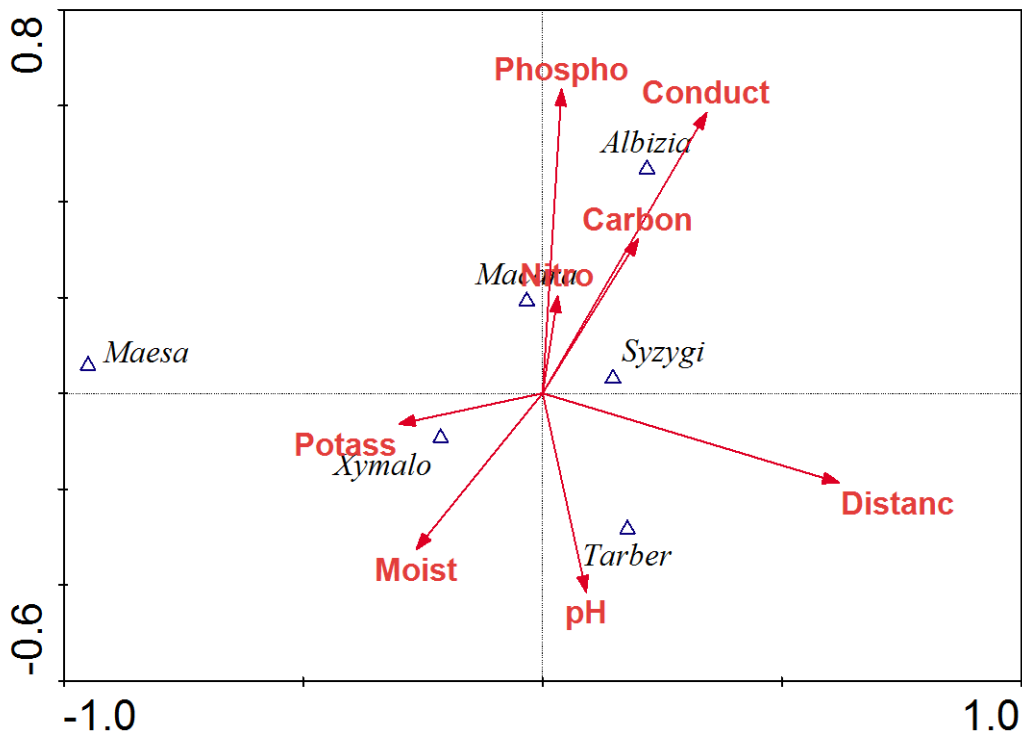


Figure 12: Canonical Correspondence Analysis (CCA) triplot of six adaptable and dominant tree species and environmental variables in five forest fragments, Taita Hills. Potass = Potassium, Moist = Soil moisture, Conduct = Electrical conductivity, Distanc = Distance from forest edge, Phospho = Phosphorus, Nitro = Nitrogen

4.5 Interaction of the most abundant species with environmental variables

The CCA model was significant for electrical conductivity ($F_{(1000)} = 3.48$; $p=0.001$), the distance from forest edge ($F_{(1000)} = 2.89$; $p=0.001$) and soil pH ($F_{(1000)} = 1.77$; $p=0.020$). However, the CCA model was not significant for potassium ($F_{(1000)} = 1.60$; $p=0.064$), phosphorus ($F_{(1000)} = 0.98$; $p=0.461$), moisture ($F_{(1000)} = 1.48$; $p=0.087$), organic carbon ($F_{(1000)} = 1.47$; $p=0.081$) and nitrogen ($F_{(1000)} = 1.56$; $p=0.052$). The distance from the forest edge had a highly positive correlation with species such as *Tabernaemontana stapfiana*, *Psychotria petiti*, *Chassalia discolor*, *Newtonia buchananii*, *Oxyanthus speciosus*, *Pauridiantha paucinervis* and *Strombosia scheffleri* among others (Figure 13). On the other hand, there was a strong positive correlation between potassium and *Xymalos monospora*, *Lepidotrichilia volkensis*, *Cussonia spicata* and *Neoboutonia macrocalyx*. *Dasylepis integra*, *Brucea antidysenterica*, *Garcinia volkensis*, *Phoenix reclinata* and *Pleiocarpa pycnantha* had a high positive correlation with soil pH. Phosphorus had a large positive correlation with *Turraea holstii* and *Millettia oblata*. Electrical conductivity and nitrogen had a strong positive relationship with *Albizia gummifera* and *Cola greenwayi* whereas *Crabia zimmermannii* exhibited slightly positive correlation with organic carbon. There was no relationship between *Rytigynia uhligii* and *Maytenus senegalensis* with any of the environmental variables.

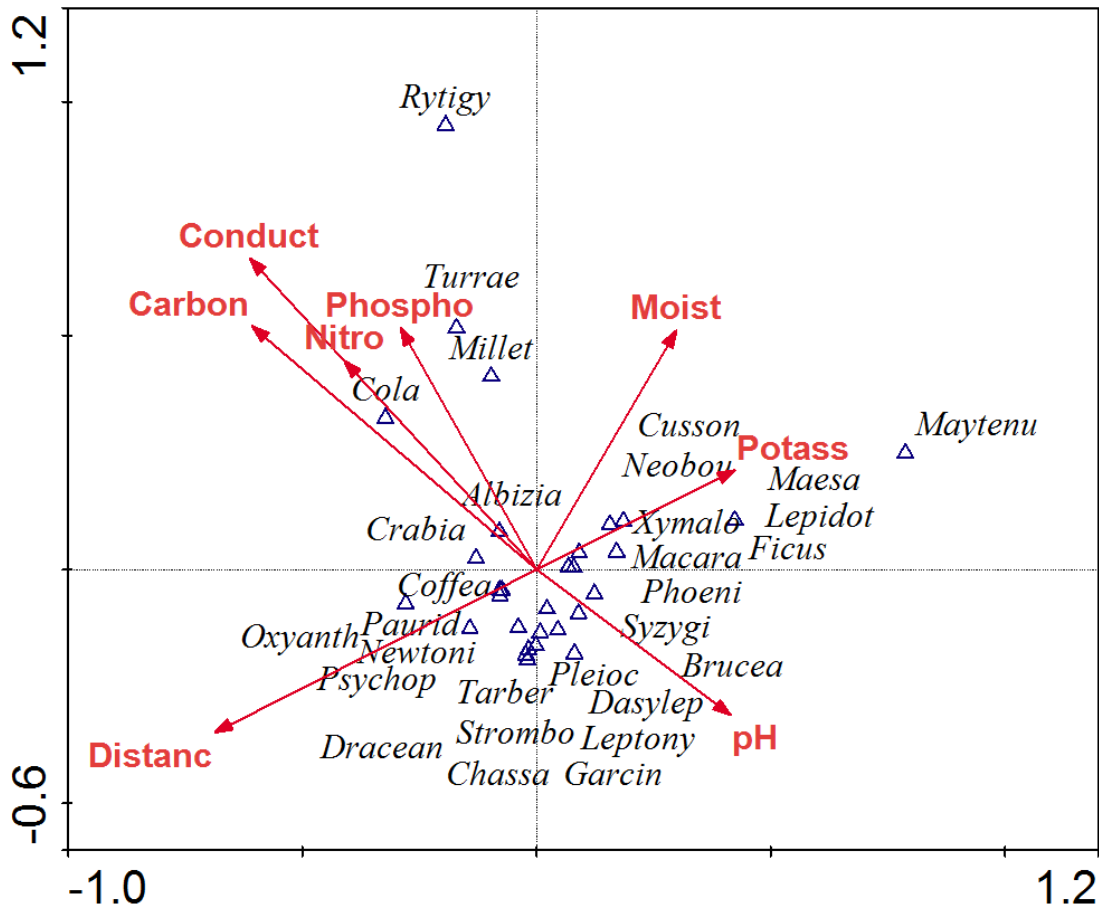


Figure 13: Canonical Correspondence Analysis (CCA) of most abundant tree species and environmental variables for the five forest fragments in Taita Hills. Potass = Potassium, Moist = Soil moisture, Conduct = Electrical conductivity, Distanc = Distance from forest edge, Phospho = Phosphorus, Nitro = Nitrogen

4.6 Interaction of less abundant species with environmental variables

The CCA model was significant for soil moisture ($F_{(1000)} = 3.30$; $p=0.044$), nitrogen ($F_{(1000)} = 2.02$; $p=0.017$) and the distance from the forest edge ($F_{(1000)} = 1.96$; $p=0.021$). However, the model was not significant for organic carbon ($F_{(1000)} = 1.37$; $p=0.202$), electrical conductivity ($F_{(1000)} = 0.79$; $p=0.658$), soil pH ($F_{(1000)} = 0.70$; $p=0.754$), potassium ($F_{(1000)} = 0.66$; $p=0.752$) and phosphorus ($F_{(1000)} = 0.43$; $p=0.934$). Less abundant species across the five forest fragments such as *Prunus africana* and *Allophylus abyssinicus* showed a highly positive correlation with nitrogen and phosphorus (Figure 14). On the other hand, *Podocarpus latifolius*, *Ochna holstii*,

Aphloia theiformis and *Polyscias fulva* had positive correlation with organic carbon, electrical conductivity and the distance from the forest edge. *Lobelia giberroa* had a high positive correlation with soil moisture and potassium. *Vernonia auriculifera* and *Agelaea pentagyna* had a strong positive correlation with soil pH. Similarly, *Aningeria robusta* and *Sorindeia madagascariensis* were correlated with soil pH.

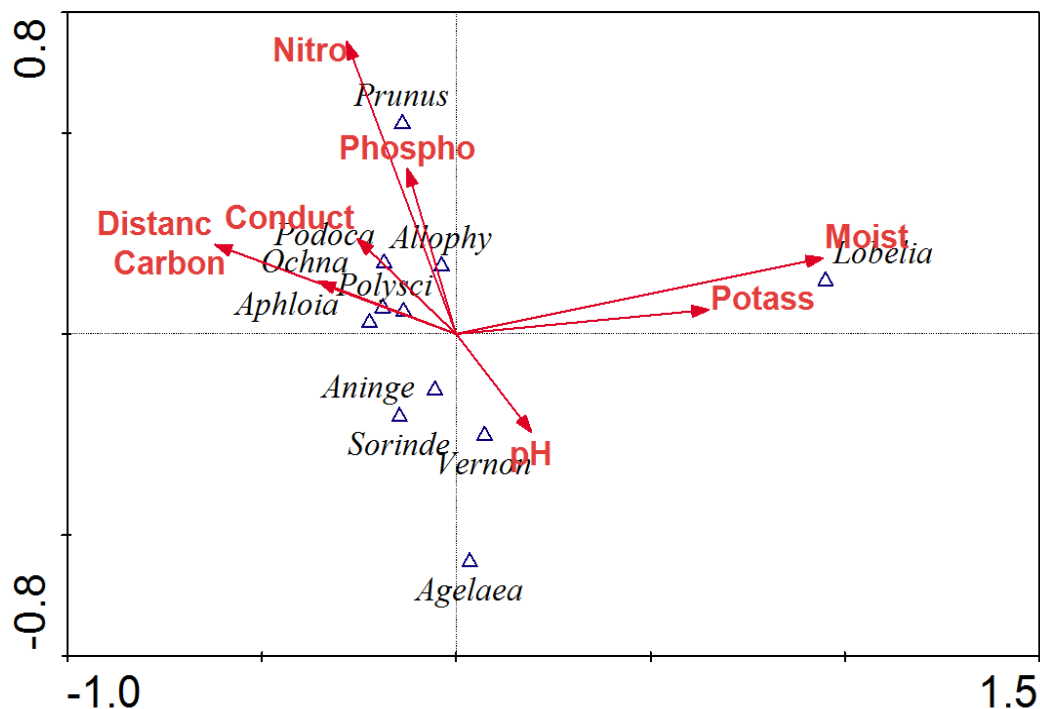


Figure 14: Canonical Correspondence Analysis (CCA) of less abundant tree species and environmental variables in the five forest fragments of Taita Hills. Potass = Potassium, Moist = Soil moisture, Conduct = Electrical conductivity, Distanc = Distance from forest edge, Phospho = Phosphorus, Nitro = Nitrogen

4.7 Interaction between dominant and less abundant tree species

The CCA model was significant for *M. lanceolata* ($F_{(1000)} = 2.19$; $p=0.015$) but not for *A. gummifera* ($F_{(1000)} = 1.17$; $p=0.309$), *M. conglomerata* ($F_{(1000)} = 0.83$; $p=0.560$), *S. guineense* ($F_{(1000)} = 1.10$; $p=0.316$), *T. stapfiana* ($F_{(1000)} = 0.82$; $p=0.592$) and *X. monospora* ($F_{(1000)} = 1.33$; $p=0.216$). There was a strong positive correlation between *S. guineense* and *X. monospora*

(Figure 15). Similarly, *M. conglomerata* had a large positive correlation with *A. gummifera*. However, *T. stapfiana* had a large negative correlation with *M. lanceolata*. Considering the relationship between key dominant and adaptable species and less abundant species, *S. guineense* and *X. monospora* had high positive correlation with *Ochna holstii* while *A. gummifera* and *M. conglomerata* had a large positive correlation with *Aphloia theiformis* and *Polyscias fulva*. *Tabernaemontana stapfiana* had a large positive correlation with *Allophylus abyssinicus*, *Aningeria robusta*, *Podocarpus latifolius* and *Prunus africana* whereas *M. lanceolata* showed a highly positive correlation with *Agelaea pentagyna*.

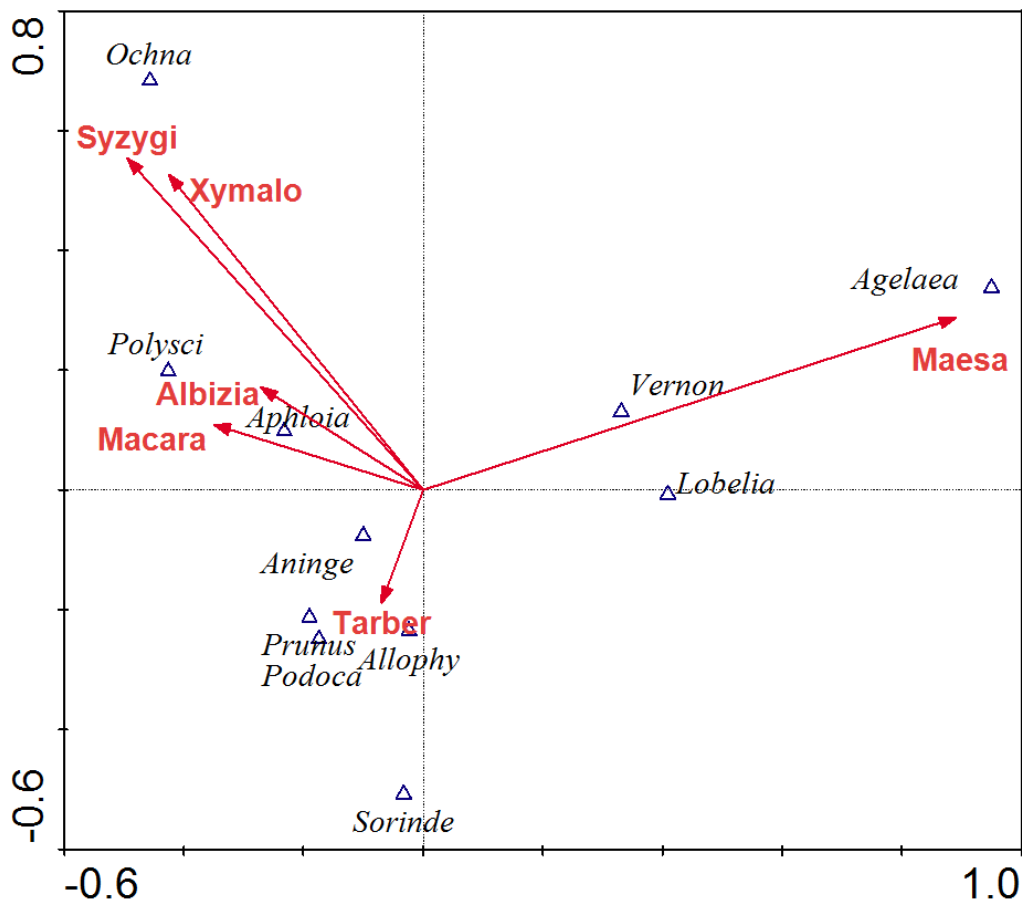


Figure 15: Canonical Correspondence Analysis (CCA) of six key dominant and adaptable tree species and less abundant tree species in the five forest fragments of Taita Hills. Albizia = *A. gummifera*, Macara = *M. conglomerata*, Maesa = *M. lanceolata*, Syzygi = *S. guineense*, Tarber = *T. stapfiana* and Xymalo = *X. monospora*

4.8 Forest structure, edge effect and carbon sequestration potential of the forest fragments

The structure of natural forests serves as a benchmark for understanding carbon storage potential for reduced carbon emissions. The stand density (stems ha⁻¹), diameter at breast height (DBH), height and wood specific gravity are the key attributes of forest structure that influences the ability of the forest to provide ecosystem services including the quantity of carbon sequestered. Stand density, DBH, height and wood specific gravity determinations for forest fragments along the forest edge-interior gradient provided the basis for isolating variables that were influenced by edge effect. The results highlighting how the forest edge affects the tree sizes (DBH), height, wood specific gravity, stand density and hence carbon stock are presented in sub-sections 4.8.1, 4.8.2 and 4.8.3.

4.8.1 Effect of forest edge on diameter at breast height, height and wood specific gravity

Generally, there was significant variation in mean DBH among the forest fragments ($F_{(1, 4)} = 4.73$; $p=0.001$). Chawia had the highest mean DBH of 22.7 ± 3.0 cm followed by Vuria (21.9 ± 4.0 cm), Fururu (20.0 ± 3.3 cm) and Ngangao (16.4 ± 1.4 cm). Mbololo had the lowest mean DBH of 13.7 ± 0.9 cm. However, the DBH did not significantly vary in the five forest fragments along the gradient from the forest edge to the interior (Table 6). In Chawia, no significant difference in DBH was observed along the gradient from forest edge to the interior ($F_{(1, 2)} = 0.1$; $p=0.909$). The DBH in Chawia was 23.7 ± 4.9 cm on the forest edge but increased to 24.0 ± 6.8 cm in the intermediate habitat of the forest before declining to 21.1 ± 4.8 cm in the forest interior. A similar trend occurred in Fururu where the mean DBH on the forest edge, intermediate forest habitat and forest interior were 14.6 ± 5.4 , 25.8 ± 8.8 and 22.0 ± 4.3 cm respectively. Nevertheless, the DBH along the gradient from the forest edge to the interior in Fururu was not significantly different ($F_{(1, 2)} = 0.92$; $p=0.409$). In Mbololo, the DBH did not vary significantly along the gradient ($F_{(1, 2)} = 0.12$; $p=0.884$). Thus, the DBH in Mbololo was 13.6 ± 1.4 cm on the forest edge, 14.4 ± 2.1 cm in the intermediate forest and 13.3 ± 1.4 cm in the forest interior. The DBH in Ngangao reduced from 16.6 ± 2.6 cm on the forest edge to 15.7 ± 2.2 cm in the intermediate forest habitat before increasing to 17.0 ± 2.5 cm in the forest interior although the variation was not significant along the forest edge-interior gradient ($F_{(1, 2)} = 0.08$; $p=0.921$). Similarly in Vuria, there was no significant difference in DBH along the gradient from the forest edge to the interior

($F_{(1, 2)} = 0.19$; $p=0.827$). Thus, the DBH in Vuria was 25.2 ± 9.6 cm on the edge of the forest, 22.8 ± 7.0 cm in the intermediate forest and 19.1 ± 5.6 cm in the interior of the forest.

