IMPACTS OF BUSH ENCROACHMENT BY *Euclea divinorum* ON WILDLIFE SPECIES DIVERSITY AND COMPOSITION IN OL PEJETA CONSERVANCY IN LAIKIPIA, KENYA

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A THESIS SUBMITTED TO SCHOOL OF NATURAL RESOURCES AND ENVIRONMENTAL STUDIES IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE CONFERMENT OF THE DEGREE OF MASTER OF SCIENCE IN WILDLIFE MANAGEMENT, KARATINA UNIVERSITY

SEPTEMBER, 2018

DECLARATION

Declaration by Student

This thesis is my original work and has not been presented for conferment of a degree in any other University or for any other award.

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DEDICATION

I dedicate this work to my family for the support and encouragement through the course, and my siblings for their steadfast support and concern.

ACKNOWLEDGEMENT

This work was funded by Rufford Foundation, Rufford Small Grants (RSG) for Nature Conservation United Kingdom (UK).

In a special way, I thank my two supervisors Dr. Mwangi J. Kinyanjui from Karatina University and Dr. Johnstone Kimanzi from University of Eldoret for their guidance and encouragement.

I am grateful to Ol Pejeta Conservancy (OPC) staff particularly, Carol Ngw'eno, Bernard Chira and the entire team from Ecological Monitoring Unit (EMU) for their unrivalled support.

I thank Dr. Duncan Kimuyu for the camera traps used for wildlife surveys.

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LIST OF ABBREVIATIONS AND ACRONYMS

ASL	Above Sea Level
DEM	Digital Elevation Models
DOS	Dark Object Subtraction
EDA	Exploratory Data Analysis
EMU	Ecological Monitoring Unit
ESRI	Environmental Systems and Research Institute
ETM	Enhanced Thematic Mapper
GIS	Geographic Information Systems
GLM	Generalized Linear Models
GPS	Global Positioning Systems
Ha	Hectares
HCA	Hierarchical Cluster Analysis
IUCN	International Union for Conservation of Nature
KML	Keyhole Markup Language
LAI	Leaf Area Index
LiDAR	Light Detection and Ranging
MODIS	Moderate-Resolution Imaging Spectroradiometer

NDVI	Normalized Difference Vegetation Index
NIR	Near Infra-red
NPP	Net Primary Production
OLI	Operational Land Imager
OPC	Ol Pejeta Conservancy
PA	Protected Area
PAST	Paleontological Statistic Software Package for Education and Data Analysis
QGIS	Quantum Geographic Information Systems
ROI	Region of Interest
SCP	Semi-Automatic Classification Plugin
SPOT	Satellite Pour l'Observation de la Terre
TM	Thematic Mapper
TOA	Top of Atmosphere
USGS	United States Geological Survey
UTM	Universal Traverse Mercator
VIS	Visible wavelength
WGS	World Geodetic System

ABSTRACT

Savannah landscapes are extensively social-economically important ecosystems which support livelihoods. Despite their importance, they are facing a biome shift due to natural and anthropogenic induced perturbations leading to increase in woody species, a phenomenon referred to as bush encroachment. In Ol Pejeta Conservancy (OPC), *Euclea divinorum*, unpalatable woody species has become a concern due to its invasion into other habitat types which can potentially affect resources for various feeding guilds, consequently affecting ecosystem services. This study examined vegetation cover changes from 1987 to 2016, topographic features attributable to these cover changes, differences in species diversity and composition in encroached and non-encroached habitats as well as habitat preference or avoidance by various feeding guilds in the conservancy. Landsat images acquired during dry seasons were processed and classified into various vegetation cover types. Infra-red motion triggered camera traps were deployed in 2km by 2km grids for 14 days and nights in June 2016 to examine species diversity, composition and habitat preference or avoidance by various feedings guilds in OPC. Results revealed that *E. divinorum* cover increased upward significantly from 1987-2016 (Mann Kendall test for trend analysis *tau 1, n=6,* $p<0.01$). Further, digital elevation models, contours and slope based normalized difference vegetation index had influence on encroachment patterns by *E. divinorum.* Shannon Weiner diversity revealed that *s*pecies diversity and richness was higher in *E. divinorum* and lowest in Open grassland dominated areas while Hierarchical Cluster Analysis revealed that percentage similarity in species composition was highest between *E. divinorum* and mixed bushland habitats. Jacobs' Index means revealed that *E. divinorum* habitat was significantly avoided by all feeding guilds ($t_1=2.253$, $d.f=3$, $p<0.01$) while *A. drepanolobium* dominated habitats were significantly preferred ($t_1=2.353$, $d.f= 3$, $p=0.03$). The findings show that increase in *E. divinorum* cover, which has higher species diversity and evenness, however is avoided by all feeding guilds in OPC. As such, there is need to actively manage encroaching species as well as further research on impacts of encroachment on grass biomass and diversity. These findings are beneficial to policy makers regarding management of healthy ecosystems.

CHAPTER ONE

INTRODUCTION

1.1 Background

Woody species are increasingly encroaching grasslands and mixed bushland globally (Dickie, Schnitzer, Reich, & Hobbie, 2007). These invaders often form persistent patches which alter composition and structure of the plant community in savannah (Wangen & Webster, 2006). Savannahs are defined as tropical seasonal ecosystems with continuous grass layer, mixed with forbs and sedges with variable cover of trees and shrubs. They are characterized by distinct dry and wet seasons. In Africa, savannah ecosystems have been widely relied on for livestock production and wildlife conservation especially wild herbivores (Devine, McDonald, & Maclean, 2017). However, these ecosystems are shrinking rapidly (Oba, Syvertsen, & Stenseth, 2000) a major cause of their decline in coverage hence world savannah ecosystems are declining/ altered by a phenomenon called "bush encroachment" (Van Auken, 2000). Bush encroachment is increase in woody vegetation density, cover and biomass in savannah and rangeland ecosystems (Oba et al., 2000).

The increase in woody cover is attributed to overgrazing due to positive correlation between grazing pressure and increased woody cover in savannah. Other possible causes are increased precipitation rates (Joubert, Rothauge, & Smit, 2008), fire suppression and favourable edaphic conditions and increasing carbon IV oxide (Sankaran, Ratman, & Hanan, 2008; Oba et al., 2000). Moisture is a limiting factor in these savannah ecosystems exacerbated by low or erratic precipitation patterns. Hence, savannahs are fragile ecosystems that are sensitive to perturbations resulting to bush encroachment or habitat quality degradation.

The effects of increase in woody species varies remarkably, such as land cover change from grassland to forested bushlands resulting to decreased grass biomass and by extension increase in fire intolerant woody species that can potentially affect species composition. In extreme case woody encroachment can result in ecosystems structure and functioning decline as well as irreversible landscape degradation (Tobler, Cochard, & Edwards, 2003; Khavhagali & Bond, 2008). Woody encroachment in savannah ecosystems is emerging as a new threat in these landscapes. At the extreme, land cover changes in these ecosystems impede visibility (Riginos & Grace, 2008), increase in perceived predation risk and reduced penetrability by medium to large herbivores. Increasingly, *E. divinorum* a woody species is considered as an encroacher species within its range in many parts. It is fast growing, unpalatable and fire-resistant woody species (Sharam, Sinclair, & Turkington, 2006).

In Ol Pejeta Conservancy (OPC) in Laikipia County Kenya, *E. divinorum* is regarded as a local encroacher species. *E. divivnorum* is an ever-green woody species multi stemmed with highly branched crowned growing up to about 6m at maturity but occasionally grows up to 15m (Orwa et al., 2009). It is hardy and fast grower especially if in its natural habitat however when disturbed can become invasive due to its ability to develop suckers from its roots (Van Wyk & Van Wyk, 1997). *E. divinorum* is listed as least concern in the International Union for Conservation of Nature (IUCN) according to Raimondo et al. (2009). This plant species dioecious flowering in August to December and with cup shaped flowers, produces a fleshy berry, purple in colour when ripe edible fruits majorly fed on by birds (Orwa et al., 2009). The species in native in most of East and South African countries namely; Kenya, Uganda, Tanzania, Zimbabwe, South Africa, Swaziland, Namibia and Botswana (Orwa et al., 2009). In most areas in the conservancy, this plant species is present and higher in areas such as valley bottoms and drainage channels where soil depth and moisture content are significantly high (Wahungu et al., 2012). Encroachment in isolated ecosystems may result to decline and/or extinction of native species and can potentially affect species diversity, distribution, abundance (Towns, Atkinson, & Daugherty, 2006).

Invasion has become a great concern and threat to conservation efforts, a wide spread ecological problem affecting savannah due to its associated costs in eradicating established invasive/encroaching species (Tobler et al., 2008). It is worth acknowledging that no single approach can be employed to prevent, eradicate, manage or control invasive/encroacher species hence a combination of various techniques is preferred where its applicability is best. In most cases, mechanical, chemical and biological control techniques have been used widely to manage invasive species and restore degraded ecosystems.

Fire as a management tool has been used to control invasive species. In OPC, prescribed burning was employed as a way of controlling *E. divinorum* encroachment but abandoned when a study conducted revealed deleterious effects of fire on other plant species (Wahungu, Mureu, & Macharia, 2009). Furthermore, use of fire despite many associated benefits such as removal of moribund grass, remains a debatable subject (Sharam et al., 2006). This study reports on changes in cover dominated by *E. divinorum* from 1987 to 2016 a period chosen due to spatiotemporal comparability of the Landsat sensors, topographic features and their influence on encroachment patterns, species diversity, composition and preferential habitat use.

1.2 Statement of the Problem

Increase in woody species through encroachment in savannah ecosystems poses a serious threat to ecosystems function especially tree-grass coexistence. Grass-tree balance influence grassland/rangeland economic services, biodiversity conservation and ecosystem function at local and landscape scales (Riginos & Grace, 2008; Gemedo, Maass, & Isselstein, 2006). In OPC, a facility which is actively managed for livestock production and wildlife conservation, encroaching species are becoming a major concern to management. *E. divinorum* encroachment towards *Acacia drepanobium,* grasslands and other open bush land vegetation cover types can potentially affect food resources for mega faunas in these ecosystems especially the critically endangered Eastern Black Rhino (*Diceros bicornis,* (IUCN 2011 Red Listing) and vulnerable African Elephants (*Loxodonta africana*, IUCN 2008 Red Listing) among other herbivores. This encroachment can as well potentially reduce the available ranging lands and to some extend exterminate some of the wild flora and fauna.

1.3 Justification and Significance

Monitoring of ecosystems function and health is critical for ecosystems service realization. *E. divinorum* is an encroacher species (locally) and interferes with species diversity (Towns et al., 2006) in savannah ecosystems hence, understanding factors that contribute to its encroachment as well as impacts on other habitat types are important for management of these landscapes. As such, the research findings are crucial for wildlife and rangeland managers to inform sound decision making regarding management of these ecosystems for sustainable development.

1.4 Study Objectives

1. To determine changes in the area under *E. divinorum* dominated cover from 1987 to 2016 in OPC.
2. To examine topographic features attributable to encroachment patterns by *E. divinorum* in OPC.
3. To compare wildlife species diversity and composition within encroached and "non- encroached" habitats in OPC.
4. To determine habitat preference for encroached and "non-encroached" habitat among various feeding guilds in OPC.

1.5 Statistical Hypotheses

H_{01}: The area under *E. divinorum* vegetation cover has not changed significantly from 1987 to 2016 in OPC.

$H_{02:}$ Topographic features are not attributable to encroachment patterns by *E. divinorum* in OPC

H_{03}: Wildlife species diversity and composition is the same in encroached and "non-encroached" habitats in OPC.

$H_{04:}$ There is no significant preference for encroached and "non-encroached" habitats among various feeding guilds in OPC.

CHAPTER TWO

LITERATURE REVIEW

INTRODUCTION

African savannahs contribute remarkable revenue from game viewing in tourism industry as well as provide Ideal rangelands as grazing fields thus livestock production and further to larger extend as it is being observed currently, these landscapes are being converted for irrigated agriculture to support the burgeoning human population. Their attached utility value by local communities especially nomadic cannot be underrated due to its enormous contribution to their wellbeing. Changes in these ecosystems such as bush encroachment among others have led to rise in recurrent conflicts over grazing resources. These conflicts are expected to occur occasionally if these land cover changes that alter/limit availability of a central resource continue to take place in these ecosystems. Bush encroachment is proliferation of woody species often unpalatable to both domestic and wild herbivores suppressing grass/leaves for grazers/browsers and to the extreme resulting to closed habitats impenetrable by these feeding guilds and increase in perceived predations risks. This phenomenon leads to reduction in carrying capacity of these savannahs according to Ward (2005) thus artificial shrinking of the available ranging land especially herbivores.

2.1 Bush Encroachment in Savannah and its Implications

Savannah structure and dynamics are driven by an array of factors which determines vegetation structure and composition. Chiefly, they can be grouped in two categories namely; bottom up and top down factors. Bottom up determinants include available moisture, soil types, nutrients and topographical gradients which vary on temporal and spatial scales from local to global scales (Joubert et al., 2008) while top down determinants include fire regimes (frequency, severity and duration), as well as herbivory (Van Langevelde et al., 2003; Sankaran, Ratman, & Hanan, 2004) and other human induced perturbations. In this regard, structure and dynamism in savannah occurs as a function of top down factors (disturbances) acting within the constraints of bottom up factors. Although savannah ecosystems support an enormous community of both plant and animal species, they have continuously been exploited for livestock production, fuelwood, agroforestry, agriculture and infrastructural developments. As a result of these often-uncontrolled human socio-economic exploitation of these ecosystems, vegetation structure, composition productivity, biodiversity and distribution have changed drastically (Hudak, Fairbanks, & Brockett, 2004; Foley et al., 2005).

In response to these changes in savannah, protected areas (PA) have been designated for biodiversity conservation to curb further alteration and maintain savannahs in their pristine or near pristine conditions. Nevertheless, these PA are experiencing characteristically unstable vegetation structure and composition due to the earlier mentioned dynamic effects of both bottom up and top down drivers of savannah landscapes (Hudak & Wessman, 2001; Hudak, Wessman, & Seastedt, 2003; Hudak et al., 2004). Woody cover in savannah ecosystems is a very important biophysical variable in determining the status of savannah (Gareth, February, & Verboom, 2007). As such, investigations in to spatial context of woody cover resource has been considered a key component in understanding patterns and distribution of species habitat requirement or habitat preference hence species density and diversity (Mutanga et al., 2004; Mutanga & Rugege, 2006).

