VEGETATION AND HABITAT USE AMONG LARGE HERBIVORES IN KIMANA WILDLIFE SANCTUARY, AMBOSELI ECOSYSTEM, KENYA

BY

KIPKOSGEI LYNN JEPKOECH

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DECLARATION

Declaration by the Candidate

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Kipkosgei Lynn Jepkoech …………………… Date ……………………

(SES/PGB/02/10)

Declaration by Supervisors

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

Doctor Shem M. Mwasi ……………………….. Date ………………….

(The School for Field Studies, Centre for Wildlife Management, Kenya)

Professor John W. Kiringe ………………...... Date ………………..

(The School for Field Studies, Centre for Wildlife Management, Kenya)

DEDICATION

I dedicate this research thesis to my supervisors and family; my husband David and my children Deborah, Norah and Collins.

ABSTRACT

Understanding the composition and structure of vegetation is important in conservation and management of large herbivores worldwide. The aim of the study was to obtain information on the composition and structure of vegetation and habitat use among large herbivores in Kimana Wildlife Sanctuary (KWS), Amboseli ecosystem, Kenya. The objectives of the study were to characterize and to classify vegetation in KWS, to determine the structure and composition of vegetation in KWS, to determine the population density of large herbivores in KWS and to determine the seasonal variations in habitat selectivity, niche breadth and habitat overlap indices among large herbivores in KWS. Five vegetation types were classified and mapped using remote sensing but seven vegetation types were further classified using physiognomic features and dominant species. Data on vegetation and large herbivore were collected between May, 2012 and December, 2012 during both the dry and wet seasons using Stratified Research Design. Random and systematic sampling methods were used for sampling vegetation using Point–Centered Quarter method, Belt transect method, Descending Step Point method and Pasture Disc Meter method while random sampling method was used in large herbivore size determination. Variations were observed across vegetation types in woody species richness, diversity, evenness, similarity and Importance Value Index (IVI). Frequency of woody plant stems (dbh≥5cm) showed an inverted J-shaped structure in *Acacia tortilis* woodland, wooded grassland and sparse shrubland but a J-shaped structure in *Acacia xanthophloea* woodland. Frequencies of grass ecological categories differed significantly in all the vegetation types, with the exception of open grassland. Increaser II grass species; *C. dactylon*, *S. fimbriatus* and *H. schimperii* dominated in all the vegetation types, with the exception of wooded grassland, which was dominated by *P. stramineum*, an increaser I grass species. Grass standing crop biomass, grass basal cover, grass height and inter-tuft distance also varied across vegetation types. A total of 3,983 individuals of large herbivores were counted, 2,153 in the dry season and 1,830 in the wet season. Large herbivore overall density was 1.980 ± 0.236 and 1.803 ± 0.2156 during the dry and the wet seasons respectively. Plains zebras (*Equus quagga* Boddaert) were the densest species during the dry, 6.793 ± 0.871 and the wet season, 6.473 ± 0.848 and Cape elands significantly increased their population densities during the wet season. Larger herbivores showed greater selection for wooded grassland during the dry (≈71%) and the wet (≈85%) seasons. Niche breadth indices were low among the large herbivores during the dry (≈73%) and the wet (≈87%) seasons. Habitat overlap indices were high among the large herbivores during the dry (≈67%) and the wet (≈51%) seasons. In conclusion, the structure and composition of vegetation varied across and within vegetation types, cape elands significantly increased their population densities during the wet season and the large herbivores were highly selective able to specialize on a few habitat types, hence, they greatly overlapped in habitat use.

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CHAPTER ONE

INTRODUCTION

1.1 Background information

Savanna is an important component of vegetation on the earth surface that provides important resources including habitat and food to both animals and humans. Savanna vegetation covers approximately 18% of the earth's total land area (Michelakis *et al.,* 2014) and approximately 65% of total land area in Africa (Otieno *et al.,* 2005).

Savanna vegetation is generally characterized by a discontinuous cover of woody plants (trees & shrubs) intermixed with a continuous cover of herbaceous species (grasses and forbs) (van Langevelde *et al.,* 2003). However, there is existing evidence to show that the composition and structure of savanna vegetation can vary both spatially and temporarily across and within ecosystems due to factors such as climate, herbivory, fires, soils, herbivory and human activities (Gandiwa *et al.,* 2011; Gandiwa *et al.*, 2013; Zisanza-Gandiwa *et al.*, 2013). For instance, vegetation in Gonarezhou National Park (GNP), southeastern Zimbabwe have been shown to vary across different areas due to factors such as fires, soil types, herbivory and human activities (Zisanza-Gandiwa *et al.,* 2013).

Variability of savanna vegetation has been shown to be important in influencing diversity, abundance, distribution, habitat utilization and spatial relationships among large herbivores (Cromsigt & Olff, 2006; Cromisgt *et al.,* 2009; Mwasi *et al.,* 2013; Kleynhans *et al.,* 2010). In Africa, the importance of vegetation variability in supporting species coexistence through habitat partitioning has been demonstrated in many conservation areas (Mwangi & Western, 1998; Cromsigt & Olff, 2006; Cromsigt *et al.,* 2009; Mwasi *et al.,* 2013; Kleynhans *et al.,* 2010). In South Africa, for instance, multiple species of large herbivores were found to coexist through partition of diversity of habitats in Hluhluwe iMfolozi Park (Kleynhans *et al.,* 2010). Also larger ruminants and differently sized non-ruminants were found to coexist with ruminants of medium and smaller sizes through partitioning of habitats in Hluhluwe in iMfolozi Park (Cromsigt & Olff, 2006; Cromsigt *et al.,* 2009). Apart from vegetation variability, habitat partitioning among large herbivores in savanna ecosystems have been suggested to be determined by differences in herbivore body size, digestive physiology, seasonality and presence of megaherbivores (Cromsigt & Olff, 2006; Cromsigt *et al.,* 2009; Kleynhans *et al.,* 2010).

Although variability of savanna vegetation can facilitate species coexistence through habitat partitioning, species have been shown to overlap in their resource use, particularly among species of similar body sizes (Kleynhans *et al.,* 2010; Macandza *et al.,* 2012; Mwasi *et al.,* 2013; Owen-Smith *et al.,* 2015). Such overlaps often lead to important ecological processes such as competition or facilitation among species (Mudhusudan, 2004; Young *et al.,* 2005; Odadi *et al.,* 2011).

In Kenya, typical savanna vegetation occur within the Arid and Semi-Arid Lands (ASALs), which cover over 80% of the country's total land area (Musyoki *et al.,* 2012). The vegetation in this area is generally characterized by abundance of grasses of the species *Cenchrus ciliaris* L., *Pennisetum strameneum* Rich., *Penicum maximum* Jacq, *Themeda triandra Forssk, Cynodon dactylon L.* and *Enteropogon macrostachyus* Hochst. Ex. Rich (Mganga *et al.,* 2010; Kioko *et al.,* 2012; Mganga *et al.,* 2013; Mureithi *et al.,* 2014) and woody plants of the species *Acacia tortilis* (L.) Del., *Acacia xanthophloea* Benth., *Acacia drepanolobeum* Harms., *Salvadora persica*

L., *Balanites aegyptica* (L.) Del., *Balanites glabra* Mildbr & Schiecht and *Commiphora africana* A. Rich. (Dharani *et al.,* 2006; Lekoyiet, 2006; Mureithi *et al.,* 2014). However, like in similar ecosystems worldwide, the composition and structure of vegetations in ASALs of Kenya is highly variable due to effects of climate, topography, soils, fire, herbivory and human activities (Riginos & Grace, 2008; Shisanya *et al.,* 2011; Kioko *et al.,* 2012; Mureithi *et al.,* 2014; Kilavi, 2015).

The vegetation in ASALs in Kenya is important in providing numerous resources to about 25% of the nation's human population, mostly the pastoral and agro-pastoral communities (Ottichilo *et al.,* 2000; Macharia & Ekaya, 2005). The communities living in ASALs majorly relay and utilize plant resources for various purposes including fencing, construction, fuel provision, medicinal purposes, ornamentals and as a source of forage for over 75% of the country's livestock (Macharia & Ekaya, 2005; Kiringe & Okello, 2005; Okello *et al.,* 2015). The vegetation in Kenya's ASALs is also important in supporting the high abundance of the country's wildlife resources, which play significant roles in supporting the country's economic development. In the year 2011, for instance, it was estimated that wildlife resources accounted for 70% Gross Tourism Earnings (GTE), 25% GDP and 10% total Formal Sector Employment (FSE) (Government of Kenya (GoK), 2012).

Wildlife in the ASALs is often conserved in protected areas such as Amboseli National Park, Maasai Mara National Reserve and Tsavo National Parks. However, over 80% of wildlife in the country is conserved in communal lands bordering protected areas (GoK), 2012). In Amboseli ecosystem, for example, Amboseli National Park was delineated by the government in 1974 for protection of a high diversity of wildlife, but over 80% of the wildlife in the park relay on the surrounding

Maasai Group Ranches for migration and alternative feeding and breeding ground, especially, during the wet seasons (Okello *et al.,* 2011).

Despite their importance, ASALs have continued to experience land degradation, land use and land cover changes, land fragmentation, land subdivision and compression resulting from rapid human population growth, expansion of human settlements and intensification of land uses (Kioko & Okello, 2010; Symbua, 2013; Bhola *et al.,* 2013; Ogutu *et al.,* 2014). Land degradation in ASALs is attributed mainly to effects of overgrazing activities that causes changes in the functional characteristics of native vegetation (Macharia & Ekaya, 2005; Kioko *et al.,* 2012). Overgrazed areas are often characterized by unpalatable and annual herbaceous plant species, loss of woody vegetation and encroachment by bushes (Macharia & Ekaya, 2005; Lekoyiet, 2006; Kioko *et al.,* 2012). The dominance of unpalatable and annual grasses may result in low forage potentials, which may affect the ability of rangelands from supporting grazers (Young-Zhong *et al.,* 2005; Kioko *et al.,* 2012). Overgrazing can also reduce grass height and grass cover, which can have direct effects on ASAL soils by exposing the surface layer to wind, hence, increased soil erosion (Kioko *et al,* 2012). Furthermore, overgrazing can increase coarseness in surface soil, can reduce soil nitrogen and calcium and can lower soil pH (Young-Zhong *et al.,* 2005; Kioko *et al,* 2012).

As a result of land degradation and other threat factors such as land use changes, bush meat trade, poaching, human-wildlife conflicts, rising human population pressures, encroachment and loss of migratory corridors and dispersal areas, government policies, competition with livestock, economic and socio-cultural transformation, the population of wildlife in Kenya has declined significantly in both protected and non-protected areas (Bhola *et al.,* 2013; Ogutu *et al.,* 2013; Ogutu *et al.,* 2014). In a recent survey, for example, the population of large herbivores in eastern and western Kajiado was found to have declined by 67% between 1977 and 2011 (Ogutu *et al.,* 2014). This study, therefore, provide valuable information on vegetation and habitat use among large herbivores in KWS for the sustainable conservation of wildlife in Amboseli ecosystem.

1.2 Statement of the problem

In Amboseli ecosystem, rainfall patterns are seasonal (Altmann *et al*., 2002), thus, large herbivores migrate seasonally between protected areas such as Amboseli National Park and adjacent Maasai Group Ranches in search of forage, water and alternative breeding sites (Okello *et al.*, 2011).

The Maasai Group Ranches have been significant migratory corridors and dispersal areas for the protected areas in the ecosystem (Okello, 2012). However, due to increasing anthropogenic activities, these significant areas have been fragmented and wildlife habitats lost (Kihima & Nyamasyo, 2014). This situation often results to contraction of wildlife in dispersal areas, blockage of migratory corridors and subsequent insularization of conservation areas (Okello & Kioko, 2010; Okello, 2012; Mose *et al.*, 2012).

Kimana wildlife sanctuary (KWS) is one of the important conservation areas in Amboseli ecosystem that serves as a wildlife migratory corridor and a wet season concentration area for wildlife (Okello *et al.,* 2011). However, with increasing fragmentation and loss of wildlife habitats in the Group Ranches, wildlife tends to concentrate in the sanctuary during both the dry and the wet seasons. Livestock also depend on the sanctuary for grazing, especially during the dry season, when forage resources are scarce. The high concentration of wildlife in the sanctuary in all seasons is likely to cause continuous and/or heavy grazing, which may impact on vegetation with consequence on its structure and composition. Also, their impacts on vegetation may become intensified during the dry season, when forage resources are scarce (Muthoni *et al.,* 2014)

In addition to the effects of herbivores, the composition and structure of vegetation is likely to be influenced by other factors including climatic factors, edaphic factors and human activities, as it has been observed in other savanna areas of Africa (Gandiwa *et al.,* 2011; Gandiwa *et al.,* 2013; Zisanza-Gandiwa *et al.,* 2013) Variations in the composition and structure of vegetation both spatially and temporarily due to biotic and abiotic factors have conservation implication for wildlife management because it has the potential to influence diversity, abundance, distribution, habitat utilization and spatial relationships among large herbivores (Cromsigt & Olff, 2006; Cromsigt *et al.,* 2009; Kleynhans *et al.,* 2010). Therefore, this study aimed at understanding the composition and structure of vegetation and habitat use among large herbivores in KWS, Amboseli ecosystem.

1.3 Justification and significance of the study

Despite the importance of KWS in supporting wildlife resources in Amboseli ecosystem, the compositon and structure of its vegetation and habitat use among large herbivores remains poorly understood.

The composition and structure of vegetation in ASALs in Kenya is influenced by several factors such as fire, climate, soils, herbivory and human activities (Gachimbi, 2002; Western & Maitumo, 2004; Macharia & Ekaya, 2005; Riginos & Grace, 2008; Shisanya *et al*, 2011; Kilavi, 2015). Vegetation variability is important influencing species diversity, abundance, distribution, habitat utilization and spatial relationships

(Cromsigt & Olff, 2006; Cromsigt *et al.*, 2009; Kleynhans *et al.*, 2010), hence, it is important to understand the composition and structure of vegetation and habitat use among large herbivores in KWS, which is important for sustainable conservation of wildlife.

The study contributes to the general knowledge and understanding of the structure and composition of vegetation and habitat use among large herbivores during different rainfall seasons in KWS. It provides important information for restoration of degraded habitats in the sanctuary and also new information to add on the existing literature in this field of study for academic and research work. It provides insights on the suitability of the sanctuary as a seasonal migratory corridor and a dispersal area for wildlife occurring in Amboseli National Park and other neighboring protected areas. Finally, it will guide effective management of the sanctuary including conservation of its wildlife resources.

1.4 Objectives of the study

1.4.1 General objective

The general objective of the study was to obtain baseline information on the composition and structure of vegetation and habitat use among large herbivores in KWS, Amboseli ecosystem.

1.4.2 Specific objectives

The specific objectives for the study were;

1. To characterize and to classify vegetation in KWS
2. To determine the composition and structure of vegetation in KWS

3. To determine seasonal population sizes of large herbivores in KWS
4. To determine the seasonal variations in habitat selection, niche breadth and habitat overlap indices among large herbivores in KWS

1.4.3 Research questions

1. What are the major vegetation types in KWS?

2. What is the structure and composition of vegetation in KWS?

3. What is the composition and abundance of large herbivores in KWS?

4. To what extent do the large herbivores vary their habitat selectivity, niche breadth and habitat overlap during the dry and the wet seasons in KWS?

1.5 Scope of the study

The study focused on the structure and composition vegetation and habitat use among co-occurring large herbivores in KWS. The study was conducted over a period of eight months, during the dry (July to September) and the wet (October to December) seasons in the year 2012, which corresponds to the periods of minimum and maximum abundance of forage resources. The vegetation types studied were taken to represent the various habitat types available for large herbivore use.

CHAPTER TWO

LITERATURE REVIEW

2.1 Classification, characterization and mapping of vegetation communities in savanna ecosystems

Vegetation is the plant cover of the Earth and it is compost of different types of plant species growing in a very great diversity of assemblages. In different savanna regions, vegetation have been characterized, classified and mapped for different ecological purposes. Kleynhans *et al.*, (2010) classified the vegetation in HiP, South Africa into thicket, open woodland and forest.

In Kenya, vegetation has been characterized, classified and mapped mostly in conservation areas. In the former Narok District, Trump (1972) recognized 18 vegetation types of ecological significance. He also recognized 10 major climax vegetation types that included bamboo, forests, evergreen and semi-deciduous bushlands, acacia woodland, *Acacia commiphora* bushed grassland, wooded grassland and montane grasslands. Trump (1972) also derived nine minor vegetation types from the major vegetation types. The vegetation on the impeded drainage soils were separately mapped and classified into three vegetation types.

van Essen *et al.*, (2002) assessed woody vegetation at Ol Choro Oiroua Conservancy in Maasai Mara. They identified six plant communities including *Olea africana-Euclea divinorum* forest community, *Croton dichogamus-Euclea divinorum* forest community, *Croton dichogamus-Acacia brevispica* low thicket, *Rhus natalensis - T. triandra* tall closed shrubland, *Tarchonanthus camphoratus-T.-triandra* low closed woodland, *Euclea racemosa-T. triandra* tall closed shrubland and *T.-triandra-Cynodon dactylon* short closed grassland. The study acknowledged changes in the woodland communities of the Mara. Walpole *et al.*, (2004) identified thirteen woody habitats in Maasai Mara reserve based on dominant species and used them as the basis for analyzing the densities, browse availability and species richness of different habitats.

Mutangah (1989) differentiated the vegetation in Lake Nakuru National Park into *A. xanthophloea*, *Euphorbia candelabrum* Kotschy and *Olea europaea* L. African mill forests. However, in a more detailed floristic survey, (Mutangah, 1994) further differentiated and classified the vegetation in Lake Nakuru National Park into twenty four sub-types, which were presented in a physiognomic vegetation map. Mwasi *et al.*, (2013) in a study that investigated habitat segregation patterns between similar sized large herbivores in the park recognized 9 vegetation types. Ng'weno *et al.*, (2009) in a study that looked at distribution, density and impact of invasive plants in the Park recognized 8 vegetation types. Mwangi & Western (1998) in their study that determined fluctuations in food supply identified and worked on four vegetation types; open grassland, shoreline grassland & bushed grassland.

In Amboseli ecosystem, Okello (2005) classified the vegetation in Kuku Group Ranch (KGR) into grassland, open woodland, open shrubland, dense woodland, dense shrubland and riverine woodland. In Kimana Group Ranch (KGR), Kioko *et al.*, (2012) classified vegetation basing on the physiognomic characteristics into wooded grassland, bushed grassland, grassland and dwarf shrub grassland.

Lekoiyet (2006) in a comparative study identified and classified woodlands in Kimana and Eselenkei Group Ranches into *A. tortilis* woodland and *A. xanthophloea* woodland. In her study, she compared the composition and structure of woody vegetation in the woodlands of conserved and communal sites of the Group Ranches. The works of Lekoyiet (2006) were limited to the woodlands only hence other vegetation types in KWS were not classified and considered for detailed floristic study.

2.2 Composition and structure of vegetation in savanna ecosystem

2.2.1 Composition and structure of woody vegetation

In savanna ecosystems, vegetation is generally characterized by a discontinuous layer of woody plants (trees & shrubs) and a continuous layer of grasses (van Langevelde *et al.,* 2003). However, several studies have provided evidence to show that woody vegetation across and within ecosystems is highly variable both spatially and temporary due to influences from climate, differences in soil types and disturbances such as hervivory, fire and human activities (Gandiwa *et al.,* 2011; Gandiwa *et al.,* 2013; Gandiwa *et al.,* 2014).

Wessel *et al,* (2010) studied the structure of woody vegetation along a land use gradient in the South Africa’s rangelands using airborne light detection and ranging (LiDAR). They compared canopy cover and height distribution and canopy cover on areas with different management strategies. It was observed that large trees (>7m) were clearly valued and conserved in communal rangelands and trees under 5m were of low prevalence.

Diversity and structure of woody vegetation has been studied across areas with different soil types in Gonarezhou National Park in Zimbawe. For instance, Gandiwa *et al.* (2011) compared the structure and composition of woody vegetation across areas characterized by rhyolite, malvernia and granophyte bedrocks. They found significant difference in tree height, densities, basal area and species diversity across the three study sites (Gandiwa *et al.,* 2011). Gandiwa *et al.*, (2014) in a study that compared the structure and composition of woody vegetation between siallitica and rogosol soils in the same park found similarities in woody vegetation structure between siallitic and regosol soil stratum but woody species diversity was found to be significantly higher in siallitica soils stratum compared to rogosol soil stratum. The results from this study suggested that soil variations are important in influencing woody vegetation diversity and structure within ecosystems.

