INFLUENCE OF GRAZING MANAGEMENT PRACTICES ON VEGETATION COMPOSITION, SOIL ORGANIC CARBON AND GREENHOUSE GAS EMISSIONS IN SEMI ARID GRASSLANDS OF MAKUENI COUNTY, KENYA

Hillary Kipngetich Rotich (Bsc.)

A Thesis Submitted to Graduate School in Partial Fulfilment for the Requirement of the Degree of Masters of Science in Range Management in the Department of Land Resource Management and Agricultural Technology (LARMAT), University of Nairobi

DECLARATION

I hereby declare that this thesis is my original work and has not been presented in any other university

HILLARY KIPNGETICH ROTICH

Sign…………………………… Date…14/08/2018……

This thesis has been submitted with our approval as university supervisors;

Prof. RICHARD ONWONGA

Sign…………………………… Date…14.08.2018……

Department of Land Resource Management and Agricultural Technology,

University of Nairobi

DR. JUDITH S. MBAU

Sign…………………………… Date…15/08/2018……

Department of Land Resource Management and Agricultural Technology,

University of Nairobi

DR. OSCAR KIPCHIRCHIR KOECH

Sign…………………………… Date…14/8/2018……

Department of Land Resource Management and Agricultural Technology,

University of Nairobi

DR. DAVID PELSTER

Sign…………………………… Date…14/08/2018……

International Livestock Research Institute (ILRI), Nairobi

DEDICATION

This thesis is dedicated to my dear mother Mrs. Jane Samoei and my family members for their endless support during my study. Thank you!

ACKNOWLEDGEMENT

I sincerely thank the Australian government through the System for Land Based Emission Estimation in Kenya (SLEEK) initiative for funding my master studies. I also acknowledge the International Livestock and Research institute (ILRI) through Dr. Merbold Lutz and Dr. David Pelster for their great support in terms of greenhouse gas measurements and analysis.
My heartfelt appreciation goes to my university supervisors and mentors Dr. Richard Onwonga, Dr. Judith S.Mbau and Dr. Oscar Koech for their sustained guidance and mentorship and positive criticism of my work. I feel greatly indebted to them for making this study a success.

I also wish to acknowledge Prof. Jesse Njoka for the fatherly advice during the hard times of my academic journey; to me you are more of a father than a lecturer. Thanks to the Erasmus + Program through Dr. Judith S. Mbau, Dr. Stephen Mureithi and Prof. Jesse Njoka, which gave me the conducive environment at Jagiellonian University-Krakow Poland to write this thesis, God bless you all.

I wish to acknowledge Dr. Oscar K Koech for facilitating the greenhouse gas measurement workshop at Aberyswyth University in United Kingdom; the workshop equipped me with measurement techniques which made my field work easy.

To my classmates Stanley Jawuor and Joseph Ombega and my friends Felix, Bernice, Collins, Gitau, Janet and Harrison, I greatly appreciate your moral and spiritual support.

Above all, I acknowledge and thank the Almighty God for seeing me through this enormous journey. Your mercies and support gave me strength throughout this study. Indeed you are the Lord my God, always walking with me.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS AND ACRONYMS

ASAL	Arid and Semi-Arid Lands
CBD	Convention for Biological Diversity
FAO	Food and Agricultural Organisation
GC	Gas Chromatogram
GHG	Greenhouse gas
GOK	Government of Kenya
IPCC	Intergovernmental Panel for Climate Change
ILRI	International Livestock and Research Institute
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SSA	Sub Saharan Africa
UN	United Nations
UNDP	United Nations Development Programme
UNEP	United Nations Environmental Programme

GENERAL ABSTRACT

Rangeland degradation threatens the productivity of rangelands, soil organic carbon storage and offset of greenhouse gasses. A key component for sustaining production in rangeland ecosystems is the maintenance of soil organic matter (SOM), which is strongly influenced by land management. Many management techniques intended to increase forage production may potentially increase SOM, thus sequestering atmospheric carbon (C) and other greenhouse gasses such as nitrous oxide and methane. Effective management activities require an understanding of the impact of livestock grazing on these ecosystems. The purpose of this study was to investigate the influence of grazing management practices on vegetation composition, soil organic carbon and greenhouse gas emissions. The study was conducted on a commercial grazing land. The study followed a pseudo-replication design in which there were three treatments: (1) continual grazed, (2) rotational grazed and (3) and ungrazed Vegetation attributes, soil parameters and GHG fluxes were measured in each of the three sites. Soil samples for measurement of soil organic carbon concentration were taken up to a depth of 1.2m, at an interval of 0-10, 10-20cm, 20-30cm, 30-60cm, 60-90cm and 90-120cm. Greenhouse gas samples were collected using the static chamber method, and carried out for a period of three months covering the dry and wet season as well as a transition period.

The herbaceous biomass production was significantly higher ($p \leq 0.5$) in rotationally grazed areas compared to both continually grazed and ungrazed areas. Percentage plant cover was significantly higher under rotationally grazed areas compared to both continually grazed and ungrazed areas which were not significantly different. There was a significant difference in plant species richness between the three grazing management practices ($p \leq 0.05$). The plant species richness was highest in rotationally grazed areas, followed by continually and ungrazed areas with mean species numbers of, 13.4±1.82, 11.9±1.74 and 9.67±2.24, respectively. Rotationally grazed site had significantly higher species diversity followed by continual and ungrazed area with mean values of 3.08±0.65, 2.88±.072 and 2.43±0.02, respectively. The ungrazed site recorded significantly ($p \leq 0.05$) higher soil organic carbon concentrations than rotational and continuously grazed sites for all soil depths up to 1.2m depth. Cumulative soil CO_2 fluxes were highest from continual grazing system, followed by rotational grazing and lowest in ungrazed area, having 2357±123.9, 1285±123.9 and 1241±143 CO_2 (kg ha^{-1} 3 month), respectively. The N_2O and CH_4 fluxes were also highest in continual grazing and lowest in ungrazed site, with 677.9±130.1, 208.6±127.3 and 162.2±150.3 (gm ha^{-1} 3 month) and CH_4, 232.7±126.6, 173.1±126.6 and 80±46.2 (gm ha^{-1} 3 month), respectively. The rotationally grazed site showed enhanced plant species biomass production, percentage cover and species diversity compared to the continuously grazed area. Soil organic carbon concentrations were significantly higher at the ungrazed site followed by rotationally grazed and continual grazing system respectively. Greenhouse gas emissions were significantly higher in continuously grazed area with ungrazed site having the lowest emissions. From the above findings, study has demonstrated that rotational grazing management system has the potential to significantly improve the range productivity, increase soil organic carbon and reduce emissions of greenhouse gasses.

CHAPTER ONE

GENERAL INTRODUCTION

1.1 Background

Globally, rangelands occupy more than half of the world's land area and about 69% of the world's agricultural land (Faostat, 2009). In Kenya, rangelands cover approximately 88% of the land mass. Apart from being vital for domestic livestock production, rangelands are also important habitats for wild flora and fauna (Riisgaard and Halberg, 2010; Galvin, 2009). The major use of rangelands is for livestock production, mainly through pastoralism and a few commercial farms. An estimated 25 million pastoralists and 240 million agropastralists primarily depend on livestock production for their income (FAO, 2009).

Livestock grazing is the central component of land management (Galvin, 2009). The decline of rangeland resources compromises the traditional mobility within the pastoralist system (Gao *et al*., 2009; Maestre *et al*., 2009). The ability of dryland communities to cope with the challenges of complex and dynamic system is usually undermined by rangeland degradation. Rangeland's natural resources are experiencing rapid degradation, thus reducing their contribution to livestock feed supply. Pastoral systems are losing resilience as traditional coping mechanisms are failing due to increasing environmental degradation coupled by absence of national policies to address the problem (Kassahun *et al*., 2008). These lands also contributes a significant percentage to the biosphere-atmosphere exchange of the three greenhouse gasses, with fluxes closely linked to rangeland management practices (Soussana *et al.*, 2010).

Rangelands are exploited by livestock through grazing which influences the sustainability of grazing lands with respect to grazing intensity (Kioko *et al*,2012). Specifically, high grazing intensity have a huge effects on the botanical composition and species diversity of the grazed pasture by depressing the vigor of dominant species, consequently leading to colonization by highly competitive and tolerant plant species (Kgosikoma, 2012). When the duration and frequency of grazing and the rest periods are controlled, rotational grazing can improve the state and health of the range and also biomass production (Jacobo *et al.*, 2006). Proper utilization creates environmental conditions that prevent the survival of invasive weed species, while favouring recruitment and survival of palatable forage/browse species thus increases forage quality. Livestock through grazing affects the composition and productivity of plants through change of plant nativity, recruitment, and mortality (Adler *et al*, 2005) .The effect as a result of grazing influence may cause changes in community structure and function (Fortin *et al*, 2003). An ecosystem may maintain its stability and resistance to changes produced by grazing up to a certain threshold beyond which further changes increasingly being accentuated by stochastic abiotic factors such as rainfall. Therefore, management of plant communities for sustainable livestock production requires seasonal integration of information on plant species composition and production across the expansive and often heterogeneous rangelands.

Soil forms the largest terrestrial reservoir of carbon (C) and can store about three times as much C as the atmosphere can sequestered mainly in decomposed plant litter and residues. Global estimation shows that, rangelands have the potential to sequester carbon in the soil at the rate of 0.5 petagrams (Pg.), hence they play vital role in managing global warming. The net global warming potential from greenhouse gas exchanges within savannah grasslands is currently not known and it is also hard to determine the net carbon (C) balance because the annual grassland carbon sequestration is matched by a correspondingly large C loss through respiration. The type and condition of the ecosystem i.e. species composition, soil, management and climate dictates C sequestration potential of grasslands (Conant *et al.*, 2001; Follett and Reed, 2010). Soil and vegetation management could be a viable option to conserve and sequester C and hence mitigate atmospheric accumulation of CO_2 (Post and Kwon, 2000), although improved estimates of C balances are also required. Recently, rapid losses of soil C associated to intensive livestock grazing have been reported for tropical savannas. Rangeland management modifies Soil organic carbon storage (Conant *et al*., 2001; Schuman *et al*., 2002), potentially increasing C sequestration (Follet *et al*., 2001). The management of this lands primarily affects soil organic carbon storage by modifying C inputs to the soil, including root turnover and C allocation between roots and shoots (Ogle and Reynolds, 2004).

Previous studies done on impacts of grazing shows that repeated grazing reduces plant growth and productivity, whereas light-to-moderate grazing cause suppression of growth with occasional growth enhancement (Green , 2000). Selective defoliation modifies species composition, that often leads to low productivity and undesirable plant compositions. The livestock hoof action deteriorates soil aggregate stability and compact the soil surface through trampling and treading leading to increase in the soil bulk density. The unfavourable changes in soil physical properties may cause a decline in water infiltration and root growth. Livestock also influences the soil biogeochemical processes through the addition of nutrients in the form of faecal and urine. Barger in their studies reported that, grazing has the potential to influence rangeland carbon dynamics by altering plant litter chemistry (Barger *et al*., 2004), litter production, plant biomass allocation patterns, and the spatial distribution of nutrients (Evans *et al*, 1997). Furthermore, other study reported that, depending on the intensity, grazing pressure may slow decomposition rates by decreasing plant litter carbon to nitrogen (C: N) ratio, or due to decreased standing biomass, may accelerate the decomposition by increasing soil temperature (Welker *et al*., 2004).

Despite the above mentioned rangeland carbon dynamics in response to grazing, most results from previous research showed a wide variation ranging from positive to negative to no response and little has been done on grazing rangeland soils. Therefore, there is need to document the impact of grazing management practices on soil organic carbon stock in rangelands.

The net global warming potential from greenhouse gas exchanges within savannah grasslands is currently not known and it is also hard to determine the net carbon (C) balance because the annual grassland carbon sequestration is matched by a correspondingly large C loss through respiration. Soil and vegetation management could be a viable option to conserve and sequester C and hence mitigate atmospheric accumulation of CO_2. Rangelands have been estimated to act as a small C sink but there are still extremely large uncertainties (Janssens *et al.,* 2003). Nitrous oxide production in soils occurs as a result of both the oxidative process of nitrification and the reductive process of denitrification (Davidson *et al*,, 1991; Granli *et al.*, 1994). Soil physical properties such as coarse soil texture usually promote nitrification; on the other hand, fine soil texture and high organic C content promote denitrification, where sometimes the two processes go on simultaneously within soils (Davidson, 1991). Rangelands can also be a sink of CH_4 (Ambus *et al.*, 1995; Mosier, 1998). Smith in his studies reported that, in aerobic soils, CH_4 is taken up through oxidation by methanotropic bacteria contributing to about 5.8% to the global atmospheric CH_4 sink (Maljanen *et al,* 2003; Scholes *et al.,* 2003)

GHG fluxes from grasslands ecosystems are intimately linked to grazing management. In grasslands, CO_2 is exchanged within the soil and vegetation, N_2O is emitted by soils and CH_4 is emitted by animals and exchanged with the soil. When CO_2 exchange with vegetation is included on net GHG exchange calculation, these ecosystems are usually considered GHG sinks (Allard *et al*., 2007; Soussana *et al*., 2007).

The choices taken when managing grasslands to reduce GHG budget may involve important trade-offs. Allard et al. (2007) in their studies of greenhouse gas net exchange from grasslands including CO_2 exchange with the vegetation observed a net CO_2 equivalent sink activity, but with different trade-offs. Enteric emissions strongly affected greenhouse gas budget in intensively and extensively managed grassland with an average of 70% offset of total CO_2 sink as observed by Allard et al. (2007). On the other hand, Soussana et al., (2007) observed that addition of CH_4 and N_2O emissions from pasture soils to CO_2 sink activity of grasslands resulted in relatively small offset of total CO_2 sink activity (19% average), though the small trade-off but was not enough to affect the CO_2 equivalent sink potential of the sites studied.

With emission reduction targets in place at the present, measures to reduce greenhouse gas emissions through land management are being investigated as a potential means to mitigate climate change. Driven by this, there is a strong demand from policy-makers for simple emission factors, which quantify the effect of land management activities on the net emission of greenhouse gases, but as yet these are poorly developed (Couwenberg *et al*., 2011). As part of this, there is need for a better understanding of the vegetation composition, vertical distribution of soil organic carbon and emissions of methane, carbon dioxide and nitrous oxide from grassland soils in response to the type of grazing management regimes in rangelands.

1.2 Statement of the problem

The decline of productivity due to degradation and reduced ecological function of rangeland ecosystems due to human activity, are ecological and environmental problems recognised world-wide to have far-reaching implications in global warming and reduced rangeland productivity (Dlamini *et al*. 2015). Livestock exploits rangelands through grazing which influences the sustainability of rangelands (Kioko *et al*,2012). Specifically, high grazing intensity have a huge effects on the botanical composition and species diversity of the grazed pasture by depressing the vigor of dominant species, consequently leading to colonization by highly competitive and tolerant plant species (Kgosikoma, 2012). Overgrazing is one of the primary contributors to grassland degradation around the world, through reduction in vegetation cover, degradation of topsoil, causing soil compaction as a result of trampling, reduction in soil infiltration rates, and enhancement of the susceptibility of soils to erosion (Hilker *et al*. 2014). The present understanding of GHG emissions in Africa is particularly limited when compared to the potential of the continent as both a GHG sink and source. Lack of data on GHG emissions from African natural grasslands hinder the progress of our understanding of GHG emissions in the continent (Bombelli *et al*., 2009; Valentini *et al*., 2014). In order to identify mitigation measures and other climate smart interventions for the region, there is urgent need to quantify baseline GHG emissions, as well as understand of the impacts of deterrent land-use management strategies on GHG emissions (e.g., Palm *et al*., 2010). Livestock grazing is one of the most effective strategies to prevent grassland degradation and to support sustainable grazing (Wiesmeier *et al.* 2012). However, response of rangeland ecosystems to livestock grazing is mediated by the type of grazing management strategy employed (Butof *et al.* 2013). Therefore, it is important to take into account the influence of grazing management practices on vegetation, soil organic carbon and greenhouse gas emissions into account when making recommendations for proper grazing management practices to be adopted.

1.3 Justification

The understanding grazing management effects on GHG emissions and the soil organic carbon stocks is important to provide effective management options in the rangelands. This may contribute to improving productivity as well as managing the impacts of climate change.

In order to identify mitigation measures and other climate smart interventions for the Country, it is important to quantify baseline GHG emissions, as well as understand the impacts of different grazing management strategies on GHG emissions. The ability of rangelands to sustainably support production depends on the amount of soil organic carbon, where livestock grazing can either increase or reduce the amount of carbon in the soil. Rangeland plant composition and productivity is influenced by livestock through grazing. The effect as a result of grazing causes changes in plant community structure and function. Therefore, management of plant communities for sustainable livestock production requires seasonal integration of information on plant species composition and production across the expansive and often heterogeneous rangelands. There is need to identify the appropriate grazing management practices that will increase soil organic carbon stocks and hence increasing rangeland productivity while offsetting greenhouse gas emissions. The present study findings will contribute towards guiding policy formulations for sustainable rangeland management at both county and national levels of governance.

1.4 Study Objective

1.4.1 Overall Objective

To contribute to sustainable rangeland management through evaluation of soil organic carbon stocks, vegetation composition and GHG emissions in different grazing management regimes for climate change mitigation.

1.4.2 Specific Objectives

1. To evaluate the influence of grazing management practices on plant species composition.
2. To determine the influence of grazing management practices on soil organic carbon stocks.
3. To predict the long term impact of grazing management practices on soil carbon stocks.
4. To determine the influence of grazing management practices on Methane, Nitrous oxide and Carbon dioxide emissions.

1.5 Hypothesis

1. There is no effect of grazing management practices on plant species composition
2. Grazing practices has no influence on soil organic carbon stocks.
3. There is no significant relationship between different grazing regimes and future accumulation of soil organic carbon stocks.
4. There is no significant relationship between grazing management practices and methane, Nitrous oxide and carbon dioxide emissions.

1.6 Outline of the Thesis

This thesis has seven chapters. Chapter one presents the general introduction, problem statement, justification, broad and specific objectives, outline of the thesis and key definitions. Chapter two discusses the general literature review while chapter three contains the general materials and methods. In chapter four, the influence of grazing management practices on plant species diversity is presented and discussed. Chapters five and six outlines the influence of grazing management regimes practices on soil organic carbon and the influence of different grazing management practices on methane, Nitrous oxide and carbon dioxide emissions, respectively. The general conclusions and recommendations are listed in Chapter seven.

1.7 Definition of key terms in this thesis

Drylands: Regions where the mean annual precipitation to potential evapotranspiration ratio is less than 0.65 (Middleton and Thomas, 1997).

Rangeland: Land with renewable, multiple use natural resources to both animals and humans and on which native vegetation, predominantly grasses, grass-like plants, forbs, or shrubs which are suitable for grazing or browsing use grow (Moghaddam, 2000).

Grazing management: refers to the manipulation of livestock grazing to accomplish a desired result.

CHAPTER TWO: LITERATURE REVIEW

2.1 Extent and importance of rangelands

Rangelands make approximately 50% of the global landmass and about 69% of the world's agricultural land (Faostat, 2009). Apart from being vital for domestic livestock production, rangelands are also important habitats for wild flora and fauna (Riisgaard, and Halberg, 2010; Galvin, 2009). The major use of rangelands is for livestock production, mainly through pastoralism. An estimated 25 million pastoralists and 240 million agropastralists primarily depend on livestock production for their income (FAO, 2009). Livestock movement is the central component of land management (Galvin, 2009). The decline of rangeland resources compromises the traditional mobility within the pastoralist system, with some of the obstacles to pastoral mobility being: loss of grazing land to agriculture, poor management of watering point management, conflicts and insecurity, shifting boundaries (county, national and regional), and social change necessitated by changing human aspirations and economic needs (Gao *et al*., 2009; Maestre et al., 2009).

The ability of communities living in rangelands to cope with the challenges of complex and dynamic system is usually undermined by rangeland degradation. Rangeland's natural resources are experiencing rapid degradation, thus reducing their contribution to livestock feed supply. Pastoral systems are losing resilience as traditional coping mechanisms are failing due to increasing environmental degradation coupled by absence of national policies to address the problem (Kassahun *et al*., 2008).

2.2 Livestock grazing and rangeland soil carbon pool

Soil forms the largest terrestrial reservoir of carbon (C) and can store about three times as much C as the atmosphere can sequestered mainly in decomposed plant litter and residues. Recently, rapid losses of soil C associated to intensive livestock grazing have been reported for tropical savannas. Milchunas, (2006) reported that the impact of grazing management on the soil biogeochemical processes that regulate rangeland carbon dynamics is not well understood due to heterogeneity in grassland types. In his study, out of the 34 data sets he evaluated to compare soil carbon of grazed and protected areas, about 40% indicated an increase in soil carbon due to grazing and about 60% showed a decrease or no response to grazing (Milchunas, 2006). Studies done previously showed that the impact of grazing on ecosystem processes is influenced by the extent of the removal of photosynthetic biomass (defoliation), which is determined in part by grazing intensity; treading and trampling and faecal and urine depositions (Heitschmidt *et al*., 2004).