Encroachment of savannah ecosystems is becoming an ecological problem, a challenge for habitat ecologists and natural resource managers. Bush encroachment is typically a gradual replacement of grass and forbs by woody species (Van Auken, 2009). Further, encroachment is considered as the most extensive and threatening life form in range degradation (Briggs et al., 2005; Blaum, Rossmanith, & Jeltsch, 2006) whose implications can span vast areas of arid and semiarid landscapes globally (Asner et al., 2012). According to Ward (2005), bush encroachment significantly reduces carrying capacity of land for both livestock and wild herbivores if their key resources are replaced, (Wessels et al., 2006; Mutanga & Rugege, 2006) typical examples of some wild animals affected include but not limited to black rhinos and elephants (*Acacia drepanolobium* forms a key diet in OPC hence its decline potentially affects their survival triggering management interventions).

Encroachment by woody resources may influence fire regimes thus occurrence, severity, intensity and duration in savannah ecosystems (Hudak & Brockett, 2004). As such, these land cover changes can potentially change persistence of biodiversity, soil moisture content levels, climate change and climate variability at temporal and spatial scales (Li et al., 2007). Increase in woody cover translates to increase in water demand and/or use by plants (Kim & Jackson, 2011; Nosetto et al., 2012), and consequently affect/ alter energy balance through changes in albedo (Beltran-Przekurat et al., 2008).

Particularly, in protected area, ecotourism is a major source of conservation income as such, due to bush encroachment it may suffer significantly if there is poor visibility for game viewing (Wigley, Bond, & Hoffman, 2009). Conversely, proliferation by woody species can also be beneficial to people's economic endeavours depending on land uses for example providing timber and wood products for construction of shelters, fencing and firewood among other uses (Wigley et al., 2009). Further, in conservation areas, increase in woody species can increase food for browsing wild animals and increased avifauna diversity especially those which rely on woody species for shelter and fruits for food as their main source of diet (Wigley et al., 2009). In areas prone to soil erosion such as riparian zones and water shades, increase in woody species controls soil erosion through soil anchoring by roots. Despite the associated positive and negative impacts, drivers of encroachment by woody plant species are poorly documented according to Joubert (2007).

2.2 Fire and Herbivory as Forms of Disturbance in Savannah Ecosystems

The African continent is referred to as "fire continent" with wide spread biomass burning forming an integral part of functioning African grasslands and savannah (Ward, 2005). Climatic conditions are the driving form of fire ecology in African savannahs which have distinct dry and wet seasons, natural ignition sources such as lightning and flammable fuel loads during the dry seasons (Ward, 2005). Fire ecology is defined as response of biotic and abiotic components of the ecosystems after fire regimes (Ward, 2005). Fire regimes have changed with increasing population where natural fires have successively been suppressed by anthropogenic fires. In most tropical savannah, fires are initiated and controlled by humans hence occurrence in human influenced.

African savannah ecosystems are prone to fires which plays a vital role in determining composition and structure of these ecosystems. In the absence of fire, several savannahs could potentially develop in to closed thickets and forest, however over period this has seen development of fire tolerant species and fire depended flora (Bond, Woodward, & Midgley, 2005). Fire has the ability to dictate changes that occur in savannah plant composition hence success of using fire relies on understanding fire and its impacts (fire and tree mortality). Pastoralists and rangeland managers have widely used fire to manipulate tree-grass cover ratios (Oba et al., 2000). Pastoralists or conservationists burn grasslands and savannah ecosystem in African to remove moribund grass/unpalatable resources to improve quality of grazing resource for domestic and wild animals (Parr & Anderson, 2006).

Another reason for burning is the need to remove/suppress spread of encroaching woody species which have been identified to have deleterious effect on grazing resources (Navashni, Trollope, & Brian, 2006). This has been achieved through prescribed burning and understanding fire regimes, fire intensity and fire severity by controlling fuel loads comprising of dry biomass (plant debris). The most common types of fires in savannah are surface fires which burn as either back fire or head fire (Trollope, 2011). Under extreme fire conditions, crown fires can also occur and are sustained by abundance and continuity of aerial fuel loads.

Research work investigating effects of surface fires on grass swards reported significant differences between head and back fires. Back fires significantly suppressed regrowth of grass compared to head fires (Trollope, 2011). This is due to critical threshold temperatures maintained at 95^{o}C for approximately 20 seconds longer compared to head fires according to Trollope (2011). Further, more heat is released to the ground by back fires compared to head fires and the implications are that shoot apices of grass are adversely affected during back fires than during head fires (Trollope, 2011). Different fire types have different impacts on grass swards (Trollope, Fyumagwa, & Trollope, 2003). Height of fire flames contributes to increase in temperatures hence top kill is severe during head fires as opposed to back fires due to differences in height of the flames. Understanding of the fire ecology can be relied on especially during prescribed burning meant to control or suppress encroachment by woody species.

Despite these numerous benefits of fire in management of savannah ecosystems, they do contribute significantly to woody vegetation perturbations. Impacts of fire on other plant species are poorly documented since most fire managers focus on achieving certain objectives as opposed to overall ecosystem integrity. Savannah ecosystems support wildlife conservation and livestock production (Grace et al., 2006). In some cases, savannahs are purely managed for livestock production or wildlife conservation. However, recently both livestock production and wildlife conservation are being integrated to maximize profits. Whenever carrying capacities are exceeded, tree/grass ratio is affected thus altering plant community composition as such, a form of disturbance is impacted on these ecosystems. Further, wild animals especially mega fauna such as elephants are known drivers of changes in savannah ecosystems (Bond 2008; Pringle et al., 2015). This is due to their ability to open up closed habitats by knocking down trees. These forms of disturbances may by pass human eye since their cumulative impacts will need longer periods to be detectable. Using modern technology to process satellite images, these disturbances can be flagged out and enhance better understanding of these ecosystems.

2.3 Remote Sensing and Geographic Information Systems (GIS) in Habitat Monitoring

Intensive ground surveys cannot keep pace with rapid land cover/change over large areas since they involve “wait and see” protocol hence new technologies are necessary (De Sherbinin, 2005). These intensive ground surveys on-site field measurements require lots of funds and human capital investment which may be nearly impossible to obtain for long term studies. Information and data needs have been growing in scope and complexity (De Sherbinin, 2005) hence this has revolutionised ecological techniques.

Collecting information about a given object or phenomenon without making any physical contact has led to revolution in monitoring and management of ecosystems. This approach is referred to as remote sensing. Ecological remote sensing can be divided in to three main parts. First, land cover classification which is the physiographical characteristics of the surface environment based on land cover types (Imam, Kushwaha, & Singh, 2009). Firstly, it entails clustering of image pixel in to relatively similar pixel with same properties. Secondly, measurement of ecosystems functions at spatial scales such as leaf area index (LAI) and net primary productivity (NPP) through normalized difference vegetation index (NDVI) (Tagesson, Eklundh, & Lindroth, 2009) and thirdly, change detection thus flagging out land cover changes over a series of time for a given area hence providing an Ideal way of monitoring significantly large ecosystems (Pellikka et al., 2009). All the above-mentioned approaches can be used to study an ecosystem for better management. Different satellite sensors offer data in different spatial and temporal resolution hence limitations in applicability of the data with respect to purpose. One of the limiting factors is commercial aspect of high-resolution spatial data and supporting processing software. Further to this, geometric scale has become a hindrance especially if the area under study is small hence data acquired by satellite with high spatial resolution are required.

Land cover change and land use information can be obtained from the medium to course resolution acquired from satellites such as Landsat, Satellite Pour l'Observation de la Terre (SPOT) and Moderate Resolution Imaging Spectroradiometer (MODIS) (Wulder et al., 2004) whilst fine scale disturbance/ cover changes can be monitored using fine scale spatial and temporal resolution sensors. As a result, high resolution remote sensed data sets reduce the problem of pixel mixture which is a pronounced challenge with medium to course resolutions (Hirose et al., 2004; Lu & Weng, 2007). Often, this involves high cost in getting such data for instance, Quickbird which is among those with finest resolution such as IKONOS though too expensive to acquire. Several methods can be explored in order to derive desired outputs such as spatial data with fine pixel resolution. Light Detection and Ranging (LiDAR), geometrical-optical method with high optical and resolution can be used to detect fine scale disturbances. Landsat Images supported by Google Earth Engine can be used to study vegetation cover over time as well.

2.4 Species Richness and Diversity: Camera Trap Approach

Land use and land cover change has profound implication on animal species composition and distribution (Ward, 2005). Synergistic interaction (primary and secondary drivers of ecosystem) in savannah ecosystems can alter ecological services and functions whose consequence can modify species habitat utilisation and behaviour. In monitoring of species diversity and richness several approaches have been employed (Mounir & Zuhair, 2012) however there exists some drawbacks involved. Among the difficulties two are more bulging: inability to survey the entire area of interest and inability to detect all animals (Thompson, 2004).

With advancement in technology, new methods have been developed to reduce disturbance, cost and document even rare and elusive wild animal species (Rowcliffe & Carbone, 2008). This has led to development of camera traps that are infra-red and motion triggered or body heat triggered (Balme, Hunter, & Slotow, 2009; Mccarthy et al., 2008). Interests and increase in camera trap use success has led to dramatic increase in number of publications involving their application (Rowcliffe & Carbone, 2008). Camera traps have and are still being used to understand habitat preference and occupancy (Bowkett, Rovero, & Marshall, 2007). Infra-red camera trap varies in size, functionality and use. In regard to these differences, they are different in prices hence factor that can hinder getting quality data for ecological monitoring work.

Cameras traps are efficient in conditions that hinder direct observation or ineffective direct surveys hence it has been made to possible study nocturnal animals or those that warry of human being or use microsites within a given habitat (Larrucea, Brussard, Jaeger, & Barrett, 2007; Mccarthy et al., 2008). They can provide nearly accurate estimation of species abundance especially terrestrial mammals and birds >1 kg though this will rely mostly on camera traps position and settings. Further, they can allow study of species diversity in a given habitat. Despite their use in species abundance estimation there are potential for biasness due to differential detectability of the species.

In case baits are used, animals may spend more time in front the camera resulting to numerous photo which may misinform the researcher and to deal with this problem one can discard photos of same species captured within a set time (Larrucea et al., 2007; Tobler et al., 2008). Going forward camera traps offer ideal approach to survey of mammals especially if vast areas are to be covered and long-term monitoring as opposed to direct survey. Cost wise is also way below compared to amount of finances required for long term monitoring using conventional ways.

Bush encroachment potentially influences abiotic and biotic factors interactions. Changes in vegetation structure often resulting in increased woody plant species can alter available light, nutrients and water affecting majorly grass and dictating patterns of resource availability according to Lett and Knapp (2003). These dynamic changes trigger different responses to woody species encroachment by biotic factors. Increase in woody species can lead to decline in quality of habitats at local scale affecting both domestic and wild animal populations especially grazers and has been attributed to decline in species richness and diversity (Sirami et al., 2009; Blaum et al., 2009: Blaum, et al., 2006). Predator prey dynamics also change as function of bush encroachment for instance cheetahs are known to utilise open grasslands however in the encroachments habitats they utilise open patches within the encroached habitats (Hamilton, 1986) due to reduced and prey detectability (Muntifering et al., 2006) with elevated chances of injury while hunting in encroached habitats. These changes in vegetation structure affect hunting strategies for most carnivores which are of concern due to their position in food chain.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Study Area

The project site was Ol Pejeta Conservancy (OPC) which covers 90,000 acres (360km^2) and is a classic example of an African savannah. It lies between Mt. Kenya and the Aberdare Mountains (0^0 7.288’N, $36^0$42.384’E and 0^0 8.634’N, 37^0 0.605’E) ($0^0$1.831’S, $36^0$46.578’E and $0^0$5.7025’S $37^0$2.492’E). It lies at an average altitude of 1810m and has a mean annual rainfall of 739mm with mean maximum and minimum temperatures of 28^{o}C and 12^{o}C, respectively.

OPC is a privately-owned conservancy primarily established as a black rhino sanctuary but currently has abundant wildlife species, including Elephants (*Loxodonta Africana)*, Black Rhino (*Diceros bicornis)*, Northern White Rhino (*Ceratotherium simum cottoni*), Buffaloes (*Syncerus caffer*), Grevy’s Zebra (*Equus grevyi*), Plains Zebra (*Equus burchellii)*, several species of medium-sized gazelles, Lions (*Panthera leo*), Cheetahs (*Acinonyx jubatus*), Spotted hyaena (*Crocuta crocuta*), Striped hyaena (*Hyaena hyaena*), and Black backed Jackals (*Canis mesomelas*), among others.

The conservancy is also a chimpanzee sanctuary providing refuge for rescued chimpanzee from black markets. The conservancy has become a successful conservation site with integrated livestock production. There are several seasonal rivers and one permanent river Ewaso Nyiro River with its source at Mt Kenya and drains to Lorian Swamp, several boreholes and man-made dams. Major land cover types include grasslands, *Acacia drepanolobium*, *Acacia xanthophloea*, *Euclea divinorum*, and mixed woodlands. The conservancy is surrounded by an electric fence with three “corridors” to allow movement of wild animals in and out (but movement of rhino species is prevented due to the risks involved). The conservancy is surrounded by agro-pastoral communities and towards the north by other adjoining conservancies. Map of the study site showing OPC, Major towns and other facilities in Laikipia County is as shown in Figure 1.