Banda *et al.,* (2006) studied the structure and composition of vegetation along a protection gradient of Miombo woodlands in western Tanzania. The study assessed species richness, basal area, stem density and unique species in a protected area within a national park, game controlled area, forest reserve and unrestrict open area. The results showed basal area to be highest in game controlled area; unique species was high in all areas, except in national parks and stem density and species richness was high in game controlled area and forest. The low basal area, unique species and stem density and species richness found in the national parks conradicts the assumption by conservation managers on the total protection of areas to protect biodiversity including plants and argued that protection in East Africa is geared towards animals, hence, protected areas are usually poor in flora.

Dharani *et al.,* (2006) compared the structure and composition of *A. xanthophloea* woodlands in Lake Nakuru National Park. They selected four sites and estimated plant density, height, canopy cover and basal area of woody species. The study found *A. xanthophloea woodland* to dominate all the study sites, with importance value index that ranged between 35.3 and 60. Despite, the relative density and regeneration of *A. xanthophloea* trees differed across the *A. xanthophloea* woodland sites. They argued that knowledge on tree size, structure, species composition and regeneration patterns are important in understanding the state and dynamics of woodlands, which is important for parks and land managers to design methods to achieve a sustainable use of woody resources.

Lekoiyiet (2006) in a study that compared the structure and composition of woody vegetation between *A. tortilis* and *A. xanthophloea* woodlands in conserved and communal sites of Eselenkei Group Ranch (EGR) and KGR in Amboseli ecosystem recorded a total of 30 woody species belonging to 10 families in both the study sites. They found woody species diversity to be higher in the communal area than the conservation area of EGR. They also found higher woody species diversity in the *A. tortilis* woodland of the communal area and lower woody species diversity in *A. xanthophloea* woodland of the conserved site. They found *Commiphora schimperi* (Berg) Engl. to have a high Importance value index (155.61) in the *Acacia commiphora* bushland in Eslenkei and *A. tortilis* had the highest IVI value in *A. tortilis* woodland in Kimana. However, they found similarities in Shannon-Weiner diversity index, woody plant densities and seedling/sapling regeneration between the study sites. *Acacia xanthophloea* and *A. tortilis* trees showed an inverted J-shaped distribution at the conserved and communal sites. In this study, factors such as charcoal production, fencing and construction were cited to be the major factors influencing woody species (Lekoyiet, 2006).

However, studies of Lekoiyiet (2006) failed to assess the structure and composition of woody vegetation across different vegetation types within the study sites. This study therefore compared the structure and composition of woody vegetation in KWS.

2.2.2. Composition and structure of herbaceous vegetation

Overgrazing and selective grazing by herbivores has been shown to influence negatively the composition and structure of herbaceous vegetation and soil characteristics in savanna ecosystems. A larger number of field studies focusing on the effects of grazing management systems on rangeland vegetation clearly show variations in the structure and composition of herbaceous vegetation between areas of heavy or continuous grazing and areas of light or moderate grazing (Rutherford & Powrie, 2009; Kioko *et al,* 2012; Zarekia *et al.,* 2013). For example, in an experimental study to investigate the effects of livestock grazing on the Steppe rangelands of Iran, Zarekia *et al.* (2013) compared species composition, canopy cover percentage, production and species height between areas of different grazing regimes. The study reported increased composition of increaser III grasses and reduced canopy cover percentage, grass biomass and species height in areas experiencing continuous grazing throughout the years compared to enclosures for four years. They expressed that rotation grazing system and appropriate grazing capacity can help conserve the vegetation and soils of rangelands.

Rutherford & Powrie (2009) reported a significant decline in the canopy cover, species number of annual and perennial life forms, species diversity and local extinction of some plant species, mainly graminoids, in areas with high grazing intensity in rangelands of South Africa. Another study by Rutherford *et al.,* (2012) showed that areas under intensive grazing had reduced grass canopy cover and height of graminoid plants. Furthermore, Rutherford & Powrie (2013) in a comparative study across rangeland biomes of South Africa, found large herbivores to alter species composition by replacing perrenial plants with annual plants under intensive grazing.

In the semi arid savannas of Swaziland, Tefera *et al.,* (2010) found perennial grasses such as *C. ciliaris*, *P. maximum* and *Heteropogon contortus* L. to dominate in low stocking rate and annual grasses such *Ragus berteronianus* Schult. and *Aristida bipartita* (Nees) Trin. Rupr. dominated in high stocking rate.

In East Africa region, heavy or continuous grazing has been cited as the major factor causing significant variations in the structure and composition of herbaceous vegetation across grazing lands. Tefera *et al.,* (2007) in Ethiopia observed a lower density of palatable herbaceous plants in communal grazed lands than in ranches. Tessema *et al.,* (2011) found higher herbaceous density, total abundance, basal cover and aboveground biomass and a lower percentage of bare ground in lightly grazed sites compared with the heavy grazed sites. Angassa (2014) in rangelands of southern Ethiopia found plots experiencing light and moderate grazing to be characterized by high herbaceous species richness and abundance as compared with plots experiencing heavy grazing.

Researches done in rangelands of Kenya have also shown that high grazing intensity affects herbaceous vegetation negatively. In the rangeland of northern Kenya, Keya (1998) studying herbaceous layer production and utilization by herbivores under different ecological conditions found high standing crop biomass of between 55.3 and 4320.1 kg/ha for grasses, forbs and dwarf shrubs under non-grazing conditions and an increase in grass standing biomass under grazing conditions.

Muthoni *et al.,* (2014) have argued that ungulate herbivory overrides rainfall impacts on herbaceous re-growth and residual biomass. In their study they used enclosure experiment to investigated impacts of ungulate herbivory on herbaceous re-growth and residual biomass in Lake Naivasha. The study showed that intensive grazing significantly reduced re-growth during the dry season and mean aboveground biomass was significantly higher in un-grazed than grazed treatments but was significantly different in un-grazed treatments in the two study seasons. Grazed aboveground biomass was significantly different for all monthly comparisons, except between June and September of 2011 and high grazing intensities (0.59-0.75) were recorded in sites dominated by short-medium height grasses.

Ng'weno *et al.,* (2009) studied the distribution, density and impact of invasive plants in Lake Nakuru National Park. In their study they compared grass biomass in invaded grasslands and non-invaded grasslands of various grassland types. Grass biomass was higher in non invaded grasslands (37.2g m $^{-2}$) than in invaded grasslands (22g m^{-2}). Also, *Sporobolus spicatus* grassland type was found to have the lowest mean forage grass biomass of 23.88 ± 5.53g m^{-2} and 13.46 ± 4.97 g m^{-2} for non-invaded and invaded grassland patches respectively.

In the rangelands of central Kenya, Mureithi *et al.,* (2014) investigated impacts of community conservation management on herbaceous layer and soil nutrients in Laikipia County. The study compared vegetation in conservation zones, buffer zones and high intensity use zones. They found areas of high intensive use to be characterized by low herbaceous diversity, species richness, basal cover, herbage cover and relative abundance of both annual and perennial grasses. These areas were also found to have higher percentage of bare ground compared to conservation and buffer zones.

Berliner & Kioko (1999) in their study that investigated long-term impacts of mowing and ungulate exclusion on natural unfertilized rangelands of Athi Kapiti plains found an increase in the relative percentages of *P. mezaneum* and *P. stramineum* and a decrease in the relative percentages of *Microchloa kunthii* Desv. and *Sporobolus ioclados* (Trin.) Nees. Imbahale *et al.* (2008) in the same area studied resource utilization by large migratory herbivores. The study compared grass biomass in plots between months and vegetation types in different grazing management systems. They found monthly mean plant biomass among plots within the open grassland in the conservation area to be different, but grass biomass in the open and bushed grasslands in the community grazing area was not different.

Kamau (2004) studied forage diversity and impact of grazing management on rangeland ecosystem in Mbeere District Kenya. The study compared species richness biomass and canopy cover between open sites and enclosures. It also compared species richness, diversity and evenness between the open sites and enclosure in the dry and wet seasons. She found total biomass and total species richness to be higher in enclosures than in open sites, but herbaceous species richness was higher in open site than in enclosure. Furthermore, herb and shrub cover was higher in open site and enclosure respectively, but herbaceous biomass and tree cover were relatively the same in the open site and enclosure.

Kioko *et al.,* (2012) working in the rangelands of southeastern Kenya indicated that continuous grazing causes loss of vegetation with negative, long-term effects on grass functional qualities and forage production. They investigated impacts of livestock grazing on herbaceous vegetation in KGR and designated the grassland areas into three; grassland from previous Maasai settlement that had been abandoned for over twenty years; grassland excluded from livestock grazing for eight years; dry season grazing area and continuous grazing area where grazing occurred throughout all seasons. They collected data on grass species composition, grass height, inter-tuft distance, grass standing biomass and soil characteristics. They found most of the study sites to be dominated by increaser I and Increaser II grass species. Areas under continuous grazing were found to have low grass height, high inter-tuft distance, low forage potential, high soil erosion potential, low contents of exchangeable calcium, total nitrogen and soil PH values compared with the other study sites. However, the study found no significant changes for grass biomass between the dry season grazing area and the area excluded from livestock grazing for eight years.

2.3 Seasonal population sizes of large herbivores in savanna ecosystems

In savanna ecosystems, large herbivore densities have been surveyed between seasons and studies report seasonal variations among a few species. Chamaille-Jammes *et al.,* (2009) in a study that investigated seasonal density estimates of common large herbivores in Hwange National Park, Zimbabwe found local densities of large herbivores to fluctuate throughout the year, with larger increase in the late dry season in the main camp. However, Africa elephants ((*Loxodonta africana* Blumenbach) in this season were abundant (>3 individuals/km^2) in all the studied areas. The observed seasonal fluctuations in densities in this study were attributed to seasonal changes in surface water availability across the park.

In Simanjiro plain, Tanzania, Kahurananga (1981) estimated populations, densities and biomass of large herbivores and found Plains zebras and Common wildebeest (*Connochaetes taurinus* Burchell) to have the highest population, with a peak of 10.96km^2 and 7.44/km^2 for zebras and wildebeests respectively during the wet season. Cattle (*Bos indicus* L.) were found to have the highest population with peak estimate of 44.83/km^2 during the dry season. Contrary, Kiffner *et al.,* (2016) in a study that investigated trends in seasonal population densities of wild species in Tarangire National Park, Tanzania, found Maasai giraffes (*Giraffa camelopardalis* L.), Plains zebras and Common wildebeest to have significantly lower population densities during the rainy season than the dry season.

Mwangi and Western (1998) in a general study that investigate habitat selection by large herbivores in Lake Nakuru National Park, Kenya, compared the population density of large herbivores between species, habitats and seasons. The population density of large herbivores was found to be significantly different between species and habitats, but seasons.

2.4 Seasonal selection of habitat by large herbivores in savanna ecosystems

Selection of habitats by large herbivores is often related to several factors, including forage availability, forage abundance and quality, predation risks, water availability, topography and habitat heterogeneity (Sitters *et al.*, 2009; Groom & Harris, 2010; Owen-Smith *et al.*, 2015).

However, in savanna ecosystems, where rainfall patterns are seasonal, habitats selection by large herbivores usually varies according to the season, due to the spatial and temporal variability of forage resources. Thus, Macandza *et al.*, (2012) in a study that compared habitat and resource use between rare and abundant species in Kruger National Park, South Africa found that large herbivores vary the range of habitats selected during different seasons. In the study, buffaloes (*Syncerus caffer* Sparrman) were found to use a wide range of habitats during the wet season, but shifted towards lowlands during the late dry season, zebras used habitat type characterized by relatively open woody cover throughout the year and sable used a narrow range of habitats characterized by taller and denser woody vegetation during the wet and early dry season, but used bush savanna during the late dry season.

Research done by Dekker *et al.,* (1996) to investigate spatial and temporal distribution of ungulates and identify habitat variables that determine ecological separation of ungulates in Mopeni veld, South Africa found large herbivores to exhibit varying degrees of habitat selectivity. Owen-Smith *et al.,* (2015) in Kruger National Park found wildebeest herds to use grazing lawn grasslands during the wet season but shifted to seep-zone grasslands in the late dry season.

Okane, *et al.,* (2013) in their study that investigate the effects of resource limitation on habitat usage by the browser guild in HiP found greater selection of vegetation types to occur in the dry season, with greater selection occurring in the second, more severe dry season than in the first, less severe dry season. Mwangi and Western (1998) in Lake Nakuru National Park, Kenya, found most large herbivores to select a narrow range of habitats during both the dry and the wet seasons in the park. Waterbucks, however, according to the study, used a wider range of habitats during both the dry and the wet seasons.

2.5 Seasonal habitat niche breadths among large herbivores in savanna ecosystems

Few researchers have studied the effects of rainfall seasonality on habitat niche breadths pattern among large herbivores in savanna ecosystems (Ahrenstani *et al.,* 2012; Mwasi *et al.,* 2013). In a comparative study of co-occurring assemblages of four large herbivores; chital (*Axis axis* Erxleben), sambar (*Cervus unicolor* Kerr), guar (*Bos gaurus* Smith) and Asia elephant (*Elephas maximus* L.) in Bandipur and Mudumalai forests, South India, habitat niche breadth values were found to be high for elephants and low for guar during the dry and the wet seasons (Ahrenstani *et al.,* 2012).

In contrast, a few other studies could not show any difference between large herbivore in habitat niche-breadth values during the dry and the wet seasons. For instance, Mwasi *et al.,* (2013) found similar sized large herbivores to have low habitat niche breadth patterns during the dry and wet seasons in Lake Nakuru National Park.

2.6 Seasonal habitat overlap among large herbivores in savanna ecosystems

Several studies have shown large herbivores to coexist through habitat selection and partitioning of critical resources such as habitats and diets and several factors including body sizes, digestive physiologies and seasonality are considered important determinants of ecological separation among large herbivores (Gromsigt & Olff, 2006; Gromsigt *et al.,* 2009; Mwasi *et al.,* 2013; Kleynhans *et al.,* 2010). However, in savanna ecosystems where rainfall is seasonal, seasonality of resources is considered important in determining niche partitioning (Kleynhans *et al.,* 2010). For instance, in Lake Nakuru National Park, Kenya, Mwasi *et al.,* (2013) found large grazers to have low degree of habitat overlap during the dry season than the wet season.

Dekker *et al.,* (1996) investigated habitat partitioning by ungulates in the Mopeni veld, South Africa and found the studied ungulates to exhibit seasonal changes in the use of plant communities. In the warm, dry season, the ungulates were found to be widely separated in their use of plant communities, but considerable overlap was observed among several species in the wet season and cool, dry season.

Kleynhans *et al.,* (2010) in Hluhluwe iMfolozi Park, South Africa, investigated resource partitioning among six large herbivores; impala (*Aepyceros malapus* Lichtenstein), warthog (*Phacochoerus aethiopicus* Pallas), wildebeest, buffalo, zebra and white rhinoceros (*Ceratotherium simum* Burchell). Habitat utilization differences among the species were found to be generally small and did not vary between the dry and the wet seasons. The dry season resource partitioning among species in the park was attributed to body mass differences except for the white rhinoceros, a megaherbivore. They, therefore, concluded that coexistence of savanna herbivores was mostly through size-driven resource partitioning in the dry season, with the exception of the white rhinoceros.

Okane *et al.,* (2013) working in HiP, South Africa, studied overlap and seasonal shifts in use of woody plants amongst a guild of savanna browsers. The results showed overlap in habitat use to be higher for all large herbivores pairs in the wet seasons and 95% of overlap values were higher for all herbivores pairs in the dry seasons. However, 80% of overlap values were lower during the dry season compared with the preceding wet seasons in both the study years.

Traill (2004) studied seasonal utilization of habitat by large grazing herbivores in semi-arid Zimbabwe and found high degree of niche overlap among grazer species during the hot, wet season and cool, dry season. The study also found species to ecologically separate in their use of habitats during the hot dry season. Traill (2004) associated the distribution and ecological segregation of large herbivores in this study mostly with distance from water, grass sward height, time since last burned, woody plant density and by the presence of predominant grasses *Urochloa mosambicensis* (Hack.) Dandy*, Panicum maximum, Heteropogon contortus* (L.) P. Beauv. ex Roem. & Schult. *and Digitaria eriantha* Steud. Basing on the study results, Trail (2004) recommended estimation of the population trends and carrying capacity of herbivores and monitoring of vegetation.

In a study to investigate the effects of introduced exotic cattle into native African herbivore assemblages, Voeten & Prins (1999) compared resource partitioning between zebras, wildebeests and cattle in Tarangire region, Tanzania. Zebras, wildebeests and cattle overlapped in habitat use during the wet season, but wildebeests and zebras segregated in their use of habitat types during the dry season. However, cattle overlapped with zebras during the early wet season, but overlapped with wildebeests during the early dry season.

Mwasi *et al.*, (2013) in Lake Nakuru National Park, Kenya, studied seasonal resource use and niche breadth in an assemblage of six coexisting grazers of different body weights and predicted that overlap in resource use would be smallest among grazers with similar body weights in the dry season. The study, however, found overlap in resource use to be consistently higher during both the dry and the wet seasons and suggested that the co-occurring grazer assemblages in the Park could be interacting competitively.

2.7 Knowledge gap

From the reviewed literature, it is evident that there is no information on the composition and structure of vegetation and on habitat use among large herbivores in KWS of which this study generated.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Study area

3.1.1 Location and size

Kimana Wildlife Sanctuary is found within Amboseli ecosystem in Oloitokitoki sub-county, Kajiado County, southeastern part of Kenya (Figure 3.1). Amboseli ecosystem is situated between Chyulu Hills and Tsavo West National Park south of Mount Kilimanjaro and covers an area of 5700 km^2 (Tuqa *et al.,* 2014). It lies between longitude 36^0 5′ and 37^0 5′ East and latitude 1^0 0′ and 3^0 0′ South (Kihima & Nyamasyo, 2014). Kimana Wildlife Sanctuary is located in the northeastern side of KGR and covers an area of about 22.5 km^2 of the total 251 km^2 land area of KGR (Okello *et al.,* 2011).

3.1.2 Climate

Amboseli ecosystem is typical of a “semi-arid” under Agro-Ecological Zone V1, as defined by Pratt and Gwynne (1977). Precipitation is generally low and is partly influenced by the relief conditions of Mount Kilimanjaro (Okello *et al*., 2011). Rainfall occurs in two seasons, with the short season occurring around October to November and the long season occurring around March to early June (Altmann *et al*., 2002). Mean annual rainfall varies greatly, from 150 mm to 200 mm per year, but it may be relatively high during the two seasons (Altmann *et al*., 2002).

Temperatures are continuously warm to hot and varies within seasons. The daily temperatures range between 12^0C and 35^0C. Lowest temperatures occur between June and August and highest temperatures occur between September and February (Altmann *et al.,* 2002).