The livestock hoof action deteriorates soil aggregate stability and compact the soil surface through trampling and treading leading to increase in the soil bulk density. The unfavourable changes in soil physical properties may cause a decline in water infiltration and root growth. Livestock also influences the soil biogeochemical processes through the addition of nutrients in the form of faecal and urine. Barger in their studies reported that, grazing has the potential to influence rangeland carbon dynamics by altering plant litter chemistry (Barger *et al*., 2004), litter production, plant biomass allocation patterns, and the spatial distribution of nutrients (Evans *et al*, 1997).

Despite the above mentioned rangeland carbon dynamics in response to grazing, most results from previous research showed a wide variation ranging from positive to negative to no response. For instance, Gill (2007) in his evaluation of the influence of 90 years of protection from grazing on carbon dynamics in subalpine rangeland, reported that livestock grazing had no significant impacts on total soil carbon or particulate organic matter, although active soil carbon content increased and that the loss of carbon from the active carbon pool was higher in grazed sites than in ungrazed areas. The implication of these results is that grazing may convert the relatively recalcitrant carbon pool into easily mineralizable carbon fraction.

Conant *et al* (2003) in their study done in pastures of Virginia, USA, showed that the major factor controlling soil organic carbon dynamics is actually grazing management, they found that soil organic carbon averaged 8.4 Mg C ha^{-1} more under intensive management or short rotation grazing than extensively grazed or hayed sites. Naeth *et al*. (1991) observed a negative impact on soil organic matter with heavy intensity or early season grazing, as compared to light intensity or late season grazing in the grasslands of Alberta, Canada. Heavy grazing resulted in significant reductions in height of standing and fallen litter, and a decrease in live vegetative cover and organic matter mass. Consequences of increasing atmospheric CO_2 concentration can affect rangeland carbon storage by accelerating the photosynthesis rate, which increases biomass production, and lowers the decomposition rate (Ingram *et al*., 2008; Svejcar *et al*., 2008). Therefore an appropriate grazing management regime needs to be adopted in order to offset the increase of atmospheric CO_2

2.3 Soil Organic Carbon Storage and Sequestration

Soil organic carbon pool has been widely appreciated in the global C cycle (Jones *et al.*, 2005; David Powlson, 2005). The SOC pool is comprised of animal and plant residues at various stages of decomposition, chemical and microbiological breakdown products, and the bodies of microorganisms and small animals (Lal, 2008). Carbon dioxide emissions from rangeland practices are caused by removal of plant species biomass, increases in the mineralization rates due to changes in soil temperature and moisture and losses by leaching and erosion (Lal, 2003). However, under appropriate management, the C in the soil's biomass can be permanently stored becoming a form of C sequestration (Lal, 2008).

Besides the climate change mitigation, terrestrial C sequestration has a series of ancillary benefits (Lal, 2008). Organic C plays a crucial role in preventing and mitigating soil degradation. The importance of soil organic carbon on soil structure are many; soil structure is affected by increased soil organic carbon directly through the stabilization of aggregates and indirectly through increased aeration, increased infiltration rates, increased water holding capacity and decreased surface crusting (Craswell and Lefroy, 2001; Martius *et al*., 2001).

The estimates of soil organic carbon storage globally range from 1200 to 1600pg in the top 1m soil depth, which is mostly the case in arid and semi-arid regions (Díaz-Hernández *et al.*, 2003). Below the 1m depth profile, soil carbon storage is rarely estimated at the natural landscapes (Li *et al*., 2007). The big question which remains answered is how much carbon is underestimated in global budgets below the first meter with respect to grazing management practies. It was estimated that, there is a 60% increase in the global soil organic carbon (SOC) storage with depth extended to 2.0m (Batjes, 1996). A recent estimate shows that there is a 56% increase in soil organic carbon storage at global level when the third meter of soil is also included in the estimates (Jobbágy and Jackson, 2000). Mounting the global estimates of deep soil organic carbon storage reflected an increasing need for an understanding of the importance of deep soil carbon (Veldkamp *et al* 2003,).

2.4 Effects of grazing practices on vegetation composition and aboveground biomass production

Rangeland ecosystems are largely impacted by land uses (Conant *et al*., 2001), which have a significant effect on the environment globally. Livestock exploits rangelands through grazing and grazing intensity influences the sustainability of grazing lands (Kioko *et al*,2012).

Specifically, high grazing intensity have a huge effects on the botanical composition and species diversity of the grazed pasture by depressing the vigor of dominant species, consequently leading to colonization by highly competitive and tolerant plant species (Kgosikoma, 2012).

Livestock through grazing affects the composition and productivity of plants through change of plant nativity, recruitment, and mortality (Adler *et al*, 2005) .The effect as a result of grazing influence may cause changes in community structure and function (Fortin *et al*, 2003). An ecosystem may maintain its stability and resistance to changes produced by grazing up to a certain threshold beyond which further changes increasingly being accentuated by stochastic abiotic factors such as rainfall. Proper utilization creates environmental conditions that prevent the survival of invasive weed species, while favouring recruitment and survival of palatable forage/browse species thus increases forage quality. Therefore, management of plant communities for sustainable livestock production requires the use of proper grazing management practices across the expansive and often heterogeneous rangelands.

2.5 Grazing practices and greenhouse gas emissions

The influence of grazing on the productivity of rangelands, greenhouse gas emissions and soil organic carbon content are variable and depend on local climate, soil, topography, plant community, and grazing timing and intensity (Jackson et al, 2007) . In general, the livestock are the greatest sources of greenhouse gases from rangelands (Beauchemin et al, 2010). Grazing can change the physical characteristics of soil through compaction or otherwise physically disturb soils, particularly when animal densities are high for prolonged periods and when soils are saturated (Greenwood and McKenzie, 2001). Compaction leads to decreased plant growth and thus C uptake from the atmosphere. Grazing through soil compaction can also affect the soil hydrologic cycle by decreasing infiltration rates, lowering soil aeration, and altering the composition and diversity of soil biota consequently leading to promotion of CH4 and N2O production (Asner et al., 2004). Grazing animals can also destroy surface soil structure, disturb surface crusts, and create micro-topography (Trimble and Mendel 1995), which can increase soil C losses through erosion agents such as wind and water.

The structure and function of plant communities can be directly affected by grazing through selective plant removal, defoliation, and changing the amount and composition of residual biomass (Ingram *et al.,* 2008; Rook *et al.*, 2004). The amount of defoliation also affects subsequent forage production by changing light competition and residual biomass (Collins et al., 1998). When vegetation is removed, soil moisture decreases and temperature increases (Asner *et al*., 2004), which can lead to increased decomposition at the soil surface and less transfer of plant litter into the soil organic matter pool. Finally, grazers through their excrement redistribute nutrients. The dung and urine patches usually concentrate nutrients and act as hotspots of organic matter deposition, with impacts on plant community composition and growth (Veldkamp *et al*., 2003). Greenhouse gas emissions from soils can also be affected by livestock due to differences in animal size and behaviour, manure production and composition and grazing preferences (Mekuria and Veldkamp, 2012), However, there are no current studies reporting the influence of grazing management practices on greenhouse gas emissions. It is against this background that the current study therefore attempts to estimate the emissions of GHG from soil under different grazing management systems, the influence of these grazing systems on vegetation and the vertical distribution of soil organic carbon and then predict the long term impact of grazing systems on SOC.

CHAPTER THREE: MATERIALS AND METHODS

3.1 STUDY AREA

3.1.1 Location

The study was conducted in Yaoni commercial ranch located in Makueni County, approximately 125 km southeast of Nairobi, Kenya (Figure 1).). This is a commercial grazing ranch in south eastern rangelands of Kenya. Makueni County borders Kajiado to the West, Taita Taveta to the South, Kitui to the East and Machakos to the North. It lies between Latitude 1º 35´ and 30^0 00’ South and Longitude 37º10´ and 38º 30´East. The county is largely arid and semi-arid and usually prone to frequent droughts. The dominant vegetation type is grasses with scattered acacia species shrubs. The ranch has been used for livestock (cattle) grazing for the last 70 years under both rotational and continuous grazing system.

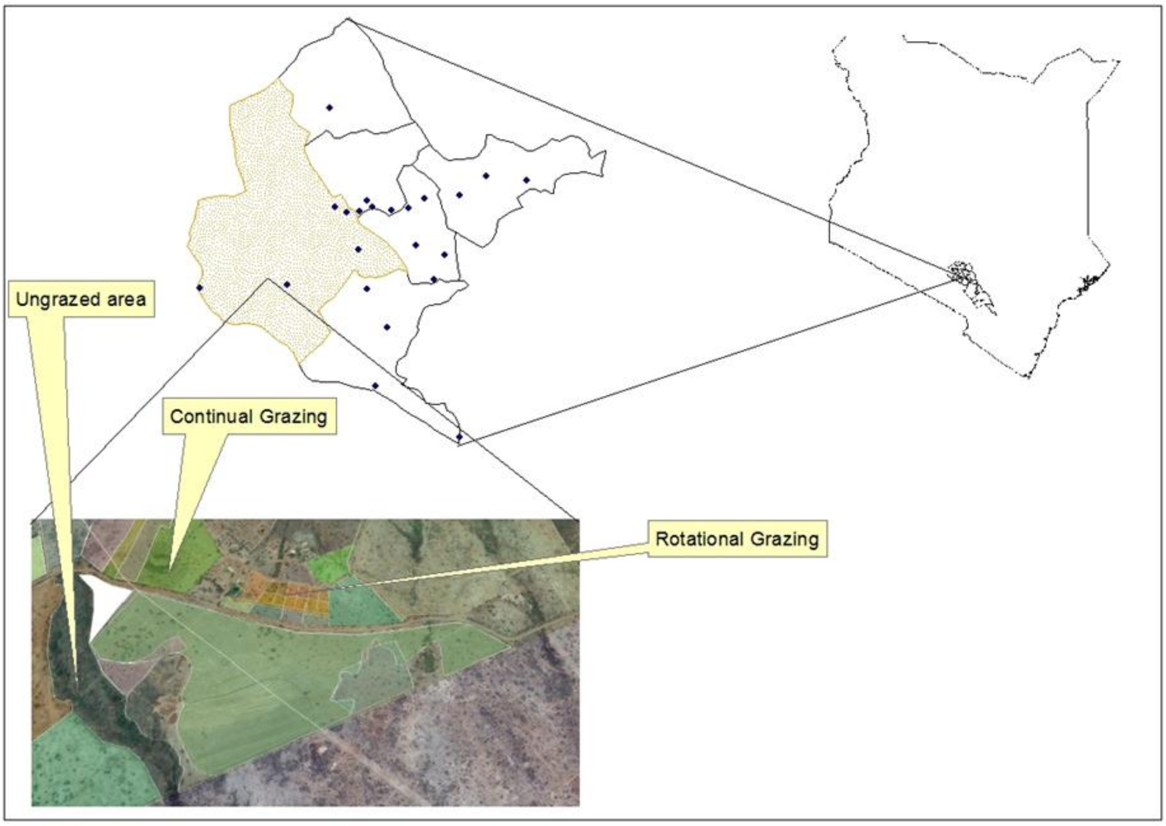


Figure 1: Map of the study area in relation to map of Kenya

3.1.2 Agro-economic potential/economic activities

Yohani falls under agro-ecological zone IV and V (Jaetzold *et al*., 2006). In terms of agro-ecological potential, Jaetzold et al. (2006), classifies the area as a ranching zone naturally suited for extensive livestock production and wildlife utilization.

3.1.3 Climate

Yohani ranch lies at an altitude of between 1200-1400 metres above sea level. Rainfall and temperatures are influenced by altitude. Rainfall is bimodal with long rains falling between the months of March to May and short rains in October to December. Total annual rainfall is between 400 and 600mm (KMS KMD/FCST/5-2016/SO/01). In between the rainy seasons, the area experiences intervening dry spells in January/February as well as July to September. Provide temperature ranges with seasonality variations.

3.1.4 Topography, geology and soils

The terrain is characterized by plains to the North, undulating hills to the South. The geology of the study area is characterized by relatively deep over-burden, with very few exposures of the underlying basement rock. The basement system are crystalline rocks of pre-cambrian age often occurring as fine-grained schists and course gneisses, that have been invaded by pink quartzo feldspathic pegmatites (Kurrent Technologies, 2011).

3.1.5 Natural vegetation

The study area is mainly covered by savannah woodland with a significant woody plant component. The natural vegetation of the study area consists of predominantly, *Themeda triandra*, a tufted perennial grass species that is preferred by grazers, and *Themeda -Balanites* or Themeda-Acacia wooded grassland (Kinyua *et al.*, 2000).

3.2 Experimental design and Treatments

The study followed a pseudo-replication design involving two grazing systems and ungrazed area: continuous grazing, rotational grazing and ungrazed area (control A section of the ranch was converted from continuous grazing into rotational grazing for the last six years at the time the study was conducted. The second sampling block was an area that has been under continuously grazing for the past 30 years. Under the rotational grazing, cattle are moved between paddocks after a period of two months. The ungrazed area consist of an abandon land area that has not been grazed for more than 30 years due to a deep gully due to erosion creating and isolation that is inaccessible by livestock as shown in (Figure 1). The specific research methodology per objective are presented in the independent papers presented as chapters below.

CHAPTER FOUR: VEGETATION COMPOSITION IN RELATION TO GRAZING MANAGEMENT PRACTICES IN SEMI-ARID LANDS, MAKUENI COUNTY, KENYA

Abstract

Livestock grazing practices in the rangelands are being recognized as management tools for environmental protection and increased livestock productivity. However, improper livestock grazing practices exploits rangelands hence threatening the sustainability of these ecosystems and the livelihoods of the pastoral. Overgrazing of the rangeland ecosystem has been largely reported to reduce pasture productivity and increase environmental degradation. The aim of this study was to evaluate the influence of grazing practices on vegetation composition in Yoani, a commercial grazing land within Makueni County. The study followed a pseudo-replication design in which there were three treatments: (1) continual grazed, (2) rotational grazed and (3) and ungrazed. Shannon-Wiener's diversity index was used to determine species diversity. The Rotationally grazed site had significantly higher ($p \leq 0.05$) herbaceous biomass production compared to both continually grazed and ungrazed areas. Biomass yield was significantly higher ($p \leq 0.05$) in continual grazing than the ungrazed site. Percentage cover of herbaceous plants was significantly higher under rotationally grazed areas compared to both continually grazed and ungrazed areas which were not significantly different. There was no significant difference in plant species richness between the three sampling blocks ($p \leq 0.05$). However, plant species richness was highest in rotationally grazed areas, followed by continually and ungrazed areas with mean species numbers of 13.4±1.82, 11.9±1.74 and 9.67±2.24, respectively. The rotationally grazed site had higher species diversity followed by continual and ungrazed area with mean values of 3.08±0.65, 2.88±.072 and 2.43±0.02 respectively.

Key words: Plant Biomass, Plant Cover, Species Diversity and Richness, Grazing management.

4.1 Introduction

Globally, properly managed grazing lands incorporates the most important land use practices (Liebig *et al*., 2006) and covers about 25% of earth's land surface (Asner *et al*., 2004; Einstein, 2010). The significant area of rangelands globally makes them a useful resource for grazing, biodiversity conservation and a source of livelihood, specifically for rural communities (Ericksen *et al*., 2009). Most of the global's rangelands are believed to be degraded as a result of excessive livestock grazing (Milton *et al*, 1994). Livestock grazing influences the plant community structure and ecosystem functioning which is a key issue in the management of rangelands in order to maximize livestock production and sustainability (Jacobo *et al*, 2006). Evidence exists to the effect that livestock grazing strongly influences the structure, richness, and composition of plants in rangelands (Porqueddu *et al.*, 2016;), for instance, the short term effects of grazing have been reported to influence the structure of plant communities through defoliation and reduction of plant tissues while causing changes in botanical composition and species diversity in the long term through selective grazing (Jacobo *et al.*, 2006). Changes in plant species composition are mostly due to the substitution of palatable by unpalatable species ,this have been observed to lead to an increase in annual plants species following rangeland degradation (Tarhouni *et al*, 2007), as well as a decline in plant species diversity and rangeland productivity (Cingolani *et al*, 2005). When unpalatable species become dominant, it becomes difficult to reverse the effects hence lowering the rangeland productivity (Westoby *et a*l, 1989).Rotational grazing may be used as a useful management method to preserve species diversity and rangelands productivity (Gamoun, 2014). It is also a preferred practice for conserving biological soil crusts and the ecological services they provide in nitrogen fixation and soil stabilization (Liu *et al*., 2009). Moreover, low to light grazing intensity can increase production compared to no grazing. However, the extent of grazing may affect the photosynthesis process which maximizes manufacturing of plant food and depends on the growing conditions of the harvesting stage within the seasons hence there is need to use a proper grazing management strategy (Patton *et al*, 2007). Furthermore, Grazing influences soil properties and hydrology, which may result to critical outcomes affecting plant growth and productivity in the rangelands where water scarcity is a common phenomenon (Jeddi *et al*, 2010).

Rangelands in the tropics are highly dominated by Savanna grasslands with most grass species being highly tolerant to grazing, however, the common high grazing intensity coupled with frequent droughts increasingly lead to shift in species composition and decline in soil fertility and biomass productivity (Van Auken, 2009). Poor grazing practices leads to overgrazing which negatively influence the botanical composition and species diversity. Continuous grazing results to increase in more competitive and drought tolerant grass species but of low feed value to animals while selective grazing of palatable herbaceous vegetation by grazing animals encourages the establishment of annuals and unpalatable plant species (Fensham *et al*, 2010). Rangeland vegetation does not always respond in a linear way to grazing intensity, partly because local environmental conditions such as high rainfall and soil fertility regulate the plants' ability to cope with grazing pressures. However, herbaceous biomass appears to be more responsive to differences in grazing intensities across grazing management systems. Well managed rangelands usually exhibits a higher herbaceous biomass production than the poorly managed grazing lands , which could be due to the higher grazing intensities of the latter compared with the former .

The concept of rangeland management in Kenya has become widely accepted and implemented. When natural vegetation becomes severely degraded, management with an aim of increasing productivity has been proven unlikely. However, this situation can be remedied if restoration work is undertaken (Le Houérou 2002; Gamoun *et al*. 2012), therefore rangeland protection becomes necessary for sustainable management and increased resilience

(Gamoun *et al*. 2011; Gamoun 2014). In practice, grazing management is simply controlling where and when animals graze over the landscape (Norton *et al.* 2013). Additionally, it is the manipulation of the soil–plant–animal complex of the grazing land in pursuit of a desired result (Allen *et al.* 2011). The effects of grazing management practices such as controlled stocking densities and grazing systems on plant species diversity and plant biomass production or functional groups may have important consequences for ecosystem function (Hickman *et al.* 2004). Although a lot of studies have been done on the impact of grazing on vegetation dynamics, we did not find literature on studies comparing the impact of different grazing management systems on vegetation cover, biomass, diversity and richness in the southern rangelands of Kenya. Therefore, we investigated the influence of two grazing systems (continual and rotational) and ungrazed area on herbaceous vegetation composition

4.2 Materials and methods

4.2.1 Treatments and experimental design

Detailed descriptions of the site are as described in chapter three of this thesis. The study followed a pseudo-replication design in which there were three treatments: (1) continual grazed, (2) rotational grazed and (3) and ungrazed. Four transects were laid 50m apart from each other in the individual. Demarcation of $10m^2$ were done at an interval of 20m along the transect within which five quadrats were laid down within the $10m^2$ plot.

4.2.2 Vegetation data collection

Vegetation data was collected using the Quadrat method by Garg and Vyas, (1975). Demarcation of 100 m^2 sub- plot within grazing systems was done and five 1x1m quadrats laid out. A quadrat was placed at each of the four corners of the $100m^2$ plot and the 5th quadrant placed at the centre of the plot. Plant functional types were identified as annuals, perennials, forbs. They were then clipped, weighed and put in their respective sample bags for biomass determination. The herbaceous vegetation cover was estimated by visual method within each quadrat with the help of Taxonomist from the University of Nairobi. Biomass was determined by weighing the oven dried vegetation sample. Species diversity and richness were determined using Shannon Weiner's diversity index (1963) as described by (Krebs, 1989). Species richness was calculated as the total number of species per quadrat (Polley *et al*, 2005).

Shannon-Weiner's Diversity index (H');

$$H' = -\sum [\left(\frac{n_1}{N}\right) \times \ln\left(\frac{n_1}{N}\right)]$$

Where; n_1 = number of individuals of each species,

N = Total number of individuals (or amount) for the site

Ln = the natural log of the number.

4.2.3 Statistical Analysis

Data collected on vegetation attributes was subjected to analysis of variance (ANOVA) using Genstat Discovery 15th edition statistical software. Turkey's HSD post hoc was used to separate treatment means.