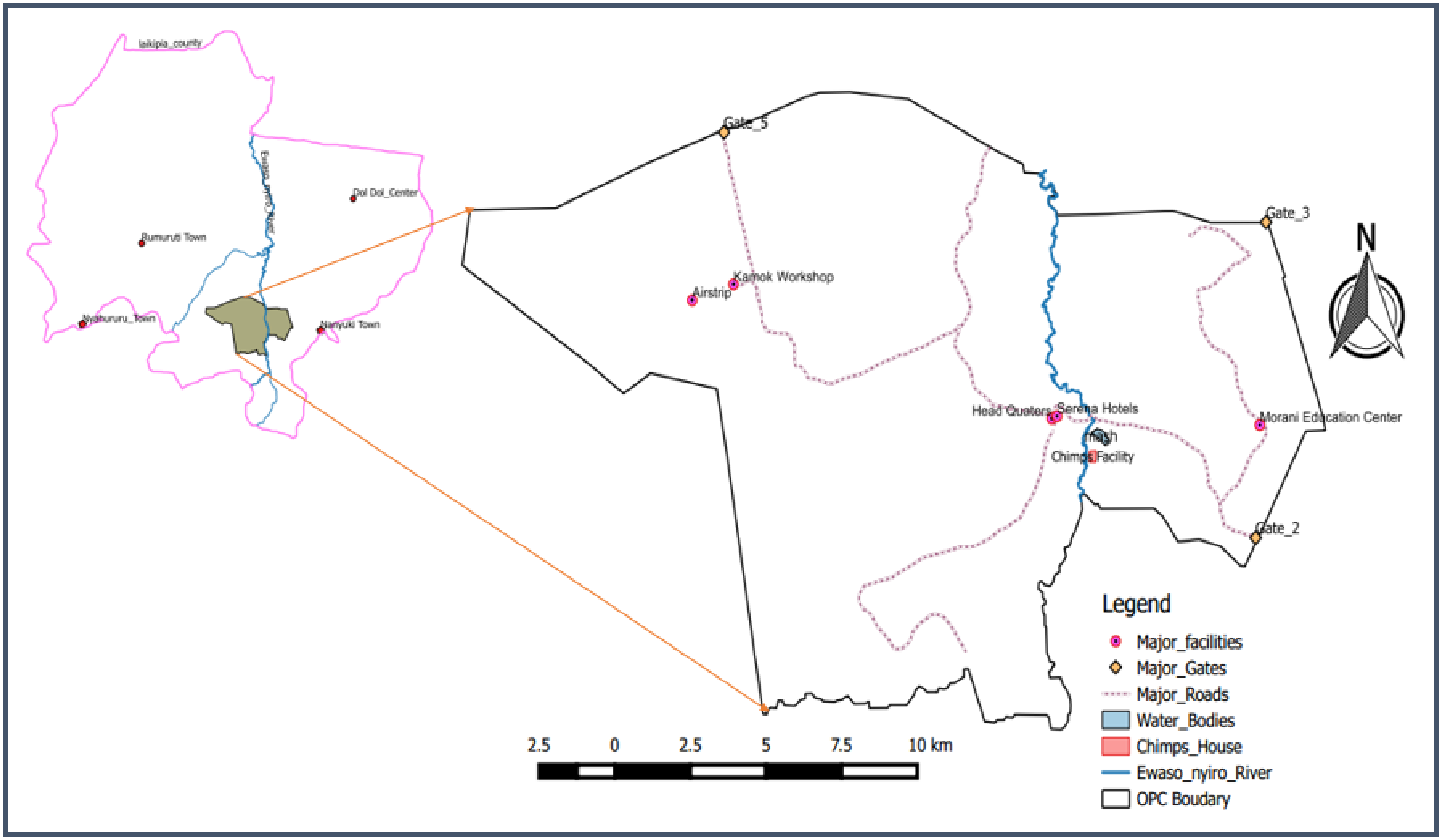


Figure 1: Map of Ol Pejeta Conservancy

3.2 Methods for Image Acquisition

Land cover thematic shapefiles of the Ol Pejeta Conservancy (OPC) created by the Ecological Monitoring Unit (EMU), a research unit in OPC with support from Environmental Systems Research Institute (ESRI) were used to demarcate the study area. An overlay of the shapefiles on the Google Earth Satellite Layer on a Quantum Geographic Information Systems (QGIS) platform was used for creation of new layers based on observable features for the year 2016. To allow perfect overlay of the features, the shapefiles were projected to the Universal Transverse Mercator (UTM) Zone 37N. The GPS points were converted to polygons and edited to precisely show areas currently occupied by *E. divinorum* species using a current Google Earth Satellite Layer (for 2016) as the reference.

3.2.1 Landsat Imagery Data Source and Materials

Landsat Imagery time series data were obtained from United States Geological Survey website (USGS, 2016) as the primary data source for general land cover classification. Landsat imageries acquired during dry season either February or March for easy detection of *E. divinorum* an ever-green woody species and cloud free (< 3%) in the year 1987, 1995, 2000, 2005, 2010 and 2016 provided multitemporal data. The Landsat 5 Thematic Mapper (TM), Landsat 7 Enhanced Thematic Mapper (ETM)+ and Landsat 8 Operational Land Imager (OLI) were appropriate for general land cover trends and change analysis. The sensors had comparable spatial (pixel) resolution of 30m and temporal resolution of 16days hence the research considered the above-mentioned years for vegetation classification. The oldest image with similar characteristics was acquired in 1987 and recent was acquired in 2016.

Details of the images used in this study are given in Table 1.

Table 1: Details of images used in the study

Satellite/Sensor	Date of acquisition	Path/Row	Spatial/ Temporal resolution
Landsat_5/TM	Feb_25_1987	168/60	30m/16 days
Landsat_5/TM	May_22_1995	168/60	30m/16 days
Landsat_7/ETM +	Feb_02_2000	168/60	30m/16 days
Landsat_7/ETM+	Feb_02_2005	168/60	30m/16 days
Landsat_5/TM	Feb_02_2010	168/60	30m/16 days
Landsat_8 OLI	March_28_2016	168/60	30m/16 days

Source: USGS website (USGS, 2016)

3.3 Pre-Classification of Digital Images Processing

3.3.1 Top of Atmosphere (TOA) Reflectance

As the light passes through the atmosphere, it interacts with other particulate maters such as haze, water vapour and smoke among others, hence can considerably affect the signal before and after interacting with the object in question (Chavez, 1996; Lillesand & Kiefer, 2004). As a result, this may necessitate in situ atmospheric correction. In order to achieve better and clear Landsat scenes, TOA reflectance was performed using the algorithms as developed for Semi-Automatic Classification (SCP) Plugin Version 5.0 of QGIS software.

3.3.2 Dark Object Subtraction (DOS1)

Dark object subtraction (DOS) is a family of image based atmospheric correction techniques which include DOS1, DOS2, DOS3, DOS4. These techniques have one assumption according to Chavez (1996) that, within an image some pixels are incomplete shadows which are received by satellite as a function of atmospheric scattering. This assumption leverages the fact that on the earth surface, few targets may be black or assumed one (1) percent reflectance which is better that zero. Here, for the purpose of this study DOS1 technique was used as described by Luca (2016) (Semi-Automatic Classification Plugin in QGIS release 4.8.0.1).

3.3.3 Image Re-Projection and Band Compositing

After the various image corrections, all images were re-projected to World Geodetic System (WGS) 84 Universal Traverse Mercator (UTM) Zone_37N and other vector data were re-projected to this projection system. The raster images were then clipped using a vector mask boundary of the study area.

3.3.4 Image Classification

Multitemporal Landsat images TM, ETM+ and OLI of 1987, 1995, 2000, 2005, 2010 and 2016 were used to study land cover dynamics with more focus on changes in *E. divinorum* as the species of concern for this study. Here, classification used supervised classification technique an algorithm that uses spectral signature to identify different materials in an image and finally generate a thematic map of the land cover. In order to minimise potential of vegetation cover type mix up while classifying, google earth image satellite layer 2016 was used to precisely map different cover types. Further, an option in the classification plugin for Normalised Difference Vegetation Index (NDVI) was activated to display NDVI values of different cover types. This was used to enhance classification accuracy. This was the starting point to enhance classification accuracy. Further to this, thirty (30) ground truthing sites were generated randomly, coordinates loaded in to GPS and later visited to compare similarity between spectral output from Landsat and on ground reality.

In order to analyse images through supervised classification semi-automatic classification for QGIS (also known as supervised classification) was used. This process requires creation of temporary region of interest (ROI) as vector file(s) which is saved as classification signature file in the plugin. Vegetation cover was assigned one macro class identity and separated finer in to five micro class identities to achieve desired results. Maximum likelihood algorithm which calculates probability distribution for the classes, using the Bayes' theorem estimating if a pixel belongs to a particular land cover class was used (Richards & Jia, 2006). This classification algorithm is preferred over the other algorithms due to its ability to use well developed probability theory (Richards & Jia, 2006). Assessment of spectral distance (spectral separability) to minimise classification errors was executed using Jeffries-Matusita Distance where if asymptotic distance is 2 the signatures are completely different whilst, if it is 0 signatures are identical (Richards & Jia, 2006).

3.3.5 Accuracy Assessment and Classification Report

In order to evaluate the reliability/correctness of the classification output, the random ROI creation option in SCP was used to generate a total of thirty (30) samples for reference purposes for error matrix calculation (Luca, 2016). Here, overall classification for classification report 1987 was 80%, 1995 86%, 2000 84%, 2005 87%, 2010 89% and 2016 88%. Finally, a classification report was generated giving proportion of each land cover and total areas occupied by each class in Hectares. Image processing and classification were done using Open source software QGIS.

3.4 Topographic Features Attributable to Encroachment Patterns

3.4.1 Slope Based Normalized Difference Vegetation Index

Topographic features were examined using Normalized Difference Vegetation Index (NDVI) with reference to slope in OPC. Here, the NDVI equation was used to compute NDVI values as shown (Rouse et al., 1974)

$$\mathrm{NDVI} = \frac{(\mathrm{NIR} - \mathrm{VIS})}{(\mathrm{NIR} + \mathrm{VIS})}$$

Where NIR and VIS stands for spectral reflectance measurements acquired in near infra-red and visible regions respectively. Values range from -1 to +1 where values close to +1 are an indication of dense canopy with high chlorophyll content whilst close to -1 are land cover/or bodies with low chlorophyll content and those without chlorophyll content such as water bodies and bare soils.

3.4.2 Contours and Elevation Overlaid on Vegetation Map

Contours and elevation map were created using Google Earth to generate coordinates and altitude of various points in OPC. A path was run over the google earth layer covering the area of interest. A file in Keyhole Markup Language (KML) format was generated which was uploaded to TCX converter (a freeware software for extraction of elevation) for extraction of the altitude. The file was converted in to recognizable file in QGIS and digital elevation models (DEM) as well as contours generated.

3.5 Wildlife Survey for Species, Richness, Diversity and Composition

To determine species diversity and richness and compare between encroached and non-encroached sites, the entire OPC map was divided in to 2x2km grid for infra-red motion triggered camera trap deployment and further in three sectors namely Eastern, Southern and Northern. Camera traps Reconyx RM^{45} Hyperfire model and Bushnell Model were deployed systematically at the centroid of each grid within a given Land cover type taking cognize of animals' trails or paths to maximize animal photo captures. Camera traps were either mounted on a tree or housed in a metal cage for the case of open grassland and placed at knee height (50cm) above the ground surface. They were set to remain active for 24hrs with no delay in between photo taking session and in rapid fire mode.

Camera traps remained in the field for 14 consecutive days and nights and serviced after the seventh day to check cameras' battery level, memory card storage status and general condition of the camera trap. In between the two deployment sessions, there was a break of three days to allow for battery charging and cleaning of the storage cards in preparation for the next deployment event. Finally, all the data recording camera trap location, habitat type, species names, group size and time, were downloaded from the memory card and cleaned (removing false triggers, duplicates, and blurred images) in readiness for analyses as described by Rowcliffe and Carbone (2008).

A total of thirty-six (36) camera traps were deployed in the entire study area where nine (9) camera traps were used in each of the four vegetation classes under consideration thus *E. divinorum*, *A. drepanolobium*, Open grassland and Mixed bushland. A. *xanthophloea* dominated habitat was exclude for animal species due it its small areas coverage (<1%) by the year 2016 only major cover was present inside the Chimpanzee enclosure. Equal number of camera traps per habitat type were deployed in order to account for differences in area coverage by different habitat types. Total sampling effort was calculated as total number to cameras used multiplied by 24hrs they were set active multiplied by number of days they remained in the field hence;

Totals sampling effort 36 camera traps x 24hrs x 14 days = 12096 Hours

3.6 Data Analysis

3.6.1 Landsat Image Analysis

Preliminary analyses of the Landsat classification output results were done in QGIS using the default user options in the SCP plugin version 5.0 (Luca, 2016). Here, results such as land cover class proportions for further analyses were generated. Further, exploratory data analyses (EDA), Mann Kendall test for trend analysis as described by Gilbert (1987) were performed using R statistical software (R Studio Core Team development, version 3.1.2 2013).

3.6.2 Species Richness and Dominance Across Habitat Types

Species richness is defined as variety of species/number of different species in a given habitat under consideration whilst species dominance is defined as most conspicuous and abundant species in a given habitat under consideration (Shannon & Weaver, 1949). Simpson's index of dominance (C) (Shannon & Weaver, 1949) is calculated as

$$C=\Sigma(P_i)^2$$

Where C = is the Simpson's index of dominance

P_i = proportion of species i in the community

This is interpreted using the theoretical values ranging from 0 to 1 where, if values are close to 0 it's an indication of a more even community while high values indicate less even or more dominated community.

3.6.3 Species Diversity and Evenness

Species diversity and evenness were computed using the Shannon wiener index (Shannon & Weaver, 1949) which is a robust index since it takes in to account for both species abundance and evenness on the species present (Krebs, 1999). Species evenness refers to how close in numbers each species in an environment is. It's a measure of biodiversity which quantifies how equal a community is numerically.

Shannon Weiner index is calculated as;

$$H = -\sum_{i=1}^{s} (P_i * \ln P_i)$$

Where,

H = the Shannon diversity index

P_i = fraction of the entire population made up of species i

S = numbers of species encountered

$\sum$ = sum from species 1 to species S

This was executed using Paleontological Statistic Software Package for Education and Data Analysis version 1.0.0.0 (PAST).

3.6.4 Species Composition

Species composition refers to identity of all species that make up a community in a given ecosystem. Hierarchical Cluster Analysis (HCA), a multivariate test which groups observations by dissimilarity or similarity (Gauch, 1982) was used to compare species composition in the four habitats cover types namely *Euclea divinorum* dominated area, *Acacia drepanolobium* dominated, Open grassland and mixed bushland. Linkage method was performed using Bray Curtis similarity analysis which uses species abundance data was performed and consequently cluster analysis used to generate dendrogram showing species composition similarity across the vegetation cover types.

3.6.5 Habitat Preference or Avoidance Analysis by Various Feeding Guilds in Ol Pejeta Conservancy

Preference and avoidance of habitat was tested using Jacobs' Index (Jacobs, 1974) modification of a simple Ivlev Index (Ivlev, 1961) which is a more robust test and is not affected by bias to rare habitat types and nonlinearity. Further, increasing heterogeneity is bound, defined and does not lack symmetry between selected and rejected values, hence Jacobs' index was deemed appropriate (Jacobs, 1974). Jacobs' index is expressed as:

$$D_i = \frac{r_i - p_i}{r_i + p_i - 2\, r_i p_i}$$

Where; r*i* proportion of observations in habitat *i* (habitat utilised) of that species and p*i* is the proportion of habitat *i* available in the study area.

Mean value Jacobs' indices for each habitat were tested using one sample t-test for significance preference or avoidance against a mean of 0. Further, one-way analysis of variance was used to test significance avoidance or preference across the four vegetation types that were considered. Prior to these tests, normality tests were performed to ascertain that data did not violate assumptions for parametric test (Palomares et al., 2001) The Jacobs' Index values ranges from -1 for avoidance through 0 for random selection to +1 for preference.

CHAPTER FOUR

RESEARCH FINDINGS, PRESENTATION AND INTERPRETATION

4.1 Vegetation Map of OPC

Landsat image of 2016 produced the five major land cover types of focus which include *A_xanthophloea*, *E. divinorum*, Open grassland, *A. drepanolobium*, and Mixed bushland. These vegetation land cover types were in the following proportions; *E. divinorum* 49.7%, Open grassland 24.2%, *A. Drepanolobium* 17.0%, mixed bushland 8.8% and *A_xanthophloea* 0.3%. Further, area coverage for each vegetation cover type is as shown in Table 2 while the vegetation map is as shown in Figure 2.