Comparing the height of trees among the five forest fragments revealed that trees were significantly taller in Chawia (12.6 ± 1.4 m) than Ngangao (11.6 ± 0.7 m), Fururu (11.4 ± 1.6 m), Mbololo (10.1 ± 0.4 m) and Vuria (7.1 ± 0.7 m) ($F_{(1, 4)} = 3.20$; $p=0.013$). Forest edge did not significantly affect the height of trees along the forest edge-interior gradient in the five fragments (Table 6). In fact, there was no significant variation in height along the gradient from forest edge to the interior in Chawia ($F_{(1, 2)} = 0.21$; $p=0.808$). Trees occurring in the forest interior in Chawia were the tallest with a mean height of 13.5 ± 2.8 m, followed by those in the intermediate forest habitat (13.1 ± 3.0 m) and forest edge (11.4 ± 1.7 m). Similarly, height was not significantly different along the forest edge-interior gradient in Fururu ($F_{(1, 2)} = 2.17$; $p=0.131$). The tallest trees in Fururu occurred in the interior of the forest (15.3 ± 3.6 m) followed by intermediate forest (10.7 ± 1.3 m) with the edge of the forest having relatively shorter trees (7.9 ± 1.7 m). In Mbololo, the height did not vary significantly along the gradient ($F_{(1, 2)} = 0.95$; $p=0.387$) with the tallest trees occurring in the intermediate habitat (10.9 ± 0.9 m). Trees on the forest edge in Mbololo were the shortest with mean height of 9.3 ± 0.7 m. Similarly, the height of trees in Ngangao was not significantly different along the gradient from forest edge to the interior ($F_{(1, 2)} = 0.54$; $p=0.587$) even though the trees occurring in the interior of the forest were marginally taller (12.6 ± 1.5 m) compared to the forest edge (11.3 ± 1.0 m) and intermediate forest (10.9 ± 1.2 m). In Vuria, the intermediate forest had the tallest trees (8.1 ± 1.7 m) followed by the forest edge (6.8 ± 1.6 m) and lastly the forest interior (6.5 ± 0.9 m). However, the variation in height in Vuria was not significantly different ($F_{(1, 2)} = 0.40$; $p=0.678$).

Mbololo had significantly higher wood specific gravity (0.540 ± 0.007 gcm⁻³) among the five forest fragments while Fururu had the lowest (0.481 ± 0.007 gcm⁻³) ($F_{(1, 4)} = 5.31$; $p=0.001$). However, wood density in Chawia was not significantly different along the forest edge-interior gradient ($F_{(1, 2)} = 0.76$; $p=0.473$) even though it was higher on the edge of the forest (0.537 ± 0.025 gcm⁻³) followed by the forest interior (0.502 ± 0.017 gcm⁻³) and the intermediate forest (0.499 ± 0.034 gcm⁻³) (Table 6). Similarly, there was no significant variation in wood specific gravity along the gradient in Fururu ($F_{(1, 2)} = 0.58$; $p=0.567$). In fact, the wood specific gravity in Fururu on the forest edge (0.487 ± 0.012 gcm⁻³), intermediate forest (0.467 ± 0.016 gcm⁻³) and forest interior (0.483 ± 0.011 gcm⁻³) were comparable. However, in Mbololo, there was a

significant variation in wood specific gravity along the forest edge-interior gradient ($F_{(1, 2)} = 8.87$; $p=0.001$) and was higher in the forest interior ($0.564\pm 0.012 \text{ gcm}^{-3}$) followed by intermediate forest ($0.553\pm 0.013 \text{ gcm}^{-3}$) while the forest edge had the lowest wood specific gravity ($0.503\pm 0.007 \text{ gcm}^{-3}$). In the case of Ngangao, wood specific gravity did not vary significantly along the gradient ($F_{(1, 2)} = 1.39$; $p=0.253$). The specific wood gravity was $0.507\pm 0.011 \text{ gcm}^{-3}$ on the edge of the forest, $0.502\pm 0.011 \text{ gcm}^{-3}$ in the intermediate forest and $0.529\pm 0.015 \text{ gcm}^{-3}$ in the interior of the forest. In Vuria, wood specific gravity was not significantly different along the forest edge-interior gradient ($F_{(1, 2)} = 0.87$; $p=0.434$). The wood specific gravity was $0.507\pm 0.011 \text{ gcm}^{-3}$ on the edge of the forest, $0.502\pm 0.011 \text{ gcm}^{-3}$ in the intermediate forest and $0.497\pm 0.009 \text{ gcm}^{-3}$ in the forest interior (Table 6).

Table 6: Comparison of DBH, height and wood specific gravity along the forest edge-interior gradient. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same column are not significantly different at $p \leq 0.05$ level

Forest fragment	Forest edge-interior gradient	DBH (cm)	Height (m)	Wood specific gravity (gcm^{-3})
Chawia	Forest edge	23.7±4.9 ^a	11.4±1.7 ^a	0.537±0.025 ^a
	Intermediate forest	24.0±6.8 ^a	13.1±3.0 ^a	0.499±0.034 ^a
	Forest interior	21.1±4.8 ^a	13.5±2.8 ^a	0.502±0.017 ^a
	<i>p</i> value	0.909	0.808	0.473
	LSD	15.5	7.4	0.070
Fururu	Forest edge	14.6±5.4 ^a	7.9±1.7 ^a	0.487±0.012 ^a
	Intermediate forest	25.8±8.8 ^a	10.7±1.3 ^a	0.467±0.016 ^a
	Forest interior	22.0±4.3 ^a	15.3±3.6 ^a	0.483±0.011 ^a
	<i>p</i> value	0.409	0.131	0.567
	LSD	17.0	8.0	0.038
Mbololo	Forest edge	13.6±1.4 ^a	9.3±0.7 ^a	0.503±0.007 ^a
	Intermediate forest	14.4±2.1 ^a	10.9±0.9 ^a	0.553±0.013 ^b
	Forest interior	13.3±1.4 ^a	10.1±0.7 ^a	0.564±0.012 ^b
	<i>p</i> value	0.884	0.387	0.001
	LSD	4.5	2.2	0.031
Ngangao	Forest edge	16.6±2.6 ^a	11.3±1.0 ^a	0.507±0.011 ^a
	Intermediate forest	15.7±2.2 ^a	10.9±1.2 ^a	0.502±0.011 ^a
	Forest interior	17.0±2.5 ^a	12.6±1.5 ^a	0.529±0.015 ^a
	<i>p</i> value	0.921	0.587	0.253
	LSD	6.8	3.5	0.035
Vuria	Forest edge	25.2±9.6 ^a	6.8±1.6 ^a	0.503±0.013 ^a
	Intermediate forest	22.8±7.0 ^a	8.1±1.7 ^a	0.482±0.012 ^a
	Forest interior	19.1±5.6 ^a	6.5±0.9 ^a	0.497±0.009 ^a
	<i>p</i> value	0.827	0.678	0.434
	LSD	21.2	3.9	0.033

4.8.2 Effect of forest edge on forest stand density

Besides DBH, height and wood specific gravity, stand density (stems ha⁻¹) is another important attribute of the forest structure that influences the ability of the forest to provide ecosystem services including carbon sequestration. Stand density evaluations for the five forest fragments along the forest edge-interior provided information for relating edge effect to stand density. Generally, the stand density was not significantly different among the five forest fragments and ranged from 202±24.5 to 250±43.9 ($F_{(1,4)} = 0.92$; $p=0.450$). In Chawia, there was a clear pattern of changes in stand density, which increased towards the forest interior (Figure 16). The stand density of 220±47.9 stems ha⁻¹ in the forest interior was slightly higher compared to 183±36.4 stems ha⁻¹ at the forest edge and 200±36.9 stems ha⁻¹ in the intermediate forest although the variations were not significant ($F_{(1,2)} = 0.13$; $p=0.875$).

Similarly, the stand density in Fururu increased along the gradient from the edge of the forest to the interior (Figure 16). However, the increase in the stand density along the forest edge-interior gradient was not significantly different ($F_{(1,2)} = 0.11$; $p=0.898$). The stand density was 231±69.2, 238±70.6 and 277±84.9 stems ha⁻¹ for forest edge, intermediate forest and the forest interior respectively.

Similar to Chawia and Fururu, the stand density in Mbololo increased from 193±14.4 stems ha⁻¹ at the edge of the forest to 205±20.5 and 224±18.1 stems ha⁻¹ in the intermediate forest and forest interior respectively (Figure 16). However, this increase in stand density along the forest edge-interior gradient did not vary significantly ($F_{(1,2)} = 0.99$; $p=0.373$). The stand density along the forest edge-interior gradient in Ngangao did not differ significantly ($F_{(1,2)} = 0.23$; $p=0.793$). Nevertheless, the forest interior exhibited a marginally higher stand density of 262±43.4 stems ha⁻¹ compared to 250±41.6 stems ha⁻¹ in the intermediate forest and 216±31.8 stems ha⁻¹ at the edge of the forest (Figure 16).

However, the stand density in Vuria exhibited a different pattern from the other four forest fragments (Chawia, Fururu, Mbololo and Ngangao) and the pattern was asymmetrical (Figure 16). The stand density increased from 157±57.1 stems ha⁻¹ at the edge of the forest to 300±98.2 stems ha⁻¹ in the intermediate forest before declining to 227±107.9 stems ha⁻¹ in the forest interior. However, the variation in the stand density along forest edge-interior gradient was not significantly different ($F_{(1,2)} = 1.01$; $p=0.381$).

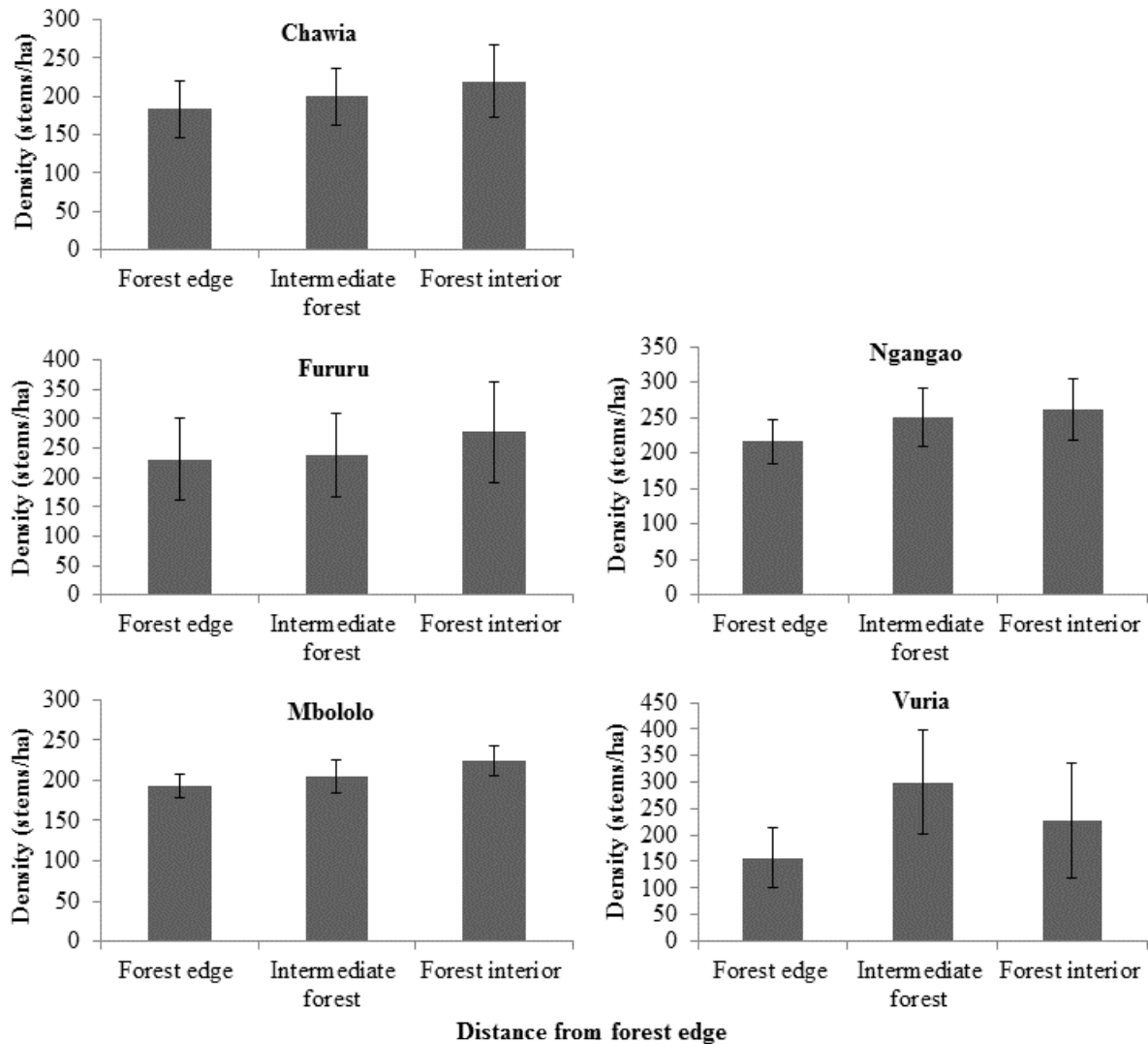


Figure 16: Comparison of mean stand density (\pm SE) along the forest edge-interior gradient for the five forest fragments in Taita Hills

4.8.3 Effect of forest edge on carbon stock

Although carbon stock varied among the forest fragments and along the gradient from the edge of the forest to the forest interior, the variations were not significant among the forest fragments ($F_{(1, 4)} = 0.64$; $p=0.634$). The variations in carbon stock along forest edge-interior gradient in the five forest fragments are presented in Figure 17. In Chawia, the carbon stock at the edge of the forest ($194.5 \pm 30.1 \text{ Mg ha}^{-1}$) was lower than in the intermediate forest ($284.5 \pm 82.94 \text{ Mg ha}^{-1}$) and the forest interior ($260.6 \pm 41.52 \text{ Mg ha}^{-1}$). However, the carbon stock

in Chawia did not vary significantly along the forest edge-interior gradient ($F_{(1, 2)} = 0.35$; $p=0.704$). Similarly, the carbon stock in Fururu along the gradient from the forest edge to the forest interior was not significantly different ($F_{(1, 2)} = 0.1$; $p=0.908$). Nevertheless, the carbon stock increased from $292.7 \pm 190.2 \text{ Mg ha}^{-1}$ on edge of the forest to $333.9 \pm 170.4 \text{ Mg ha}^{-1}$ in the intermediate forest and further to $394.0 \pm 150.4 \text{ Mg ha}^{-1}$ in the interior of the forest.

The carbon stock along forest edge-interior gradient in Mbololo was not significantly different ($F_{(1, 2)} = 0.61$; $p=0.544$); even though carbon stock was higher in the intermediate forest ($275.6 \pm 145.2 \text{ Mg ha}^{-1}$) than in the forest interior ($244.2 \pm 62.6 \text{ Mg ha}^{-1}$) and at the forest edge ($142.3 \pm 28.9 \text{ Mg ha}^{-1}$). Despite the fact that there was higher carbon stock in the intermediate forest in Ngangao ($247.3 \pm 103.1 \text{ Mg ha}^{-1}$) than at the forest edge ($239.7 \pm 73.1 \text{ Mg ha}^{-1}$) and in the forest interior ($232.1 \pm 58.7 \text{ Mg ha}^{-1}$), the carbon stock was not significantly different along the forest edge-interior gradient ($F_{(1, 2)} = 0.01$; $p=0.991$).

The carbon stock in Vuria did not vary significantly from the forest edge to the forest interior ($F_{(1, 2)} = 0.54$; $p=0.590$). The carbon stock in Vuria was higher in the intermediate forest ($260.0 \pm 127.9 \text{ Mg ha}^{-1}$) followed by forest edge ($187.4 \pm 138.5 \text{ Mg ha}^{-1}$) and the forest interior ($118.5 \pm 48.8 \text{ Mg ha}^{-1}$).

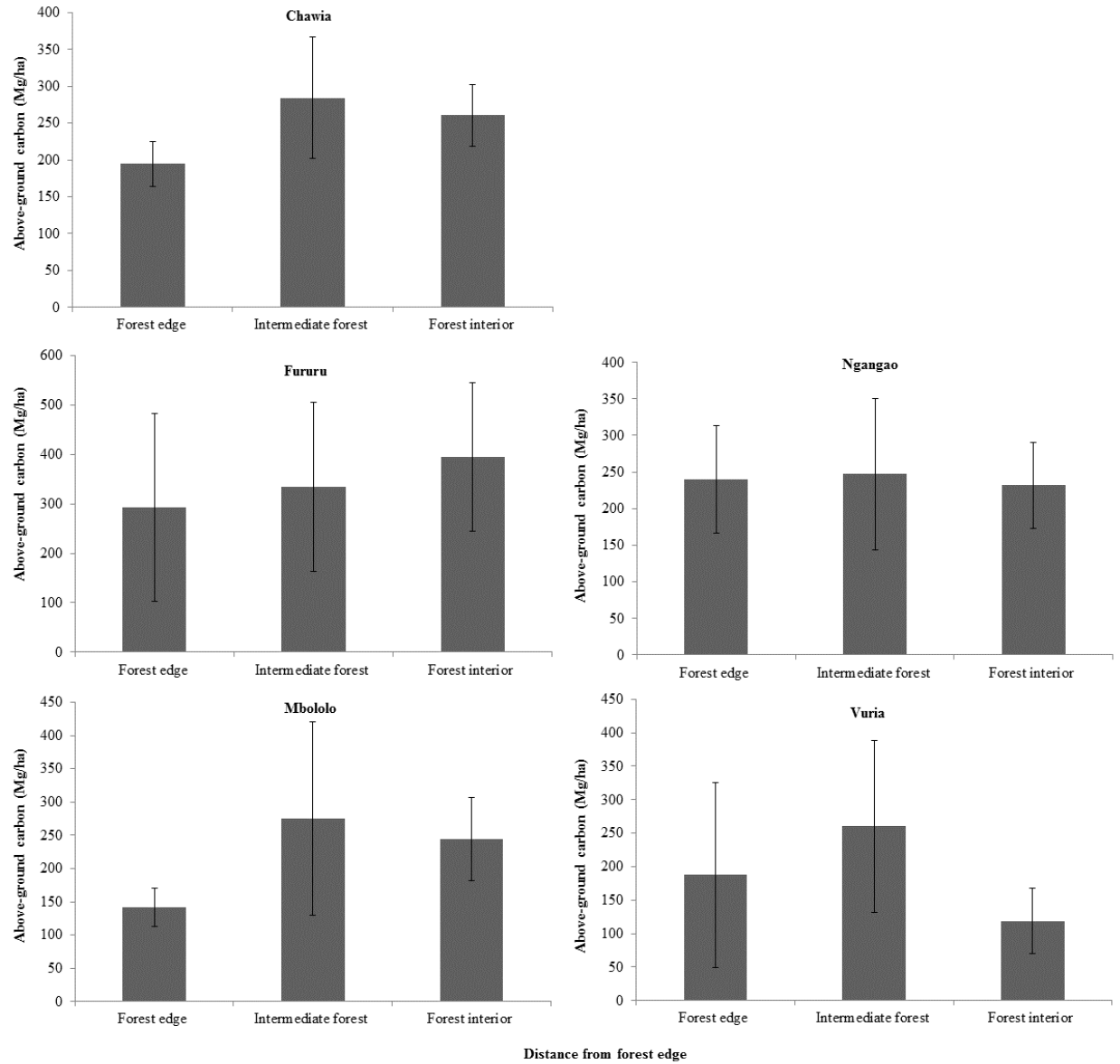


Figure 17: Comparison of mean above-ground carbon stock (\pm SE) along the forest edge-interior gradient for the five forest fragments in Taita Hills

4.8.4 Changes in carbon stock of the five forest fragments between 1973 and 2016

Generally, the carbon stock stored by the five forest fragments considerably reduced between 1973 and 2016 due to the loss in forest cover (Figure 18). In the past 43 years (1973-2016), deforestation of five major forest fragments in Taita Hills reduced the carbon stock stored by these forest fragments by 55.8 Gg as the forested area decreased. Carbon stock held in Chawia substantially reduced from 58.3 Gg in 1973 to 39.2 Gg in 2016 while Fururu's capacity to sequester carbon reduced slightly from 19.4 Gg to 18.8 Gg during the same period. In Mbololo, the carbon stock held by the forest declined by 9.5 Gg from 69.5 Gg in 1973 to 60.0 Gg in 2016. Ngangao and Vuria too lost considerable quantities of carbon seized between 1973 and 2016. Whereas Ngangao lost 10.2 Gg between 1973 and 2016, Vuria lost 16.4 Gg within the same period.