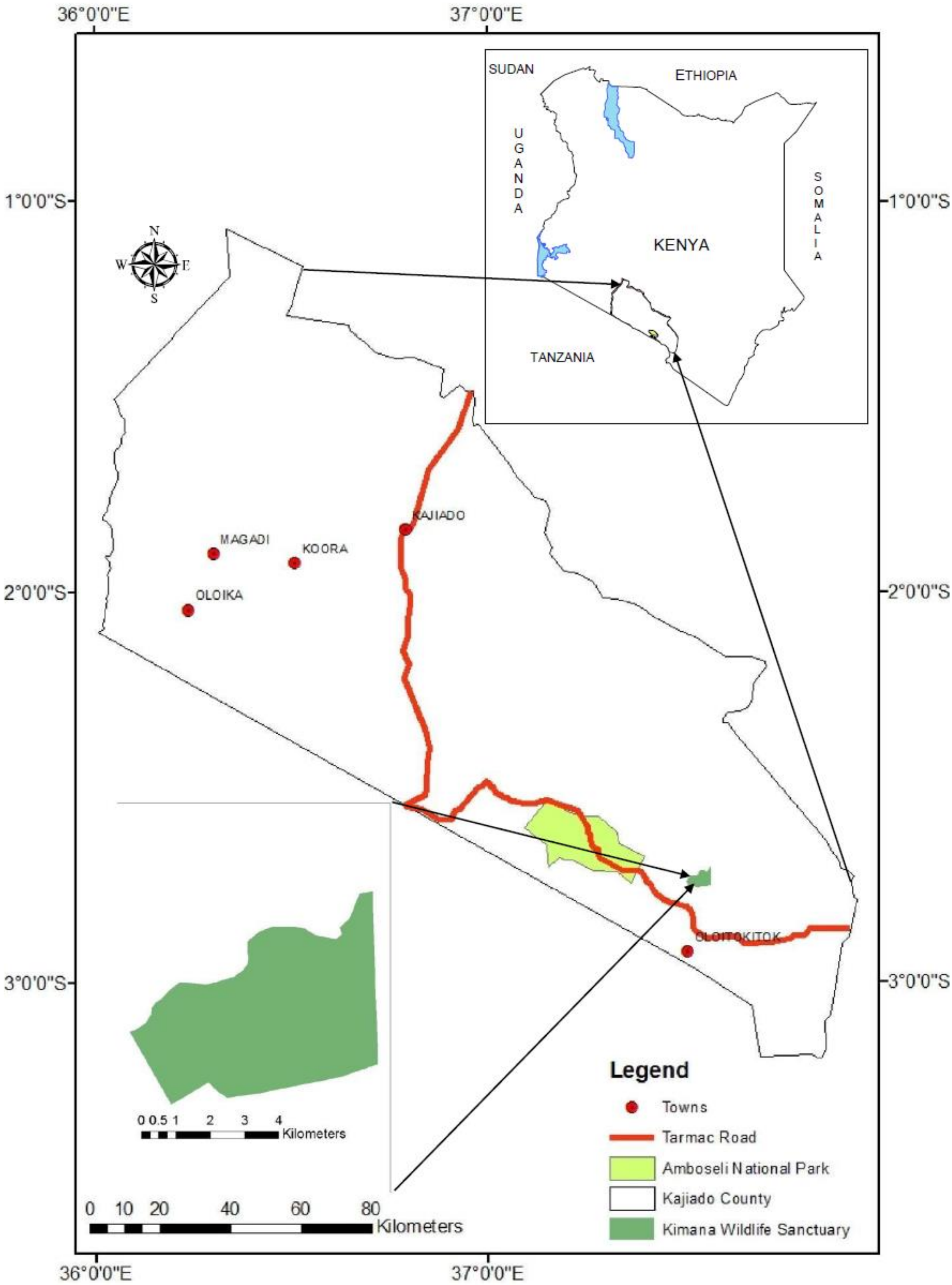
3.1.3 Geology, landforms and soils

The area is generally characterized by undulating uplands and plains and the soils are variable depending on parent material and the landforms (Gachimbi, 2002). The uplands have moderately deep and firm clay soils formed from basement system rocks rich in ferromagnesian minerals (Ferral-chromic Luvisols). The plains have very deep, friable and firm sandy clay soils formed from undifferentiated basement system rocks. The plains of Amboseli basin have soils varying depth, saline-sodic clay soils. River alluvial plains have deep, well drained, cracking clay soils (Luvisols to Vertisols).

Soil fertility is generally moderate, but in cultivated areas, organic carbons and phosphorus are generally low due to continuous cultivation and high mineralization rate of soil organic carbon resulting from high temperatures and adequate moisture (Gachimbi, 2002).

3.1.4 Fauna and flora

Vegetation of Amboseli ecosystem is diverse in terms of physiognomy and floristics and according to Western (1983) they include open grassland, bushland and wooded grassland. Esikuru (1998) classified vegetation in Amboseli basing on cover into riverine vegetation, swamps, bushland, forested woodland, shrubland and wooded grassland. The vegetation types in KGR include wooded grassland, bushed grassland, grassland and dwarf shrub grassland (Kioko *et al.,* 2012).


36°0'0"E
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SUDAN
ETHIOPIA
UGANDA
SOMALIA
KENYA
TANZANIA
N
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E
S
MAGADI
KOORA
KAJIADO
OLOIKA
OLOITOKITOK
0 0.5 1 2 3 4
Kilometers
0 10 20 40 60 80
Kilometers
Legend
Towns
Tarmac Road
Amboseli National Park
Kajiado County
Kimana Wildlife Sanctuary

Figure 3.1: Spatial location of Kimana Wildlife Sanctuary
(Source: Author, 2016)

Amboseli ecosystem has a variety of plant life-forms; trees, shrubs, herbs, sedges and grasses (Lekoyiet, 2006). Some of the trees and shrubs present include *A. tortilis*, *A. xanthophloea*, *A. drepanolobium*, *B. glabra, S. persica* and *Cordia monoica* Roxb. *Acacia xanthophloea* tree is dominant in riparian areas while other trees are found in the dry sites (Okello *et al.*, 2011). The dominant perennial grasses are *C. ciliaris* and *Chloris roxburghiana* Schult (Okello *et al.*, 2011).

Large herbivores are the most conspicuous animals in Amboseli ecosystem and occur in high diversity and abundance (Okello, 2005). They include African elephant, Cape buffalo, Maasai Giraffe, impala, Plains zebra, warthog, Common wildebeest, cattle, Thompson's gazelle (*Gazella thomsonii* Günther), Grants' gazelle (*Gazella granti* Brooke), Common waterbuck (*Kobus ellipsiprymnus* Ogilbyi), Cape eland (*Tragelaphus oryx* Pallas), Bohor reedbuck (*Redunca arundinum* Boddaert), goats (*Capra aegagrus hircus* Erxleben) and sheep (*Ovis aries* L.) (Okello, 2005).

Vervet monkeys (*Cercopithecus aethiops* L., Sykes monkeys (*Cercopithecus mitis* Wolf) and olive-backed baboons (*Papio cynocephalus Anubis* Lesson) are common primates of Amboseli ecosystem. The most common predator species include spotted hyena (*Crocuta crocuta* Erxleben), black-backed jackal (*Canis mesomelas* Schreber), Golden jackal (*Canis aureus* L.), Cheetah (*Acinonyx jubatus* Schreber), Bat-eared fox (*Otocyon megalotis* Desmarest) and lions (*Panthera leo* L.) (Okello, 2005).

3.1.5 Land uses

Amboseli ecosystem has variable land uses. Traditionally, subsistence pastoralism was practiced by the Maasai community, but this has been replaced by agriculture, agro-pastoralism and other human developments (Okello & D’Amour, 2008; Okello, *et al.,* 2011). The changes in land uses is associated with poverty, droughts, education level, cultural practices, agricultural expansion, human population dynamics and changing land tenure policy (Okello & D’Amour, 2008; Kihima & Nyamasyo, 2014). Agricultural expansion and human population growth have resulted in an increase in farmlands and human settlements and a decline in forestland, grassland, wetland and woodland (Kihima & Nyamasyo, 2014). Changes in land uses may result in a decline in wildlife ungulate numbers, habitat destruction, increased human-wildlife conflicts and human- human conflicts, land degradation and displacement of wild ungulates by livestock (Okello, 2005; Okello *et al.*, 2011; Kihima & Nyamasyo, 2014).

3.2 Research design

A pilot study was carried out in 2011 to help evaluate the vegetation types and to determine the most appropriate research design for the study. The vegetation in the study area was observed to be horizontally and vertically heterogeneous (Pratt & Gwynne, 1977); hence stratified research design was used to sample vegetation and to count large herbivores.

3.3 Characterization and classification of vegetation

3.3.1 Classification using remote sensing

Using remote sensing, a LANDSAT image of the study area acquired from United states Global Land Cover Facility Website (*http://glcf.umd.edu/data/landsat/*) (2016) was used in the classification. Supervised classification method (maximum livelihood) to cluster vegetation types implemented in ArcGIS Map 10.2 software (ESRI, 2014) (Figure 3.2). Training samples were used to identify and classify vegetation types. Histograms tool and statistical tool (ESRI, 2014) was used in evaluation to ensure enough separation between the classes identity as training sample. Overlapping classes were merged to avoid confusion in the final classification. A signature file tool was created using the created signature file tool to generate classified image with five types of vegetation.

3.3.2 Classification using physiognomic characteristics and dominant plant species

Vegetation was visually stratified into vertical and horizontal components (Muller-Dombois & Ellenberg, (1974). The vertical vegetation component was stratified into two distinct layers based on the plant life form; the canopy layer, which comprised of woody plants, trees (rooted, woody, self-supporting plants ≥ 3m in height with one or few definite trunks) and trees/or shrubs (rooted woody self-supporting, multi-stemmed or single stemmed plants greater than 1m in height) and the ground layer, which comprised of herbaceous plants (plants that do not have persistent stem above ground). Similarly, the horizontal vegetation component was stratified into six distinct stands based on the physiognomic features such as plant height and cover (Pratt & Gwynne, 1977). All the five vegetation types classified (woodlands, grassland, herb grassland, wooded grassland, sparse shrubland) were taken to represent different sampling strata and habitat types for animals. Woodlands and grasslands were further classified based on the dominant plant species (Pratt & Gwynne, 1977; Muller-Dombois & Ellenberg, 1974).

3.4. Sampling of woody vegetation

Point-Centered Quarter (PCQ) method was used to assess woody plants in *A. tortilis* woodland, *A. xanthophloea* woodland and sparse shrubland. In *A. tortilis* and *A.*

xanthophloea woodlands , sampling was done along 2 parallel, 1km line transects at intervals of 100m, for a total of 9 points per transect and 18 points for the two sampled line transects in each vegetation type. In wooded grassland, sampling was done along 4 parallel, 1km line transects at intervals of 100m, for a total of 7 points per transect and 28 points for this vegetation type. In sparse shrubland, sampling was done along 4 parallel, 500m line transects at intervals of 100m, for a total of 4 points per transect and 16 points for this vegetation type.

At each sampling point, a cross shaped wooden frame was laid on the sampling point to divide the area near the point into four 90^0 quarters of the compass. The nearest woody plant to the point in each quarter was sought and species with diameter at breast height (dbh) of ≥ 5cm and a height of ≥ 1m was identified and classified using the scientific names (Agnew & Agnew, 1994; Beentje, 1994; Noad & Birnie 1994) and the following information was recorded; the quarter number, species name, distance (m) from the point to the centre of the trunk, the dbh (m) and canopy diameters (D_1 & D_2). The distance (m) from the quarter point to the first woody plant and its dbh (m) was measured (cm). The voucher specimens of sampled species were later collected, mounted and deposited in the herbarium of the University of Eldoret.

Belt Transect Method (Mueller-Dombois & Ellenberg, 1974) was used to sample woody plants in wooded grassland, since the woody plants were very scattered. The belt transects used were 10m wide and 1km long. Sampling was done along 4 parallel, 10m x10m belt transects placed at intervals of 100m along the transect for a total of 7 points per transect and 28 points for all the 4 transects sampled in this vegetation type. All the woody plants with dbh of ≥ 5cm and a height of ≥ 1m were identified and classified according to Beentje, (1994) and Noad & Birnie, (1994) and their dbh and canopy dimensions; the widest (D_1) and the dimension perpendicular to the widest (D_2) recorded. The voucher specimens of sampled species were later collected, mounted and deposited in the herbarium of the University of Eldoret.

3.4.1 Data processing

3.4.1.1 Point-Centre-Quarter data

i) Mean area per individual

The distance from the point to the nearest woody species in all the quarters was totaled and averaged to get the mean point-to individual distance. The mean obtained was then squared to give the mean area per individual.

ii) Total density

The total densities of species in the area sampled were calculated as follows (Brower & Zar, 1990):

$$TD = {}^{u}/_{A} \qquad [3.1]$$

Where TD = total density

u = Number of area units used in expressing density

A = mean area per individual

iii) Relative density

The relative density of species i was determined using the following equation (Brower & Zar, 1990):

$$RDi = {}^{ni}/_{\sum n} \qquad [3.2]$$

Where RDi = relative density

ni = number of individuals of species i

$\sum n$ = total number of individuals of all species counted

iv) Absolute species density

The absolute density for each species was determined using the following equation (Brower & Zar, 1990):

$Di = (RDi)(TDi)$……………………………………………… [3.3]

Where Di = absolute density for species i

RDi = relative density for species i

TDi = total density of species i

v) Cover

The coverage of species i was determined as follows (Brower & Zar, 1990):

$Ci = {}^{(ai)(Di)}/_{ni}$ ……………………………………………... [3.4]

Where Ci = density of species i

ai = sum of the basal area for species i

Di = density of species i

vi) Relative cover

The relative cover of species i was determined as follows (Brower & Zar, 1990:

$RCi = {}^{Ci}/_{\sum C}$ .... [3.5]

Where RCi = relative coverage for species i

$\sum C$ = total cover for all species

vii) Frequency

The frequency of species i was determined using the following equation (Brower & Zar, 1990):

$fi = {}^{ji}/_{k}$ .... [3.6]

Where fi = frequency of species i

ji = number of sampling points where species i was recorded

k = total number of points sampled

viii) Relative frequency

The relative frequency of species i was determined as follows (Brower & Zar, 1990):

$Rfi = {}^{fi}/_{\sum f}$ .... [3.7]

Where Rfi = relative frequency for species i

$\sum f$ = total of the frequencies of all species

ix) Importance value

The importance value index for species i was determined using the following equation (Brower & Zar, 1990):

$IVi = RDi + Rfi + RCi$.. [3.8]

Where IVi= importance value index of species i

x) Canopy cover

Canopy cover of species i was determined using the following equation (Ko *et al.,* 2009):

$\mathrm{C} = 2\pi\ (D1^2 + D2^2/2^{1/2}$... [3.9]

Where C = Canopy cover

D_1 and D_2 = Two perpendicular canopy diameters recorded at 90^0

π = Constant, 3.14

xi) Relative canopy cover

The relative canopy cover of species i was determined as follows (Ko *et al.*, 2009)

$RCi = {}^{Ci}/_{\sum C}$.. [3.5]

Where; RCi = relative canopy coverage for species i

$\sum C$ = total canopy cover for all species

xii) Size class distribution

The distribution of size classes for woody plants were estimated using the histograms constructed from the frequency of woody plants (%) (Y-axis) categorized into diameter at breast height (dbh) classes (cm) (X-axis) (Peter, 1996). However, the regeneration status of woody plants across the vegetation types was determined based on the profile depicted by the size classes.

xiii) Species richness

Species richness was expressed as the total number of species within an area

xiv) Species diversity

Species diversity was estimated using the following equation (Krebs, 1998):

$H' = -\sum[(Pi) * In(Pi)]$……………………………………………… [3.10]

Where H' = Shannon –Weiner Diversity index

$\sum$ = Summation

Pi = Proportion of total sample belonging to species i

xv) Species evenness

Species evenness of species was estimated using the following equation (Krebs, 1998):

$J = {}^{H'}/_{InS}$……………………………………………………...... [3.11]

Where J = Species evenness

H' = Shannon – Weiner diversity index

InS = Maximum diversity possible

3.4.1.2 Belt transect data

i) Density

The density of species i was determined using the following equation (Brower & Zar, 1990)

$Di = {}^{ni}/_{A}$……………………………………………………….. [3.12]

Where Di = density of species i

ni = total number of individuals counted for species i

A = total area sampled

ii) Relative density

Species relative density was calculated as described in equation (3.2)

iii) Cover

The cover of species i was determined as follows (Brower & Zar, 1990):

$Ci = {}^{ai}/_{A}$ ……………………………………………………….. [3.13]

Where Ci = coverage

ai = total area covered by species i

A = total area sampled

iv) Relative cover

Species relative coverage was calculated as described in equation (3.7)

v) Frequency

Species frequency was calculated as described in equation (3.6)

vi) Relative frequency

Species relative frequency was calculated as described in equation (3.7)

vii) Importance value indices (IVI)

Importance value indices of species i was estimated as in equation (3.8)

viii) Species richness

Species richness was estimated as in 3.4.1.1 (xii)

ix) Species diversity

Species diversity was estimated as in equation (3.10)

x) Species evenness

Species evenness was estimated as in equation (3.11)

xi) Size class distribution of woody plants

The distribution of size classes for woody plants was estimated as in 3.4.1.1 (xii)

xii) Jaccard’s similarity coefficient

Similarity in the composition of woody species was estimate using the equation below (Janson & Vegelius, 1981):-

$$SJ = {}^{A}/_{(A+B+C)} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots [3.14]$$

Where SJ = Jaccard's similarity index

A = number of woody plants present in the two vegetation types under consideration

B = number of woody plants present in vegetation type 1 but absent in vegetation type 2

C = number of woody plants present in vegetation type 2 but absent in vegetation type 1

Jaccard's similarity coefficients range from 0 (complete dissimilarity) to 1 (identical).

3.5 Sampling of herbaceous vegetation

Descending Step Point Method (DSPM) (Trollope, 2004) was used to collect data across vegetation types for determination of grass frequencies and inter-tuft distances. Data were collected by working a maximum of 20 steps (approx. 1m each) from each sampling point along 2 parallel, 1km transects at intervals of 100m for a total of 200 points per transect and 400 points per vegetation type.

At every step, a metallic pin was vertically dropped on the ground from a height of approximately 1.5m high. The herbaceous plants (i.e. grasses/forbs/sedges) hit by the pin was identified and classified to species level as in Agnew & Agnew, (1994) and Clayton, (1982). The grasses hit were grouped into species ecological categories (decreaser, increaser I and increaser II grass species) (Trollope, 2005). The hit was also recorded whether on bare ground, litter, dung or rock. The nearest herbaceous plant to each hit point was also identified and classified to the species level as in Agnew & Agnew, (1994) and Clayton, (1982) and the distance (cm) between each hit point and the nearest herbaceous plant was also recorded. The voucher specimens of sampled species were later collected, mounted and deposited in the herbarium of the University of Eldoret.

Disc Pasture Meter (Gachuli, 2000) was used to measure the grass standing crop biomass and grass height. The Disc Pasture Meter used consisted of a disc/plate made of acrylic plastic (plexiglass), with diameter of 45cm and weight of 1.5 kg and a calibrated metal stick that is 60 cm long. During sampling the plastic disc was dropped down along the metal stick from a height of approximately 60m and the settling grass height was observed and recorded. A quadrant frame of 0.25m^2 was placed at each sampling point and herbaceous cover was estimated to the nearest 10%.

Aboveground foliage of mixed grasses were also harvested from the quadrat frames using a pair of scissors and packaged in well labeled paper bags. The mixed grass samples were taken into laboratory at Masinde Muliro University of Science and Technology, where they were dried to a constant weight at 70^{o}C for 48 hours. The dry weights of the mixed grasses were recorded against the height of the mixed grasses in the field. These were used to develop a model for further estimation of grass standing crop biomass in each vegetation type.

3.5.1 Data processing

i) **Frequency**

The frequency of species i was determined using the following equation (Brower & Zar, 1990):

$$fi = {ji}/{k} \qquad \text{[3.15]}$$

Where fi = frequency of species i

ji = number of sampling points where species i was recorded

k = total number of points sampled

ii) **Relative frequency**

The relative frequency of species i was determined as follows (Brower & Zar, 1990):

$$Rfi = {}^{fi}/_{\sum f} \text{ .... } [3.16]$$

Where Rfi = relative frequency for species i

$\sum f$ = total of the frequencies of all species

iii) Calibration of Disc Pasture

The harvest method was used to calibrate the Disc Pasture used in measuring the grass standing biomass (Ganguli *et al.*, 2000) and regression analysis was used to develop a linear relationship between the grass height (cm) and the grass weight (g) (Figure 4.3). The regression equation developed was as follows:-

$$y = 0.054x + 2.515 \text{ .... } [3.17]$$

Where y = the measured grass biomass (g/m^2)

x = the grass height (cm)

R^2 = the coefficient of determination.

Grass height was a significant predictor of grass standing crop biomass ($p = 0.001$) (**Error! Reference source not found.**3).

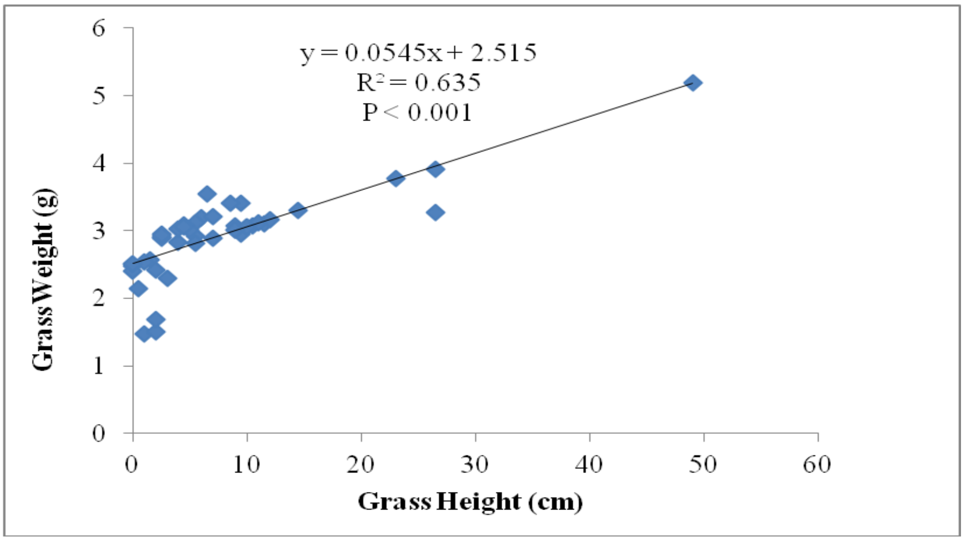


Figure 3.2: Relationship between grass height (cm) and grass weight (g) in Kimana Wildlife Sanctuary

3.6 Identification and counting of large herbivores

Large herbivores were identified and classified using field guidelines for African large mammals in Estes (1991) and Kingdon (1997). Counting was done between 8.00 am and 10.00 am by a team comprising of one researcher and two community rangers. Direct counts were carried out along 5 parallel line transects (Burnham *et al.*, 1980) of varying lengths that were located at intervals of 1km and oriented to transverse each vegetation type.