4.3 Results and discussion

4.3.1 Influence of grazing management on herbaceous plant species diversity and richness

The total number of perennial herbaceous plant species recorded were 20 under continual grazing, 16 and 11 under rotationally grazed and ungrazed areas respectively. The total number of forbs recorded in continual grazing system were 19 while those recorded under rotational grazing system and ungrazed area were 13 and 6, respectively (Tables 1 and 2). The perennials with the highest density under rotational grazing and continuous grazing systems were *Cynodon dactylon*, *Erragrostis superba* and *Cyperus spp*. While the dominant perennial plant species found under ungrazed area were *Themeda triandra*, *Cynodon plectostachyus and Panicum maximum.* The forbs that dominated rotationally grazed areas were *Amarandus hypritis*, *Tagetes minuta* and *Ipomea mombasana* while those which are dominant under continuous grazing management system were *Indigofera spicata and Oxygonum sinuatum* .The ungrazed areas were dominated by *Comelina benghalensis* and *Chlorophylum spp*

There was significant difference in plant species richness between the three grazing management systems ($p \leq 0.05$). The plant species richness was highest in rotationally grazed areas, followed by continually and ungrazed areas with mean species numbers of, 13.4±1.82, 11.9±1.74 and 9.67±2.24, respectively (Figure 2 and 3). Rotationally grazed site had significantly higher species diversity followed by continual and ungrazed area with mean values as follows 3.08±0.65, 2.88±.072 and 2.43±0.02, respectively.

Table 1. Perennial plant species dominating different grazing management systems

Herbaceous plant species	Rotational	Continual	Ungrazed
Cynodon dactylon	306	116	2
Erragrostis superba	59	14	5
Cyperus spp	52	29	0
Digitaria macroblephera	21	17	4
sporobolus fimbriatus	15	21	0
Bothriochloa insculpta	11	28	0
Chloris roxburghiana	13	35	3
Cynodon plectostachyus	9	2	43
Cyperus rotundus	9	27	2
pennisetum mensianum	7	3	0
Sporobolus pyramidalis	6	16	0
Aristida teniensis	6	4	0
cymbopogon excavatus	4	0	2
Digitaria velutina	2	0	3
Panicum maximum	2	3	28
Centrias ciliaris	2	10	2
Themeda triandra	0	0	60
Hyperenia litonia	0	4	0
Erragrostis tenuifolia	0	5	0
hyparrhenia rufa	0	2	0
Microchloa kunthii	0	3	0
Enteropogon contortus	0	3	0
digitaria scalarum	0	2	0
Total number of species	**16**	**20**	**11**

Table 2. Forbs dominating different grazing management systems.

Herbaceous plant species	Grazing management systems Rotational	Continual	Ungrazed
Amarandus hypritis	54	8	0
Tagetes minuta	34	2	0
Ipomea mombasana	24	7	0
solanum incunum	22	14	0
Comelina africana	16	0	0
Lukas martinensis	8	2	2
Justicia anceliana	8	9	4
Chlorophylum spp	7	0	9
comelina latifolia	7	0	0
Indigofera spicata	5	23	0
Comelina benghalensis	4	6	19
Tribulus terrestris	4	0	0
Oxygonum sinuatum	3	19	0
ocimum basilicum	0	2	0
Sita ovada	0	5	0
Tephrosia pumila	0	10	0
Acanthospermum	0	12	0
Potroclin somalensis	0	1	0
seteria pallitefusica	0	2	0
Bitten billosa	0	17	3
vilanthus mandela sp	0	1	0
pollyghala spinosphtera	0	3	3
pollyghalus spinofthera	0	2	0
Total number of species	**13**	**19**	**6**

Table 3: Tree species dominating the grazing management systems

	Rotational	Continual	Ungrazed
Acacia tortilis	7	2	0
Combratum mole	1	1	0
Acacia senegal	1	5	3
Commiphora africana	1	0	0
Ormocarpum kirkii	1	0	0
Dumbega rotundifolia	0	1	0
Acacia merifera	0	2	10
Acacia nilotica	0	3	0
balanitis egyptica	0	1	5
Balanitis egyptica	0	1	2
Omocabum trachybum	0	0	1
Zizibhum spp	0	0	1
Total number of species	5	9	6

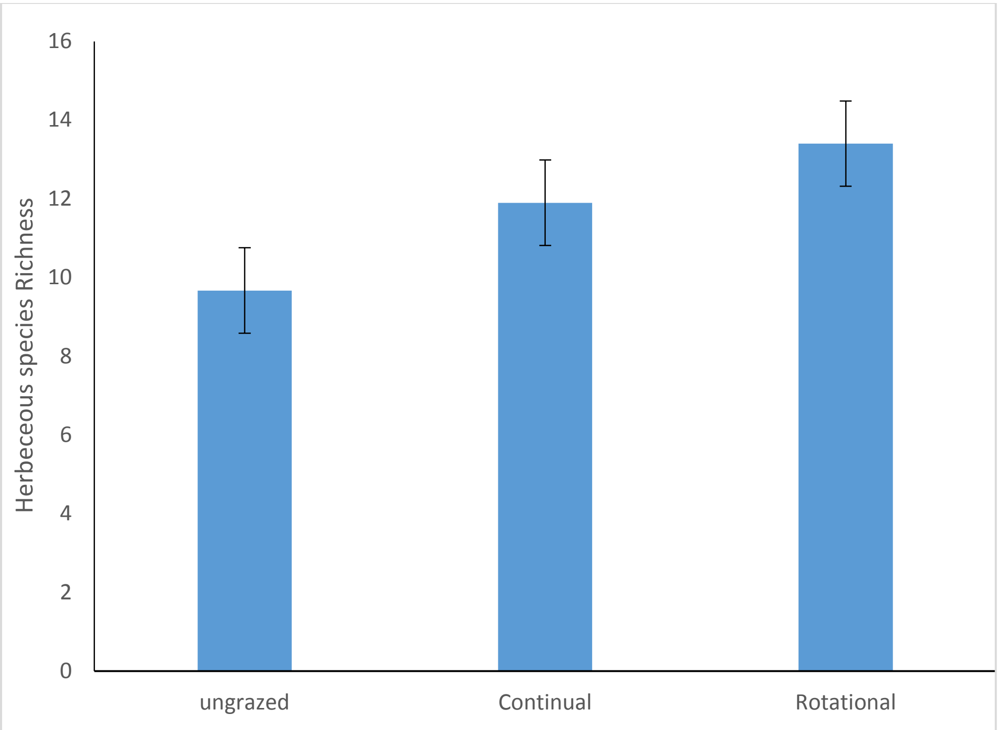


Figure 2: Herbaceous plant species richness across different grazing management systems

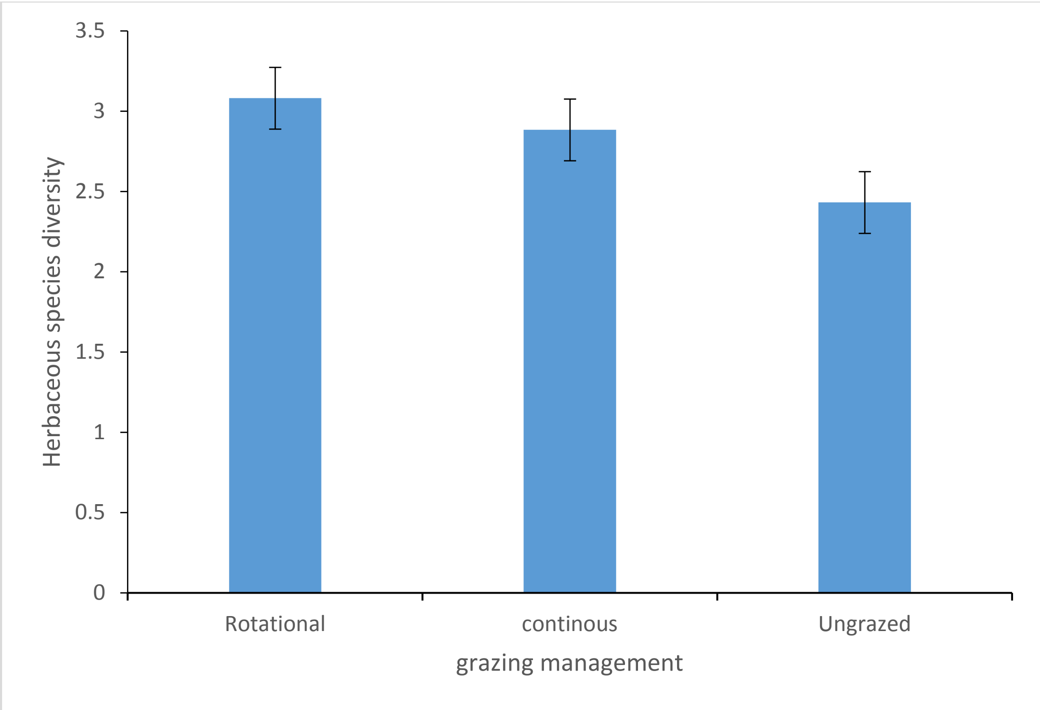


Figure 3: Herbaceous species diversity under different grazing management systems.

The dominance of *Cynodon dactylon* under both the rotationally and continuous grazing systems can be attributed to the fact that this has the ability to withstand heavy grazing pressure and can adapted to various climatic and edaphic conditions(Rita *et al.*, 2012; Egeru *et al.*, 2015). The observed high number of forbs under continuous grazing system can be attributed to the high grazing pressure which reduces the density of palatable plants forcing the animals to forage on species of low nutrition value. As a result, most palatable herbaceous plant species cease to grow, consequently leading to emergence of forbs. Similar results were reported by Mudongo *et al.*, (2016) who observed that continuous grazing led to diminished perennial grass and increase in forbs and woody species. The low density of forbs in the ungrazed area can be due to the canopy cover effects from the dominant woody species which result to insufficient resources such as sun light hence poor growth. The dominant tree species in the rotationally grazed site were *Acacia tortilis* while *Acacia Senegal and Acacia* *merifera* dominated the continuous grazing system site and the ungrazed area. *Digitaria velutina* was the only annual which was present and dominant across the three sites. This could be attributed to its nature of spreading slowly but steadily by seeds and rooting at the nodes of shoots, hence its ability to take advantage of many environments.

The observed low plant species diversity in ungrazed areas can be attributable to the presence of a few dominant tree stands that tap the largest share of the habitat resources (nutrients and light). Belsky, (1992) reported that plant species favoured by lack of livestock disturbance through grazing always tend to outcompete plants with smaller statures, an argument that was also supported by (Pekin *et al*, 2015). On the other hand, we recorded lower herbaceous plant species diversity and richness in the continuously grazed areas compared to the rotationally grazed areas. This implies that the high livestock grazing pressure in the continually grazed areas led to decreased herbaceous species diversity and richness in this semi-arid rangeland.

The high herbaceous plant species diversity in rotationally grazed areas may be as a result of the effects of livestock grazing that results in opening up of the canopy, hence giving chance for regeneration of gap opportunistic plant species. The observed high plant species diversity and richness can also be attributed to grazing which may have reduced competition among plant species through selective grazing on palatable plant species hence giving equal chance for the various plant species to grow.

Our results are in agreement with the intermediate disturbance hypothesis proposed by Connell (1978), whereby the models and metadata analysis have indicated that species richness and the Shannon Wiener diversity index are strong predictors of the intermediate disturbance hypothesis (Svensson *et al*, 2012). The mechanism underlying the intermediate disturbance hypothesis is centred on a complex interplay between life history, biotic interaction and historical disturbance regime (Catford *et al.*, 2012). The increased availability of plant requirements, such as light, following disturbances through livestock grazing explains why high diversities were observed in rotationally grazed areas.

According to Robert and Gilliam, (2003) intermediate disturbance causes changes in local microclimates by opening up space in the canopy, resulting in the availing of resources such as sun-light, which would otherwise not be accessible to understory plants. Physical disturbances prevent competitively dominant species from excluding other species from the community (Mackey and Currie, 2001). This brings about a trade-off between plant species ability to compete and tolerate various forms of disturbance. Species diversity is low at extremely low levels of disturbance because only the best competitors dominate and persist within community (Connell, 1978).

This concept is in agreement with the findings of this study, in which sampling site under the ungrazed area where disturbance was low displayed low plant species diversity. However, in the severely and the highly disturbed areas, a few species persisted or repeatedly colonised after every similar regime of disturbance, thus resulting in low species diversities. This concept also applies to the findings of this study in which continually grazed areas had lower species diversities than rotationally grazed areas this could be probably due to the difference in disturbance from grazing livestock. Continually grazed area experienced high grazing pressure. This was evident by the fact that it was dominated by unpalatable plant species. Therefore, the balance between competitive exclusion and the loss of competitive dominants through disturbance is attained at intermediate disturbances (Mackey and Currie, 2001) which in our study, we can put rotationally grazed site under this category due to minimal disturbance with respect rest periods from grazing.

4.3.2. Effects of grazing management on herbaceous biomass production and ground cover

The above-ground herbaceous biomass production was significantly different in rotationally grazed site *($P \leq 0.05$)* than both continually grazed and ungrazed sites (Figure 4), with the rotational grazed site having the highest herbaceous biomass. The herbaceous vegetation cover was significantly different across the three grazing management systems (Fig 5) with the rotationally grazed site having the highest percent herbaceous cover, followed by continual and ungrazed sites, with the following percentages (27.62±3.76,37.22±2.6 and 55.78±2.56), respectively.

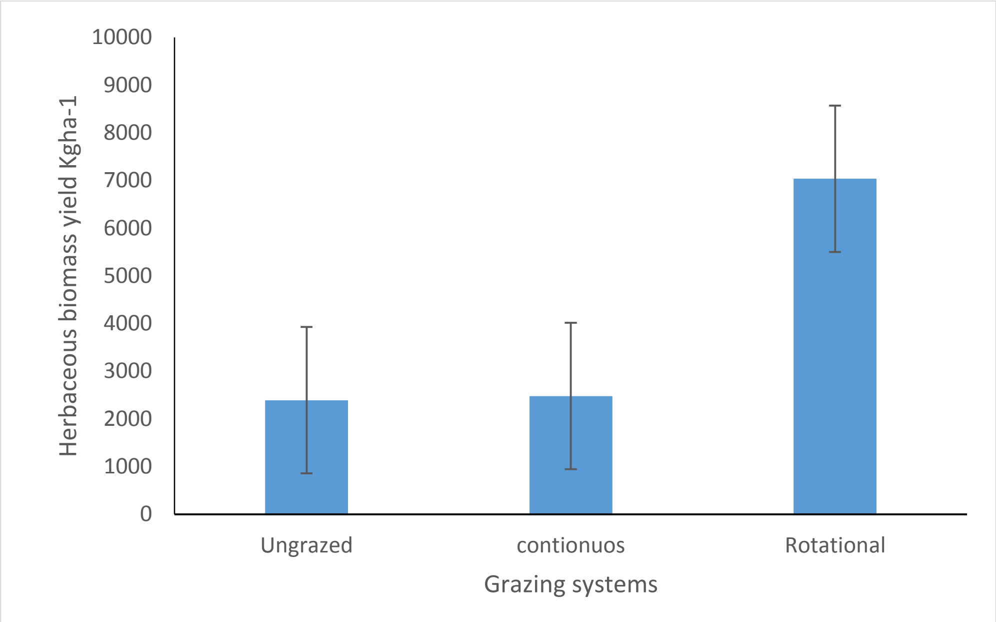


Figure 4: Herbaceous biomass yield (Kgha-^{1}) across different grazing management systems.

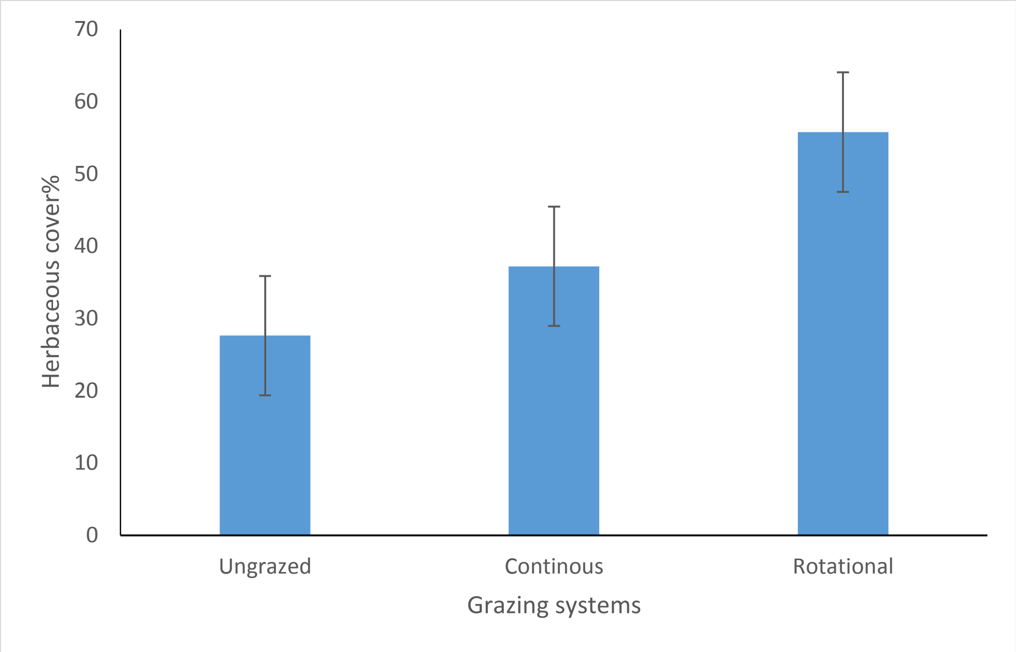


Figure 5: Herbaceous percentage cover under different grazing management systems.

Both enhanced biomass production and herbaceous cover in rotational grazed sites could be attributed to higher forage recovery time which gives plants enough time to regrow as a result of rest periods. Furthermore, plant species under rotational grazing have higher competitive advantage due to less selectivity by grazing livestock, these leads to high growth rate of palatable plant species which usually results to a higher biomass compared with the unpalatable plant species found under continual grazing as a result of selective grazing. These results are similar to those observed by Alphayo, (2015) while studying the influence of holistic grazing management on biomass production. The low herbaceous cover and biomass yield in ungrazed site is associated to canopy effects from the dense wooded species which may have affected growth of the herbaceous vegetation due to shedding effects as a result of canopy cover, resulting to low photosynthetic rates and thus reduced growth.

The low herbaceous cover and biomass production under continuous grazing management can be due to the fact that livestock graze continuously until forage is reduced to critical low levels that affect their recovery. This exposes plants to frequent defoliation, which can be detrimental to plant productivity. The low biomass observed in continual grazing sites was partly due to individual plants being subjected to multiple and severe defoliations without enough time for regrowth. The high frequency of livestock grazing invariably must have led to a decline in the plant's productivity, root biomass and vigour. The low aboveground herbaceous cover and biomass yield in continual grazing sites could have negative effects on soil chemical and physical properties, leading to further reduction in herbage productivity. The difference in biomass and cover between the rotational and continual grazing systems can also be attributed to the influence of livestock grazing on the composition and structure of the plant community, primarily by modifying the competitive interactions via selective feeding of livestock.

Similar results were reported by Gebremeskel, (2006) who found more biomass production under moderate grazing regimes that are well utilized by grazing animals than areas that had been severely and continuously grazed in the semi-arid lands of Ethiopia. Our results were also in agreement to those of (Jacobo *et al.,* 2006) who reported that in time-controlled grazing systems, the frequency and duration of grazing and the rest periods is of importance to plant species recovery from defoliation and gain vigor for their survival, thus resulting in increased biomass yield. Other studies have also reported similar results whereby in the well-structured grazing system, high biomass has been realised and attributed to adequate recovery time allowed after defoliation than in the continuous grazing that is always subjected to high grazing pressure without rest (Radford *et al.,* 2008, Steffens *et al.,* 2008).

The low herbaceous biomass observed in the continually grazed site can be attributed to high percent utilization of pasture in the continually grazed areas. The low biomass production and herbaceous percentage cover under the continual grazing system can also be attributed to the reduced plant leaf area by grazing animals and with insufficient or no time to recover which affects negatively the absorption of active radiation for photosynthesis. This is evidenced by the low biomass production which is as a result of reduced plant's ability to convert light energy into chemical energy for production of biomass. The functioning, growth and development of plant is normally affected by limited conversion of energy (Li *et al*., 2013). The root system is also greatly affected by high grazing pressure because the energy to support the root biomass and new root production is reduced hence affecting the root longevity. When plants are subjected to high grazing pressure, their ability to access the required water and nutrients for their survival is undermined (Holechek, 2001) leading to low plant biomass as was observed in the continually grazed sites.

Kgosikoma, (2012) observed that grazing intensity is the major factor determining the influence of grazing on the ecosystem, and that continuous grazing leads to overuse of forage resources, which affects the ability of plants to regrow after defoliation hence low aboveground herbaceous biomass and cover while working in semi-arid rangelands of Botswana. In a study on the linkages between land use change, land degradation and biodiversity across East Africa. Maitima et al., (2009), reported grazing intensity to have profound impact on biodiversity and emphasised the need for balance between land use and degradation.

4.4 Conclusions

The rotationally grazed area had significantly higher species richness, herbaceous biomass production and cover compared to both the continual grazing system and ungrazed area. There was no difference in plant species diversity between the two grazing systems (rotational and ungrazed) but the ungrazed site had significantly lower plant species diversity. The reduced biomass production in the continuous grazing system is attributable to the overgrazing of the continuously grazed site. This study confirmed that rotational grazing system on arid rangelands is an effective tool for sustainable grazing management. By controlling the time livestock graze in an area, managers conserve biodiversity, increase primary productivity and ground cover while ensuring the continued productivity of forage. We therefore recommended that, grazing period be strictly monitored with respect to the available pasture to ensure that the time spent by grazing animals in a given range site does not deteriorate the environment.