Table 2: Proportion of various cover types for year 2016

Cover type	***Area Cover (Ha)***	***(% of vegetation type)***
E.divinorum	14456.0	49.7
Open grassland	7051.7	24.2
A. drepanolobium	4950.3	17.0
Mixed_bushland	2574.0	8.8
A. xanthophloea	85.5	0.3
Total	**29117.41**	**100%**

Further as shown in Figure 2 *E. divinorum* has significant cover in eastern and southern parts of the conservancy. Additionally, it is found in areas along deep valleys and thus in areas in low elevation and deep black cotton soils. Northern sector of the conservancy has less cover of *E. divinorum* hence highly dominated by *A. drepanolobium*, open grassland and mixed bushland towards the north-west direction.

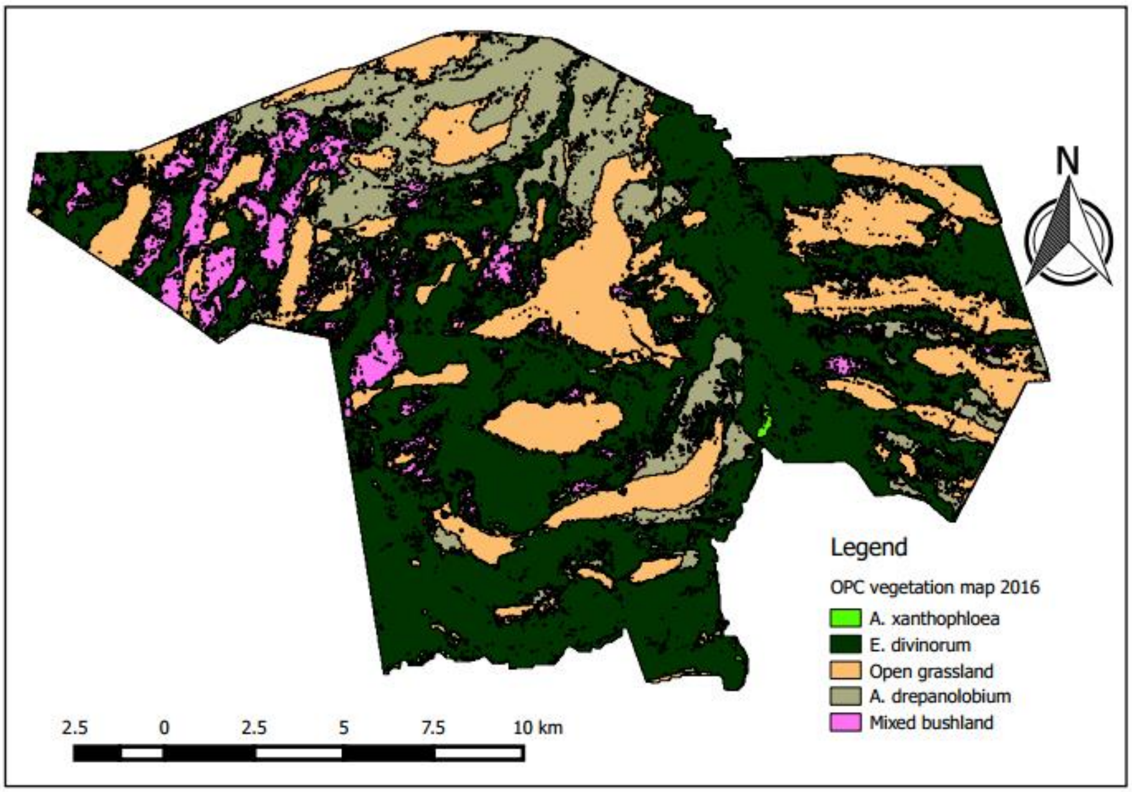


Figure 2: Vegetation cover map of Ol Pejeta Conservancy in 2016

4.1.1 Land Cover Changes with Reference to *E. divinorum*

Vegetation classes produced from image analysis include Open grassland, *A. drepanolobium*, *E. divinorum*, mixed bushland and *A. Xanthophloea* a riverine vegetation. Users' local knowledge-based approach was used to achieve vegetation classes as mention above. However, in some images more classes were identified such as swamps and water bodies (area coverage <1%) which came to existence as a function of human intervention to provide more water for animals recently. Figure 3 shows *E. divinorum* cover in OPC in different years from 1987 to 2016.

Between the year(s) 1987 and 1995 the area of *E. divinorum* cover increased by 531.4ha while between years(s) 1995 and 2000 its cover increased by 806.9ha. Between 2000 and 2005 the area in *E. divinorum* cover increased by 1910.0ha whereas between 2005 and 2010 and between 2010 and 2016 *E. divinorum* increased by 5121.1ha and 2491.4ha respectively. This increase was gradual but increased after 2000 whilst the greatest change in cover was between 2005 and 2010 as shown in Figure 3. Statistical analysis using Mann Kendall test for trend analysis was performed which revealed a significant monotonic (upward) increase in *E. divinorum* cover from 1987 to 2016 (tau 1, $p < 0.01$).

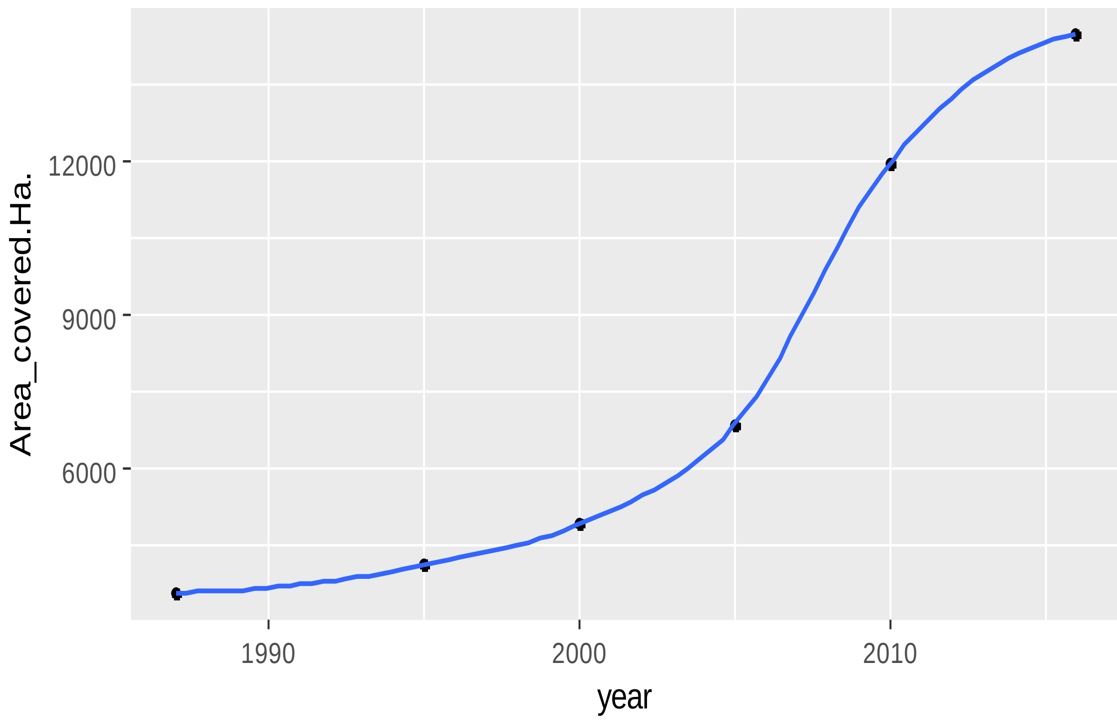


Figure 3: *Euclea divinorum* cover trends in OPC between 1987 and 2016

The rate of change between 1987 and 1995 annual increment was 66.4ha/Yr. whilst annual increment rate between 1995 and 2000 and between 2000 and 2005 were at 161.4ha/Yr and 381.6ha/Yr, respectively. Rates of change between 2005 and 2010 and between 2010 and 2016 were 1024.3ha/Yr and 415.2ha/Yr.

4.1.2 Overall Land Cover Changes in Ol Pejeta Conservancy

It was evident from the study that, some land cover classes continuously increased in cover while others showed decrease between the first two years and increased in cover in the subsequent years. Further, the Land cover class of riverine *A. xanthophloea* was nearly completely lost <1% cover by the year 2016. Overall vegetation cover changes and trends in OPC for the study period are as shown in Figure 4.

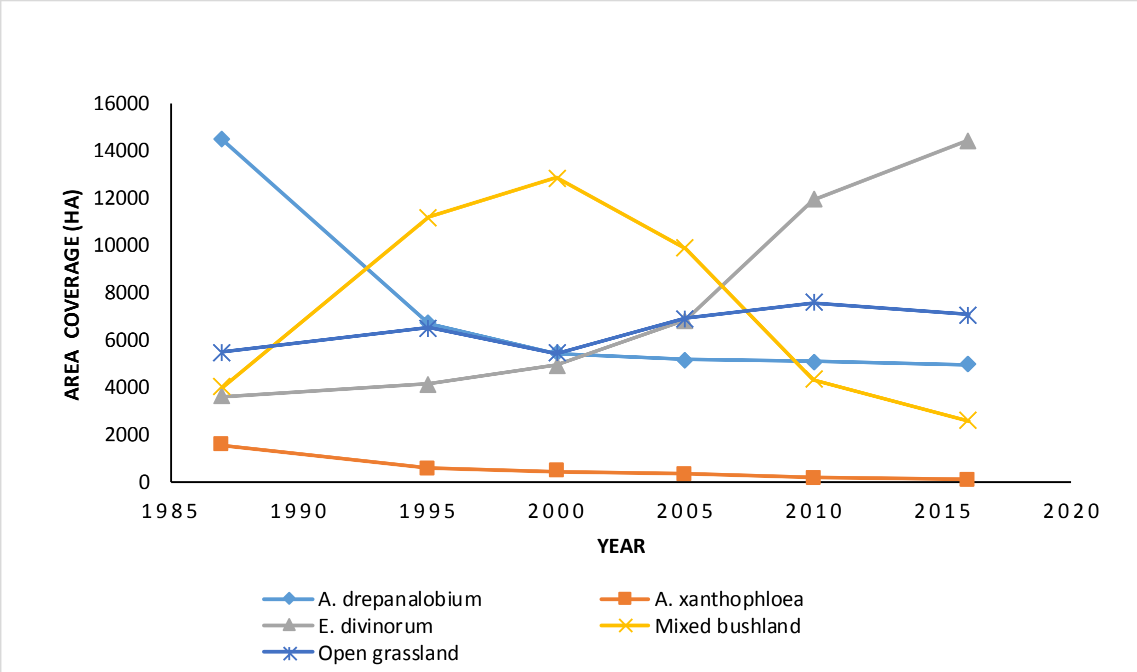
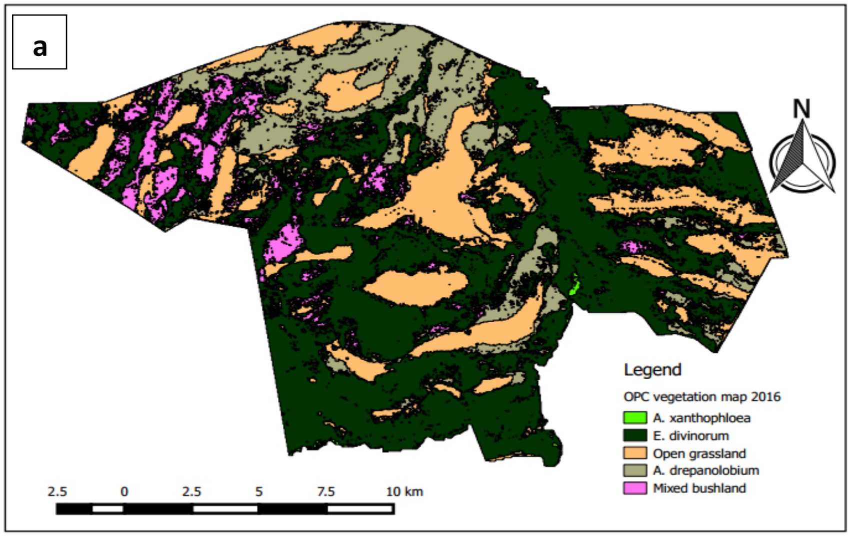


Figure 4: Vegetation cover types trends in Ol Pejeta Conservancy

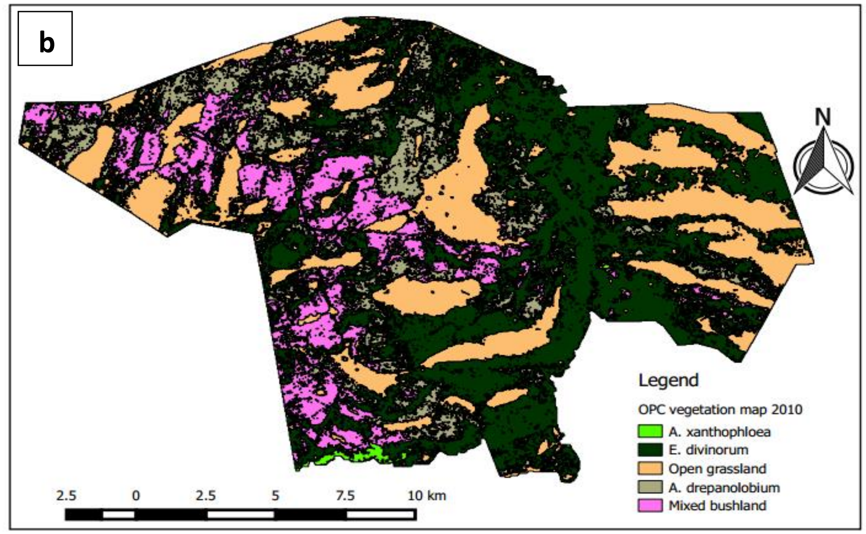
From the exploratory land cover changes in the Figure 4, it's evident that land cover changes are quite dynamic in the sense that there is increase or decrease in certain land cover classes or continuous decrease/increase in other class covers. Particularly, *E. divinorum* has increased in cover throughout the study period whereas *A. drepanolobium* and *A. xanthophloea* cover decreased in cover throughout the study period. On the other hand, Open grassland, and mixed bushland either increased or decreased in cover in different time periods. Open grassland class, between 1987 and 1995 there was increase in cover +1025.6ha followed by a decrease between 1995 and 2000 by -1060.5ha and an increase in between 2000 and 2005 by +1665.4ha, between 2005 and 2010 645.1ha and finally a decrease between 2010 and 2016 by -511.0ha. Overall there was an increase in open grass cover in the entire study period 1987-2016 by +1558.7ha.

