EFFECTS OF VEGETATION COVER AND TOPOGRAPHIC POSITIONING ON SOIL ORGANIC CARBON, SOIL AGGREGATES AND WATER INFILTRATION RATES IN LAIKIPIA GRASSLANDS, KENYA

MUTUKU DAISY MUTHIKE

A THESIS SUBMITTED TO THE SCHOOL OF AGRICULTURE AND BIOTECHNOLOGY IN PARTIAL FULFILMENT OF THE REQUIREMENT FOR THE CONFERMENT OF THE DEGREE OF MASTER OF SCIENCE IN SOIL SCIENCE, KARATINA UNIVERSITY

NOVEMBER, 2020

DECLARATION

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

Name: Daisy Mutuku

Reg No: A301/2150P/201

Signature …………………….....................Date……………………..……..

DECLARATION BY SUPERVISORS

We confirm that the work reported in this thesis report was carried out by the candidate under our supervision and has been submitted with our approval as University supervisors.

Name: Dr. Hellen Kamiri

Department of Agricultural Sciences, School of Agriculture and Biotechnology,

Karatina University

Signature …………………….....................Date……………………..……..

Name: Dr. Mugo Mware

Department of Natural Resources, School of Natural resources and Environmental Studies,

Karatina University

Signature …………………….....................Date……………………..……..

DEDICATION

This work is dedicated to my family,

Thank you for your unconditional love and support,

I push myself beyond the limit because of you.

ACKNOWLEDGEMENTS

I am grateful to my lead supervisor, Dr. Hellen Kamiri for her guidance, encouragement, patience and professional support during this study. I also thank my second supervisor Dr. Mugo Mware, who inspired and supported me throughout this study.

My gratitude goes to the Higher Education Loans Board (HELB) and Kenya Forestry Research Institute (KEFRI) for their financial support. Special thanks go to the USAID under NAS-PEER project for funding this research. I also thank the University of Nairobi and Karatina University for offering their facilities and services for laboratory analysis.

I thank Mpala Research Institute for their support during field work and Ilmotiok community, specifically the research assistants who participated in one way or another during this research. I am also grateful to the technical support I received from Mr. Ferdinand Anyika, Mr. John Kimotho and Mr. Bonface Muliro, for committing their time to assist me with my samples, I am forever grateful.

Many thanks go to my family for moral and prayer support and for believing in my hard work. I further extend my appreciation to Angela, Janeth, Kariuki, Kioko and Sainepo for your impeccable advice, moral support and true friendship when I really needed it; truly the sky is the limit.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS AND ACRONYMS

ANOVA: Analysis of Variance

AS: Aggregate Stability

ASAL: Arid and Semi-Arid Land

ASL: Above Sea Level

BD: Bulk Density

C: Carbon

Cm: Centimetres

CO_2: Carbon dioxide

DF: Degrees of Freedon

FAO: Food Agriculture Organization

g: grams

GEF: Global Environmental Facility

GHGs: Green House Gases

Gt: Giga tonnes

Ha: Hectare

HELB: Higher Education Loans Board

HS: Hillslope

HW: Headwater

IR: Infiltration rate

KEFRI: Kenya Forestry Research Institute

Kg: Kilogram

LSD:	Least Significant Difference
M:	metres
Mha:	Million hectares
Ml:	millilitre
Mm:	milimetres
Mt:	Metric tonnes
N:	Normal
NRCS:	Natural Resource Conservation Service
NS:	Not Significant
P:	Probability
Pg:	Petagrams
PLA:	Plateau
RIP:	Riparian
SAS:	Statistical Analysis System
SOC:	Soil Organic Carbon
SOCs:	Soil Organic Carbon stocks
SOM:	Soil Organic Matter
TZ:	Topographic Zone
UNEP:	United Nations Environmental Programme
UNIC:	United Nations Information Centre
USDA:	United States Department of Agriculture
VC:	Vegetation Cover

WMO: World Meteorological Organization

WMO: World Meteorological Organization

μm: Micrometre

ABSTRACT

Soil degradation activities in semi-arid grasslands have contributed to loss of vegetation cover, removal of surface soil, reduced soil organic matter and instability of soil aggregates thus reducing water infiltration rates in the soil. Soil losses are partly influenced by human-induced practices such as grazing, bush clearing and cultivation. The aim of this study was to investigate the influence of vegetation cover types and topographic positioning on soil organic matter, aggregate stability and water infiltration rates. The study was carried out in two semi-arid grasslands: Mpala and Ilmotiok ranches in Laikipia County, Kenya. Three vegetation cover types; (Tree, Grass and Bare gounds) and four topographic positions (Hillslope, Headwater, Riparian, and Plateau) were evaluated. A reconnaissance survey was done to demarcate a sampling grid of 4*4 km^2 area in each site. A stratified sampling design was then used to demarcate four transect plots (Riparian-RIP, Plateau-PLA, Headwater-HW and Hillslope-HS). Within the four plots, three subplots comprising of Grass fields, Bare grounds and Tree fields were identified. Soils were sampled within a 5 m radius at 0-10, 10-20, 20-30, 30-40, 40-50 cm depths for laboratory determination of organic carbon, bulk density and water stable aggregate during the dry season of May to August 2016. Organic matter was estimated from the organic carbon content by multiplying percent organic carbon with a factor of 1.724. Data was subjected to two-way analysis of variance using SAS statistical software. A multiple comparison test (T-test) using LSD at P=0.05 was done to separate the means of various soil parameters and to isolate the significant differences between each of the vegetation cover types, topographic zones and soil depths in both sites. All statistical tests were considered significant at the level of $P < 0.05$. The study revealed that soil organic carbon and organic matter (SOM) differed significantly at (P=0.0001) among the vegetation cover types, topographic zones and soil depth for both sites. Soil aggregate stability varied significantly between topographic zones (P=0.0124) but not between the vegetation cover types and soil depth in Ilmotiok site. Mpala site showed a significant difference in aggregate stability between the topographic zones (P=0.0152). However, no significant difference was observed in variation of aggregate stability between the vegetation cover types and soil depth (P=0.8998; P=0.8284), respectively. In Ilmotiok site, the highest infiltration rate was recorded in the Tree covered fields (73.3 mm/hr) and decreased in Grass fields and Bare grounds at 25 and 17 mm/hr, respectively. The Headwater zones had the highest infiltration rates (73.3 mm/hr) while the lowest infiltration rates were (0 mm/hr) in the Hillslope zones. The infiltration rates in Mpala site were highest in Bare grounds (37.8 mm/hr) and lowest in Tree fields with 5.7 mm/hr. The Headwater zones had the highest infiltration rates followed by Hillslope zones with (8.9 mm/hr) while the Riparian zones had the lowest infiltration rates (0.00 mm/hr). Overall, the study revealed that topographic positions and grazing management influenced soil properties in the grasslands, hence the need for holistic grazing management strategies that will promote restoration of these degraded areas and support the livelihood of the people in these areas.

CHAPTER 1: INTRODUCTION

1.1 BACKGROUND INFORMATION

Grasslands can be defined as ground covered by vegetation dominated by grasses with 10-40% tree and shrub cover. Grasslands' coverage is approximated at 40% globally and 10% of this coverage constitutes global soil carbon storage (Suttie *et al*., 2005). Grasslands are also valued for their ability to sequester soil organic carbon (SOC), depending on land management strategies because of their importance in the global carbon cycle (Franzluebbers & Doraiswamy, 2007). They also sustain the world's livestock through the production of forage (FAO, 2010). However, grasslands' functioning is disrupted by land degradation activities such as overgrazing, urban expansion and infrastructure development (UNEP, 2007). Presently, anthropogenic driven environmental changes affect a greater proportion of earth's grassland ecosystems thus altering their capacity to provide ecosystem functions and services needed by rural livelihoods (Cardinale, *et al*., 2012).

The reduction in the capacity of semi-arid wooded grasslands to carry out their main ecosystem functions has been described as land degradation by UNEP, (2007). Land degradation has been attributed to human disturbances such as accidental grassland fires, overgrazing, livestock trampling, and soil erosion processes. The loss of soil nutrients and decline in soil organic matter content is a result of grassland burning which releases about three times as much gas and particle emissions to the atmosphere as deforestation (Goldammer, 2000; Bagamsah, 2005). Overgrazing has been found to reduce water infiltration, soil moisture and fertility, accelerate runoff and soil erosion, increase soil bulk density, penetration resistance, soil ammonia and nitrate content and alter soil microbial activity (Czegledi & Radacsi, 2005). The loss of soil organic matter and disruption of soil aggregates has also been linked to soil movement through erosion processes (Itanna *et al*., 2011).

Soil aggregation is related to SOC content, and SOC is often increased in both macro and micro-aggregates (Cambardella & Elliot, 1994). Therefore, soil organic carbon can encapsulate within stable aggregates and protect the soil against microbial processes and enzymatic reaction (Lal, 2003). Poor soil management practices can affect the soil aggregate fractions (macro and micro aggregate) whose function in soil carbon sequestration is undeniable. Therefore, the re-distribution of soil organic matter in water-stable aggregates is dependent on proper soil use practices (Simansky & Kovacik, 2014). Von Lutzow *et al*., (2006) indicated the interactive functions of both soil organic matter and stable aggregates in improving the soil cohesive strength, water movement processes and resistance to erosion, and the overall increase in soil fertility. Thus, soil fertility and the biological diversity of semi-arid grasslands are greatly affected by any change in the quality and quantity of soil organic matter (SOM) (Kaiser *et al*., 2010).

The chemical and physical characteristics of soil are significantly affected by the quality of soil organic carbon which is categorized as a key component of soil quality assessment by Lal, (2004). Christensen, (2001) indicated the main consequence of land degradation on soil organic matter quality and quantity in grasslands as SOC stock depletion. In addition, the concentration of SOC in the top soil layer is estimated at 60-70% in grassland soils; therefore, any external soil disturbance contributes to depletion of SOC and overall decline in soil fertility. Eventually, the productivity of grasslands is affected, particularly production of forage for livestock and loss of biodiversity (Dong *et al*., 2012; Ruiz-Sinoga & Diaz, 2010). The importance of protecting the soil resource has been over emphasized by Du Preez *et al.,* (2011).

Limited knowledge exists on vegetation-soil interactions and about how vegetation status, including degradation is reflected in soils. Past studies have reported significant differences in soil properties under vegetation disturbed to various degrees in several aspects including chemistry and physical attributes such as soil organic matter and soil aggregation properties (Li *et al*. 2009). Research shows that vegetation structure (canopy cover, sapling density, litter depth, and woody debris) may be a main factor influencing soil and water loss (Casermeiro *et al*., 2004). Vegetation plays an important role in water and soil erosion control. Vegetation cover can reduce the kinetic energy of raindrops. Vegetation covered plots and the litter layer protect soil surfaces, increase soil surface roughness, impede overland flow and increase infiltrating time (Zheng *et al*., 2008). Vegetation root development can improve soil physical properties (e.g., soil strength, shear strength, structural stability and aggregate stability), which are closely related to soil erodibility, resulting in a contribution to soil loss control (Gao et al., 2009).

Topographic features such as slope and catchment area, control rates of redistribution of soil across sloping landscapes and have an effect on the amount and quality of soil organic carbon content that is found across the landscape in semi-arid grasslands (Yoo *et* *al*., 2005). At eroding portions of the landscape, soil erosion represents a widespread cause of soil quality loss and a threat to critical soil ecosystem services (Dominati *et al*., 2010) due to redistribution of soil particles and SOC across the landscape. Further, high rates of erosion diminish soil productivity, can influence soil carbon (C) sequestration potential, and alter soil hydrological properties (Gessler *et al*., 2000). Other researchers have emphasized the role of the physical environment, together with biotic and abiotic conditions, in affecting SOC's persistence in dynamic landscapes that experience erosion and terrestrial sedimentation or deposition (Berhe *et al*., 2012; Van den Bygaart *et al*., 2012).

The need to fully understand the concentrations of SOC and SOM content, the stability of soil aggregates and water infiltration rates as affected by vegetation cover, topographic positioning and soil depth led to the design of this study to enhance efforts for reversing land degradation menace and restoration of degraded soils in semi-arid grasslands.

1.2 STATEMENT OF THE PROBLEM

Human-induced activities including; overgrazing, soil erosion processes and runoff lead to loss of vegetation and gradually soil and land degradation in grasslands. Soil materials are transported from the steeper slopes as a result of the topographic position of the land and are deposited on the lowland areas leading to loss of essential soil elements such as carbon and nitrogen, and therefore influence the cycling of both elements in soils (Quinton *et al*., 2006). Eventually, soil organic matter content reduces, the existing soil aggregates are disrupted and water infiltration rates also reduce (Celik, 2005). Loss of essential soil elements such as nitrogen, phosphorus, sulfur, and micronutrients as a result of decreased SOM content could result to a decline in soil stability, fertility and grassland survival (Itanna *et al*., 2011). The availability of enough water in the soil for plant and vegetation growth could be influenced by unstable soil aggregates which lead to formation of a hard physical crust on the soil surface, resulting to decreased infiltration rates which further result to increased runoff and soil erosion processes (NRCS, 2011). Knowledge on the influence of vegetation covers and topographic positioning on soil organic carbon and organic matter content, soil aggregate stability and water infiltration rates particularly in semi-arid grasslands in Laikipia is scarce. Therefore, there was need to conduct this study in order to develop proper land management practices to restore and conserve grassland ecosystems in Kenya.

1.3 RESEARCH OBJECTIVES

1.3.1 Broad Objective

The overall objective of the study was to evaluate the effects of vegetation cover and topographic positioning on soil organic carbon and organic matter content, soil aggregates stability, and infiltration rates in semi-arid grasslands of Laikipia County.

1.3.2 Specific Objectives

1. To determine the effects of vegetation cover, topographic positioning and soil depth on soil organic carbon and organic matter content in semi-arid grasslands
2. To evaluate the effects of vegetation cover, topographic positioning and soil depth on soil aggregate stability in semi-arid grasslands

3. To determine the effects of vegetation cover and topographic positioning on water infiltration rates in semi-arid grasslands
4. To assess the influence of fencing/protected ranching on soil properties

1.4 HYPOTHESES

1. Soil organic carbon and organic matter content does not differ significantly under different vegetation cover, topographic positions, soil depths
2. Soil aggregate stability does not differ significantly under different vegetation cover, topographic positions and soil depth
3. Water infiltration rates does not differ significantly under different vegetation cover, topographic positions and soil depths
4. Fencing/protected ranching does not significantly influence soil properties

1.5 JUSTIFICATION

The interactions between soil organic matter, soil aggregate stability and water infiltration rates have been found to improve the overall soil fertility in grasslands when utilization of land is well managed. Proper land use practices ultimately contribute to sustainable productivity of grasslands with plant and vegetation growth and eventually support of community livelihoods. The influence of vegetation cover and topographic positioning on the levels of organic matter, available water in soil for plant growth and the stability of soil aggregates in semi-arid grasslands is fundamental to the development of conservation and restoration measures in these grasslands. Therefore, the need for up to date information on the status of soil degradation led to the design of this study to evaluate the effects of vegetation covers and topographic positioning on soil organic carbon and organic matter content, soil aggregate stability and water infiltration rates in semi-arid grasslands.

1.6 SCOPE OF THE STUDY

Grasslands are vulnerable ecosystems to land/soil degradation activities and continue to deteriorate due to poor management strategies. This study focused on evaluating how vegetation cover and topographic positioning in the grasslands have influenced soil properties. Of particular interest was soil organic carbon (SOC), soil aggregate stability and water infiltration rates. Soil organic carbon plays a major role in soil fertility improvement and the overall grassland production. The relationship between SOC and soil aggregate stability is positively influential in improving soil organic matter and soil water functions. As a result, the interaction of these three soil parameters is key in improvement of soil quality. The research was carried out between the months of June and August in 2016. The study was conducted in a degraded- wooded open grassland (Ilmotiok group ranch) and a protected wooded grassland (Mpala ranch) in Laikipia, Kenya. Laikipia region lies on the leeward side of Mt. Kenya, receiving approximately 500- 1000 mm of rainfall annually with temperature range of 22-26 ^{0}C. Sampling was done along a degradation gradient based on topographic transects and vegetation cover types. Soil samples were collected and analyzed in the laboratory using standard soil analysis procedures. After soil analysis, the data was analyzed using descriptive statistics and the research findings were reported in chapter five of this document.

1.7 LIMITATIONS OF THE STUDY

Like any other research, challenges were inevitable. Firstly, traversing of wild animals between the two ranches created interference with data collection activities where the plots were located. This meant data could only be collected with the guidance of a field guide officer. Secondly, the heavy rains in May to June destroyed transport networks and movements were restricted to when the rivers/streams were not overflowing. Additionally, the ranches were located in a poor communication network area. The use of walkie-talkies from the field guide officers was the only way of communication within the research locations. Finally, since the research area was on a community group ranch; Ilmotiok Ranch, despite having research permits, the owners were not ready to allow foreign movements within their land until the community elders intervened.

1.8 DEFINITION OF TERMS

Topographic position: The arrangement of the physical features of an area of land

Topographic zone: It is a plot area of 200 by 150 m in size, distinguished by the slope of the area.

Vegetation cover: It is field coverage with grass, tree/shrub vegetation, or an open bare land.

Soil organic carbon: It is a measureable component of soil organic matter and has an important role in the physical, chemical and biological function of agricultural and grassland soils.

Soil organic matter: The fraction of the soil that consists of plant or animal tissue in various stages of breakdown (decomposition).

Soil aggregate stability: Is a measure of the ability of soil aggregates to resist degradation when exposed to external forces such as water erosion and wind erosion, shrinking and swelling processes, as well as tillage.

Infiltration rate: The rate at which water moves into the ground at any given moment, regardless of the current soil saturation.

Land degradation: Deterioration in the quality of land, its topsoil, vegetation, and/or water resources, caused usually by excessive or inappropriate exploitation.

CHAPTER 2: LITERATURE REVIEW

2.1. Overview of Grassland Ecosystems in the World

Vegetation types in the world are diverse and can be grouped into different categories including; forest which constitutes closed tropical rainforest of lowlands, closed evergreen montane, sub-montane forests and mosaic forest/savanna; woodland and shrubland vegetation which includes savanna and dry forest; grassland; swamp and mangrove vegetation (Wondimagegn, 2019).

Grasslands are defined as areas dominated by grasses and forbs and have few or no trees. The savannas (tropical grasslands) are listed among the most endangered ecosystems in the world (Hoekstra *et al*., 2005). According to Wang & Fang, (2009) grasslands' coverage in the world is about two fifths in which about 70% are areas under agricultural production (Conant, 2012). In addition, ten percent of this coverage consists of the terrestrial biomass contributing to about 20-30% of the soil organic carbon pool in the world.

More than half of the world's livestock are raised in the semi-arid and arid lands where approximately 40% of the human population in the world is found (UNIC, 2011). The potential of grasslands to sequester atmospheric CO_2 as stable carbon in the soil could contribute to the mitigation of climate change (Allard *et al*., 2007). However, the accumulation and storage of carbon in grasslands is particularly influenced by biotic factors such as the grazing intensity (McSherry & Ritchie, 2013).

Research highlights that a great percentage of grasslands have become degraded mainly due to human-induced land use practices and are vulnerable to changes in the environment, including a shift in climate change, increasing atmospheric carbon dioxide concentrations, overgrazing activities, and natural disturbances such as increased woody plant encroachment (Kemp *et al*., 2013). The productivity of grasslands is highly affected by land degradation (UNEP, 2007) and therefore, any conservation mechanisms that restore the soil and vegetation will ultimately contribute to sequestration of carbon in the soil. The pastoral system, which is the backbone of the dry arid and semi-arid lands, has also been affected by the decline in grassland resources especially the quantity and quality of livestock feed necessitated by increasing environmental degradation (Kassahun *et al*., 2008).

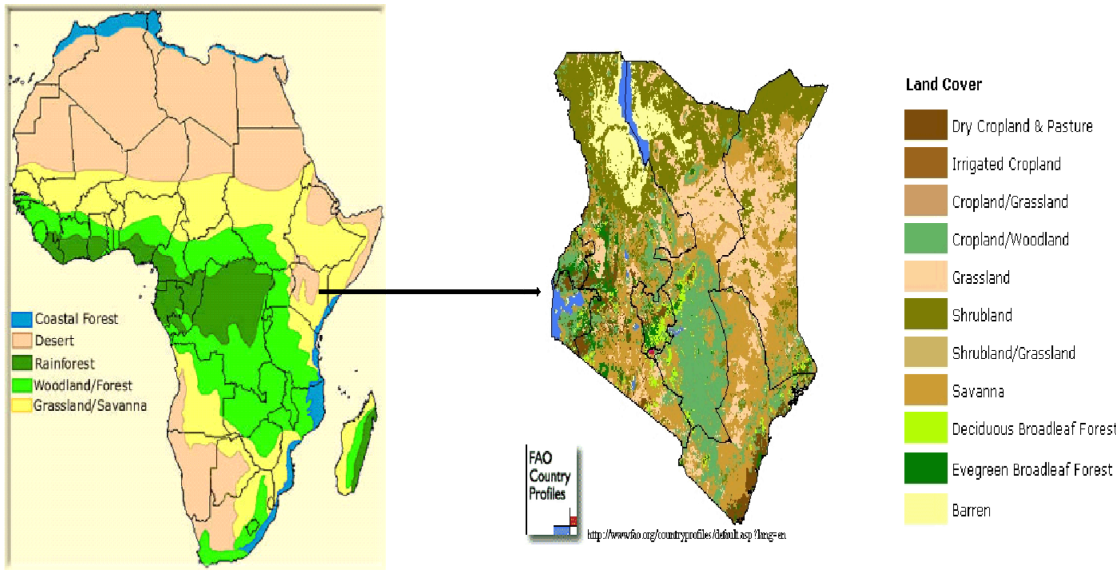

Figure 2.1: A map showing the distribution of major vegetation types in Africa and Kenya. (Adapted from; http://apassionforscience.pbworks.com/f/africaveg.jpg)

2.2 Grassland Ecosystems in Africa

A large percentage of land in Africa is classified as arid and semiarid lands (ASALs) whose communities are majorly dependent on livestock production for survival (Beatrice *et al*, 2012). About 73% of the land in East Africa is termed dry land with annual rainfall in the region ranging from 150 mm in the arid and semi-arid areas to 2000 mm in the wet regions (Mati, 2005) (Figure 2.1). The rainfall in the region is not uniformly spread, but comes as intense storms such that sometimes a single rainfall event may exceed the average monthly amount (Hussein *et al*., 2006). Such irregularity not only causes soil erosion but also results in inadequate supply of water for vegetation growth which leads to degradation. The African grasslands are vulnerable to degradation due to over exploitation of land to grazing activities which have consequences on the sustainability of these lands with respect to the grazing intensity (Kioko *et al*., 2012). High grazing intensity affects the production, composition and diversity of plant species leading to destruction of the most tolerant species for livestock grazing (Kgosikoma, 2012).