CHAPTER FIVE: SOIL ORGANIC CARBON CONTENT AND STOCKS AS INFLUENCED BY GRAZING MANAGEMENT IN SEMI-ARID GRASSLANDS OF KENYA

Abstract

Rangeland cover approximately 85% of Kenya's land mass and is a major resource for livestock farming. However, these lands are stressed differently by various land management practices which in turn have negative impacts on soil health. Our study aimed at determining the influence of grazing management systems on soil organic carbon content and stocks in Yoani ranch located in the southern rangelands of Kenya. The study followed a pseudo-replication design in which there were three treatments: (1) continual grazed, (2) rotational grazed and (3) and ungrazed .Soils were sampled up to a depth of 1.2m, at an interval of 0-10, 10-20cm,20-30cm, 30-60cm,60-90cm and 90-120cm for determination of soil organic carbon content and stocks. The ungrazed site recorded significantly ($p \leq 0.05$) higher soil organic carbon content and stocks followed by rotationally grazed site and continuously grazed sites which had the least SOC up to 1.2m depth. The predicted results showed that the rate of SOC stock [t/ha] change was positively higher under rotational grazing system in comparison to ungrazed and continual grazing system for the modelling period of 2015-2064. In the absence of grazing, the system was predicted to accumulate 19.22 Mg C ha^{-1} of SOC at the rate of 0.369 Mg C $ha^{-1}yr^{-1}$, whereas rotational grazing system was predicted to accumulate 30.46 Mg C ha^{-1} at the rate of 0.61 Mg C $ha^{-1}yr^{-1}$. The continual grazing management system resulted in accrual of 18.49 Mg C ha^{-1} at the rate of 0.37 Mg C ha^{-1} yr^{-1} over 50 years. Thus, rotationally grazing management system has the greatest potential of accumulating soil organic carbon in semi-arid grasslands.

Key words: Soil organic carbon content; Carbon stocks; Grazing practices; Rangelands

5.0 Introduction

Rangelands are important ecosystems that occupy more than half of the earth's terrestrial surface and are characterised by low-stature vegetation, owing to temperature and moisture restrictions (Brown *et al*, 2008). On global scale, rangelands provide up to 70% of the forage for livestock and contribute to the livelihoods of more than 800 million people (Derner and Schuman, 2007). Rangelands also provide essential ecosystem services, climate regulation, carbon sequestration as well as the mitigation of greenhouse gases such as carbon dioxide (Briske *et al.*, 2008; Brown *et al.*, 2008). Africa's rangelands cover about 43% of its landmass and are characterized as woodlands, Shrublands and/or grasslands (Hoffman and Vogel, 2008).

In Kenya the primary users of rangelands are pastoral communities whom practice extensive grazing. Besides, a smaller number of agro-pastoralists and commercial farms exist. Livestock production in these areas has gained importance due to the increasing human population and hence increased food demand. Consequently, this has led to an increased grazing pressure in most of the rangeland areas.

Forage quality and quantity are determinants of rangeland's sustainability and profitability (Briske et al., 2008). However, rangelands have experienced soils and vegetation degradation due to overgrazing, climate change and plant invasions (Asner *et al.*, 2004, Manjarrez-Dominguez *et al* , 2015). Thus, management practices that will favour plant production and enhance livestock productivity have considerable potential to restore or even increase grassland soil carbon (C) storage and provide a potential positive feedback on the global C cycle (Derner and Schuman, 2007; Follett *et al*, 2010; P Smith *et al.*, 2010). Currently, there is widespread interest in harnessing the large soil carbon sequestration potential from rangelands to offset global greenhouse gas (GHG) emissions, due to their vastness, history of degradation, and potential for improved management. Their ability for soil C storage is estimated to be in a similar order of magnitude as that of croplands and forests (Pete Smith *et al.,* 2008).

Grazing directly affects the structure and function of plant communities through selective plant removal (HilleRisLambers *et al*,, 2010; Huntsinger, Bartolome *et al.*, 2007), defoliation, and changing the amount and composition of residual biomass (Bartolome *et al.*, 1992;). The level of defoliation also affects subsequent forage production by changing light competition and residual photosynthetic tissues. With the change in vegetation cover caused by grazing, soil moisture decreases and temperature increases (Asner et al., 2004; Bremer *et al.*, 2001), which can result in more decomposition at the soil surface and less transfer of plant litter into the soil organic matter pool.

The impact of grazing management on soil biogeochemical processes that regulate rangeland carbon dynamics have not been well studied and understood due to heterogeneity in grassland. Bekker et al, (2006), reported an increase in soil organic carbon in ungrazed areas compared to areas under grazing. The impact of grazing on ecosystem processes is stimulated by the removal of photosynthetic biomass (defoliation), which is determined through grazing intensity; treading and trampling and fecal and urine depositions (Heitschmidt *et al.,* 2004).

The quantity of defoliation relies upon plant morphology, production levels, and the availability of water and nutrients. Selective defoliation modifies species composition, resulting in low productivity and undesirable plant compositions. Livestock treading compact the soil surface, increasing the soil bulk density while hoof action deteriorates soil aggregate stability. Adverse changes in soil physical structure may also result to a decline in water infiltration and root development.The addition of nutrients through fecal and urine affects the soil biogeochemical methods. Altogether, grazing has the capability to steer rangeland carbon dynamics through changing plant litter chemistry (Neff *et al*, 2009), plant biomass allocation patterns clutter production, and the spatial distribution of nutrients (Grayston *et al.*, 1997).

Regardless of the above noted grazing effect on rangeland carbon dynamics, most research results showed a wide variation ranging from positive to negative to no response. Gill (2007) evaluated the impact of ninety years of protection from grazing on carbon dynamics in subalpine rangeland and reported that livestock grazing had no significant impacts on soil carbon or particulate organic matter, however active soil carbon content increased. The loss of carbon from the active carbon pool become higher in grazed plots (4.6% of general carbon) than in ungrazed plots (3.3% of general carbon). Those results suggest that grazing can also convert the recalcitrant carbon pool into easily mineralizable carbon fraction. The exclusion of grazing prompted an increase in annual forbs and grasses without dense fibrous rooting structure conducive to soil organic matter formation and accumulation (Reeder and Schuman, 2002).Similarly, other scientists have reported the various impacts of grazing management on rangeland soil organic matter, for instance, a study by Conant (2003) in pastures of Virginia, United States of America, found that soil organic carbon averaged 8.4 mg C ha^{-1} more under intensive management or light rotation grazing than considerably grazed sites. Naeth et al. (1991) reported a negative impact on soil organic carbon in heavy intensity or early season grazing, compared to light grazing in the grasslands of Alberta, Canada. Heavy grazing resulted in vast discounts in height of status and fallen clutter, and a decrease in stay vegetative cover and biomass production.

Furthermore, the vertical distribution of Soil carbon storage is rarely estimated at the natural landscapes (Wang *et al*., 2010).How much carbon is underestimated in global budgets below the first meter is surely a bigger question mark. Batjes (1996) estimated a 60% increase in the global soil organic carbon (SOC) storage with depth extended to 2.0 m. A recent estimate of SOC storages has reported with a 56% increase at global level with the third meter of soil was also included (Jobbágy and Jackson, 2000) . An increasing understanding of the importance of deep soil carbon is reflected in the mounting global estimates of soil carbon storages (Veldkamp *et al.,* 2003). The biomes with the most SOC at 1–3 m depth were tropical evergreen forests and tropical grasslands/savannas (Jobbágy and Jackson, 2000). Soil C pool that remains poorly understood is its vertical distribution, especially the differences of this vertical distribution across management types. Most previous studies on soil carbon storage have been focused on upper layer, although deeper profile was known to be important in soil carbon storage (Mi *et al*., 2008).

In present time, it is essential to estimate mitigation potential of grazing lands because of their vast area. There are several methods of estimating changes in soil organic carbon. The IPCC Guidelines (IPCC 2006) provide a three-tiered approach where the tier 3 is based on dynamic model if the data is available. Models of soil organic carbon dynamics are a well-known tool to simulate the response of the soil system to environmental disturbances and are widely used to study changes in SOC stocks (Peltoniemi et al., 2007). The main characteristics of the most popular process-oriented SOC turnover models have been frequently discussed in the literatures (G. Wang, Huang, Wang, Yu, and Zhang, 2013). From literature, it has been observed that CENTURY, RothC and DNDC are the most frequently used models to simulate SOC dynamics spatially (Yu and Yao, 2003). RothC- 26.3 was originally developed and parameterized to model the turnover of organic C in arable topsoil's from the Rothamsted long-term field experiments. Later, it was extended to model the turnover in the grassland and in woodland, and operate in different soils under different climates (Coleman *et al.*, 1997). RothC-26.3 has been tested against long-term experiments in a range of soils and climatic conditions in Western and Central Europe and some parts of Africa (Coleman *et al.*, 1997; Falloon and Smith, 2002).

Currently, there are few experimental data regarding the potential for carbon sequestration in tropical drylands, particularly for the land management practices associated with grasslands (Farage *et al*., 2007; Favretto *et al*., 2016). Although field experimentation is required to confirm the ability of particular soils and agricultural practices to sequester carbon, modelling provides a means by which the overall feasibility of a variety of land management practices can be assessed (Smith *et al*., 2010). Modelling soil carbon dynamics also provides a vehicle by which the most promising methodologies can be selected for further investigation. This is particularly relevant to soil carbon sequestration as it can take several years before measurable changes are evident in the field following alteration in agricultural practice (Poussart *et al*., 2004; Rasmussen *et al*., 1998).

Although the effects of grazing impact on soil C storage been extensively studied in a wide range of ecosystems worldwide (Pelster *et al*., 2015) there is no existing literature on studies done to examine the linkage between SOC storage and grazing systems in the southern eastern rangelands of Kenya. Understanding the effects of livestock grazing management systems on SOC stocks is of essence for rational utilization of grassland. Therefore, we investigate the influence of three different grazing management systems (continual, and rotational grazing system and an ungrazed area) on SOC. This study also investigated the potential for carbon sequestration under different grazing management systems in semi-arid grasslands of Makueni County, Kenya using RothC modelling approach over a period of 50 years.

5.1 Materials and methods

5.1.1 Treatments and Experimental design

The study was conducted at Yohani ranch in Makueni County. The study followed a pseudo-replication design in which there were three treatments: (1) continual grazed, (2) rotational grazed and (3) and ungrazed. Four transects were laid 50m apart from each other in the individual treatments thereafter soil were sampled at an interval of 20m along a 100m transect. Detailed descriptions of the site are as described in chapter three of this thesis.

5.1.2 Soil Sampling

Soil was sampled at intervals of 0–10, 10-20, 20-30, 30-60, 60-90 and 90-120 cm depth using a soil auger. Soils were sampled at intervals along a 100m transect. Within each sampling point, two soil samples were mixed to form a composite soil sample per respective depths. The composited samples were packed in a well labelled polythene bags for transportation to ILRI Mazingira Centre for soil organic carbon analysis. In addition, undisturbed soil samples for each depth were collected using core rings with a defined volume (100 cm^3) from each sampling plot for bulk density determination.

5.2.3 Soil Analysis

The soil samples were air-dried and passed through a 2 mm sieve. In order to determine soil total C and N content, 20 g of sieved soils were dried at 40°C for 48hrs and thereafter ground with a hammer mill (RetschMM400Mixer Mill, Retsch GmbH, Germany). A 20g subsample was then analysed for C content using a high-temperature oxidative combustion system (Elementar Vario Max Cube). The bulk density for each depth was estimated by core ring method (Blake, 1965).

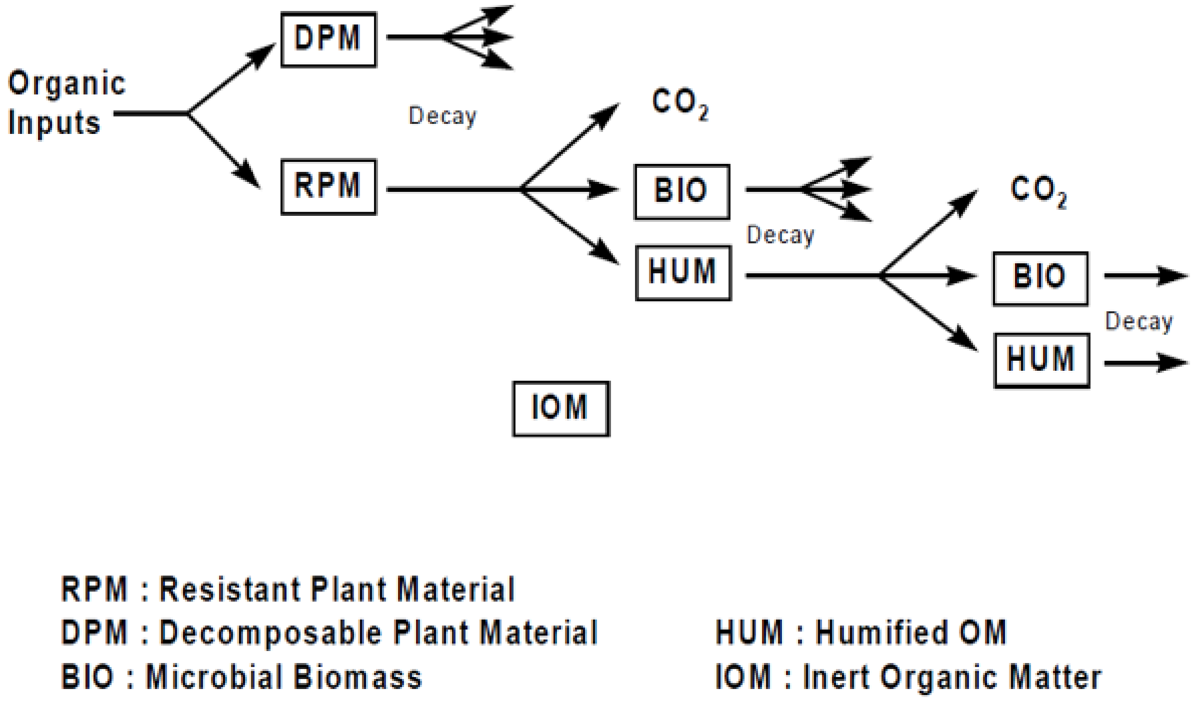
Soil carbon stocks were calculated using the following equation 1:

SOC_{Stock}=c × BD × D

Where SOC is the soil organic carbon stock (Mg C ha-1) (Were, 2015). c represents the carbon concentration (%), BD bulk density (gcm-3) and D the respective soil depth (m). where:

5.2.4 Roth C Model Description and Data requirements

The RothC-26.3 model is a SOM decomposition model that divides incoming plant residues into decomposable plant material (DPM) and resistant plant material (RPM); these both decompose to form microbial biomass (BIO), humified organic matter (HUM) and evolved CO_2 (Coleman and Jenkinson, 1996). The model also includes an inert pool of organic matter (IOM). Roth-C is one of the most widely used SOC models (Jenkinson *et al*., 1991; Juma, Izaurralde *et al*., 1996) and has been evaluated in a wide variety of ecosystems including croplands, grasslands and forests (Falloon and Smith, 2002). The schematic structure of the Roth-C model depicts plant residues entering the soil environment, undergoing decomposition by the soil microbial biomass to form several pools with the evolution of CO_2.



5.2.5 Data requirements

The Roth-C model requires three types of data:(a) Climatic data; monthly rainfall (mm), monthly evapotranspiration (mm), average monthly mean air temperature (° C); (b) Soil data; clay content (%), inert organic carbon (IOM), initial soil organic carbon (SOC) stock (t C ha^{-1}), depth of the soil layer considered (cm); (c) Land use and land management data; soil cover, monthly input of plant residues (t C ha^{-1}), monthly input of farmyard manure (FYM) (t C ha-1), residue quality factor (DPM/RPM ratio) (Jenkinson *et al*., 1991).

5.2. Model Calibration procedure

For calibration, RothC model was run through an equilibrium period (7000–10,000 years) standard procedure to represent soil and vegetation conditions prior to human disturbance (Parton *et al*., 1987). The equilibrium files for all the land management units (monthly plant residue input, soil cover, monthly farm yard manure) were prepared. The model was set up to simulate the characteristics of the study site, including the land management represented by monthly C inputs (t C ha^{-1}) and soil cover (Coleman, and Jenkinson, 1990) (whether soil is covered by a crop or is fallow). All soil pool data were converted from g C/kg soil to t C/ha for use in the model. Climate data for Makueni County were acquired from Kenya meteorological department. Land management data with a DPM/RPM (decomposable plant material/resistant plant material) ratio of 0.67 and an inert organic matter (IOM) content of 4.83, 4.5504 and 3.798 t C ha^{-1} for ungrazed site, rotationally and continually grazed sites respectively were approximated using equation (1) proposed by Falloon *et al*, 1998 (because the radiocarbon content was not known) were used to simulate equilibrium land use at the study site.

IOM = 0.049TOC1.139 (1)

Where: TOC is Total organic carbon, t C ha^{-1}

IOM is inert organic matter, t C ha^{-1}

Model input files were created using the input data collected from the site, and model runs executed to accurately simulate management regimes at the site studied.

5.2.1 Litter collection and soil carbon content determination

Quadrat method was used to collect litter samples as described by Garg and Vyas, (1975). In each plot, a 100 m^2 sub- plot was demarcated and five 1x1m quadrats laid out. A quadrat was placed at each of the four corners of the 100m^2 plot and the 5th quadrant placed at the center of the plot. There was a total of 135 quadrats were used for the three grazing systems, each grazing system having 45 quadrats. Litter samples were collected from the 1×1m quadrats and packed in well labelled polythene bags. The litter samples were oven dried and thereafter ground with a hammer mill (RetschMM400Mixer Mill, Retsch GmbH, Germany). A 10g subsample was then analysed for C concentrations using a high-temperature oxidative combustion system (Elementar Vario Max Cube).The result were converted to T C/ha. Plant residue input was calculated by dividing this amount with the number of years the place has been under grazing to get the yearly residue input. Then divide by 12 to get the monthly residue input.

Table 4.The proportion of carbon ($tCha^{-1}$) assumed added in plant material each year under different grazing management systems based on RothC model calculations

Month	Rotation grazing	Continual grazing	Ungrazed
January	0.05	0.015	0.022
February	0.05	0.015	0.022
March	0.05	0.015	0.022
April	0.05	0.015	0.022
May	0.1	0.037	0.05
June	0.15	0.037	0.05
July	0.1	0.037	0.05
August	0.15	0.037	0.05
September	0.1	0.035	0.05
October	0.1	0.035	0.05
November	0.05	0.035	0.02
December	0.05	0.035	0.02

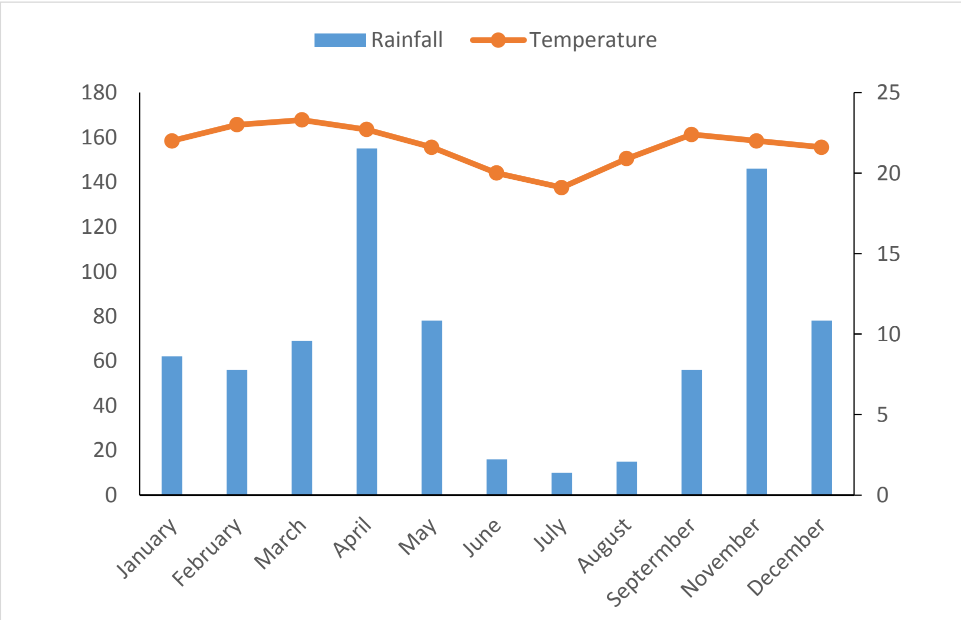


Figure 6: Averages monthly rainfall and temperature data from 1950-2014).