Garmin Global Positioning System 12, (GPS 12) handset was used to measure the length of each transect, to mark the observation point and to maintain a particular walking direction along the line transect. The perpendicular distance from the observer to the large herbivore or group of large herbivores was estimated using a laser rangefinder and the following information was recorded in the field each time a large herbivore or a group of large herbivores was spotted; the GPS coordinates of the exact location occupied by the observer, perpendicular sighting distance, species, vegetation type and total number observed. Animal age, sex, age structure and activity for large herbivore were also recorded although they were not used to make any calculations in this study. For livestock, similar information was recorded, but separation by sex and age was not done and mixed flocks of sheep and goats were collectively counted as shoats.

3.6.1 Data processing

i) Population density

Large herbivore counts from foot transects were separated into data for dry season and data for wet season. Animal densities (animals/km^2) for the observed large herbivores during the two seasons were estimated according to the following equation (Northon-Griffiths, 1978):-

$Di = {ni}/{a}$ …………………….. [3.18]

Where Di = population density of species i

ni = total number of individuals of species i

a = area of the sampled site

However, the total area of the sampled site was obtained using the following formulae;-

$a = l \ x \ w$ …………………………………………………………..... [3.19]

Where; = l = length of the foot transect

w = Width of the foot transect

ii) Habitat selectivity indices

Selection of various habitats by the large herbivores was determined using Savage selectivity index (Manly & Mcdonald, 1993) as follows:-

$wi = {}^{ui}/_{pi}$ ………………………………………………………… [3.20]

Where wi = Savage selectivity inde

ui = Proportion of observations recorded in a given habitat

pi = Proportion of that habitat against total available habitat

Savage selectivity index varies from 0 (maximum negative selection) to ∞ (maximum positive selection).

iii) Habitat niche breadth indices

Observation on how uniformly each species of large herbivore utilized habitat resources in the study area was determined using Levins' measure of niche breadth (Levin, 1968) as follows:-

$B = {}^{1}/_{\sum p^2 i}$ ………………………………………………………… [3.21]

Where B = Levins' measure of niche breadth

i = Resource type

p = Proportion of individuals using resource i

The Levins' scale was standardized as recommended by Grossman (1986) into a scale of 0 to 1 using the following equation:-

$$BA = {(B-1)}/{(N-1)} \text{ ……………………………………………….. [3.22]}$$

Where; BA = Standardized niche breadth

N = Total number of resource items for the species of interest

Niche breadth of large herbivores was considered low (0 - 0.39), intermediate (0.40 – 0.60) or high (0.61 - 1) as recommended by Grossman (1986).

iv) Habitat overlap indices

The degree to which various species of large herbivores overlapped in their use of the available habitats was measured using the Pianka's measure of niche overlap (Pianka, 1973) as follows:-

$$Ojk = {(\Sigma Pij * Pik}/{(P^2ij * P^2ik} \text{ …………………………………….. [3.23]}$$

Where; Ojk = Piankas' measure of overlap between species j and species *k*

Pij = the proportion that resource i is of the total resource used by species j

Pik = the proportion that resource i is of the total resource used by species *k*

For the purpose of seasonal comparisons, habitat niche overlap indices were categorized into dry season and wet season. Piankas' overlap index ranges between 0 (no overlap) to 1 (complete overlap). Overlap of habitat use between the large herbivores was considered low (0-0.39), intermediate (0.4-0.6) or high (0.61-1) as recommended by Grossman, (1986). Comparisons between the dry and the wet season were obtained using frequency distributions of the habitat overlap values among the large herbivores.

3.7 Data analysis

The standing crop biomass, inter-tuft distance, grass cover and grass height were tested for normality and homogeneity of variance using Shapiro-Wilks test ($p \leq 0.05$) and Levene's test ($p \leq 0.05$) respectively. The data for standing crop biomass was not normally distributed ($p \leq 0.001$) and homogeneity of data was violated ($p \leq 0.001$), hence, the data was log ($\log_{10}$) transformed to reduce heteroscedasticity.

One-Way ANOVA, at 5% level of significance was used to test for statistical difference in grass standing crop biomass, inter-tuft distance, grass cover and grass height across vegetation types. Post hoc analysis for variables with significant differences was carried out using Tukey Honestly Significant Difference (HSD) ($P \leq 0.05$).

An independent t-test ($P \leq 0.05$) was used to compare mean densities for the large herbivores between the dry and the wet seasons.

CHAPTER FOUR

RESULTS

4.1Characterization, classification and mapping of vegetation

Five distinct vegetation types were characterized, classified and mapped using remote sensing method in KWS (Figure 4.1). These were woodlands, wooded grassland, sparse shrubland, grassland and herb grassland.

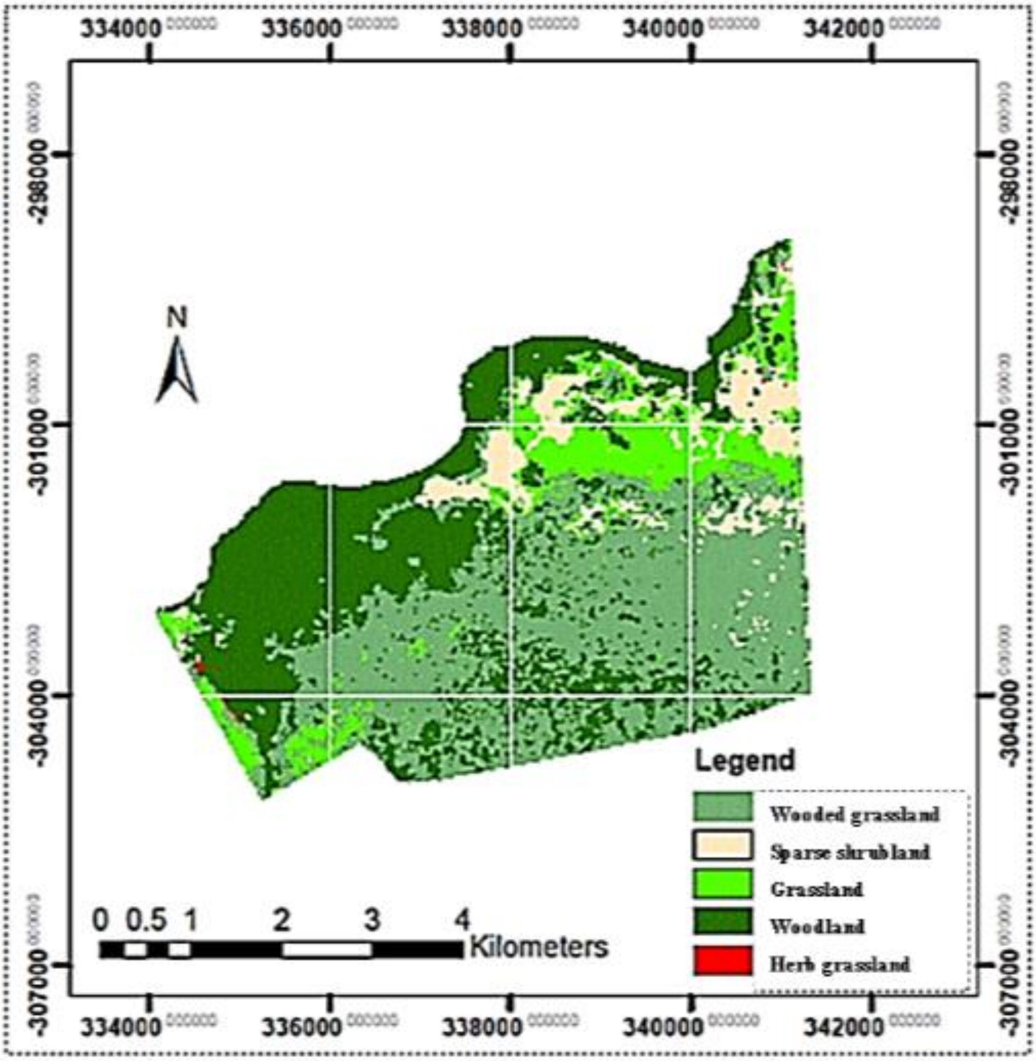


Figure 4.1: Spatial distribution of vegetation types in Kimana Wildlife Sanctuary (Source: Author, 2016)

However, further classification based on physiognomic features and dominant species (Pratt & Gwynne, 1977) resulted into 7 vegetation types, which included *A. tortilis* woodland, *A. xanthophloea* woodland, wooded grassland, sparse shrubland, *S. fimbriatus* grassland, *C. ciliaris* grassland and herb grassland (Table 4.1).

Table 4.1: Classified vegetation types based on the physiognomic features and dominant species in Kimana Wildlife Sanctuary

Mapped Vegetation	**Characterized Vegetation**	**Dominant life Form**	**Canopy cover (%)**	**Herbaceous cover (%)**
Woodland (W)	*Acacia tortilis* woodland	Trees	22.42	15
	A. xanthophloea woodland	Trees	34.38	70
Wooded grassland (WG)	Wooded grassland	Trees	34. 39	18.5
Shrubland (S)	*Sparse shrubland*	Shrubs	8.8	2.78
Grasslands (G)	*S. fimbriatus* grassland	Grasses	-	21
	C. ciliaris grassland	Grasses	-	26.47
Herb grassland (HG)	Herb grassland	Herbs	**-**	**-**

Acacia tortilis woodland was dominated by *A. tortilis* species, with canopy cover of 22.42% and herbaceous cover of 15%. *Acacia xanthophloea* woodland was dominated by *A. xanthophloea* species, with canopy cover of 34.38% and herbaceous cover of 70%. Wooded grassland was dominated by *A. tortilis*, with canopy cover of 34.39% and herbaceous cover of 18.5%. Sparse shrubland was dominated by *B. glabra* shrub with canopy cover and herbaceous cover of 8.8% and 2.78% respectively. *Sporolobus* *fimbriatus* grassland was dominated by *S. fimbiratus* grass with herbaceous cover of 21% and *C. ciliaris* grassland was dominated by the *C. ciliaris* grasses with herbaceous cover of 26.47%. Herb grassland was not characterized using the physiognomic characteristics and dominant plant species because it was too small to be recognized during the field survey.

4.2 Composition and structure of woody plants

4.2.1 Woody species richness, diversity (H′) and evenness (J)

A total of 17 woody species belonging to 7 different families were recorded in the sanctuary (**Error! Reference source not found.**. However, their composition differed across the vegetation types with 7 woody species belonging to 6 different families occurring in wooded grassland; *A. tortilis* (family Fabaceae), *B. glabra* (family Balanitaceae), *S. persica* (family Salvadoraceae), *C. monoica* (family Boraginaceae), *Comiphora africana* (family Burseraceae), *Maerua edulis* (family Capparidaceae) and *Azima tetracantha* (family Boraginaceae).

In *A. tortilis* woodland, 5 woody plants belonging to 5 different families were recorded; *A. tortilis* (family fabaceae), *B. glabra* (family Balanitaceae), *Lycium europeaum* (family Solanaceae) and *Cordia monoica* (family Boraginaceae).

In sparse shrubland, 4 woody plants belonging to 3 different families were recorded; *A. tortilis* (family Fabaceae), *B. glabra* (family Balanitaceae), *C. monoica* (family Boraginaceae), *Acacia drepanolobium* (family Fabaceae) and *Acacia mellifera* (family fabaceae). *Acacia xanthophloea* woodland recorded only 1 woody species, *Acacia xanthophloea* (family Fabaceae).

Shannon-Weiner diversity indices (H′) for woody plants ranged from 0 to 1.48 (Table 4.2). Wooded grassland had the highest diversity (H′ = 1.48), *A. tortilis* woodland had the lowest diversity (H′ = 0.98) and *A. xanthophloea* woodland was not diverse. Species evenness (J) ranged from 0 to 0.76 and wooded grassland had the highest species evenness (0.76) and species evenness in *A. xanthophloea* woodland was zero.

Table 4.2: Woody species diversity and evenness in different vegetation types in Kimana Wildlife Sanctuary

Vegetation Type	**Species Richness**	**Species diversity (H′)**	**Species evenness (J)**
Wooded grassland	7	1.48	0.76
Acacia tortilis woodland	5	0.98	0.06
Sparse shrubland	4	1.36	0.48
Acacia xanthophloea woodland	1	0	0

4.2.2 Similarity in the composition of woody species

Similarity in the composition of woody species between vegetation types ranged from 0 to 0.98 (**Error! Reference source not found.**3). Species similarity was higher between *A. tortilis* woodland and wooded grassland (0.93).

Table 4.3: Similarity in the composition of woody species between vegetation types in Kimana Wildlife Sanctuary

Vegetation type	**ATW**	**WG**	**SSL**	**AXW**
ATW	1.0			
WG	0.93	1.0		
SSL	0.85	0.58	1.0	
AXW	0	0	0	1.0

Key: *Acacia tortilis* woodland (ATW); wooded grassland (WG); sparse shrubland (SSL); *Acacia xanthophloea* woodland (AXW)

The second highest index was observed between *A. tortilis* woodland and sparse shrubland (0.85). Wooded grassland and sparse shrubland pairs were third, with a similarity index of 0.58. However, no similarity was observed between *A. xanthophloea* woodland and the other vegetation types; *A. tortilis* woodland, wooded grassland and sparse shrubland (**Error! Reference source not found.**3).

4.2.3 Importance Values Index (IVI) of woody species

Importance value indices for woody species in *A. tortilis* woodland ranged from 5.24 to 159.4 with *A. tortilis* recording the highest IVI, 159.4 followed by *B. glabra*, 51.33. *Salvadora persica* had an IVI of 17.66 while both *L. europeaum* and *C. monoica* had IVI of 5.24 (Table 4.4).

In wooded grassland, IVI of woody species ranged from 4.71 to 118.58. *Acacia tortilis* recorded the highest IVI, 118.58 while *S. persica*, *B. glabra*, *C. africana*, M. edulis, *A. tetracantha* and *C. monoica* recorded IVI of 36.54, 20.5, 9.98 and 4.71 respectively (Table 4.4).

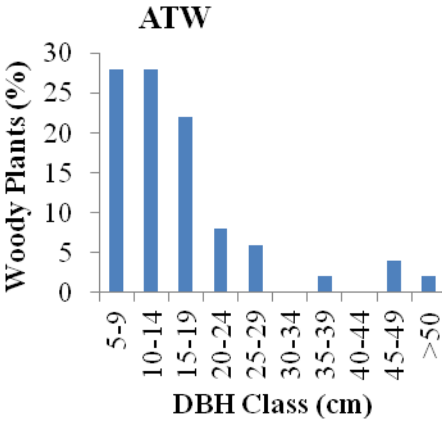
In sparse shrubland, IVI for woody species ranged between 6.68 and 82.79 with *B. glabra* recording the highest IVI, 82.79. Both *A. tortilis* and *A. drepanolobium* recorded an IVI of approximately 66 while both *A. mellifera* and *C. monoica* recorded an IVI of 6.68. *Acacia xanthophloea* species in *A. xanthophloea* woodland contributed an IVI of 225 (Table 4.4).

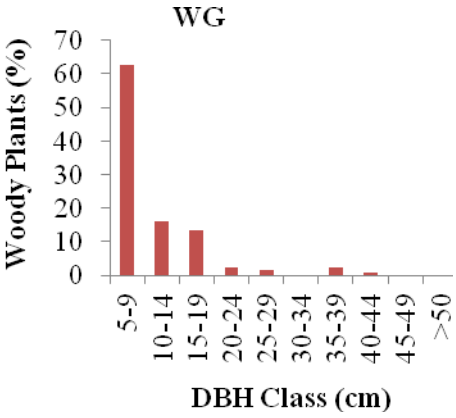
Table 4.4: Importance Value Indices (IVI) of woody species in different vegetation types in Kimana Wildlife Sanctuary

Vegetation Type	**Species**	**Relative density**	**Relative dominance**	**Relative frequency**	**IVI**
***Acacia tortilis* Woodland**	*Acacia tortilis*	20.83	85.41	53.21	159.45
	Balanites glabra	11.11	11.98	28.21	51.3
	Salvadora persica	4.16	2.6	10.9	17.66
	Lycium europeaum	1.39	0	3.85	5.24
	Cordia monoica	1.39	0	3.85	5.24
Wooded Grassland	*Acacia tortilis*	34.29	63.73	20.56	118.58
	Salvadora persica	37.14	25.48	37.78	100.4
	Balanites glabra	14.29	4.47	17.78	36.54
	Comiphora africana	7.14	4.47	8.89	20.5
	Maerua edulis	2.86	1.01	6.11	9.98
	Azima tetracantha	2.86	0.43	6.11	9.4
	Cordia monoica	1.43	0.4	2.88	4.71
Sparse shrubland	*Balanites glabra*	15.9	40.23	26.66	82.79
	Acacia tortilis	0.09	46.64	20	66.73
	Acacia drepanolobium	13.6	13.12	39.88	66.6
	Acacia Mellifera	0.022	0	6.66	6.68
	Cordia monoica	0.022	0	6.66	6.68
***Acacia xanthophloea* Woodland**	*Acacia xanthophloea*	25	100	100	225

4.2.4 Size class distribution of woody plants

The proportions of occurrence of woody plant stems with dbh ≥ 5cm differed across vegetation types (Figure 4.2). These were shown to be decreasing with increasing dbh size classes in *A. tortilis* woodland, wooded grassland and sparse shrubland, with a higher proportion of stems occurring in the smaller dbh size classes and a lower proportion of stems or their absence in the higher dbh size classes (Figure 4.2). Contrary, the frequency of woody plant stems were shown to be increasing with increasing dbh size classes in *A. xanthophloea* woodland, with a higher proportion of stems occurring in higher dbh size classes than in the lower dbh classes (Figure 4.2).





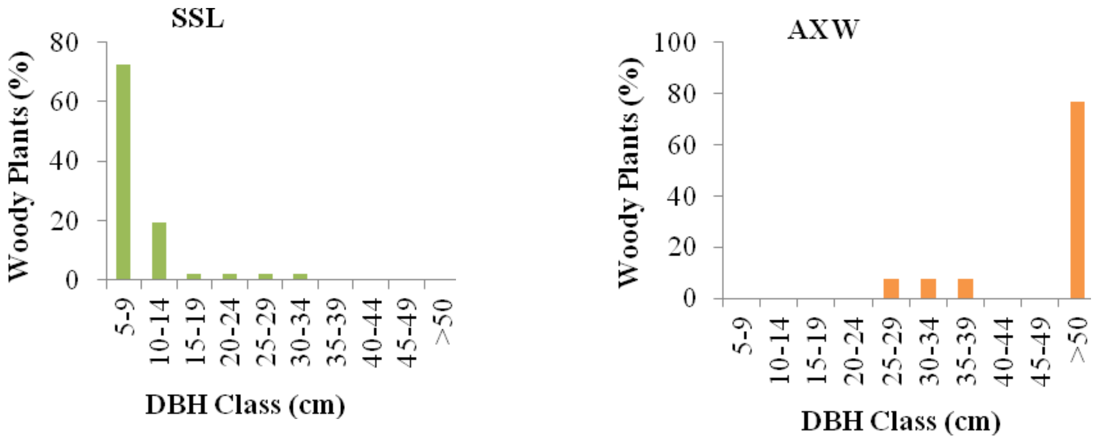


Figure 4.2: Size class (cm) distribution of woody plants in different vegetation types in Kimana Wildlife Sanctuary

Key*:* *Acacia tortilis* woodland (ATW); wooded grassland (WG); sparse shrubland (SSL); *Acacia xanthophloea* woodland (AXW)

4.3 Structure and composition of herbaceous vegetation

4.3.1 Ecological composition of grasses

A total of eight (8) species of grasses belonging to three ecological categories; decreaser, increaser I and increaser II grass species were recorded in the study area (Table 4.5). *Cenchrus ciliaris* was the only decreaser grass species encountered in the study area and occurred in wooded grassland and *S. fimbriatus* grassland. Three species of increaser I grass species occurred in the sanctuary namely; *P. mezaneum, P. perpureum* and *P. stramineum*. Similarly, three species of increaser II grass species occurred in the sanctuary namely; *Cynodon dactylon, Harpachne schimperii and Eragrotis tenuifolia* (Table 4.5)

Table 0.5: Composition of herbaceous vegetation in different vegetation types in Kimana Wildlife Sanctuary

Herbaceous plant species	**Ecological category**	**ATW (%)**	**WG (%)**	**SFG (%)**	**SSL (%)**	**AXW (%)**	**CDG (%)**
Cenchrus ciliaris	Decreaser	0	5.1	0	0	0	41.05
Decreaser total		**0**	**5.1**	**0**	**0**	**0**	**41.05**
Pennisetum mezaneum	Increaser I	0	2.24	0	0	0	0
Pennisetum perpureum	Increaser I	0	3.06	0	0	0	0
Pennisetum stramineum	Increaser I	11.81	23.06	0	1.58	0	0
Increaser I total		**11.81**	**28.36**	**0**	**1.58**	**0**	**0**
Cynodon dactylon	Increaser II	86.11	0	0	0.4	0	49.47
Harpachne schimperii	Increaser II	0	3.06	0	7.91	0	0
Sporolobus fimbriatus	Increaser II	0.69	8.37	21.81	0	67.8	9.47
Eragrotis tenuifolia	Increaser II	0	0	0	3.95	0	0
Increaser II total		**86.8**	**11.43**	**21.81**	**12.26**	**67.8**	**58.94**
χ^2 – Value		**131.42**	**28.97**	**0.087**	**12.25**	**13.828**	**26.95**
P – Value		**0.001**	**0.001**	**0.768**	**0.002**	**0.001**	**0.001**

Key*:* *Acacia tortilis* woodland (ATW); wooded grassland (WG); *Sporolobus fimbriatus* grassland (SFG); sparse shrubland (SSL); *Acacia xanthophloea* woodland (AXW) and *Cynodon dactylon* grassland (CDG)

In *A. tortilis* woodland, the proportions for the ecological categories of grasses were 0% increaser I grasses, 11.81% decreaser grasses and 86.8% of the increaser II grasses. However, increaser II grass species were significantly more abundant compared with increaser I grass species (χ^2 = 56.818; df = 1; p = 0.001), where the proportion of the increaser II grass species being 7 times higher compared with a lower value of 11.81% decreaser I grass species (Table 4.5).