In Kenya, over 80 percent of the land area is classified as ASALs and majority of this area is affected by moderate to severe land degradation and desertification (UNEP, 2007), with approximately ten million people living in these areas. Additionally, the current rapid increase in population and the associated demand for land in the high potential areas has forced people to migrate to the ASALs (Akuja *et al*., 2005), resulting in severe land degradation. To ensure sustainable productivity of these lands, efficient strategies have to be devised to fight land degradation (Mganga *et al*., 2010), which has been a dilemma to many researchers because efficient soil conservation techniques are usually not a priority among farmers and pastoralists due to their indirect and long term benefits to them.

2.3 Significance of Grassland Ecosystems

Grasslands are a source of livelihoods for about 800 million people in the world and 10 million people in the Kenyan population (White *et al.*, 2000). This means, on a daily basis, almost every human being benefits from grasslands in their products such as food, fiber and fuel. Many smallholder farmers in Africa also rely on grasslands for their livestock feeds which contribute immensely to food security through production of meat and milk (O'Mara, 2012). In addition, grasslands play a major role in improving grassland productivity through increased soil fertility, reduced water and wind erosion and surface runoff processes (Sochorec *et al.,* 2015). Grasslands also protect the soil surface from intense falling raindrops and disperse their energy, slowing down surface runoff and increasing water infiltration.

2.4 Concept of Land Degradation

Land degradation is a process in which the value of the biophysical environment is negatively affected by a combination of human induced processes acting upon the land (Lal, 2010). It describes how land resources have changed for the worst (Bai *et al*., 2008). Land degradation and soil degradation are often used interchangeably and the two are closely linked, as soil degradation processes constitute the most significant land degradation processes (Stocking & Murnaghan, 2000).

Land degradation processes have been categorized by World Meteorological Organization, (2005) as chemical degradation comprising water and wind erosion processes, acidification, salinization, and fertility depletion; physical degradation which is mostly crusting and compaction on the soil surface and biological degradation including decline in soil organic carbon, and reduced ecosystem biodiversity. Land degradation is a world menace as it is continuously affecting food security among countries and creating poverty particularly to the vulnerable rural people (Bezuayehu *et al*., 2002). Moreover, it is a threat to the fight against climate change (Neely *et al*., 2009) and may induce migration (Requier-Desjardins, 2008). Paulos, (2001) indicates that several factors influenced by man that could contribute to degradation include soil factors, topography and agro-ecological parameters. The severity and extent of land degradation is increasing across the world, with statistics showing that 10% of grasslands are undergoing degradation (Blaikie & Brookfield, 2015).

2.5 Causes of Soil and Land Degradation

The interaction between the natural and the human social systems has been found to cause land degradation thereby determining the success or failure of a land resource (Berry, 2003). Several factors have been identified as causes of land degradation by WMO, (2005) including; socioeconomic factors (poor land management practices, income, human health and institutional support), biophysical factors (unsuitable land use), and political factors (lack of incentives and political instability).

The reliance of a country's growing population on exploitative agricultural practices by small scale farm holders has geared the increase in land degradation (Gebreyesus & Kirubel, 2009) Again, human impact in form of unsustainable land use management can further lead to degeneration of the arid lands (i.e. degradation like salinization, water logging, erosion and desertification) affecting a significant percentage of the human population (UNIC, 2011). The depletion of the soil organic carbon pool has also been linked to soil degradation in the semi-arid grasslands making it a reliable indicator for monitoring soil degradation (Krupenikov *et al*., 2011).

2.6 Effects of Land Degradation

Socio-economic effects of land degradation

High persistent poverty levels across the world are associated with land degradation which has not only threatened food security but also the ecosystem functioning and provision of environmental valuable services (Bossio *et al*., 2004). Consequently, the health and wellbeing of the human population in the dry lands is adversely altered (Shibru, 2010). Economically, the effects of land degradation are severe for the rural people in Sub-Saharan Africa whose 90% source of income is agricultural production (Project Development Facility, 2007).

Ecological effects of land degradation

The loss of habitats and ecosystem biodiversity, shift in water flows and sedimentation of rivers and dams has been linked to land degradation in the environment (Project Development Facility, 2007). Further, environmental degradation has led to reduced soil fertility through loss of soil surface materials which affect the type of vegetation growth in grasslands, as well as the changes in surface water volumes and overall reduction in available water for plant productivity (Berry 2003). Mulugeta, (2004) found that land degradation has threatened agricultural production and biological resources in rural areas. Global Environmental Facility, (2006) emphasized that land degradation has adversely affects the global carbon cycle and the water cycle essential in cycling of nutrients important for plant productivity in grasslands.

Losses of forest cover through deforestation affects the circulation of air and may influence climate change patterns, as a result of land degradation. In addition these effects translate to change in river flows, siltation of lakes and dams, and erosion processes which reduce the productivity of grassland ecosystems (Project Development Facility, 2007).

Depletion of nutrients is a form of land degradation, because it been reported to reduce soil ecosystem services by 60% between 1950 and 2010 particularly in the sub-Saharan rural countries (Leon & Osorio, 2014). More than 500 million hectare (Mha) in the dry land areas has resulted from soil degradation (Lamb *et al*., 2005), which amounts to 33% of earths' land coverage (Bini, 2009).

2.7 Mechanisms Used to Prevent Land Degradation and Enhance Restoration Strategies

Land degradation prevention mechanisms depend on the form of degradation. Addition of nutrients through soil amendments and restoration of vegetation cover to depleted soil has been used in in several studies to reverse the effects of land degradation (Scherr & Yadav, 1996). However, there are challenges in reversing some forms of land degradation than others such as total loss of the top soil to erosion (Coxhead & Ygard, 2008). Restoration of vegetation cover can be adopted in the affected lands to enhance reduced removal and transportation of soil surface materials, and reduced runoff speed and erosion processes in the soil (Coxhead & Ygard, 2008).

Improved land management practices could promote grassland productivity and soil carbon replenishment (Wang *et al*., 2011). Also, identifying the most affected areas by land degradation by estimating the extend of degradation and monitoring the consequences of rehabilitation actions (Campell *et al*., 2003) as well as assisting in the planning and designing of appropriate mitigation measures could contribute to reversing or prevention of the degradation menace.

2.8 Soil Organic Carbon in grasslands

Grassland soils are considered important reservoirs because of their potential to store carbon (C) for many years (Brevik & Homburg, 2004; Yan-Gui *et al*., 2013). The storage of C translates to over 71% of the world's terrestrial C pool (Lal, 2009). The top 100 cm of soil is approximated to store 1200 to 1600 Pg of soil organic carbon in arid and semi-arid regions worldwide (Diaz-Hernández *et al.*, 2003). In the past, studies have shown differences in C storage rates in soils depending on land use and management practices particularly overgrazing activities clay mineralogy and climate patterns (Wang *et al*., 2010), soil textural properties, topography and slope (Hontoria *et al*., 2004), and tillage practices (Umakant *et al*., 2010).

The effects of poor land management could lead to carbon losses making the soils a source of greenhouse gas (GHG) emissions (Ciais *et al*., 2010; Powlson *et al*., 2011). Soil organic carbon lost in form of CO_2 amounts to 42-78 billion of tons of carbon with negative consequences on climate change and food security (Lal, 2004). These greenhouse gas emissions particularly accumulation of CO_2 in the atmosphere can be prevented through proper soil and vegetation management (Post & Kwon, 2000).

Fifty percent (50%) of all soil organic matter consists of soil organic carbon (Wilke, 2005). SOC offers buffering capacity to several soil properties such as soil temperature, water quality and soil pH (Pattanayak *et al.*, 2005). SOC content increases in the soil create greater pore spaces which contribute to more water and nutrients retained in the soil (Greenhalgh & Sauer, 2003).

2.9 Soil Organic Matter in Grasslands

Soil organic matter (SOM) functions are well described by Johnston *et al.,* (2009) therefore; loss of SOM is considered a threat to sustainable productivity of grasslands (Amundson *et al*., 2015). The availability of SOM contributes to increased soil fertility, biodiversity growth, retention of water quality, resistance to erosion and runoff processes, and climate change mitigation (Adhikari & Hartemink, 2016).

Conservation and proper management of SOM is essential to ecosystem functioning as it provides services which can be described as soil fertility functions (Feller *et al.,* 2001). Appropriate management of SOM minimizes agricultural impact on the environment through carbon sequestration, erosion control and preservations of soil biodiversity (Six *et al.,* 2002). Loss of SOM will therefore reduce soil fertility, degrade soil structure and water holding capacities and ultimately, leads to land degradation (Lawal *et al.,* 2009). These losses are particularly critical in semi-arid grasslands because of the effects of global warming which accelerate SOM decomposition thereby, reducing SOM levels (Jenkinson *et al*., 1991).

2.10 Soil Aggregate Stability

Soil aggregates are the basic units of soil structures and contribute to carbon sequestration and stabilization in the soil (Brevik *et al*., 2015). The ability of the soil to retain its arrangement of solids and pore spaces after exerting external or mechanical stress or destructive forces is described as soil aggregate stability (Diaz-Zorita *et al.,* 2002). Aggregate stability is also directly and indirectly linked to soil erosion and degradation (Mataix-Solera *et al*., 2011).

Past studies reported that soil stability for example, water-stable aggregates is a crucial determinant of ecosystem functions (Six & Paustian, 2014). Good soil structure is fundamental to the storage and stabilization of soil organic carbon (Gelaw *et al*., 2013). Over exploitation of land could often result to destruction of soil structures and the foundations for organic carbon sequestration and increased levels of soil erosion (Tang *et al*., 2014).

Soil aggregation is important in protecting organic matter (Von Lutzow *et al*., 2006) and improving overall soil fertility. This is achieved through reduced soil erosion and increased water infiltration rates, and water holding capacities in the soil (Oades, 1984). Therefore, aggregate stability is a critical aspect of soil quality because it determines root penetration, soil organic matter stabilization, susceptibility of soil to compaction and soil erodibility (De Gryze *et al*., 2006).

Several researchers in the past have identified a positive association between plant biomass and soil aggregate stability. Further, plant roots and plant cover (Peres *et al*., 2013; Chaudhary *et al*., 2009) are positively linked to aggregation of soils. This relationship between plant biomass and aggregate stability of soils is also determined by factors that influence plant productivity such as soil moisture content and water infiltration capacity, available plant nutrients and the composition of plant species (Reinhart *et al*., 2016).

2.11 Water Infiltration In Grassland Ecosystems

The flow of water into the soil from the surface by downward or gravitational movement is described as infiltration (Zhang *et al*., 2010) and it is a key process in the water cycle since it controls the surface water–groundwater relationship (Ward & Robinson, 1989). Soil properties play a crucial role in this process and soils are the interface through which water infiltration occurs (Cerda, 1997). Any ecosystem modifications produce changes in infiltration that can accelerate erosive processes (Cerda, 1998).

The size and amount of pore spaces in soils is a crucial determinant of soil water content in grasslands (Tuller & Or 2004). However, degradation processes disrupt the soil structure and breakdown organic matter which affects the soil moisture content. The differences in the mineral composition and the geometry of pore network of many grassland soils can affect soil water retention (Li *et al*., 2009 and Meskini *et al*., 2014).

Soil structure influences water movement into the soil profile and therefore controls soil system functioning. This facilitates development of vegetation cover, and both factors indirectly affect soil erosion rates (Mataix-Solera *et al*., 2011). Land management practices can significantly influence water infiltration properties (Zhou *et al*., 2008).

Soils exposed to human impact particularly compaction by livestock are often stripped of organic-rich upper horizons, thereby increasing bulk density and reducing water infiltration rates (Li & Shao, 2006). Decreased water infiltration in grasslands resulting from human land use, has been previously demonstrated to be a direct consequence of altered soil hydrology (Leblanc *et al*., 2008). Overgrazing activities leads to deformation of the top soil which induce changes to the pore structure thus impacting air and water conductivities in the soil profile and soil water retention characteristics (Krummelbein *et al*., 2006; Kutilek *et al*., 2006).

Infiltration is a fundamental process in ecology because of its contribution to water availability for plants and its impacts on soil surface runoff and erosion processes in the semi-arid landscapes (Michaelides *et al*., 2009). The amount of runoff controls the vegetation composition and pattern in grasslands which define the amount and variation of infiltration rates in the soil (Van Schaik, 2009). Human induced activities such as grazing disturbance may lead to changes in vegetation and soil properties and disrupts soil water properties and gradually changes the functioning of the ecosystem to a non-productive state (Briske *et al*., 2005). These changes can negatively affect soil quality because the removal of fine soil particles and litter by erosion reduces organic matter and nutrient concentration in the soil (Schiettecatte *et al*., 2008). The reduction in available soil nutrients and degradation of soil physical conditions limits plant and vegetation growth (Bisigato & Bertiller, 2004).

2.12 Research Gaps

Grasslands have numerous functions to the human population in the world, yet the land degradation menace is increasingly threatening the survival of the grasslands which is the source of rural livelihood. Soil organic matter is crucial in formation of soil aggregates therefore any losses of SOM could reduce soil cohesion causing instability of aggregates which could lead to reduced water infiltration rates. Past studies have reported on the effects of topography on SOC content as well as the role of vegetation cover on SOC content of soils in different environments. However, there is still scarce information in literature on how vegetation and topographic positioning influence SOM content, aggregate stability and infiltration rates in semi-arid grasslands.

CHAPTER 3: MATERIALS AND METHODS

3.1 DESCRIPTION OF STUDY SITE

3.1.1 General Description of the Study Region

The study was carried out in two grassland regions in Ewaso Nyiro in Laikipia County, Kenya; in a degraded open grassland (Ilmotiok group ranch) and a protected grassland (Mpala ranch). The two sites have varying grazing management systems. Mpala ranch lies beneath the shadow of Mt. Kenya, in the center of Laikipia County. Mpala is a private commercial ranch which is fenced and protected and incorporates wildlife conservation with controlled livestock grazing. Ilmotiok group ranch borders Mpala ranch to the North-west and it is occupied by pastoral communities, who practice open uncontrolled grazing of livestock herds including cows, poultry, donkeys, goats, sheep and camels.

The two ranches in Laikipia County are located in agro-ecological zone (AEZ) IV with rainfall ranges of 400 mm to 800 mm annually (Sombrek *et al*., 1982) (Figure 3.2). Altitude range of the sampling sites in Mpala was between 1627 to 1686 meters above sea level (ASL) while in Ilmotiok ranch the sampling sites were located between altitudes 1638 to 1661 meters ASL. The common vegetation comprises of drought tolerant trees and grass species largely classified as grassland, bushland, woodland and dry forest.

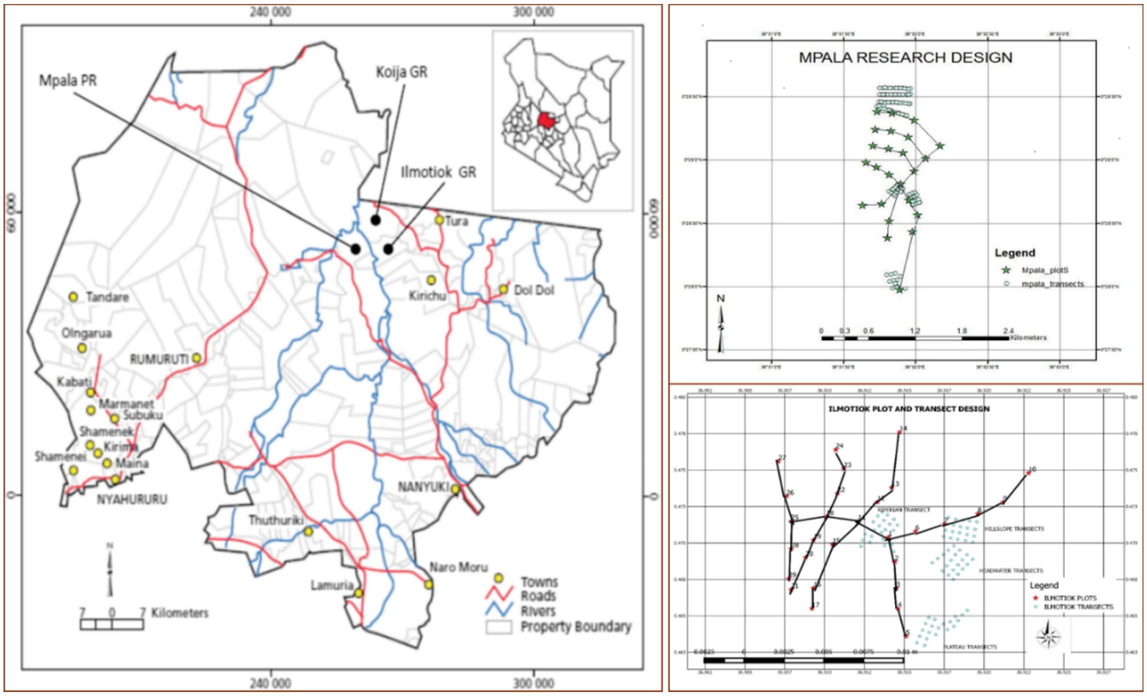

Figure 3.1: Location of study sites and sampling plots in Mpala and Ilmotiok ranches in Laikipia County, Kenya. (Adapted from Butynski & De Jong, 2014). *(Source with modifications:* KEFRI- NASPEER project)

3.1.2 Climate

The area receives bimodal type of rainfall which fall in two seasons; the main wet season occurs during April to May, often accounting for 80% of total annual rainfall, while a second wet season occurs later in the year in October-November. Daily temperatures vary with altitude and season; mean temperatures generally range within 22-26°C and temperature minimum and maximum are 6-14°C and 35°C, respectively (Figure 3).

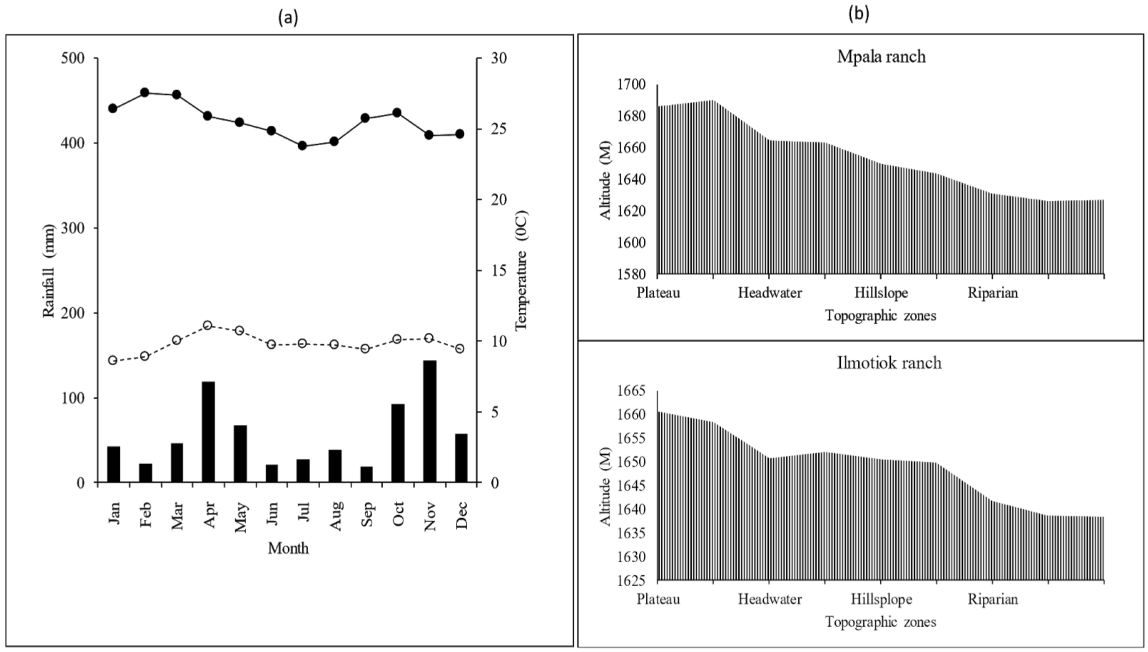

Figure 3.2: Map (a) showing monthly rainfall in (mm) shown as bars minimum (o) and maximum air (●) temperature (°C) for Laikipia County and (b) topographic orientation of the study sites. Rainfall and temperature data are based on 10-year average. Map adopted from Kibet *et al*., (2016); sampling positions adopted with modifications from KEFRI- NASPEER project.

3.1.3 Soil and Vegetation

Soils are dominated by Ferric and Chromic Luvisols (red sandy loams) and Pellic Vertisols (black cotton soils), (Butynski & De Jong, 2014). The vegetation in the two sites is comprised of grass, shrubs/tree covers and to some extent bare patches in severely degraded areas. The grass cover is dominated by *Themeda triandra*, *Pennisetum mezianum*, *P. straminium*, *P. massaiense*, , *P. kikuyuensi*, *Bothriochloa insculpta, Cenchrus ciliaris*, *Chloris* spp. *Digiteria* spp. and *Cynodon* spp. Areas which are overgrazed are dominated by unpalatable species of *Harpachne schimperi* and Aristida *adoensis*. Major tree species which dominate the region include *Acacia Senegal*, *A. brevispica*, *A. drepanolobium*, *A. gerrardii, A. atbaica*, *A. melliphera*, *A. tortilis*, *Combretum* and *Terchonanthus* spp. and shrubs; *Croton dichogamous*, *Grawier similis*, *Euphorbia busei* (Young *et al*., 2013; Lalampaa *et al*., 2016).

3.2 METHOD OF DATA COLLECTION

3.2.1 Site Selection and Experimental Design

A reconnaissance survey of the region was undertaken prior to the initiation of the study by a multidisciplinary team consisting of ecologists, geographers and soil scientists to identify and demarcate sampling sites and fields based on degradation gradient (degraded- wooded open grassland and protected wooded grassland); topographic transect (Plateau, Hillslope, Head water, Riparian) and vegetation cover (Tree cover, Grass cover and Bare ground). The sampling sites comprised of a 4 km radius in each of the sites (Mpala and Ilmotiok ranches) hereby referred to as 'super sites'. In each super site, four topographic transects each measuring 200 m long were demarcated. Topographic transects were classified as Hillslope (HS), Headwater (HW), Plateau (PLA) and Riparian (RIP) and differentiated based on position in the grassland. The sampling set up is shown in Figure 3.3 and Figure 3.4. The upper zone comprising of the Hillslope has moderate to high slope and highly covered by woody vegetation. In the Headwater and Plateau, the zones are covered by grass and woody vegetation while Riparian zones are located at the lower plains with tree and woody vegetation as the main vegetation type. The Riparian zones are depositional areas with high wet soil in some areas. Further descriptions of the topographic setting are described in Table 3.1.