5.2.3 Running the model

5.2.3.1 Simulation procedure

The initial carbon content of the soil organic matter pools and the annual plant addition to the soil were obtained by running the RothC model to equilibrium under constant environmental conditions (Coleman and Jenkinson, 1996). The constant climatic conditions were taken to be the average of climate data from 1950 to 2010 By initializing the model in, and running the model from 1950, potential initialization effects are minimized (Smith *et al*., 2005). RothC is known to be relatively insensitive to the distribution of C inputs through the year; the proportions of plant material added to the soil in each month were set to describe the pattern of inputs for a typical grassland (Table 4). Plant cover was assumed to occur all year round in grasslands. Initially, the total plant input was obtained by dividing litter carbon content by the number of years the area has been under the different management to get the annual plant residue input and by 12 to get the monthly plant residue input: this was not intended to represent the actual plant addition, but provided the first point in the initialization, indicating how the annual plant addition should subsequently be adjusted.

The adjusted annual plant addition was then redistributed through the months, and the equilibrium run repeated. This iteration was continued until the measured and the simulated carbon contents of the soil were within 0.00001 tC ha^{-1} .Having determined the plant additions and carbon contents of the soil organic matter pools, the simulations were run forward from 2015 to 2064 using the measured climate and simulated plant residue input data

5.2.3.2 Statistical analysis

Analysis of variance (ANOVA) was performed to determine if the measured soil organic stocks were significantly different among the grazing regimes. Significant differences for the analysis of variance were accepted at $P \leq 0.05$. Tukey's HSD post hoc was used to separate means of the measured soil attributes under the various grazing treatments.

5.3 Results and Discussions

5.3.1. Influence of grazing management on soil bulk density

Along the soil profile, a significant difference ($p \leq 0.05$) in soil bulk density up to 0.3m depth was observed with continuously grazed site showing the highest values followed by rotationally grazed area and ungrazed site, respectively, however the latter were no significantly different in bulk density at the lower depths across the management regimes (Table 5)

Table 5: Soil bulk density (gcm-3) across grazing management systems

Depth(m)	Ungrazed	Rotational	Continuous	Cv%
0-0.1	1.22±0.02a	1.45±0.01b	1.57±0.02c	3.2
0.1-0.2	1.17±0.02a	1.38±0.04b	1.46±0.04c	3.9
0.2-0.3	1.13±0.02a	1.24±0.01b	1.28±0.003c	1.9
0.3-0.6	1.06±0.03a	1.14±0.08a	1.22±0.03a	7.6
0.6-0.9	1.08±0.02a	1.1±0.02a	1.14±0.24a	19.8
0.9-1.2	1.07±0.02a	1.08±0.02a	1.18±0.006b	1.8

Means with different letters within the row are significantly different ($P \leq 0.05$)

The significantly high soil bulk density under continual grazing system compared to rotationally grazed and ungrazed areas can be attributed to animal trampling which may have had severe effect on soil compaction. The lower bulk densities in rotational grazing sites can be due to the minimum livestock impact and loafing due to short duration grazing that gives maximum rest to the grazed area. The low bulk density in the ungrazed area can be attributed to lack of grazing disturbance from livestock. The decrease in bulk densities down the soil profile across the grazing management systems is due to the fact the pressure exerted by grazing animals is concentrated on the upper soil layers. Similar effects have been observed for other grassland ecosystems (Daniel *et al.*, 2002; Savadogo *et al*., 2007; Stavi *et al*., 2008; Wei *et al.*,2011) , and also (Steffens et al., 2008) monitored significant higher bulk densities in a semi-arid steppe in heavily grazed plots compared with ungrazed plots. Igwe (2005) found that areas grazed continuously exhibited higher bulk density than the areas under moderate grazing in south eastern Nigeria. This finding was attributed to consistent animal trampling that increases soil compaction.

5.3.2 Effect of livestock grazing management on Soil Carbon concentrations and stocks

The ungrazed site showed higher SOC concentrations and stocks than the grazed sites along the soil profile (Fig 7 and Table 6) respectively. Similarly, the rotationally grazed site showed higher SOC concentrations and stocks along the entire sampled soil depths compared to continual grazing system. However, the difference in SOC concentration was not statistically different ($p \geq 0.05$) between continual and rotational grazing systems at the lower depths (0.3M- 1.2M.)

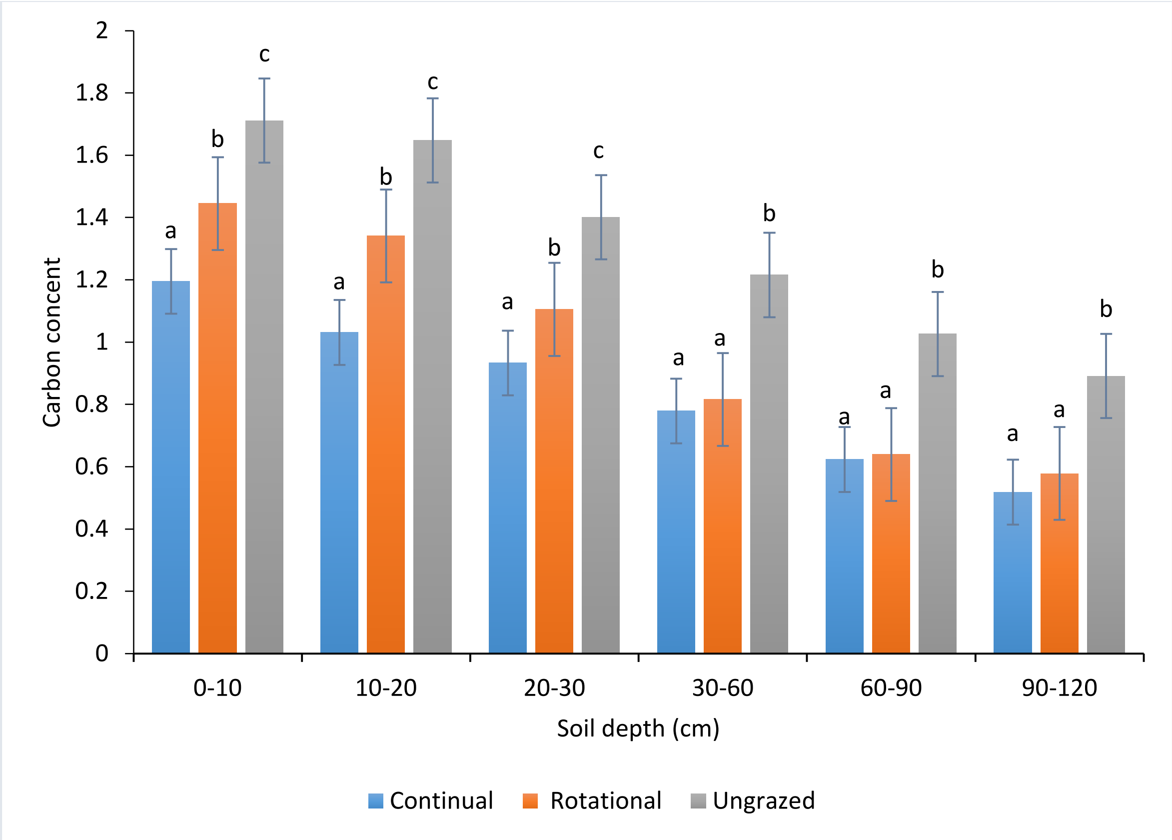


Figure 7: Soil organic carbon content across different grazing management system. Different letters indicate significant differences (p≥0.05) per depth across grazing systems.

Table 6: Soil organic carbon stocks (MgCha-1) across different grazing management systems along the soil profile (Means + SE)

Depth(M)	Continual grazing	Rotational grazing	Ungrazed
0-0.1	18.66±2.82a	20.9±1.83a	20.95±1.43a
0.1-0.2	14.95±2.00a	18.85±0.0.51a	19.42±1.65a
0.2-0.3	11.97±2.41a	13.67±2.11ab	15.93±4.73b
0.3-0.6	26.79±2.378a	28.63±2.522a	38.76±3.57b
0.6-0.9	21.1±2.37a	23.14±2.87a	33.47±2.82b
0.9-1.2	17.05±0.57a	22.03±1.65b	28.6±2.04c

Means with different letters within the row are significantly different ($P \leq 0.05$)

The difference in SOC concentrations and stocks across grazing systems can be attributed to the difference in herbaceous vegetation cover. The higher aboveground standing biomass under rotational grazing and ungrazed sites significantly reduced loss of organic matter and nutrients from the soil-plant system through soil erosion. Also, more stubble biomass is expected under rotational grazing sites than continual grazing sites, which means a conversion of atmospheric carbon through the process of photosynthesis into carbon and nitrogen compounds that are returned to the soil through litter fall and dead plant materials. The higher SOC content stocks in the rotational grazing management system can therefore be attributed to increased litter accumulation.

The low herbaceous plant cover under the continual grazing system as a result of high grazing pressure exposes soils to sun rays leading to increased soil temperatures and evapotranspiration rates hence increased decomposition of organic matter resulting in higher losses of carbon. 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 ungrazed and rotationally grazed 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 these results into low SOC accumulation for continual grazing system.

The findings collaborates with those of Sanjari et al, (2008), who pointed out that, a relative increase in soil organic matter under time controlled grazing as opposed to under continuous grazing was attributed to higher rates of grass growth and rest periods that lead to an increase in litter accumulation and subsequent increase soil organic carbon under time rotational grazing compared with continuous grazing. In a study by Han *et al.,* (2008) on the effects of grazing intensity on soil carbon in Mongolia, they reported less organic carbon in areas with high grazing intensity than areas under low grazing intensities and attributed the results to high net primary production under light grazing intensities.

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 their review of different effects of livestock grazing on soil organic carbon reported divergent results where grazing increased SOC while in some instances, it reduced or had no influence on SOC. Derner and Schuman (2007) in their study on the potential of rangelands in sequestrating carbon, noted that grazing usually increases carbon storage on C4 dominated grasslands.

5.3.3 Projected impact of grazing management systems on soil organic carbon stocks for a 50 year period.

In the absence of grazing, the ungrazed site was predicted to accumulate 19.22 Mg C ha^{-1} of SOC at the rate of 0.369 Mg C $ha^{-1}yr^{-1}$, whereas rotational grazing system was predicted to accumulate 30.46 Mg C ha^{-1} at the rate of 0.61 Mg C $ha^{-1}yr^{-1}$.The continual grazing management system resulted in accrual of 18.49 Mg C ha^{-1} at the rate of 0.37 Mg C ha^{-1} yr^{-1} over 50 years.

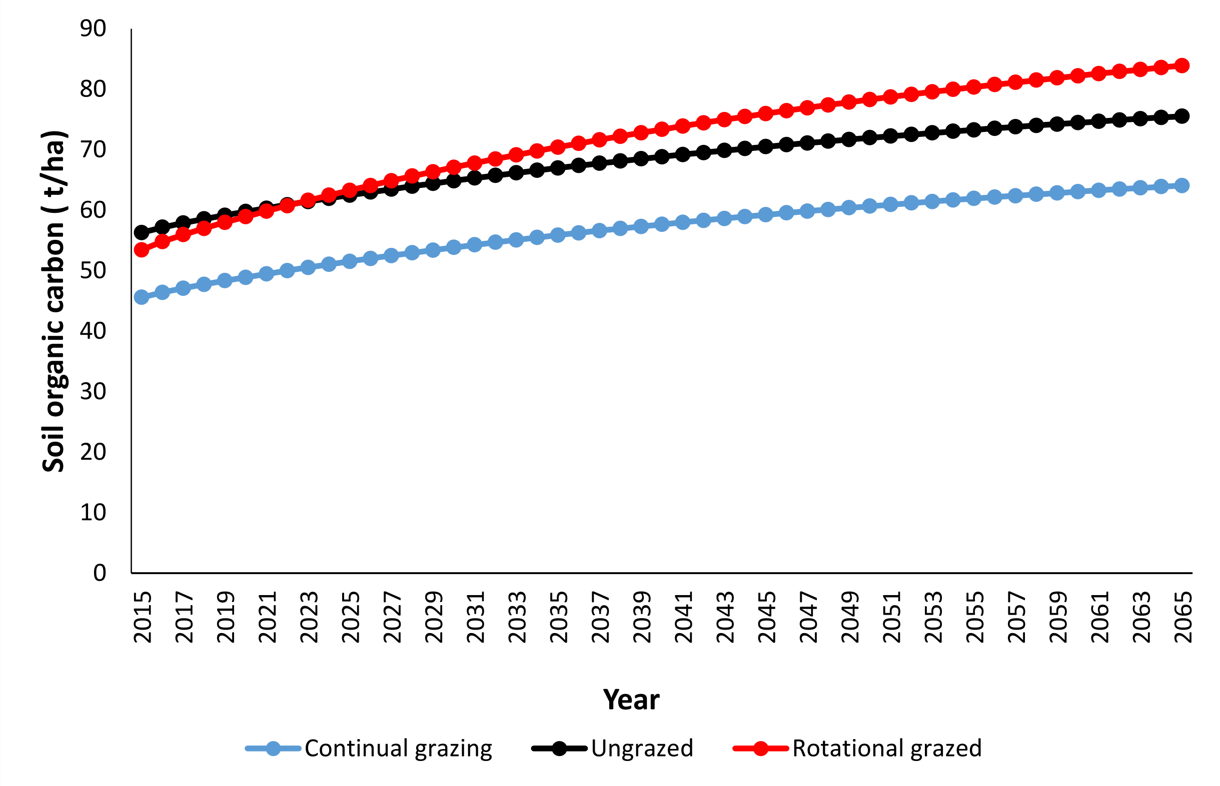


Figure 8: Predicted Soil organic Carbon Stocks in a 50 year period under different grazing management systems

The predicted higher SOC stocks in rotationally grazed area than in both continual and ungrazed areas can be attributed to plant biomass returning into soil. Under rotational system, the input of above-ground plant materials is continuous throughout the year in a significant amount compared to continually grazed areas due to enhanced biomass production. The modelling results indicate a higher proportion of total plant material in rotational grazed systems entering into soil for carbon sequestration compared with both continually grazed and ungrazed areas

Furthermore, the effect of grazing livestock on soil physical characteristics can explain the differences in the predicted high accrual of soil carbon under rotational and continual systems. The predicted low rate of accumulation of soil organic carbon under continual grazing system can be due to livestock trampling as a result of high grazing pressures which leads to soil organic carbon losses.

The low herbaceous plant cover under the continual grazing system as a result of high grazing pressure exposes soils to sun rays leading to increased soil temperatures and evapotranspiration rates hence increases the decomposition of organic matter resulting in higher losses of carbon from the soil. The difference between the ungrazed site and rotationally grazed site could be attributed to the difference in the quality litter inputs. The litter input in the ungrazed site is majorly form the wood vegetation which may take time to be incorporated in the soils as opposed to the ones in the rotationally grazed site which are majorly herbaceous vegetation.

Our finding are in agreement with studies done by Fearnside and Barbosa (1998), who showed that trends in soil C were strongly influenced by pasture management. Sites that were judged to have been under poor management generally lost soil carbon, whereas sites under good management accumulated soil carbon. Trumbore et al. (1995) reported soil C losses in overgrazed pasture in the Amazon region. According to Neill et al. (1997), degraded pastures with little grass cover are less likely to accumulate soil C because inputs to SOC from pasture roots will be diminished. Barger in his studies reported that, grazing has the potential to influence rangeland carbon dynamics by altering plant litter chemistry, litter production, plant biomass allocation patterns, and the spatial distribution of nutrients Furthermore, he reported that, depending on the intensity, grazing pressure may slow decomposition rates by decreasing plant litter carbon to nitrogen (C: N) ratio, or due to decreased standing biomass, may accelerate the decomposition by increasing soil temperature.

5.4 Conclusion

The study showed that the highest levels of soil organic carbon content and stocks were observed in the ungrazed area followed by the rotationally and continuously grazed areas respectively. The predicated soil organic content showed that rotationally grazed systems can sequester up to 6.5 t C over 50 years, yielding an average 0.46 t C ha^{-1} $year^{-1}$. In the absence of grazing, the system was predicted to accumulate 19.22 Mg C ha^{-1} of SOC at the rate of 0.369 Mg C $ha^{-1}yr^{-1}$, whereas the continual grazing management system resulted in accrual of 18.49 Mg C ha^{-1} at the rate of 0.37 Mg C ha^{-1} yr^{-1} over 50 years. All the grazing systems were predicted to accumulate SOC. The results highlighted the importance of well managed pasture in soil carbon sequestration as a potential measure for mitigating climate change. These shows, that the success of all grazing systems when constrained by similar ecological variables, the difference in their performance with respect to their impact on soil organic carbon is as a result of the effectiveness and the efficiency with which the grazing management practices are used rather than ecological variables.

CHAPTER SIX: EFFECTS OF GRAZING MANAGEMENT ON GREENHOUSE GAS EMISSIONS IN SOUTHERN RANGELANDS OF KENYA

Abstract

Rangelands ecosystems play a critical role in regulating the emission and uptake of the most important greenhouse gases (GHGs) such as CO_2, CH_4, and N_2O. However, the effects of grazing management on GHG fluxes in the semi-arid lands of East Africa remain unclear. The present study compared the effects of three grazing systems on cumulative CO_2, CH_4, and N_2O fluxes in the semi-arid grazing land ecosystem in Yohani ranch Makueni County, Kenya. The study followed a pseudo-replication design in which there were three treatments: (1) continual grazed, (2) rotational grazed and (3) and ungrazed. Greenhouse gas samples were collected using the static chamber method for a period of three months covering the dry and wet season as well as a transition period. Cumulative soil CO_2 fluxes were highest in continual grazing system (2357±123.9 kg ha^{-1} 3 month), followed by rotational grazing (1285±123.9 kg ha^{-1} 3 month) and lowest in the ungrazed (1241±143 CO_2 kg ha^{-1} 3 month) respectively. The three month cumulative N_2O and CH_4 fluxes were also highest in continual grazing and lowest in ungrazed site 677.9±130.1, 208.6±127.3 and 162.2±150.3 (gm ha^{-1} 3 month) and CH_4, 232.7±126.6, 173.1±126.6 and 80±46.2 (gm ha^{-1} 3 month) respectively. The results suggest that the continual livestock grazing system increases emissions of GHGs compared to rotational grazing.

Key words: Greenhouse gas fluxes; Grazing management systems; Rangelands

6.0 Introduction

Greenhouse gas (GHG) emissions were estimated globally to be 49×109 $MgCO_2$ eq. in 2010 (Change, 2014a, 2014b), with approximately 21.2–24% (10.3–12×109 $MgCO_2$ eq.) of emissions originating from soils in agricultural, forestry and other land use (Tubiello et al., 2015). Annual CO_2 and CH_4 emissions from agriculture were estimated to be 5.2–5.8×109 $MgCO_2$ eq.yr^{-1} in 2010 (FAO, 2014; Tubiello et al., 2013), with approximately 4.3– 5.5×109 $MgCO_2$ eq.yr^{-1} attributable to land-use change (Change, 2014a). Greenhouse gas fluxes in Africa play vital role in the global GHG budget (Bombelli *et al*., 2009; Ciais *et al.*, 2011; Hickman *et al*., 2014; Thompson *et al.*, 2014; Valentini et al., 2014). For instance, in sub-Saharan Africa, Nitrous oxide (NO_2) emissions has been estimated to contribute between 6 and 19% of the global total changes in soil fluxes in SSA with high inter annual variations in the tropical and subtropical environments (Thomas and Rosenstock, 2016). Rangelands are important ecosystems not only because of the vast area they occupy and their contribution as major feed resource to livestock, but are also important ecosystems in the global budget of the greenhouse gases (GHGs); carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). In Kenya, rangelands occupy more than three quarters of land and the primary users of these are pastoral communities who practice extensive grazing. However, there are some agro-pastoralists and commercial ranching farms. Livestock production in these areas has gained importance due to increased human population resulting to increased food demand. Consequently, this has led to increase in grazing pressure in most of the rangeland areas. Rangelands have experienced soils and vegetation degradation due to overgrazing, climate change and plant invasions. Thus, management practices that will favour plant production for increased livestock productivity is desirable. This should have potential to restore or even increase grassland soil carbon storage and provide a potential positive feedback on the global carbon cycle (Smith *et al*., 2010).

The extent to which rangeland act as a net sink or a source for GHGs is determined particularly by their management (Conant *et al*., 2001). Rangeland degradation through inappropriately adopted livestock grazing management can lead to a net release of GHGs, predominantly in the form of CO_2 that deplete soil organic carbon stocks (Lal, 2004). On the other hand, efficient grassland management and restoration of degraded sites, e.g. through planned grazing or exclusion, offer large GHG mitigation potential, mainly through the sequestration of atmospheric CO_2 (Conant *et al*., 2001; Lal, 2004; Smith, 2008). A complete understanding of agriculture's impact on radiative forcing and the accurate quantification of GHG mitigation potentials require field-level measurements of CO_2, CH_4, and N_2O (Robertson *et al*., 2000). Complete field-level GHG fluxes of agricultural systems have been extensively measured in intensive and semi-intensive European grasslands (Soussana *et al*., 2010) in the Midwest and Northeast United States on croplands and unmanaged sites (Robertson *et al.,* 2000; Mosieretal. 2006) and on moderately and heavily grazed pastures of the temperate steppe in the United States (Liebig *et al*., 2010). However, we didn't find literature on the GHG balance of livestock farming in Kenyan rangelands. Furthermore, the influence of grazing management on the balance of GHGs is not yet well established, and further research is needed to quantify the effect of grazing management in GHGs emissions (Gill *et al*., 2010;).