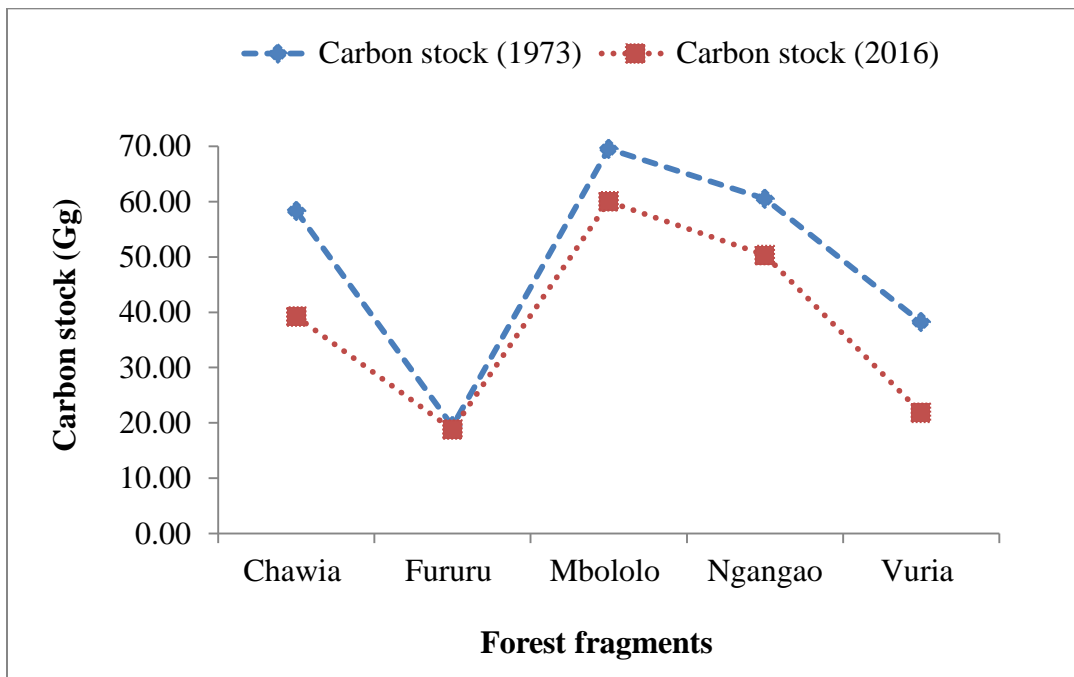


Figure 18: Changes in carbon stock of the five forest fragments in Taita Hills between 1973 and 2016

4.9 Effect of fragment size and forest edge on species richness

In total, 61 tree species belonging to 58 genera were identified and recorded in the five forest fragments (Appendix VII). The species richness varied among the forests with Chawia having 21 different species representing 21 genera while Fururu had 16 different species representing 16 genera. Forty-six different species were recorded in Mbololo representing 46 genera while in Ngangao, 39 different species representing 37 genera were recorded. In Vuria, which had the least number of species, only 10 different species representing 10 genera were recorded. Thus, large forest fragments (Mbololo and Ngangao) had high number of different tree species compared to small forest fragments (Chawia, Fururu and Vuria).

Comparative analyses of species richness along forest edge-interior gradient within each of the forest fragment showed no significant variation in all the forest fragments irrespective of their sizes (Table 7). The species richness in Chawia of 4.5 ± 1.9 at the edge of the forest, 3.0 ± 0.8 in the intermediate forest and 5.0 ± 0.8 in the forest interior did not vary significantly ($F_{(1, 2)} = 0.77$; $p=0.490$). Similarly, post hoc Tukey's HSD test at 95% probability level confirmed that the interior of Fururu had higher species richness (4.3 ± 1.9) than forest edge (4.0 ± 0.4) and intermediate forest (2.5 ± 0.5), which did not differ significantly ($F_{(1, 2)} = 0.76$; $p=0.507$). In Mbololo, the forest interior had higher species richness (9.3 ± 0.6) which did not vary significantly ($F_{(1, 2)} = 0.62$; $p=0.303$) from 8.9 ± 0.9 and 8.2 ± 0.9 at the forest edge and intermediate forest respectively. The intermediate forest in Ngangao exhibited greater species richness of 8.0 ± 1.0 which was not significantly different from 7.4 ± 1.1 and 7.4 ± 1.0 at the forest edge and in the forest interior respectively ($F_{(1, 2)} = 0.12$; $p=0.886$). Species richness in Vuria did not vary significantly along the gradient from the edge of the forest to the interior ($F_{(1, 2)} = 3.70$; $p=0.090$) although higher species richness occurred in the forest interior (3.7 ± 0.3) followed by the intermediate forest (2.7 ± 0.3) and the forest edge (2.3 ± 0.3).

Table 7: Comparison of mean species richness along the gradient from forest edge to the interior for the five forest fragments in Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same row are not significantly different at $p \leq 0.05$ level

Forest fragment	Area (ha)	Species richness			<i>p</i> value	LSD
		Forest edge	Intermediate forest	Forest interior		
Chawia	86.0	4.5±1.9 ^a	3.0±0.8 ^a	5.0±0.8 ^a	0.490	1.125
Fururu	5.0	4.0±0.4 ^a	2.5±0.5 ^a	4.3±1.9 ^a	0.507	1.108
Mbololo	200.0	8.9±0.9 ^a	8.2±0.9 ^a	9.3±0.6 ^a	0.544	0.303
Ngangao	120.0	7.4±1.1 ^a	8.0±1.0 ^a	7.4±1.0 ^a	0.886	0.470
Vuria	1.0	2.3±0.3 ^a	2.7±0.3 ^a	3.7±0.3 ^a	0.090	0.427

Evaluations of species richness on the forest edge, in the intermediate forest and in the forest interior provided a basis for biodiversity comparisons among the five forest fragments. Large forest fragments had higher species richness in all the habitats along the forest edge-interior gradient compared to small forest fragments (Table 8). There was a significant variation in species richness at the edge of the forest among the forest fragments ($F_{(1, 4)} = 6.59$; $p=0.001$). Mbololo had the highest species richness at the edge of the forest (8.9±0.9) followed by Ngangao (7.4±1.1) while Vuria had the lowest species richness (2.3±0.3). In the intermediate forest, the species richness was significantly higher in Mbololo (8.2±0.9) and Ngangao (8.0±1.0) than Chawia, Vuria and Fururu which had species richness of 3.0±0.8, 2.7±0.3 and 2.5±0.5 respectively ($F_{(1, 4)} = 9.97$; $p=0.001$). Similarly, the species richness in the forest interior was significantly higher in Mbololo than the rest of the forest fragments ($F_{(1, 4)} = 7.12$; $p=0.001$). The forest interior had species richness of 9.3±0.6 in Mbololo, 7.4±1.0 (Ngangao), 5.0±0.8 (Chawia), 4.3±1.9 (Fururu) and 3.7±0.3 in Vuria.

Table 8: Comparison of mean species richness at the forest edge, intermediate forest and forest interior for the five forest fragments of Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same row are not significantly different at $p \leq 0.05$ level

Forest edge- interior gradient	Forest fragment					<i>p</i> value	LSD
	Chawia	Fururu	Mbololo	Ngangao	Vuria		
Forest edge	4.5±1.9 ^{ab}	4.0±0.4 ^{abc}	8.9±0.9 ^c	7.4±1.1 ^{bc}	2.3±0.3 ^a	0.001	0.649
Intermediate forest	3.0±0.8 ^a	2.5±0.5 ^a	8.2±0.9 ^b	8.0±1.0 ^b	2.7±0.3 ^a	0.001	0.673
Forest interior	5.0±0.8 ^a	4.3±1.9 ^a	9.3±0.6 ^b	7.4±1.0 ^{ab}	3.7±0.3 ^a	0.001	0.485

4.10 Effect of fragment size on species diversity

The present findings revealed that large forest fragments had significantly higher species diversity than small forest fragments ($F_{(1, 4)} = 13.26$; $p=0.001$). Ngangao exhibited the highest diversity (2.86 ± 0.09) followed by Mbololo (2.84 ± 0.08), Fururu (2.09 ± 0.19) and Chawia (1.79 ± 0.29) while Vuria had the lowest species diversity (1.68 ± 0.22) (Table 9).

Table 9: Comparison of mean species diversity among the five forest fragments in Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same column are not significantly different at $p \leq 0.05$ level

Forest fragment	Area (ha)	Shannon-Wiener diversity index (H')
Chawia	86.0	1.79±0.29 ^a
Fururu	5.0	2.09±0.19 ^a
Mbololo	200.0	2.84±0.08 ^b
Ngangao	120.0	2.86±0.09 ^b
Vuria	1.0	1.68±0.22 ^a
<i>p</i> value	-	<0.001
LSD	-	0.47

Generally, large forest fragments exhibited high species diversity compared to small forest fragments along the forest edge-interior gradient. Moreover, large forest fragments showed significantly higher species diversity at the forest edge than small forest fragments ($F_{(1,4)} = 5.03$; $p=0.006$). Similarly, small forest fragments had significantly lower species diversity in the intermediate forest compared to the large forest fragments ($F_{(1,4)} = 5.90$; $p=0.003$). Regarding the forest interior, large forest fragments were significantly more species diverse than small forest fragments ($F_{(1,4)} = 3.38$; $p=0.030$) (Table 10).

Table 10: Comparison of mean Shannon-Wiener diversity indices at the forest edge, intermediate forest and forest interior across the five forest fragments in Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same column are not significantly different at $p \leq 0.05$ level

Forest fragment	Area (ha)	Shannon-Wiener diversity index (H')		
		Forest edge	Intermediate forest	Forest interior
Chawia	86.0	1.60±0.65 ^{ab}	1.52±0.51 ^a	2.25±0.33 ^{ab}
Fururu	5.0	2.12±0.42 ^{ab}	1.84±0.05 ^{ab}	2.30±0.47 ^{ab}
Mbololo	200.0	2.79±0.11 ^b	2.40±0.16 ^b	2.99±0.11 ^b
Ngangao	120.0	2.72±0.17 ^b	2.98±0.10 ^b	2.88±0.22 ^{ab}
Vuria	1.0	1.20±0.32 ^a	1.99±0.29 ^{ab}	1.87±0.48 ^a
<i>p</i> value		0.006	0.003	0.030
LSD		0.9635	0.3850	0.8071

4.11 Effect of fragment size on carbon stock of the forest fragments

The results of overall average carbon stocks per unit area for each of the forest fragments are presented below (Table 11). Despite the differences in the sizes of the forest fragments and the levels of disturbance, the carbon stock per unit area did not significantly differ among the forest fragments ($F_{(1,4)} = 0.64$; $p=0.634$). Nevertheless, Chawia had marginally more carbon stock per unit area ($352.3 \pm 83.1 \text{ Mg ha}^{-1}$) followed by Fururu ($341.1 \pm 98.4 \text{ Mg ha}^{-1}$), Ngangao ($239.9 \pm 46.9 \text{ Mg ha}^{-1}$) and Mbololo ($219.7 \pm 51.1 \text{ Mg ha}^{-1}$). Vuria had the least carbon stock per unit area ($180.6 \pm 56.5 \text{ Mg ha}^{-1}$).

Table 11: Comparison of mean aboveground carbon stock in the five forest fragments of Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same column are not significantly different at $p \leq 0.05$ level

Forest fragment	Above-ground carbon stock (Mg ha ⁻¹)
Chawia	352.3±83.1 ^a
Fururu	341.1±98.4 ^a
Mbololo	219.7±51.1 ^a
Ngangao	239.9±46.9 ^a
Vuria	180.6±56.5 ^a
<i>p</i> value	0.634
LSD	259.1

4.12 Effect of forest edge on edaphic factors

Evaluation of seven edaphic factors along the gradient from the forest edge to the forest interior ascertained the changes that occur because of the edge effect. The soil physical and chemical properties measured include moisture content (%), nitrogen (%), phosphorus (ppm), potassium (ppm), pH, electrical conductivity (mS/cm) and organic carbon (%). The results on the vicissitudes in soil physical and chemical properties along forest edge-interior gradient appear in sub-sections 4.12.1, 4.12.2, 4.12.3 and 4.12.4.

4.12.1 Relationship between edaphic factors along the forest edge-interior gradient

Soil pH, electrical conductivity (EC), nitrogen, organic carbon, potassium, phosphorus and moisture are fundamental indicators of soil health and hence affect the growth performance of trees and other vegetation in a natural forest. Therefore, soil pH, EC, nitrogen, organic carbon, potassium, phosphorus and moisture were analysed for the relationship with one another and with distance from the forest edge to ascertain their interactions.

Spearman's rank correlation analysis revealed a highly significant negative linear relationship between soil pH and electrical conductivity (EC) ($r = -0.592$; $p=0.001$) (Table 12). Similarly, a highly significant negative relationship was observed between soil pH and organic carbon ($r = -0.573$; $p=0.001$). Soil pH also showed significant negative relationship with nitrogen content ($r = -0.487$; $p=0.001$), phosphorus ($r = -0.428$; $p=0.001$) and distance from the forest

edge ($r = -0.374$; $p=0.001$). However, soil pH had a positive relationship with soil moisture even though the relationship was not significant ($r = 0.043$; $p=0.723$). Electrical conductivity (EC) had a highly significant positive relationship with organic carbon ($r = 0.698$; $p=0.001$) and nitrogen content ($r = 0.696$; $p=0.001$). Equally, EC correlated positively with phosphorus ($r = 0.352$; $p=0.002$) and distance from the forest edge ($r = 0.260$; $p=0.027$). However, EC had insignificantly negative relationship with potassium ($r = -0.177$; $p=0.137$) and soil moisture ($r = -0.034$; $p=0.779$).

Moreover, Spearman's correlation coefficient (r) indicated that organic carbon had a highly significant positive relationship with nitrogen content ($r = 0.693$; $p=0.001$), phosphorus ($r = 0.431$; $p=0.001$) and distance from the forest edge ($r = 0.421$; $p=0.001$) (Table 12). Potassium ($r = -0.047$; $p=0.698$) and soil moisture ($r = -0.209$; $p=0.078$) had negative relationship with organic carbon although the relationship was not significant. A highly significant positive relationship was observed between nitrogen content and phosphorus ($r = 0.404$; $p=0.001$). In addition, the distance from the forest edge exhibited a significant positive relationship with nitrogen content ($r = 0.244$; $p=0.039$). Even though soil moisture was positively correlated with nitrogen content, the relationship was not significant ($r = 0.099$; $p=0.410$). Nitrogen content in the soil was negatively affected by potassium along the forest edge-interior gradient though not significantly ($r = -0.077$; $p=0.518$). Phosphorus showed a positive relationship with potassium ($r = 0.109$; $p=0.361$), moisture ($r = 0.028$; $p=0.817$) and distance from the forest edge ($r = 0.062$; $p=0.603$) but the relationship was not significant.

While the level of potassium concentration in the soil was positively correlated to soil moisture, the relationship was not significant ($r = 0.028$; $p=0.817$). On the other hand, potassium had a significant negative relationship with the distance from the forest edge ($r = -0.282$; $p=0.016$). Further, soil moisture had a negative relationship with the distance from the forest edge. However, the relationship was insignificant ($r = -0.184$; $p=0.123$).

Table 12: Spearman rank correlation analysis of soil pH, electrical conductivity, nitrogen, organic carbon, potassium, phosphorus and moisture along forest edge-interior gradient. EC = Electrical conductivity; OC = Organic Carbon; N = Nitrogen; P = Phosphorus; K = Potassium; MS = Moisture and DFE = Distance from Forest Edge

Soil property	pH	EC (mScm)	OC (%)	N (%)	P (ppm)	K (ppm)	MS (%)	DFE (m)
pH	1.000							
EC (mScm)	-0.592**	1.000						
OC (%)	-0.573**	0.698**	1.000					
N (%)	-0.487**	0.696**	0.693**	1.000				
P (ppm)	-0.428**	0.352**	0.431**	0.404**	1.000			
K (ppm)	0.368**	-0.177	-0.047	-0.077	0.109	1.000		
MS (%)	0.043	-0.034	-0.209	0.099	0.028	0.228	1.000	
DFE (m)	-0.374**	0.260*	0.421**	0.244*	0.062	-0.282*	-0.184	1.000

**Correlation is significant at the 0.01 level (2-tailed)

*Correlation is significant at the 0.05 level (2-tailed)

Correlation is not significant for values without asterisks at both 0.01 and 0.05 levels (2-tailed)

4.12.2 Effects of forest edge on soil moisture content

The percentage soil moisture content varied significantly along the forest edge-interior gradient in Chawia ($F_{(1, 2)} = 47.47$; $p=0.001$), Fururu ($F_{(1, 2)} = 12.96$; $p=0.001$), Ngangao ($F_{(1, 2)} = 9.97$; $p=0.001$) and Vuria ($F_{(1, 2)} = 11.25$; $p=0.001$). However, no significant variations in soil moisture content were observed in Mbololo ($F_{(1, 2)} = 1.45$; $p=0.235$) (Table 13). Although the soil moisture varied significantly in Chawia, Fururu, Ngangao and Vuria along the forest edge-interior gradient, the pattern was non-linear. Chawia had significantly higher soil moisture content in the forest interior while in Fururu and Ngangao, the intermediate forest exhibited significantly higher soil moisture content than the forest edge and interior. Soil moisture content in Vuria was higher at the edge of the forest than in the intermediate forest and forest interior.

Table 13: Comparison of mean soil moisture (percentage) along the forest edge-interior gradient for the five forest fragments in Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same row are not significantly different at $p \leq 0.05$ level

Forest fragment	Forest edge-interior gradient				
	Forest edge	Intermediate forest	Forest interior	<i>p</i> value	LSD
Chawia	1.50±0.00 ^a	1.65±0.04 ^a	2.23±0.08 ^b	<0.001	0.171
Fururu	2.77±0.15 ^b	3.51±0.53 ^b	1.75±0.11 ^a	<0.001	0.692
Mbololo	1.53±0.05 ^a	1.64±0.07 ^a	1.65±0.04 ^a	0.235	0.154
Ngangao	1.25±0.02 ^a	1.35±0.02 ^b	1.24±0.02 ^a	<0.001	0.057
Vuria	3.82±0.44 ^b	2.56±0.12 ^a	2.48 ±0.12 ^a	<0.001	0.555

4.12.3 Effect of forest edge on soil pH and electrical conductivity

Soil pH and electrical conductivity (EC) are important indicators of soil health and hence affects nutrients availability, microbial activities and the growth of trees and other vegetation in a natural forest. Soil pH and EC were evaluated along the forest edge-interior gradient to establish the changes that occurred because of the edge effect created by forest fragmentation. The results showed no significant variations in soil pH along the forest edge-interior gradient in Chawia ($F_{(1, 2)} = 0.36$; $p=0.700$) and Fururu ($F_{(1, 2)} = 0.41$; $p=0.667$). However, Mbololo ($F_{(1, 2)} = 15.48$; $p=0.001$) and Ngangao ($F_{(1, 2)} = 13.21$; $p=0.001$) had significantly higher soil pH at the edge of the forest than in the intermediate forest and the forest interior. Vuria had significantly lower soil pH at the edge of the forest than in the intermediate forest and the forest interior ($F_{(1, 2)} = 3.63$; $p=0.042$). The EC was not significantly different along the forest edge-interior gradient in Chawia ($F_{(1, 2)} = 0.11$; $p=0.896$), Mbololo ($F_{(1, 2)} = 1.01$; $p=0.367$), Ngangao ($F_{(1, 2)} = 0.60$; $p=0.555$) and Vuria ($F_{(1, 2)} = 0.15$; $p=0.861$). Fururu had significantly higher EC at the edge of the forest than the forest interior and intermediate forest ($F_{(1, 2)} = 3.44$; $p=0.048$) (Table 14).