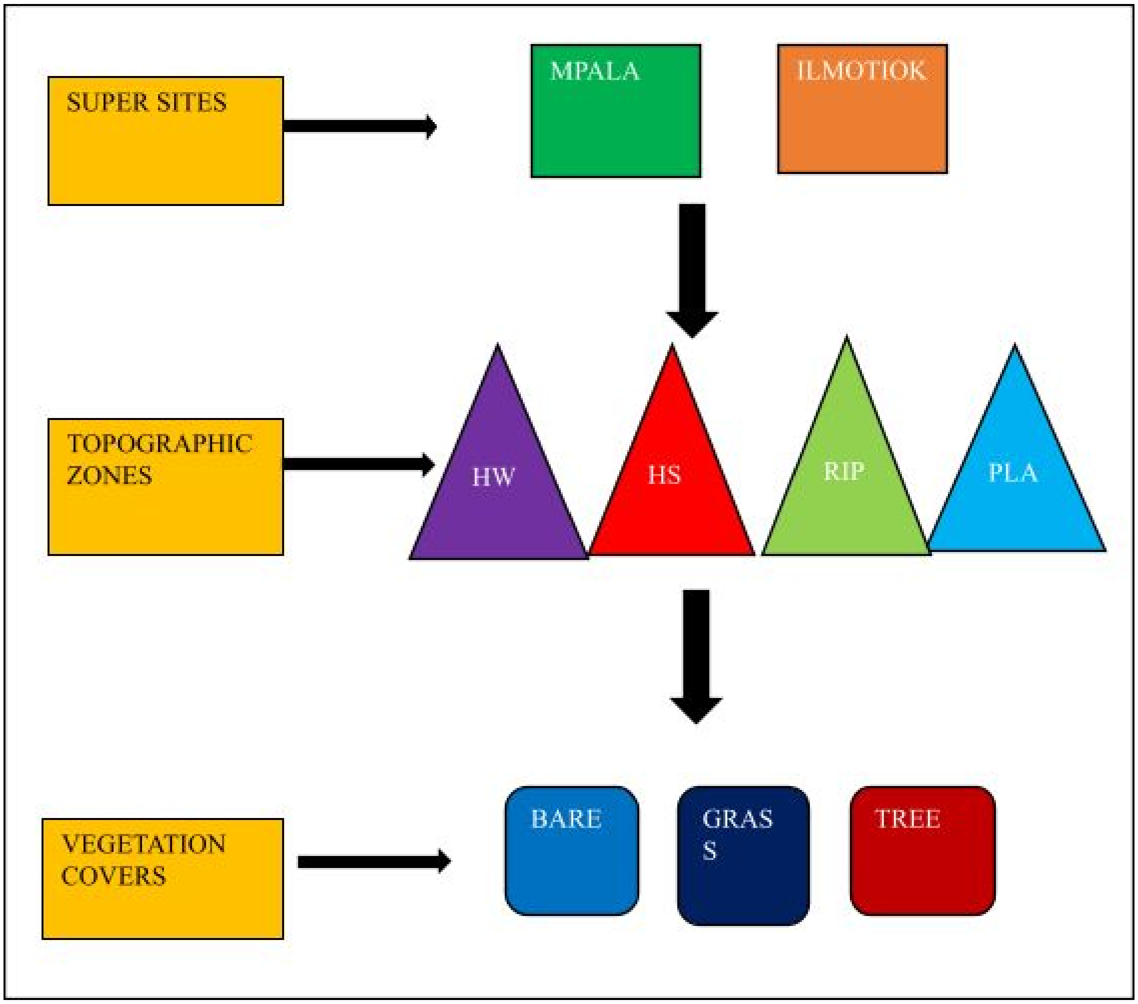

Figure 3.3: Experimental layout showing the sampling positions. ***Hillslope (HS), Headwater (HW), Plateau (PLA) and Riparian (RIP)***

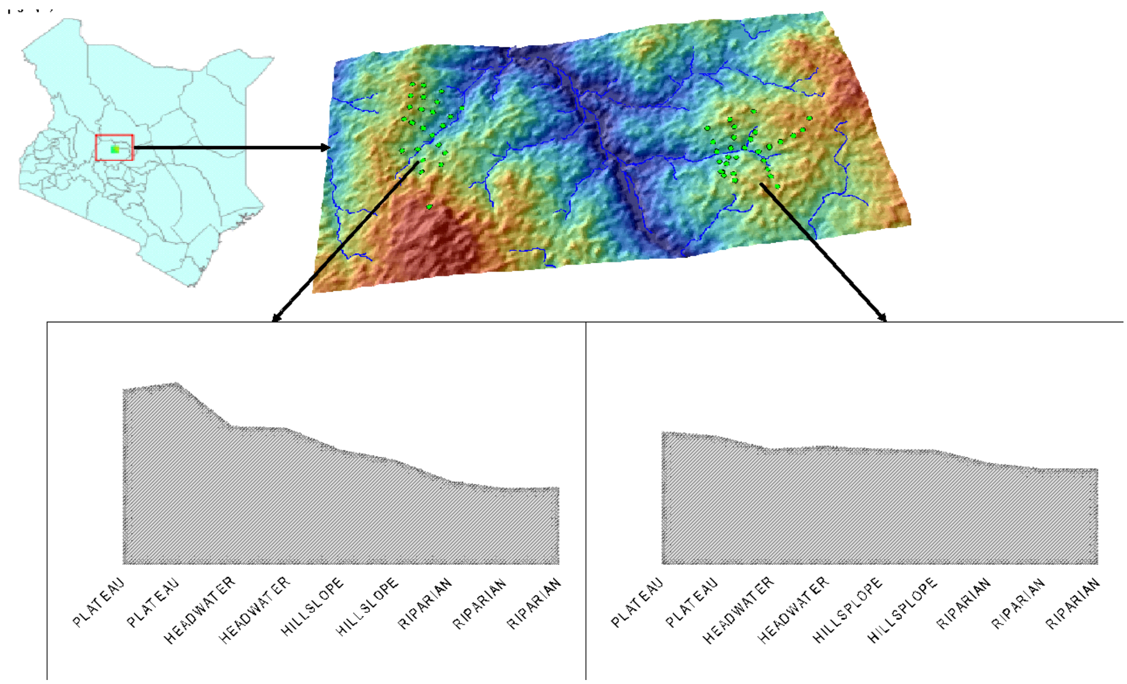

Figure 3.4: Location of study sites and sampling plots in Mpala and Ilmotiok ranches in Laikipia County, Kenya. ***(Source with modifications:*** **KEFRI- NASPEER project).**

In each topographic transect, three fields of size 200m*150m were delineated based on vegetation cover: thus (1) Tree cover (fields with more than 50% vegetation cover of trees or shrubs; (2) Grass cover (fields with natural grass species with moderate grazing); and (3) Bare grounds (fields with sparse grass vegetation or open grounds reflecting degradation after several years of intensive land use). The vegetation cover types are further described in Table 3.2. Similar vegetation cover types within the same topographic transect were identified as replicates provided soil conditions were the same and the soil showed low heterogeneity.

Table 3.1: **Description of the topographical zones in the two sites (Mpala and Ilmotiok)**

Topographic Zones	Slope (%)	Dominant soil texture	Vegetation cover	Surface characteristics
Hillslope (HS)	>10	Sandy-silt loams, gravel and stones	High coverage of bare grounds	Gravel and stones, erosion features, removal of top soil
Headwater (HW)	5-10	Sandy clay soils,	High grass vegetation cover and medium tree/woody vegetation cover	Medium bare ground areas
Plateau (PLA),	<5	Sandy loam soils.	Medium grass and tree/woody vegetation cover	Medium bare ground areas
Riparian (RIP)	<2	Sandy clay, Wet soils,	High coverage by tree/woody vegetation, high canopy shade cover	Deposited soil materials from upstream

Mpala Ranch; Elevation at 1700-2000 m A.S.L (Kareri, 2009)

The three replications of the same vegetation cover in the different topographic transects at each site were located within 50 m from each other. This approach substituted space for time and therefore the reference vegetation cover (Bare ground) and the topographic transect sampling units were located such that differences in geologic, topographic and climatic conditions were negligible. Therefore, any differences in soil properties under these conditions were assumed to be attributed to the differences in vegetation cover. This approach has been employed in several studies such as Kamiri *et al*., (2013) when studying land use changes in wetland systems in East Africa and in Ethiopia by Bewket & Stroosnijder (2003). The three vegetation cover types were nested on the topographic transect plots established based on a randomized complete block design for measuring soil properties (Figure 3.3).

Table 3.2: Description of the sampling units within the selected topographic zones in the study areas (Mpala and Ilmotiok)

Vegetation cover	Land use description	Land use history	Main vegetation type
Tree cover	Grassland vegetation moderately disturbed by grazing and human activity, with tree canopy coverage near 50%	Wildlife conservation	Tree dominated by acacia species; *Acacia drepanolobium*
Grass cover	Grass vegetation with 40-70% canopy cover	Moderate grazing during the past 10 years	Grass dominated by *Themeda triandra*, *Cynodon dactylon*.
Bare ground	Areas with minimal vegetation left to regenerate after extended period of grazing	Intensive grazing over the past 10 years	Few species of grass *Penisetum kikuyiensis*

Tree: fields with tree cover; Grass: grass covered fields; Bare ground: fields with minimal vegetation.

3.2.2 Soil Sampling

In each vegetation cover type, a 3 × 3 m subplot was randomly measured and soil samples were taken from 0-10, 10-20, 20-30, 30-40, and 40-50 cm depth layer using a 5 cm diameter soil auger. For each depth, the soil was randomly sampled from six soil cores within a 5 m radius and bulked to prepare a composite sample. This was done to increase the precision of estimate over the entire population, as soils of each replicate are relatively homogeneous.

Similar vegetation cover types located within 50 m from each other in the same topographic transect were identified as replicates provided soil conditioned were the same and the soil showed low heterogeneity. About 500 g of soil was collected per replicate for chemical and physical analysis in the laboratory. A total of 360 soil samples equivalent to two sites, four plots, three treatments, three replications, and five sampling depths (2*4*3*3*5) were collected. Soil samples were sorted and labelled according to the various sites, plots, transects, treatments and depths to reduce risk of mixing. The soil samples were air-dried, ground and sieved to pass through a 2 mm sieve for subsequent analysis in the laboratory. Water infiltration rate was measured in situ, on the soil surface using a minidisk infiltrometer (Zhang, 1997)

Figure 3.5: Soil sampling process in Bare grounds in Ilmotiok and Mpala ranches, Laikipia County

3.2.3 Soil Analysis

A routine soil analysis was done to measure soil bulk density and texture as part of the main analysis. Organic carbon and aggregate stability were analyzed based on recommended procedures for tropical soils (Okalebo *et al.,* 2002) as described in subsequent sections.

Soil aggregate stability

The wet-sieving method described by Six *et al*., (2002) was used to determine soil aggregate stability. The soil was first passed through an 8 mm sieve from where 50 g was weighed for wet-sieving. The 50 g sample was submerged in water over a 2 mm sieve for 5 minutes to allow slaking, followed by sieving for 2 minutes to obtain large macro aggregates. The soil that passed through the sieve was re-sieved for 2 minutes using a 250 µm (micrometre) sieve to obtain the small macro aggregates. The aggregates not captured by the 250 µm sieve were then sieved for 2 minutes using a 53 µm sieve to obtain the micro aggregates. After sieving with a 53 µm sieve, 250 ml of the filtrate containing silt and clay was obtained. The four different samples were dried at 60°C for 48-72 hours before weighing. Sand content (used for sand correction) of the aggregates (>53 µm) was determined by dispersing a subsample of aggregates from selected soils with sodium hexametaphosphate. For isolation of micro aggregates enclosed in macro aggregates, 5 g of macro aggregates (2.5 g each of large and small macro aggregates) was moistened with water (50 ml) and kept overnight at 4°C before isolation. Isolation was then done according to the procedure of Six *et al*., (2000). Macro aggregates were shaken with 50 glass beads over a 250 µm sieve and small macro aggregates were retained. Micro aggregates were immediately flushed onto a 53 µm sieve (by steady water flowing through the isolation device) and later sieved to leave only the water-stable ones. Isolated silt and clay was obtained by drying 250 ml of the filtrate at 60 °C for 72 hours.

Calculation: (source: Kemper and Rosenau, 1986)

$$\%\, WSA = \frac{((Ma+s)-Ms)}{(Mt-Ms)} * 100$$

Where:

WSA- Water stable aggregates;

$Ma+s$ - Mass of resistant aggregate plus sand (g)

Ms- Mass of sand fraction alone (g)

Mt - Total mass of the sieved soil (g).

Soil organic carbon

The Walkley-Black method, (1934) was used for determination of soil organic carbon. One (1 g) sample of soil previously passed through a 2 mm sieve to remove the coarse fraction, and ground to pass a 0.5 mm sieve was weighed into an Erlenmeyer (conical) flask. Ten (10 ml) of 1N (Normal) potassium dichromate was added using a pipette, and mixed with the whole sample by swirling gently. Twenty (20 ml) of concentrated 36 N sulphuric acid was transferred into a measuring cylinder. This was poured in a steady stream into the center of the soil-dichromate mixture. There was an immediate reaction and considerable heat was produced: it was placed on a sheet of asbestos and allowed to cool for about 20 minutes. Distilled water was added to bring the volume to about 200 ml. Approximately 5.0 ml 85% orthophosphoric acid (H_3P0_4) and about 5.0 ml of diphenylamine sulphonate indicator were added. Titration was done with 0.5 N ferrous sulphate (or ferrous ammonium sulphate), the exact normality of which was obtained by titrating 10.0 ml of dichromate in a blank. As the end point is approached the turbid dark blue colour became greenish, changing to a clear pale green quite sharply at the end point itself. Percent carbon in the sample was formulated as below (Walkley & Black, 1934);

$$\%\ Soil\ organic\ carbon\ = \frac{(V\ Blank - V\ Sample) * 0.3 * N}{Wt}$$

***Where*:**

N = Normality of Ferrous sulphate (N=10/Vblank)**,**

V Blank = volume of Ferrous sulphate solution required to titrate the blank**,**

V Sample = volume of Ferrous sulphate solution required to titrate the sample.

Wt = weight of the soil sample

Calculation of SOM

Soil organic matter (SOM) bears roughly 58% (the "van Bemmelen factor") carbon. Therefore, SOM was calculated by multiplying SOC with a factor of 1.724 (Pribyl, 2010).

Water infiltration rate

Infiltration rate of soil was measured in situ, within the same soil sampling locations using a mini disk Infiltrometer as detailed by Zhang (1997). First, the starting water volume was recorded. At time zero, the Infiltrometer was placed on the surface, assuring that it made solid contact with the soil surface. The volume was recorded at regular time intervals as the water infiltrated into the soil. The time interval was 30 minutes between readings for 2.5 hours. The cumulative infiltration vs. time was measured and the results were fitted with the function:

$$I = C_1 t + C_2 \sqrt{t}$$

***Where*:**

I = infiltration rate, C_1 (ms^{-1}) and C_2 ($ms^{-1/2}$) are soil parameters. C_1 is related to hydraulic conductivity, and C_2 is related to soil sorptivity.

The hydraulic conductivity of the soil (k) was then computed from $\mathbf{k=C_1/A}$ where, C_1 is the slope of the curve of the cumulative infiltration vs. the square root of time, and A is a value relating the van Genuchten parameters for a given soil type to the suction rate and the radius of the Infiltrometer disk. The van Genuchten parameters for the 12 texture classes were adopted from Carsel & Parrish (1988).

3.3 DATA ANALYSIS

Data on the soil parameters was subjected to two-way Anova separately for each site using Statistical Analysis System (SAS) software to test for significant differences in variation of soil organic carbon, organic matter, water stable aggregates and water infiltration rates among the vegetation covers and topographic zones. A multiple comparison test (T-test) using LSD at $P=0.05$ was done on the vegetation cover types and topographic zones to separate the means and isolate the significant difference between each other, for all the soil parameters.

Additionally, a multiple linear regression analysis was done to identify the influence of the vegetation covers and topographic zones on each of the soil parameters in the two sites. Statistical tests were considered significant at the level of $P < 0.05$ unless otherwise stated.

A t-test analysis (unpaired two samples assuming unequal variances) was done to compare the mean SOC, SOM, bulk density (BD), soil organic carbon stocks (SOCs), aggregate stability (AS), and infiltration rate (IR) in Mpala (protected) with Ilmotiok ranches. The purpose of the analysis was to determine if fencing/protected ranching influenced the said soil properties. The three treatment means (Grass fields, Tree fields and Bare grounds) were tested at 0.05 level of significance for SOC, SOM, BD, SOCs, AS, and IR .

CHAPTER 4: RESULTS

4.1 INTRODUCTION

This chapter presents the findings of the laboratory analyses of soil samples for SOC, SOM, SOCs, bulk density, soil texture and aggregate stability collected in three vegetation cover types along four topographical transects at soil depth intervals of 10 cm between 0-50 cm in the two ranches; Mpala and Ilmotiok in Laikipia grasslands in Kenya. Further, the results for water infiltration rate which was measured directly in the field have also been presented in this section. The results presented consist of four sub-sections; effects of (i) vegetation cover and topographic positions on SOC and SOM content, (ii) vegetation cover and topographic positions on soil aggregate stability, (iii) vegetation cover and topographic positions on water infiltration rate and (iv) the effects of grazing and grassland management on soil properties in the two sites (Mpala and Ilmotiok ranches).

4.2 EFFECTS OF VEGETATION COVER AND TOPOGRAPHIC POSITIONING ON SOIL ORGANIC CARBON A ND ORGANIC MATTER CONTENT

4.2.1 Soil Organic Carbon as Affected by Soil Vegetation Cover and Topographic Zones

The highest soil organic carbon (expressed in g/kg) in Ilmotiok site was in Tree cover fields with a mean of 43.32 g/kg and the lowest was in Grass covered fields with 7.6 g/kg SOC content in Hillslope and Plateau zones, respectively. The mean soil organic carbon content recorded for each vegetation cover type from the highest to the lowest ranged from; 43.32-8.36; 42.18-7.6; and 34.2-8.74 g/kg in Tree covered fields, Grass cover fields and Bare grounds, respectively (Figure 4.1). The mean soil organic carbon varied between topographic zones and vegetation cover types in the following order (RIP$^{>}$ HS$^{>}$ HW$^{>}$ PLA; RIP$^{>}$ HW$^{>}$ PLA$^{>}$ HS; RIP$^{>}$ HW$^{>}$ HS$^{>}$ PLA) in Bare grounds, Grass and Tree vegetation covers, respectively.

In Mpala ranch, organic carbon content was highest in RIP zones and lowest in HS and PLA zones in Bare grounds, Grass and Tree covered fields, respectively. Soil organic carbon ranged between 36.86-9.6; 38-8.36 and 50.54-9.12 g/kg in Bare grounds, Grass and Tree covered fields respectively (Figure 4.1). Tree covered fields had the highest organic carbon content of 50.54 g/kg, followed by Grass and Bare fields (38; 36.86 g/kg), respectively. The mean carbon concentrations recorded followed the order; (RIP$^{>}$ HW$^{>}$ PLA$^{>}$ HS); (RIP$^{>}$ HW$^{>}$ HS$^{>}$ PLA); RIP$^{>}$ HW$^{>}$ HS$^{>}$ PLA) in Bare grounds, Grass and Tree cover fields, respectively.

4.2.2 Soil Organic Carbon as Influenced by Soil Depth

Soil organic carbon content was higher in the top soil horizon (0-10, 10-20, 20-30 cm) compared to sub-soil horizon (30-40 and 40-50 cm depth), with the lowest organic carbon content observed at 40-50 cm soil horizon for all vegetation cover types. Tree covered fields had the highest organic carbon content of 43.32 g/kg at the top soil layer (0-10 cm), Grass covered fields with intermediate carbon of 42.18 g/kg while Bare grounds had the lowest organic carbon content of 34.2 g/kg in Hillslope and Riparian zones in Ilmotiok site, respectively. Soils in Mpala ranch followed a similar trend with Tree covered fields recording the highest organic carbon content at the top soil layer (0-10 cm) of 50.54 g/kg; Grass fields 38 g/kg and Bare grounds 36.86 g/kg in Riparian zones for all three vegetation cover types (Figure 4.1).