In wooded grassland, the frequency of occurrence of ecological grass categories was 0%, 5.10% and 28.36% of the increaser II grasses, decreaser grasses and the increaser I grasses respectively. However, increaser I grass species were significantly more abundant compared with the other grass species (χ^2 = 19.409; df = 2; p = 0.001) (Table 4.6).

In sparse shrubland, the proportions of ecological categories of grasses were 0%, 1.58 % and 12.26% of the decreaser, increaser I grasses and increraser II grasses respectively. However, their proportions differed significantly (χ^2 = 7.413; df = 1; p = 0.002), where the proportion of increaser II grass species being 11 times higher compared with a lower value of 1.58% of increaser I grass species (Table 4.6).

In *S. fimbriatus* grassland, the proportion of grass ecological categories was 0%, 0% and 21.8% decreaser, increaser I and increaser II respectively and these did not differ significantly ($p \geq 0.05$) (Table 4.6).

In *C. dactylon* grassland, the proportions of ecological categories was 41.05% and 58.94% of the decreaser and the increaser II grass species respectively, which did not differ significantly (χ^2 = 26.95; df = 1; p = 0.001) as tested using Chi-square test of fitness (Table 0.). *Acacia xanthophloea* woodland was dominated by one grass species, *S. fimbriatus*, which had a higher frequency of 67.80% (Table 4.6).

4.3.2 Grass standing crop biomass

Grass standing crop biomass mean values ranged from 367.80 ± 46.35kg/ha to 3093.10 ± 582.79kg/ha in sparse shrubland and wooded grassland respectively (Table

4.6). *Acacia tortilis* woodland recorded the second highest grass standing crop biomass, 1510.30 ± 263.58kg/ha. *Acacia xanthophloea* woodland, *S. fimbriatus* grassland and *C. dactylon* grassland had mean values of 2255.20 ± 262.67kg/ha, 1273.00 ± 242.873kg/ha and 536.84 ± 46.91kg/ha respectively (Table 4.6).

Table 0.6: Mean grass standing crop biomass, inter-tuft distance, grass height and grass cover in different vegetation types in Kimana Wildlife Sanctuary

Vegetation Type	Grass standing crop (kg/ha)	Inter-tuft distance (cm)	Grass height (cm)	Grass cover (%)
WG	3093.10 ± 582.79^{b}	8.65 ± 0.57^{ab}	7.54 ± 1.28^{b}	18.50 ± 10.00^{b}
AXW	2255.20 ± 262.67^{ab}	4.90 ± 0.81^{ab}	7.54 ± 1.87^{b}	70.00 ± 3.66^{bc}
ATW	1510.30 ± 263.58^{a}	10.56 ± 0.75^{b}	3.73 ± 0.56^{b}	15.00 ± 4.01^{b}
SFG	1273.00 ± 242.89^{a}	9.93 ± 1.42^{ab}	4.01 ± .69^{ab}	21.00 ± 2.45^{a}
CDG	536.61 ± 0.84^{a}	9.47 ± 0.37^{ab}	10.71 ± 1.48^{a}	26.47 ± 5.37^{c}
SSL	367.80 ± 46.35^{a}	15.81 ± 1.81^{c}	2.70 ± 0.36^{ab}	2.78 ± 1.21^{ab}

Means with different superscript letter within the same column differ significantly (Tukey HSD test p < 0.05).

Key: Wooded grassland (WG); *Acacia xanthophloea* woodland (AXW); *Acacia tortilis* woodland (ATW); *Sporolobus fimbriatus* grassland (SFG); *Cynodon dactylon* grassland (CDG) and sparse shrubland (SSL).

One-Way ANOVA revealed a significant difference in the mean grass standing crop biomass across vegetation types (F = 13.11; df = 5, 334; p = 0.007). Tukey HSD revealed a significantly higher mean grass standing crop biomass in wooded grassland compared with lower values in sparse shrubland, 367.80 ± 46.35kg/ha (p = 0.018), *C. dactylon* grassland, 537.84 ± 47.53kg/ha (p = 0.004) and *S. fimbriatus* grassland, 1164.10 ± 230.15kg/ha (p = 0.032).

4.3.3 Inter-tuft distance between the grass swards

The mean inter-tuft distance between the grass swards ranged from 4.90 ± 0.8cm to 15.81 ± 1.81cm in *A. xanthophloea* woodland and sparse shrubland respectively (Table 4.6). *Acacia tortilis* woodland, wooded grassland, *S. fimbriatus* grassland and *C. dactylon* grassland had mean inter-tuft distance of 10.56 ± 0.75cm, 8.65 ± 0.57cm, 9.93 ± 1.42cm and 9.47 ± 0.37cm respectively. The mean inter-tuft distance varied significantly between the vegetation types (F = 5.699; df = 5,662; p = 0.001). Tukey HSD test found a significantly higher mean inter-tuft distance in shrubland, 15.81 ± 1.81cm, compared with lower values of 4.90 ± 0.8cm in *A. xanthophloea* woodland, (p = 0.001), 8.65 ± 0.57cm in wooded grassland, (p = 0.001), 9.47 ± 0.37cm in *C. dactylon* grassland (p = 0.014), 9.93 ± 1.42cm in *S. fimbriatus* grassland (p = 0.033) and 10.56 ± 0.75cm in *A. tortilis* woodland (p = 0.047). Similarly, mean inter-tuft distance between the grass swards in *A. tortilis* woodland was two times higher compared with lower values of 4.90 ± 0.8cm in *A. xanthophloea* woodland (p = 0.009) (Table 4.6).

4.3.4. Grass basal cover

The Mean grass cover ranged from 2.78 ± 1.21% to 70.00 ± 3.66% in sparse shrubland and *A. xanthophloea* woodland respectively (Table 4.6). The difference in mean grass basal area differed across vegetation communities differed significantly (F = 12.21; df = 5, 53; p = 0.01). Tukey HSD test revealed a significant higher mean grass basal cover in *A. xanthophloea* woodland compared with *A. tortilis* woodland (p = 0.01), wooded grassland (p = 0.01), *S. fimbritus* grassland (p = 0.01), sparse shrubland (p = 0.01) and a higher mean grass basal cover in wooded grassland compared with sparse shrubland (p = 0.043).

4.3.5 Grass height

Mean grass height ranged from 3.73 ± 0.56 to 10.71 ± 1.54 in *S. fimbriatus* grassland and *A. xanthophloea* woodland respectively (Table 4.6). However, the mean heights differed significantly across the vegetation types (F = 10.33; df = 5, 295; p = 0.001). Tukey HSD, revealed a significantly higher mean height *in A. tortilis* woodland compared with *S. fimbriatus* grassland (p = 0.008), in wooded grassland compared with *S. fimbriatus* grassland (p = 0.01), *C. dactylon* grassland (p = 0.001) and sparse shrubland and in *A. xanthophloea* woodland compared with *S. fimbriatus* grassland (p = 0.01), sparse shrubland (p = 0.002) and *C. dactylon* grassland (p = 0.01).

4.4 Population size and density of large herbivores in Kimana Wildlife Sanctuary

Sixteen species of large herbivores were encountered, which included African elephant, Cape buffalo, impala, Plains zebra, Thompson’s gazelle, Grant’s gazelle, warthog, wildebeest, common waterbuck, Cape eland, Bohor reedbuck, Maasai giraffe, cattle, donkeys, goats and sheep (Table 4.7).

Table 0.7: Seasonal population sizes and densities (animals/km^2) of large herbivore in the Kimana Wildlife Sanctuary

	Population size		**Population density**		
	Seasons				
Large herbivores	**Dry**	**Wet**	**Dry**	**Wet**	**P-value**
Wild herbivores					
Plains zebra	488	456	6.80 ± 0.87	6.47 ± 0.85	0.791
Grant's gazelle	262	248	3.37 ± 0.53	3.59 ± 0.59	0.78
Impala	261	221	3.36 ± 0.80	3.19 ± 1.20	0.904
Cape eland	218	4	2.85 ± 1.15	0.05 ± 0.05	0.024*
Maasai giraffe	146	89	2.03 ± 0.56	1.33 ± 0.43	0.338
Warthog	103	67	1.60 ± 0.47	1.15 ± 0.33	0.449
Thomson's gazelle	100	111	1.07 ± 0.31	1.58 ± 0.30	0.25
Wildebeest	76	143	1.03 ± 0.49	1.89 ± 0.52	0.238
Common waterbuck	30	51	0.39 ± 0.22	0.70 ± 0.33	0.445
Bohor reedbuck	10	15	0.29 ± 0.167	0.22 ± 0.10	0.718
African elephant	5	52	0.07 ± 0.06	0.77 ± 0.53	0.186
Cape buffalo	0	15	0	0.22 ± 0.15	0.148
Sub total	**1699**	**1472**	**1.91 ± 0.21**	**1.76 ± 0.20**	**0.616**
Domestic herbivores					
Cattle	411	388	5.12 ± 2.31	5.58 ± 2.00	0.883
Shoats	115	25	1.59 ± 0.88	0.32± 0.32	0.195
Donkeys	10	0	0.12 ± 0.08	0	0.126
Sub total	**536**	**413**	**2.28 ± 0.86**	**1.97 ± 0.70**	**0.784**

Total	**2235**	**1885**	**1.98 ± 0.24**	**1.80 ± 0.22**	**0.582**

Significance difference (p ≤ 0.05) between means (M ± SE) for large herbivore densities between seasons are indicated by (*)

A total of 3683 large herbivores were counted in the sanctuary, with 2153 and 1530 large herbivores counted during the dry and wet seasons respectively (Table 4.8). The population of large wild herbivores was 1699 and 1472 during the dry and the wet seasons respectively, while that of large domestic herbivores were 536 and 413 during the dry and the wet seasons respectively (Table 4.7).

Mean population densities for all the large herbivore were 1.98 ± 0.241and 1.80 ± 0.22 during the dry and the wet season respectively (Table 4.7). However, the mean values did not differ significantly between the two seasons ($p > 0.05$). The mean population densities for the wild large herbivores was 22.85 individuals/km^2 and 21.15 individuals/km^2 during the dry and the wet seasons respectively (

Table 0.). However, for domestic large herbivore the densities were 2.28 ± 0.86 and 1.97 ± 0.70 during the dry and the wet seasons respectively (Table 4.7). The differences in the mean population densities for both wild and domestic large herbivores were not significant during the two study seasons ($p \geq 0.05$) as tested using independent t-test.

Plains zebras recorded the highest population density in both dry and wet seasons and African elephants recorded the lowest mean population density during the dry season. Cape buffalo and Bohor reedbuck recorded the lowest during the wet season (Table 4.7). Cape buffaloes and donkeys were not encountered in the study area during the dry season and the wet season respectively (Table 4.7).

Comparisons of the mean population densities for individual large herbivore species did not differ significantly for all the species, except, for Cape eland, which was significantly higher in the dry season than the wet season ($t = 2.36$; $df = 35$; $p = 0.02$) (Table 4.7).

4.5 Seasonal habitat selectivity pattern between large herbivores

Cape eland, African elephants, Grant's gazelles, Plains zebras, wildebeests, Thomson's gazelle, Cape buffaloes, Bohor reedbuck, warthog, cattle and donkeys maintained high habitat selectivity throughout the dry and wet seasons (Table 4.8;

Table 4.9). During both dry and the wet seasons, Plains zebras and wildebeests used wooded grassland and *S. fimbriatus* grassland, Cape eland, African elephant and Grant's gazelle used wooded grassland while warthogs used wooded grassland and *C. dactylon* grassland (Table 4.8; Table 4.9). Thomson's gazelles used *S. fimbriatus* grassland and sparse shrubland during the dry season, but used wooded grassland and sparse shrubland during the wet season (Table 4.8; Table 4.9).

Bohor reedbucks used only sparse shrubland during the dry season (Table 4.8). However, they used sparse shrubland and wooded grassland in the wet season (Table 4.9). Cape buffaloes selected wooded grassland in the wet season, but they were absent in the sanctuary in the dry season (Table 4.8; Table 4.9). Cattle and donkeys selected wooded grassland and shrubland during the dry season, but in the wet season, cattle selected shrubland, while donkeys were absent in the sanctuary.

Impalas constantly maintained low habitat selectivity patterns during the dry and wet seasons through the use of a wide range of vegetation types. During the dry season, they selected wooded grassland, sparse shrubland and *C. dactylon* grassland, but selected *A. tortilis* woodland, wooded grassland, sparse shrubland and *A. xanthophloea* woodland during the wet season.

Common waterbucks, Maasai giraffes and shoats varied their selectivity for habitats during the dry and the wet seasons (Table 4.8; Table 4.9). Common waterbucks exhibited high habitat selectivity during the dry season by preferring a narrow range of vegetation types, which included *S. fimbriatus* grassland and sparse shrubland (Table 4.8; Table 4.9), but exhibited low selectivity during the wet season by utilizing a wider range of vegetation types, which included wooded grassland, *S. fimbriatus* grassland and sparse shrubland.

Contrary, Maasai giraffes exhibited low selectivity during the dry season by utilizing a wide range of vegetation types, which included wooded grassland, *S. fimbriatus* grassland and sparse shrubland (Table 4.8; Table 4.9), but exhibited high selectivity during the wet season by preferring a narrow range of vegetation types which included wooded grassland and *C. dactylon* grassland. Similarly, shoats were observed to be less selective during the dry season, utilizing a wide range of vegetation types including, sparse shrubland, *A. xanthophloea* woodland and *C. dactylon* grassland, but in the wet season they were highly selective utilizing sparse shrubland only (Table 4.8; Table 4.9).

Table 0.8: Savage indices of habitat selection by large herbivores during the dry season in Kimana Wildlife Sanctuary

	Habitat type					
Large herbivore	**ATW**	**WG**	**SFG**	**SSL**	**AXW**	**CDG**
Cape buffalo	-	-	-	-	-	-
Cape eland	0.47	4.93	0.03	0.00	0.00	0.00
African elephant	0.00	6.17	0.00	0.00	0.00	0.00
Maasai giraffe	0.33	1.78	1.36	3.19	0.41	0.79
Grant’s gazelle	0.46	3.77	0.76	0.00	0.16	0.93
Impala	1.09	1.14	0.06	2.22	1.19	0.25
Bohor reedbuck	0.00	0.62	0.00	10.23	0.00	0.00
Thomson's gazelle	0.76	0.99	2.87	1.48	0.00	0.00
Warthog	0.58	1.80	0.36	0.00	0.42	4.73
Common waterbuck	0.00	0.41	0.00	4.92	4.31	0.00
Wildebeest	0.00	3.17	3.58	0.00	0.00	0.00
Plains zebra	0.17	4.09	1.18	0.54	0.00	0.76
Cattle	0.22	3.15	0.00	4.51	0.00	0.00
Donkeys	0.95	3.09	0.00	1.14	0.00	0.00
Shoats	0.00	0.00	0.00	5.04	2.40	3.57
Frequency (%)	**7.14**	**71.43**	**28.57**	**57.14**	**21.43**	**14.29**

Key: *Acacia tortilis woodland (ATW); Wooded grassland (WG); Sporolobus fimbriatus grassland (SFG); Sparse shrubland (SSL); Acacia xanthophloea woodland (AXW) and Cynodon dactylon grassland (CDG)*

Table 0.9: Savage indices of habitat selection by large herbivore during the wet season in Kimana Wildlife Sanctuary

	Habitat type					
Large herbivore	**ATW**	**WG**	**SFG**	**SSL**	**AXW**	**CDG**
Cape buffalo	0.00	6.17	0.00	0.00	0.00	0.00
Cape eland	0.00	6.17	0.00	0.00	0.00	0.00
African elephant	0.00	5.82	0.00	0.00	0.50	0.00
Maasai giraffe	0.96	1.39	0.00	0.00	0.58	3.89
Grant’s gazelle	0.22	4.01	0.86	0.82	0.59	0.00
Impala	0.19	2.51	0.00	2.98	0.82	1.33
Bohor reedbuck	0.00	3.29	0.00	5.30	0.00	0.00
Thomson's gazelle	0.17	4.12	0.33	1.33	0.47	0.58
Warthog	0.04	3.96	0.22	0.00	0.51	3.25
Common waterbuck	0.00	2.90	1.01	0.45	3.04	0.00
Wildebeest	0.00	3.58	3.03	0.00	0.00	0.09
Plains zebra	0.03	4.13	1.11	0.87	0.47	0.45
Cattle	0.28	0.29	0.00	5.24	0.13	4.59
Donkeys	-	-	-	-	-	-
Shoats	0.00	0.00	0.00	11.36	0.00	0.00
Frequency (%)	**0**	**85.71**	**21.42**	**35.71**	**7.14**	**28.57**

Key: *Acacia tortilis woodland (ATW); Wooded grassland (WG); Sporolobus fimbriatus grassland (SFG); Sparse shrubland (SSL); Acacia xanthophloea woodland (AXW) and Cynodon dactylon grassland (CDG)*

The frequency distribution of habitat selection by the large herbivores indicates high selection for wooded grassland in the dry (≈71%) and in wet (≈85%) seasons, followed by sparse shrubland with frequency distribution of (≈57%) in the dry season and (≈35%) in the wet season. *Acacia tortilis* woodland, *S. fimbriatus* grassland, *C. dactylon* grassland and *A. xanthophloea* woodland had frequency distribution of less than 30% in the dry and the wet seasons (Figure 4.3).

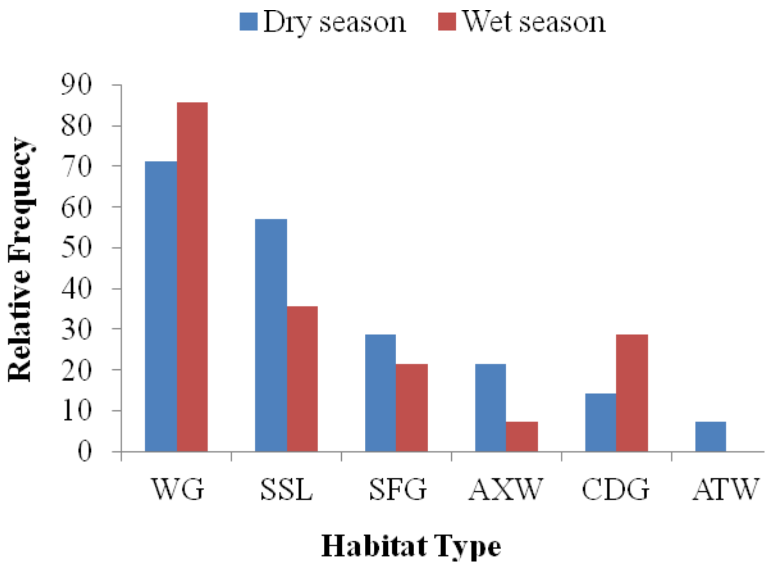


Figure 0.3: Frequency (%) distribution of large herbivores among different habitats during the dry and wet seasons in Kimana Wildlife Sanctuary

Key: *Acacia tortilis* woodland (ATW); wooded grassland (WG); *Sporolobus fimbriatus* grassland (SFG); sparse shrubland (SSL); *Acacia xanthophloea* woodland (AXW) and *Cynodon dactylon* grassland (CDG)

4.6 Seasonal niche breadth indices for large herbivores

Habitat niche breadth values for the large herbivores ranged from 0 to 0.71 during the dry season and from 0 to 0.53 during the wet season, with Maasai giraffes (0.71) and impalas (0.47) recording the highest values during the dry and the wet seasons respectively (Figure 4.4).