Table 14: Comparison of mean soil pH and electrical conductivity along the forest edge-interior gradient for the five forest fragments in Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same row are not significantly different at $p \leq 0.05$ level

Forest fragment	Soil property	Forest edge-interior gradient				LSD
		Forest edge	Intermediate forest	Forest interior	<i>p</i> value	
Chawia	pH	5.58±0.26 ^a	5.34±0.14 ^a	5.62±0.32 ^a	0.700	0.720
	EC (mS/cm)	0.107±0.019 ^a	0.123±0.019 ^a	0.116±0.032 ^a	0.896	0.070
Fururu	pH	6.02±0.19 ^a	6.01±0.13 ^a	5.85±0.10 ^a	0.667	0.430
	EC (mS/cm)	0.132±0.021 ^b	0.080±0.010 ^a	0.093±0.011 ^{ab}	0.048	0.043
Mbololo	pH	6.10±0.19 ^b	5.25±0.17 ^a	4.79±0.14 ^a	<0.001	0.475
	EC (mS/cm)	0.138±0.022 ^a	0.174±0.028 ^a	0.183±0.021 ^a	0.367	0.067
Ngangao	pH	5.23±0.102 ^b	4.69±0.069 ^a	4.72±0.077 ^a	<0.001	0.239
	EC (mS/cm)	0.155±0.037 ^a	0.123±0.017 ^a	0.121±0.013 ^a	0.555	0.07
Vuria	pH	5.26±0.09 ^a	5.64±0.105 ^b	5.67±0.153 ^b	0.042	0.351
	EC (mS/cm)	0.074±0.008 ^a	0.069±0.011 ^a	0.076±0.009 ^a	0.861	0.028

4.12.4 Effect of forest edge on soil organic carbon, nitrogen, phosphorus and potassium

In Chawia, there were no significant variations in organic carbon ($F_{(1, 2)} = 1.10$; $p=0.345$), nitrogen ($F_{(1, 2)} = 0.16$; $p=0.855$), phosphorus ($F_{(1, 2)} = 1.38$; $p=0.265$) and potassium ($F_{(1, 2)} = 2.44$; $p=0.102$) along the forest edge-interior gradient (Table 15). Similarly, no significant variations in organic carbon ($F_{(1, 2)} = 0.56$; $p=0.576$), nitrogen ($F_{(1, 2)} = 2.01$; $p=0.156$), phosphorus ($F_{(1, 2)} = 0.38$; $p=0.690$) and potassium ($F_{(1, 2)} = 1.67$; $p=0.209$) were observed in Vuria along the gradient from the edge of the forest to the forest interior. In Fururu, soil nitrogen was significantly higher at the forest edge than in the intermediate forest and the forest interior ($F_{(1, 2)} = 3.77$; $p=0.038$). However, organic carbon ($F_{(1, 2)} = 0.36$; $p=0.703$), phosphorus ($F_{(1, 2)} = 0.00$; $p=0.997$) and potassium ($F_{(1, 2)} = 1.35$; $p=0.278$) were not significantly different along the forest edge-interior gradient (Table 15). Potassium varied significantly along the forest edge-interior gradient in Mbololo ($F_{(1, 2)} = 6.58$; $p=0.002$) while organic carbon ($F_{(1, 2)} = 1.68$; $p=0.192$), nitrogen ($F_{(1, 2)} = 0.79$; $p=0.458$) and phosphorus ($F_{(1, 2)}$)

= 0.85; $p=0.430$) were not significantly affected by the edge effect. Similarly, organic carbon ($F_{(1, 2)} = 1.17$; $p=0.321$), nitrogen ($F_{(1, 2)} = 1.05$; $p=0.358$) and phosphorus ($F_{(1, 2)} = 0.93$; $p=0.404$) were not significantly different along the forest edge-interior gradient in Ngangao. However, potassium was significantly higher at the edge of the forest than in the intermediate forest and forest interior ($F_{(1, 2)} = 3.64$; $p=0.035$) (Table 15).

Table 15: Comparison of mean organic carbon, nitrogen, phosphorus and potassium along the forest edge-interior gradient for the five forest fragments in Taita Hills. The presented values are means with standard error of mean. Values with similar letters in superscripts in the same row are not significantly different at $p \leq 0.05$ level. OC = Organic Carbon (%); N = Nitrogen (%); P = Phosphorus (ppm) and K = Potassium (ppm)

Forest fragment	Soil property	Forest edge	Intermediate forest	Forest interior	<i>p</i> value	LSD
Chawia	OC (%)	6.98±0.86 ^a	6.94±0.98 ^a	9.52±2.06 ^a	0.345	4.05
	N (%)	0.59±0.17 ^a	0.63±0.15 ^a	0.72±0.18 ^a	0.855	0.48
	P (ppm)	5.0±0.58 ^a	7.0±1.26 ^a	11.0±4.62 ^a	0.265	8.0
	K (ppm)	120.87±15.39 ^a	89.74±9.16 ^a	88.14±9.85 ^a	0.102	33.96
Fururu	OC (%)	4.94±0.78 ^a	4.17±0.65 ^a	4.74±0.62 ^a	0.703	2.025
	N (%)	0.79±0.04 ^b	0.64±0.05 ^{ab}	0.57±0.08 ^a	0.038	0.17
	P (ppm)	5.0±1.02 ^a	5.0±0.64 ^a	5.0±0.40 ^a	0.997	2.20
	K (ppm)	111.96±21.12 ^a	77.81±13.06 ^a	83.79±11.0 ^a	0.278	46.45
Mbololo	OC (%)	9.14±0.92 ^a	10.08±1.01 ^a	11.56±0.87 ^a	0.192	2.641
	N (%)	0.81±0.078 ^a	0.81±0.08 ^a	0.93±0.08 ^a	0.458	0.23
	P (ppm)	14.0±1.84 ^a	12.0±1.63 ^a	15.0±1.75 ^a	0.430	4.9
	K (ppm)	222.20±31.92 ^b	143.10±15.77 ^a	112.60±13.82 ^a	0.002	62.09
Ngangao	OC (%)	6.91±0.68 ^a	7.09±0.80 ^a	8.59±1.05 ^a	0.321	2.44
	N (%)	0.60±0.07 ^a	0.53±0.06 ^a	0.67±0.06 ^a	0.358	0.19
	P (ppm)	8.0±1.32 ^a	12.0±2.79 ^a	11.0±1.44 ^a	0.404	5.60
	K (ppm)	96.48±15.94 ^b	54.16±6.04 ^a	72.35±8.96 ^{ab}	0.035	31.74
Vuria	OC (%)	5.591±0.44 ^a	4.84±0.60 ^a	5.12±0.44 ^a	0.576	1.47
	N (%)	0.58±0.06 ^a	0.57±0.06 ^a	0.45±0.04 ^a	0.156	0.16
	P (ppm)	18.0±4.26 ^a	23.0±4.81 ^a	23.0±5.24 ^a	0.690	14.0
	K (ppm)	163.10±36.51 ^a	246.50±38.83 ^a	221.50±21.26 ^a	0.209	96.69

CHAPTER FIVE

DISCUSSION

5.1 Introduction

The findings of the study are discussed by elucidating the evidence used in supporting or rejecting the hypotheses tested. The discussions provide interpretations of the findings and meaning associated with the changes in broad patterns observed in tree species diversity, species richness, the abundance of dominant and adaptable tree species, edaphic factors and carbon sequestration potential along the forest edge-interior gradient. Additionally, there is an elucidation of the patterns observed in species richness, tree diversity and carbon stock in the forest fragments of different sizes. The validation of the research findings takes the form of comparisons and contrasts of the present findings with related studies reported by other authors.

5.2 Forest cover change

The forest loss for the five forest fragments in Taita Hills between 1973 and 2016 has been relatively rapid. Approximately 5.4 ha of the studied forest fragments were lost annually. However, the forest loss varied among the five forest fragments. The highest forest loss occurred in Vuria and the lowest in Fururu. Vuria lost 91.0 ha between 1973 and 2016 while Fururu lost a meagre 1.8 ha. Chawia lost 54.2 ha whereas Mbololo and Ngangao lost 43.5 and 42.4 ha respectively during the same period. The variations in the forest area/cover lost among the forest fragments may be due to a number of factors. The legal and administrative status of the forest may have affected the rate of forest loss as well as the forest cover/area lost. The forest fragments under the management of Kenya Forest Service (KFS) exhibited low rate of forest loss compared to forest fragments managed/owned by the County government (formerly Trust land/County council forests). Mbololo, Ngangao and Fururu which are gazetted forests managed by KFS lost the smallest area compared to Chawia and Vuria which are County forests under the management of Taita Taveta County. On average, Mbololo, Ngangao and Fururu lost 29.2 ha from 1973 to 2016 compared to an average forest area of 72.6 ha lost in Chawia and Vuria. In fact, the forest cover loss witnessed in KFS owned and managed forest fragments was less than half of the loss suffered by forest fragments managed by the County government. This is because the number of staff responsible for day-to-day management and conservation of the forests at County level is inadequate; this hinders effective management of the forests on sustainable basis.

Furthermore, the gazettement of the forest fragments and the year of gazettement are important factors that explained the variations in the forest area lost and the associated rate of forest loss for KFS managed forest fragments. Fururu which according to KFS (2016) was gazetted in 1991 lost only 4.9 ha (8.7%) while Ngangao which was gazetted in 2003 lost 24.5 ha (10.5%) of its forest cover between 1973 and 1987. During the same period, Mbololo which was gazetted in 1991 (KFS, 2016) lost 41.8 ha (13.2%) of its forest cover. However, after the gazettement, the forest loss decreased substantially with Fururu losing merely 1.9 ha (3.6%) of its forest cover between 1987 and 2001. Ngangao and Mbololo lost 12.2 ha (5.4%) and 1.4 ha (0.5%) respectively between 1987 and 2001. It is worth noting that large areas of the gazetted forest fragments were lost before forest gazettement but the loss decreased after the gazettement. Vuria which is a County managed forest (not gazetted) lost 66.8 ha (31.6%) of its forest cover between 1973 and 1987 which was about four times the forest area Fururu lost during the same period. Moreover, the forest loss in Vuria from 1973 to 1987 was three and two and half times the loss that occurred in Ngangao and Mbololo respectively over the same period.

There was extensive loss in forest area in Vuria (24.2 ha) between 1987 and 2001. The forest loss of 24.2 ha (16.7%) recorded in Vuria was thus four and half times the loss witnessed in Fururu and three and thirty three times the loss witnessed in Ngangao and Mbololo respectively during the same period. This implies that if Vuria had been gazetted at the same time as Fururu, Ngangao and Mbololo, the forest area lost between 1987 and 2001 could have been probably reduced between three and thirty three times. Alternatively, if Community Forest Associations (CFAs) in Vuria had been empowered to effectively participate in the management of the forest, the forest area lost between 1987 and 2001 could be smaller compared to the forest area lost (24.2 ha) as reported in the present study.

The annual rate of forest cover loss was lower in forest fragments that were gazetted in early 1990s compared to those that were gazetted after the year 2000. Fururu, gazetted 12 years before Ngangao had an annual rate of forest loss of 0.07% that was remarkably lower than 0.4% witnessed in Ngangao. Chawia and Vuria, not gazetted to date, exhibited annual rate of forest loss of 1.3% and 2.1% respectively. Therefore, it is imperative to note that forest fragments not gazetted by KFS and under the management of County government showed high annual rate of forest loss compared to the gazetted forests managed by KFS. The gazettement of forest

fragments improved the management of the forests by reducing the rate of forest loss. Therefore, gazettement of forests is fundamental in halting deforestation and for biodiversity conservation.

The annual rate of population growth in the sub-locations neighbouring the forest fragments studied influenced the rate of forest loss and the forest cover/area lost in some forest fragments and not others. In Vuria where the annual rate of forest loss was high (1.0%), the annual growth rate of population in the neighbouring Mwanda sub-location was also high (5.6%). Similarly, Chawia which had the second highest annual rate of forest loss of 0.8%, the annual rate of population growth in the neighbouring Chawia and Bura sub-locations was relatively high (1.9%). Whereas the annual population growth rate for Mgambonyi sub-location which borders Ngangao was relatively high (1.2%), the annual rate of forest loss was low (0.4%). The annual rate of population growth in Mbololo and Ronge sub-locations that borders Mbololo is 8.6%, which is comparatively high and uncorrelated to the annual rate of forest loss of 0.3%. Fururu whose neighbouring sub-locations (Kidaya Ngerenyi and Kishamba) had annual population growth rate of 1.8% that was comparable to that of the sub-locations neighbouring Chawia had very low rate of forest loss annually (0.07%) that was eleven times less than Chawia. The impact of population growth on the rate of forest loss varied greatly among the forest fragments studied. Consequently, it is clear that the increase in population alone can only partially explain the changes occurring in the forest fragments. Other factors such as incidences of forest fires, management system and gazettement and/or non-gazettement of the forests could be interacting with changes in human population thereby altering the areas of the forest fragments and the quality of ecosystem services provided by these forests.

The largest forest area across the five forest fragments was lost between 1973 and 1987. During this period, Chawia lost 54.6 ha, Fururu (4.9 ha), Mbololo (41.8 ha), Ngangao (24.5 ha) and Vuria (66.8 ha). This massive loss in forest cover during this period could be attributed to the tragedy of the commons whereby local communities freely extracted materials from the forest fragments in unsustainable manner due to uncoordinated management. Forest management in Taita Hills was not well coordinated from 1973 to 1977 (Beentje, 1988). It was until 1977 that well-coordinated forest management started in Taita Hills following a presidential decree banning logging in the forests without a permit (Beentje, 1988). Besides, the first District Forest Officer (DFO) was posted to Taita Taveta County (formerly Taita Taveta District) in 1982. Prior to the posting, forest management responsibilities in Taita Taveta County were within the

domain of Provincial Forest Office in Mombasa. Management of the forests from the Mombasa office was not effective due to the distance and this contributed to poor enforcement that resulted into wanton encroachment of the forest fragments for agriculture and human settlement.

The loss in forest cover of between 3.2% and 43.0% for the five forest fragments studied compares well with forest cover loss of between 3.8% and 40.0% in a period of 50 years previously reported by Eva *et al.* (2006) and Harper *et al.* (2007) for Afromontane forests in Madagascar. The Afromontane forests in Madagascar have high flora and fauna biodiversity similar to Taita Hills' forests. Furthermore, related studies on forest cover change using time-series analysis have shown that Kakamega and Nandi forests lost 31.0% of the original area in a period of 29 years (Lung and Schaab, 2010); this compares well with the forest loss of 32.7% reported in Chawia and the overall average loss for the five forest fragments of 23.2%. According to Beentje (1988), forest loss in Taita Hills since 1960's has been substantial. Vuria has lost 99% since 1960's, Chawia (85%), Ngangao (50%) and below 50% for Mbololo (Beentje, 1988). The present findings show that since early 1970's, Vuria lost 43.0%, Chawia (32.7%), Ngangao (16.8%) and Mbololo (13.7%). Consequently, the linear trend in forest cover loss reported by Beentje (1988) is analogous to the trend observed in the current study whereby Vuria lost the largest forest area followed by Chawia, Ngangao and Mbololo. The annual rate of deforestation of 0.5% reported in this study compares well with the annual rate of deforestation in humid tropical forests of Africa of 0.4-0.5% (FAO, 2001; Hansen *et al.*, 2008) but is higher than the global annual deforestation rate of 0.08-0.2% as earlier reported by other authors (Keenan *et al.*, 2015; Sloan and Sayer, 2015). Moreover, the annual rate of deforestation of 0.5% reported in this study is higher than national annual rate of deforestation of Kenya of 0.3% reported by Wu (2011).

In 2016, the areas for the five forest fragments studied were as follows: Chawia (111.4 ha), Fururu (55.1 ha), Mbololo (273.0 ha), Ngangao (209.6 ha) and Vuria (120.7 ha). The area for Chawia of 111.4 ha reported in this study compares well with 111.3 ha previously reported by Pellikka *et al.* (2009) but differs from 86.0 ha reported by KFS (2016) and Mbuthia (2003). The present findings on the forest area for Vuria also deviate from earlier reports by Run (1995) and Lanne (2007) of 100.0 ha and 91.0 ha respectively. Preceding studies have estimated the area for Fururu as 14.1 ha (KFS, 2016), 62.1 ha (Pellikka *et al.*, 2009) and 13.0 ha (Mbuthia, 2003). Consequently, the estimated forest area for Fururu in the current study of 55.1 ha compares well

with findings of Pellikka *et al.* (2009) but differs from forest area reported by KFS (2016) and Muthia (2003). The method of assessment used by KFS is not clearly stated and therefore the difference in the area of Fururu presented in this study and the area reported by KFS for the same forest fragment may be due to the differences in the methods used. The present study estimates the area for Mbololo to be 273.0 ha which is similar to 272.7 ha reported by KFS (2016). Only KFS has quantified the area for Mbololo. The technical and financial support from the Japanese government facilitated this assessment in 2016. Before 2016, the forest area for Mbololo used by KFS for management planning was based on underestimated forest area of 200.0 ha which was 26.7% less than the actual forest area. This means that KFS has been allocating insufficient human and financial resources for the management and conservation of the forest and hence compromising management effectiveness.

The area for Ngangao (209.6 ha) reported in this particular study is quite similar to 206.6 ha reported in earlier studies (Pellikka *et al.*, 2009). However, the area for Ngangao reported in this study was different from 113.0 ha (Run, 1995), 123.0 ha (Muthia, 2003) and 136.0 ha (Adriaensen *et al.*, 2006) and 139.0 ha (Lanne, 2007) as reported in previous studies undertaken at different times. According to the present study results, Vuria had a total forest area of 120.7 ha which is close to 115.0 ha reported by KFS (2016). Prior to this study, only one comprehensive quantitative study to determine the area for Vuria existed (KFS, 2016). Before KFS and present studies were undertaken, records had indicated that the area for Vuria was 1.0 ha (Beentje, 1988). Thus, management decisions for Vuria by the forest managers depended on exceedingly underestimated forest area. As a result, inadequate resources, both human and financial were made available for management and conservation of Vuria. This also explains why Vuria has lost the largest forest area for the last 43 years compared to other forest fragments whose estimated areas used in decision making prior to the study carried out by KFS in 2016 were almost similar to KFS estimates of 2016.

There is great similarity in the areas of the three forest fragments (Chawia, Fururu and Ngangao) in Taita Hills given by Pellikka *et al.* (2009) and the present study. This implies that with proper analysis and interpretation, the use of Landsat images and aerial photographs in quantitative assessment of the changes in forest cover/area provides accurate results that are comparable. On the other hand, there is great variation in the areas of the forest fragments given by other authors with the exception of Pellikka *et al.* (2009) and this could be due to the

differences in the analysis and interpretation methods used. Moreover, in the previous studies except Pellikka *et al.* (2009), the methods used are inadequately described and hence unclear and this most likely reduced the accuracy of the results. However, the areas of the forest fragments reported in the present study are fairly accurate and useful for adoption by forest managers and policy makers for making management decisions for sustainable conservation of the forest fragments in order to safeguard the rich flora and fauna diversity found in these forest fragments.