Although the effects of grazing impact on greenhouse gas emissions have been studied in a wide range of ecosystems worldwide, there is no existing literature on studies done to examine the linkage between greenhouse gas emissions and grazing systems in Kenyan rangelands (Pelster *et al*., 2015). Worthwhile, the effects of livestock grazing management systems on greenhouse gas emission need to be understood in order to sustainably utilize grassland and determine "low emission development strategies" (LEDS). Therefore, we investigated the influence of two grazing systems (continual and rotational grazing systems) on GHG emissions and compared to an ungrazed area in the south eastern rangeland of Kenya.

6.1 Materials and methods

6.1.1 Experimental design and treatments

The treatments are presented in chapter three under general materials and methods chapter. In each grazing system, four sampling points were randomly selected, and within each sampling point, four chambers were installed to form a 10m x10m square.

6.1.2 Flux measurements

The concentration of CO_2, N_2O and CH_4 were measured using the static greenhouse gas chamber approach (Pelster *et al*., 2015). In each grazing system, four sampling points were randomly selected, and within each sampling point, four chambers were installed to form a 10m x10m square. The, chambers consisted of a collar (0.27m×0.372m×0.1m) and a lid (27×37.2×12.5cm) made of plastic. The collars were inserted up to 10cm into the ground. The lids were equipped with 50cm long (2.5cm diameter) vent tubes, thermometers to measure internal temperature, fan and a gas sampling ports.

During measurements, the lid was placed on the collar and both increments were tied with clamps with a gasket between the lid and the collar for airtight seal. Chamber bases were inserted at least one week prior to the first greenhouse gas concentration measurements and remained in place throughout the three months sampling period. During each sampling event, chambers were closed for 30 min and thereafter, four samples taken at 10 min intervals (0, 10, 20, and 30 min) from each individual chamber. A gas pooling technique was employed where 15 ml of gas was sampled in each of the four chambers within a sampling point. This was done using a 60ml propylene syringe with Luerlocks and immediately transferred into 20 ml glass vials fitted with crimp seals (Butterbach-Bahl *et al.*, 2016). The first 30ml of the sample were used to flush the vial and the remaining 30ml filled the vial in order to pressurize it to reduce the likelihood of contamination with ambient air. Samples were analyzed within 36 hours after every sampling period.

Concentrations of CO_2, N_2O, and CH_4 were analyzed using a gas chromatograph (model 8610C; SRI) equipped with two detectors; a flame ionization detector (FID) comprising of a Platinum catalyzed methanizer for catalytic conversion of CO_2 to CH_4 and for subsequent detection of CH_4 and CO_2 and an electron capture detector (ECD) to detect N_2O. A mixture of CO_2 and N_2 pre-mixed in the ratio of 5:95, was used as the ECD Make-up gas to improve on the detector sensitivity.

The analytes were separated on (3m, 1/8″) chromatographic columns packed with Hayesep D stationary phase at an isocratic oven temperature program of 70°C. ECD and FID detectors temperatures were set at 350°C. 99.999%. White spot Nitrogen was used as carrier gas at flow rates of 25ml min^{-1} on both FID and ECD lines. Gas concentrations of samples were calculated based on the peak areas measured by the gas chromatograph relative to the peak areas measured from calibration gases run at two calibration levels. This was done four times each day. Calibration gases ranges from (4.28-8.3097 ppm) for CH_4, (400-810.5 ppm) for CO_2 and (0.36-0.7606 ppp) for N_2O. Concentrations were then converted to mass per volume using the Ideal Gas Law (pV = nRT) and measured chamber volume, internal chamber air temperature, and atmospheric pressure determined during sampling. GHG fluxes were calculated using linear regression of gas concentrations versus chamber closure time.

6.1.3. Statistical analysis

Gas flux data was subjected to analysis of variance (ANOVA) using GenStat Discovery 15th edition statistical software. Tukey's HSD post hoc was used to separate the treatment means.

6.2 Results and Discussions

6.2.1 Influence of grazing management on N_2O fluxes

Significantly higher soil nitrous oxide fluxes ($p \leq 0.05$) were obtained in the continuously grazed site (677.9±130.1gmha^{-1} 3 months) compared to both rotationally grazed and ungrazed site (208.6±127.3 and 162.2±150.3 gmha^{-1} 3 months), respectively, with the ungrazed area having the lowest nitrous oxide fluxes (Figure 9).

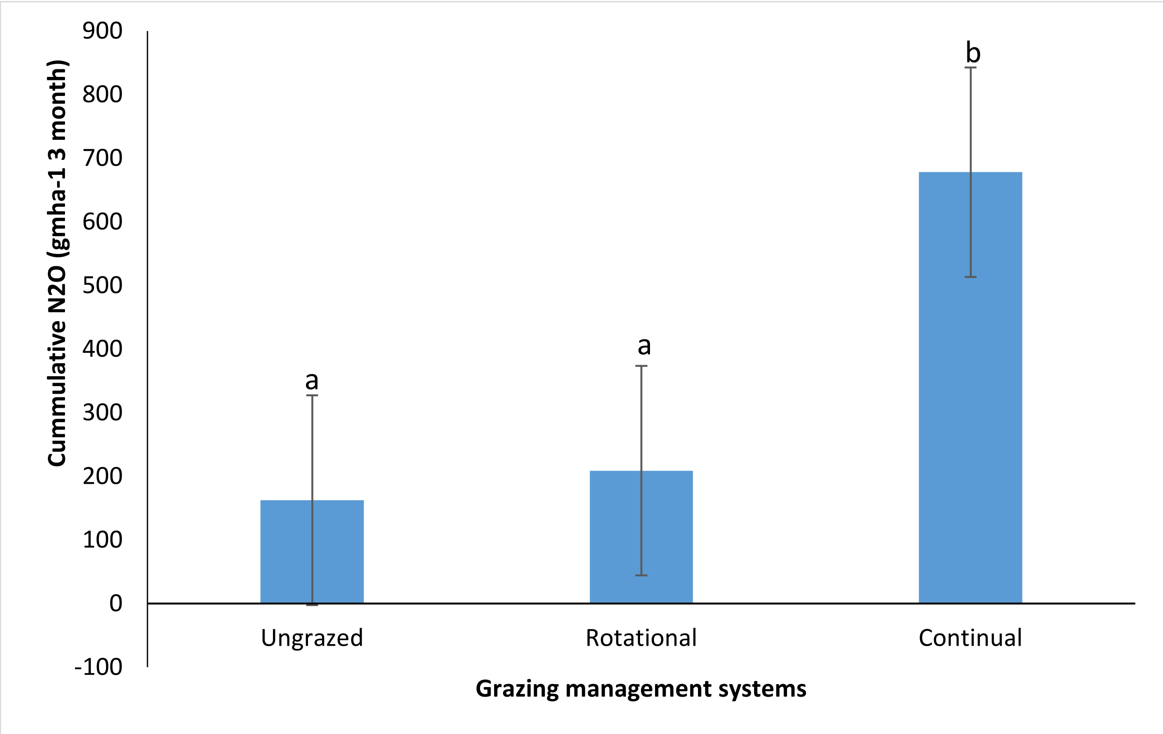


Figure 9: Cumulative flux of N_2O across grazing management systems.

The high N_2O emission rates observed in grazed pastures than ungrazed area can be attributed to N and C from the deposition of animal excreta to the soil and anaerobic conditions as a consequence of soil compaction caused by animal treading. Trampling compacts the soil affecting the abiotic soil characteristics such as pore size, soil moisture and soil aeration. The low air filled pores as a result of soil compaction leads to a decreased oxygen concentration and more anaerobic conditions which can result in high denitrification, thus leading to a significant loss of inorganic N via gaseous emissions. Similar results were found by Douglas and Crawford (1993) who reported N_20 emissions and denitrification rates were two times higher in compacted than in uncompacted grassland soils. The same results were also reported by Torbert and Wood (1992) who found an increase in bulk density resulting to increased total "N losses by a factor of 3.6 and attributed this to denitrification as a major cause of the observed N loss. Accordingly, moisture effects on soil nitrous oxide fluxes are a result of the limitations of oxygen diffusion into the soil and expansion of soil anaerobiosis, which in turn promotes reductive microbial processes such as denitrification under continual grazing system (Patra *et al*., 2005)

The observed high N_2O fluxes in the continual grazing system can also be attributable to the low vegetation cover which may have led to high soil temperatures. Continuous grazing system reduces above ground vegetation cover compared to rotationally grazing management system due to high grazing pressure on continual basis throughout the year. The low vegetation cover leads to increased soil temperatures under continual grazing which increase N_2O emissions. This can be explained by an expansion in the soils pores that are triggered by the accelerated soil respiration leading to increased denitrification rate, and the temperature sensitivity of the underlying enzymatic processes causing N_2O release. Grazing livestock reduce aboveground herbaceous cover which increases evaporation at the soil surface, resulting in higher salinity and increase in denitrification rates Our observation of high fluxes from continual grazing agrees with other previous studies which have reported an increase in nitrous oxide fluxes with increase in soil temperature (Brumme, 1995; Dinsmore *et al.*, 2009; Dobbie and Smith, 2003; Schindlbacher *et al*., 2004). The present study findings collaborates with those recorded in a temperate steppe grassland in Inner Mongolia (Du 2006), but differ from those reported by Jiang *et al.* (2010) working in the alpine grassland, where N_2O emissions were found to peak under higher soil moisture conditions.

The high emissions from continual grazing can also be attributable to high grazing frequency because of the presence of active hot spots of urine and dung depositions, due to continuous grazing throughout the year. According to Nunez and others (2007) the nutrient cycle is influenced by grazing animals who can return as much as 80% of consumed N in the form of dung and urine. Other studies, for example, Wolf and others (2010), found higher N2O emissions with an increase in stocking rate (LU ha^{-1}). Restricted grazing not only reduces N-input from urine, but also reduces hoof compaction of the soil. Luo and others (2008) reported a total reduction of N_2O emission by 7–11% under restricted grazing regimes. Similarly, (Nunez and others 2007) in their study of effects stocking rate on soil nitrogen, reported that, the application of the concept of carrying capacity (the number of animals that the pasture can support) can significantly reduce the N_2O losses from soils.

6.2.2 Influence of grazing systems on soil CO_2 fluxes

The continually grazed site showed significantly higher CO_2 fluxes ($P \leq 0.05$) than both rotational and ungrazed sites (Fig 10). Rotationally grazed site had slightly higher CO_2 fluxes than the ungrazed site though were not statistically significant ($P \leq 0.05$). The three month cumulative emissions for continual, rotational grazing and ungrazed area were 2357±122.1, 1285±123.9 and 1241±143 CO_2 (kg ha^{-1} 3 month) respectively.

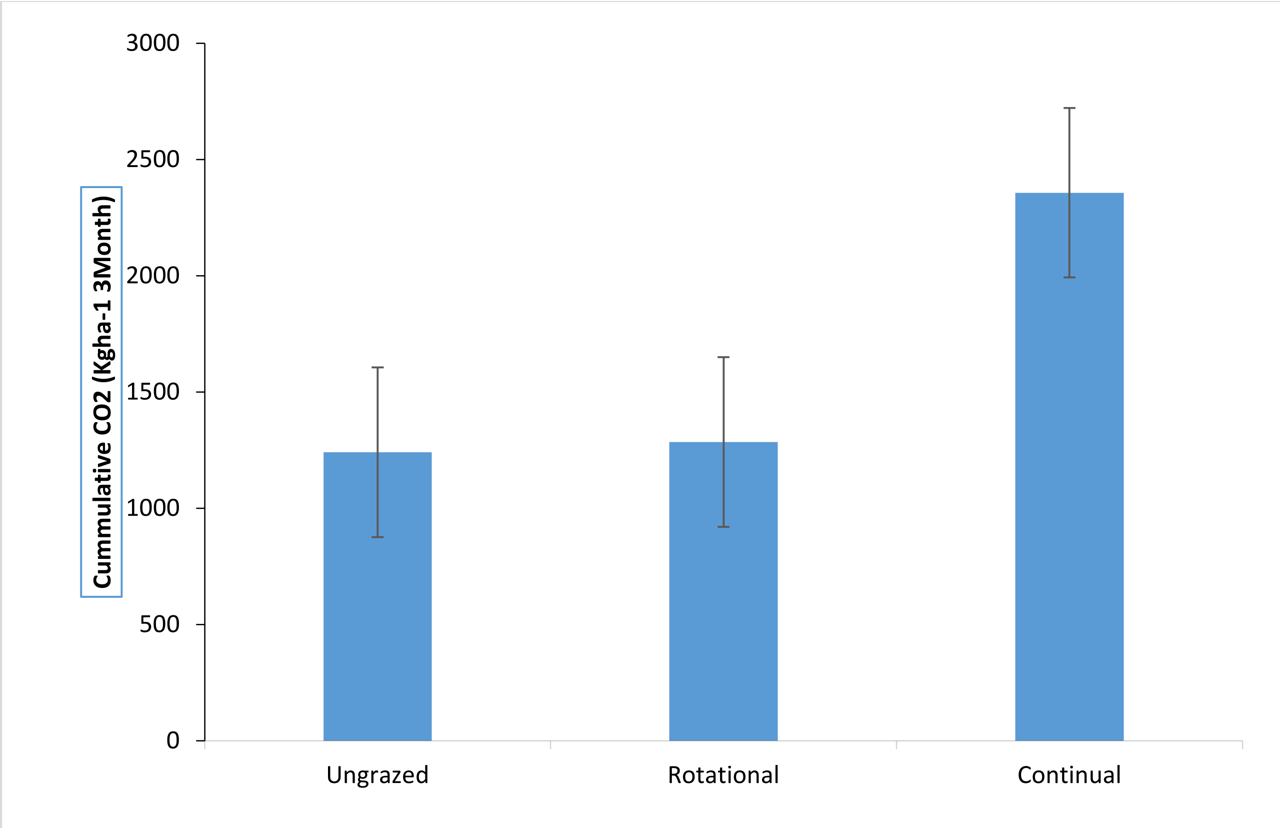


Figure 10: Cumulative fluxes of CO_2 across grazing management systems.

The observed high $C0_2$ fluxes in continuous grazing system can be attributed to the effects of grazing livestock on soil temperature, soil moisture, plant litter and the amount of urine and dung deposition which influences the $C0_2$ fluxes. Urine and dung deposition acts as hot spots for the emissions of $CO_{2.}$ Soil temperature influences enzyme kinetics and metabolic turnover rates of nitrifiers and denitrifiers. Microbial activity and organic matter mineralization are higher at higher soil temperature, thereby resulting in higher CO_2 emissions. The observed low soil CO_2 fluxes under the rotational grazing and ungrazed land can be attributed to shedding effect that results from adequate herbaceous and woody vegetation cover, respectively, which tends to reduce the soil temperatures. The low herbaceous plant cover under the continual grazing system exposes soils to sun rays leading to increased soil temperatures and evapotranspiration rates hence increased decomposition of organic matter resulting to carbon loss from the soil. Moreover, the high carbon dioxide emissions under continual grazing can likely be attributed to decrease in photosynthesis and carbon transportation to the roots occasioned by the low vegetation as a result of grazing pressure. The high CO_2 emissions under the continual grazing can also be attributed to the grazing influence on topsoil carbon sequestration. Mechanism such as trampling induces soil compaction and deterioration of the soil structure. This is followed by mineralization of organic carbon which is released due to destruction of macro-aggregate structures.

Similar results were reported by Zhou, (2013) in a study where it was reported reported that, at constant temperature, wetter soils emitted more CO_2 due to better conditions for microbial respiration. This observation is only true until the point of water saturation when CO_2 fluxes tend to decrease, as those conditions favor the development of anaerobiosis, slowing down the decomposition of organic matter and reducing CO_2 diffusion to the atmosphere (Knowles *et al.,* 2015).

6.2.3 Grazing management influence on CH_4 fluxes

The three-month cumulative emission of CH_4 was highest at continually grazed site and lowest at ungrazed area, i.e. the emissions for continual, rotational grazing and ungrazed area were, 232.7±126.6, 173.1±126.6 and 80±46.2 (gm ha^{-1} 3 month) respectively (Fig11).

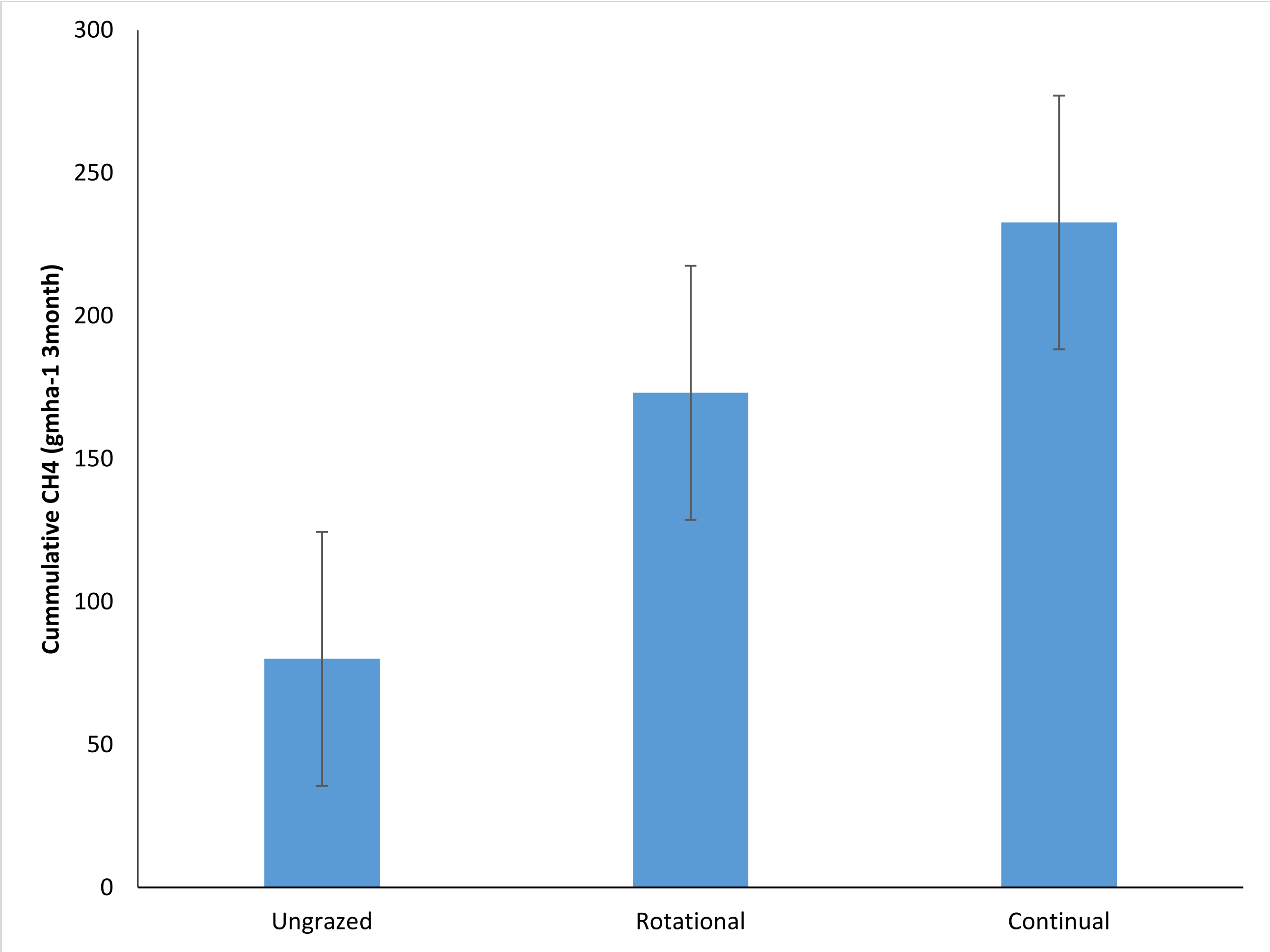


Figure 11: Cumulative flux of CH4 across grazing management systems

The reduced production of CH_4 in areas under rotational grazing and ungrazed sites can be attributed to low soil bulk density from grazing effects which is important for CH_4 uptake since atmospheric CH_4 is the only source for methanotrophs communities in the soils. When the diffusion rate of CH_4 from the atmosphere to soil layers is affected by trampling, the oxidation rate of CH_4 is also affected as a consequence. Thus, an improvement in the physical structure of the soil in the form of a lower bulk density is likely to be the main reason for the reduction of CH_4 production in the rotationally and ungrazed sites. Our findings collaborates with those done by Shi (2017) on the effects of grazing on CO_2, CH_4, and N_2O fluxes in three temperate steppe ecosystems reported an increase in CH_4 emissions with increased bulk density due to trampling by livestock.

The high emission under continual grazing system could be due to high soil moisture caused by changes in soil bulk density. The low air filled porosity as a result of soil compaction leads to more anaerobic conditions which lowers the diffusion rate between the atmosphere and soil, as the only way for CH_4 absorption by methanotrophs, hence leading to emission of CH_4. Our findings collaborates a study done by Paz-Ferreiro (2012) who reported an increase in CH_4 emissions with increasing grazing intensity in temperate grasslands.