Several large herbivores maintained low habitat niche breadth values (0 to 0.39) during the dry and wet seasons. However, niche breadths for Maasai giraffes (0.71), warthogs (0.50), impalas (0.47) and Thomson's gazelle (0.47) were generally high during the dry season than the wet season for the Maasai giraffes (0.4), warthogs (0.22), impalas (0.53) and Thomson's gazelle (0.47).

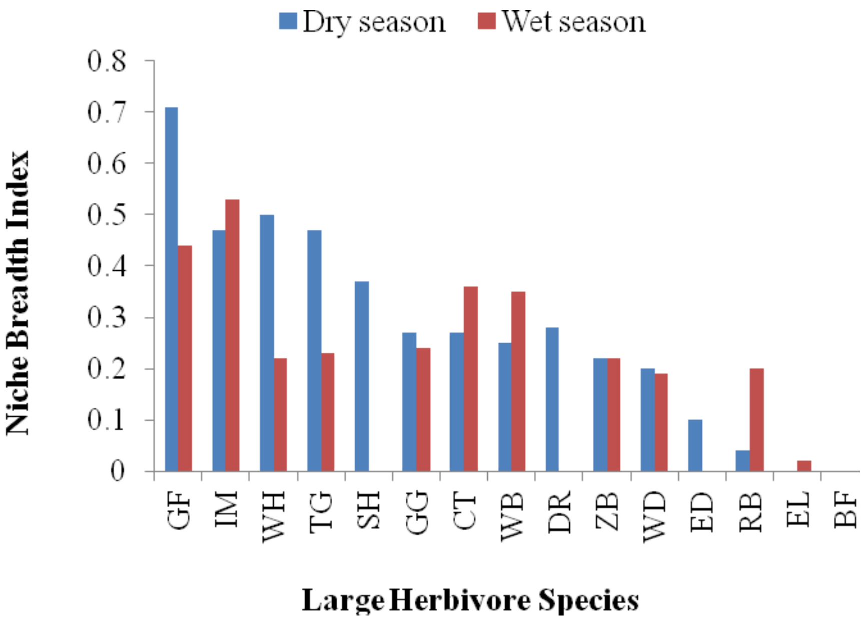


Figure 4.4: Levin's indices of habitat niche breadth among large herbivores in Kimana Wildlife Sanctuary

Key: Masaai giraffe (GF), impala (IM), warthog (WH), Thomson's gazelle (TG), Shoats (SH), Grant's gazelle (GG), Cattle (CT), Common waterbuck (WB), donkey (DR), Plains zebra (ZB), wildebeest (WLB), Cape eland (ED), Bohor reedbuck (RB), African elephant (EL) and Cape buffalo (BF).

Niche breadth values at low level (0 to 0.39) were generally more frequent among the large herbivores, occurring at approximately 73% and 87% of the species during the dry and wet seasons respectively (Figure 4.5). Intermediate level (0.40 to 0.60) was less frequent among species, with frequency distribution of approximately 20% and 13% during the dry and wet seasons respectively. Similarly, high level (0.60 to 1) was less frequent among species during the dry season, occurring in approximately 7% of the species.

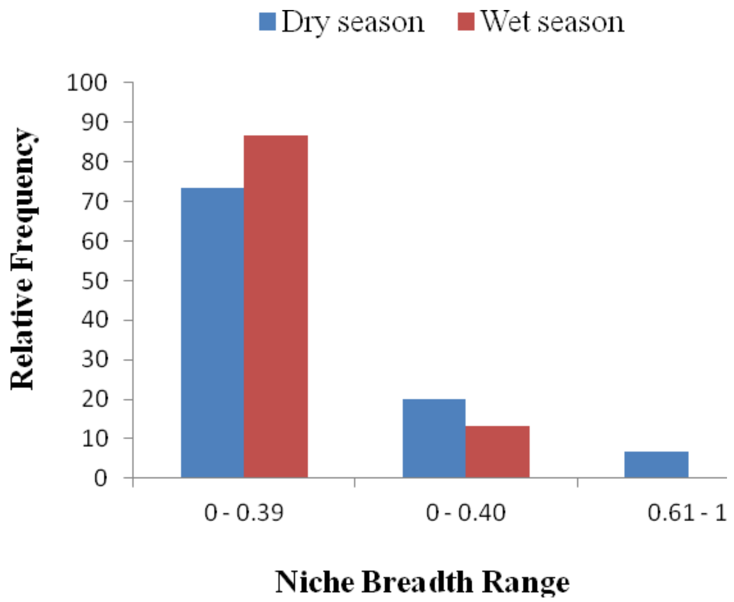


Figure 4.5: Frequency (%) distribution of the Levin's indices of habitat niche breadth among the large herbivores in Kimana Wildlife Sanctuary

During the dry season, habitat niche breadth mean values were 0.71, 0.48 and 0.20 at high, intermediate and low levels respectively (Figure 4.4.6). However, during the wet season, the mean values were 0.49 ± 0.05 and 0.15 ± 0.04 at intermediate and low levels respectively. These mean values differed significant during the dry season (F = 14.814; df = 2, 11; p = 0.001) and the wet season (F = 10.297; df = 1, 12; p = 0.008). Post Hoc tests were not performed for these results because the high level range had less than two cases during the dry and wet seasons (Figure 4.6).

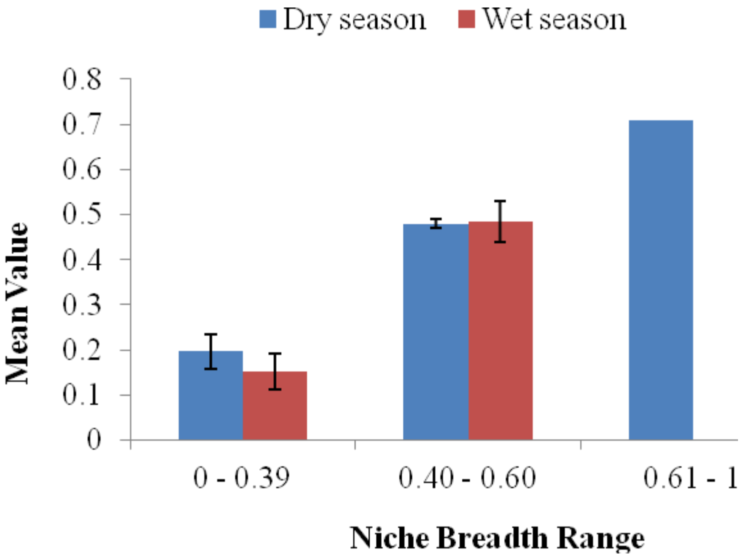


Figure 4.6: Means of the Levin's indices of niche breadth among large herbivores in Kimana Wildlife Sanctuary

4.7 Seasonal habitat overlap among the large herbivores

Overlap in habitat use among coexisting large herbivores were observed to occur in a continuum of low (0 to 0.39), intermediate (0.4 to 0.60) and high (0.61 to 1) throughout the dry and the wet seasons, with values ranging from 0 to 0.97 and 0 to 1 recorded during the dry and the wet seasons respectively (Table 4.10). During the dry season, highest overlap index was observed between elands and elephants, 0.97 and the lowest was observed between shoats and Grant's gazelles, 0.06. However, no overlap occurred between the following species: shoats and Cape elands, 0, shoats and African elephants, 0, shoats and Cape buffaloes, 0 and between shoats and wildebeest, 0.

However, during the wet season, highest overlap index (a perfect overlap) was observed between the following species: Cape eland and Cape buffalo, 1, African elephants and Cape elands, 1, African elephants and Cape buffaloes, 1, Thomson’s gazelles and Grant’s gazelles, 1, Plains zebras and Grant’s gazelles, 1, Plains zebras and Thomson’s gazelles, 1, the lowest was observed between shoats and waterbucks, 0.07 and no overlap was observed between shoats and wildebeests, 0, shoats and warthogs, 0, shoats and Cape elands, 0, shoats and Cape buffaloes, 0 and between shoats and Maasai giraffes, 0.

Habitat overlap indices were constantly high during the dry and the wet seasons for the following species: Grant’s gazelles and Cape elands, Grant’s gazelles and warthogs, Grant’s gazelles and wildebeests, Thomson’s gazelles and wildebeests, impalas and warthogs, impalas and Maasai giraffes, Plains zebras and Cape elands, Plains zebras and Grant’s gazelles, Plains zebras and impalas, Plains zebras and warthogs and Plains zebras and wildebeests. However, it was constantly low during both the dry and wet seasons between the following pairs of species: Maasai giraffes and Cape elands, shoats and Thomson’s gazelles, shoats and warthogs, shoats and Plains zebras and between shoats and donkeys.

Overlap in habitat use was observed to be decreasing between several species during the wet season. However, it was increasing between Maasai giraffes and all the other species during this season.

Table 4.10: Pianka's indices of habitat overlap among the large herbivores during the dry and wet seasons in Kimana Wildlife Sanctuary

Animal	Season															
ED	Dry	_														
	Wet	1														
EL	Dry	_	0.97													
	Wet	1	1													
GF	Dry	_	0.39	0.62												
	Wet	0.4	0.4	0.4												
GG	Dry	_	0.97	0.93	0.74											
	Wet	0.97	0.94	0.97	0.49											
IM	Dry	_	0.52	0.33	0.71	0.56										
	Wet	0.8	0.79	0.8	0.6	0.86										
RB	Dry	_	0.12	0.11	0.66	0.1	0.4									
	Wet	0.75	0.75	0.75	0.23	0.79	0.92									
TG	Dry	_	0.43	0.29	0.77	0.55	0.2	0.27								
	Wet	0.98	0.98	0.98	0.51	1	0.91	0.85								
WB	Dry	_	0.19	0.11	0.46	0.12	0.29	0.66	0.18							
	Wet	0.77	0.77	0.81	0.38	0.86	0.75	0.63	0.83							
WH	Dry	_	0.64	0.55	0.61	0.73	0.61	0.06	0.43	0.13						
	Wet	0.92	0.92	0.93	0.58	0.91	0.85	0.69	0.94	0.77						
WLB	Dry	_	0.7	0.72	0.71	0.72	0.25	0.08	0.7	0.08	0.46					
	Wet	0.81	0.81	0.81	0.33	0.89	0.64	0.61	0.84	0.76	0.78					
ZB	Dry	_	0.96	0.96	0.77	0.97	0.65	0.28	0.52	0.15	0.66	0.86				
	Wet	0.97	0.97	0.97	0.43	1	0.85	0.8	1	0.85	0.93	0.91				
CT	Dry	_	0.79	0.78	0.89	0.76	0.59	0.69	0.45	0.48	0.49	0.56	0.81			
	Wet	0.08	0.08	0.09	0.5	0.19	0.66	0.57	0.28	0.13	0.3	0.08	0.21			
DK	Dry	_	0.9	0.77	0.75	0.89	0.83	0.24	0.62	0.18	0.7	0.56	0.82	0.78		
	Wet	_	_	_	_	_	_	_	_	_	_	_	_	_		
SH	Dry	0	0	0	0.55	0.06	0.41	0.74	0.18	0.84	0.38	0	0.1	0.45	0.11	
	Wet	0	0	0	0	0.1	0.5	0.66	0.18	0.07	0	0	0.12	0.76	0	
		BF	**ED**	**EL**	**GF**	**GG**	**IM**	**RB**	**TG**	**WB**	**WH**	**WLB**	**ZB**	**CT**	**DK**	**SH**

Key: Masaai giraffe (GF), impala (IM), warthog (WH), Thomson's gazelle (TG), Shoats (SH), Grant's gazelle (GG), Cattle (CT), Common waterbuck (WB), donkey (DR), Plains zebra (ZB), wildebeest (WLB), Cape eland (ED), Bohor reedbuck (RB), African elephant (EL) and Cape buffalo (BF).

Habitat overlap indices were generally high for several large herbivores occurring at approximately 67% and 51% of the species during the dry and wet seasons respectively (Figure 4.7). Intermediate values were less frequent occurring at approximately 5% and 15% of the species during the dry and the wet seasons respectively. Similarly, low values were less frequent, ranging at approximately 27 % and 33 % during the dry and wet seasons respectively.

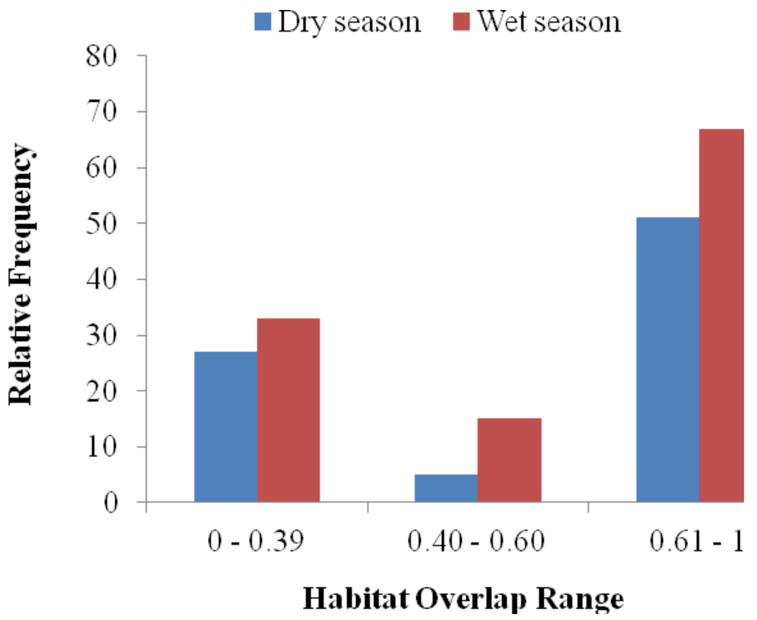


Figure 4.7: Frequency (%) distribution of the Pianka's indices of habitat overlap among large herbivores during the dry and wet seasons in Kimana Wildlife Sanctuary

The means for the higher values were 0.85 ± 0.01 and 0.77 ± 0.02 during the dry and the wet seasons respectively and differed significantly between the dry and wet seasons (t = -3.28; df = 97; p = 0.001) (Figure 4.8). The mean values at intermediate level were 0.45 ± 0.01 and 0.49 ± 0.25 during the dry and the wet seasons, but were non-significant between the dry and wet seasons (t = 0.26; df = 27; p = 0.80. The mean values at low level were 0.16 ± 0.19 and 0.13 ± 0.25 during the dry and wet seasons respectively, but were non-significant (t = 1.08; df = 51; p = 0.29).

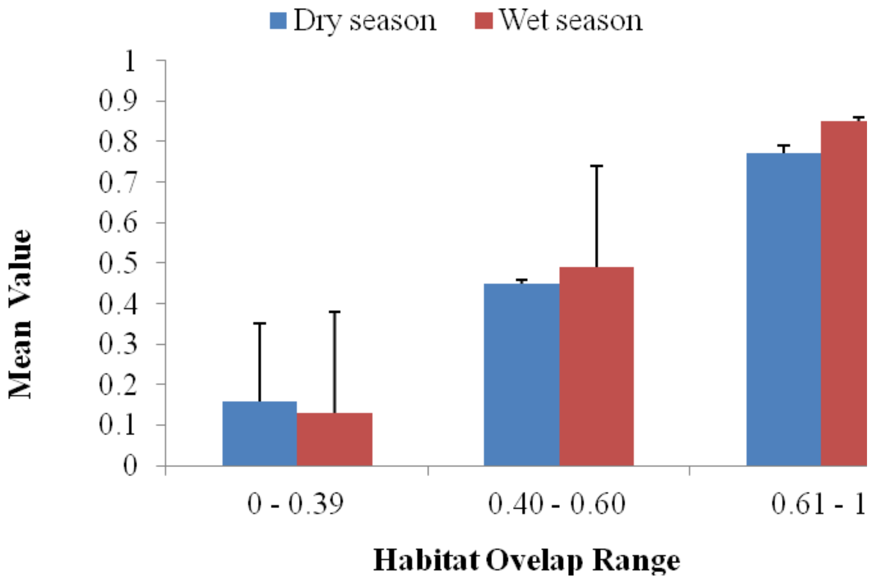


Figure 4.8: Means of the Pianka's indices of habitat overlap at low, intermediate and high range during the dry and wet seasons

CHAPTER THREE

CHAPTER FIVE

DISCUSSION

5.1 Characterization, classification and mapping of vegetation types

The results of remote sensing showed that the study area consisted of five distinct vegetation types which were woodlands, wooded grassland, sparse shrubland, grassland and herb grassland. However, classification based on physiognomic features of vegetation and dominant plant species resulted to seven distinct vegetation types which included *A. tortilis* woodland, *A. xanthophloea* woodland, wooded grassland, sparse shrubland, *S. fimbriatus* grassland, *C. ciliaris* grassland and herb grassland.

The result suggested that the study area was heterogeneous in its physiognomy, a structure that is characteristic to savanna vegetation and typical of vegetation in semi-arid regions of Kenya (Lekoiyet, 2006; Okello, 2005; Kioko *et al.,* 2012; Okul, 2014). The vegetation types reported in this study were within the range of Kioko *et al.* (2012) in KGR, who classified vegetation on the basis of physiognomic characteristics into four distinct types. Okul, (2014) classified vegetation of Maasai Mara conservancies into nine types .Generally, heterogeneity of vegetation in savannas often results from various factors including climate, soils and disturbances from fires, herbivores and human activities (Gandiwa *et al.,* 2011; Gandiwa *et al.,* 2013; Zisanza-Gandiwa *et al.,* 2013).

5.2 Composition and structure of vegetation types

The study recorded a total of 17 woody species belonging to seven different families in all the six vegetation types. Okul (2014) recorded a total of 86 woody species in conservancies of Maasai Mara. Welpola *et al.* (2004) recorded a total of 82 individual woody species in Maasai Mara National Reserve. Lekoyiet (2006) recorded a total of 41 individual woody species in the conserved and communal sites of Kimana and Eselenkei Group ranches in Amboseli ecosystem. The results, therefore, indicates that the sanctuary was composed of a lower number of woody species as compared with some of the protected areas, conservancies and communal lands in semi-arid regions of Kenya. The relatively low number of woody species could be attributed to effects of climate, soil factors and disturbances such as herbivory and human activities.

However, woody plant species richness was found to vary across the vegetation types, with wooded grassland recording the highest species richness and *A. xanthophloea* woodland the lowest. The observed variations could be due to difference across sites in conditions important for growth and survival of woody plants including soil conditions and effects of disturbances such as fire and herbivory. Lekoyiet (2006) found highest species richness in the *Commiphora* bushlands and *A. tortilis* woodlands and low species richness in *A. xanthophloea* woodlands of Eselenkei and Kimana Group Ranches.

Similarly, woody species diversity and evenness were low in all the vegetation types. However, it differed across the vegetation types, with wooded grassland and sparse shrubland recording the highest species diversity and evenness than *A. tortilis* woodland. *Acacia xanthophloea* woodland had zero species diversity and evenness, due to dominance of *A. xanthophloea.* The low woody plant species diversity and evenness in the sanctuary and its variation across the vegetation types could be attributed to effects of disturbance, soil types and human activities.

The woody species diversity and evenness reported in this study were within the range of Okul (2014) in Maasai Mara conservancies, who reported a diversity value of between 0.22 and 2.37 across the 9 vegetation formations. Lekoyiet (2006) recorded a diversity value of between 0.181 and 1.764 and evenness value of between 0.130 and 0.713 across woodlands of conserved and communal sites of Eselenkei and Kimana Group Ranches.

Similarity in the composition of woody plants also proved to be different across vegetation types, with woody plants in *A. tortilis* woodland showing high similarity in their composition with those in wooded grassland and sparse shrubland. This could be due to overlap in the composition of woody plants including *A. tortilis*, *B. glabra* and *S. persica* across the vegetation types. However, the composition of woody plants in *A. xanthophloea* woodland was dissimilar with those of other vegetation types.