Another class cover under consideration is the mixed bushland, this vegetation class increased in cover between 1987 and 1995 and between 1995 and 2000 by 7184.7ha and 1691.54ha, respectively followed by decrease from 2000-2005 by -3001.3ha, 2005-2010 a decrease by -5556.7ha and a further decrease between 2010 and 2016 by 1747.9ha. However, there was an overall decrease in class cover in the entire study period by 1429.7ha. In proportions, different land cover classes were as reported in different years. In the year 1987 *A. drepanolobium* covered 49.7%, Open grassland 18.9%, mixed bushland at 13.8% followed closely by *E. divinorum* at 12.4% and least *A. xanthopholea* at 5.3%. in the year 1995 different landcover proportions were as follows mixed bushland at 38.4%, *A. drepanolobium* at 23.1%, Open grassland at 22.4%, *E. divinorum* at 14.2% and finally *A. xanthophloea* at 1.92%.

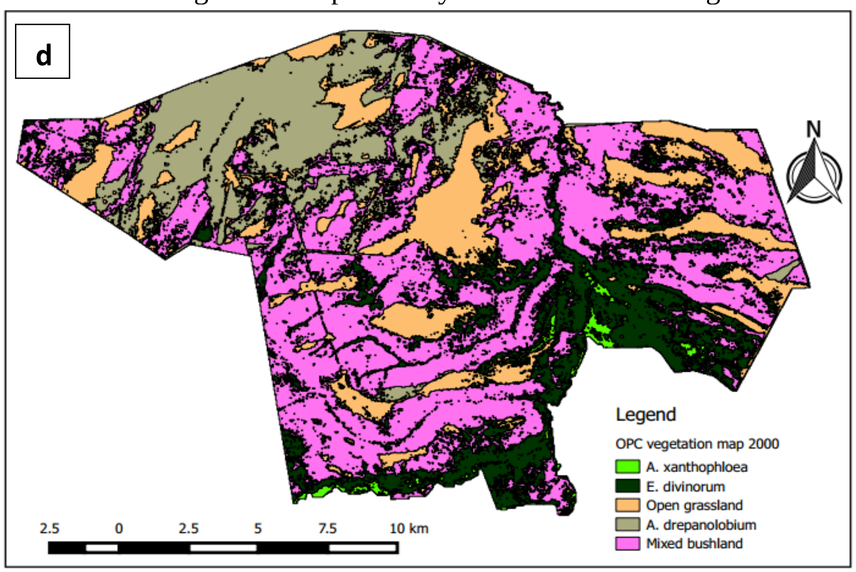
In the year 2000 cover proportions were as follows: mixed bushland covered 44.2%, Open grassland covered 18.7% followed closely by *A. drepanolobium* at 18.6%, *E. divinorum* covered 16.9% while *A. xanthophloea* came distance at 1.5%. In the year 2005 Mixed bushland covered 33.9%, Open grassland at 23.8% followed closely by *E. divinorum* at 23.5%, *A. drepanolobium* at 17.8% and finally *A. xanthoploea* 1.0%. In 2010 cover proportions were as follows *E. divinorum* at 41.1%, followed by Open grassland at 26.0%, *A. drepanolobium* at 17.5%, mixed bushland at 14.9% and finally *A. xanthophloea* at 0.5%. Lastly, the year 2016 cover proportions were as follows: *E. divinorum* at 49.6%, Open grassland at 24.2%, *A. drepanolobium* at 17.0%, mixed bushland 8.8% and finally *A. xanthophloea* at 0.3% as such due to insignificant coverage as omitted in species diversity and composition survey. The different land cover in various years are shown in Figure 5 ab,c,d,e and f.



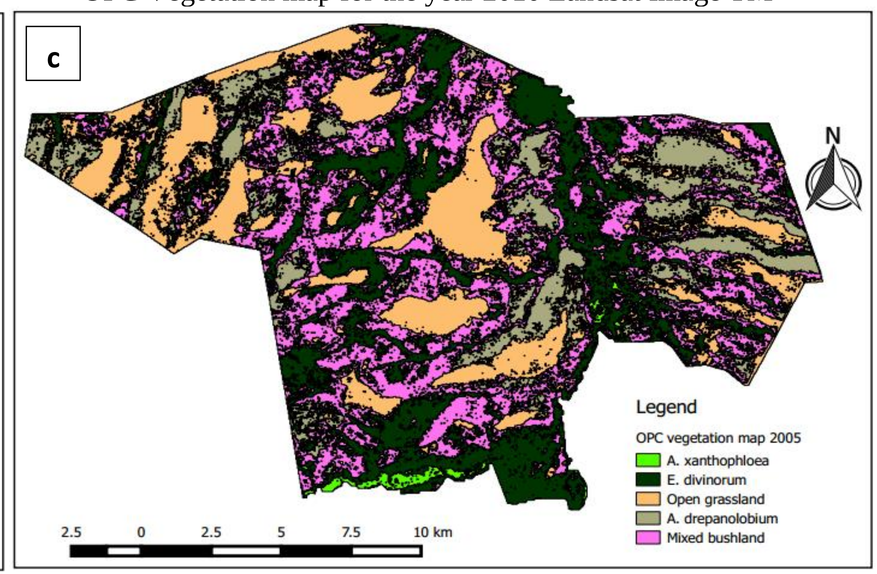
OPC Vegetation map for the year 2016 Landsat image OLI



OPC Vegetation map for the year 2010 Landsat Image TM

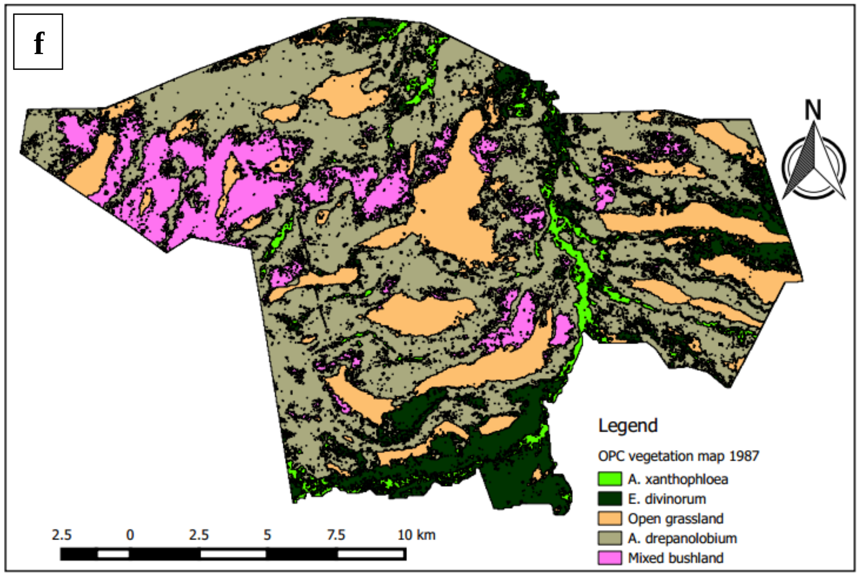


OPC Vegetation map for the year 2000 Landsat image TM

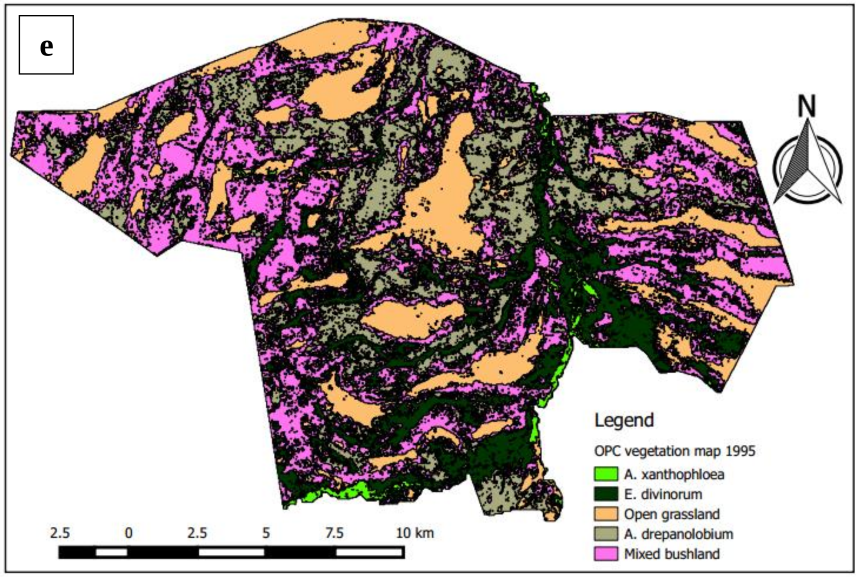


OPC Vegetation map for the year 2005 Landsat Image ETM+

Figure 5a: OPC vegetation maps for 2000, 2005, 2010 and 2016



OPC Vegetation map for the year 1987 Landsat Image TM



OPC Vegetation map for the year 1995 Landsat Image TM

Figure 5b: OPC Vegetation maps for 1987 and 1995

4.2. Topographic Features Attributable to Encroachment Patterns by *E. divinorum*

The results showed that areas with high NDVI value (NDVI maps 1987, 1995 2005 and 2016) were those along deep channels and valleys ideally areas in low elevation were covered by dense *E. divinorum* as in Figure 6 a,b,c and d. this was possible given that the images classified were acquired during dry season hence possible to detect the evergreen *E. divinorum* dominated habitat. NDVI results revealed that in 1987, areas close to the river creating a thin band were covered by evergreen *E. divinorum* whilst in the year 2000, the species spread outwards notably in the southern and southern eastern parts. In the year 2005, *E. divinorum* spread further especially in low elevated areas (deep valleys) in the entire southern and eastern as well in some parts in the northern region. In the year 2016, *E. divinorum* spread further towards the northern western parts in the conversancy predominantly in the low elevated areas.

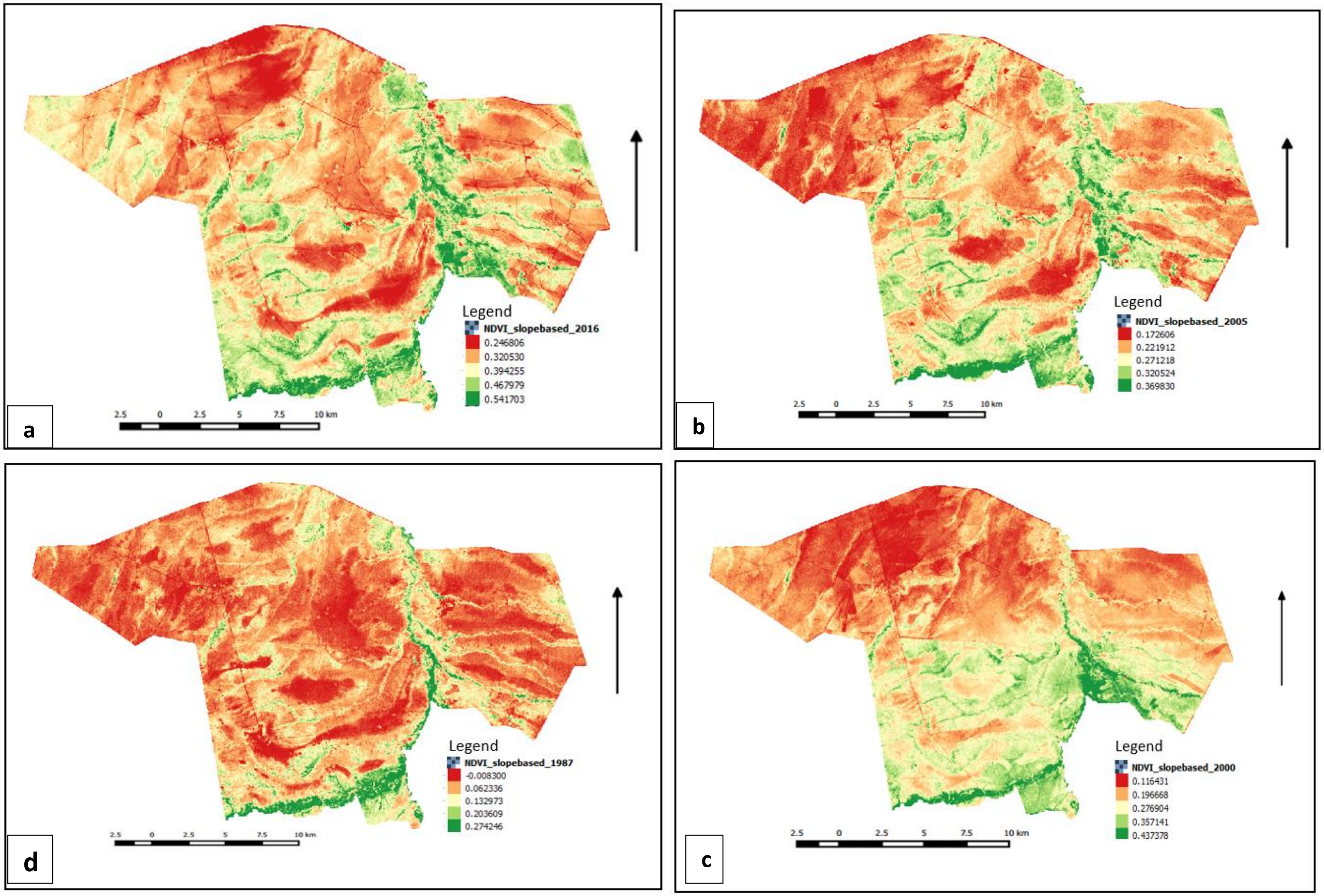


Figure 6: NDVI map for 1987, 2000, 2005 and 2016

4.2.1 Contours and Elevation Overlaid on Vegetation Map

Overlay of both topographic features, which include contours and elevation (observable features) revealed that areas below 1800m above sea level (ASL) were covered by *E. divinorum* as in Figure 7.a, b, c and d. Additionally, topology analysing using DEM and contours revealed the lowest point at about 1762.48 m ASL and highest point at about 1917.64m ASL translating to a range in altitude of 155.16m in the conservancy. Further, overlay of both topographic features, which include contours and elevation (observable features) on vegetation map of 2016 showed that areas below 1800m above sea level (ASL) were covered by *E. divinorum*. However, this encroachment appears to be expanding towards areas even at higher altitudes than 1800m ASL.