5.3 Trends in forest cover change for undisturbed, disturbed and open ground forest areas

The trend analysis in forest cover/area change between 1973 and 2016 for undisturbed, disturbed and open ground areas within the forests showed great variation for the forest fragments studied. In Chawia, the undisturbed forest area decreased between 1973 and 1987. This decrease could be attributed to rapid population increase witnessed between 1969 and 1979 that increased the demand for wood products particularly poles for construction and firewood for domestic energy needs as people encroached the forest for settlement. The population of Chawia sub-location, which borders the forest, increased from 21,635 in 1969 to 25,654 in 1979 translating to 18.6% increase in 10 years' period (KNBS 1969; 1979). Moreover, there was no formal protection of the forest from 1973 to 1977 (Beentje, 1988). During this period, communities entered into the forest without restrictions and extracted wood products hence causing heavy disturbance that resulted into increased areas of disturbed sites within the forest; consequently reducing the area under pristine forest. Between 1987 and 2001, the state of Chawia forest fragment improved following the increase in undisturbed forest area. This improvement was because of enrichment planting of exotic tree species (*Cupressus lusitanica*, *Grevillea robusta* and *Eucalyptus* spp) in the disturbed sites within the forest (Pellikka *et al.*, 2009). The reduction in undisturbed forest area between 2001 and 2012 may be due to forest fires that occurred in 2003 and 2006. According to KFS records on fire incidences, the estimated combined area burned in Chawia in 2003 and 2006 was approximately 7.0 ha. Occurrence of forest fires is common in Taita Hills forests during the dry season (December-March) and whenever the fires occur, they are usually very intense and spread quickly due to the presence of dense layer of litter on the forest floors that usually acts as fuel. Over time, the burned sites within the forest recovered resulting into increased undisturbed forest area between 2012 and 2016. Moreover, rehabilitation of degraded sites in Chawia initiated in 2011 by KEFRI in

partnership with a local CBO, Chawia Environmental Group also contributed to increased area of restored forests.

The undisturbed forest area decreased substantially in Fururu between 1973 and 1987. Rapid population growth in the neighbouring Kidaya Ngerenyi and Kishamba sub-locations was responsible for the heavy loss witnessed during the same period. The population of the two sub-locations grew from 10,117 in 1969 to 12,314 in 1979; translating to 17.8% increase (KNBS, 1969; 1979). Most of the forest degradation that caused a decline in undisturbed forest area and hence an increase in the area of disturbed sites in the forest probably happened before 1979 when the management of Fururu was not coordinated and there was illicit encroachment for agriculture and human settlement. The slight decrease of undisturbed forest area between 1987 and 2001 and the subsequent increase between 2001 and 2012 may be due to the gazettelement of the forest fragment in 1991 that streamlined the management making it effective and the establishment of Eucalyptus plantations on previously bare areas (Pellikka *et al.*, 2009). The decrease in undisturbed forest area observed in 2016 was because of forest fire that occurred in January 2015 destroying an estimated area of 5.0 ha according to KFS fire incidences records.

Similar to Fururu, the undisturbed forest area in Mbololo reduced by 31.3% between 1973 and 1987 followed a slight increase between 1987 and 2001. As the undisturbed forest area increased, the disturbed forest decreased. Lack of formal protection as well as increase in population of Mbololo and Ronge sub-locations, which borders the forest, explains why disturbed areas/sites within the forest increased between 1973 and 1987. The population of Mbololo and Ronge sub-locations increased from 14,394 in 1969 to 26,767 in 1979; this represents 86.0% increase (KNBS, 1969; 1979). The massive growth in the population augmented the demand for wood products especially poles and timber for construction of new homes resulting into uncontrolled harvesting of trees and hence opening up the forest and creating artificial gaps. This reduced the area of undisturbed forest but increased the area for disturbed forest. With the gazettelement of Mbololo in 1991, a formal management system was established and this explains the slight improvement in the forest condition between 1987 and 2001. Moreover, due to accessibility challenges, the exotic plantations (*C. lusitanica* and *P. patula*) reflect no symptoms of extraction since they were planted in the late 1980's in areas opened up by illegal logging and this has helped in maintaining the forest in good condition. No fire incidences have occurred in Mbololo in the last 34 years (1982-2016) and this together with

the impact of gazettement explains the improved forest condition between 2001 and 2016 as confirmed by the reduction in the areas for disturbed sites and open grounds within the forest fragment.

In comparison to Chawia, Fururu and Mbololo, Ngangao exhibited a different trend in forest cover change for undisturbed forest area. The undisturbed forest increased by 8.1% between 1973 and 1987 whereas the disturbed forest area decreased by 28.6% during the same period. Several factors contributed to the increase in the undisturbed forest in Ngangao (1973-1987). Firstly, fast growing succession species such as *Phoenix reclinata* and *Dracaena steudneri* var. *kilimanjaro* rapidly colonized the abandoned open areas especially at the top of Ngangao forest closing up the gaps. Secondly, the establishment of exotic plantations of *C. lusitanica* and *P. patula* in mid 1970s in open areas and particularly in the northern part of the forest (the area bordering Kitumbi primary school); these plantations covered previously bare areas resulting into an increase in undisturbed forest area. Moreover, the population increase in Mgambonyi sub-location between 1969 and 1979 when the forest had no formal management system was low. The population increased by 11.6% (KNBS, 1969; 1979). However, from 1987 to 2016, the undisturbed forest area decreased by 10.2 ha (6.6%) while the disturbed forest area increased by 12.8 ha (47.8%) during the same period. The open ground too increased by 8.8 ha (52.1%) between 2012 and 2016. This means that the gazettement of Ngangao in 2003 has not had a positive impact on the ecological integrity of the forest. Recurrence of forest fires is responsible for the increase in the areas of disturbed and open ground within the forests. According to fire incidences records at KFS Wundanyi office, Ngangao has had three incidences of forest fire that destroyed approximately 30.0 ha of the forest. Besides, in 2011, 5.0 ha of Ngangao were lost to forest fire. In early 2016, forest fire burned 6.5 ha of the forest. Although the estimated areas of the forest lost to fires might not be very accurate, they provide a satisfactory explanation why the disturbed and open ground areas have been increasing lately.

In Vuria, the undisturbed forest area greatly declined by 76.9 ha (57.7%) between 1973 and 2016. Similarly, the disturbed forest area decreased by 44.1% between 1973 and 1987 but then slightly increased by 18.3% between 1987 and 2001. Vuria is a County government managed forest (formerly managed by the County council of Taita Taveta) and therefore the enforcement of forest protection laws has been rather weak due to lack of capacity by the County government resulting in subsequent continuous degradation. Besides, the population of Mwanda

sub-location that borders the forest increased substantially from 9,349 in 1969 to 14,553 in 1979, translating to 55.7% increase over a period of 10 years (KNBS, 1969; 1979). The increase in population was rather rapid and hence responsible for the decline in the undisturbed forest area. Uncontrolled grazing of livestock beyond the carrying capacity is common in the forest. Large numbers of cattle and sheep grazed in the forest fragment all year round. The livestock browses on the vegetation including the regenerates (seedlings and saplings) causing heavy disturbance and hindering the recovery of degraded sites which could have otherwise recovered through natural succession process. At the hill top of Vuria, the forest was cleared for setting up of radio and mobile telephone towers and access road constructed from the edge of the forest to the towers; an action that created perpetual open areas within the forest. Vuria experiences frequent incidences of forest fires. Most of the open and degraded areas within Vuria forest support a dense matrix of *Blotella cursii*, an invasive fern that dries up during the dry season forming massive fuels and hence increasing the fire risk. According to fire records at KFS Wundanyi station, the latest incident of fire occurred in 2009 and it affected approximately 2.5 ha of the forest. The decrease in the undisturbed forest area in Vuria and the accompanying increase in open and disturbed forest areas may be explained by rapid increase in human population, overgrazing, frequent forest fires and ineffectiveness in the management of the forest due to inadequate human and financial capacity of the County government of Taita Taveta.

5.4 Effect of forest edge on tree species diversity

The effect of forest edge on tree species diversity was evident in all the five forest fragments studied. The tree species diversity was higher in the forest interior than in the intermediate forest and at the forest edge in all the five forest fragments irrespective of the status of gazettement and the management system. The present findings are congruent with earlier reports which indicated that tree species diversity is relatively lower at the edge of the forest than in the forest interior (Laurance *et al.*, 2006; Benítez-Malvido *et al.*, 2014; Kacholi, 2014; Dantas de Paula *et al.*, 2016; Their and Wesenberg, 2016). The forest edge may be distinguished from the forest interior by the changes in microclimate, including incoming solar radiation, temperature, wind speed and evapotranspiration (Ranney *et al.*, 1981). Hence, the microclimate at the forest edge differs from that of the forest interior in attributes such as incident light, humidity, ground and air temperature, and wind speed (Ranney *et al.*, 1981). The changes in

microclimate affect biological processes such as litter decomposition and nutrient cycling as well as the forest structure, composition of vegetation and ecological functioning along the forest edge (Bennett and Saunders, 2010). The microclimate changes triggers biotic responses that permit only tree species adapted to forest edge micro-climatic conditions to grow at the edge of the forest (Kupfer and Malanson, 1993; Laurance, 2008). These forest edge adapted species are often species found in the early stages of forest succession (Kupfer and Malanson, 1993). Whereas forest edge adapted tree species are resilient to microclimate changes and increases in abundance, forest edge sensitive species are unable to survive in the newly created conditions and hence decline becoming locally extinct. The extinction of edge sensitive species that could not survive the new micro-climatic conditions created by edge effect is responsible for low species diversity witnessed at the edges of the five forest fragments studied.

5.5 Interaction between environmental variables and the abundance of dominant and adaptable tree species

The abundance of six dominant and adaptable species studied (*M. conglomerata*, *A. gummifera*, *S. guineense*, *X. monospora*, *T. stapfiana* and *M. lanceolata*) was greatly influenced by the distance from the forest edge, electrical conductivity, potassium and soil moisture. In contrast, soil pH, nitrogen, phosphorus and organic carbon did not influence the abundance of the six species in the five forest fragments studied. *M. conglomerata*, *A. gummifera*, *S. guineense*, *X. monospora*, *T. stapfiana* and *M. lanceolata* are secondary succession species commonly found on the forest edges and in open gaps within the forest (Aerts *et al.*, 2011). Since the species are secondary succession species typical of forest edges and gaps, their density and frequency were high at the forest edge but decreased along the gradient from the forest edge to the forest interior. Moreover, the six common and dominant species are shade intolerant and hence are associated with different intensities of forest disturbances (Wekesa *et al.*, 2016). Therefore, they are more common on the forest edges that are usually disturbed than in the forest interior that are in most cases intact.

Soil moisture was another important environmental variable that significantly affected the abundance of *M. conglomerata*, *A. gummifera*, *S. guineense*, *X. monospora*, *T. stapfiana* and *M. lanceolata* in Taita Hills forest fragments. Earlier studies have shown that increased penetration of solar radiation, lowering of soil moisture, dehydration and damaging winds levels amplifies

disturbance on the forest edges altering the ecosystem functioning (Laurance *et al.*, 1998; Laurance *et al.*, 2000). The increased solar radiation and wind causes soil drought, lowering the amount of available water for vegetation (Camargo and Kapos, 1995; Laurance *et al.*, 2000). To survive, the six dominant and adaptable species are well adapted to micro-climatic conditions at the forest edge and particularly the low amount of available water for absorption and use in life supporting processes.

Besides distance from the forest edge and soil moisture, the abundance of the six species was determined by the level of potassium in the soil and electrical conductivity (EC). Similar to soil moisture, level of potassium in the soil was between low and moderate on the forest edges in all the five forest fragments studied. This means that *M. conglomerata*, *A. gummifera*, *S. guineense*, *X. monospora*, *T. stapfiana* and *M. lanceolata* are able to survive in forest areas with low to moderate levels of potassium. This validates the fact that these species can effectively endure the micro-climatic conditions on disturbed forest edges and open habitats such as low moisture and potassium levels as reported in previous studies (Aerts *et al.*, 2011; Wekesa *et al.*, 2016). Therefore, the six species are likely to occur frequently in areas within the forest fragments where the concentration of potassium is low to moderate. On the other hand, the EC on the forest edges was normal for plant growth in all the forest fragments studied and therefore, the key six species successively survive in these soils. Consequently, any changes in EC either positive or negative changes going beyond the normal levels could cause a major shift in the distribution and occurrences of the six species in the forest altering their abundance.

The levels of nitrogen and organic carbon in soil were adequate along the forest edge-interior gradient in the five forest fragments. The levels of nitrogen and organic carbon in the soil ranged from 0.45% to 0.93% and 4.17% to 11.0% respectively. According to Okalebo *et al.* (2002), the levels of nitrogen and organic carbon above 0.3% and 1.5% respectively are adequate for plant growth. Furthermore, the soils were slightly acidic at the forest edge, intermediate forest and forest interior in all the forest fragments studied. The abundance of *M. conglomerata*, *A. gummifera*, *S. guineense*, *X. monospora*, *T. stapfiana* and *M. lanceolata* was symptomatic of the absence of the effect of soil pH, nitrogen and organic carbon as the nutrients were sufficiently available for absorption and use by the tree species. Surprisingly, despite the fact that phosphorus was not adequate along the gradient from the forest edge to the forest interior, the abundance of these species had no relationship with the deficiency along the forest edge-interior

gradient. There is evidence in the literature that phosphorus availability in montane forests can be low and declines with elevation (Tanner *et al.*, 1998; Benner *et al.*, 2010). However, efficient cycling of phosphorus through higher resorption efficiency of phosphorus as the plants tissues senesce makes the nutrient available to the plants in sufficient quantities (Dalling *et al.*, 2016). Seemingly, the six species have an inherent mechanism that enhances efficient cycling of phosphorus thereby increasing the available phosphorus to the trees.

Individually, there was great variation in the abundance of each of the six dominant and adaptable tree species in the five forest fragments studied. The abundance of *A. gummifera* was determined by the levels of salinity in the soil, soil moisture content and organic carbon. *A. gummifera*, a pioneer species (Mullah *et al.*, 2013), is commonly found in disturbed and open habitats on the forest edges (Aerts *et al.*, 2011; Mullah *et al.*, 2013; Wekesa *et al.*, 2016) and this explains why the species was more abundant in forest areas with saline soils with low levels of moisture. Typically, forest edges have low levels of moisture in the soil due to heavy winds and increased penetration of solar radiation (Laurance *et al.*, 1998; Laurance *et al.*, 2000). According to Sakin (2012), there is a significant relationship between organic carbon and nitrogen because of litter decomposition and mineralization through microbial activities. Subsequently, the abundance of *A. gummifera*, a prolific nitrogen fixer, may be due to the influence of organic carbon, symptomatic of the relationship between mineralized carbon and nitrogen as indicated by Sakin (2012). The abundance of *M. conglomerata* was determined primarily by phosphorus and nitrogen levels in the soil. The positive correlation of *M. conglomerata* with phosphorus could be explained by the effect that pH has on the availability of phosphorus in the soil. Thus, *M. conglomerata* was most abundant in forest areas within the forest fragments that had the highest available soil phosphorus and lowest soil pH. Previous studies have shown that in the tropics, phosphorus is the nutrient that limits the growth of forests (Vitousek, 1984) and that its availability is dependent on soil pH levels (Brady, 1984). This result shows the distribution of *M. conglomerata* to be dependent on available phosphorus at low pH levels. In addition, the abundance of *M. conglomerata* may be dependent on nitrogen level in the soil. Omoro *et al.* (2011) has also reported a significant positive relationship between *M. conglomerata* and nitrogen level in the soil. However, the role played by nitrogen in influencing the abundance of *M. conglomerata* could be minor as earlier reported (Omoro *et al.*, 2011).

The abundance of *X. monospora* was dependent on the levels of soil moisture and potassium. Characteristically, *X. monospora* is well adapted to forest areas with low levels of soil moisture and potassium. Hence, the findings of this study conform to results of earlier studies (Omoro *et al.*, 2010; Aerts *et al.*, 2011; Wekesa *et al.*, 2016) which indicated that the species is common in open and disturbed forest sites such as forest gaps and edges where the levels of soil moisture and potassium are low. On the other hand, *T. stapfiana* was dependent on the soil pH and the distance from the forest edge. However, the present results showing that the abundance of *T. stapfiana* is dependent on the soil pH deviate from an earlier report, which indicated that the species abundance was not dependent on any of the soil factors (Omoro *et al.*, 2011). Many authors (Omoro *et al.*, 2011; Aerts *et al.*, 2011; Wekesa *et al.*, 2016) have reported that *T. stapfiana* is a secondary succession species associated with disturbance; typically, forest edges are disturbed and this demonstrates why the abundance of this particular species is dependent on the forest edge-interior gradient.

The abundance of *S. guineense* was marginally dependent on the levels of organic carbon and electrical conductivity in the soil. Typically, *S. guineense* is found in tropical mountain forests (FAO, 2000) and in the Taita Hills, where *S. guineense* is endemic (Chege and Bytebier, 2005; Omoro *et al.*, 2011), its frequencies was relatively higher in all the five forest fragments studied regardless of the levels of soil organic carbon and electrical conductivity. The occurrence of *S. guineense* in all the five forest fragments studied may be due to natural distribution of the species rather than the effects of soil electrical conductivity and organic carbon.

There were no correlations between *M. lanceolata* and soil variables. The findings of this study indicate that the abundance of *M. lanceolata* was not related to any of the soil variables studied. This is not surprising because previous studies have shown that soil parameters are not the only abiotic factors that affect the spatial differentiation of plant communities; instead other factors like geologic and anthropogenic activities influences the spatial abundance of plant species (Gruszczynska *et al.*, 1991; Vitousek *et al.*, 1996; Omoro *et al.*, 2011).

5.6 Interaction of the most abundant species with environmental variables

Unlike the six dominant and adaptable species whose abundance was mainly influenced by the distance from the forest edge, electrical conductivity, potassium and soil moisture, the probability of occurrence of the thirty tree species categorized as most abundant (six dominant

and adaptable species included) was mainly influenced by the distance from forest edge, potassium and soil pH. Besides, electrical conductivity, nitrogen, phosphorus and organic carbon marginally affected the probability of occurrence of the most abundant tree species. Soil moisture did not affect the occurrence of the most abundant trees species. The distance along forest edge-interior gradient influenced the occurrence of 36.7% of the species classified as most abundant species followed by potassium (23.3%), soil pH (20.0%), phosphorus (6.7%) and organic carbon (3.3%). The electrical conductivity and nitrogen collectively accounted for 6.7% of species occurrence for the most abundant tree species. The occurrence of the remaining 3.3% of the most abundant species was not dependent either on any of the soil variables studied or the distance along forest edge-interior gradient. As argued by several authors (Gruszczynska *et al.*, 1991; Vitousek *et al.*, 1996; Omoro *et al.*, 2011), soil parameters are not the only abiotic factors that affect the spatial differentiation of plant communities. Instead, other factors like geologic and anthropogenic activities influence the spatial abundance of plant species (Gruszczynska *et al.*, 1991; Vitousek *et al.*, 1996; Omoro *et al.*, 2011).

The distance along the forest edge-interior profoundly influenced the occurrence of most abundant species and this could be due to edge effect. The physical changes associated with edge effect adversely affect biological processes such as litter decomposition and nutrient cycling as well as the structure and composition of the forest thereby altering the ecological functioning along forest edge-interior gradient (Bennett and Saunders, 2010; Magnago *et al.*, 2015). Moreover, tree species are habitually associated with soils that differ in several physical, chemical and biotic properties (Ayres *et al.*, 2009) and these soil properties vary along the gradient from the edge of the forest to the forest interior (Bergès *et al.*, 2013). Therefore, the large variation in soil properties and ecological functioning along the forest edge-interior gradient explains the great influence of distance from forest edge on species distribution. Additionally, potassium, soil pH and phosphorus had a substantial effect on the distribution of the most abundant species. Evidently, potassium seems to play a key role in the productivity and sustenance of the forest fragments hence its dynamics has ecological implications on the occurrence of the most abundant tree species. The extant findings are in agreement with reports of earlier studies that have shown that there is a correlation between potassium and woody species abundance in tropical forests (Tripler *et al.*, 2006).