The reported similarities and dissimilarities in the composition of woody plants between and dissimilarity reported between *A. xanthophloea* woodland and other vegetation types could be attributed to influences from soil conditions, herbivory and anthropogenic activities. Okul (2014) showed that the composition of woody plants across vegetation formations generally differed, though the composition of some woody plant species overlapped across the vegetation formations in the conservancies of Maasai mara.

Relative importance value indices differed across vegetation types, with woody plants in the wooded grassland recording the highest IVI as compared with woody plants in the other vegetation types. The IVI disparity between the sites could be attributed to differences across the vegetation types in species richness, densities, dominance and frequencies of woody species present. For instance, the wooded grassland which had the highest IVI had the highest richness and some of the woody plants present such as *A. tortilis* and *S. persica* had high densities, dominance and frequencies as compared with woody plants in the other vegetation types.

Similarly, the IVI for the individual woody plants varied across the vegetation types. *Acacia tortilis* dominated *A. tortilis* woodland and wooded grassland, *B. glabra* dominated sparse shrubland and *A. xanthophloea* dominated *A. xanthophloea* woodland. The observed result could be attributed to mainly to differences across vegetation types in soil conditions, as *A. xanthophloea* trees, which require high moisture for growth and survival (Otieno *et al.*, 2005) occurred along the riparian zone with the other plant species such as *A. tortilis*, *S. persica* and *B. aegyptica* occurred in the dry areas. Other factors including herbivory and human activities can cause variations in IVI for species across vegetation types. Similarly, Okul (2014) found different vegetation formations to be dominated by different woody species in conservancies of Maasai Mara. In Lake Nakuru National Park, Dharani *et al.,* (2006) found *A. xanthophloea* to dominate *A. xanthophloea* woodland.

The frequency of woody plant stems (dbh≥5cm) showed an inverted J-shaped structure in *A. tortilis* woodland, wooded grassland and sparse shrubland. This indicates that woody plants in lower demographic classes were more frequent in comparison with woody plants in higher demographic classes. Several studies have reported an inverted J-shaped structure in the population of woody plants and have suggested to represent a high regeneration and a lower recruitment potentials of woody plants into higher DBH classes due to disturbances such as fire, herbivory and human activities (Lekoyiet, 2006). However, the frequency of woody plant stems in *A. xanthophloea* woodland showed a J-shaped structure. This indicates that woody plants in higher demographic classes were more frequent compared to the woody plants in lower demographic classes. A similar structure for *A. xanthophloea* trees have been noted in Lake Nakuru National Park (Darahani, (2006) and Amboseli basin (Western & Maitumo, 2004), where climate, increased soil salinity and elephant browsing have been cited to be important determinants.

Generally, therefore, the results suggest that the structure and composition of woody vegetation may be similar or dissimilar across vegetation types. This is a common observation in many savannas of Africa, as similar researches have also found either similarities or dissimilarities in the composition and structure of woody vegetation (Okul, 2014). The similarities or dissimilarities in the composition and structure of vegetation across sites have been suggested to result from effects of climate, soil conditions and disturbances such as fires, herbivory and human activities (Okul, 2014).

Spatial variation in the structure and composition of woody vegetation has been pointed out by several findings to be important in influencing the abundance, distribution, habitat use patterns and spatial relationships among large herbivores (Cromsigt & Olff, 2006; Cromsigt *et al.,* 2009; Kleynhans *et al.,* 2010; Mwasi *et al.,* 2013). According to these studies, therefore, variability in the structure and composition of vegetation reported in this study can influence abundance, distribution, habitat use patterns and spatial relationships among large herbivores.

The frequencies of grass ecological categories; decreaser grass species, increaser I grass species and increaser II grass species differed significantly in all the vegetation types, with the exception of open grassland. Increaser II grass species such as *C. dactylon*, *S. fimbriatus* and *H. schimperii* dominated in all the vegetation types, with the exception of wooded grassland, which was dominated by *P. stramineum*, an increaser I grass species.

In contrast, a decreaser grass species often considered more palatable to grazers, *C. ciliaris* (Odadi *et al.,* 2007; Angassa, 2014), occurred at low frequency (< 50%) in wooded grassland and *C. dactylon* grassland, but was absent in all the other vegetation types. Its occurrence in wooded grassland and *C. dactylon* grassland could be explained by its ability to resist disturbances including high grazing pressure and drought due to strong fibrous root systems that are more than 2m deep, high germination capacity, high drought tolerance, quick response to rainfall patterns and its allelopathic traits (Mganga *et al.,* 2013; Angassa, 2014). However, with increasing densities of grazers and mixed feeders, this grass species is expected to decrease significantly or to disappear completely from wooded grassland and *C. dactylon* grassland as observed in the other vegetation types.

Other decreaser grasses that are considered more palatable and preferred as forage by grazers such as *P. maximum* and *T. triandra* (Odadi *et al.,* 2007; Richie, 2014), which have been reported to occur in this ecosystem, though in low frequencies, (0.15%) (Kioko *et al.,* 2012), were completely absent in all the vegetation types in the present study. Their absence is probably attributed to their inability to tolerate heavy grazing pressures (Kioko *et al.,* 2012).

The observed variations in the composition of the ecological categories of grasses in most of the vegetation types could be due to selective grazing of highly palatable and more preferred increaser grass species such as *C. ciliaris* by the large herbivores as compared with less palatable and less preferred increaser I and Increaser II grass species such as *C. dactylon, S. fimbriatus* and *H. schimperii* (Odadi, 2007; Trollope & Trollope, 2011). However, the dominance of the increaser grasses species in most of the vegetation types suggests that most of the vegetation types are under conditions of over-utilization (Trollope & Trollope, 2011). This could also suggest degradation of herbaceous vegetation due to effects of continuous grazing or high grazing pressures (Kioko *et al.,* 2012; Angassa, 2014). However, the dominance of *P. strameneum,* a more palatable and preferred increaser II grass species in wooded grassland indicates conditions of under-grazing or selective grazing, which often results from improper management of rangelands (Trollope & Trollope, 2011).

Herbaceous plant structure (grass standing biomass, grass basal cover, grass height and inter-tuft distance) was found to vary substantially across the vegetation types. Wooded grassland reported the highest grass standing crop biomass as compared with the other vegetation types. This could be due to accumulation of dead material resulting from excessive self-shading of less palatable grass species such as *P. strameneum,* a species that was found to dominate this vegetation type or moribund grasses with limited re-growth potential. The high accumulation of grass standing crop biomass reported in wooded grassland could represent important fuel load for fires that could develop at high intensity, which can be used to remove moribund, to control bush encroachment and to stimulate new grass tillers for grazers (Trollope & Trollope, 2011).

Riparian soils are often characterized by a combination of soil conditions that render them more suitable for growth and survival of vegetation (Richardson *et al.,* 2007). Accordingly, *A. xanthophloea* woodland, which occurred in the riparian zone of River Kimana, recorded the lowest inter-tuft distance but the highest grass basal cover, a relatively higher grass standing crop biomass and grass height, hence this vegetation type could be less susceptible to soil erosion. In contrast, sparse shrubland, which occurred in the drier site, recorded the highest inter-tuft distance but the lowest grass standing crop biomass, grass height and basal cover. This shows that shrubland was poorly covered by herbaceous vegetation, hence, could be more susceptible to soil erosion.

Several studies in rangelands have noted significant low grass biomass production, grass height, basal and canopy cover percentage and an increase in inter-tuft distance in areas of continuous or heavy grazing (Kioko *et al.,* 2012; Zarekia *et al.,* 2013). Therefore, the observed structure of herbaceous species in shrubland could be resulting from grazing and trampling effects of herbivores, particularly livestock which showed a higher selectivity for this vegetation type during both the dry and the wet seasons. Other than effects of herbivory, other factors such as soil conditions, fires and human activities may have influenced the composition and structure of herbaceous vegetation (Zisanza-Gandiwa, 2013).

5.3 Population size of large herbivores between seasons

The study showed that the combined population densities of all large herbivores, densities of large wild herbivores and of large domestic herbivores did not differ significantly during the dry and the wet seasons. Similarly, the population densities for individual animals did not differ, except for Cape elands, whose densities increased significantly during the wet season. The lack of significant changes in the densities of large herbivores could suggest wet and dry season concentration of large herbivores in the sanctuary probably due to availability of diverse and suitable habitats. However, the significant increase in the densities of Cape elands during the wet season could indicate seasonal migration of Cape elands, probably due to seasonal variations in forage resources, a pattern that has been observed for cape elands in Lake Nakuru National Park (Mwangi & Western, (I998).

Seasonal migration by wild herbivores is considered a survival strategy important for maximizing species resource requirements while minimizing predation risks (Hopcraft *et al.,* 2010), for increasing resource availability (Georgiadis *et al.,* 2003) and for minimizing competition (Ottichillo *et al.*, 2000). Accordingly, the seasonal migration by the cape elands could help the species survive by maximizing resource requirement, minimizing predation risks, increasing resource availability and minimizing competition.

5.4 Seasonal selection of habitats by large herbivores

Selection of habitats by large herbivores varied across vegetation types during the dry and the wet seasons. This could be due to differences across vegetation types in the composition and structure of their vegetation. For instance, wooded grassland, which was characterized by a high species richness, diversity, evenness, IVI, grass standing crop biomass and high frequency of Increaser I grass species was highly selected by the large herbivores compared with the other vegetation types during the dry and wet seasons. This suggests that wooded grassland provided forage, thermal cover and nesting cover for both browsers and grazers in the sanctuary. This could indicate the importance of wooded grassland as a major habitat type for large herbivores in the sanctuary. Groom & Harris (2010) showed that large herbivores, especially grazers, were more distributed in areas were grass biomass was higher in Amboseli ecosystem.

In contrast, sparse shrubland, which was characterized by low species richness, IVI, standing crop biomass, grass height and grass cover but higher frequency of increaser II grass species and inter-tuft distance compared with the other vegetation types, was generally avoided by several large herbivores during both the dry and the wet seasons. However, it was highly selected by livestock including cattle and shoats during both the dry and wet seasons. Greater selection of shrubland by livestock could be due to interspecific competition for forage resources, which probably resulted from direct displacement of large wild herbivores by livestock. This could suggest that the sanctuary is not compatible for livestock grazing and wildlife conservation.

Livestock have been reported to displace wildlife in ecosystems. Studies in Amboseli ecosystem have shown that livestock, especially shoats displace wildlife (Okello, 2015). Studies in other ecosystems have shown wildlife to avoid habitats used by livestock. For instance, Namgail *et al.,* (2007), found Tibetan argali (*Ovis ammon hodgsoni*) to avoid habitats used by livestock; goats and sheep in Gya-Miru Wildlife Sanctuary, Ladakh, India.. Nevertheless, selection of habitat used by livestock by wild large herbivores may also be influenced by disturbances caused by herders and herding dogs (Namgail *et al.,* 2007) and this could have also influenced the use of shrubland by large wild herbivores in the present study. Large wild herbivores, however, have been observed to spatially overlap with livestock in the same ecosystem (Sitters *et al.,* 2009).

The results from this study could suggest that habitat use by large herbivores during the dry and wet seasons may be due to variation in the structure and composition of vegetation and disturbance from herded livestock. However, other factors such as forage availability, abundance and quality, predation risks, water availability, topography and habitat heterogeneity and could influence habitat use by large herbivores (Sitters *et al.,* 2009; Groom & Harris, 2010; Owen-Smith *et al.,* 2015).

5.5 Seasonal habitat niche breadth indices among large herbivores

The study found several large herbivores to consistently maintain low habitat niche breadth indices during the dry and wet seasons, which suggested specialization on habitat use. The low habitat niche breadth indices could be due to relatively low selectivity by the large herbivores during both the dry and the wet seasons. Mwasi *et al.,* (2013), reported low habitat niche breadth values in a grazer assemblage during the dry and the wet seasons in Lake Nakuru National Park, Kenya.

5.6 Seasonal habitat overlap indices among large herbivores

Habitat overlap indices were generally high among the large herbivores during the dry and the wet seasons. This imply that there was minimal ecological separation in habitat use among large herbivore during the dry and the wet seasons, which could be due to greater selection of habitats offering abundant forage of high quality by the large herbivores and minimal use of habitats with limited resources.

The reported high overlap indices could suggest high potential for resource competition among the large herbivores during both the dry and the wet seasons (Bilal & Turner, 2012), but it is likely that the results also indicate shared resource utilization and lack of competition among the large herbivores (Mwasi *et al.,* 2013). Therefore, conclusions about competition as a structuring mechanism among the large herbivores is controversial, due to the fact that competition can occur between species only under conditions of habitats overlap, diet overlap and resource limitation (Mudhusudan, 2004; Trail, 2004; Mwasi *et al.,* 2013).

Despite the high habitat overlap indices reported among large herbivores during both the dry and the wet season, overlap indices between giraffes and the other large herbivores was shown to be low during the wet season. This observation could indicate coexistence between giraffes and other large herbivores during the wet season and segregation during the dry season, which could be resulting from differences between the giraffes and the other large herbivores in their feeding strategies. Giraffe was the only obligate browser encountered in the sanctuary. This finding is similar with that reported between giraffes and other large herbivores in Lake Nakuru National Park (Mwangi & Western, 1998).

Higher overlap in habitat use among large herbivores has been reported by other studies in savannas of Africa. Kleynhans *et al.*, (2010) found a high overlap in habitat use for six wild grazer species during the dry and the wet seasons in Hluhuwe-iMfolozi Park, South Africa. Mwasi *et al.*, (2013) also found a higher overlap in habitat use for similar sized grazers during the dry and the wet seasons in Lake Nakuru National Park.

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

The vegetation in KWS was characterized by seven distinct vegetation types that included *A. tortilis* woodland, *A. xanthophloea* woodland, wooded grassland, sparse shrubland, *C. dactylon* grassland and *S. fambriatus* grassland and herb grassland.

The structure and composition of woody vegetation in KWS varied across vegetation types, with woody plants in wooded grassland recording a higher species richness, diversity and evenness, high relative canopy cover and high woody plant IVI.

The composition of grass categories varied within vegetation types, except in open grassland, which had similar composition due to dominance of an increaser II grass species, *S. fimbriatus* and absence of the other ecological categories of grasses. Similarly the structure of herbaceous plants (grass standing crop biomass, grass basal cover, grass height and inter-tuft distance) varied across all the vegetation types.

Sixteen species of large herbivores occurred in the sanctuary and their population densities did not vary between the dry and the wet season, except for Cape elands whose density increased during the wet season.

Large herbivores were more frequent in wooded grassland during the dry and wet seasons. Several large herbivores consistently maintain low habitat niche breadth indices during the dry and wet seasons. Habitat overlap indices among the large herbivores were generally high during the dry and the wet seasons.

6.2 Recommendations

6.2.1 Recommendations for wildlife management

The diversity of vegetation types in the sanctuary should be maintained so that it can continually provide diversity of habitats that provide a greater number of important niches for wildlife use

Most of the vegetation types were dominated by increaser grass species, which indicated conditions of over-utilization; hence, the frequencies of decreaser grasses of high palatability and more preferences as forage by large herbivores should be increased by reseeding using native grass species, which are more palatable and resistant to high grazing intensity. *Cenchrus ciliaris* is an example of such grass species and since it occurred in wooded grassland and *C. dactylon* grassland, though in low frequencies, demonstrated its potential for restoration of all the vegetation types.

As large herbivores, particularly, the Cape elands concentrate in the sanctuary during the wet season and disperse during the dry season, their conservation crucially depends on habitat conditions in both the sanctuary and surrounding areas, hence, wildlife conservation efforts should be aimed at restoring the conditions of various habitat types within the sanctuary and to properly plan land uses in the surrounding areas to prevent further fragmentation and loss of wildlife habitats.

Specialization by large herbivores on habitat types during the dry and the wet seasons can cause narrow tolerance and resilience to spatial and temporal changes and this can cause rarity and/or low density of species, which might cause species decline and extinction. Therefore, the population densities of large herbivores in the study area should be monitored and habitat types in poor conditions be restored to promote greater use of all the habitats.

6.2.2 Recommendations for further research

Several suggestions were made to explain various findings in this research study, including suggestions on the underlying factors causing similarities or dissimilarities in the composition and structure of woody and herbaceous vegetation, high selectivity by large herbivores during the dry and wet seasons, low niche breadth during the dry and wet seasons and high overlap in habitat use during the dry and wet seasons. Therefore, further research is recommended to find out the factors causing variations in the composition and structure of woody and herbaceous vegetation across and within vegetation types.

Similarly, further studies should focus on factors causing high habitat selectivity, low niche breadth and high overlap in habitat use among the large herbivores during the dry and wet seasons in KWS. Data on diet use is required to provide more accurate conclusions regarding competition as a structuring mechanism among the studied large herbivores during the dry and wet seasons.

REFERENCES

Agnew, A., & S., A. (1994). *Upland Kenya wild flowers* (2nd ed.). Nairobi: East Africa Natural History Society.

Ahrestani, F. S., Heikonig, I. M., & Prins, H. H. (2012). Diet & habitat niche relationships within an assembalage of large herbivores. *Tropical Ecology, 28*, 385-394.

Altmann, J., Altmann, S., & Roy, S. (2002). Dramatic change in local climate patterns in the Amboseli basin, Kenya. *African Journal of Ecology, 40*, 248-251.

Angassa, A., Megersa, B., Markemann, A., Ogutu, J., Piepho, H., & Zarate, A. (2014). Livestock diversification: an adaptive strategy to climate and rangeland ecosystem changes in Southern Ethiopia. *Human Ecology, 42*(4), 509-520.

Banda, T., Schwartz, M., & Caro, T. (2006). Woody vegetation structure and composition along a protection gradient in a Miombo ecosystem of western Tanzania. *Forest Ecology and Management, 230,* 179-185.

Barbour, M., Burk, J., & Pitts, W. (1987). *Terrestrial plant ecology.* California: The Benjaminn/Cumming Publi. Co. Inc. .

Beentje, H. (1994). *Kenya trees, shrubs and lianas.* Nairobi: National Meseums of Kenya.

Berliner, D., & Kioko, J. (1999). The effect of long-term mowing and ungulate exclusion on grass species composition and soil nutrient status on the Athi-Kapiti plains, central Kenya. *16*(2-3), 62-70.

Bhola , N.; Ogutu, J.O.; Piepho , H.; Said, M.Y.; Reid , S.R.; Hobbs, N.T.; Olff, H.;. (2012). Comparative changes in density and demography of large herbivores in the mara reserve and its surrounding human-dominated pastoral ranches in Kenya. *Biodiversity and Conservation, 21*(6), 1509-1530.

Brower, J., & Zar, J. (1990). *Field and laboratory methods for general ecology.* (3rd ed.). Dubuque,: Brown publishers.

Chamaille-Jammes, S., Valeix, M., Bourgarel, M., Murindagomo, F., & Frutz, H. (2009). Seasonal density estimates of common large herbivores in Hwange National Park, Zimbabwe. *African Journal of Ecology, 47*(4), 804-808.

Clayton, W. (1982). *Graminae (part III). Flora of tropical East African* . Rotterdam: Crown Agents for Overseas Governments and Administration.

Cromisgt, J. G., Prins, H. H., & Olff, H. (2009). Habitat heterogeneity as a driver of ungulate diversity and distribution pattern: interaction of body mass and digestive stragey. *Diversity and Distribution, 15*, 513-522.

Cromsigt, J., & Olff, H. (2006). Resource partitioning among savanna grazers mediated by local heterogeneity: an experimental approach. *Ecology, 87*, 1532-1541.

Dekker, B., Van Rooyen, N., Bothma, J., & Du, P. (1996). Habitat partitioning by ungulates on a game ranch in the Mopani veld. *South Africa Journal of Wildlife Research, 26*(4), 117-122.

Dharani, N., Kinyamario, J., & Onyari, J. (2006). Structure and composition of Acacia xanthophloea woodland in Lake Nakuru National Park, Kenya. *African Journal of Ecology, 44*(4), 523-530.

ESRI. (2014). ArcGIS software user library: Spatial analysis tool (Image Classification). *Forest Policy and economics, 9*(5), 558-577.

Estes, R. D. (1991). *The Behavior Guide to African Mammals.* London, USA: University of Califonia Press, Ltd.

Facility, U. S. (2016, 04 51TH). *W.W.W.* Retrieved from http://glcf.umd.edu/data/landsat

Gachimbi, L. (2002). *Technical report of soil survey and sampling: Amboseli-Loitokitok Transect.* Lucid Technical Report.