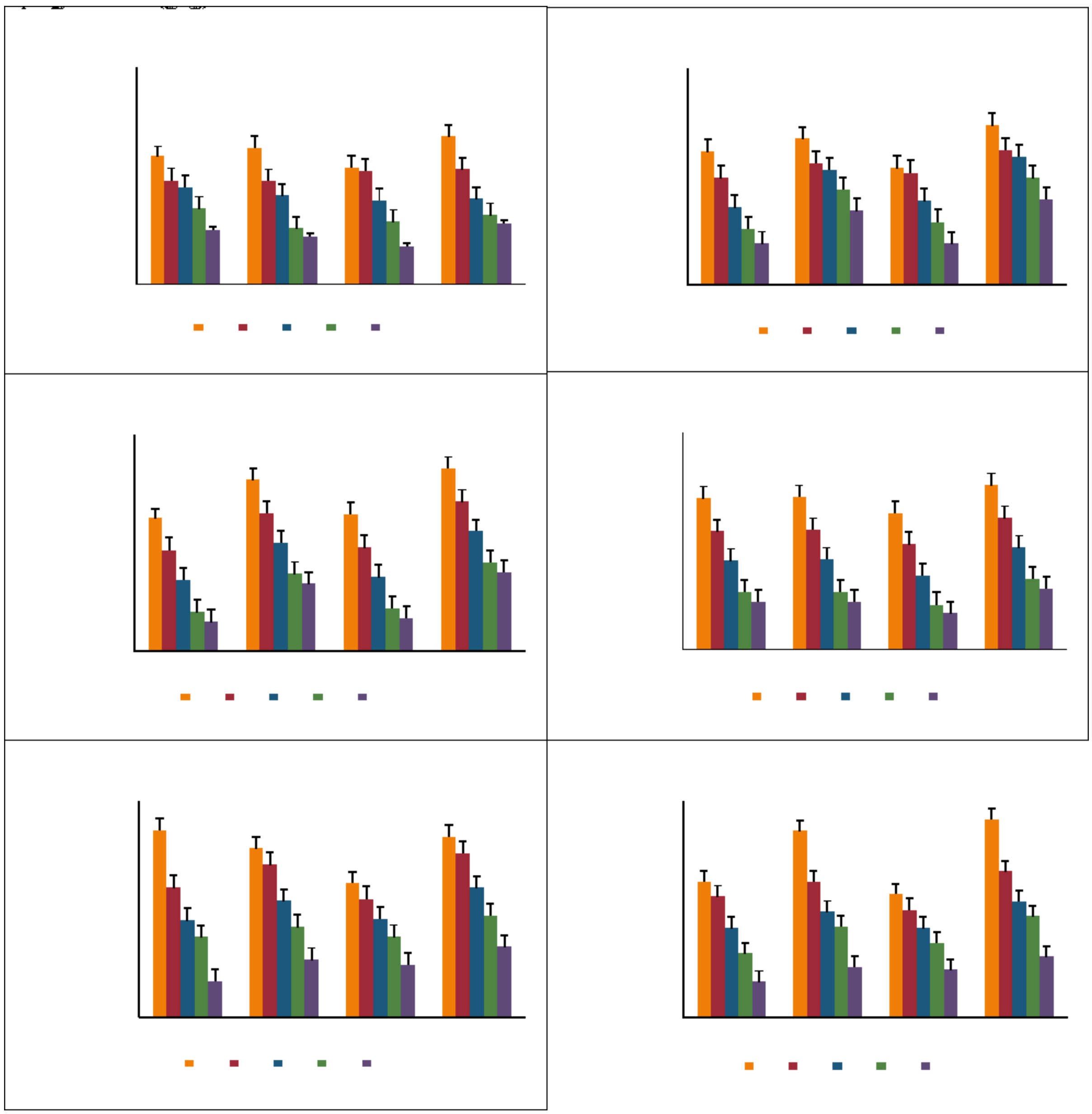

Figure 4.1: Soil organic carbon as influenced by vegetation covers, topographic positions and soil depth in Ilmotiok and Mpala sites, Laikipia County. MPA: Mpala ranch, ILM: Ilmotiok ranch

The highest mean organic carbon content in Ilmotiok site in the top soil layer (0-30 cm) was 36.61 g/kg and the lowest was 23.47 g/kg in Riparian and Hillslope zones, respectively. The mean organic carbon content in Mpala site in the top soil layer (0-30 cm) was highest at Riparian zones with 39.14 g/kg and lowest at Plateau zones with 24.32 g/kg SOC content. Further, the sub-soil layer (30-50 cm), registered a mean organic carbon content of 19.95; 7.85 g/kg and 22.23; 9.31 g/kg in Ilmotiok and Mpala sites, respectively. Soil organic carbon content differed significantly (P=0.0001) across the vegetation cover types, topographic zones and soil depth in both Mpala and Ilmotiok sites (Table 4.1). A multiple comparison test showed that Tree and Grass fields; Tree fields and Bare grounds; and Grass fields and Bare grounds were significantly different between each other in Ilmotiok site. The case was similar in Mpala site except for Grass fields and Bare grounds, which were not significantly different between each other. The RIP and HW zones; RIP and HS ; RIP and PLA; HW and HS; HW and PLA were significantly different ($P<0.05$) between each other, but HS and PLA zones were not significantly different between each other in both Mpala and Ilmotiok sites (Table 4.2 and 4.3). When soil organic carbon was regressed on soil organic matter, aggregate stability, infiltration rate and bulk density to explore which soil parameters influenced SOC, only soil organic matter significantly (P=0.0001) influenced soil organic carbon as shown in the regression model below.

SOC = -0.02.55 + 5.937 SOM (R^2 = 0.9998; P=0.0001)

Thus, besides SOC varying across vegetation types, it was also influenced by SOM, which also varied across vegetation types, but other parameters had no significant effect on it (Table 4.5).

Table 4.1: Anova table showing the significant differences for mean soil organic carbon, soil organic matter, aggregate stability and water infiltration rate between topographic zones, vegetation cover types and soil depth.

		Ilmotiok ranch			Mpala ranch		
Soil parameter	F value	TZ	VC	DEPTH	TZ	VC	DEPTH
Soil organic carbon	F Value	16.77	17.52	112.93	28	17.15	132.79
	Pr > F	*0.0001*	*0.0001*	*0.0001*	*0.0001*	*0.0001*	*0.0001*
Soil organic matter	F Value	*17.51*	17.77	115.60	22.62	17.13	133.12
	Pr > F	*0.0001*	0.0001	0.0001	0.0001	0.0031	0.0001
Aggregate stability	F Value	4.01	1.33	0.99	3.83	0.11	0.37
	Pr > F	*0.0124*	*0.2726*	*0.4227*	*0.0152*	*0.8998*	*0.8284*
Infiltration rate	F Value	*7.45*	*0.87*	*23.02*	*7.25*	*6.20*	*1.51*
	Pr > F	*0.0003*	*0.4260*	0.0001	*0.0004*	*0.0039*	*0.2147*

Legend: TZ-Topographical zones (Hillslope, Headwater, Riparian, and Plateau); VC-Vegetation cover types (Grass, Tree and Bare covered fields). Depth: 0-10, 10-20, 20-30, 30-40, and 40-50 cm.

Table 4.2: Multiple comparison tests of soil properties between vegetation covers in Mpala and Ilmotiok sites

Soil Parameter	Differences between means		
	Vegetation type comparison	Ilmotiok Ranch	Mpala Ranch
Soil organic carbon	Tree-Grass	0.0930*	0.1330*
	Tree-Bare	0.1465*	0.0905*
	Grass-Bare	0.0535*	0.0425ns
Soil organic matter	Tree-Grass	0.1590*	0.0905*
	Tree-Bare	0.2520*	0.1330*
	Grass-Bare	0.0930*	0.0425^{ns}
Aggregate stability	Tree-Grass	ns	Ns
	Tree-Bare	ns	Ns
	Grass-Bare	ns	Ns
Infiltration rate	Tree-Grass	ns	-0.039^{ns}
	Tree-Bare	ns	-0.468*
	Grass-Bare	ns	0.429*

*Significant differences at 0.05 level are indicated by *; ns means not significant*

4.2.3 Soil Organic Matter as Affected by Vegetation Cover and Topographic Positioning

The mean SOM content was highest (7.54%) in Hillslope (HS) zones and lowest in Grass covered areas (1.17%) in Ilmotiok site. In Mpala, SOM content was highest in Tree covered fields with 8.79%, and lowest in Grass covered fields with 1.45%, in RIP and PLA zones, respectively. SOM decreased with depth, in both sites (Figure 4.2). Soil organic matter content followed the order: (RIP> HS >HW> PLA; RIP >HW> PLA> HS; RIP> HW> HS >PLA) in Bare grounds, Grass and Tree covered fields, respectively in Ilmotiok site. A similar trend was observed for Grass cover types in Mpala ranch while Bare and Tree covered fields had the following trend; RIP >HW> PLA> HS and RIP> HW> PLA> HS, respectively.

Table 4.3: Multiple comparisons of soil properties between different topographic zones in Mpala and Ilmotiok sites

	Differences between means Ilmotiok Ranch					
	Topographic zones					
Soil Parameter	RIP-HW	RIP-HS	RIP-PLA	HW-HS	HW-PLA	HS-PLA
SOC	0.07267*	0.1553*	0.1853*	0.08267*	0.11267*	0.0300^{ns}
SOM	0.12267*	0.2693*	0.32267*	0.14667*	0.2*	0.0533^{ns}
AS	0.4467^{ns}	0.800*	0.3600^{ns}	0.7133*	0.4467^{ns}	1.160*
IR	0.334^{ns}	1.3667*	0.4633^{ns}	1.7007*	0.7973*	0.9033*
	Differences between means Mpala Ranch					
SOC	0.074*	0.18733*	0.21533*	0.1133*	0.1413*	0.028^{ns}
SOM	0.122*	0.3213*	0.366*	0.1993*	0.244*	0.0446^{ns}
AS	0.0353^{ns}	0.1777^{ns}	0.5607*	0.142^{ns}	0.596*	0.738*
IR	0.7476*	0.2513^{ns}	0.1633^{ns}	0.7273*	0.6393*	0.688^{ns}

*Legend; SOC- soil organic carbon; SOM: soil organic matter; AS: soil aggregate stability; IR: infiltration rate. RIP-Riparian; PLA-Plateau; HS-Hillslope; HW-Headwater zones. Significant differences at 0.05 level are indicated by *; ns-not significant*

4.2.4 Soil Organic Matter as Influenced by Soil Depth

Soil organic matter decreased with depth in both Ilmotiok and Mpala, with the highest organic matter registered in the top soil (0-10 cm), and the lowest in 40-50 cm soil depth (Figure 4.2). Tree covered fields recorded the highest organic matter content at the top horizon of 7.54%, Grass covered fields had intermediate levels of 7.34% while Bare grounds had the lowest organic matter of 5.95%, in Hillslope and Riparian zones in Ilmotiok site, respectively. A similar trend was observed in Mpala ranch, with Tree fields recording the highest SOM content of 8.79%, Grass fields had 6.61% and Bare grounds 6.41% in Riparian zones, respectively. The sub-soil horizon had lower levels of organic matter content compared to the top soil horizon (Figure 4.2), but similar trends of Tree covered fields with the highest organic matter were observed in Mpala and Ilmotiok sites. Tree covered fields had 3.47% organic matter, Grass fields 3.37% and Bare grounds 3.15% in Riparian zones, respectively in Ilmotiok. In Mpala, Bare grounds had 3.60% SOM content, Grass covers 2.64% and Tree covers 1.95% in Riparian zones, respectively.

Soil organic matter (SOM) differed significantly at ($P=0.0001$) between the vegetation cover types, topographic zones and soil depth for both Ilmotiok and Mpala sites (Table 4.1). A multiple comparison test between each of the vegetation cover types showed that Tree and Grass fields; Tree and Bare grounds were significantly different ($P<0.05$) between each other but Grass and Bare ground vegetation covers were not significantly different. Likewise, a multiple comparison test was done between each of the topographic zones, and it was observed that, RIP-HW; RIP-HS; RIP-PLA; HW-HS; HW-PLA were significantly different ($P<0.05$) between each other, however, HS-PLA zones were not significantly different. When soil organic matter was regressed on soil organic carbon, aggregate stability, infiltration rate and bulk density to explore which soil parameters influenced SOC, only soil organic matter significantly ($P=0.0001$) influenced soil organic carbon as shown in the regression model below.

$$\mathbf{SOM = 0.0508 - 0.0518\ PH + 0.00752\ BD + 1.659\ SOC}\ (\boldsymbol{R^2=0.9998;\ P=0.0001})$$

Thus, besides SOM varying across vegetation types, it was also influenced by SOC, soil pH, and bulk density which also varied across vegetation types.

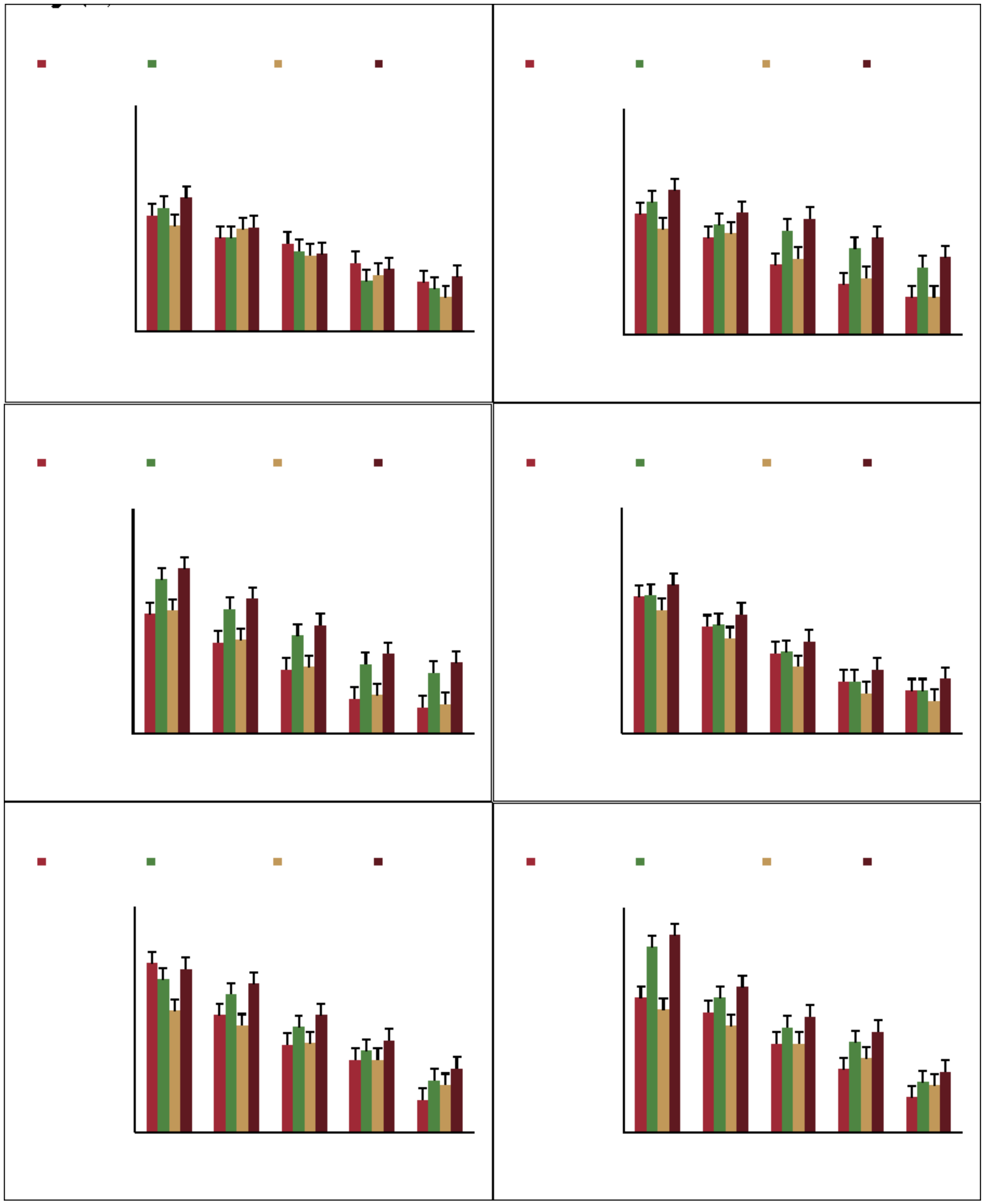

Figure 4.2: Soil organic matter as influenced by topographic positions, vegetation covers and soil depth in Ilmotiok and Mpala sites, Laikipia County.

4.2.5 Soil Carbon Stocks as Influenced by Vegetation Cover, Topographic Positions and Soil Depth

Soil organic carbon stocks (SOCs) varied significantly and decreased with increasing soil depth in the three vegetation cover types in both Mpala and Ilmotiok sites (Figure 4.3). The highest SOCs occurred at the top 0-10 cm. Grass and tree fields recorded the highest SOCs at 11.4 t C/ha and 12 t C/ha in Ilmotiok and Mpala ranches, respectively. The lowest SOCs were observed at 40-50 cm along the soil profile for both sites, with Grass fields recording 1.8 t C/ha in Ilmotiok ranch. The decreasing carbon stock with increasing soil profile depth was constant except in Bare grounds at the 20-30 cm depth in Ilmotiok site, and the 10-20 cm depth in Bare grounds and Grass fields in Mpala site.

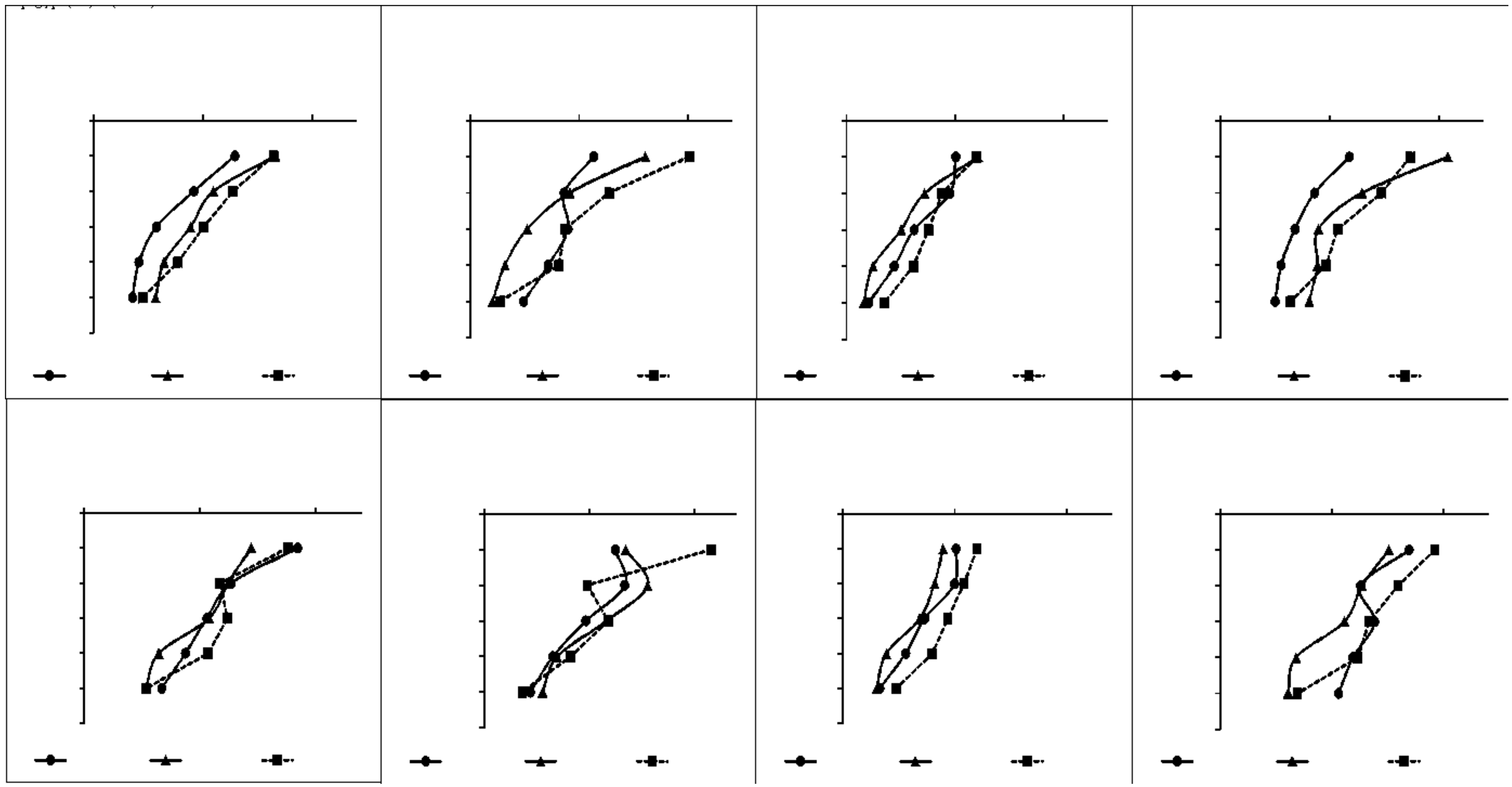

Figure 4.3: Soil carbon stocks as affected by soil vegetation covers, topographic positions and soil depth in Mpala and Ilmotiok sites, Laikipia County. ILM: Ilmotiok ranch; MPA: Mpala ranch

4.2.6 Soil Bulk Density and Texture of Top-Soil in Relation to Vegetation Covers and Topographic Positions

Soil bulk density and texture were determined for the top 0-20cm. Soil bulk density (BD) of soils in Ilmotiok site was highest in Grass cover fields with 1.02 $g.cm^{-3}$ and lowest in Bare grounds with 0.76 $g.cm^{-3}$ in Hillslope and Riparian zones respectively. In Mpala, the highest BD was in Bare grounds with 1.07 $g.cm^{-3}$ and the lowest in Grass covered fields with 0.73 $g.cm^{-3}$ in Headwater and Plateau zones respectively (Table 4.4).

Grass covered fields in Ilmotiok site, had sandy clay loam soils across the topographic zones, while Bare grounds and Tree covered fields had sandy loam soils in the top 0-20 cm depth. In Mpala, Headwater and Plateau zones had similar textural classes across the three vegetation covers (sandy loam), while in Riparian zones, Grass and Tree cover fields had clay soils, and Bare grounds sandy loams. In Hillslope zones, Bare grounds had sandy clay loam soils and Grass and Tree covered fields had sandy loam soils, respectively (Table 4.4).

Riparian zones with sandy loam texture had higher aggregate stability in Ilmotiok zones while Mpala Riparian zones with clay texture had the lowest aggregate stability. Hillslope zones in Ilmotiok had higher soil aggregate stability and Riparian zones had the lowest in Mpala. Water infiltration rate was generally low in Mpala compared to Ilmotiok (Figure 4.4).

Table 4.4: **Soil bulk density and textural classes of the surface horizon**

Site	Topographic zone	Vegetation Cover	BD $g.cm^{-3}$	Sand	Clay	Silt	Textural classes (0-20cm depth)
Ilmotiok Ranch	Hillslope	Bare	0.85	82	10	8	Loamy sand
		Grass	1.02	59	28	13	Sandy clay loam
		Tree	0.95	73	17	10	Sandy loam
	Headwater	Bare	0.89	72	21	8	Sandy clay loam
		Grass	0.83	69	23	8	Sandy clay loam
		Tree	0.85	70	18	12	Sandy loam
	Riparian	Bare	0.76	79	16	5	Sandy loam
		Grass	0.92	63	28	10	Sandy clay loam
		Tree	0.86	80	14	6	Sandy loam
	Plateau	Bare	0.85	80	10	8	Sandy loam
		Grass	0.79	69	23	8	Sandy clay loam
		Tree	0.80	69	19	13	Sandy loam
Mpala Ranch	Hillslope	Bare	1.05	69	20	11	Sandy clay loam
		Grass	1.03	81	12	6	Sandy loam
		Tree	1.01	79	12	10	Sandy loam
	Headwater	Bare	1.07	78	11	11	Sandy loam

		Grass	0.94	77	13	10	Sandy loam
		Tree	0.77	69	14	17	Sandy loam
	Riparian	Bare	0.94	76	16	8	Sandy loam
		Grass	0.89	13	76	11	Clay
		Tree	0.86	38	43	18	Clay
	Plateau	Bare	0.87	80	10	8	Sandy loam
		Grass	0.73	67	14	17	Sandy loam
		Tree	0.87	79	12	10	Sandy loam

Table 4.5: Effects of vegetation cover types and topographic positions on soil parameters (multiple regression analysis) in Ilmotiok and Mpala sites

Variables	**Df**	**Regression Model**[†]	**P value**	**R^2**
Bulk density	6	BD = -5.7785 + 5.901 PH + 13.31SOM	0.0001	0.9895
Soil organic carbon	6	SOC = -0.02.55 + 5.937 SOM	0.0001	0.9998
Soil organic matter	6	SOM= 0.0508 - 0.0518 PH+ 0.00752 BD + 1.659 SOC	0.0001	0.9998

SOM- Soil organic matter; SOC – Soil organic carbon; and BD -Bulk density

Soil bulk density varied significantly across the topographic zones and soil depth (P=0.0001) in Mpala and Ilmotiok sites. However no significant differences were observed across the vegetation covers P=0.2472 and P=0.01816 in Ilmotiok and Mpala, respectively. In Ilmotiok site, a multiple comparison test showed that RIP and HS; RIP and PLA; HW and HS; HW and PLA; HS and PLA zones varied significantly ($P<0.05$) between each other, except RIP and HW zones which did not differ significantly between each other. In Mpala, RIP and HS; HW and HS; HW and PLA; HS and PLA differed significantly ($P=0.05$) between each other, but RIP and HW; RIP and PLA zones were not significantly different between each other.