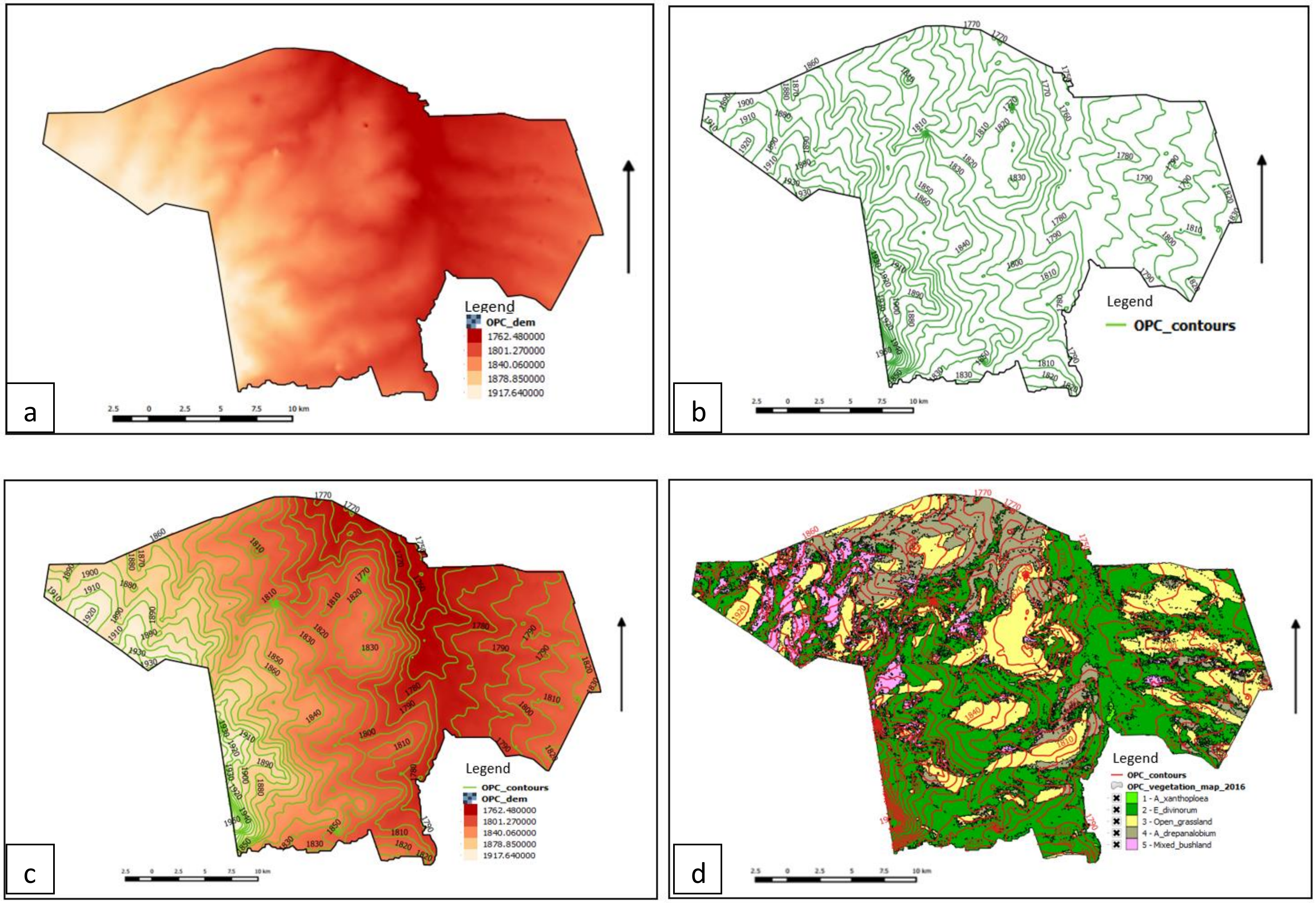


Figure 7: DEM, contours and vegetation overlays for OPC

4.3 Wildlife Survey for Diversity and Richness Assessment

4.3.1 Species Richness across the Habitat Types

In the four habitat types the animal taxa recorded were as follows; *A. drepanolobium* dominated habitat had higher species richness with 23 animal species followed by Open grassland with 21 species, *E. divinorum* dominated habitat had 19 species while mixed bushland had 15 species (refer animal species list in the appendix). Figure 8 shows species richness across four habitat types in OPC.

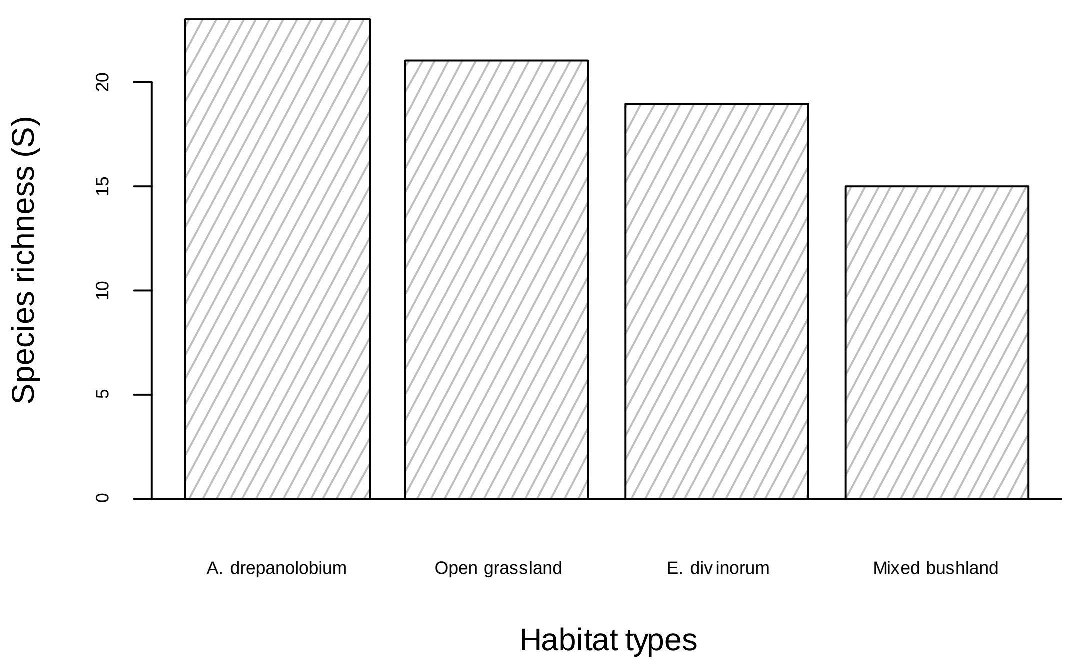


Figure 8: Species richness across four habitats in OPC

4.3.2 Species Dominance and Simpson's Dominance Index (D)

Species dominance, defined as most conspicuous and abundant species, was also compared across the four habitat types. Plains zebra was the most dominance animal species in Open grassland, Mixed bushland and *A. drepanolobium* dominated habitat. Simpsons species dominance (D) was higher in Open grassland (D= 0.334) followed closely by mixed bushland (D= 0.302), followed by *A. drepenolobium* dominated habitats at (D= 0.197) and lastly *E. divinorum* dominated had (D= 0.154). Graphically Simpson's dominance index is as shown in Figure 9.

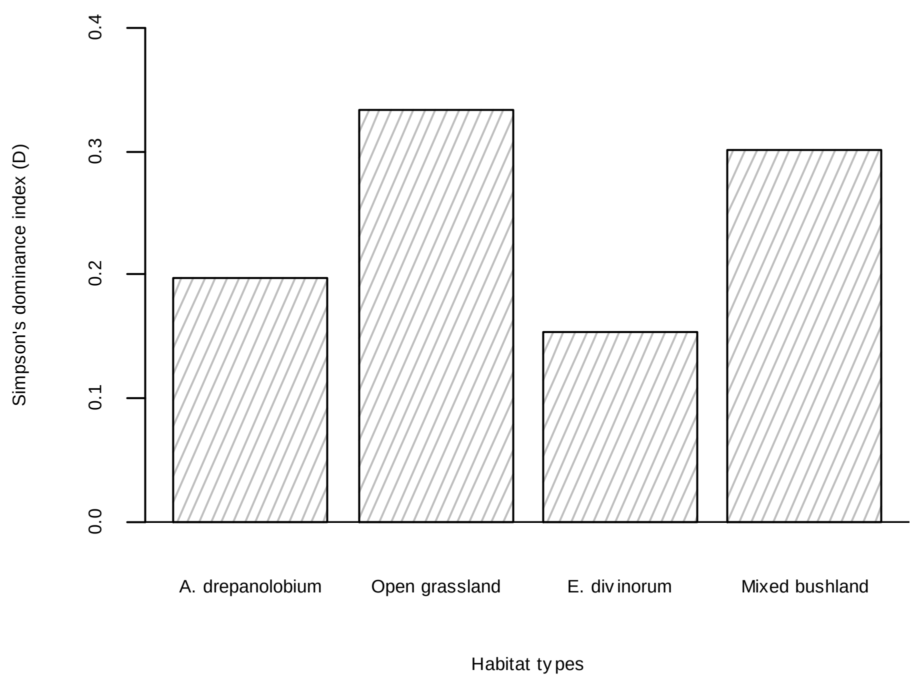


Figure 9: Species dominance across four habitats in OPC

4.3.3 Species Diversity and Evenness

Animal species diversity (H) was higher in *E. divinorum* dominated habitats at 2.291, *A. drepanolobium* dominated habitats had 2.058, Mixed bushland at 1.728 and Open grassland with least index value of 1.715 as shown in Figure 10 (a). Evenness (H/S) was highest in area under *E. divinorum* at 0.5201, followed by mixed bushland at 0.3751, then closed by *A. drepanolobium* at 0.3404 and finally Open grassland at 0.2647 as shown in figure 10 (b).

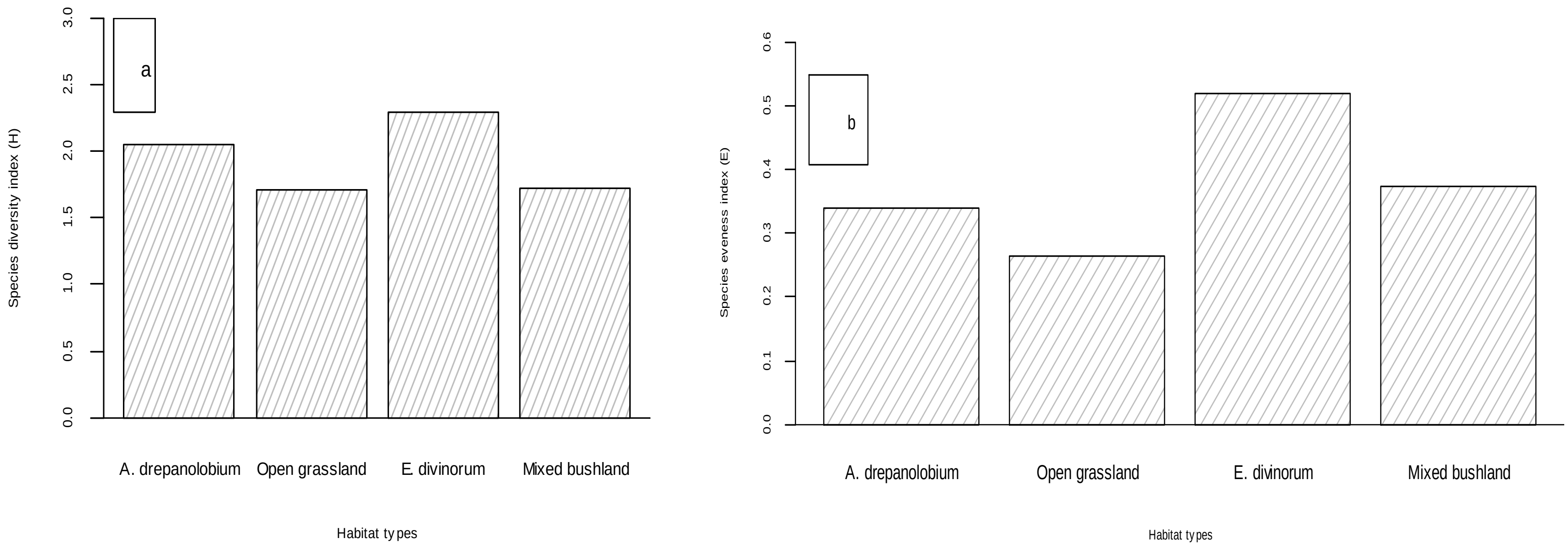


Figure 10: Species diversity index (a) and species evenness index (b) across four habitats in OPC

4.3.4 Species Composition

Hierarchical Cluster Analysis (HCA) compared species composition across the four habitat types namely *A. drepanolobium* dominated habitat, *E. divinorum* dominated habitat, Mixed bushland and Open Grassland. *E. divinorum* dominated habitat and mixed bushland habitats shared 45% similarity in species composition. This implies that 45% of species found in both mixed bushland *E. divinorum* dominated habitat were recorded in both habitats. Further, *E. divinorum* dominated habitat and Mixed bushland shared 39% similarity in species composition with *A. drepanolobium* dominated habitat. On the other hand, open grassland habitat shared 27% similarity in species composition with three habitat types namely, *E. divinorum* dominated habitat, *A. drepanolobium* dominated habitat and mixed bushland as shown in the Figure 11.

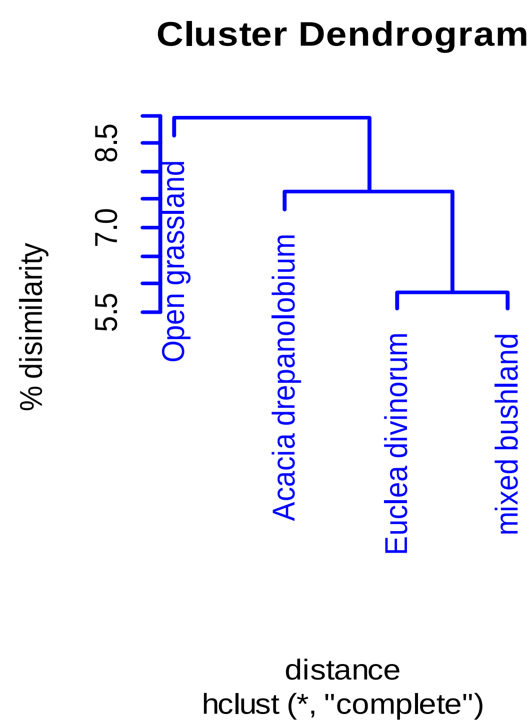


Figure 11: Dendrogram showing species composition

4.4 Habitat Preference or Avoidance by Feeding Guilds in Ol Pejeta Conservancy

Carnivores showed highest preference for *A. drepanolobium* dominated habitat (D=0.469) followed by Open grassland (D=0.327) whilst they least preferred Mixed bushland (D=0.066) habitats. On the other hand, *E. divinorum* dominated habitat was the most avoided habitat by carnivores (D= -0.698) s shown in Figure 12a. Grazers showed great preference for open grasslands (D=0.773) and *A. drepanolobium* dominated habitat (D=0.040). However, *E. divnorum* dominated habitat (D= -0.917) and Mixed bushland D= -0.192) habitats were avoided as shown in Figure 12b.