Gandiwa, E., Chikorowondo, G., Zisadza-Gandiwa, P., & Muvengwi, J. (2011). Structure and composition of Androstachys johnsonii woodland across various stata in Gonarezhou National Park, southeastern Zimbabwe. *Tropical conservation science, 4*(2), 218-229.

Gandiwa, E., Heutkonig, I., Gandiwa, P., Matsvayi, W., Van der westhuizen, H., & Gwenya, M. (2013). Large herbivore dynamics in northern Gonarezhou National Park, Zimbabwe. *Tropical Ecology, 54*(3), 345-354.

Gandiwa, E., Zisadza-Gandiwa, P., Goza, D., Mashapa, C., & Muboko, N. (2014). Diversity and structure of woody vegetation across areas with different soils in Gonarezhou National Park, Zimbabwe. *Southern Forests, 76*(2), 111-116.

Gandiwa, P., Chinoitezvi, E., & Gandiwa, E. (2013). The structure and composition of woody vegetation in two important bird areas in Southern Zimbabwe. *The Journal of Animal and Plant Sciences, 23*(3), 813-820.

Ganguli, A. C., Vermeire, L. T., Mitchell, R. B., & Wallace, M. C. (2000). Comparison of four non-destructive techniques for estimating standing crop in short-grass plains. *Agronomy Journal*, 1211-1215.

Georgiadis, N., Hack, M., & Turpin, K. (2003). The influence of rainfall on zebra population dynamics: implications for management. *Journal of Applied Ecology, 40*(1), 125-136.

Goqwana, W. M., & Trollope, W. W. (2005). Simplified techniques for assessing the condition of the grass sward in the Cymbopogon Themeda veld in the southern free state and northern Eastern cape provinces of South Africa. *Grassroots: Newsletter of the Grassland Society of Southern Africa., Vol. 5* (1).

Groom, R., & Harris, S. (2010). Factors affecting the distribution patterns of zebra and wildebeest in a resource-stressed environment. *African Journal of Ecology, 48*(1), 159-168.

Grossman, G. (1986). Food resources partitioning in a rocky intertidal fish assemblage. *Journal of Zoology, 1,* 317-355.

Hopcraft, J., Olff, H., & Sinclair, A. (2010). Herbivores, resources and risks: Alternating regulation along primary environmental gradients in savannas. *Trends Ecological Evolution, 25,* 119-128.

Imbahale, S., Githaiga, J., Chira, R., & Said, M. (2008). Resource utilization by large migratory herbivores of the Athi-Kipiti ecosystem. *African Journal of Ecology, 46*(1), 43-51.

Janson, D., & Vegelius, J. (1981). Measures of ecological association. *Oecologia, 49,* 371-376.

Kahurananga, J. (1981). Population estimates, densities and biomass of large herbivores in Simanjiro Plains, northern Tanzania. *African Journal of Ecology, 19*(3), 225-238.

Kamau, P. (2004). *Forage diversity and impact of grazing management on rangeland ecosystems in Mbeere District, Kenya.* LUCID working paper no. 36, International Livestock Research Institute, Nairobi.

Kenya, G. o. (2012). *The national wildlife conservation and management policy, 2012.* Nairobi: Government Press.

Keya, G. (1998). Herbaceous layer production and utilization by herbivores under different ecological conditions in an arid savanna of Kenya. *Agri. Ecosys. Environ, 69*, 55-67.

Kiffner, C., Hopper, R., & Kioko, J. (2016). Trends in seasonal population densities of wildlife species in Tarangire National Park, Tanzania. *African Journal of Ecology*.

Kilavi, M. (2015, August 5th). *Analysing the temporal characteristices of extreme temperature events over the ASALs and the costal regions of Kenya as an Indicator of Climate change*. Retrieved from URL: http://erepository:uonbi.ac.ke:8080/xm/ui/handle/123456789/20737

Kingdon, J. (1997). *The Kingdon Field Guide to African Mammals.* London, USA.: Academic press.

Kioko, J., & Okello, M. (2010). Land use cover and environmental changes in a semi-arid rangelands, Southern Kenya. *Journal of Geography and Regional Plannning, 3*(11), 322-326.

Kioko, J., Kiringe, J. W., & Ole seno, S. (2012). Impacts of livestock grazing on a savanna grassland in Kenya. *Journal of Arid Land, 4*(1), 29-35.

Kiringe, J., & Okello, M. (2005). Use and availability of tree and shrub resources on Maasai communal rangelands near Amboseli, Kenya. *Range Forage Science., 22*(1), 37-46.

Kleynhans, E., Jolles, A., Boss, R., & Olff, H. (2010). Resource partitioning among multiple niche dimensions in different sized African savanna grazers. *Oikos, 120,* 591-600.

Ko, D., Bristow, N., Greenwood, D., & Weisberg, P. (2009). Canopy cover estimation in semi-arid woodlands: comparisons of field-based and remote sensing methods. *Forest science, 55*(2), 132-141.

Krebs, C. J. (1998). *Ecological Methodology.* (2nd edn. ed.). New York: Addison Wesley Longman.

Kurgat, B., Golicha, D., Giese, M., Kuria, S., & Asch, F. (2014). *Relationship between vegetation cover types and soil organic carbon in the rangelands of Northern Kenya.* Hohenheim University. Retrieved from Open URL.

Lekoyiet, S. T. (2006). *Woody species comparisons between conserved and communal sites and woody plant use by local communities: A case study of Eselenkei and Kimana Group Ranches in Southern Kajiado, Kenya.* Addis Ababa University. Unpublished thesis.

Levin, R. (1968). *Evolution in changing environments: Some theoritical explorations.* Princeton, NJ: Princeton University Press.

Macandza, V., Owen-smith, N., & Cain III, J. (2012). Dynamic spatial partitioning and coexistence among tall grass grazers in an African savanna. *Oikos, 121*, 891-898.

Macharia, P., & Ekaya, W. (2005). The impact of rangeland condition and trend on the grazing resources of a semi-arid environment in Kenya. *Journal of Human Ecology*, *17*(2), 143-147.

Manly, B., & Mcdonald, L. (1993). *Resource Selection by Animals. Statistical Design and Analysis for Field Studies.* Chapman and Hall.

Martin, J., Benhamous, S., Yoganand, K., & Owen-Smith, N. (2015). Coping with spatial heterogeneity and temporal variability in resources and risks: Adaptive movement behaviour by a large grazing herbivore. *PloS one, 10*(2). doi:e0118461

Mganga, K., Musimba, N., Nyangito, W., Nyariki, D., & Mwang'ombe, A. (2010). Improving hydrological properties of degraded soils in Semi-Arid Kenya. *Journal of Environmental Science and Technology, 3*, 217-225.

Mganga, K., Musimba, N., Nyariki, D., Nyangito, M., & Wangombe, A. (2013). The choice of grass species to combact desertification in Semi-Arid Kenyan rangelands is greatly influenced by their forage value for livestock. *Journal of the British Grassland Society*.

Michelakis, D., Staurt, N., Lopez, G., Linares, V., & Woodhouse, I. (2014). Local - Scale Mapping of Biomass in Tropical Lowland Pine Savannas using ALOS PALSAR. *Forests*, *5*, 2377-2399.

Mose, V., Nguyen-Huu, T., Auger, P., & Western, D. (2012). Modelling herbivore population dynamics in the Amboseli National Park, Kenya: Application of spatial aggregatio of variables to derive a master model. *Ecological Complexity, 10*, 42-51.

Mudhusudan, M. D. (2004). Recovery of wild large herbivores following livestock decline in a tropical Indian wildlife reserve. *Jounal of African Ecology, 41*, 858-869.

Mueller-Dombois, D., & Ellenberg, H. (1974). *Aims and methods of vegetation ecology*. New York: John Wiley and Sons.

Mureithi, R., Koech, O., Kinuthia, S., & Wanjogu, R. (2014). Effects of different ooil water content and seed storage on Quality of Six Range Grasses in the Semi-Arid ecosystem of Kenya. *Environment and Ecology Research, 2*(7), 261-271.

Musyoki, J., Luvanda, A., & Kitheka, E. (2012). Sources of natural dyes and tannins among the somali community living in Garissa county, Kenya. *International Joural of Biodiversity and Conservation, 4*(12), 400-410.

Mutangah, J. (1989). The vegetation of Lake Nakuru National Park, Kenya: A list of the vegetation types with annoted species list. *Journal of East African Natural History, 78*(1), 71-80.

Muthoni, F., Groen, T., Skidmore, A., & van Oel, P. (2014). Ungulate herbivory overrides rainfall impacts on herbaceous regrowth and residual biomass in a key resource area. *Journal of Arid Environments, 100*, 9-17.

Mwangi, E. M., & Western, D. (1998). Habitat selection by large herbivores in Lake Nakuru National Park, Kenya. *Chapman and Hall, 7*, 1-8.

Mwasi, S., Van Wieren, S., Heitkonig, I., & Prins, H. (2013). Seasonal resource use and niche breadth in an assemblage of coexisting grazers in a fenced park. *Open Journal of Ecology, 3*(6), 383-388.

Namgail, T., Fox, J. L., & Bhatnaga, Y. V. (2007). Habitat shift and time budget of the Tibetan Agali; The influence of livestock grazing. *Ecological Restoration*(22), 25-31.

Ng'weno, C., Mwasi, S., & Kairu, J. (2010). Distribution, density and impact of invasive plants in Lake Nakuru National Park, Kenya. *African Journal of Ecology, 48,* 905-913.

Noad, T., & Birnie, A. (1994). *Trees of Kenya: A full illustration field guide.* Nairobi.

Northon-Griffiths, M. (1978). *Counting animals.* Nairobi: African Wildlife Leadership foundation .

Nyamasyo, K., & Kihima, O. (2014). Changing Land Use Patterns and their Impacts on wild ungulates in Kimana Wetland Ecosystem, Kenya. *International Journal of Biodiversity*.

Odadi, O., Young, T., & Okeyo Owuor, J. (2007). Effects of wildlife on cattle diets in Laikipia, Rangeland, Kenya. *Rangeland Ecology & Management, 60*, 179-185.

Odadi, W. O., Karachi, M. K., Abdultazak, S. A., & Young, T. P. (2011). African wild ungulate compete or facilitate cattle depending on season. *Science, 333,* 1795.

Ogutu, J. O., Owen - Smith, N., Piepho, H. P., & Said, M. Y. (2011). Continuing wildlife population declines and range constraction in the Mara region of Kenya during 1977-2009. *Journal of Zoology, 284*, 99-109.

Ogutu, J., Piepho, H., Said, M., & Kifugo, S. (2014). Herbivore dynamics and range contraction in Kajiado County Kenya: climate and land use changes, population pressures, governance, policy and human-wildlife conflicts. *The Open Ecology Journal, 7*(1).

Ogutu, J.O.; Owen-Smith , N.; Piepho, H.; Said , M.Y.; Kifugo, S.C.; Reid, R.S.; Gichohi, H.; Kahumbu , P.; Andanje , S.;. (2013). Changing wildlife population in Nairobi National Park and adjoining Athi-Kiputiei plains. Collapse of the migratory wildebeest. *The open conservation biology journal, 7*(11), 11-26.

O'kane, C., Duffy, K., Page, B., & Macdonald, D. (2013). Effects of resource limitation on habitat usage by the browser guild in Hluhluwe-Imfolonzi Park, South Africa. *Journal of Tropical Ecology, 29*(01), 39-47.

Okello, M. (2012). The contraction of wildlife dispersal areas by human structures and activities in Mbirikani Group Ranch in the Amboseli Ecosystem, Kenya. *International Journal of Biodiversity and Conservation, 4*(6), 243-259.

Okello, M. M. (2005). An assessment of the large mammalian component of the proposed wildlife sanctuary site in Maasai Kuku Group Ranch near Amboseli, Kenya. *South African Journal of Wildlife Research, 35*(1), 63-76.

Okello, M., & D'Amour, D. E. (2008). Agricultural expansion within Kimana eclectric fences and implications for natural resource conservation around Amboseli National Park, Kenya. *Arid Environments, 72*, 2179-2192.

Okello, M., & Kioko, J. (2010). Contraction of wildlife dispersal area in Olgulului-Ololorashi Group Ranch around Amboseli National Park, Kenya. *The Open Conservation Biology Journal, 4*, 34-45.

Okello, M., Njumbi, S., Kiringe, W., & Isiiche, J. (2015). Habitat use and preference by the African elephant outside of the protected area, and management implications in Amboseli landscape, Kenya. *International Journal of Biodiversity and Conservation, 7*(3), 211-236.

Okul, D. (2014). *An assessment of woody vegetation structure in Maasai Mara conservancies of Narok county, Kenya.* Doctoral dissertation, University of Nairobi.

Otieno, D., Schmidt, M., Kinyamario, J., & Tenhunen, J. (2005). Responses of Acacia tortilis and Acacia xanthophloea to seasonal changes in soil water availability in the sevanna region of Kenya. *Journal of Arid Enviroment*, *62*, 377-400.

Ottichilo, W. K., de Leeuw, J., Skidmore, A. K., Prins, H., & Said, M. Y. (2000). Population trends of large non-migratory wild herbivores and livestock in the Masai Mara ecosystem, Kenya, 1977 and 1997. *African Journal of Ecology, 38*, 2002-216.

Owen-Smith, N., Jodie, M., & Yoganand, K. (2015). Spatially nested niche partitioning between syntopic grazers at foraging arena scale within overlapping home ranges. *Ecosphere, 6*(9), 1-17.

Pianka, E. R. (1973). The structure of lizard communities. *Annual Review of Ecology and Systematics, 4*, 53-74.

Porensky, L. M., & Veblen, K. E. (2012). Grasses and browsers reinforce landscape heterogeneity by excluding trees from ecosystem hotspots. *Oecologia, 168*, 749-759.

Pratt, D. J., & Gwynne, M. D. (1977). *Rangeland management and ecology in East Africa.* London, USA.: Hodder & Stoughton.

Richards, J. (1999). *Remote sensing digital image analysis*.

Richardson, D., Holmes, P., Esler, K., Galatowitsch, S., Stomberg, J., Kirkman, S., . . . Hobbs, R. (2007). Riperian vegetation degradation, alien plant invasions, and restoration prospects. *Diversity and Distribution, 13*, 126-139.

Riginos, C., & Grace, J. B. (2008). Savanna tree density, herbivores and the herbaceous community: bottom-up vs. top-down effects. *Ecology, 89*,, 2228-2238.

Rutherford, M., & Powrie, L. (2009). Severely degraded dune of the southern Kalahari; Local extincton, persistence and natural re-establishment of plants. *African Journal of Ecology, 48*(4), 930-938.

Rutherford, M., & Powrie, P. (2013). Impacts of heavy grazing on plant species richness. A comparison across rangeland biomes of South Africa. *South African Journal of Botany, 87,* 146-156.

Rutherford, M., Powrie, L., & Thompson, D. (2012). Impacts of high utilization pressure on biodiversity components in Colophospermum mopane savanna. *African Journal of Range & Forage Science, 29*(1), 1-11.

Sankaran, M., Ratnam, J., & Hanan, N. (2008). Woody cover in African Savannas: the role of resources, fire and herbivory. *Global Ecological Biogeography, 17*, 236-245.

Shisanya, C., Recha, C., & Anyamba, A. (2011). Rainfall variability and its impacts on normalized difference vegetation index in Arid and Semi-Arid lands of Kenya. *International Journal of Geoscience, 2*, 36-47.

Sitters, J., Heitkoning, I., Holmgren, M., & Ojwang, G. (2009). Herded cattle and wild grazers partition water but share resources during dry year in East African savanna. *Biological Conservation, 142*, 738-750.

Syombua, M. (2013). Land use and land cover changes and their implications for human-wildlife conflicts in the semi-arid rangelands of southern Kenya. *Journal of Geography and Regional Planning, 6*(5), 193-199.

Tafera, S., Snyman, H., & Smit, G. (2007). Rangeland dynamics in southern Ethiopia: (1) Botanical composition of grasses and soil characteristics in relation to land-use and distance from water in semi-arid Borana rangelands. *Journal of Environmental Management, 85*, 429-442.

Tessema, Z., de Boer, W., Baars, R., & Prins, H. (2011). Changes in soil nutrients, vegetation structure & herbaceous biomass in response to grazing in a semi-arid savanna of Ethiopia. *Journal of Arid Environments*, *75*, 662-670.

Traill, L. (2004). Seasonal utilization of habitat by large grazing herbivores in semi-arid Zimbabwe. *South African Journal of Wildlife Research, 34*(1), 13-24.

Trollope, W. (2004). Prescribed Burning in African Grassland and Savannas for Domestic Livestock System. *Grassroots: Newslettr of Grassland Society of South Africa., 4*.

Trollope, W., Trollope, L., Naylor, S., Goode, R., de Bruyn, P., Cheswick, S., & Schutte, C. (2011). The Mun-Ya-Wana Private Game Reserve, South Africa - a leader in using a decision support system for prescribed Burning. *The 5th International Wildland Fire Conference.* Sun city, South Africa.

Trump, E. (1972). *Vegetation and land use survey of Narok district.* Nairobi: UNDP/FAO/GOK.

van Essen, L., Bothma, J., Du, P., Rooyen, v., & Trollope, W. (2002). Assessment of the woody vegetation of Ol Choro Oiroua, Masai Mara, Kenya. *African Journal of Ecology, 40*, 76-83.

van Langevelde, F.; van de Vijver, C.; Kamur, L.; van de Kloppel, J.; de Ridder, N.; van Andel, J.; Skidmore, A.K.; Heame, J.W.; Stroosnijder, L.; Bond, W.J.; Prins, H.H.T.; Rietkerk, M.;. (2003). Effects of fire and herbivory on the stability of savanna ecosystems. Ecology. *Ecology*, 337-350.

Voeten, M. M., & Prins, H. H. (1999). Resource partitioning between sympatric wild & domestic herbivores in the Tarangire region of Tanzania. *Oecologia, 120*, 287-294.

Walpole, M., Nabaala, M., & Matankory, C. (2004). Status of the Mara woodlands in Kenya. *African Journal of Ecology, 42*(3), 180-188.

Western, D., & Maitumo, D. (2004). Woodland loss and restoration in a savanna park: a 20-year experement. *African Journal of Ecology, 42*, 111-121.

Young-Zhong, S., Yu-Lin, L., Jian-Yuan, C., & Wen-Zhi, Z. (2005). Influence of continuous grazing and livestock exclusion on soil properties in a degraded sandy grassland, Inner Mongolia, Northern China. *Catena, 59*, 267-278.

Zarekia, S., Arzani, H., Jafari, M., Javadi, S., & Esfahan, E. (2013). Changes of vegetation structure and biomass in response to the livestock grazing in steppe rangelands of Iran. *Journal of Animal and Plant Science, 23*(5), 1466-1472.

Zisanza-Gandiwa, P., Mango, L., Gandiwa, E., Goza, D., Parakasingwa, C., Chinoitezvi, E., . . . Murengwi, J. (2013). Variation in woody vegetation structure and composition in a semi-arid savanna of Southern Zimbabwe. *International Journal of Biodiversity and Conservation*, *5*(2), 71-77.

APPENDICES

Appendix I. Composition of woody plants in different vegetation types in Kimana Wildlife Sanctuary

Vegetation type	Family	Species	Life form
***Acacia tortilis* Woodland**	Balanitaceae	*Balanitis glabra*	Shrub
	Boraginaceae	*Cordia monoica*	Shrub
	Fabaceae	*Acacia tortilis*	Tree
	Salvadoraceae	*Salvadora persica*	Shrub
	Solanaceae	*Lycium europeanum*	Shrub
Wooded grassland	Balanitaceae	*Balanities glabra*	Shrub
	Boraginaceae	*Cordia monoica*	Shrub
	Burseraceae	*Comiphora africana*	Shrub
	Capparidaceae	*Maerua edulis*	Shrub
	Fabaceae	*Acacia tortilis*	Tree
	Salvadoraceae	*Azima tetracantha*	Shrub
		Salvadora persica	Shrub
Shrubland	Balanitaceae	*Balanities glabra*	Shrub
	Boraginaceae	*Cordia monoica*	Shrub
	Fabaceae	*Acacia drepanolobiun*	Shrub
		Acacia tortilis	Tree
***Acacia xanthophloea* Woodland**		*Acacia xanthophloea*	Tree

Appendix II: Vegetation types in Kimana Wildlife Sanctuary

***Acacia tortilis* woodland (Source, Author, 2016)**

Acacia xanthophloea **woodland (Source, Author, 2016)**

Sparse shrubland (Source, Author, 2016)

Wooded grassland (Source, Author,2016)

***Sporolobus fimbriatus* grassland (Source, Author, 2016)**

***Cynodon dactylon* grassland (Source, Author, 2016)**