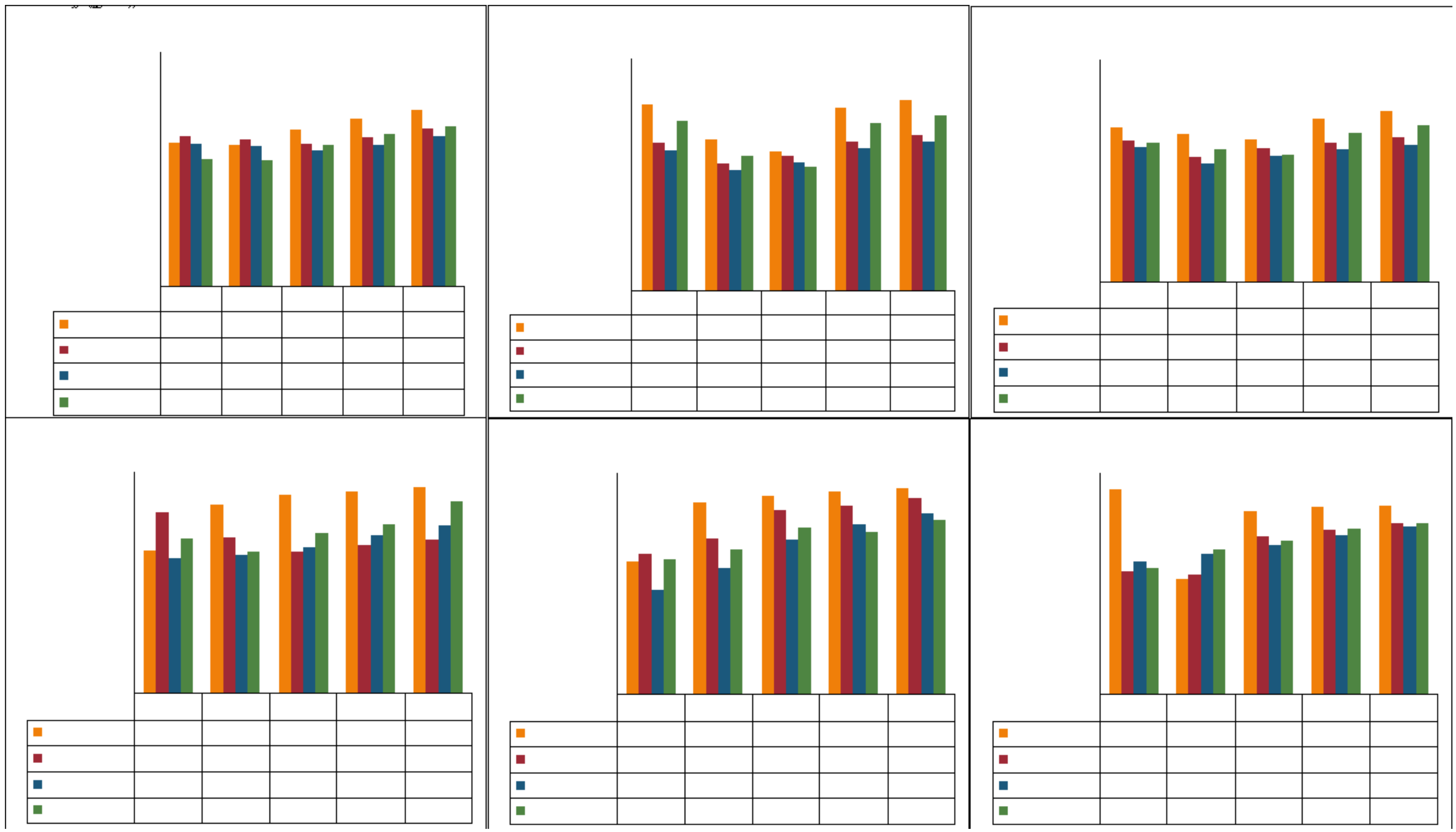

Figure 4.4: Soil bulk density as affected by vegetation covers and topographic positioning in Mpala and Ilmotiok, Laikipia County. MPA: Mpala ranch, ILM: Ilmotiok ranch

4.3 EFFECTS OF VEGETATION COVER AND TOPOGRAPHIC POSITIONING ON SOIL AGGREGATE STABILITY

4.3.1 Soil Aggregate Stability as Influenced by Vegetation Cover and Topographic Positions

Soil aggregate stability (AS) in Ilmotiok site ranged from 0.4-4.1%, 0.5-3.7% and 1.2-3.6% in Bare grounds, Grass and Tree vegetation covers, respectively, across the topographic zones. The aggregate stability in Mpala site ranged between 0.1-3.7-%, 0.9-2.6% and 0.8-2.8% in Bare grounds, Grass and Tree covered fields across the topographic zones (Figure 4.5). Riparian zones showed high aggregate stability while HS and HW zones recorded low AS across the vegetation cover types, in Ilmotiok site. The trend was different for Mpala site with HS zones recording the highest aggregate stability and RIP zones the lowest in Bare grounds, Grass and Tree covered fields, respectively (Figure 4.5**).** Soil aggregate stability varied significantly between topographic zones (P=0.0124). However, there was no significant difference in variation of aggregate stability among the vegetation cover types (P=0.2726) and soil depth (P=0.4227) in Ilmotiok site Similarly, Mpala site showed a significant difference in aggregate stability among the topographic zones (P=0.0152). However, no significant difference was observed in variation of aggregate stability among the vegetation cover types and soil depth (P=0.8998; P=0.8284) respectively.

4.3.2 Soil Aggregate Stability as Influenced by Soil Depth

Soil aggregate stability fluctuated with soil depth, with major fluctuations observed between 10-20 cm, 20-30 cm and 30-40 cm in both Mpala and Ilmotiok sites (Figure 4.5). In Ilmotiok, the AS in the top 0-10 cm horizon ranged from 1.4-2.7% in Bare grounds; 0.8-2.7% in Grass fields and 1.2-2.5% in Tree covered fields. Further, Riparian and Headwater zones had the highest aggregate stability of 2.7% in Bare grounds and Grass fields while Plateau zones had the lowest AS of 0.8%, respectively. In the sub-soil horizon, aggregate stability ranged from 1.2-3.2%; 0.5-2.2%; 1.8-3.4% in Bare grounds, Grass and Tree cover fields respectively with Plateau zones recording the highest soil aggregate stability while Headwater zones had the lowest in the sub-soil horizon. In Mpala, the soil aggregate stability of the top soil layer ranged between 0.8-1.6%; 1.9-2.1%; 1.9-2.8% in Bare grounds, Grass and Tree covered fields, respectively, with the highest aggregate stability registered in Headwater zones and the lowest in Plateau zones. The aggregate stability in the sub-soil horizon (10-20 cm) ranged between 0.1-2.3%; 1.4-2.6%; 0.8-2.1% in Bare grounds, Grass and Tree cover fields, respectively, where the highest and lowest aggregate stability for the sub-soil horizon was recorded in Headwater zones (Figure 4.5).

A multiple comparison test on soil aggregate stability showed that the vegetation cover types were not significantly different between each other in both Ilmotiok and Mpala. Riparian and Hillslope zones; Headwater and Hillslope zones differed significantly ($P<0.05$) between each other. However, no significant differences were observed between RIP and HW; RIP and PLA, HW and PLA and HS and PLA zones in Ilmotiok zones, respectively. In Mpala, RIP and HS; HW and PLA; HS and PLA differed significantly ($P<0.05$) between each other. However, the RIP and HW; RIP and HS; and HW and HS were not significantly different between each other (Table 4.2 and 4.3).

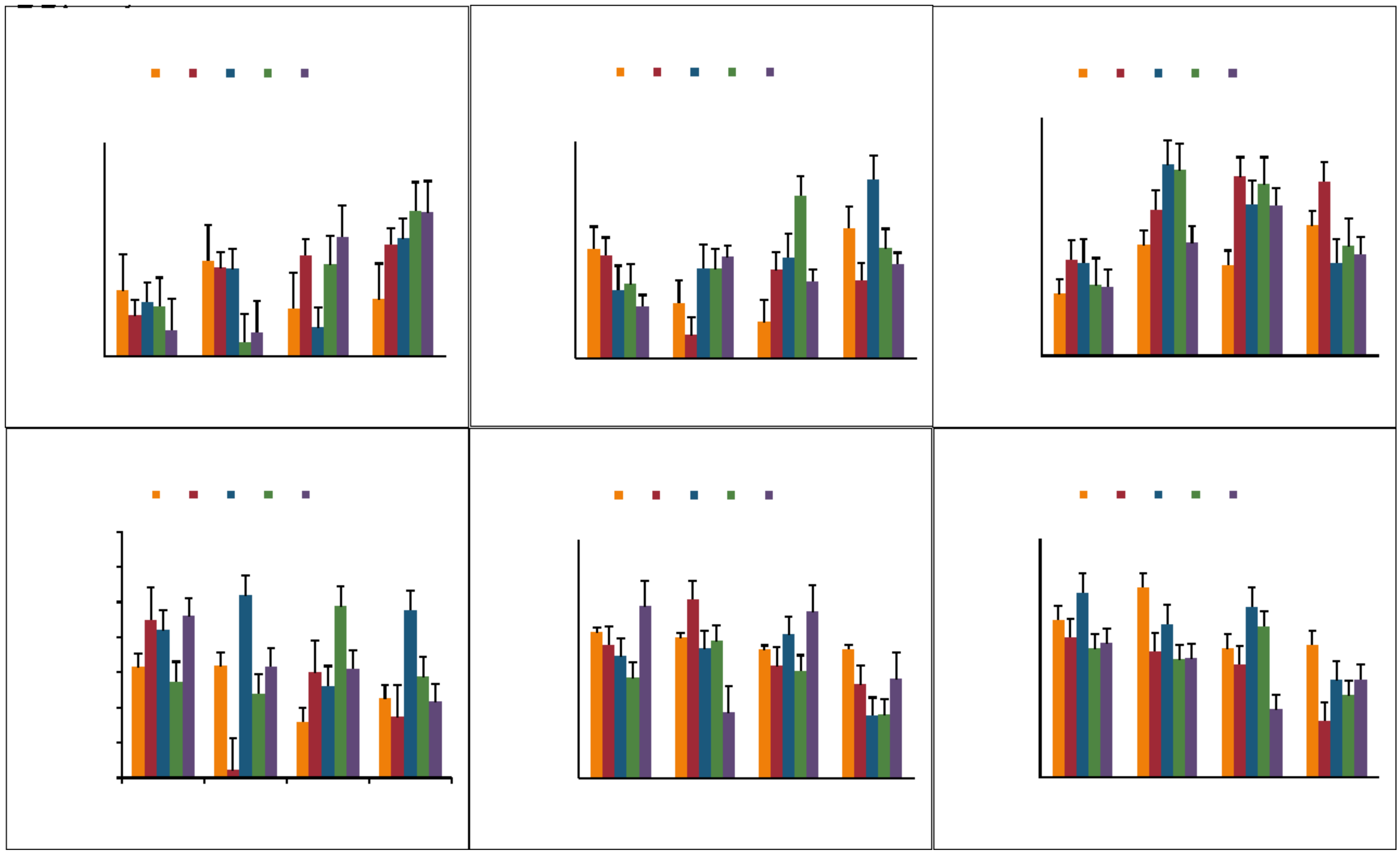

Figure 4.5: **Soil aggregate stability as affected by soil vegetation covers in Mpala and Ilmotiok sites, Laikipia County. MPA: Mpala ranch, ILM: Ilmotiok ranch**

4.4 Effects Of Vegetation Cover And Topographic Positioning On Water Infiltration Rate

The highest infiltration rate was recorded in the Tree covered fields (73.3 mm/hr) and decreased in Grass fields (25 mm/hr) and Bare grounds with 17 mm/hr, respectively in Ilmotiok site. The highest infiltration rates were observed in HW zones (73.3 mm/hr) and the lowest infiltration rates were recorded in HS zones (0 mm/hr) in Tree covered fields (Figure 4.6).

In Mpala, the infiltration rates ranged between 0.9-37.8 mm/hr in Bare grounds; 0.0-8.9mm/hr in Grass fields and 0.7-5.7 mm/hr in Tree fields, respectively. Headwater zones had the highest infiltration rates of 37.8 mm/hr in Bare grounds while the lowest infiltration rates were observed in Riparian zones with 0.00 mm/hr in Grass fields (Figure 4.6).

Cumulative infiltration in Ilmotiok site ranged from 26.7-128.7 mm/hr; 7.9-122.7 mm/hr and 6.9-106.2 mm/hr in Tree fields, Bare grounds and Grass fields, respectively. The highest (128.7 mm/hr) and lowest (6.9 m/hr) cumulative infiltration occurred in HW zones after 2.5 hours. In Mpala, Bare grounds recorded a cumulative infiltration of 1.5-95.5 mm/hr; Grass fields 2.2-23.7mm/hr and Tree fields 2.2-21.3 mm/hr. Headwater zones had the highest cumulative infiltration (95.5 mm/hr) and the lowest was in Riparian zones (1.5 mm/hr) in Mpala (Figure 4.6).

Water infiltration rate (IR) of soils differed significantly among the topographic zones (P=0.0003). However, infiltration rate was not significantly different among the vegetation covers (P=0.4260) in Ilmotiok site. In Mpala, there was a significant difference in variation of infiltration rate among the topographic zones (P=0.0004) and vegetation cover types (P=0.0039). A multiple comparison test showed that RIP and HS zones; HW and HS; HW and PLA; and HS and PLA were significantly different between each other in Ilmotiok site. However, RIP and HW; and RIP and PLA zones were not significantly different between each other.

In Mpala, the multiple comparison test indicated that Tree fields and Bare grounds; and Grass fields and Bare grounds differed significantly ($P<0.05$) between each other, but Tree and Grass fields did not differ significantly between each other. The topographic zones; RIP and HW; HW and HS; and HW and PLA zones differed significantly ($P<0.05$) between each other. However, RIP and HS; RIP and PLA; HS and PLA were not significantly different between each other (Table 4.2 and 4.3).

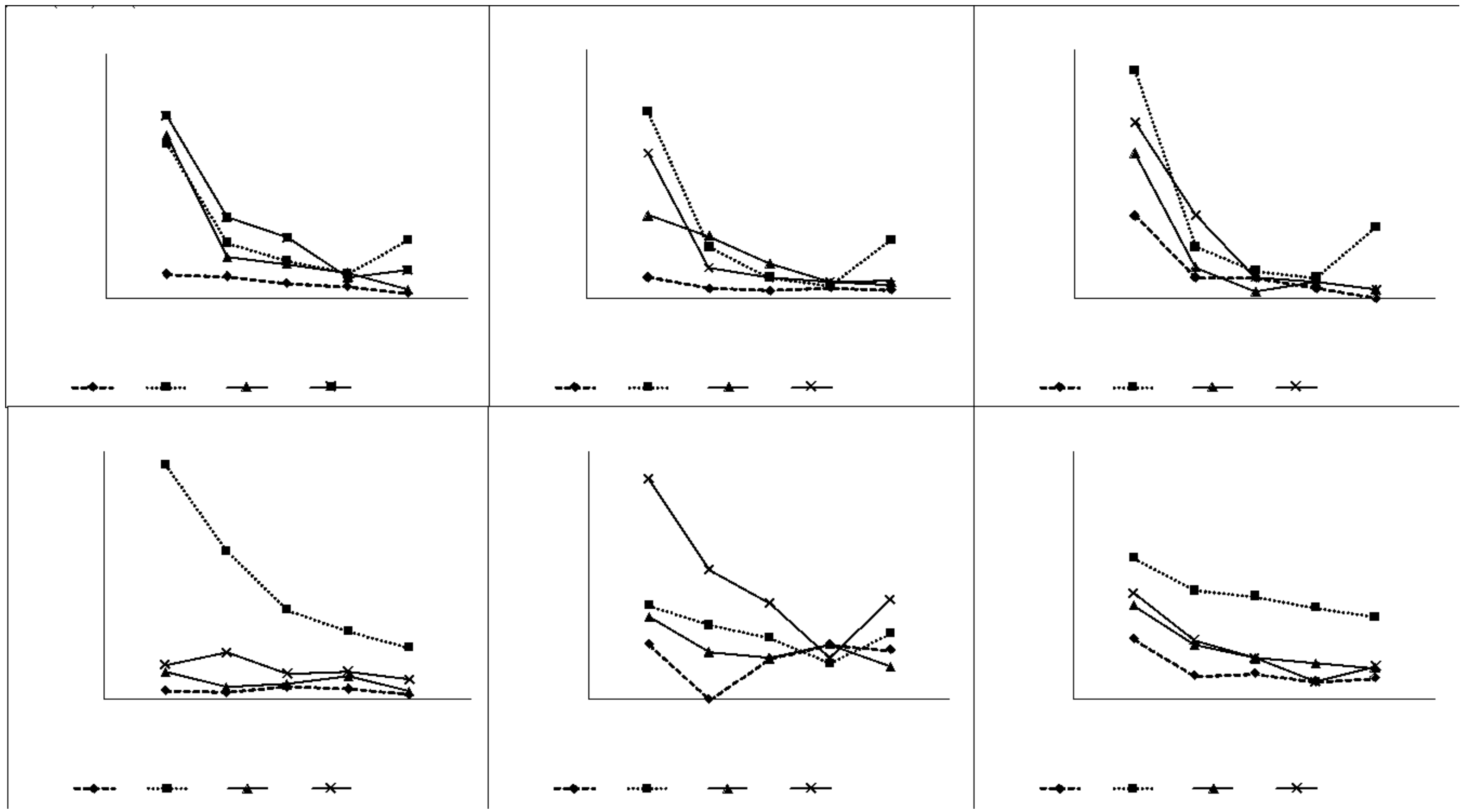

Figure 4.6: Water infiltration rate as affected by vegetation cover types and topography in Ilmotiok and Mpala sites, Laikipia County. MPA: Mpala ranch, ILM: Ilmotiok ranch

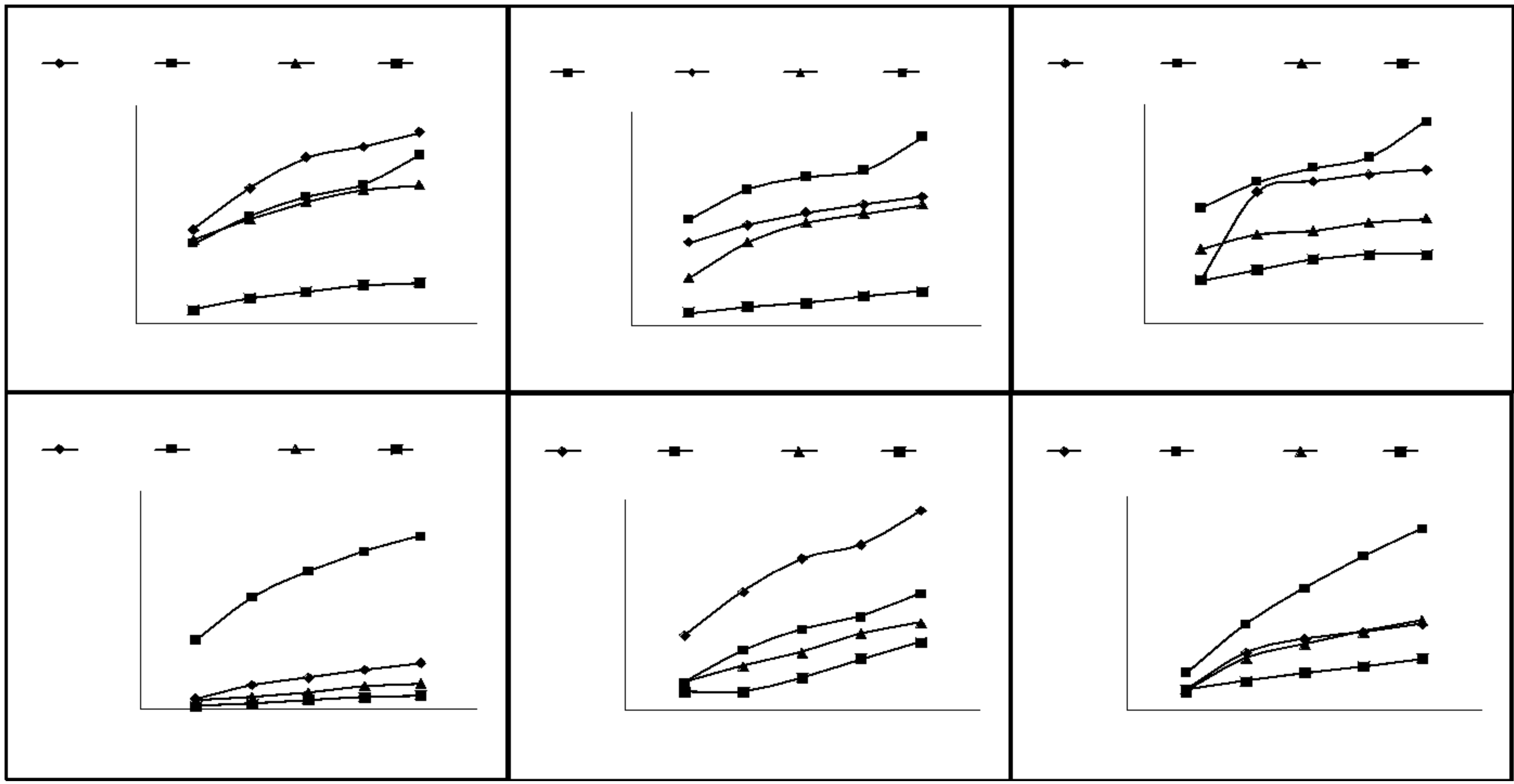

Figure 4.7: Cumulative infiltration as affected by vegetation cover types and topographic zones in Ilmotiok and Mpala sites, Laikipia county. MPA: Mpala ranch, ILM: Ilmotiok ranch

4.5 Effects Of Grazing Management On Soil Properties In Ilmotiok And Mpala Ranches

The results of a t-test analysis (unpaired two sample assuming unequal variances) showed that SOC and SOM content, were not significantly different between the two sites in the three vegetation covers (Bare ground, Grass cover and Tree cover), respectively. In soil bulk density test, all the three vegetation covers were significantly different between the two sites (P=0.01; P=0.008; P=0.0002) in Tree fields, Grass fields and Bare grounds, respectively. Soil organic carbon stocks were significantly different between the two sites (P=0.01) in Bare grounds only. Soil aggregate stability was significantly different between the two sites (P=0.03; P=0.05) in Tree fields and Bare grounds respectively. Water infiltration rates were significantly different between the two sites (P=0.004; P=0.006; P=0.02) in Tree fields, Grass fields and Bare grounds, respectively.