Browsers showed preference for *A. drepanolobium* dominated habitat (D= 0.674) and Mixed bushland habitat (D=0.175) but avoided both *E. divinorum* dominated habitat (D= -0.673) and Open grasslands (D= -0.116) dominated habitats. Finally, mixed feeders preferred habitats dominated by Open grassland (D=0.688) and *A. drepanolobium* dominated habitat (D=0.523) but avoided *E. divinorum* dominated habitat (D= -0.858) and Mixed bushland(D=-0.420) dominated habitats as shown in Figure 12d. Overall habitat preference or avoidance across the four habitats by all feeding guilds in OPC as shown in Figure 12.

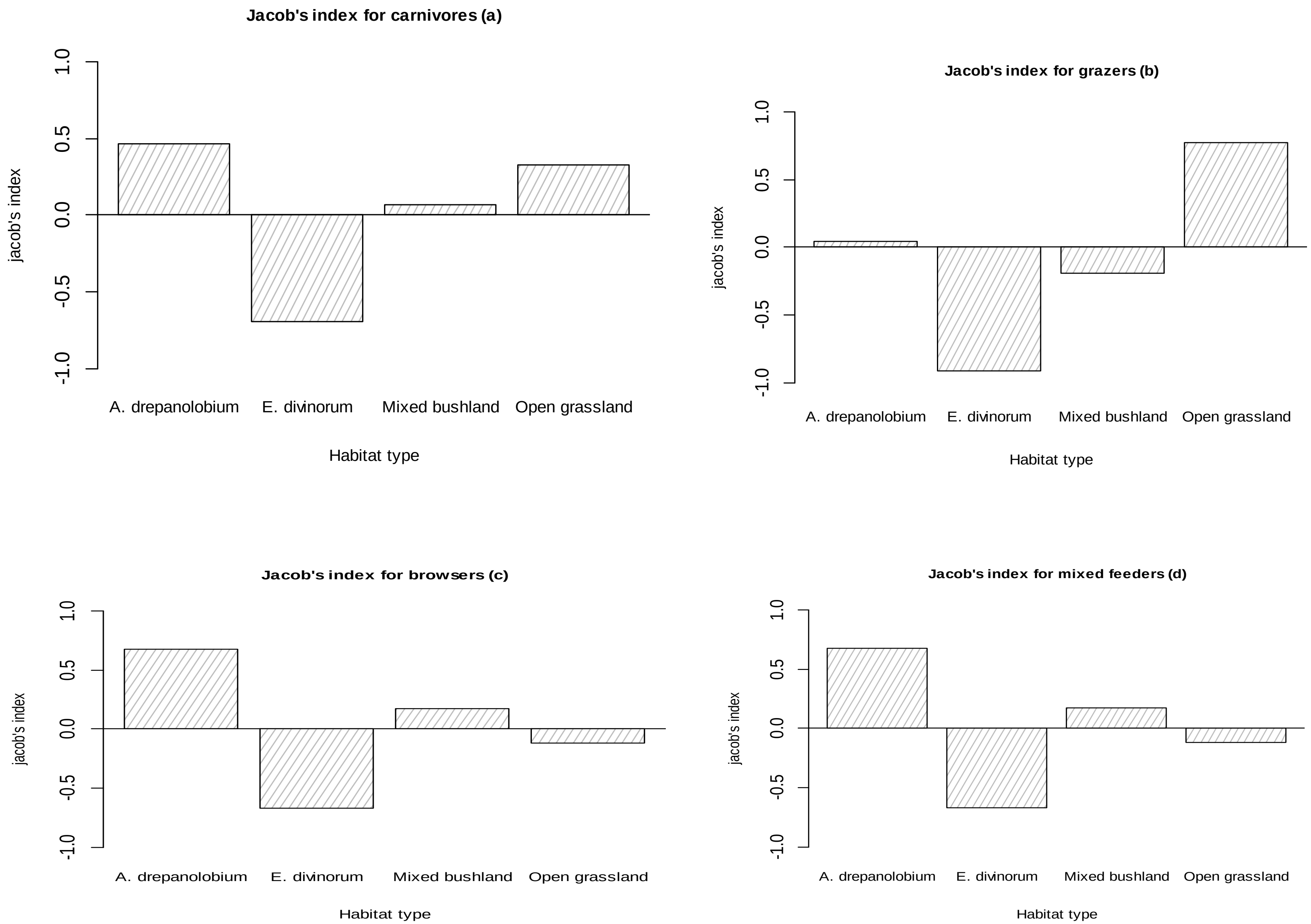


Figure 12: Habitat preference or avoidance (Jacobs' Index) for four feeding guilds in OPC

When a mean preference or avoidance of all habitats by various feeding guilds in OPC was computed, *E. divinorum* dominated habitat and mixed bushland were avoided by all guilds, however *E. divinorum* dominated habitat was significantly avoided ($t_1=2.253$, $d.f=3$, $p=<0.01$) than mixed bushland ($t_1=2.353$, $d.f=3$, $p=0.27$). On the other hand, *A. drepanolobium* dominated habitat and Open grassland were both preferred by all guilds, however, *A. drepanolobium* dominated habitat were significantly preferred ($t_1=2.353$, $d.f= 3$, $p =0.03$) compared to open grasslands ($t_1=2.353$, $d.f=3$, $p=0.06$) as shown in Figure 13.

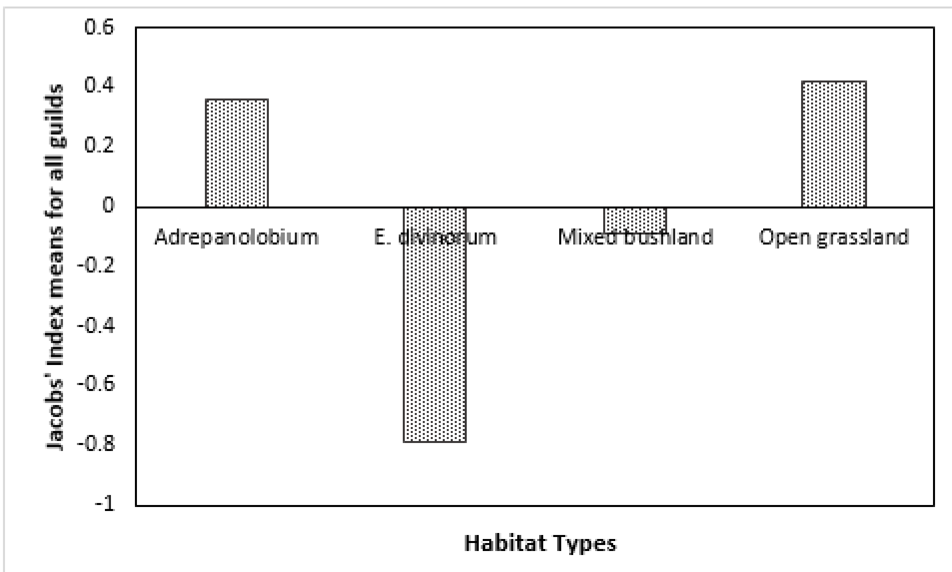


Figure 13: Means for Jacobs index across the habitat types among feeding guilds in OPC

When ranked on either preference or avoidance, open grassland habitat was the most prefered followed by *A. drepanolobium* dominated habitat while the most avoided habitat was *E. divinorum* dominated habitat but mixed bushland dominated habitat was randomly selected as shown in the Table 3.

Table 3. Ranking of Preference or avoidance of habitats by all feeding guilds based on Jacobs index means

Habitat Type	Jacobs Index(mean)	Rank	conclusion	P value for avoidance or preference
A. drepanolobium	0.359183	2	preferred	0.03
E. divinorum	-0.78668	-2	Avoided	0.01
Mixed bushland	-0.09283	-1	Randomly selected	0.27
Open grassland	0.418201	1	Preferred	0.06

Further, analysis revealed that Jacobs index mean when compared for significance avoidance or preference across all the habitat types where, *E. divinorum* dominated habitat was significantly ($F_{(3,12)}$ *15.268, p= <0.01*) avoided by all guilds while *A. drepanolobium* dominated habitat was significantly preferred by all feeding guilds as shown in the Figure 14. Further, Tukey Honestly Significance Difference test revealed that *E. divinorum* Jacobs index mean was significantly smaller than all other means of other habitat types as shown in Figure 14.

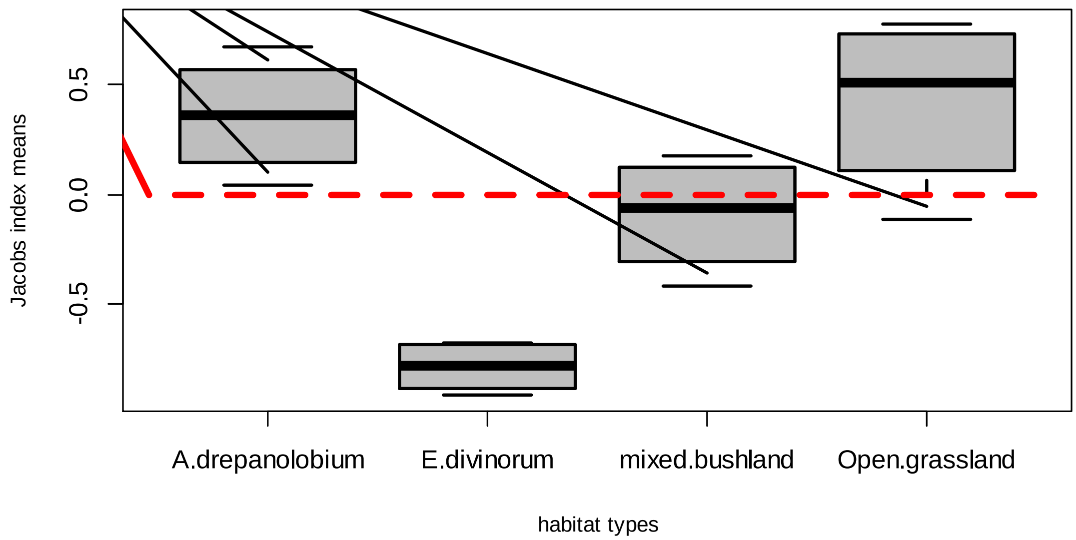


Figure 14: Means of Jacobs' Index across the four habitats in OPC

4.5 Interpretation

4.5.1 *Euclea divinorum* Dominated Habitat and Other Habitat Type Cover Changes

The findings of this study revealed that *Euclea divinorum* dominated habitats spread spatially in OPC covering nearly half of the entire conservation area by 2016. This significant increment in areas covered by *E. divinorum* dominated habitats over time implies that, other habitat cover types in OPC have been reduced/exterminated due to such increase in coverage by the species under consideration. Notably, *A. drepanolobium*, mixed bushland and *A. xanthophloea* dominated habitat are the most vulnerable to encroachment by *E. divinorum*. The increment in cover by *E. divinorum* was gradual, however after sometime the changes in its cover increased sharply. The patterns exhibited by spread of *E. divinorum* in this study are consistent with Skellam's (1951) diffusion model for invasive species whereby at the start there is low recruitment rates but over time the recruitment rate increases consequently and cover increases exponentially (encroachment from infested zones to transition zones and finally establishment in the un-infested zone).

Whilst there has been increase in cover by *E. divinorum*, on the other hand *A. xanthophloea* and *A. drepanolobium* have also reduced in coverage notably. As these dynamics in cover changes take place, other habitats have increased and decreased over the entire study period as for the case of mixed bushland majorly composed of other woody species such as *Scutia myrtina, Rhamnus staddo, Euclea divinorum, Acacia drepanolobium, Rhus natalensis and Carissa edulis* at no relative abundance. Remarkably, *E. divinorum* dominated habitat has been spreading in to vacant niches in habitat with low densities or areas devoid of the encroaching species (Wahungu et al., 2012). The finding of this study is in tandem with other research done in OPC where they reported increase in spatial coverage by *E. divinorum* dominated habitat (wahungu et al., 2012) though they reported that there were no significant in increment in cover by *E. divinorum* dominated habitat. Given that the woody species under consideration is unpalatable to both wild and domestics animals Smith and Goodman (1987), there is a potential of affecting their resource base indirectly by augmenting loss of resources through replacement (extermination of pasture biomass majorly grass).

Disturbances in savannah ecosystems have been mooted as possible driver for changes in savannah landscapes (Van Langevelde et al., 2003). These disturbances can range from human induced land cover changes such as prescribed burning, climatic induced factors such as droughts and rainfall to herbivory and pastoralism (Jeltsch, Weber, & Grimm, 2000; Van Langevelde et al., 2003). As such in OPC, where there is an increasing population of elephants, giraffes and black rhinos (personal communication, Bernard Chira) their herbivory (with preference to *A. drepanolobium*) impact is giving *E. divinorum* dominated habitat an advantage over *A. drepanolobium* dominated habitat with regards to their success in re-establishment.

Earlier research work conducted in the conservancy reported that there was high levels of damage (herbivory) to the *A. drepanolobium* dominated habitat whose net reduction can potentially explain the encroachment by *E. divinorum* dominated habitat (Birkett & Stevens-Woods, 2005). As a result of herbivory *E. divinorum* has been taking over areas previously dominated by *A. drepanolobium*, Mixed bushland and *A. xanthophloea* Grazing management systems/regimes such as over stocking can lead to degradation of the ecosystems functions thus altering grass-woody interactions. This kind of land management systems can potentially suppress grass biomass thus facilitate encroachment by woody species in arid and savannah ecosystems however there is paucity of information on cattle ranching augments encroachment in OPC.

Herbivory effect especially by mixed feeders can influence heterogeneity of savannah floral composition. This holds true if the woody species in question are palatable. According to Wahungu et al., (2012) mega herbivores such as elephants have the ability to open up bushy habitats and regulate woody species density thus minimize net effect of encroachment. In OPC, there are low levels *E. divinorum* dominated habitat damage owing to the fact that the species is unpalatable (Smith & Goodman, 1987). Elephant's herbivory net effect on *E. divinorum* plant species is insignificant hence may not contribute significantly to opening up of habitats under *E. divinorum* dominated habitat. This observation is consistent with other work done in Seregeti that elephants herbivory had no significant effect on *E. divinorum* plant species (Sharam et al., 2006).

In summary, elephants can suppress or open up closed woody vegetation especially if composed of palatable species whilst on the other hand pure grazers such as cattle can suppress grass hence alter its competitiveness with other plant species. Such interactions coupled with rainfall and other disturbances have potential to augment encroachment by woody species as well as determine dominant plant cover (Accatino, Vezzoli, Donzelli, & Scholes, 2010). In OPC, insignificant net effect of herbivory on *E. divinorum* is favouring its recruitment at the expense of other palatable woody species thus colonising vacant niches.

These changes in structure of savannah and semi-arid ecosystems from grass to increased bushy or woody species has remained a subject of debate. As such, possible theories and supposition have been postulated to elucidate this phenomenon as observed over time in savannah ecosystems. Climate change, high levels of herbivory, changing fire regimes (fire severity, duration and frequency) according to Sankaran et al. (2004), changes in competitiveness of grass, seed dispersal by animals and combination of all these factors have been suggested to be responsible for encroachment (Van Auken, 2000; Herrmann, Anyamba, & Tucker, 2005; Scanlon et al., 2005). According to Wahungu et al. (2012) prescribed fire in OPC was adopted to curb spread of *E. divinorum* dominated habitat however, the net effect had insignificant impact instead exacerbated spread of encroacher species.