The higher emissions under continually grazed area can be due to the fact that cattle are always present in the area hence continuously depositing dung and urine on daily basis thus increasing the emissions of CH_4. Grazing animals on continuous basis reduces standing biomass and vegetation cover which ultimately increase soil evaporation hence an increased water stress, which could inhibit the activities of methanotrophs. The difference in CH4 emissions between the continual and rotational grazing systems can be attributed also to the populations of soil methanotrophs .Zhou *et al*. (2008) found that the population of methanotrophs were lower in heavily grazed sites compared with the lightly and moderately grazed sites.

6.2.4 Greenhouse gas balance across grazing management systems

Based on the $C0_2$ equivalent calculations, the study revealed that the continual grazing management regime emits up to 87kgha^{-1} CO_2 equivalent of the greenhouse gases, while the rotational grazing system and ungrazed areas leads to carbon sequestration with a negative carbon balance of (-54.47kgha^{-1} and -105.56Kgha^{-1}) respectively (Fig 12).

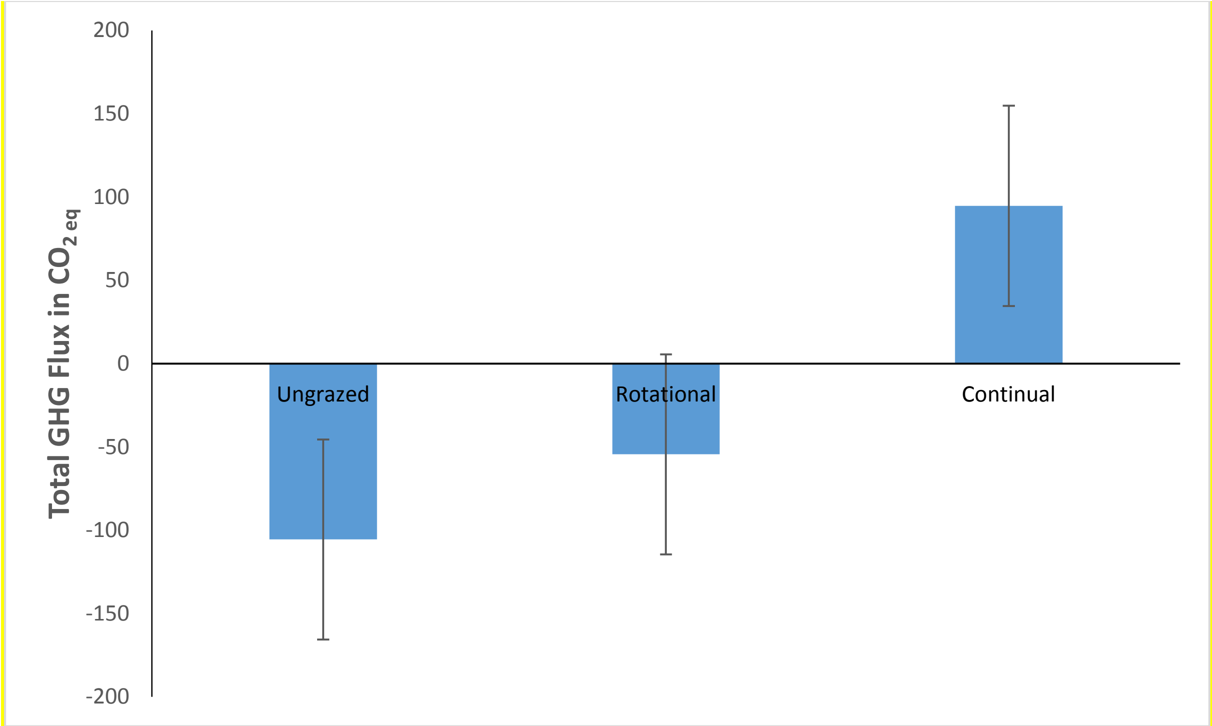


Figure 12: Total GHG balances across grazing management systems

Our results clearly demonstrated the need for taking into account the fluxes of all three gases (CO_2, N_2O and CH_4) when calculating the greenhouse gas balance of a grassland. As expected, reducing grazing pressure strongly reduced both GHG emissions. Hence, grassland management methods that reduce herbaceous vegetation cover, biomass production and increases soil bulk density, may not be appropriate as a mitigation option in semi-arid grasslands.

6.3 Conclusion

Despite the relatively short study period, the findings did indicate that Continuous grazing system had the highest emissions of the three greenhouse gases followed by rotational grazing system while the ungrazed site had the lowest emissions. The results suggest that the continual livestock grazing system leads to high soil emissions of GHG compared to rotational grazing and that grazing exclusion holds large potential to restore the semi-arid soils to sequester atmospheric GHGs. The balance between grazing management system predominantly determines GHG balances of grass-based livestock farming in this region. Therefore, for reduced GHG balance of grazing lands, rotational management system needs to be adopted.

CHAPTER SEVEN: GENERAL DISCUSSION, CONCLUSIONS AND RECOMMENDATIONS

7.1 General discussion

The ungrazed site recorded significantly ($p \leq 0.05$) higher soil organic carbon concentrations and stocks followed by the rotationally grazed area and lowest continuously grazed sites for all soil layers up to 1.2m depth. The predicted results showed that the rate of SOC stock [t/ha] change was positively higher under rotational grazing system followed by ungrazed and continual grazing system respectively for the modelling period of 2015-2064. The higher aboveground standing biomass under rotational grazing and ungrazed sites significantly reduced loss of organic matter and nutrients from the soil-plant system through soil erosion. The presence of three which create a microclimate in areas under rotational grazing and ungrazed can also explain the high soil organic carbon due their effect on soil moisture and temperatures 7). The difference in SOC content and stocks across the grazing systems can also be attributed to the difference in the herbaceous vegetation cover Furthermore, less pore space limits gas exchange and cut back root growth. The two mechanisms counsel that soil compaction reduces plant production and thus SOC storage. This findings collaborates with those of Sanjari et al, (2008), who pointed out that, a relative increase in soil organic matter under time controlled grazing as opposed to under continuous grazing was attributed to higher rates of grass growth and rest periods that lead to an increase in litter accumulation and subsequent increase soil organic carbon under time rotational grazing compared with the continuous grazing.

Rotationally grazed area had higher biomass production, herbaceous cover and species diversity and richness followed by continually grazed area while the ungrazed site had the lowest. Improved biomass yields, cover and high species diversity in rotation grazed areas was largely attributed to the flexibility in the management in which grazing frequency, durations and the rest periods are efficiently controlled compared to continuous grazing areas.

The emissions of the three greenhouse gases (CO_2, N_2O and CH_4) were higher in continually grazed area, followed by rotationally grazed area and lowest in the ungrazed area. The observed high fluxes in continuous grazing system can be attributed to the effects of grazing livestock on soil temperature, soil moisture, plant litter and the amount of urine and dung deposition which influences the GHG fluxes. Soil temperature influences enzyme kinetics and metabolic turnover rates of nitrifiers and denitrifiers (Lecain et al., 2000). Microbial activity and organic matter mineralization are higher at higher soil temperature, thereby resulting in higher CO_2, N_2O and CH_4 emissions. Our findings collaborates with those done by Shi (2017) on the effects of grazing on CO_2, CH_4, and N_2O fluxes in three temperate steppe ecosystems reported an increase in GHG emissions with increased bulk density due to trampling by livestock under continuous grazing system.

7.2 General Conclusions

The current study was carried out to investigate the effect grazing management practices on biomass production, cover, species diversity, soil organic carbon content and stocks and greenhouse gas emissions. The rotationally grazed area had higher biomass production, cover and species diversity followed by the continuous grazing system and ungrazed site respectively. In terms of soil organic carbon, the ungrazed site had the highest soil organic carbon content followed by the rotational grazing system while the continuous grazing system had the least. The emissions of GHG (CO_2, N_2O and CH_4) were highest under continuous grazing system, followed by rotational grazing system and least in ungrazed site. The present study suggests that rotational grazing is a viable management tool for the productivity of livestock-disturbed rangeland ecosystems that are ecologically vulnerable to overgrazing and degradation. The balance between grazing management system predominantly determines GHG balances of grass-based livestock farming in this region. Therefore, for reducing GHG balances of grazing lands, rotational management system needs to be adopted.

7.3 General Recommendations

- To increase soil organic carbon and primary productivity in semi-arid grasslands in Kenya, rotational grazing management should be encouraged.
- A study needs to be done to assess the temporal and spatial effects of grazing management systems on greenhouse gas fluxes in semi-arid grasslands due to their heterogeneity nature.
- Soil microorganisms play a vital role in determining the organic matter dynamics in the soil and greenhouse gas emissions, this study did not investigate the effect of the different grazing management on soil microorganism. Therefore, further research on the effects of grazing management on soil microorganisms would help to further reveal the mechanisms underlying the observed enhancement of the measured soil organic carbon and the low soil emissions of the three greenhouse gasses under rotational grazing management.

REFERENCES

Abel, N., and Blaikie, P. (1989). Land degradation, stocking rates and conservation policies in the communal rangelands of Botswana and Zimbabwe. *Land Degradation and Development, 1*(2), 101-123.

Adler, P. B., Milchunas, D. G., Sala, O. E., Burke, I. C., and Lauenroth, W. K. (2005). Plant traits and ecosystem grazing effects: comparison of US sagebrush steppe and Patagonian steppe. *Ecological Applications, 15*(2), 774-792.

Allard, V., Soussana, J.-F., Falcimagne, R., Berbigier, P., Bonnefond, J.-M., Ceschia, E., . . . Martin, C. (2007). The role of grazing management for the net biome productivity and greenhouse gas budget (CO 2, N 2 O and CH 4) of semi-natural grassland. *Agriculture, Ecosystems and Environment, 121*(1), 47-58.

Alphayo, L. (2015). *Effects of holistic grazing management on soil physico-chemical properties and herbaceous vegetation production in Naibunga Conservancy, Laikipia County, Kenya.* University of Nairobi.

Ambus, P., and Christensen, S. (1995). Spatial and seasonal nitrous oxide and methane fluxes in Danish forest-, grassland-, and agroecosystems. *Journal of Environmental Quality, 24*(5), 993-1001.

Asner, G. P., Elmore, A. J., Olander, L. P., Martin, R. E., and Harris, A. T. (2004). Grazing systems, ecosystem responses, and global change. *Annu. Rev. Environ. Resour., 29*, 261-299.

Bakker, E. S., Ritchie, M. E., Olff, H., Milchunas, D. G., and Knops, J. M. (2006). Herbivore impact on grassland plant diversity depends on habitat productivity and herbivore size. *Ecology Letters, 9*(7), 780-788.

Barger, N. N., Ojima, D. S., Belnap, J., Shiping, W., Yanfen, W., and Chen, Z. (2004). Changes in plant functional groups, litter quality, and soil carbon and nitrogen mineralization with sheep grazing in an Inner Mongolian grassland. *Journal of Range Management, 57*(6), 613-619.

Bartolome, J. W., and McClaran, M. P. (1992). Composition and production of California oak savanna seasonally grazed by sheep. *Journal of Range Management*, 103-107.

Batjes, N. H. (1996). Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science, 47*(2), 151-163.

Bayer, C., Mielniczuk, J., Martin-Neto, L., and Ernani, P. R. (2002). Stocks and humification degree of organic matter fractions as affected by no-tillage on a subtropical soil. *Plant and Soil, 238*(1), 133-140.

Beauchemin, K. A., Janzen, H. H., Little, S. M., McAllister, T. A., and McGinn, S. M. (2010). Life cycle assessment of greenhouse gas emissions from beef production in western Canada: A case study. *Agricultural Systems, 103*(6), 371-379.

Bell, L. W., Kirkegaard, J. A., Swan, A., Hunt, J. R., Huth, N. I., and Fettell, N. A. (2011). Impacts of soil damage by grazing livestock on crop productivity. *Soil and Tillage Research, 113*(1), 19-29.

Belsky, A. J. (1992). Effects of grazing, competition, disturbance and fire on species composition and diversity in grassland communities. *Journal of Vegetation Science, 3*(2), 187-200.

Bergman, C. M., Fryxell, J. M., Gates, C. C., and Fortin, D. (2001). Ungulate foraging strategies: energy maximizing or time minimizing? *Journal of Animal Ecology, 70*(2), 289-300.

Bilotta, G., Brazier, R., and Haygarth, P. (2007). The impacts of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed grasslands. *Advances in agronomy, 94*, 237-280.

Blake, G. (1965). Bulk density. *Methods of Soil Analysis. Part 1. Physical and Mineralogical Properties, Including Statistics of Measurement and Sampling*(methodsofsoilana), 374-390.

Bolwig, S., Ponte, S., Du Toit, A., Riisgaard, L., and Halberg, N. (2010). Integrating poverty and environmental concerns into value-chain analysis: a conceptual framework. *Development Policy Review, 28*(2), 173-194.

Bombelli, A., Henry, M., Castaldi, S., Adu-Bredu, S., Arneth, A., De Grandcourt, A., . . . Rasile, G. (2009). The Sub-Saharan Africa carbon balance, an overview.

Booker, K., Huntsinger, L., Bartolome, J. W., Sayre, N. F., and Stewart, W. (2013). What can ecological science tell us about opportunities for carbon sequestration on arid rangelands in the United States? *Global Environmental Change, 23*(1), 240-251.
Bremer, D. J., Auen, L. M., Ham, J. M., and Owensby, C. E. (2001). Evapotranspiration in a Prairie Ecosystem. *Agronomy Journal, 93*(2), 338-348.
Briske, D. D., Derner, J., Brown, J., Fuhlendorf, S., Teague, W., Havstad, K., . . . Willms, W. (2008). Rotational grazing on rangelands: reconciliation of perception and experimental evidence. *Rangeland Ecology and Management, 61*(1), 3-17.
Brown, J. R., and Thorpe, J. (2008). Climate change and rangelands: responding rationally to uncertainty. *Rangelands, 30*(3), 3-6.
Brumme, R. (1995). Mechanisms of carbon and nutrient release and retention in beech forest gaps. *Plant and Soil, 168*(1), 593-600.
Buckeridge, K. M., and Jefferies, R. L. (2007). Vegetation loss alters soil nitrogen dynamics in an Arctic salt marsh. *Journal of Ecology, 95*(2), 283-293.
Butterbach-Bahl, K., Kiese, R., and Liu, C. (2011). Measurements of biosphere–atmosphere exchange of CH4 in terrestrial ecosystems. *Methods Enzymol, 495*, 271-287.
Butterbach-Bahl, K., Sander, B. O., Pelster, D., and Díaz-Pinés, E. (2016). Quantifying greenhouse gas emissions from managed and natural soils *Methods for Measuring Greenhouse Gas Balances and Evaluating Mitigation Options in Smallholder Agriculture* (pp. 71-96): Springer.
Catford, J. A., Daehler, C. C., Murphy, H. T., Sheppard, A. W., Hardesty, B. D., Westcott, D. A., . . . Horvitz, C. C. (2012). The intermediate disturbance hypothesis and plant invasions: Implications for species richness and management. *Perspectives in Plant Ecology, Evolution and Systematics, 14*(3), 231-241.
Change, I. P. o. C. (2014a). *Climate Change 2014–Impacts, Adaptation and Vulnerability: Regional Aspects*: Cambridge University Press.
Change, I. P. O. C. (2014b). IPCC. *Climate change*.
Chou, W. W., Silver, W. L., Jackson, R. D., Thompson, A. W., and ALLEN-DIAZ, B. (2008). The sensitivity of annual grassland carbon cycling to the quantity and timing of rainfall. *Global change biology, 14*(6), 1382-1394.
Ciais, P., Gervois, S., Vuichard, N., Piao, S., and Viovy, N. (2011). Effects of land use change and management on the European cropland carbon balance. *Global change biology, 17*(1), 320-338.
Cingolani, A. M., Noy-Meir, I., and Díaz, S. (2005). Grazing effects on rangeland diversity: a synthesis of contemporary models. *Ecological Applications, 15*(2), 757-773.
Coleman, K., and Jenkinson, D. (1996). RothC-26.3-A Model for the turnover of carbon in soil *Evaluation of soil organic matter models* (pp. 237-246): Springer.
Coleman, K., Jenkinson, D., Crocker, G., Grace, P., Klir, J., Körschens, M., . . . Richter, D. (1997). Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. *Geoderma, 81*(1-2), 29-44.
Collins, S. L., Knapp, A. K., Briggs, J. M., Blair, J. M., and Steinauer, E. M. (1998). Modulation of diversity by grazing and mowing in native tallgrass prairie. *science, 280*(5364), 745-747.
Conant, R. T., Paustian, K., and Elliott, E. T. (2001). Grassland management and conversion into grassland: effects on soil carbon. *Ecological Applications, 11*(2), 343-355.
Connell, J. H. (1978). Diversity in tropical rain forests and coral reefs. *science, 199*(4335), 1302-1310.
Couwenberg, J., Thiele, A., Tanneberger, F., Augustin, J., Bärisch, S., Dubovik, D., . . . Skuratovich, A. (2011). Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. *Hydrobiologia, 674*(1), 67-89.
Craswell, E., and Lefroy, R. (2001). The role and function of organic matter in tropical soils *Managing Organic Matter in Tropical Soils: Scope and Limitations* (pp. 7-18): Springer.
Curran Cournane, F. (2010). *Impacts of livestock grazing on soil physical quality and phosphorus and suspended sediment losses in surface runoff.* Lincoln University.

Daniel, J., Potter, K., Altom, W., Aljoe, H., and Stevens, R. (2002). Long-term grazing density impacts on soil compaction. *Transactions of the American Society of Agricultural Engineers, 45*(6), 1911-1915.

Davidson, E., Hart, S., Shanks, C., and Firestone, M. (1991). Measuring gross nitrogen mineralization, and nitrification by 15 N isotopic pool dilution in intact soil cores. *Journal of Soil Science, 42*(3), 335-349.

Davidson, E. A., and Janssens, I. A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature, 440*(7081), 165-173.

De Deyn, G. B., Cornelissen, J. H., and Bardgett, R. D. (2008). Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecology Letters, 11*(5), 516-531.

DeLonge, M. S., Owen, J. J., and Silver, W. L. (2014). Greenhouse Gas Mitigation Opportunities in California Agriculture.

Derner, J., and Schuman, G. (2007). Carbon sequestration and rangelands: a synthesis of land management and precipitation effects. *Journal of Soil and Water Conservation, 62*(2), 77-85.

Díaz-Hernández, J. L., Fernández, E. B., and González, J. L. (2003). Organic and inorganic carbon in soils of semiarid regions: a case study from the Guadix–Baza basin (Southeast Spain). *Geoderma, 114*(1), 65-80.

Dinsmore, K. J., Skiba, U. M., Billett, M. F., Rees, R. M., and Drewer, J. (2009). Spatial and temporal variability in CH 4 and N 2 O fluxes from a Scottish ombrotrophic peatland: implications for modelling and up-scaling. *Soil Biology and Biochemistry, 41*(6), 1315-1323.

Dobbie, K. E., and Smith, K. A. (2003). Nitrous oxide emission factors for agricultural soils in Great Britain: The impact of soil water-filled pore space and other controlling variables. *Global change biology, 9*(2), 204-218.

Dong, Y., Zhang, S., Qi, Y., Chen, Z., and Geng, Y. (2000). Fluxes of CO2, N2O and CH4 from a typical temperate grassland in Inner Mongolia and its daily variation. *Chinese Science Bulletin, 45*(17), 1590-1594.

Dudney, J., Hallett, L. M., Larios, L., Farrer, E. C., Spotswood, E. N., Stein, C., and Suding, K. N. (2016). Lagging behind: have we overlooked previous-year rainfall effects in annual grasslands? *Journal of Ecology*.

Einstein, A. (2010). The Future of Big Ecology: IGBP, AmeriFlux, NEON, and Other Major Initiatives. *Big Ecology: The Emergence of Ecosystem Science*, 145.

Elsgaard, L., Görres, C.-M., Hoffmann, C. C., Blicher-Mathiesen, G., Schelde, K., and Petersen, S. O. (2012). Net ecosystem exchange of CO 2 and carbon balance for eight temperate organic soils under agricultural management. *Agriculture, Ecosystems and Environment, 162*, 52-67.

Evans, N., Avis, A., and Palmer, A. (1997). Changes to the vegetation of the mid-Fish River valley, Eastern Cape, South Africa, in response to land-use, as revealed by a direct gradient analysis. *African Journal of Range and Forage Science, 14*(2), 68-74.

Falloon, P., and Smith, P. (2002). Simulating SOC changes in long-term experiments with RothC and CENTURY: model evaluation for a regional scale application. *Soil use and management, 18*(2), 101-111.

FAO, U. (2014). FAOstat. *Retrieved Feb, 2014*.

Faostat, F. (2009). Statistical databases. *Food and Agriculture Organization of the United Nations, Rome*.

Farage, P., Ardö, J., Olsson, L., Rienzi, E., Ball, A., and Pretty, J. (2007). The potential for soil carbon sequestration in three tropical dryland farming systems of Africa and Latin America: A modelling approach. *Soil and Tillage Research, 94*(2), 457-472.