CHAPTER 5: DISCUSSION

5.1 Influence of Vegetation Cover on Soil Organic Carbon and Organic Matter Content in Grassland Systems of Laikipia County

The study findings revealed that soil organic carbon differed significantly (P=0001) among the vegetation cover types, topographic zones and soil depth in both Mpala and Ilmotiok sites. The findings agree with studies by Dunkerley, (2002) and Ludwig *et al*., (2005) who found that vegetation cover controls the physical and chemical properties of soil, which in turn determine the plant composition in semi-arid landscapes. In the present study, Grass and Tree covered fields stored more soil organic carbon than Bare grounds in both Mpala and Ilmotiok sites because they (Grass and Tree fields) generated more plant litter than Bare grounds, and most of it was returned to the soil through litter fall.

Further, plant residues provide a good carbon source for microbial processes which are conducive to carbon fixation from the atmosphere, thereby gradually increasing SOC content (Yang *et al*., 2011; Schulze *et al*., 2009). The soil surface of Bare grounds had long been exposed hence their soil organic matter content was low compared to Tree and Grass cover types (Zhao *et al*., 2010). In addition, Grass and Tree fields provide better microclimate due to the adequate herbaceous and woody vegetation cover which reduces soil temperatures and evapotranspiration rates in the soil. Areas with undisturbed soils are characterized by a reduced oxidation of the sub-soils which creates a favorable environment for the microbial organisms and facilitates the decomposition of soil organic matter (Theirfelder & Wall, 2012; Fontaine *et al*., 2007).

In addition, the low SOC and SOM contents in Bare grounds can be explained by continuous heavy grazing which depletes most of the soil organic carbon pools and eventually increases soil erosion processes (Ritchie *et al.*, (2012). According to Derner *et al.*, (2006) intensive grazing activities on the land decrease above ground litter deposition and below ground carbon allocation hence the low carbon contents in Ilmotiok site. Sanjari *et al.*, (2008) also noted that an area under controlled grazing contributed to higher rates of vegetation growth (grass) which leads to an increase in litter accumulation and a substantial increase of 1.37 ton/ha carbon in the top 10 cm compared to a continuously grazed site.

The Riparian (RIP) zones had higher SOC and SOM contents compared to the other topographic zones which is attributable to the soil textural characteristics (fine silt and clay particles) in the riparian zones. Similarly, Yang *et al*., (2008) explains that high clay and silt content significantly increases SOC density in the soil. Studies in the past agree that fine particles tend to stabilize and retain more organic matter and have higher water retention capacity than coarser soil particles (Gregorich *et al*., 1994; Gomez-Plaza *et al*., 2001).

5.2 Effects of Topographic Positioning on Soil Carbon and Organic Matter

The results of Wiaux *et al*., (2014) indicated that organic carbon storage on the foot slope were 2.5 times higher than other slope positions along an eroding hillslope situated on cropland in Belgium. Hancock *et al*., (2010) found similar results in an undisturbed environment in Australia. These authors found that higher SOC content is likely to relate to a higher aboveground biomass on the lower slope positions. This explains the high SOC and SOM contents in Riparian zones than Hillslope zones in both Mpala and Ilmotiok site.

SOC and SOM content along an eroding hill slope (HS and HW zones) are lower compared to the depositional areas because soil thickness is reduced due to removal of soil surface materials which are deposited at the bottom of the slope (Berhe *et al*., 2008). Again, at foot slope positions (RIP zones), SOC accumulation and burial results in greater SOC content and protection from further decomposition (Fontaine *et al*., 2007; Berhe & Kleber, 2013; Doetterl *et al*., 2016). On the contrary, other researchers have also hypothesized that soil erosion leads to exposure of protected SOC to decomposition hence high SOM levels in some scenarios (Fierer *et al*., 2003).

5.3 Soil Organic Carbon and Organic Matter Content as Influenced by Soil Depth

The top 0-30 cm layer of soil had a higher concentration of SOC and SOM in both sites (Figure 4.1 and 4.2) explained by the active mineralization and immobilization processes of carbon. Similarly, Batjes, (1996) found that 50 % of SOC content appeared in the 0-30 cm soil layer. In addition, Jobbágy & Jackson, (2000) and Civeira *et al*., (2012) found similar results in forest soils. This is explained by the large amounts of CO_2 that are released when tropical soils are deforested, changes in land use, or increased oxidation of superficial peat layers on drainage.

Soil organic carbon concentration and SOM decreased with soil depth in all the topographic zones and vegetation cover types in the study areas. This has been explained by Jackson *et al.,* (1996) who reported that the distribution of roots depending on the vegetation cover type (grass, tree or bare ground), can affect the vertical placement of carbon in the soil. Again, plant litter fall from Grass and Tree covered areas contributes to higher SOM contents on the top soil layer which decreases along the soil profile due to decomposition processes, or through soil surface heating as a result of higher temperatures Jobbágy & Jackson (2000).

5.4 Effects of Bulk Density on Soil Organic Carbon and Organic Matter Content

The low bulk densities in Mpala ranch were conditioned by high SOC and SOM contents compared to Ilmotiok ranch. This effect is attributable to controlled grazing and minimum soil disturbance in the private ranch (Tuffour *et al.*, 2014). Higher bulk densities in Ilmotiok ranch can be linked to low SOC and SOM contents as well as soil compaction due to heavy animal trampling (Wolf, 2011). Li *et al.*, (2008) and Igwe, (2005) agree that reducing soil bulk densities to solve soil compaction problems is highly dependent on reduced soil disturbances and increasing organic matter content. Several researchers have also found a strong positive correlation between organic matter and bulk density of soils (Sakin *et al*., 2011).

5.5 Soil Carbon Stocks as Influenced by Vegetation Cover and Topographic Positioning

The high SOCs reported in Tree covered fields compared to Grass fields and Bare grounds in both sites may be attributed to higher litter input and soil moisture found under the canopies of the tree vegetation cover types (Figure 4.3). Further, microbial action on the litter fall is increased due to the micro-climate created by the vegetation cover hence high organic matter content, which subsequently improves soil aggregate stability (Sainepo *et al*., 2018).

Low carbon stocks observed in Ilmotiok ranch may be linked to the subsequent disruption of carbon inputs and excessive harvesting of the above ground biomass by livestock which has been reported to alter the carbon cycle within the ecosystem (Sainepo *et al.,* 2018). Excessive removal of herbaceous material has led to exposure of the surface to harsh temperature and surface runoff, which further aggravates the situation. According to Murty (2002), this exposure hastens the litter turnover rates, leading to SOM oxidation, expediting CO_2 release into the atmosphere. On the other hand, low organic carbon inputs in the soil and soil compaction as a result of high livestock movement and trampling actions, and soil surface removal could contribute to low SOCs in the study area (Muya, 2011). Likewise, the area has low and variable precipitation, with high solar radiation (Jaetzold, 2010) which discourages SOC build up.

Studies by He *et al.,* (2008) and Yang *et al.,* (2011) have further shown that severely grazed areas experience a significant loss of aboveground biomass which reduces the amount of organic matter and other essential nutrients in the soil. Similar results have been reported by Fernandez *et al.,* (2008). Uncontrolled grazing activities have been found to reduce the source of SOC from above and below ground inputs because of increased CO_2 fluxes from the atmosphere (Frank *et al.,* 2002).

Soil carbon stocks decreased with increasing soil profile depth in both Mpala and Ilmotiok sites. This could be explained by decreasing OM content and bulk density with increasing soil profile depth as a result of low clay concentrations along the soil profile (Tsui *et al.,* 2013; Minasny *et al.,* 2006; Li *et al.,* 2010).

Lower carbon stocks were observed at Plateau zones compared to Riparian zones in both sites (Figure 4.3). This could be attributable to the positioning of the Plateau zones on the gentler slopes while the Riparian zones were found at the foot slope, where the rich organic materials of the soil are transported from the steep slopes because of erosion and runoff processes through the gentler slopes and deposited on the foot slopes (Xu & Wan, 2008). Other studies have found low levels of SOC stocks in the higher slope position as opposed to the foot slope positions (Muya *et al.,* 2011).

Several researchers agree that topographic positions influence runoff processes and drainage, soil temperature, and soil erosion and consequently soil properties (Fissore *et al*., 2017) which in turn, affect plant production and OM decomposition. Moreover, topographic positions are also considered to affect the spatial variability of soil properties along the slope (Hancock *et al*., 2010).

5.6 Carbon Sequestration in Mpala and Ilmotiok Ranches

Soil carbon content varied significantly (P=0.0001) among the topographic zones, vegetation covers and soil depths in the two sites. The two sites showed a similarity in carbon content decline with depth, with the highest carbon levels noted in the top and sub-soil horizons. The deeper the profile, the lower the carbon content obtained. Higher carbon contents were observed in Tree and Grass covered fields compared to Bare grounds in both sites (Figure 4.1). This could be attributable to the higher litter fall in Tree covered areas, which triggered organic matter decomposition in the soil. Bare grounds were highly affected as more trampling by livestock occurred, making the soil compact, and affecting soil mechanisms that support organic matter decomposition. The topography of the study area affected the carbon content storage in the soil. Higher soil carbon was deposited in Riparian zones than the other topographic zones (Plateau, Headwater and Hillslope) (Figure 4.1). Hillslope areas with the most soil degradation processes taking place, had lower carbon levels while Headwater and Plateau zones followed the same trend. However, soil re-vegetation measures are important in the restoration of bare lands and in controlling the movement and speed of soil materials from the higher sloping areas for carbon sequestration process to take place in the two sites. Further, Mpala being a protected ranch, showed a higher recovery potential to soil carbon losses and gradual carbon sequestration because of grazing management compared to Ilmotiok group ranch.

5.7 Soil Aggregate Stability as Influenced by Vegetation Cover, Topographic Positioning and Soil Depth

The results of the aggregate stability determination showed that aggregation processes lead to more stable soil aggregates on vegetated land than on Bare grounds, which agrees with the findings of Imeson & Verstraten, (1989). Again, vegetation cover through litter fall contributes to organic matter input in the soil which is an influencing factor of soil aggregate stability. This is due to contributions of organic matter from the litter. Also, vegetation enhances rainwater infiltration and offers more favorable micro-climate beneath plants due to shadow. These conditions should generate a more active fauna and flora and as a consequence, improvement of soil aggregates (Zhang, 1997). Grass vegetation cover type is associated with high aggregate stability because of its dense root system which results to high microbial biomass and carbohydrate production (Gale *et al*., 2000). Aggregation is achieved through excretion of organic substances that act as binding glue in grass covered areas.

Several studies have shown positive associations of aggregate stability with plant cover (Rillig *et al*., 2002; Rezaei *et al*., 2006). However, others studies have had negative results on the relationship between water stable aggregates and plant or vegetation cover (O'Dea, 2007; Piotrowski *et al*., 2004).

In Vertisols, drying grassroots add carbon input and thus stimulate microbial biomass (Tisdall & Oades, 1982). Chevallier *et al*., (2001) reported that the restoration of soil carbon stock in a Vertisol under pasture was caused by abundance of grass roots. Plant roots affect macro-aggregation through organic debris and colloids whereas micro-aggregation is mainly influenced by extracellular polysaccharides produced by bacteria (Albrecht *et al*., 1998).

On the contrary, vegetation covers and soil aggregate stability were not statistically significant in both Mpala and Ilmotiok which is in agreement with the findings of O'Dea, (2007); Wang, (2010); Reinhart *et al*., (2015) and Barto *et al*., (2010). These studies did not detect positive correlations between measures of soil aggregate stability with plant cover, total peak above ground biomass and root biomass. Further, the results of a multiple regression analysis also showed that soil aggregate stability was not significantly influenced by other soil properties such as, SOC, SOM, bulk density and water infiltration rate.

Unexpectedly, the Tree and Grass cover fields had low water-stable soil aggregates compared to Bare grounds. These findings concur with Peres *et al*., (2013) who reported that plant species within a vegetation matrix may affect soil aggregate stability. Further, binding agents, such as fungal hyphae and plant roots of grasses and trees may stabilize large aggregates but are temporary and unstable (Ayoubi *et al*., 2012).

5.8 Water Infiltration as Affected by Vegetation Cover and Topographic Positioning

Water infiltration rate differed significantly between vegetation cover types in Mpala site. This is in agreement with previous studies by Rietkerk *et al*., (2000), who showed that total vegetation cover significantly affects infiltration and runoff during simulated intense rainfall events of 4-6 $mmMin^{-1}$. However, the case is different at Ilmotiok site, where no significant difference in infiltration rate between vegetation cover types was found. This is possibly because infiltration was measured directly using a mini-disk infiltrometer placed on the bare ground (Bower, 1986). As such, the measurements are independent of vegetation cover and it is unlikely that vegetation differences can explain the differences in water infiltration observed in this site.

As has been largely documented in grassland ecosystems, the type and spatial distribution of vegetation have spatial influence on soil surface characteristics and infiltration (Van Schaik, 2009). Accordingly, the results clearly showed that infiltration rates are not uniform within the Mpala and Ilmotiok ranches (Figure 4.6). Soil protection factors, mainly perennial grasses and litter cover, affect infiltration/runoff and soil detachment by raindrops (Chartier & Rostagno, 2006). Further, these findings may be attributed to land management which can affect infiltration, which is of great importance to vegetation restoration and crop production in semi-arid areas (Wang *et al*., 2008). Huang *et al*., *(*2010) reported that stable infiltration rate for grass cover was higher than that for bare soils, based on artificial rainfall experiments in Yangling, Shaanxi Province, China. In addition, studies on Andisols have drawn attention to the degradation of structural properties and decrease in infiltration capacity caused by a change in land use (Zehetner & Miller, 2006). Studies by Pirastru *et al*., (2013) determined that the change in land use affects the hydro physical properties of soil.

Tree covered areas in Mpala had low infiltration rate (0.7 mm/ hr) compared to Ilmotiok site (73.3 mm/hr) which could be associated to a higher organic litter accumulation and presence of materials such as lignin that give soils hydrophobic properties. Moreover, the infiltration rate of soil in Ilmotiok, was measured higher than the soils of Mpala owing to the initial soil moisture of the grassland and soil compaction resulting from livestock trampling.

The infiltration rate of soils was low in Bare grounds compared to Tree and Grass fields due to the compaction of soils created by animals as they compress over the bare soils (Thurow *et al*., (1988). Similar findings on the impact of uncontrolled grazing activities on soil aggregate stability were reported by Castellano & Valone, (2007). On the other hand, factors such as high SOC concentration in Mpala might have contributed to increased soil moisture retention (Hudson, 1994). Soil textural characteristics might have influenced the infiltration rates of Mpala soils because of the high clay content in Riparian zones, hence low infiltration rates.

5.9 Relationship between Water Infiltration Rate and Other Soil Properties

Soil texture variability resulting from different topographic positioning of the sampling sites was associated with different infiltration rates in the study. The low infiltration rate in Riparian zones might be attributed to the particle size distribution and other physical properties in the different soil layers. The highest infiltration rate in Hillslope zones might be attributed to the high sand content of over 80% and low clay content. This behavior is attributed to the fact that the coarse textured soils in the Hillslope zones have higher macro pores than the fine textured soil layers at Riparian zones. Additionally, the soils in the Riparian zones are enriched in clay content, mainly smectitic (Bouza *et al*., 2007), therefore, are associated with lower soil infiltration rate. Moreover, the basic infiltration rate has been to increase as macro pores increase because the coarse textured soils have higher macrospores than the fine textured ones. The findings also showed that, as sand content increased in Hillslope zones, water movement increased due to the relative increase in soil pores and the converse is true, Clemmens (1983).

Soil infiltration rates can also be linked to other factors such as the soil bulk density (USDA-NRCS, 2001). The low infiltration rates recorded in Bare grounds could possibly be due to cattle trampling effect among other factors. Studies show that soil texture controls the infiltration rate and the amount of water that can be stored in a given thickness of soil (USDA-NRCS, 2001). Clay soils for instance provide the highest surface area, but if clay content is high enough to restrict air and water movement, these critical variables may limit its productivity. Sandy soils on the other hand have high rates of water infiltration but are low in productivity because they do not retain water or nutrients (USDA-NRCS, 2001). The ideal substrate is therefore texturally balanced soil in the loam range (Tiedemann & Lopez, 2004).

Overgrazing has been found to reduce water infiltration, soil moisture and fertility, accelerate runoff and soil erosion, increase soil bulk density, penetration resistance, soil ammonia and nitrate content and alter soil microbial activity (Czegledi & Radacsi, 2005). In other studies, paired grazing has been previously found to have significantly affected infiltration rate (from 9.85 to 6.00 cm/hr), soil water content (from 24% to 15%) and greater soil loss to erosion in the arid environment (from 288.74 to 525.91 kg ha^{-1}) (Giese *et al*., 2013).

However, a multiple regression analysis indicated that infiltration rate was not significantly affected by soil properties such as bulk density, SOC, SOM, aggregate stability which is contrary to the findings by Castellano & Valone, (2007) which showed positive results on the relationship between infiltration rates with other soil properties.

Sarah (2002) found that shrubs affect soil properties through soil-vegetation interaction that resulted in higher infiltration rate and greater soil water retention capacity than in inter-shrub areas after rain. Similar results were found by Wang *et al., (*2008), who observed that the topsoil layer under high vegetation cover, as with the highly organic fine-grained soils and litter layer, reduced the thermal conductivity and increased the water infiltration and water holding capacity, altering the active soil-water-heat relationship.

5.10 Effects of Grazing and Grassland Management on Soil Properties in Ilmotiok and Mpala Ranches

Grazing management influenced the soil parameters in the two sites. Soil organic carbon and organic matter content were significantly different in Bare grounds among the vegetation covers in the two sites. Controlled grazing in Mpala ranch influenced the Bare grounds coverage and the amount of SOC and SOM contents found in these areas. Continuous heavy grazing in Ilmotiok ranch created more bare surfaces leaving the soil vulnerable to degradation processes hence reduced organic matter. For instance, the bulk density of soils was statistically significant between the two sites in the three vegetation cover types. Changes in bulk density were conditioned by increased SOC/SOM thus

Mpala ranch had higher levels of SOC and SOM and thus lower BD compared to Ilmotiok site. This is attributable to heavy grazing which reduces the organic litter in the soil, as well as livestock trampling creating soil compaction, resulting in high bulk densities. Grazing management in Mpala ranch explains the differences in aggregate stability of soils between the two sites. Tree fields and Bare grounds differed significantly in aggregate stability between the two sites. Hence the high soil aggregate stability found in Mpala site than in Ilmotiok site.

Water infiltration rates differed significantly (P=0.001; P=0.002; P=0.0043) in Tree fields, Grass fields and Bare grounds between the two sites. Thus, managed grazing activities in Mpala describe the high initial soil moisture contents thus low infiltration rates compared to Ilmotiok ranch. Soil organic carbon stocks differed significantly between the two sites, particularly in Tree fields and Bare grounds. This is attributable to conservation of the Mpala ranch hence, higher levels of SOCs than in Ilmotiok ranch. Finally, the open grassland (Ilmotiok ranch) can be considered different in ecology and soil characteristics from the protected grassland (Mpala ranch).

Controlled grazing at Mpala site was important in giving the grazed plants an ample recovery time/period which then increases the above ground biomass and organic matter pool after incorporation. Reeder & Schuman, (2002) in their study found similar results with areas that were under low/slightly grazed having more soil organic carbon than areas that were heavily grazed areas. They argued that the observed increase in SOC was because of the increase in the rates of nutrient cycling, annual shoot turnover and altered plant species composition. High grazing intensity has been reported to reduce the amount of organic carbon (Han *et al.*, 2008) compared to areas under low grazing intensities because of high net primary production in such areas. Furthermore, the presence of deep-rooted plants (trees) that gradually decompose when plant dies in combination with leaf litter decomposition may have contributed to high SOC in a controlled/managed grazing area than in continually grazed area. A few shrubs, grasses and herbs with shallow roots contribute to annual litter deposition that is also suppressed by herbivores and this resulted into low SOC accumulation in the area under open/uncontrolled grazing.

In contrast to the findings of this study, Ingram *et al.*, (2008) reported that areas under continuous heavy grazing had more organic carbon stocks than areas that were lightly grazed, because of the higher root mass they observed in areas under high grazing pressure. Pineiro *et al.*, (2010) in his review of differences on the effect of grazing on soil organic carbon revealed divergent results where grazing increased SOC while in some instances, it reduced or had no influence on SOC. A study in the semi-arid grassland biome of South Africa revealed that communal farms with continuous grazing were generally depleted of nutrient stocks, and nutrient depletion increased with increasing grazing intensity (Kotzé *et al*., 2013). Soil movement such as through erosion is usually associated with the loss of organic matter, nitrogen, phosphorus, potassium and other essential plant nutrients (Itanna *et al*., 2011).

CHAPTER 6: CONCLUSION AND RECOMMENDATION

6.1 Conclusion

1. Topographic positioning of the land and vegetation cover affects SOC and SOM contents in grasslands. The Tree fields and Riparian zones contained higher SOC and SOM contents while Bare grounds and Hillsope zones contained the lowest SOC and SOM contents. The level of SOC and SOM contents affected other soil properties such as bulk density. Low bulk densities in the soil were conditioned by reduced levels of SOC and SOM in the two sites. Soil depth also significantly affected the levels of SOC and SOM in the two sites.
2. Topographic positioning significantly influenced water stable aggregates in the two sites. However, vegetation cover and soil depth had no significant effect on the aggregate stability of soils in Mpala and Ilmotiok ranches.
3. The findings revealed that topographic zones significantly affected the infiltration rates in both sites. Vegetation cover significantly influenced infiltration rates in Mpala ranch but the opposite was true for Ilmotiok ranch. The Riparian zones had the lowest infiltration rates while the Hillslope zones had the lowest infiltration rates
4. Grazing management influenced the soil properties in the two sites. Controlled grazing in Mpala ranch influenced SOC and SOM contents, aggregate stability and water infiltration rates. Continuous grazing in Ilmotiok ranch created more bare surfaces whcih influenced the amount of litter returned to the soil, thus affecting SOC

and SOM contents. Consequently, this affected the aggregation and infiltration rate of soils.

The study therefore concludes that SOM is an indicator of soil quality assessment in semi-arid grasslands. Land degradation processes as a result of land use practices in this area affect the functioning of SOM in the ecosystem which further affects productivity in the rural livelihoods.