Introduction of cattle (pure grazers especially in large numbers) in grassland ecosystems has been cited as the major driver of encroachment (Van Auken, 2000). However, relatively low herbivory pressure can be tolerated by plants without conspicuous changes in plant productivity, biomass reproduction but higher pressure can affect these factors. In OPC, where there are mixed feeders, browsers and pure grazers, there is a potential that their herbivory effect has benefited spread of *E. divinorum* plant species over time. However, the role of cattle in their indirect facilitation of encroachment in OPC is a question for ecological research.

4.5.2 Topographic Features Attributable to Encroachment Patterns by *E. divinorum*

Topographic factors such as slope and elevation can influence micro climates and potentially determine spatial extent of encroachment. Digital elevation model (DEM) map and contours overlaid on vegetation map of 2016 showed that areas along deep valleys are covered by *E. divinorum* dominated habitats a finding in tandem with Wahungu et al., (2012). Further, slope based Normalized Difference Vegetation Index (NDVI) revealed that *E. divinorum* initially infested zones as deep valleys and consequently spreading outward through transition zones to uninfested zones. Additionally, considerably a large portion under *E. divinorum* dominated habitat is below 1800m above sea level a suggestion that the encroacher species may prefer areas up to certain altitude. Nevertheless, there are areas outside the recorded altitude infested by *E. divinorum* hence edaphic factors may be responsible for woody species encroachment an indication of topology influencing micro climate and consequently plants species distribution indirectly.

Micro topographic feature influences vegetation cover, distribution and even species present (Ma et al., 2010). Relief and topographic variables such as slope, aspect and elevation can exert site specific micro climates hence affect landcover in some area (Zhao, Yang, & Zhou, 2010). This is in consistent with observed *E. divinorum* encroachment patterns and cover. In this study, this changes in elevation and slope indirectly affect net effect of solar radiation as such influencing soil temperature, near surface soil temperature and soil moisture (Bennie et al., 2008). These differences on earth surface net effect is manifested in form on vegetation structure, distribution and growth (Bennie et al., 2008; Hofer et al., 2008; Zhao et al., 2010). The differences in topographic feature also influences water infiltration, run off, erosion, seed migration and other debris (Jiao et al., 2009). Elevation at regional scales constrained vegetation distribution (Moeslund et al., 2013) which is true for the case on bush encroachment by *E. divinorum* in OPC where is mainly in low elevations. However, it is important recognising there are other factors contributing significantly towards these vegetation distribution types such as soil factors through not examined in this study.

4.5.3 Species Richness, Dominance, Diversity and Evenness

Wild animals utilise available habitats differently whereby some may be source of food, others explored to find breeding site while others as shelter/roosting or avoid perceived risks (Riginos & Grace, 2008). Herbivores are said to exist in "landscape of fear" (Laundre, Hernandez, & Altendorf, 2001; Brown & Kotler, 2004). This phenomenon implies that herbivore occurrence in an ecosystem complex balance between predator avoidance and maximizing forage quantity/quality. According to Sirami et al., (2009) savannah ecosystems are diverse in plant community structure and composition hence they support diverse fauna. As such, abiotic and biotic factors have the ability to dictate species assemblage and space use.

Animals have different preference for certain habitats (Sinclair, Mduma & Brashares, 2003) as a function of direct and indirect effects of prey availability, detectability/cover and resource availability (Ripple & Beschta, 2004). Such factors can explain reasons for higher diversity in *E. divinorum* dominated landscapes where animals can conceal from predators hence favourable conditions for mixed feeders and browsers in a landscape with both carnivores, herbivores and other mixed feeders. Further, higher species diversity in *E. divinorum* dominated habitats can be explained existence of resource rich patches in these types of habitat cover that are chiefly avoided but animals take risk to explore them. Spatial heterogeneity of ecosystem is important for maintenance of diverse wild animal species (Scholes & Archer, 1997) and acts as buffer against changes in resources availability for wild animals in era of climate change (Wang et al., 2006). On the other hand, woody encroachment, can potentially change this heterogeneity and affect grasses, rangeland productivity and herbivores wide range of wild animals (Scholes & Archer 1997).

In OPC, *E. divinorum* dominated habitat are characterised by nearly single plant species forming a thick canopy hence appealing to wild animals especially cryptic which require such habitats for camouflage such as small carnivores. This homogeneity of landscape results in reduction of the available range and less diversity chiefly affecting pure grazers. On the other hand, if encroachment exceeds certain threshold there is a possibility of affecting economy of the area indirectly. In this regard, visitor experience may be affected negatively due to reduced visibility (Gray & Bond, 2013; Marshall, Lovett, & White, 2008) of wild animals potentially affecting number of visitors who are major sources of revenue collected in form of conservation fees. Grass and herbaceous cover and biomass may be affected negatively thus affect sustainability of subsistence and commercial livestock production as well as wildlife conservation (Archer, 2003; Richter, Snyman, & Smit, 2001). OPC as a result has initiated integrated wildlife livestock management in order manage grass swards that are unfavourable for wild animal species.

4.5.4 Species Composition

Changes in habitat structure can influence species distribution and their space utilisation. Increase in woody cover can affect species herd size, distribution depending on feeding habits and overall behaviour of the species in question. In OPC, animal species composition was higher in some habitats suggesting that more animals in the conservancy utilised these types of habitat in search of the resource rich patches that are not assessed by many animals' due to the risks associated with them (Abrams, 2007). On the other hand, pure grazers are dominant in open landscapes due to reduced predation risks attributable to vegetation structure (Dupuch, Dill, & Magnan, 2009). As a result, significant loss of open grassland and *A. drepanolobium* dominated habitat could negatively affect those species which prefer such ecosystems in long term. In the context of OPC, pure browsers and grazers are at risk due to shrinking of their preferred habitats. This may potentially expose them to increased predation risks and decline in forage quality and quantity.

Habitat preference analysis has revealed contrary that *E. divinorum* habitats are avoided by all feeding guilds despite having highest species diversity. As such, higher percentage similarity in species composition between *E. divnorum* and mixed bushland is due to small proportion of various wildlife species utilising such areas in search for patches that are resource rewarding due to limited exploitation attributable to risks involved dictated by similarity in vegetation structure.

According to Smit and Prins (2015), increase in woody cover in savannah ecosystems will become less grassy, burn less frequently and grazers will be replaced by browsers and mixed feeders especially if the encroacher species is palatable though this may not be the case in OPC because the species in unpalatable. Additionally, species composition, feeding behaviour and space use will change greatly in response to changing ecosystems in order to adapt to new vegetation structure. As a function of changing vegetation structure and composition, wild animals will respond through range shifts to other areas with favourable characteristics. For instance, ranging areas for affected guilds will increase in order to cope with changing ecosystems.

4.5.5 Habitat Preference or Avoidance by Feeding Guilds in Ol Pejeta Conservancy

Habitat quality determines species distribution and space used hence a primary concern for conservation efforts (Boyce et al., 2016). Animals use cues such as landscape structure, vegetation structure such as tree species phenology, resource availability (probability of encounter, quantity and quality), predator/parasite risks among others in order to explore a given habitat type. As such, these qualities in a given habitat determine habitat preference or avoidance by wild animals.

Habitat selection theory postulates that animal distributes and colonise habitat patches with highest fitness (Morris, 2003) hence this explains why *E. dvinorum* dominated habitat is highly avoided by all feeding guilds due to reduced fitness with regard to predator encounter rates, poor visibility and escape chances. A suggestion that *E. divinorum* dominated habitat is of poor quality in terms resources availability and high perceived predation risks (Brown & Kotler, 2004). In case of OPC bush encroachment has degraded ecosystem by reducing vegetation cover into a nearly homogenous cover composed of *E. divinorum* plant species resulting in to unfavourable habitat conditions. The study findings agree with other findings which suggest bush encroachment in an indication of ecosystems degradation (Van Auken, 2000; Tobler, 2003) manifested by its significant avoidance by feeding guilds in OPC. According to Dalle, Maass and Isselstein (2006) increase in woody cover results in reduction in grassland diversity hence loss of biodiversity, reduction in forage resources and consequently reduction in carrying capacity (artificial shrinkage of carrying capacity). From economic point of view especially in areas where ecotourism is highly relied on as source of revenue, increase in woody cover can potentially affect visitor viewing experience (Wigley et al., 2009).

CHAPTER FIVE

SUMMARY OF FINDINGS, CONCLUSIONS AND RECOMMENDATIONS

5.1 Summary of Findings

The study revealed that bush encroachment was evident in Ol Pejeta Conservancy whereby time series monitoring revealed that *A. drepanolobium* dominated landscapes where majorly affected by the encroaching species. It was also evident that areas along deep channels were fully dominated by dense *E. divinorum* species spreading outward with time. Similarly, all feeding guilds showed preference for *A. drepanolobium* dominated habitats and avoided *E. divinorum* dominated habitats despite higher in species diversity. Species composition was closest between Open grassland and *A. drepanolobium* dominated habitats.

5.2 Conclusions

The study concludes that *E. divinorum* dominated habitat increased spatially between 1987 and 2016 whilst exterminated other habitat cover types namely; *A. drepanolobium* dominated habitat, mixed bushland and *A. xanthophloea*. Further, it was cocnluded that encroachment initially started in deep valleys and over time spread to other areas covering nearly 49% of the entire conservation area. Further, that *E. divinorum* has over time reduced cover of other habitat types, hence potential threat to suitability of these ecosystems for all feeding guilds as revealed by the findings.

It was also concluded that, *E. divinorum* dominated habitats had higher species diversity and evenness compared with other habitat types in the conservancy. On the other hand, species richness and dominance were highest in open grassland habitats attributed to high number of pure grazers in OPC, that is, plains Zebra and Buffaloes which spend most of their time in open grassland. Species composition was closely similar in *E. divinorum* dominated habitat and mixed bushland attributable to their similar vegetation structure and cover whilst species composition in open grassland had highest dissimilarity from the rest. This is attributable to its uniform landcover type majorly grassland (key resource) preferentially attracting pure grazers and by extension mixed feeders. Significant differences exist in habitat preference or avoidance among feeding guilds in OPC. Preference among feeding guilds was highest for *A. drepanolobium* dominated habitat followed by Open grassland and mixed bushland selected randomly. Habitat avoidance was highest among feeding guilds for *E. divinorum* dominated habitat.

5.3 Recommendations

From the findings of this work several areas have emerged that requires further research to better understand dynamics of woody encroachment and resultant implications on species both fauna and flora composition and assemblage. As such, recommendations are in two-folds as follows:

a) Recommendations for management

i. Investigate potential negative effect on visitor experience focusing on regular visitors who may have noticed increase in *E. divinorum.*

ii. OPC should develop invasive/encroacher species management plan to control spread of *E. divinorum* species with focus on transition zones combining both mechanical and chemical control of encroaching species and further into already infested zones.

b) Recommendations for further research;

i. That there is need for a long-term study to understand dynamics of woody tree density per unit area with focus on already infested zone, transition zones and un-infested zones by *E. divinorum*.
ii. That the interactions of both livestock and wildlife are investigated and their influence on establishment of *E. divinorum* needs further research.
iii. Investigate impacts of bush encroachment by *E. divinorum* on grass production, biomass and diversity on long term basis.

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APPENDICES

Appendix 1: List of animal species detected in various habitat types

Species (common Name)	**Scientific Names**	**Habitat Types In OPC**			
		Acacia drepanolobium	***Euclea divinorum***	**mixed bushland**	**Open grassland**
Aardvark	*Orycteropus afer*	1	0	1	1
Baboon	*Olive Baboon*	1	1	1	1
Black Backed Jackal	*Canis mesomelas*	1	1	1	1
Black Rhino	*Diceros bicornis*	1	1	1	1
Buffalo	*Syncerus caffer*	1	1	1	1
Bush Duiker	*Sylvicapra grimmia abyssinicus*	1	1	0	0
Eland	*Taurotragus oryx*	1	1	1	1
Elephant	*Loxodonta africana*	1	1	1	1
Genet (*)	*Genetta tigrine*	1 *(*)*	0	0	0
Giraffe	*Giraffa reticulata*	1	1	1	1
Grants Gazelle	*Nanger granti*	1	0	0	1
Hare	*Lepus victoriae*	1	0	0	1
Hartebeest *(*)*	*Alcelaphus buselaphus*	0	0	0	1*(*)*
Impala	*Aepyceros melampus*	1	1	1	1
Leopard (*)	*Panthera pardus*	0	1 *(*)*	0	0
Lion	*Panthera leo*	1	1	0	1
Dwarf Mongoose (*)	*Helogale parvula*	0	0	0	1 *(*)*
Plains Zebra	*Equus quagga*	1	1	1	1
Serval Cat (*)	*Leptailurus serval*	1 *(*)*	0	0	0
Spotted Hyena	*Crocuta Crocuta*	1	1	1	1

Steenbok	*Raphicerus campestris*	1	1	0	0
Striped Hyena	*Hyaena hyaena*	1	1	0	1
Suni	*Neotragus moschatus*	0	0	1	0
Thompsons Gazelle (*)	*Eudorcas thomsonii*	0	0	0	1 (*)
Warthog	*Phacochoerus africanus*	1	1	1	1
Waterbuck	*Kobus ellipsiprymnus*	1	1	1	1
White-Rhino	*Ceratotherium simum*	0	1	0	0
White-tailed Mongoose	*Ichneumia albicauda*	1	0	1	1
Wildcat (*)	*Felis silvestris*	0	1 (*)	0	0
Zorilla (*)	*Ictonyx striatus*	1 (*)	0	0	0

Key: **1** denotes ***detection*** while **0** denotes ***non-detection***

***** Denotes ***detection*** only in one habitat type

Appendix 1: Research Licence permit

NATIONAL COMMISSION FOR SCIENCE, TECHNOLOGY AND INNOVATION

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Ref. No. **NACOSTI/P/18/45191/25231**

Date: **24th September, 2018**

Cyrus Mutunga Kavwele
Karatina University
P.O. Box 1957-10101
KARATINA.

RE: RESEARCH AUTHORIZATION

Following your application for authority to carry out research on ***"Impacts of bush encroachment by euclea divinorum on wildlife species diversity and composition in Ol Pejeta Conservancy in Laikipia, Kenya"*** I am pleased to inform you that you have been authorized to undertake research in **Laikipia County** for the period ending **20th September, 2019.**

You are advised to report to **the County Commissioner and the County Director of Education, Laikipia 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.

DR. MOSES RUGUTT, PHD, OGW
DIRECTOR GENERAL/CEO

Copy to:

The County Commissioner
Laikipia County.

The County Director of Education
Laikipia County.