The effect of soil pH and phosphorus levels on the occurrence of the most abundant species is typically interrelated. When phosphorus availability decreases, the soil pH increases (Omoro *et al.*, 2011; Cerozi and Fitzsimmons, 2016). The effect of phosphorus on species occurrence may be due to the effect of soil pH on the available phosphorus in the soil. Soils with pH values between 6.0 and 7.5 are ideal for phosphorus availability, while pH values below 5.5 and between 7.5 and 8.5 limits phosphorus availability to plants due to fixation by aluminum, iron and calcium often associated with soil parent materials (McCauley *et al.*, 2009). The soil pH in the five forest fragments along the forest edge-interior gradient was slightly acidic with the pH values ranging from 4.72 to 6.10. This pH range (4.72-6.10) was not ideal for phosphorus availability in the soil. Most tropical forests' soils have pH values ranging between 3.0 and 5.5 (Li *et al.*, 2013). The available phosphorus was therefore not adequate for plant growth in the forest fragments studied. This explains why the interaction between soil pH and phosphorus strongly influenced the distribution of particular tree species. For instance, *T. holstii* and *M. oblata* were well adapted in soils with low level of phosphorus while *D. integra*, *B. antidysenterica*, *G. volkensis*, *P. reclinata* and *P. pycnantha* were not. In the tropical forests, phosphorus is the critical substance that limits the growth of forest plant materials and hence strongly affects tree species distribution (Vitousek, 1984; Baribault *et al.*, 2012; Li *et al.*, 2013; Dalling *et al.*, 2016). Other studies have also revealed that soil pH level that increases phosphorus availability promotes the co-existence of tree species affecting their occurrence within a forest community (Xu *et al.*, 2016).

In forest fragments of Taita Hills, organic carbon, electrical conductivity and nitrogen levels in the soil did not limit tree species growth. This is because organic carbon and nitrogen levels in the soil were adequate in all the forest fragments studied while electrical conductivity was within the normal range required by plants for successive growth. Studies by Omoro *et al.* (2011) revealed that organic carbon and nitrogen play a minor role in species occurrence and distribution in Ngangao, Chawia and Mbololo forest fragments in Taita Hills. However, previous evidence shows that sodium level in the soil is the most important determinant of species occurrence and distribution in the forest fragments of Taita Hills (Omoro *et al.*, 2011). This study did not examine the relationship between sodium and species occurrence to allow for comparison with the results of Omoro *et al.* (2011). Besides, Omoro *et al.* (2011) did not study the

interaction between electrical conductivity and species occurrence to allow for comparison with findings reported in this study.

A small proportion of most abundant species (3.3%) did not reflect the influence of soil factors and distance along forest edge-interior gradient. Therefore, other factors such as geodynamic and anthropogenic activities may have affected the spatial distribution and occurrence of these particular species. Prior studies by Gruszczynska *et al.* (1991) and Vitousek *et al.* (1996) revealed that soil parameters are not the only factors that affect spatial occurrence and distribution of tree species; instead, other factors such as geodynamic and anthropogenic activities do.

5.7 Interaction between less abundant species with environmental variables

Evidence from this study shows that the occurrence of eleven tree species categorized as less abundant was due to the effect of organic carbon, electrical conductivity, soil pH, nitrogen, phosphorus and distance from the forest edge. On the other hand, soil moisture and potassium marginally affected the occurrence of less abundant species in all the five forest fragments studied. The organic carbon, electrical conductivity and distance from the forest edge influenced the occurrence of 36.4% of the less abundant species. Similarly, soil pH greatly affected the occurrence of 36.4% of the less abundant species. Nitrogen and phosphorus levels in the soil had a great effect on the occurrence of 18.2% of the less abundant species while soil moisture and potassium were responsible for the occurrence of 9.1% of the species. Clearly, the occurrence of the less abundant species in the forest fragments studied was not dependent on only one environmental variable but rather on a combination of either two or more variables. This is in contrast with the case of most abundant tree species whose occurrence was mostly dependent on one environmental variable. Therefore, interactions among the environmental variables are seemingly important in maintaining diversity of less abundant tree species in the fragmented forests of Taita Hills.

5.8 Interaction between dominant and less abundant tree species

Evaluation of the relationships among the dominant and adaptable tree species revealed that *S. guineense* and *X. monospora* occurred together in similar environments in the forest fragments studied. Similarly, *A. gummifera* and *M. conglomerata* were found occurring together.

However, *T. stapfiana* and *M. lanceolata* occurrences were not dependent on any of the other key dominant and adaptable species. The occurrence of *S. guineense* and *X. monospora* in similar environments could be because both species are associated with low levels of disturbance (Omoro *et al.*, 2010). The fact that *A. gummifera* and *M. conglomerata* are pioneer species explains the close association between the two species (Omoro *et al.*, 2010). Being pioneer species, *A. gummifera* and *M. conglomerata* enriches the soil with nitrogen and organic matter respectively. The roots of *A. gummifera* contain *Bradyrhizobium* bacteria that fix nitrogen in the soil while *M. conglomerata* increases humus content of the soil through heavy leaf fall particularly during the dry season (Isango, 2001; Tadesse, 2006; Mullah *et al.*, 2013). The occurrence of *T. stapfiana* and *M. lanceolata* was independent of any other key species because the species are native secondary species associated with heavy disturbance (Omoro *et al.*, 2010; Wekesa *et al.*, 2016). Heavy forest disturbance has negative consequences on soil ecology (Cambi *et al.*, 2015). Soil ecology and nutrients determines the occurrence of most tree species in natural forests (John *et al.*, 2007; Cambi *et al.*, 2015).

The occurrence of key dominant and adaptable species greatly influenced the occurrence of less abundant species. The occurrence of *T. stapfiana* and *M. lanceolata* had the greatest influence on the occurrence of the less abundant species. Conversely, the occurrence of *A. gummifera*, *M. conglomerata*, *S. guineense* and *X. monospora* least influenced the occurrence of less abundant species. *T. stapfiana* co-existed with 45.5% of the less abundant species while 18.2% of the less abundant species co-existed with *M. lanceolata*. Only 9.1% of the less abundant tree species co-existed with each of the following key species: *A. gummifera*, *M. conglomerata*, *S. guineense* and *X. monospora*. This was not surprising especially for *A. gummifera* and *M. conglomerata* because the species enrich soil through nitrogen fixation and organic matter addition (Isango, 2001; Tadesse, 2006; Mullah *et al.*, 2013); nitrogen and organic matter are not the nutrients limiting tree growth in tropical Afromontane forests (Dalling *et al.*, 2016). The less abundant species occurring in association with *T. stapfiana* such as *P. africana*, *P. latifolius*, *A. robusta* and *A. abyssinicus* are shade intolerant and are most abundant in open areas along forest edges and disturbed areas. This explains the observed association between the species and *T. stapfiana* because *T. stapfiana* is also abundant on the forest edges and in disturbed forest areas. Similarly, *P. fulva* and *A. theiformis* which were found occurring in close association with *A. gummifera* and *M. conglomerata* are light loving and often abundant on the

forest edges. *A. gummifera* and *M. conglomerata* are also light loving and therefore common on the forest edges. Moreover, *M. lanceolata* and *A. pentagyna* are shade intolerant species and are often abundant on the forest edges and this explains the occurrence of the two species together.

The occurrence of *O. holstii* was associated with *S. guineense* and *X. monospora*. Typically, *S. guineense* and *X. monospora* are associated with low levels of disturbance and are shade tolerant (Lawes *et al.*, 2007; Kebede and Isotalo, 2016). *O. holstii* is also shade tolerant and hence the environmental conditions for successive establishment and growth of *O. holstii* are similar to those of *S. guineense* and *X. monospora*; this explicates the close association among the three species. The occurrence of *Lobelia giberroa* was not associated with any of the six key species. Since *L. giberroa* is shade tolerant species that thrives well under heavy shading, the species was abundant in undisturbed forest areas with dense canopy closure. Furthermore, *L. giberroa* requires consistently moist soils and hence cannot survive on the forest edges where soil moisture is usually low.

5.9 Effect of forest edge on the forest structure and carbon stock

The stand density, DBH and height are key features of the forest structure that determine the quantity of carbon that a given forest can sequester. There was a clear pattern of changes in the stand density along the forest edge-interior gradient in four out of the five forest fragments studied. Thus, the stand density increased towards the forest interior in Chawia, Fururu, Mbololo and Ngangao. On the other hand, the stand density in Vuria lacked a clear pattern and was higher in the intermediate forest than in forest interior and at the edge of the forest. Evidently, the micro-climatic conditions created by the edge effect affected the stand density in Chawia, Fururu, Mbololo and Ngangao with the forest edge having lower stand density and the forest interior having higher stand density. The low stand density that was observed on the forest edge in Chawia, Fururu, Mbololo and Ngangao could be attributed to several factors. The soils at the forest edges have low levels of moisture that hinders seed germination leading to poor regeneration. The low soil moisture also hinders the fast growth of the regenerates into mature trees. This means that only shade intolerant tree species adapted to low soil moisture are able to survive on the forest edges. Moreover, the forest edges are exposed to strong winds, which increase the rate of transpiration in trees further reducing the available moisture for uptake by trees. The strong winds also damages juvenile and mature trees by breaking and uprooting them

respectively. It is also vital to note that due to low soil moisture on the forest edges, nutrient cycling is not efficient and as a result, the essential soil nutrients needed by trees for growth are limited. Because of the reduced moisture in the soil on the forest edges, drought-stressed trees copiously shed leaves (Carvalho and Vasconcelos, 1999) but the litter decomposition is slow due to drier forest edge conditions (Didham, 1998). Accumulation of leaf litter on the forest edges negatively affect seed germination (Bruna, 1999) and seedlings survival (Scariot, 2001) and makes forest edges vulnerable to surface fires during dry seasons (Cochrane *et al.*, 1999). The low soil moisture and inadequate essential soil nutrients along the forest edges combined with the impact of strong winds and forest fires hampers successful establishment and growth of trees on the forest edges resulting into low stand density.

The stand density in Chawia, Fururu, Mbololo and Ngangao was higher in the forest interior and intermediate forest. The reason for this may be due to efficient nutrient cycling in the intermediate forest and the forest interior and hence essential soil nutrients required for trees growth are not limiting. Secondly, due to the dense canopy cover, the soils in the forest interior are able to retain sufficient soil moisture to facilitate seed germination, promote natural regeneration and support vigorous growth of the regenerates into mature trees. Consequently, the availability of adequate moisture and nutrients in the soil to support good growth of vegetation enhances forest growth leading to high stand density. Moreover, the impact of wind is minimal in the intermediate forest and forest interior and therefore juvenile and mature trees are hardly broken and/or uprooted by the strong winds. The situation in Vuria where the intermediate forest had high stand density than the forest interior was unexpected although this may be due to the heavy livestock-grazing disturbance witnessed in forest fragment.

The present results reveal that the stand density is higher in the intermediate forest and forest interior than on the forest edges. Other studies have also shown that tree densities fluctuate more intensely over time near forest edge than in the forest interior (Laurance *et al.*, 2000; Harper *et al.*, 2005; Laurance *et al.*, 2007; Wicklein *et al.*, 2012; Kacholi, 2014; Magnago *et al.*, 2015). Moreover, it has been reported that the forest structure and composition of vegetation along the forest edge are different from the forest interior (Bennett and Saunders, 2010; Kacholi, 2014; Magnago *et al.*, 2015; Reinmann and Hutyrá, 2017), and this has been confirmed in this particular study.

Edge effect along the forest edge-interior gradient in all the forest fragments did not affect DBH and height of trees. The DBH on the forest edges was comparable to the intermediate forest and forest interior in all the five forest fragments. Similarly, the height of trees on the forest edges was comparable to the intermediate forest and forest interior. Consequently, the mean DBH and height was most probably dependent on the individual species growth characteristics rather than the effect of the forest edge. Regarding the mean DBH along the forest edge-interior gradient, the present results are similar to the results of Magnago *et al.* (2015) which showed that there was no difference in mean DBH between the forest edge and the forest interior. However, the present findings on mean height are in sharp contrast with the findings reported by Magnago *et al.* (2015). Whereas Magnago *et al.* (2015) established that the mean tree height decreased towards the forest edge, the present results indicate that the mean height was not different between the forest edges and the forest interior.

Wood specific gravity or wood density is an important attribute of the forest structure because it affects the quantity of carbon sequestered by a given tree. Therefore, wood density is a crucial variable in carbon accounting in tropical forests since it correlates with numerous morphological, mechanical, physiological, and ecological properties of trees and hence it is the best single descriptor of wood (Chave *et al.*, 2006; Otuoma *et al.*, 2016). Edge effect had no influence on wood density in Chawia, Fururu, Ngangao and Vuria along the forest edge-interior gradient. However, in Mbololo, wood density was higher in the intermediate forest and forest interior. Comparatively, there was great variation in the levels of nitrogen, organic carbon, phosphorus and potassium in the soil between the forest edges and the intermediate forest and forest interior in Mbololo. The variation accounted for the differences in wood density between the forest edge and the intermediate forest and forest interior. Evidence in the literature (Cavender-Bares *et al.*, 2004; Chave *et al.*, 2006) suggests that variation in wood density is driven by environmental conditions. This is in conformity with the findings of this study. Although, wood density varied over the plant's environment along the forest edge-interior gradient, this range was in one way or another limited, and more importantly, the mean wood density of the species as reported in the present study was highly conserved phylogenetically. The present findings support the argument advanced by Chave *et al.* (2006) that wood density in natural forests vary considerably along environmental gradients.

The carbon stock in the five forest fragments remained constant along the forest edge-interior gradient. This is probably due to compensatory effects of large trees retained at forest edges (Ziter *et al.*, 2014). Although the forest edges were disturbed and had slightly low stand densities, they had big and tall trees with large diameters such as *P. fulva*, *O. holstii*, *T. stapfiana*, *A. gummifera*, *M. conglomerata* and *S. guineense* among others that compensated for the low densities by sequestering large amounts of carbon. Clearly, stand density, DBH, height and wood density were the main factors that determined carbon stock along the forest edge-interior gradient and not the edge effects. In fact, related studies have shown that stem diameter (DBH), wood specific gravity (wood density) and height coupled with stand density are the most important determinants of above-ground carbon stock in forests (Chave *et al.*, 2005; 2014; Wekesa *et al.*, 2016; Otuoma *et al.*, 2016).

The carbon stock varied among the five forest fragments. Chawia had the highest carbon stock per unit area followed by Fururu, Ngangao and Mbololo. Vuria had the least carbon stock per unit area. These differences in carbon stocks could be attributed to the differences in tree species diversity (Wekesa *et al.*, 2016). Tree species in tropics exhibit different growth rate and patterns and as a result, they vary considerably in terms of size (DBH and height) and specific wood gravity or wood density. These variables (DBH, height and wood density) determine the quantity of carbon a tree can store. Due to differences in diversity of tree species in the five forest fragments, the DBH, height and wood density varied greatly resulting into differences in carbon stocks (Wekesa *et al.*, 2016). The values of carbon stock per unit area in this study of between 180.6 Mg ha⁻¹ and 352.3 Mg ha⁻¹ were lower than 360 Mg ha⁻¹ in Kakamega forest as reported by Glenday (2006) but higher than 58.0-94.0 Mg ha⁻¹ earlier reported for Arabuko Sokoke forest (Glenday, 2008).

5.10 Effect of fragment size and forest edge on species richness

As expected, large forest fragments had higher species richness than smaller forest fragments. Among the gazetted forest fragments, Mbololo that is the largest had the highest species richness followed by Ngangao and Fururu. In the case of non-gazetted forest fragments, Chawia the larger forest fragment had high species richness than Vuria. The results of this study are consistent with the findings of other authors. Laurance *et al.* (2002), found a positive correlation between species richness and fragment size in fragmented forests. Consequently,

small forest patches tend to have low species richness compared to large forest fragments (Laurance *et al.*, 2002; Krauss *et al.*, 2004). Moreover, Benítez-Malvido and Martínez-Ramos (2003) and Benítez-Malvido *et al.* (2014) observed that the species richness of recruited trees, lianas and herbs is lower in fragmented forests than in continuous forests; with a 1.0 ha fragment having the poorest species richness. Correspondingly, Schleuning *et al.* (2011) argues that the relatively large size of the forest fragments (>40.0 ha) mitigate the effects of forest fragmentation on species richness further validating the results of the present study which have shown that large forest fragments (Mbololo, Ngangao and Chawia) have high species richness than small forest fragments (Fururu and Vuria). Also, individual tree species have been reported to respond differently to patch area dynamics, habitat status and soil conditions (Ojoi *et al.*, 2014) and the abundance of dominant species decrease with reduction in patch area (Ojoi *et al.*, 2014). Usually, rare species can easily be lost in small forest fragments (Cagnolo *et al.*, 2006) suggesting that large forest fragments are necessary in preserving species with low local or regional abundance.

With regard to the effect of forest edge on species richness, there was a clear pattern along the forest edge-interior gradient in four out of the five forest fragments studied. In the four forest fragments (Chawia, Fururu, Mbololo and Vuria), the species richness increased towards the forest interior. However, the species richness in Ngangao was high in the intermediate forest but low in the forest interior and on the forest edge. Unexpectedly, the species richness in the forest interior and on the forest edge in Ngangao was similar. The low species richness on the forest edges of Chawia, Fururu, Mbololo and Vuria forest fragments may be due to the reduced soil moisture on the forest edges. The decreased levels of soil moisture on the forest edges triggers water stress in trees occurring on the forest edges and as a result, the trees plentifully shed leaves (Carvalho and Vasconcelos, 1999). Nevertheless, the shed leaves decompose at a very slow rate due to drier forest edge conditions (Didham, 1998; Their and Wesenberg, 2016) leading to heavy accumulation of leaf litter on the forest edges that negatively affect seed germination (Bruna, 1999) and seedlings survival (Scariot, 2001) and makes forest edges vulnerable to surface fires during dry seasons (Cochrane *et al.*, 1999). Consequently, only tree species adapted to drier conditions on the forest edges and are resistant to fire e.g. *X. monospora* are able to survive on the forest edges, this explains the low species richness observed along the forest edges of Chawia, Fururu, Mbololo and Vuria. The present study results are similar to

findings of other authors (Cagnolo *et al.*, 2006; Aguirre-Gutiérrez, 2014) that show that species richness was higher in the interior of the forest than on the forest edges.

Moreover, slope and elevation characteristics of Chawia, Fururu, Mbololo and Vuria influenced the species richness in the forest fragments. As the slope and elevation increased along the forest edge-interior gradient, species richness also increased almost proportionately. The results of this study suggest that slope and elevation characteristics are key determinants of species richness in fragmented forest ecosystems. In a similar study conducted in the mixed coniferous-non coniferous forest ecosystem in the Sierra Madre Occidental, in the northern state of Chihuahua, Mexico, the slope and elevation characteristics of forest fragments greatly influenced species richness of woody plants (Aguirre-Gutiérrez, 2014).

On the other hand, the similarity in species richness in the forest interior and on the forest edge observed in Ngangao could be attributed to the structurally rich farmland surrounding the forest fragment which alleviated the effects of forest edge on species richness as earlier observed by Schleuning *et al.* (2011). The high species richness in the intermediate forest than in the forest interior and on the forest edge may be due to edaphic factors. Occasionally in fragmented forests, soil parameters might gain relative importance and become the decisive driver of species richness (Normann *et al.*, 2016) and this appears to be the case in Ngangao where soil moisture and phosphorus were not limiting in the intermediate forest. Other related studies have shown that in the tropics, forest productivity, tree growth and species richness are functions of soil phosphorus and soil moisture (Gouvenain *et al.*, 2007; Baribault *et al.*, 2012; Marryanna *et al.*, 2012; Dalling *et al.*, 2016).