6.3 Recommendations

From the study, it is recommended that;

1. Further studies on SOC dynamics in Kenyan semi-arid grasslands are needed due to limited knowledge created by inadequate data availability and therefore stakeholders may team up together to create databases that are easily accessible by the research community for studies concerning grassland SOC storage
2. There is need to develop land management strategies that encourage pasture/fodder production and or reseeding to enhance restoration of bare areas to improve soil aggregation as well as feed for livestock to improve community livelihoods
3. It's imperative to evaluate soil water levels in semi-arid grasslands and encourage conservation agriculture to improve grassland production potential
4. There is need to adopt proper grazing management practices that incorporate closed or rotational grazing system for proper vegetation restoration, improved soil conditions and overall grassland production

REFERENCES

Adhikari, K., & Hartemink, A. E. (2016). Linking soils to ecosystem services—A global review. *Geoderma*, *262*, 101-111.

Akuja, T.E., Zaady, E., Ward D. & Gutterman, Y. (2005). Effects of soil erosion and land use on soil quality in the semi-arid Negev Desert of Israel. *East African journal of Life Sciences*, 5: 67-74.

Albrecht, A., Angers, D.A., Beare, M.H. & Blanchart, E. (1998). Déterminants organiques et biologiques de l'agrégation: implications pour la recapitalisation de la fertilité physique des sols tropicaux. *Cahiers d'Agricultures* 7: 357-363.

Allard, V., Soussana, J.F., Falcimagne, R., Berbigier, P., Bonnefond, J.M., Ceschia, E., D'hour, P., Henault, C., Laville, P., Martin, C., & Pinares-Patino, C. (2007). The role of grazing management for the net biome productivity and greenhouse gas budget (CO2, N2O, and CH4) of semi-natural grassland. *Agriculture, Ecosystems and Environment,* 121, 47–58.

Amundson, R., Berhe, A.A., Hopmans, J.W., Olson, C., Sztein, A.E., & Sparks, D.L. (2015). Soil and human security in the 21st century. Science 348, 126-107.

Bagamsah, T.T. (2005). The impact of bushfire on carbon and nutrient stocks as well as albedo in the savanna of northern Ghana. ISBN: 3-86537-406-9

Bai, Z.G., Dent, D.L., Olsson, L. and Schaepman, M.E. (2008) Proxy Global Assessment of Land Degradation. *Soil Use Management*, 24, 223-234. http://dx.doi.org/10.1111/j.1475-2743.2008.00169.x

Barto E.K., Alt F., Oelmann Y., Wilcke W., & Rillig MC. (2010). Contributions of biotic and abiotic factors to soil aggregation across a land use gradient. *Soil Biology and Biochemistry Journal*, 42(12):23–24.

Batjes, N.H., 1996. Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, 47(151 – 163).

Beatrice, N., Julius, O. and Geoffrey, K. (2012) "Drivers of the transition from pastoralism to vegetable farming in Africa's arid and semi-arid areas and implications for soil fertility management: The case of Kenyan pastoralists." *African Journal of Agricultural Research*, Vol. 7(15), pp. 2273-228

Berhe, A.A. (2012). Decomposition of organic substrates at eroding vs. depositional landform positions. *Plant Soil Journal* 350 (1–2), 261–280

Berhe, A.A., & Kleber, M. (2013). Erosion, deposition, and the persistence of soil organic matter: mechanistic considerations and problems with terminology. *Earth Surface Processes and Land forms Journal*, 38, 908–912.

Berhe, A.A., Harden, J.W., Torn, M.S., & Harte, J. (2008). Linking soil organic matter dynamics and erosion-induced terrestrial carbon sequestration at different landform positions. *Journal of Geophysical Research,* 113 (G4)

Berry, L. (2003). Land degradation in Ethiopia: its impact and extent in Berry L., Olson J., & Campbell, D. (ed(s).): Assessing the extent, cost and impact of land degradation at the national level: findings and lessons learned from seven pilot case studies. Commissioned by global mechanism with support from the World Bank.

Bewket W., & Stroosnijder L. (2003). Effects of agro-ecological land use succession on soil properties in Chemoga watershed, Blue Nile basin, Ethiopia. *Geoderma journal*, 111:85–98.

Bezuayehu T., Gezahegn A., Yigezu A., Jabbar M., & Paulos D. (2002). Nature and causes of land degradation in the Oromiya Region: socio- economic and policy research working paper. International Livestock Research Institute.

Bini, C. (2009) Soil: A precious natural resource. In *Conservation of Natural Resources*; Kudrow, N.J., (Ed.); Nova Science Publishers: Hauppauge, NY, USA; pp. 1–48.

Bisigato, A., Bertiller, M.B., (2004b). Seedling recruitment of perennial grasses in degraded areas of the Patagonian Monte. *Journal of Range Management* 57, 191–196

Blaikie, P., & Brookfield, H. (Ed(s.)). (2015). *Land degradation and society journal*. Routledge.

Bossio D., Noble A., Pretty J., & Vries F. (2004). Reversing land and water degradation: Trends and 'Bright Spot' Opportunities. Paper presented at the SIWI/CA Seminar. Stockholm, Sweden.

Bouza, P.J., Simón, M., Aguilar, J., del Valle., H.F., & Rostagno, C.M., (2007). Fibrous-clay mineral formation and soil evolution in Aridisols of northeastern Patagonia, Argentina. Geoderma 139, 38-50.

Bower, H. (1986). Intake rate. Cylinder infiltrometer. Pg 825-843. In Klute, A. (Ed.) Methods of soil analysis: Part 1. Physical and mineralogical properties, 2nd (Ed.). ASA, Madison, WI.

Brevik, E.C., & Homburg, J. (2004). A 5000 year record of carbon sequestration from a coastal lagoon and wetland complex, Southern California, USA, Catena, 57, 221–232.

Brevik, E.C., Cerdà, A., Mataix-Solera, J., Pereg, L., Quinton, J.N., Six, J., & Van Oost, K. (2015). The interdisciplinary nature of soil. 1, 117–129.

Briske, D.D., Fuhlendorf, S.D., & Smeins. F.E. (2005). State-and-transition models, thresholds, and rangeland health: a synthesis of ecological concepts and perspectives. *Rangeland Ecology & Management Journal* 58:1–10. *Water Resources Research* (in press).

Butynski, T.M., & De Jong, Y.A. (2014). Primate conservation in the rangeland agro-ecosystem of Laikipia County, central Kenya. Primate Conservation 28: 117–128. http://www.lolldaiga.com/primate-conservation-in-the-rangeland-agroecosystem-of laikipia-county-central-kenya/

Cambardella, C.A., and Elliott, E.T. (1994). Carbon and nitrogen dynamics of soil organic matter fractions from cultivated grassland soils, Soil Science Society of American Journal, 58, 123–130.

Campbell D., Gichohi H., Reid R., Mwangi A., Chege L., & Sawin, T. (2003). Interactions between people and wildlife in Southeast Kajiado District, Kenya. LUCID Working paper No. 18. International Livestock Research Institute. Nairobi.

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., ... & Kinzig, A. P. (2012). Biodiversity loss and its impact on humanity. *Nature*, *486*(7401), 59-67.

Carsel, R.F., & Parrish R.S. (1988). Developing joint probability distributions of soil water retention characteristics. *Water Resources Journal*. ; 24 (5): 755-69.

Casermeiro, M.A., Molina, J.A., Caravaca, M., Costa, J.H., Massanet, M.I.H., & Moreno, P.S. (2004). Influence of scrubs on runoff and sediment loss in soils of Mediterranean climate. Catena 57 (1), 91–107

Castellano, M., & Valone, T., (2007). Livestock, soil compaction and water infiltration rate: evaluating a potential desertification recovery mechanism. *Journal of Arid Environments*, 71 (1), 97–108.

Celik, I. (2005). Land Use Effects on Organic Matter and Physical Properties of Soil in a Southern Mediterranean Highland of Turkey. *Soil and Tillage Research Journal*, 83, 270-277.

Cerda, A. (1997). Seasonal changes of the infiltration rates in a Mediterranean scrubland on limestone. *Journal of Hydrology* 198, 209–225.

Cerda, A. (1998a). The influence of aspect and vegetation on seasonal changes in erosion under rainfall simu7lation on a clay soil in Spain. *Canadian Journal of Soil Science* 78, 321–330.

Chartier, M.P., & Rostagno, C.M. (2006). Soil erosion thresholds and alternative states in northeastern Patagonian rangelands. *Rangeland Ecology and Management Journal.*

Chaudhary, V.B., Bowker, M.A., O'Dell, T.E., Grace, J.B., Redman, A.E., … Rillig, M.C. (2009). Untangling the biological contributions to soil stability in semiarid shrublands. *Ecological Applications Journal*, 19(1), 110–22. PMID: 19323176

Chevallier, T., Blanchart, E., Girardin, C., Mariotti, A., Albrecht, A. & Feller, C. (2001). The role of biological activity (roots, earthworms) in medium- term C dynamics in Vertisol under a Digitaria decumbens (Gramineae) pasture. *Applied Soil Ecology Journal,* 16, 11-21.

Christensen, B.T. (2001). Physical fractionation of soil and structural and functional complexity in organic matter turnover. European Journal of Soil Science, 52, pp. 345-353

Ciais, P., Dolman, A.J., Dargaville, R., Barrie, L., Bombelli, A., Butler, J., Canadell, P., Moriyama, T. (2010). Geo Carbon Strategy. GEO, Secretariat, Geneva/FAO, Rome (48 pp).

Civeira, G., & Irigoin, J., & Paladino, I., (2012). Soil Organic Carbon in Pampean Agroecosystems: Horizontal and Vertical Distribution Determined by Soil Great Group. Soil Horizons. 53. 43. 10.2136/sh12-04-0015.

Clemmens, A.J. (1983). Infiltration equations for border irrigation models. P. 266-274. *In:* Advances in infiltration. Proceedings of the National Conference on Advances in Infiltration. Dec. 12-13. Chicago, Ill. ASAE Pub. I l-83. St. Joseph MO.

Conant, R.T. (2012). Grassland soil organic carbon stocks: status, opportunities, vulnerability. In: Lal, R. (Ed.), Re-carbonization of the Biosphere: Ecosystems and the Global Carbon Cycle. Springer Science + Business Media B.V.

Coxhead, I., & Ygard, R. (2008). Land Degradation. Submitted for Copenhagen Consensus Comments. University of Wisconsin-Madison and Norwegian University of Life Sciences.

Czegledi, L., & Radacsi, A. (2005). Overutilization of pastures by livestock. Grassland Studies, 3, 29-35.

De Gryze, S., Jassogne, L., Bossuyt, H., Six, J., Merckx R (2006) Water repellence and soil aggregate dynamics in a loamy grassland soil as affecte by texture. Eur J Soil Sci. 57:235– 246

Derner, J., Boutton, T., & Briske, D. (2006). Grazing and ecosystem carbon storage in the North American Great Plains. *Plant and Soil Journal* 280, 77–90.

Diaz-Hernandez, J.L., Fernandez, B., & Gonzalez, L. (2003). Organic and inorganic carbon in soils of semiarid regions: A case study from the Guadix-Baza basin (Southeast Spain). *Geoderma*. 114. 10.1016/S0016-7061(02)00342-7.

Diaz-Zorita, M., Perfect, E., Grove, J.H. (2002). Disruptive methods for assessing soil structure. *Soil and Tillage Research Journal*. Vol. 64 pg. 3–22.

Doetterl, S., Berhe, A., Nadeu, E., Wang, Z., Sommer, M., & Fiener, P. (2016). Erosion, deposition and soil carbon: a review of process-level controls, experimental tools and models to address C cycling in dynamic landscapes. *Earth-Science Reviews Journal*, 154, 102–122.

Dominati, E., Mackay, A., & Patterson, M. (2010). Modelling the provision of ecosystem services from soil natural capital. Proceedings of the 19th World Congress of Soil

Science: Soil Solutions for a Changing World, Brisbane, Australia, 1–6 August. Congress Symposium 2: Soil ecosystem Services, pp. 32–35

Dong, S.K., Wen, L., Yi, Y.Y., Wang, X.X., Zhu, L., & Li, X.Y. (2012). Soil-quality effects of land degradation and restoration on the Qinghai–Tibetan plateau. Soil Science Society of America Journal, 76, 2256–2264.

Du Preez, C.C., Van Huyssteen, C.W., & Mnkeni P.N.S. (2011). Land use and soil organic matter in South Africa 1: A review on spatial variability and the influence of rangeland stock production. *South African Journal of Science*, 107(5-6).

Dunkerley, D.L. (2002). Infiltration rates and soil moisture in a groved Mulga community near Alice Springs, arid central Australia: evidence for complex internal rainwater redistribution in a runoff run on landscape. *Journal of Arid Environment*, 51, 199–219.

Feller, C., Albrecht, A., Blanchart, B., Cabidoche, Y.M., Chevallier, T., Hartmann, C., Eschenbrenner, V., Larre-Larrouy, M.C. & Ndandov, J.F. (2001). Soil Organic carbon sequestration in tropical areas. General considerations and analysis of some edaphic determinants for lesser Antilles Soils. Nutrient Cycling in Agro-ecosystems, 61, 19-31

Fernandez, D.P., Neff, J.C., Reynolds, R.L. (2008). Biogeochemical and ecological impacts of livestock grazing in semi-arid southern-east Utah, USA. *Journal of Arid Environment*, 72, 777–791.

Fierer, N., Schimel, J.P., & Holden, P.A., (2003). Variations in microbial community composition.

Fissore, C., Dalzell B.J., Berhe, A.A. Voegtle, M., Evans, M., & Wu, A. (2017). Influence of topography on soil organic carbon dynamics in a Southern California grassland. *Catena*, Volume 149, Part 1, Pages 140-149.

Fontaine, S., Barrot, S., Bdioui, N., Mary, B., and Rumpel, C. (2007). Stability of organic carbon in deep soil layers controlled by fresh carbon supply. Nature 450: 27728

Frank, D.A., Kuns M.M., & Guido D.R. (2002). Consumer control of grassland plant production. *Ecology Journal*, 83,602–606.

Franzluebbers, A.J., & Doraiswamy, P.C. (2007). Carbon sequestration and land degradation. In: Sivakumar, M.V.K & Ndiang'ui, N. (Ed(s.). Climate and Land Degradation, 18. Springer, Berlin Heidelberg, New York, pp. 392–403.

Gale, J., Cambardella, C.A., & Bailey, T.B. (2000). Root-derived carbon and the formation and stabilization of aggregates. *Soil Science Society of America Journal*, 64, 201-207.

Gao, Y., Zhu, B., Zhou, P., Tang, J.L., Wang, T., & Miao, C.Y. (2009). Effects of vegetation cover on phosphorus loss from a hillslope cropland of purple soil under simulated rainfall: a case study in China. *Nutrient Cycling in Agro-ecosystem Journal*, 85 (3), 263–273

Gebreyesus B and Kirubel M (2009). Estimating Soil Loss Using Universal Soil Loss Equation (USLE) for Soil Conservation planning at Medego Watershed, Northern Ethiopia. *Journal of American Science*, 5(1), 58-69.

GEF, (Global Environmental Facility) (2006). Land degradation as a global environmental issue: a synthesis of three studies commissioned by the global environment facility to strengthen the knowledge base to support the land degradation focal area. Scientific and Technical Advisory Panel, Washington, DC.

Gelaw, A.M., Singh, B.R., & Lal, R. (2013). Organic carbon and nitrogen associated with soil aggregates and particle sizes under different land uses in Tigray, Northern Ethiopia, Land Degradation & Development, 7, 690–700, doi:10.1002/ldr.2261, 2013.

Gessler, P.E., Chadwick, O.A., Chamran, F., Althouse, L., Holmes, K. (2000). Modeling soil-landscape and ecosystem properties using terrain attributes. *Soil Science Society of American Journal*, 64, 2046–2056.

Giese, M., Brueck, H., Gao, Y., Lin, S., Steffens, M., Kögel-Knabner, I., Glindemann, T.,… Susenbeth, A. (2013). N balance and cycling of Inner Mongolia typical steppe: A comprehensive case study of grazing effects. *Ecological Monographs Journal*, 83, 195-219. Doi: 10.1890/12-0114.1.

Goldammer, J. (2000). Regional to Global Fires Issues: The role of vegetation fires in Africa and other continents in global processes Proceedings of a round table conference on integrated forest fire management in Ethiopia, 2000. Ministry of Agriculture, GTZ & GFMC, Addis Ababa, pp. 36 -50.

Gómez-Plaza, A., Martınez-Mena, M., Albaladejo, J. & Castillo, V. (2001), Factors regulating spatial distribution of soil water content in small semiarid catchments, *Journal of Hydrology*, 253(1), 211–226.

Greenhalgh, S., & Sauer, A. (2003). Awakening the dead zone: an investment for agriculture, water quality, and climate change. World Resources Institute (available at http://www.wri.org/publication/awakening-dead-zoneinvestment-agriculture-water-quality-and-climate-change, last accessed 2 September 2020)

Gregorich, E., Carter, M.R., Angers, D.A. (1994). Toward a minimum data set to assess soil organic-matter quality in agricultural soils. *Canadian Journal of Soil Science*, 74: 367–385.

Han, G., Hao, X., Zhao, M., Wang, M., Ellert, H.B., Willms, W., & Wang, M. (2008). Effect of grazing intensity on carbon and nitrogen in soil and vegetation in a meadow steppe in Inner Mongolia. *Agriculture, Ecosystems and Environment*, 125, 21-32. 10.1016/j.agee.2007.11.009.

Hancock, G.R., Murphy, D., & Kenneth, E. (2010). Hillslope and catchment scale soil organic carbon concentration: An assessment of the role of geomorphology and soil erosion in an undisturbed environment. *Geoderma* . 155. 36-45. 10.1016/j.geoderma.2009.11.021.

He, N.P., Yu, Q., Wu, L., Wang, Y.S., & Han, X.G. (2008). Carbon and nitrogen store and storage potential as affected by land-use in a Leymus chinensis grassland of northern China. *Journal of Soil Biology and Biochemistry*, 40, 2952–2959.

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H., & Roberts, C. (2005). Confronting a biome crisis: global disparities of habitat loss and protection. Journal of Ecology Letters, 8, 23–9.

Hontoria, C., Rodríguez-Murillo, J.C., & Saa, A. (2004). Contenido de carbono orgánico en el suelo y factores de control en la España peninsular, Edafología, 11, 149–157.

Huang, J., Wu, P.T., & Zhao, X.N., (2010). Impact of slope biological regulated measures on soil water infiltration. Transactions of the CSAE 26 (10), 29–37.

Hudson, B.D. (1994). Soil organic matter and available water capacity. *Journal of Soil Water Conservation*, 49, 189–94.

Hussein, L., Oweis, T., & Abbas, J. (2006). "The role of micro-catchment water harvesting techniques in combating desertification in the Syrian steppe." International Soil Conservation Organization Conference. 14-19 May, 2006.

Igwe, C.A. (2005). Soil physical properties under different management systems and organic matter effects on soil moisture along soil catena in south eastern Nigeria. *Tropical and subtropical agro-ecosystems Journal*, 5, 57- 68.

Imeson, A.C., & Verstraten, I.M., (1989). The micro-aggregation and erodibility of some semi-arid and Mediterranean soils. Catena Supplement, 14, 11–24.

Ingram, L., Stahl, P.D., Schuman, G.E., Buyer, J., Vance, G., Ganjegunte, G., Welker, J., & Derner, J. (2008). Grazing Impacts on Soil Carbon and Microbial Communities in a Mixed-Grass Ecosystem. *Soil Science Society of America Journal*. Doi: 72. 10.2136/sssaj2007.0038.

Itanna, F., Mats O., and Karl S., (2011). Effect of Land Use Changes on Soil Carbon Status of Some Soil Types in the Ethiopian Rift Valley. Journal of the Drylands 4(1): 289-299.

Jackson, B.R., Canadell, J.G., Ehleringer, J., Mooney, H., Sala, E.O. & Schulze, E.D. (1996). A Global Analysis of Root Distributions for Terrestrial Biomes. *Oecologia Journal*, 108, 389-411. Doi: 10.1007/BF00333714.

Jenkinson, D.S., Adams, D.E. & Wild, A., (1991). Model estimates of CO_2 emissions from soil in response to global warming. Nature, 351:304-306.

Jobbágy, E. & Jackson, B.R. (2000). The Vertical Distribution of Soil Organic Carbon and Its Relation to Climate and Vegetation. *Ecological Applications Journal*, 10, 423-436. 10.2307/2641104.

Johnston, A.E., Poulton, P.R., & Coleman, K., (2009). Chapter 1 soil organic matter: its importance in sustainable agriculture and carbon dioxide fluxes. *Advances in Agronomy* 101, 1–57.

Kaiser, M., Wirth, S., Ellerbrock, R. H., & Sommer, M. (2010). Microbial respiration activities related to sequentially separated, particulate and water-soluble organic matter fractions from arable and forest top soils, *Soil Biology and Biochemistry*, 42, 418–428.

Kamiri, H., Kreye, C., & Becker, M. (2013). Dynamics of agricultural use differentially affect soil properties and crop response in East African wetlands. *Wetlands Ecology and Management Journal.* Vol 21, Issue 4. Issue 6, pp 417-431.

Kareri, R.W., (2009). "Some aspects of the geography of Kenya." Online: http://international.iupui.edu/kenya/resources/Geography-of-Kenya.pdf

Kassahun, A., Snyman, H.A., & Smit, G.N. (2008). Impact of rangeland degradation on the pastoral production systems, livelihoods and perceptions of the Somali pastoralists in *Eastern Ethiopia Journal of Arid Environments*, 72 (7), 1265-1281.

Kemp, D. R., Han, G.D., Hou, X.Y., Michalk, D.L., Hou, F.J., Wu, J.P., & Zhang, Y.J. (2013). Innovative grassland management systems for environmental and livelihood benefits, Proceedings of the National Academy of Sciences. USA, 110, 8369–8374.

Kemper, W., & Rosenau, R. (1986). Aggregate Stability and Size Distribution. In: Klute, A., (Ed.), Methods of Soil Analysis, Part 1. Physical and Mineralogical Methods, Agronomy Monograph, *American Society of Agronomy and Soil Science Society of America*, Madison, 425-442.

Kgosikoma, O., Mojeremane, W., & Harvie, B.A. (2012). Pastoralists' perception and ecological knowledge on savanna ecosystem dynamics in semi-arid Botswana. *Ecology and Society* 17(4): 27.

Kioko, J., Kiringe, J.W., Seno, S.O. (2012). Impacts of livestock grazing on a savanna grassland in Kenya. *Journal of Arid Lands* 4(1):29-35.

Kotzé, E., Sandhage-Hofmann, A., Meinel, J.A., du Preez, C.C., & Amelung, W., (2013). Rangeland management impacts on the properties of clayey soils along grazing gradients in the semi-arid grassland biome of South Africa. *Journal of Arid Environments*, 97, 220–229.

Krummelbein, J., Wang, Z., Zhao, Y., Peth, S., Horn, R., (2006). Influence of various grazing intensities on soil stability, soil structure and water balance of grass land soils in Inner Mongolia, P.R. China. In: Horn, R., Fleige, H., Peth, S., Peng, X.(Ed(s.), *Advances in Geoecology*, vol. 38. pp. 93–101.

Krupenikov, I., Boincean, B., Dent, D., Krupenikov, I., Boincean, B., & Dent, D. (2011). Humus-Guardian of Fertility and Global Carbon Sink. In *The Black Earth: Ecological Principles for Sustainable Agriculture on Chernozem Soils*; International Year of Planet Earth; Springer: Dordrecht, The Netherlands, pp. 39–50.

Kutilek, M., Jendele, L., & Panayiotopoulos, K.P., 2006. The influence of uniaxial compression upon pore size distribution in bi-modal soils. *Soil Tillage & Research,* 86(1), 27–37.

Lal, R. (2003). Soil erosion and the global carbon budget. *Environment International*, 29: 437–450.

Lal, R. (2004). Soil carbon sequestration impacts on global climate change and food security. *Science*, 304, 1623–1627.

Lal, R. (2009) Soil degradation as a reason for inadequate human nutrition. *Food Security*, 1, 45-57.

Lal, R. (2010): Managing soils and ecosystems for mitigating anthropogenic carbon emissions and advancing global food security. *Bioscience*, 60, 708–721.

Lalampaa, P.K., Wasonga, O.V., Rubenstein, D.I., & Njoka, J.T. (2016). Effects of holistic grazing management on milk production, weight gain, and visitation to grazing areas by livestock and wildlife in Laikipia County, Kenya. *Ecological Processes*, *5*(1), 17.

Lamb, D., Erskine, P., Parrotta, J. (2005). Restoration of degraded tropical forest landscapes. *Science*, *310*, 1628–1632.

Lawal, H.M., Ogunwole, J.O., & Uyovbisere, E.O. (2009). Changes in soil aggregate stability and carbon sequestration mediated by land use practices in a degraded dry savanna alfisol tropical and subtropical agro-ecosystems, vol. 10, no. 3, pp. 423-429 Universidad Autónoma de Yucatán Mérida, Yucatán, México.

LeBlanc, J. C., Gonçalves, E. R., & Mohn, W. W. (2008). Global response to desiccation stress in the soil actinomycete Rhodococcus jostii RHA1. *Applied and environmental microbiology*, *74*(9), 2627-2636.

Leon, J., & Osorio, N. (2014). Role of litter turnover in soil quality in tropical degraded lands of Colombia. *Science World Journal, 13*.

Li, C.L., Hao, X.Y., Zhao, M.L., Han, G.D. & Willms W.D. (2008). Influence of historic sheep grazing on vegetation and soil properties of a Desert Steppe in Inner Mongolia. *Agriculture Ecosystems & Environment* 128:109-116.

Li, F.R., Zhao, W.Z., Liu, J.L., & Huang Z.G. (2009). Degraded vegetation and wind erosion influence soil carbon nitrogen and phosphorus accumulation in sandy grasslands. *Plant Soil*, 317, 79–92

Li, P., Wang, Q., Endo, T., Chao, X., & Kakubari, Y., (2010). Soil organic carbon stock is closely related to aboveground vegetation properties in cold-temperature mountainous forests. *Geoderma*, 154, 407–415.

Li, Y.Y., & Shao, M.A. (2006). Change of soil physical properties under long-term natural Vegetation restoration in the Loess Plateau of China. *Journal of Arid Environments,* 64, 77 -96.

Ludwig, J.A., Wilcox, B.P., Breshears, D.D., Tongway, D.J., & Imeson, A.C. (2005). Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes, *Ecology*, 86(2), 288–297.

Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordán, A., & Zavala, L.M., (2011). Fire effects on soil aggregation: A review. *Earth Science. Review*, 109:44-60.

Mati, B.M., (2005). "Overview of water and soil nutrient management under smallholder rainfed agriculture in East Africa." Working paper 105. Colombo, Sri Lanka. International Water Management Institute (IWMI).

McSherry, M., & Ritchie, M.E., (2013). Effects of grazing on grassland soil carbon density: a global review. *Global Change Biology,* 19, 1347–1357.

Meskini, V., Mohammadi, M., & Vanclooster, M. (2014). Predicting the Soil Moisture Retention Curve, from Soil Particle Size Distribution and Bulk Density Data Using a Packing Density Scaling Factor. *Hydrology and Earth System Sciences*, 18, 4053-4063.

Mganga, K.Z., Nyangito, M.M., Musimba, N.K.R. (2010). "The challenges of rehabilitating denuded patches of a semi-arid environment in Kenya." *African journal of environmental science and technology*, Vol 4.No. 7. pp 430-436.

Michaelides, S., Levizzani, V., Anagnostou, E., Bauer, P., Kasparis, T., & Lane, J. (2009). Precipitation: Measurement, remote sensing, climatology and modeling. *Atmospheric Research*, 94. 512-533. 10.1016/j.atmosres.2009.08.017.

Minasny, B., McBratney, A.B., Mendonça-Santos, M.L., Odeh, I.O.A., & Guyon, B., (2006). Prediction and digital mapping of soil carbon storage in the Lower Namoi Valley. *Australian Journal of Soil Research*, 44, 233–244.

Mulugeta, L. (2004). Effects of land use change on soil quality and native flora degradation and restoration in the highlands of Ethiopia. Implication for sustainable land management. Ph.D Thesis. Swedish university of Agricultural Science. Uppsala, Sweden.

Murty, D., Kirschbaum, M.F.U., Mcmurtrie, R.E. & Mcgilvray, H. (2002). Does Conversion of Forest to Agricultural Land Change Soil Carbon and Nitrogen? A Review of the Literature. *Global Change Biology*, 8, 10-123.

Muya, E.M., Obanyi, S., Ngutu, M., Sijali, I.V., Okoti, M., Maingi, P.M. & Bulle, H., (2011). The physical and chemical characteristics of soils of Northern Kenya Arid lands: Opportunity for sustainable agricultural production. *Journal of Soil Science and Environmental Management* 2(1): 18.

Neely, C., Bunning, S., & Wilkes, A. (2009). Review of evidence on dry lands pastoral systems and climate change: Implications and opportunities for mitigation and adaptation, Land and Water Discussion Paper 8, Food and Agriculture Organization of the United Nations: Rome.

Oades, J.M. (1984). Soil organic matter and structural stability: mechanism and implications for management. *Plant Soil*, 76:319–337.

O'Dea, M.E. (2007). Fungal mitigation of soil erosion following burning in a semi-arid Arizona savanna. *Geoderma*, 138(1–2):79–85.

Okalebo, J.R., Gathua, K.W., & Woomer, P.L. (2002). Laboratory methods of plant and soil analysis: a working manual. Tropical Soil Biology and Fertility Programme, Nairobi.

O'Mara, F.P. (2012). The role of grasslands in food security and climate change. *Annals of Botany*, 110, 1263–1270.

Pattanayak, S.K., McCarl, B.A., Sommer, A.J., Murray, B.C., Bondelid, T., Gillig, D. & de Angelo, B. (2005). Water quality co-effects of greenhouse gas mitigation in US agriculture. *Climatic Change,* 71: 341–372.

Paulos, D. (2001). Soil and water resources and degradation factors affecting their productivity in the Ethiopian highland agro-ecosystems. Michigan State University Press. 8(1): 1-18.

Peres, G., Cluzeau, D., Menasseri, S., Soussana, J.F., Bessler, H., & Engels, C. (2013). Mechanisms linking plant community properties to soil aggregate stability in an experimental grassland plant diversity gradient. *Plant Soil*, 373(1):285–99.

Pineiro, G., Paruelo, J., Oesterheld, M., Jobbágy, E. (2010). Pathways of Grazing Effects on Soil Organic Carbon and Nitrogen. *Rangeland Ecology and Management*, 63,109-119. 10.2111/08-255.1.

Piotrowski, J.S, Denich, T., Klironomos, J.N., Graham, J.M., & Rillig, M.C (2004). The effects of arbuscular mycorrhizas on soil aggregation depend on the interaction between plant and fungal species. *New Phytologist*, 164(2):365–73.

Pirastru, M., & Niedda, M., (2013). Evaluation of the soil water balance in an alluvial flood plain with a shallow groundwater table. *Hydrological Sciences Journal*, 58, 4, 898–911.

Post, W.M., & Kwon, K.C. (2000). Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology,* 6, 317– 327.

Powlson, D.S., Whitmore, A.P., Goulding, K.W.T. (2011). Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *European Journal of Soil Science*, 62, 42 55.

Pribyl, D.W. (2010). A critical review of the conventional SOC to SOM conversion factor. *Geoderma*, 156, 75–83.

Project Development Facility (2007). Strategic Investment Programme for Sustainable Land Management in Sub-Saharan Africa: Assessment of the Barriers and Bottlenecks to Scaling-up sustainable land management investments throughout Sub Saharan Africa. Revised Draft.

Quinton, J.N., Catt, J.A., Wood, G.A., & Steer, J. (2006). Soil carbon losses by water erosion: Experimentation and modeling at field and national scales in the UK. *Agriculture, Ecosystems & Environment,* 112 (1), 87-102.

Reeder, J.D., & Schuman, G.E., (2002). Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass range- lands. *Environmental Pollution,* 116, 457–463.

Reid, R.S., Thornton, P.K., McCrabb, G.J., Kruska, R.L., Atieno, F., & Jones, P.G. (2004). Is it possible to mitigate greenhouse gas emissions in pastoral ecosystems of the tropics *Environment, Development & Sustainability* 6, 91–109.

Reinhart, K.O., Dangi, S.R., & Vermeire, L.T. (2016). The effect of fire intensity, nutrients, soil microbes, and spatial distance on grassland productivity. *Plant Soil*. 2016:1–14.

Reinhart, K.O., Nichols, K.A., Petersen, M., & Vermeire, L.T. (2015). Soil aggregate stability was an uncertain predictor of ecosystem functioning in a temperate and semiarid grassland. *Ecosphere*, 6(11).

Requier-Desjardins, M. (2008). Social costs of desertification in Africa: The case of migration. In Future of Drylands, Lee, C., & Schaaf, T. (ed(s.). UNESCO: Paris; 569–581.

Rezaei, S.A., Gilkes, R.J., Andrews, S.S. (2006). A minimum data set for assessing soil quality in rangelands. *Geoderma*, 136(1–2):229–34.

Rietkerk, M., Ketner, P., Burger, J., Hoorens, B., & Olff, H. (2000). Multiscale soil and vegetation patchiness along a gradient of herbivore impact in a semi-arid grazing system in West Africa. *Plant ecology*, *148*(2), 207-224.

Rillig, M.C., Wright, S.F., Eviner, V.T. (2002). The role of arbuscular mycorrhizal fungi and glomalin in soil aggregation: comparing effects of five plant species. *Plant Soil*, 238(2):325–33.

Ritchie, M., Mayemba, E., McSherry, M., & Tear, T. (2012). Soil carbon dynamics in the Northern Rangelands Trust Member Conservancies, Kenya.

Ruiz-Sinoga, J.D., & Romero Diaz, A., (2010). Soil degradation factors along a Mediterranean pluviometric gradient in southern Spain. *Geomorphology*, 188, 359–368.

Sainepo, B. M., Gachene, C. K., & Karuma, A. (2018). Assessment of soil organic carbon fractions and carbon management index under different land use types in Olesharo atchment, Narok County, Kenya. *Carbon balance and management*, *13*(1), 4. Doi: 10.1186/s13021-018-0091-7

Sakin, E., Deliboran, A., & Tutar, E. (2011). Bulk Density of Harran Plain Soils in Relation to Other Soil Properties. *African Journal of Agricultural Research*, 6, 1750-1757.

Sanjari, G., Ghardiri, H., Ciesiolka, C.A.A., & Yu, B. (2008). Comparing the effects of continuous and time-controlled grazing systems on soil characteristics in Southeast Queensland. Aust. *Journal of Soil Research,* 46, 348–358.

Sarah, P. (2002). Spatial patterns of soil moisture as affected by shrubs, in different climatic conditions. *Environmental Monitoring and Assessment*, 73: 237–241

Scherr, S., & Yadav, S. (1996). Land Degradation in the Developing World: Implications for Food, Agriculture, and the Environment to 2020. Food, Agriculture and the

Environment Discussion Paper 14. International Food Policy Research Institute, Washington, D.C, pp1-29

Schiettecatte, W., Gabriels, D., Cornelis, W., & Hofman, G. (2008). Enrichment of Organic Carbon in Sediment Transport by Interrill and Rill Erosion Processes. *Soil Science Society of America Journal*, 72. 50-55.

Schulze, K., Borken, W., Muhr, J., & Matzner, E. (2009), Stock, turnover time and accumulation of organic matter in bulk and density fractions of a Podzol soil. European Journal of Soil Science, 60(4), 567–577

Shibru, T. (2010). Land Degradation and Farmers' Perception: The Case of Limo Woreda, Hadya Zone of SNNPR, Ethiopia. MSc thesis, Addis Abeba University, Addis Abeba.

Simansky, V., & Kovacik, P. (2014): Carbon Sequestration and its Dynamics in water-stable Aggregates. *Agriculture*, 60, 1, 1-9.

Six, J., & Paustian, K. (2014). Aggregate-associated soil organic matter as an ecosystem property and a measurement tool. *Soil Biology and Biochemistry,* 68:A4–A9.

Six, J., Elliott, E.T. & Paustian, K. (2000). Soil macro aggregate turnover and micro aggregate formation: A mechanism for C sequestration under no-tillage agriculture. *Soil Biology and. Biochemistry,* 32, 2099-2103.

Six, J., Feller, C., Denef, K., Ogle, S.M., de Moraes Sa, J.C., & Albrecht, A. (2002). Soil organic matter, biota and aggregation in temperate and tropical soils— effects of no-tillage. *Agronomie*, 22:755–775.

Sochorec, M., Jandák, J., Raus, J., Kvasnovský, M., Hejduk, S., & Knot, P. (2015). Influence of different grassland management on water infiltration and soil physical properties. *Bulg. J. Agric. Sci.,* 21: 573–578

Stocking, M., & Murnaghan, N. (2000). Land Degradation—Guideline for Field Assessment. Overseas Development Group, University of East Anglia, Norwich.

Suttie, J.M., Reynolds, S.G., & Batello, C. (2005). Grasslands of the World. Food and Agricultural organization of the United Nations, Rome, Italy.

Tang, F.K., Cui, M., Zhou, J.X., Yan, S., & Ding, F.J. (2014). Difference analysis of soil organic carbon pool in different forestlands returned from farmlands in karst gorge area. *Soil Sci. Water Conservation*, 4, 1–7.

Theirfelder,C.,&Wall, P.C. (2012).Effect of Conservation Agriculture Techniques on Infi ltration

and Soil Water Content in Zambia and Zimbabwe. *Soil Tillage Re*search, 105(2), 217-227

Thurow, L.T., Blackburn, H.W. & Taylor, A.C. (1988). Infiltration and Interrill Erosion Responses to Selected Livestock Grazing Strategies, Edwards Plateau, Texas. *Journal of Range Management*, 41, 296. 10.2307/3899382.

Tiedemann, A.R. & Lopez, C.F. (2004). Assessing Soil Factors in Wildland Improvement Programs. In: Monsen, S.B., Stevens, R. and Shaw, N.L. (Compilers), *Restoring Western Ranges and Wildlands*, USDA Forest Service, Rocky Mountain Research Station, Fort Collins, General Technical Report RMRS-GTR-136-Vol. 1, 39-56.

Tisdall, J.M. & Oades, J.M. (1982). Organic matter and water-stable aggregates in soils. *Journal of Soil Science,*33, 141-163

Tsui, C., Tsai, C., & Chen, Z. (2013). Soil organic carbon stocks in relation to elevation gradients in volcanic ash soils of Taiwan. *Geoderma,* 209-210. 119-127. 10.1016/j.geoderma.2013.06.013.

Tuffour, H.O., Bonsu, M., & Khalid, A.A. (2014). Assessment of soil degradation due to compaction resulting from cattle grazing using infiltration parameters. *International Journal of Scientific Research in Environmental Sciences*, 2(4), 139-149.

Tuller, M., & Or, D. (2004). Water retention and characteristic curve. In: Encyclopedia of Soils in the Environment, Hillel, D. (Ed.). Elsevier Ltd., Oxford, U.K.

UNIC (2011). "Arid lands too degraded o provide subsistence are home to one billion of the world's poorest, hungriest people." Bulletin for World Day to Combat Desertification. 17 June, 2011.

USDA-NRCS, (2001). Rangeland Soil Quality: Water Erosion. Soil Quality Information Sheet.

USDA-NRCS, (2011). National Agronomy Manual. 190-V. 4th (ed).

Van den Bygaart, A.J., Kroetsch, J., Gregorich, E.G., & Lobb, D. (2012).Soil C erosion and burial in cropland. *Global Change Biology,* 18, 1441–1452

Van Schaik, L. (2009). Spatial variability of infiltration patterns related to site characteristics in a semi-arid watershed. *Catena*, 78:36–47.

Von Lutzow, M., Kögel-Knaber, I., Ekschmitte, K., Matzner, E., Guggenberger, G., Marschner, B. & Flessa, H. (2006). Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions –a review. *European Journal of Soil Science*, 57, 426-445

Walkley, A. & Black, I.A. (1934). An examination of the Degtjareff method for determining soil organic matter and a proposed chromic acid titration method. *Soil Science* 37:29-38.

Wang, G.X., Li, Y.S., & Hu, H.C. (2008). Synergistic effect of vegetation and air temperature changes on soil water content in alpine frost meadow soil in the permafrost region of Qinghai-Tibet. *Hydrological Processes*, 22: 3310–3320

Wang, Q., Wang, S., Xu, G., & Fan, B. (2010): Conversion of secondary broadleaved forest into Chinese fir plantation alters litter production and potential nutrient returns, Plant Ecol., 209, 269–278.

Wang, S., Wilkes, A., Zhang, Z., Chang, X., Lang, R., Wang, Y., & Niu, H. (2011): Management and land use change effects on soil carbon in northern China's grasslands: a synthesis, *Agriculture, Ecosystem & Environment*, 142, 329–340.

Wang, W., & Fang, J. (2009). Soil respiration and human effects on global grasslands. *Global and Planetary Change,* 67, 20–28.

Ward, R.C., & Robinson, M. (1989). *Principles of Hydrology*. McGraw-Hill, London. 459pp.

White, R., Murray, S., Rohweder, M. (2000). Pilot analysis of global ecosystems (PAGE): G*rassland ecosystem*. Washington, DC: World Resources Institute (WRI)

Wiaux. F., Cornelis, J.T., Cao, W., Vanclooster, M., & Van Oost, K. (2014). Combined effect of geomorphic and pedogenic processes on the distribution of soil organic carbon quality along an eroding hillslope on loess soil. *Geoderma*, 216:36–47

Wilke, B.M. (2005). Determination of chemical and physical soil properties. *In* R. Margesin & F. Schinner, eds. *Manual for soil analysis: monitoring and assessing soil bioremediation, XVI,* p. 72. Berlin, Heidelberg, Germany and New York, USA, Springer. 366 pp.

Wolf, K, (2011). Effects of high-density, short-duration planned livestock grazing on soil carbon sequestration potentials in a coastal California mixed grassland (Doctoral dissertation, California Polytechnic State University, San Luis Obispo).

Wondimagegn, M. (2019). An Overview of the Major Vegetation Classification in Africa and the New Vegetation Classification in Ethiopia. *American Journal of Zoology*, Vol. 2, No. 4, 2019, pp. 51-62. doi: 10.11648/j.ajz.20190204.12

World Meteorological Organization (2005). Climate and land degradation. http://www.wmo.int/web/agm/agmp.

Xu, W., & Wan, S. (2008). Water- and plant-mediated responses of soil respiration to topography, fire, and nitrogen fertilization in a semiarid grassland in northern China. *Soil Biology & Biochemistry*, 40(3):679–87.

Yang, W.Z., Wei, C., Lian, B., Dai, C.Y., & Chen, N.L. (2011) Soil fertility characteristics under different land-use patterns in Shiyanghe Basin. *Journal of Gansu Agricultural university*. 46, 112–117.

Yang, Y., Fang, J., Tang, Y., Ji, C., Zheng, C., He, J., & Zhu, B. (2008). Storage, patterns and controls of soil organic carbon in the Tibetan grasslands, *Global Change Biology*, 14(7), 1592–1599.

Yan-Gui, S., Xin-Rong, L., Ying-Wu, C., Zhi-Shan, Z., & Yan, L. (2013). Carbon fixation of cyanobacterial-algal crusts after desert fixation and its implication to soil organic matter accumulation in Desert. *Land Degradation & Development*, 24, 342–349.

Yoo, K., Amundson, R., Heimsath, A.M., & Dietrich, W.E., (2005). Erosion of upland Hillslope soil organic carbon: couplingfield measurements with a sediment transport model. *Global Biogeochemistry* Cycles 19 (3) (doi: Gb300310.1029/2004gb002271)

Young, H.S., McCauley, D.J., Helgen, K.M., Goheen, J.R., Otárola-Castillo, E., Palmer, T.M., & Dirzo, R. (2013). Effects of mammalian herbivore declines on plant communities: observations and experiments in an African savanna. *Journal of Ecology*, *101*(4), 1030-1041

Zehetner, F., & Miller, W.P. (2006). Erodibility and runoff-infiltration characteristics of volcanic ash soils along an altitudinal climosequence in the Ecuadorian Andes. *Catena* 65, 201–213.

Zhang, R. (1997). Infiltration models for the disk infiltrometer. *Soil Science Society of America Journal*, *61*(6), 1597-1603.

Zhang, Y., Carey, S.K., Quinton, W.L., Janowicz, J.R., Pomeroy, J.W., & Flerchinger, G.N. (2010). Comparison of algorithms and parameterisations for infiltration into organic-covered permafrost soils, Hydrology and Earth System Sciences, 14, 729–750

Zhao, J.H., Gao, C., Zhang, D.L. (2010). Study on the soil organic carbon density of alpine meadow with different degradation degrees in eastern Qilian Mountains. *Acta Agrestia Sin*. 18, 21–25.

Zheng, H., Chen, F.L., Ouyang, Z.Y., Tu, N.M., Xu, W.H., Wang, X.K., Miao, H., Li, X.Q., & Tian, Y.X., 2008. Impacts of reforestation approaches on runoff control in the hilly red soil region of Southern China. *Journal of Hydrology*, 356 (1–2), 174–184

Zhou, X., Lin, H.S, & White, E.A. (2008). Surface soil hydraulic properties in four soil series under different land uses and their temporal changes. *Catena* 73: 180–188.