5.11 Effect of fragment size on tree species diversity

Generally, large forest fragments had high tree species diversity than small forest fragments. Besides, gazetted forest fragments under the management of KFS had high tree species diversity than non-gazetted forest fragments managed by the County government. Of the three gazetted forest fragments, the large fragments of Ngangao and Mbololo were more diverse than Fururu, considered as a small forest fragment. Regarding non-gazetted forest fragments, Chawia exhibited higher tree species diversity than Vuria. Previous studies have shown that the size of the forest fragment is the most important attribute that influences species diversity, stand structure and anthropogenic disturbances (Esseen, 1994; Kupfer and Franklin, 2000; Laurance *et*

al., 2002; Echeverría *et al.*, 2007). In fact, reduction in the forest area leads to considerable loss of floral diversity (Zuidema *et al.*, 1996; Laurance *et al.*, 2002; Cagnolo *et al.*, 2006; Ojoiyi *et al.*, 2014) as confirmed in this study. Moreover, Gascon *et al.* (2000) and Laurance (2008) have argued that species extinctions is more rapid in smaller forest fragments (<100 ha) than in large forest fragments (>100 ha). Consequently, large forest fragments exhibit higher species diversity than small forest fragments (Gascon *et al.*, 2000; Laurance, 2008). The present study finding that large forest fragments have higher species diversity than small forest fragments is consistent with a number of typical results in forest fragmentation studies (Gascon *et al.*, 2000; Laurance *et al.*, 2002; 2008; Ojoiyi *et al.*, 2014).

Moreover, several studies have reported that non-gazetted forests have lower tree species diversity than gazetted forests due to illegal activities such as logging, charcoal production and encroachment of forests for agriculture (McMorrow and Tali, 2001; Linkie *et al.*, 2008; Kimaro and Lulandala, 2013; Baker *et al.*, 2013; Omondi *et al.*, 2016). The low tree species diversity attributed to the weak enforcement of forest protection laws affects the ecological functioning and integrity of the forests (McMorrow and Tali, 2001; Linkie *et al.*, 2008; Kimaro and Lulandala, 2013; Baker *et al.*, 2013; Omondi *et al.*, 2016). Enforcement of forest protection laws for the non-gazetted forest fragments in Taita Hills has been frail due to lack of capacity (both financial and human) by the County government to manage the forests under their jurisdiction. Thus, the results of this study showing that the tree species diversity was low in non-gazetted forest fragments corroborate findings of past studies by other authors (McMorrow and Tali, 2001; Linkie *et al.*, 2008; Kimaro and Lulandala, 2013; Baker *et al.*, 2013; Omondi *et al.*, 2016). Obviously, the gazettelement of currently non-gazetted forest fragments could undoubtedly limit accessibility to the forest fragments and hence safeguarding floral diversity.

5.12 Effect of fragment size on carbon stock in the five forest fragments

The carbon stock per unit area was not dependent on the size of the forest fragment. Even though Chawia and Fururu were smaller than Mbololo and Ngangao, they had higher carbon stock per unit area. On the other hand, Vuria that was also smaller compared to Mbololo and Ngangao had less carbon stock per unit area. Therefore, the carbon stock per unit area in the five forest fragments was dependent on other factors and not the fragment size (Magnago *et al.*, 2016). Structural physiognomies such as the stand density and tree sizes (DBH and height)

primarily determined the carbon stock held by the forest fragments (Aldana and Stevenson, 2016). Besides, illegal harvesting of trees and forest fires substantially impacts on the carbon stocks dynamics in forest ecosystems and this explains the variability in carbon stocks among the forest fragments (Berenguer *et al.*, 2014). Severe forest fires cause tree mortality resulting into a decrease in forest carbon stability (Hurteau and Brooks, 2011). In comparison to Mbololo, Ngangao and Vuria, Chawia and Fururu have had the least number of fire incidences since 2012. Correspondingly, the removal of large trees from the forest fragments through selective illegal logging reduces the capacity of the forest fragments to sequester carbon (Wekesa *et al.*, 2016).

5.13 Interactions among the edaphic factors and the distance from forest edge

In this study, there was a negative correlation between soil pH, electrical conductivity, organic carbon, nitrogen content, phosphorus and distance from the forest edge. The soil pH was acidic in all the forest fragments along the forest edge-interior gradient and ranged from 4.9 to 6.1. Consequently, the increased soil acidity caused a reduction in the levels of nitrogen, phosphorus and organic carbon in the soil. Moreover, increased soil acidity caused a decline in soil electrical conductivity. There was a functional relationship between soil pH level and the distance from the forest edge. Soil acidity increased towards the forest interior. These results concur with findings of previous studies that have indicated that changes in soil pH considerably affect soil microbial activities and the rate of soil carbon and nitrogen cycling hence affecting the levels of nitrogen and organic carbon in the soil (Kemmitt *et al.*, 2006). Other related studies have also shown that large amount of phosphorus is released to the soil when the pH level is low (Barrow, 1984; Turner and Engelbrecht, 2011) and this has been confirmed by the present study. Moreover, phosphorus availability may be due to the mineralization of nitrogen from organic materials such as decaying litter on the forest floor (Cote *et al.*, 2000). Soil organic matter is a storehouse and supplier of nutrients such as nitrogen and phosphorus to the plants (Schulten and Schnitzer, 1998). Tropical forests' soils are very rich in organic matter with high nitrogen content and low carbon: nitrogen (C: N) ratio that mineralizes sufficient nitrogen to support plant growth hence increasing the available phosphorus (Masunga *et al.*, 2016).

The soil pH showed positive relationship with soil moisture although the relationship was not very strong. This was expected because soil pH varies with dry and wet seasons (Zhang and Wienhold, 2002; Sonko *et al.*, 2016). Soil pH values of 5.5-6.5 and 4.4-4.5 have been reported

for wet and dry seasons respectively (Sonko *et al.*, 2016). Low soil pH (≤ 4.5) which is a highly acidic condition tightly holds soil nutrients making them unavailable for plant uptake (Sonko *et al.*, 2016) while soil pH ranging between 5.5 and 6.5 which is slightly acidic makes most soil nutrients available for plant uptake (Ronen, 2007). Therefore, soil nutrients are more readily available for plant growth during the wet season than during the dry season because there is more soil moisture available in the wet season that facilitates soil nutrient release (Sonko *et al.*, 2016).

The electrical conductivity (EC) positively correlated with organic carbon, phosphorus, nitrogen content and distance from the forest edge. EC increased with increase in organic carbon and nitrogen levels in the soil. Previous studies have shown that soil EC is positively correlated with nitrogen and organic carbon concentration because of the concentration of ions in the soil, type and amount of clay and the soil bulk density (Zhang and Wienhold, 2002; Martínez *et al.*, 2009). Similarly, EC increased with increase in phosphorous concentration in the soil. This is probably because the soils are poorly aerated. A positive relationship between EC and phosphorous concentration is characteristic of poorly aerated soils (Kim *et al.*, 2007). This undoubtedly explains the positive correlation witnessed between EC and phosphorus. The distance from the forest edge also showed positive relationship with EC. The EC reduced along the forest edge-interior gradient and was high on the forest edges and low in the forest interior. Anthropogenic activities particularly crop farming near the forest edges using inorganic fertilizers could be responsible for the high EC witnessed on the forest edges. Moreover, slash and burn agricultural practice that extends to the forest edges may be another reason for the high levels of EC witnessed on the forest edges. On the other hand, EC negatively correlated with potassium and soil moisture. High EC values corresponded with low values of potassium and soil moisture content. The decrease in the potassium levels as the EC increases could be explained by the low pH values of the soil that ranged from 4.7 to 6.1 in all the five forest fragments studied. Acidic soils contain many aluminium ions that hinder potassium availability (Ranade-Malvi, 2011). The negative relationship between EC and soil moisture is not in agreement with previous report by Ekwue and Bartholomew (2010) which have shown that there is strong positive correlation between soil moisture and EC. The negative relationship between EC and soil moisture is not surprising because previous studies have shown that soil moisture is not the only factor that affect EC level in the soil; instead, other factors like soil texture, organic matter and vegetation cover do (Lück *et al.*, 2009; Costa *et al.*, 2014). In the present study

scenario, there was positive correlation between EC and organic carbon. Usually, when the EC is positively correlated with organic carbon, the correlation between EC and soil moisture is negative (Lück *et al.*, 2009; Costa *et al.*, 2014). Besides, soil moisture and EC show negative correlation when there is high moisture content in the soil, but the variability in the field is low (Costa *et al.*, 2014). Soils in Taita Hills forest fragments have high moisture content, but the variability among the forest fragments is very low (Omoró *et al.*, 2011).

There was a positive relationship between soil organic carbon, nitrogen content, phosphorus and the distance from the forest edge. The higher values of nitrogen and phosphorus corresponded with higher values of organic carbon. This may be explained by the successive decomposition of dead plant material in the forest fragments resulting into formation of humus. As the humus slowly mineralizes, it improves the soil structure, increases the rate of cations exchange capacity (CEC) and enhances nitrogen and phosphorus levels in the soil (Turner and Engelbrecht, 2011; Adhikari and Bhattacharyya, 2015; Cai *et al.*, 2016). Soil organic carbon positively correlated with the distance from the forest edge. Accordingly, the level of organic carbon in the soil increased from the forest edge to the forest interior. Typically, forest litter increases towards the forest interior. The rate of forest litter decomposition is higher in the forest interior than on the forest edges due to the high moisture levels in the forest interior which enhance soil microbial activities (Laurance, 2008) and this explains the positive relationship between soil organic carbon and the distance from the forest edge along the forest edge-interior gradient. However, both potassium and soil moisture exhibited negative relationship with organic carbon which was not significant. This was expected in Taita Hills' forest fragments because the soils are not disturbed (virgin). Thus, there is a health relationship between potassium and soil moisture with organic carbon hence making their availability adequate to support plant growth (Pandey *et al.*, 2007; Zornoza *et al.*, 2007).

The level of nitrogen in the soil positively correlated with phosphorus and the distance from the forest edge. Thus, when the values of nitrogen increased, the values of phosphorus also increased. The strong positive relationship between nitrogen and phosphorus may be due to the decomposition of dead plant material that results into formation of humus. As the humus gradually putrefies, it contributes simultaneously to the augmentation of nitrogen and phosphorus in the soil (Adhikari and Bhattacharyya, 2015; Cai *et al.*, 2016). The positive correlation between nitrogen and the distance from the forest edge to the interior may be due to the variation in the

micro-climatic conditions along the forest edge-interior gradient (Kupfer and Malanson, 1993; Laurance, 2008). Moreover, there was a positive relationship between soil moisture and nitrogen content. Earlier studies have shown that high soil moisture level has a positive effect on net nitrogen mineralization while water shortage has negative effect (Gutiñas *et al.*, 2012; Li *et al.*, 2014). Water shortage increases the osmotic pressure, which in turn increases the energy requirements of the microorganisms for osmoregulation, negatively affecting net nitrogen mineralization (Gutiñas *et al.*, 2012; Li *et al.*, 2014). In contrast, nitrogen content exhibited negative relationship with potassium. The level of nitrogen content in the soil was high while that of potassium was low. The Taita Hills' forest fragments soils are acidic and rich in kaolinite and hence have a high concentration of aluminium ions that fixes the available potassium (Omoro *et al.*, 2011; Maeda, 2012). Ranade-Malvi (2011) previously reported that soils containing kaolinite have low potassium levels.

Phosphorus exhibited positive relationships with both potassium and moisture content although the relationships were not significant. Soil properties including, pH, potassium and moisture exerts a strong control on the amounts and forms of soil organic phosphorus in tropical forests (Kozak *et al.*, 2005; Turner and Engelbrecht, 2011) creating a balance between potassium, soil moisture and phosphorus (Kozak *et al.*, 2005; Turner and Engelbrecht, 2011). The balance created makes phosphorus, potassium and soil moisture available in sufficient quantities for uptake by plants. Even though the positive relationship between potassium and soil moisture was not significant, the concentration of potassium in the soil increased with increase in soil moisture creating a balance between these two edaphic factors. Soil moisture affects potassium availability by affecting its mobility in the soil (Kuchenbuch *et al.*, 1986).

The relationship between phosphorus and distance from forest edge though positive, was not significant. On the other hand, soil moisture had a negative relationship with the distance from the forest edge but the relationship was not significant. The weak relationship between phosphorus and soil moisture and the distance from forest edge may be due to the variation in species diversity and the rate of organic matter mineralization along forest edge-interior gradient. Different species have varying leaf shedding capabilities that in turn affect the level of organic matter added in the soil. The rate of mineralization of the organic matter also varies with tree species along the forest edge-interior gradient. The forest litter, organic matter and the rate of mineralization of organic matter are key determinants of phosphorus and moisture levels in the

soils of natural forests (Cai *et al.*, 2016). There was a significant negative relationship between potassium and increasing distance from the forest edge into the forest interior. The negative relationship between potassium and the distance from forest edge may have been caused by the interaction between soil moisture and soil pH. Other studies have reported that soil moisture and pH are the key factors that determine the availability of potassium in the soil (Kuchenbuch *et al.*, 1986; Ranade-Malvi, 2011; Omoro *et al.*, 2011; Maeda, 2012).

5.14 Effects of forest edge edaphic factors

The edaphic factors varied along the forest edge-interior gradient of Taita Hills forest fragments. In Chawia, Fururu, Ngangao and Mbololo, the soil moisture was higher in the intermediate forest and forest interior than on the forest edges although the pattern was not linear. Previous research shows that increased solar radiation and damaging winds on the forest edges may cause soil drought hence lowering the amount of water available for the vegetation (Camargo and Kapos, 1995; Laurance *et al.*, 1998; Laurance *et al.*, 2000; Baimas-George, 2012). However, the soil moisture was higher on the forest edge than in the intermediate forest and forest interior in Vuria. In fact, soil moisture decreased towards the forest interior in Vuria. Compared to other forest fragments, Vuria has very steep gradient and the gradient increases sharply towards the forest interior. Water infiltration decreases with increase in the slope steepness and this may explain the low soil moisture levels in the intermediate forest and forest interior of Vuria (Huat *et al.*, 2006).

The soil pH along the forest edge-interior gradient for all the five forest fragments ranged from moderately acidic to strongly acidic. In Fururu, Mbololo and Ngangao, the soil pH was moderately acidic on the forest edge but strongly acidic in the forest interior. This was similar to reports of previous studies that have shown that soil pH declines from forest edge to the interior due to high species diversity in the forest interior whose litter is moist and richer in nutrients than the dry litter on the forest edges (Aerts *et al.*, 2006; Wuyts *et al.*, 2013). In Chawia and Vuria, the soil pH was moderately acidic on the forest edge, intermediate forest and forest interior. Chawia and Vuria have been heavily disturbed (Pellikka *et al.*, 2009) and hence the ecological conditions on the forest edges and in the intermediate forest and forest interior may not differ much; this explains the lack of variation of soil pH along the forest edge-interior gradient. The electrical conductivity (EC) was similar along the forest edge-interior gradient in Chawia,

Mbololo, Ngangao and Vuria. However, in Fururu, the EC at the edge of the forest was higher than in the forest interior and intermediate forest and this could be attributed to the anthropogenic activities particularly crop farming in the farms adjacent to the forest using inorganic fertilizers. Normally, the EC is dependent on the cation exchange capacity and salinity level in the soil and hence application of inorganic fertilizers increases electrical conductivity and pH of the soil (Grisso *et al.*, 2009).

The level of nitrogen in the soil was high (>0.3%) in all the five forest fragments along the forest edge-interior gradient. Nitrogen content in the soil varied along the forest edge-interior gradient in the five forest fragments. In Chawia and Mbololo, nitrogen increased towards the forest interior while in Fururu and Vuria, it decreased towards the forest interior. In Ngangao, nitrogen was higher in the forest interior and lower in the intermediate forest. The scenario in Chawia, Mbololo and Ngangao where nitrogen content was higher in the forest interior than on the forest edge is consistent with previous studies (Aerts and Chapin, 2000; Toledo-Aceves and García-Oliva, 2007). On the other hand, the decrease in nitrogen content towards the forest interior in Fururu and Vuria is because of two reasons. Firstly, the soils become shallower towards the interior due to the presence of numerous rocks in the interior of Fururu and Vuria as the elevation increases. The shallow soils encumber rapid mineralization of organic matter leading to low nitrogen content in the forest interior. Secondly, Fururu and Vuria occur within the most intensively cultivated farmlands. The farmlands are regularly cultivated for food crops using inorganic fertilizers. Consequently, the forest edges that border fertilized agricultural farmlands receive nitrogen from fertilizer inputs (Pocewicz and Penelope, 2007).

The level of organic carbon in the soil was high (>1.5%) in the five forest fragments along the forest edge-interior gradient. In Mbololo and Ngangao, organic carbon increased towards forest interior. Other related studies have shown that forest edge soils have significantly lower percentage carbon than forest interior soils (Toledo-Aceves and García-Oliva, 2007). On the other hand, organic carbon was higher on the forest edges in Chawia, Fururu and Vuria than in the intermediate forest and forest interior. This is in contrast with earlier report by Toledo-Aceves and García-Oliva (2007) that showed forest edge soils have lower concentration of phosphorus compared to the forest interior. Seemingly, the size of the forest fragment influenced the organic carbon dynamics along the forest edge-interior gradient. The small fragments of Chawia, Fururu and Vuria have similar trends for organic carbon levels in the soil along the

forest edge-interior gradient. On the other hand, Mbololo and Ngangao, the large forest fragments have a similar trend for organic carbon along the forest edge-interior gradient.

The level of phosphorus along the forest edge-interior gradient was higher in the forest interior than on the forest edges in Chawia, Mbololo, Ngangao and Vuria. Unexpectedly, in Fururu, phosphorus level was the same on the forest edge, intermediate forest and forest interior clearly indicating that the rocks occurring in the forest fragment were of the same type. Rocks are primary source of phosphorus (Cerozi and Fitzsimmons, 2016). The level of phosphorus in the soil in Chawia, Fururu, Mbololo and Ngangao along the forest edge-interior gradient was less than 20 ppm and hence deficient for plant growth. However, in Vuria, the level of phosphorus in the soil along the forest edge-interior gradient was sufficient for plant growth and ranged between 18 and 23 ppm. Sufficient levels of phosphorus in the soils range between 20 and 30 ppm (Okalebo *et al.*, 2002). This variation in the levels of phosphorus among the forest fragments demonstrates that the fragments have different types of rocks. On the other hand, the variation in phosphorus along the forest edge-interior gradient is probably due to differences in soil pH (Omororo *et al.*, 2011; Cerozi and Fitzsimmons, 2016).

Potassium level in the soil was high on the forest edge in Chawia, Fururu, Mbololo and Ngangao while in Vuria, it was high in the forest interior. In Chawia, potassium was adequate on the forest edge (120.87 ppm) but deficient in the intermediate forest (89.74 ppm) and the forest interior (88.14 ppm) while in Fururu and Ngangao, it was deficient (less than 117.30 ppm) on the forest edge, in the intermediate forest and the forest interior. In Mbololo, potassium was adequate on the forest edge (222.20 ppm) and in the intermediate forest (143.10 ppm) and deficient in the forest interior (112.60 ppm). The level of potassium in the soil is adequate when the concentration is greater than 117.30 ppm (Okalebo *et al.*, 2002). The variation in the levels of potassium among the forest fragments as well as along the forest edge-interior gradient is probably due to differences in soil pH and soil moisture content (Ranade-Malvi, 2011; Maeda, 2012).

Among all the forest fragments studied, Vuria was the only forest fragment that had sufficient levels of phosphorus (≥ 20 ppm) along the forest edge-interior gradient (forest edge, intermediate forest and forest interior). Therefore, phosphorus and potassium were the nutrients that limited the growth of trees and other plants in the Chawia, Fururu, Mbololo and Ngangao. The findings of this study are congruent to earlier reports that showed that phosphorus and

potassium are the critical nutrients limiting growth of trees and other plants in tropical forests (Vitousek, 1984; Wright *et al.*, 2011; Omoro *et al.*, 2011). The limitation for plant growth in tropical forests by potassium is strongest among trees in the seedlings, saplings and poles stages (Wright *et al.*, 2011).

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

The five main forest fragments in Taita Hills lost a combined total area of 232.9 ha (23.2%) of their cover between 1973 and 2016. The loss represents annual deforestation rate of 0.5%. Further loss could be minimized by radical shifts in policy paradigms that minimize deforestation and focuses on plausible forest conservation strategies.

Stand density, tree species diversity and richness were higher in the forest interior than in the intermediate forest and at the forest edge in all the forest fragments. Forest fragments should be protected from further encroachment and other human related disturbances to save the remaining biodiversity and enhance their ecological integrity.

The above-ground carbon stock was not influenced by the ‘edge effect’ and fragment size but rather by the stand density, wood density and tree sizes (DBH and height). Maintenance of a large population of tall and big trees within the forest fragments would ensure that the capacity of the fragmented forests in Taita Hills to sequester carbon does not decline.

Large forest fragments had higher tree species richness and diversity than small forest fragments. Conservation efforts should ensure that large forest fragments, which maintain a higher proportion of biodiversity, are not subjected to further fragmentation. Forest management interventions should favour increase in the connectivity between large and small forest fragments in order to maintain high species richness and diversity.

The abundance of six dominant and adaptable tree species studied (*M. conglomerata*, *A. gummifera*, *S. guineense*, *X. monospora*, *T. stapfiana* and *M. lanceolata*) was positively influenced by the distance from the forest edge, soil moisture, nitrogen, electrical conductivity and potassium. Anthropogenic impacts that influence environmental variables increase the risk associated with biodiversity erosion and loss. The environmental variables-species relationships established in this study could be utilized in selecting native tree species for rehabilitation programmes to restore the degraded sites within the forest fragments.

6.2 Recommendations

1. Chawia and Vuria should be gazetted and put under the management of Kenya Forest Service. The gazettelement will give the forest fragments legal protection, halt further

- encroachment and stabilize the forest cover. This will stop further forest loss. However, the gazettement should go hand in hand with empowering of local communities on biodiversity conservation and watershed management through Community Forest Associations (CFAs).
2. Agro-forestry belts should be established in farmlands belonging to people living adjacent to the forest fragments to ameliorate the adversarial micro-climatic conditions created on the forest edges by the edge effect, provide alternative source of wood products and protect the forest fragments from further encroachment. The amelioration of micro-climatic conditions on the forest edges can create favourable conditions for secondary forest growth and hence enhancing resilience of the forest fragments to the edge effects and conserving the remaining biodiversity.
 3. There is also urgent need to enhance connectivity of the main forest fragments to conserve the remaining biodiversity. This could be achieved by integrating a number of indigenous tree species in farming systems particularly nitrogen fixing species such as *A. gummifera* and *M. conglomerata* among others. Integration of indigenous tree species in farmlands will promote seed dispersal between the forest fragments maintaining high genetic and species diversity across the Taita Hills landscape.
 4. Degraded areas within the forest fragments including the forest edges should be rehabilitated using endemic native species to restore their ecological and economic functionality, and enhance biodiversity conservation.

6.3 Areas for further research

1. Understanding of the seed dispersal mechanisms within and between forest fragments and ascertain their role in maintaining floral diversity at both ecosystem and landscape levels.
2. Long term monitoring of effects of forest fragmentation on biodiversity and ecosystem processes to ascertain the impacts and implications on the management and conservation of fragmented forests in Taita Hills.
3. Assessing the effectiveness of Community Forest Associations (CFAs) in promoting conservation and sustainable management of the forest fragments.

4. Undertake further assessment of forest cover change using remote sensing technologies that are sensitive to cloud cover e.g. Synthetic Aperture Radar Technologies. This is because Taita Hill forests are Afromontane moist forests that are covered with clouds throughout the year.

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APPENDICES

Appendix I

List of publications

1. **Wekesa, C., Leley, N., Maranga, E., Kirui, B., Muturi, G., Mbuvi, M., and Chikamai, B. 2016.** Effects of forest disturbance on vegetation structure and above-ground carbon in three isolated forest patches of Taita Hills. *Open Journal of Forestry* 6:142-161.
2. **Wekesa, C., Maranga, E., Kirui, B., Muturi, G.M., and Gathara, M.W. 2018.** Interactions between native tree species and environmental variables along forest edge-interior gradient in fragmented forest patches of Taita Hills, Kenya. *Forest Ecology and Management* 409:789-798.

Appendix II

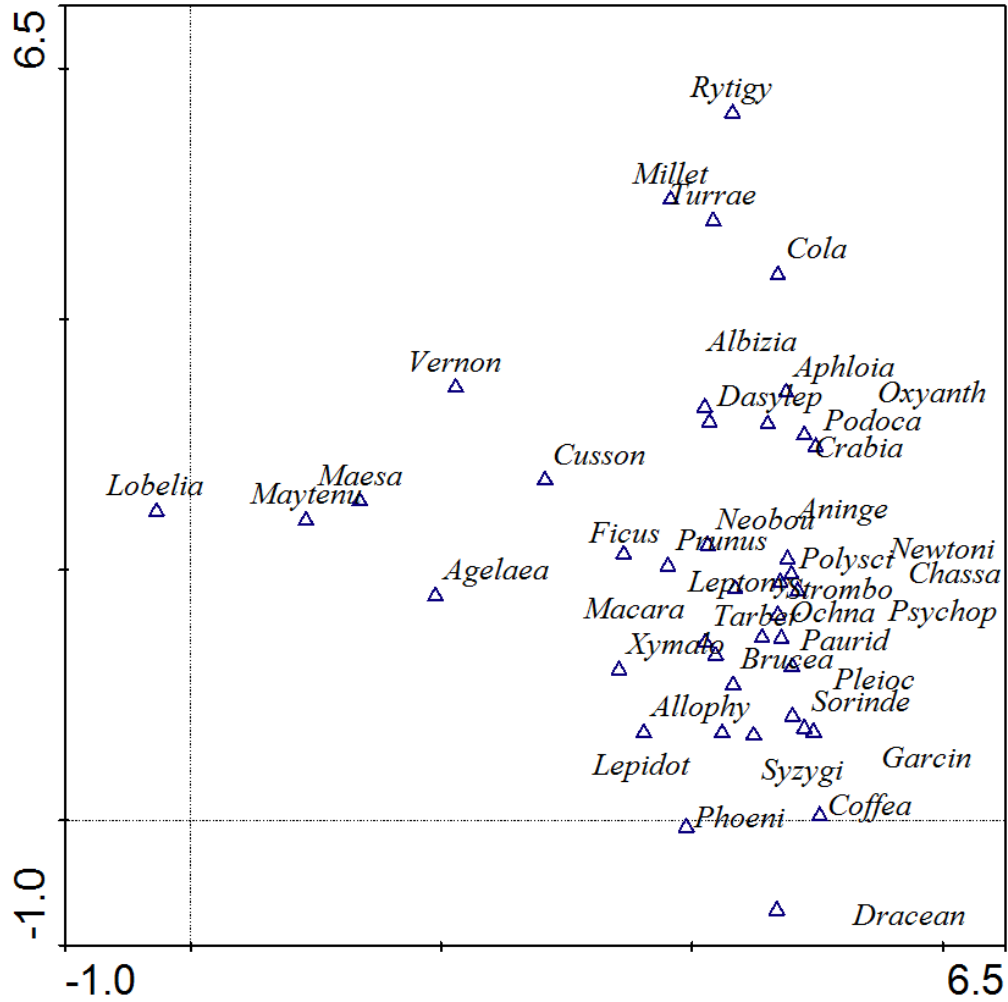
Detrended Correspondence Analysis for the length of the gradient

**** Summary ****

Axes	1	2	3	4	Total inertia
Eigenvalues	: 0.849	0.566	0.331	0.243	8.182
Lengths of gradient	: 4.872	4.884	3.131	2.715	
Cumulative percentage variance of species data	: 10.4	17.3	21.3	24.3	
Sum of all eigenvalues					8.182

Appendix III

Detrended Correspondence Analysis (DCA) for species association in the five forest fragments of Taita Hills



Appendix IV

Spearman's rank correlation analysis for edaphic factors

			Correlations							
			pH	EC (mS/cm)	C (%)	N (%)	P (ppm)	K (ppm)	Moisture (%)	Distance from forest edge (m)
Spearman's rho	pH	Correlation Coefficient	1.000	-.592**	-.573* *	-.487**	-.428**	.368**	.043	-.374**
		Sig. (2-tailed)	.	.000	.000	.000	.000	.001	.723	.001
		N	72	72	72	72	72	72	72	72
	EC (mScm)	Correlation Coefficient	-.592**	1.000	.698* *	.696**	.352**	-.177	-.034	.260*
		Sig. (2-tailed)	.000	.	.000	.000	.002	.137	.779	.027
		N	72	72	72	72	72	72	72	72
	Carbon	Correlation Coefficient	-.573**	.698**	1.000	.693**	.431**	.047	-.209	.421**
		Sig. (2-tailed)	.000	.000	.	.000	.000	.698	.078	.000
		N	72	72	72	72	72	72	72	72
	N	Correlation Coefficient	-.487**	.696**	.693* *	1.000	.404**	.077	.099	.244*
		Sig. (2-tailed)	.000	.000	.000	.	.000	.518	.410	.039
		N	72	72	72	72	72	72	72	72
	P (ppm)	Correlation Coefficient	-.428**	.352**	.431* *	.404**	1.000	.109	.028	.062
		Sig. (2-tailed)	.000	.002	.000	.000	.	.361	.817	.603
		N	72	72	72	72	72	72	72	72

	K (ppm)	Correlation	.368 ^{**}	-.177	-.047	-	.109	1.00	.228	-.282 [*]
		Coefficient				.077		0		
		Sig. (2-tailed)	.001	.137	.698	.518	.361	.	.054	.016
	N	72	72	72	72	72	72	72	72	72
	Moisture	Correlation	.043	-.034	-.209	.099	.028	.228	1.000	-.184
		Coefficient								
		Sig. (2-tailed)	.723	.779	.078	.410	.817	.054	.	.123
	N	72	72	72	72	72	72	72	72	72
	Distance from forest edge	Correlation	-.374 ^{**}	.260 [*]	.421 [*]	.244 [*]	.062	-	-.184	1.000
Coefficient				*	*		.282 [*]			
Sig. (2-tailed)		.001	.027	.000	.039	.603	.016	.123	.	
N	72	72	72	72	72	72	72	72	72	

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Appendix V

Selected photographs taken in Taita Hills during field work



Taita Hills landscape



Human settlements on the edge of the forest



Debarking of *Ochna holstii* for medicinal use



Soil sampling in the field



Soil moisture measurement in the field



Livestock grazing in Vuria forest

Appendix VI

GPS coordinates for the transects and plots

Forest fragment	Transect Number	Plot Number	Northings (S)	Eastings (E)	Elevation (m)	
Vuria	1		03.41615	038.28688	1949	
		1	03.41615	038.28693	1950	
		2	03.41624	038.28706	1951	
		3	03.41626	038.28724	1952	
	2			03.41829	038.29596	1870
		1	03.41821	038.29586	1880	
		2	03.41812	038.29585	1882	
		3	03.41808	038.29589	1902	
	3			03.40931	038.29876	1932
		1	03.40925	038.29874	1931	
		2	03.40915	038.29886	1934	
		3	03.40898	038.29893	1936	
	Fururu	1		03.43193	038.33847	1697
			1	03.43201	038.33850	1699
			2	03.43229	038.33852	1701
		3	03.43241	038.33842	1738	
2				03.43209	038.33936	1686
		1	03.43239	038.33924	1690	
		2	03.43242	038.33920	1698	
		3	03.43264	038.33896	1704	
3				03.43351	038.33969	1707
		1	03.43350	038.33962	1711	
		2	03.43332	038.33920	1780	
		3	03.43336	038.33926	1766	
Chawia	1		03.47627	038.33934	1552	
		1	03.47634	038.33941	1556	

		2	03.47667	038.33968	1570
		3	03.47691	038.34005	1586
	2		03.47855	038.33890	1537
		1	03.47856	038.33895	1534
		2	03.47876	038.33925	1655
		3	03.48045	038.34011	1395
	3		03.48460	038.34252	1575
		1	03.48452	038.34258	1571
		2	03.48442	038.34290	1593
		3	03.48429	038.34324	1577
	4		03.47881	038.34262	1593
		1	03.47882	038.34249	1609
		2	03.47882	038.34213	1600
		3	03.47855	038.34168	1593
Ngangao	1		03.37164	038.34455	1749
		1	03.37142	038.34455	1753
		2	03.37079	038.34449	1758
		3	03.37060	038.34436	1802
	2		03.35690	038.33697	1414
		1	03.36180	038.34480	1780
		2	03.36172	038.34443	1814
		3	03.36161	038.34398	1816
	3		03.35429	038.34392	1723
		1	03.35440	038.34385	1726
		2	03.35460	038.34368	1802
		3	03.35499	038.34345	1820
	4		03.35566	038.33741	1789
		1	03.35555	038.33750	1812
		2	03.35542	038.33788	1832
		3	03.35538	038.33832	1854

	5		03.37097	038.34035	1810
		1	03.37098	038.34049	1826
		2	03.37127	038.34082	1825
		3	03.37142	038.34194	1809
Mbololo Forest	1		03.33508	038.44658	1558
		1	03.33511	038.44640	1610
		2	03.33509	038.44695	1606
		3	03.33464	038.44721	1588
	2		03.33395	038.44653	1574
		1	03.33397	038.44672	1556
		2	03.33403	038.44712	1579
		3	03.33390	038.44744	1662
	3		03.32955	038.44769	1629
		1	03.32953	038.44797	1594
		2	03.32959	038.44856	1645
		3	03.32934	038.44907	1696
	4		03.32877	038.44800	1584
		1	03.32861	038.44797	1591
		2	03.32885	038.44834	1629
		3	03.32884	038.44903	1656
	5		03.33054	038.44764	1612
		1	03.33056	038.44763	1606
		2	03.33079	038.44816	1654
		3	03.33115	038.44830	1676
	6		03.32602	038.44678	1550
		1	03.32597	038.44674	1578
		2	03.32655	038.44738	1594
		3	03.32679	038.44753	1620
	7		03.32476	038.44652	1547
		1	03.32480	038.44646	1560

	2	03.32505	038.44698	1597
	3	03.32504	038.44758	1592
8		03.32404	038.44723	1601
	1	03.32404	038.44732	1589
	2	03.32440	038.44775	1627
	3	03.32448	038.44836	1639
9		03.31118	038.45862	1749
	1	03.31116	038.45851	1734
	2	03.31194	038.45829	1705
	3	03.31236	038.45866	1753

Appendix VII
Trees species of Taita Hills forest fragments

Chawia

1. *Tabernaemontana stapfiana*
 2. *Dracaena steudneri*
 3. *Strombosia scheffleri*
 4. *Garcinia volkensii*
 5. *Pleiocarpa pycnantha*
 6. *Syzygium guineense*
 7. *Albizia gummifera*
 8. *Oxyanthus speciosus*
 9. *Allophylus abyssinicus*
 10. *Leptonychia usambarensis*
 11. *Piper capense*
 12. *Xymalos monospora*
 13. *Maesopsis eminii*
 14. *Cussonia spicata*
 15. *Ficus sur*
 16. *Macaranga conglomerata*
 17. *Maesa lanceolata*
 18. *Millettia oblata*
 19. *Phoenix reclinata*
 20. *Polyscias fulva*
 21. *Turraea holstii*
-

Fururu

1. *Phoenix reclinata*
2. *Tabernaemontana stapfiana*
3. *Xymalos monospora*
4. *Brucea antidysenterica*
5. *Syzygium guineense*
6. *Cussonia spicata*

7. *Maesa lanceolata*
8. *Psychotria crassipetala*
9. *Albizia gummifera*
10. *Allophylus abyssinicus*
11. *Coffea fadenii*
12. *Dracaena steudneri*
13. *Oxyanthus speciosus*
14. *Persea americana*
15. *Prunus africana*
16. *Turraea holstii*

Mbololo

1. *Crabia zimmermannii*
2. *Strombosia scheffleri*
3. *Newtonia buchananii*
4. *Syzygium guineense*
5. *Tabernaemontana stapfiana*
6. *Garcinia volkensii*
7. *Pauridiantha paucinervis*
8. *Xymalos monospora*
9. *Cussonia spicata*
10. *Albizia gummifera*
11. *Psychotria petiti*
12. *Phoenix reclinata*
13. *Coffea fadenii*
14. *Macaranga conglomerata*
15. *Neoboutonia macrocalyx*
16. *Cola greenwayi*
17. *Lepidotrichilia volkensii*
18. *Maesa lanceolata*
19. *Pleiocarpa pycnantha*
20. *Ficus sur*

21. *Sorindea madagascarensis*
22. *Chassalia discolor*
23. *Agelaea pentagyna*
24. *Aningeria robusta*
25. *Ochna holstii*
26. *Vernonia auriculifera*
27. *Aphloia theiformis*
28. *Dracaena steudneri*
29. *Leptonychia usambarensis*
30. *Oxyanthus speciosus*
31. *Polyscias fulva*
32. *Teclea trichocarpa*
33. *Turraea holstii*
34. *Dasylepis integra*
35. *Podocarpus latifolius*
36. *Prunus africana*
37. *Allophylus abyssinicus*
38. *Bersama abyssinica*
39. *Brucea antidysenterica*
40. *Lemon spp*
41. *Memecylon teitense*
42. *Millettia oblata*
43. *Ocotea usambarensis*
44. *Rapanea melanophloeos*
45. *Rauwolfia mannii*
46. *Strychnos mitis*

Ngangao

1. *Oxyanthus speciosus*
2. *Cola greenwayi*
3. *Crabia zimmermannii*
4. *Albizia gummifera*

5. *Psychotria petiti*
6. *Dasylepis integra*
7. *Newtonia buchananii*
8. *Turraea holstii*
9. *Millettia oblata*
10. *Rytigynia uhligii*
11. *Tabernaemontana stapfiana*
12. *Cussonia spicata*
13. *Phoenix reclinata*
14. *Macaranga conglomerata*
15. *Leptonychia usambarensis*
16. *Strombosia scheffleri*
17. *Chassalia discolor*
18. *Maesa lanceolata*
19. *Pauridiantha paucinervis*
20. *Croton macrostachyus*
21. *Eucalyptus saligna*
22. *Pleiocarpa pycnantha*
23. *Podocarpus latifolius*
24. *Xymalos monospora*
25. *Allophylus abyssinicus*
26. *Aningeria robusta*
27. *Aphloia theiformis*
28. *Brucea antidysenterica*
29. *Croton megalocarpus*
30. *Cupressus lusitanica*
31. *Ekebergia capensis*
32. *Garcinia volkensii*
33. *Lepidotrichilia volkensii*
34. *Ochna holstii*
35. *Polyscias fulva*

36. *Prunus africana*
 37. *Psychotria crassipetala*
 38. *Rauvolfia mannii*
 39. *Syzygium guineense*
-

Vuria

1. *Maytenus senegalensis*
 2. *Maesa lanceolata*
 3. *Lobelia giberroa*
 4. *Millettia oblata*
 5. *Xymalos monospora*
 6. *Albizia gummifera*
 7. *Cussonia spicata*
 8. *Vernoniia ariculifera*
 9. *Prunus africana*
 10. *Solanum aculeastrum*
-

Appendix VIII
Research permit



**NATIONAL COMMISSION FOR SCIENCE,
TECHNOLOGY AND INNOVATION**

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When replying please quote

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NAIROBI-KENYA

Ref. No. **NACOSTI/P/18/65399/24891**

Date: **11th September, 2018**

Chemuku Wekesa
Egerton University
P.O. Box 536-20115
NJORO

RE: RESEARCH AUTHORIZATION

Following your application for authority to carry out research on “*Assessing the effects of forest fragmentation on forest cover dynamics, tree species diversity and carbon stock in Taita Hills, Kenya*” I am pleased to inform you that you have been authorized to undertake research in **Taita Taveta County** for the period ending **11th September, 2019**.

You are advised to report to **the County Commissioner and the County Director of Education, Taita Taveta County** before embarking on the research project.

Kindly note that, as an applicant who has been licensed under the Science, Technology and Innovation Act, 2013 to conduct research in Kenya, you shall deposit a **copy** of the final research report to the Commission within **one year** of completion. The soft copy of the same should be submitted through the Online Research Information System.

BONIFACE WANYAMA
FOR: DIRECTOR-GENERAL/CEO

Copy to:

The County Commissioner
Taita Taveta County.

The County Director of Education
Taita Taveta County.