Favretto, N., Stringer, L., Dougill, A., Dallimer, M., Perkins, J., Reed, M., . . . Mulale, K. (2016). Multi-Criteria Decision Analysis to identify dryland ecosystem service trade-offs under different rangeland land uses. *Ecosystem Services, 17*, 142-151.

Fensham, R., Fairfax, R., and Dwyer, J. (2010). Vegetation responses to the first 20 years of cattle grazing in an Australian desert. *Ecology, 91*(3), 681-692.

Follett, R. F., and Reed, D. A. (2010). Soil Carbon Sequestration in Grazing Lands: Societal Benefits and Policy Implications. *Rangeland Ecology and Management, 63*(1), 4-15. doi: http://dx.doi.org/10.2111/08-225.1

Ford, R. M. (2015). *THE EFFECTS OF FIRE AND GRAZING MANAGEMENT ON UNPALATABLE CLIMAX GRASSLANDS DOMINATED BY HYPARRHENIA HIRTA AND CYMBOPOGON VALIDUS.* University of Pretoria.

Fortin, D., Fryxell, J. M., O'brodovich, L., and Frandsen, D. (2003). Foraging ecology of bison at the landscape and plant community levels: the applicability of energy maximization principles. *Oecologia, 134*(2), 219-227.

Galbally, I. E., Kirstine, W. V., Meyer, C., and Wang, Y. P. (2008). Soil–atmosphere trace gas exchange in semiarid and arid zones. *Journal of Environmental Quality, 37*(2), 599-607.

Galvin, K. A. (2009). Transitions: pastoralists living with change. *Annual Review of Anthropology, 38*, 185-198.

Gamoun, M. (2014). Grazing intensity effects on the vegetation in desert rangelands of Southern Tunisia. *J Arid Land, 6*(3), 324-333.

Gao, Y., Schumann, M., Chen, H., Wu, N., and Luo, P. (2009). Impacts of grazing intensity on soil carbon and nitrogen in an alpine meadow on the eastern Tibetan Plateau. *Journal of Food, Agriculture and Environment, 7*(2), 749-754.

Garg, R., and Vyas, L. (1975). Litter production in deciduous forest near Udaipur (south Rajasthan), India *Tropical ecological systems* (pp. 131-135): Springer.

Gebremeskel, K. (2006). *Rangeland potential, quality and restoration strategies in north-eastern Ethiopia: a case study conducted in the southern Afar region.* Stellenbosch: University of Stellenbosch.

Gill, M., Smith, P., and Wilkinson, J. (2010). Mitigating climate change: the role of domestic livestock. *Animal, 4*(03), 323-333.

Granli, T., and Bøckman, O. C. (1994). Nitrous oxide from agriculture. *Norwegian Journal of Agricultural Sciences (Norway)*.

Grayston, S., Vaughan, D., and Jones, D. (1997). Rhizosphere carbon flow in trees, in comparison with annual plants: the importance of root exudation and its impact on microbial activity and nutrient availability. *Applied soil ecology, 5*(1), 29-56.

Green, R. A., and Detling, J. K. (2000). Defoliation-induced enhancement of total aboveground nitrogen yield of grasses. *Oikos, 91*(2), 280-284.

Greenwood, K., and McKenzie, B. (2001). Grazing effects on soil physical properties and the consequences for pastures: a review. *Animal Production Science, 41*(8), 1231-1250.

Gregorich, E., Beare, M., McKim, U., and Skjemstad, J. (2006). Chemical and biological characteristics of physically uncomplexed organic matter. *Soil Science Society of America Journal, 70*(3), 975-985.

Hamilton, J. G., Griffin, J. R., and Stromberg, M. R. (2002). Long-term population dynamics of native Nassella (Poaceae) bunchgrasses in Central California. *Madroño*, 274-284.

Han, G., Hao, X., Zhao, M., Wang, M., Ellert, B. H., Willms, W., and 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*(1), 21-32.

Heitschmidt, R., Vermeire, L., and Grings, E. (2004). Is rangeland agriculture sustainable? *Journal of Animal Science, 82*(13_suppl), E138-E146.

Herrero, M., Ringler, C., Steeg, J., Thornton, P., Zhu, T., Bryan, E., . . . Notenbaert, A. (2010). Climate variability and climate change and their impacts on Kenya's agricultural sector.

Hickman, J. E., Scholes, R. J., Rosenstock, T. S., García-Pando, C. P., and Nyamangara, J. (2014). Assessing non-CO 2 climate-forcing emissions and mitigation in sub-Saharan Africa. *Current Opinion in Environmental Sustainability, 9*, 65-72.

HilleRisLambers, J., Yelenik, S. G., Colman, B. P., and Levine, J. M. (2010). California annual grass invaders: the drivers or passengers of change? *Journal of Ecology, 98*(5), 1147-1156.

Hoffman, T., and Vogel, C. (2008). Climate change impacts on African rangelands. *Rangelands, 30*(3), 12-17.

Holechek, J. L. (2001). Western ranching at the crossroads. *Rangelands*, 17-21.

Houghton, J., Ding, Y., Griggs, D., Noguer, M., Van der Linden, P., Dai, X., . . . Johnson, C. (2001). IPCC third assessment report: climate change 2001. *Cam-bridge University Press. HoughtonIPCC Third assessment report: climate change, 20012001*.

Huang, X., McNeill, M., and Zhang, Z. (2015). Quantitative analysis of plant consumption and preference by Oedaleus asiaticus (Acrididae: Oedipodinae) in changed plant communities consisting of three grass species. *Environmental entomology*, nvv172.

Huntsinger, L., Bartolome, J. W., and D'Antonio, C. M. (2007). Grazing management on California's Mediterranean grasslands. *California grasslands*, 233-253.

Ingram, L., Stahl, P., Schuman, G., Buyer, J., Vance, G., Ganjegunte, G., . . . Derner, J. (2008). Grazing impacts on soil carbon and microbial communities in a mixed-grass ecosystem. *Soil Science Society of America Journal, 72*(4), 939-948.

Jackson, R. D., and Bartolome, J. W. (2007). Grazing ecology of California grasslands. *California grasslands: ecology and management*, 197-206.

Jacobo, Elizabeth, J., Rodríguez, Adriana, M., Bartoloni, Norberto, . . . Víctor, A. (2006). Rotational grazing effects on rangeland vegetation at a farm scale. *Rangeland Ecology and Management, 59*(3), 249-257.

Jacobo, E. J., Rodríguez, A. M., Bartoloni, N., and Deregibus, V. A. (2006). Rotational grazing effects on rangeland vegetation at a farm scale. *Rangeland Ecology and Management, 59*(3), 249-257.

Janssens, I. A., Freibauer, A., Ciais, P., Smith, P., Nabuurs, G.-J., Folberth, G., . . . Schulze, E.-D. (2003). Europe's terrestrial biosphere absorbs 7 to 12% of European anthropogenic CO2 emissions. *science, 300*(5625), 1538-1542.

Jeddi, K., and Chaieb, M. (2010). Changes in soil properties and vegetation following livestock grazing exclusion in degraded arid environments of South Tunisia. *Flora-Morphology, Distribution, Functional Ecology of Plants, 205*(3), 184-189.

Jenkinson, D. S., Adams, D., and Wild, A. (1991). Model Estimates of CO (2) Emissions from Soil in Response to Global Warming. *Nature, 351*(6324), 304.

Jobbágy, E. G., and Jackson, R. B. (2000). The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications, 10*(2), 423-436.

Jones, C., McConnell, C., Coleman, K., Cox, P., Falloon, P., Jenkinson, D., and Powlson, D. (2005). Global climate change and soil carbon stocks; predictions from two contrasting models for the turnover of organic carbon in soil. *Global change biology, 11*(1), 154-166.

Juma, N., Izaurralde, R., Robertson, J., and McGill, W. (1996). Crop yield and soil organic matter trends over 60 years in a Typic Cryoboralf at Breton, Alberta. *Soil organic matter in temperate agroecosystems. Long-term experiments in North America. CRC Press, Boca Raton, FL. Crop yield and soil organic matter trends over, 60*, 273-281.

Kamau, P. (2003). Forage diversity and impact of grazing management on rangeland ecosystems in Mbeere District, Kenya.

Kammann, C., Grünhage, L., Müller, C., Jacobi, S., and Jäger, H.-J. (1998). Seasonal variability and mitigation options for N2O emissions from differently managed grasslands. *Environmental pollution, 102*(1), 179-186.

Kanga, E. M., Ogutu, J. O., Piepho, H.-P., and Olff, H. (2013). Hippopotamus and livestock grazing: influences on riparian vegetation and facilitation of other herbivores in the Mara Region of Kenya. *Landscape and Ecological Engineering, 9*(1), 47-58.

Kassahun, A., Snyman, H., and Smit, G. (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.

Kgosikoma, O. E. (2012). Understanding the savanna dynamics in relation to rangeland management systems and environmental conditions in semi-arid Botswana.

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

Kladivko, E. J. (2001). Tillage systems and soil ecology. *Soil and Tillage Research, 61*(1), 61-76.

Krebs, C. J. (1989). *Ecological methodology*: Harper and Row New York.

Lal, R. (2003). Soil erosion and the global carbon budget. *Environment international, 29*(4), 437-450.

Lal, R. (2004). Soil carbon sequestration impacts on global climate change and food security. *science, 304*(5677), 1623-1627.

Lal, R. (2008). Carbon sequestration. *Philosophical Transactions of the Royal Society B: Biological Sciences, 363*(1492), 815-830.

Le Mer, J., and Roger, P. (2001). Production, oxidation, emission and consumption of methane by soils: a review. *European Journal of Soil Biology, 37*(1), 25-50.

Leake, J., Johnson, D., Donnelly, D., Muckle, G., Boddy, L., and Read, D. (2004). Networks of power and influence: the role of mycorrhizal mycelium in controlling plant communities and agroecosystem functioning. *Canadian Journal of Botany, 82*(8), 1016-1045.

Lecain, D. R., Morgan, J. A., Schuman, G. E., Reeder, J. D., and Hart, R. H. (2000). Carbon exchange rates in grazed and ungrazed pastures of Wyoming. *Journal of Range Management*, 199-206.

Lemus, R. W., and Rivera, D. (2011). *Pasture and Grazing Management Under Drought Conditions*: Mississippi State University Extension Service.

Li, X. L., Gao, J., Brierley, G., Qiao, Y. M., Zhang, J., and Yang, Y. W. (2013). Rangeland degradation on the Qinghai-Tibet plateau: Implications for rehabilitation. *Land Degradation and Development, 24*(1), 72-80.

Li, Z., Han, F., Su, Y., Zhang, T., Sun, B., Monts, D., and Plodinec, M. (2007). Assessment of soil organic and carbonate carbon storage in China. *Geoderma, 138*(1), 119-126.

Liebig, M., Gross, J., Kronberg, S., Hanson, J., Frank, A., and Phillips, R. (2006). Soil response to long-term grazing in the northern Great Plains of North America. *Agriculture, Ecosystems and Environment, 115*(1), 270-276.

Liebig, M., Gross, J., Kronberg, S., and Phillips, R. (2010). Grazing management contributions to net global warming potential: A long-term evaluation in the Northern Great Plains. *Journal of Environmental Quality, 39*(3), 799-809.

Lipiec, J., and Stepniewski, W. (1995). Effects of soil compaction and tillage systems on uptake and losses of nutrients. *Soil and Tillage Research, 35*(1-2), 37-52.

Liu, H., Han, X., Li, L., Huang, J., Liu, H., and Li, X. (2009). Grazing density effects on cover, species composition, and nitrogen fixation of biological soil crust in an inner Mongolia steppe. *Rangeland Ecology and Management, 62*(4), 321-327.

Mackey, R. L., and Currie, D. J. (2001). The diversity–disturbance relationship: is it generally strong and peaked? *Ecology, 82*(12), 3479-3492.

Maestre, F. T., Bowker, M. A., Puche, M. D., Belén Hinojosa, M., Martínez, I., García-Palacios, P., . . . Sánchez, A. M. (2009). Shrub encroachment can reverse desertification in semi-arid Mediterranean grasslands. *Ecology Letters, 12*(9), 930-941.

Maitima, J. M., Mugatha, S. M., Reid, R. S., Gachimbi, L. N., Majule, A., Lyaruu, H., . . . Mugisha, S. (2009). The linkages between land use change, land degradation and biodiversity across East Africa. *African Journal of Environmental Science and Technology, 3*(10).

Maljanen, M., Liikanen, A., Silvola, J., and Martikainen, P. (2003). Methane fluxes on agricultural and forested boreal organic soils. *Soil use and management, 19*(1), 73-79.

Manjarrez-Dominguez, C., Pinedo-Alvarez, A., Pinedo-Alvarez, C., Villarreal-Guerrero, F., and Cortes-Palacios, L. (2015). Vegetation landscape analysis due to land use changes on arid lands. *Polish Journal of Ecology, 63*, 167-174.

Martius, C., Tiessen, H., and Vlek, P. (2001). The management of organic matter in tropical soils: what are the priorities? *Nutrient Cycling in Agroecosystems, 61*(1), 1-6.

Medina-Roldán, E., Paz-Ferreiro, J., and Bardgett, R. D. (2012). Grazing exclusion affects soil and plant communities, but has no impact on soil carbon storage in an upland grassland. *Agriculture, Ecosystems and Environment, 149*, 118-123.

Mekuria, W., and Veldkamp, E. (2012). Restoration of native vegetation following exclosure establishment on communal grazing lands in Tigray, Ethiopia. *Applied Vegetation Science, 15*(1), 71-83.

Metera, E., Sakowski, T., Słoniewski, K., and Romanowicz, B. (2010). Grazing as a tool to maintain biodiversity of grassland-a review. *Animal Science Papers and Reports, 28*(4), 315-334.

Mi, N., Wang, S., Liu, J., Yu, G., Zhang, W., and Jobbagy, E. (2008). Soil inorganic carbon storage pattern in China. *Global change biology, 14*(10), 2380-2387.

Milchunas, D. G. (2006). Responses of plant communities to grazing in the southwestern United States.

Milton, S. J., du Plessis, M. A., and Siegfried, W. R. (1994). A conceptual model of arid rangeland degradation. *Bioscience, 44*(2), 70-76.

Mosier, A. (1998). Soil processes and global change. *Biology and fertility of soils, 27*(3), 221-229.

Neff, J., Barger, N., Baisden, W., Fernandez, D., and Asner, G. (2009). Soil carbon storage responses to expanding pinyon–juniper populations in southern Utah. *Ecological Applications, 19*(6), 1405-1416.

Nichols, K. L. (2007). *Methane and nitrous oxide fluxes from cattle excrement on C3 pasture and C4 native rangeland of the shortgrass steppe.* Colorado State University. Libraries.

Oba, G., Vetaas, O. R., and Stenseth, N. C. (2001). Relationships between biomass and plant species richness in arid-zone grazing lands. *Journal of Applied Ecology, 38*(4), 836-845.

Ogle, K., and Reynolds, J. F. (2004). Plant responses to precipitation in desert ecosystems: integrating functional types, pulses, thresholds, and delays. *Oecologia, 141*(2), 282-294.

Ontl, T. A., Hofmockel, K. S., Cambardella, C. A., Schulte, L. A., and Kolka, R. K. (2013). Topographic and soil influences on root productivity of three bioenergy cropping systems. *New Phytologist, 199*(3), 727-737.

Parton, W., Ojima, D., and Schimel, D. (1995). Models to evaluate soil organic matter storage and dynamics. *Structure and Organic Matter Storage in Agricultural Soils. New York: CRC Press/Lewis*, 421-448.

Parton, W., Schimel, D. S., Cole, C., and Ojima, D. (1987). Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Science Society of America Journal, 51*(5), 1173-1179.

Patra, A., Abbadie, L., Clays-Josserand, A., Degrange, V., Grayston, S., Loiseau, P., . . . Philippot, L. (2005). Effects of grazing on microbial functional groups involved in soil N dynamics. *Ecological monographs, 75*(1), 65-80.

Patra, D., Brookes, P., Coleman, K., and Jenkinson, D. (1990). Seasonal changes of soil microbial biomass in an arable and a grassland soil which have been under uniform management for many years. *Soil Biology and Biochemistry, 22*(6), 739-742.

Patton, B. D., Dong, X., Nyren, P. E., and Nyren, A. (2007). Effects of grazing intensity, precipitation, and temperature on forage production. *Rangeland Ecology and Management, 60*(6), 656-665.

Pekin, B. K., Endress, B. A., Wisdom, M. J., Naylor, B. J., and Parks, C. G. (2015). Impact of ungulate exclusion on understorey succession in relation to forest management in the Intermountain Western United States. *Applied Vegetation Science, 18*(2), 252-260.

Pekin, B. K., Wisdom, M. J., Endress, B. A., Naylor, B. J., and Parks, C. G. (2014). Ungulate browsing maintains shrub diversity in the absence of episodic disturbance in seasonally-arid conifer forest. *PloS one, 9*(1), e86288.

Pelster, D., Rufino, M. C., Rosenstock, T., Mango, J., Saiz, G., Diaz-Pines, E., . . . Butterbach-Bahl, K. (2015). Smallholder African farms in western Kenya have limited greenhouse gas fluxes. *Biogeosciences Discuss, 12*, 15,301-315,336.

Peltoniemi, M., Thurig, E., Ogle, S., Palosuo, T., Schrump, M., Wutzler, T., . . . Mikhailov, A. (2007). Models in country scale carbon accounting of forest soils. *Silva Fennica, 41*(3), 575.

Piñeiro, G., Paruelo, J. M., Oesterheld, M., and Jobbágy, E. G. (2010). Pathways of grazing effects on soil organic carbon and nitrogen. *Rangeland Ecology and Management, 63*(1), 109-119.

Polley, H. W., Derner, J. D., and Wilsey, B. J. (2005). Patterns of plant species diversity in remnant and restored tallgrass prairies. *Restoration Ecology, 13*(3), 480-487.

Porqueddu, C., Ates, S., Louhaichi, M., Kyriazopoulos, A., Moreno, G., Pozo, A., . . . Nichols, P. (2016). Grasslands in 'Old World'and 'New World'Mediterranean-climate zones: past trends, current status and future research priorities. *Grass and Forage Science, 71*(1), 1-35.

Post, W. M., and Kwon, K. C. (2000). Soil carbon sequestration and land-use change: processes and potential. *Global change biology, 6*(3), 317-327.

Poussart, P. F., Evans, M. N., and Schrag, D. P. (2004). Resolving seasonality in tropical trees: multi-decade, high-resolution oxygen and carbon isotope records from Indonesia and Thailand. *Earth and Planetary Science Letters, 218*(3), 301-316.

Powlson, D. (2005). Climatology: will soil amplify climate change? *Nature, 433*(7023), 204-205.

Powlson, D., Gregory, P. J., Whalley, W., Quinton, J., Hopkins, D., Whitmore, A., . . . Goulding, K. (2011). Soil management in relation to sustainable agriculture and ecosystem services. *Food policy, 36*, S72-S87.

Radford, B., Yule, D., Braunack, M., and Playford, C. (2008). Effects of grazing sorghum stubble on soil physical properties and subsequent crop performance. *American Journal of Agricultural and Biological Sciences, 3*(4), 734-742.

Rasmussen, P. E., Goulding, K. W., Brown, J. R., Grace, P. R., Janzen, H. H., and Körschens, M. (1998). Long-term agroecosystem experiments: assessing agricultural sustainability and global change. *science, 282*(5390), 893-896.

Reeder, J., and Schuman, G. (2002). Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands. *Environmental pollution, 116*(3), 457-463.

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

Roberts, M. R., and Gilliam, F. S. (2003). Response of the herbaceous layer to disturbance in eastern forests. *The herbaceous layer in forests of eastern North America. Oxford University Press, Oxford*, 302-320.

Robertson, G. P., Paul, E. A., and Harwood, R. R. (2000). Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. *science, 289*(5486), 1922-1925.

Rook, A., Dumont, B., Isselstein, J., Osoro, K., WallisDeVries, M., Parente, G., and Mills, J. (2004). Matching type of livestock to desired biodiversity outcomes in pastures–a review. *Biological conservation, 119*(2), 137-150.

Rooney, T. P., and Waller, D. M. (2003). Direct and indirect effects of white-tailed deer in forest ecosystems. *Forest Ecology and Management, 181*(1), 165-176.

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

Ryals, R., and Silver, W. L. (2013). Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands. *Ecological Applications, 23*(1), 46-59.

Sanjari, G., Ghadiri, H., Ciesiolka, C. A., and Yu, B. (2008). Comparing the effects of continuous and time-controlled grazing systems on soil characteristics in Southeast Queensland. *Soil Research, 46*(4), 348-358.

Savadogo, P., Sawadogo, L., and Tiveau, D. (2007). Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. *Agriculture, Ecosystems and Environment, 118*(1), 80-92.

Schindlbacher, A., Zechmeister-Boltenstern, S., and Butterbach-Bahl, K. (2004). Effects of soil moisture and temperature on NO, NO2, and N2O emissions from European forest soils. *Journal of Geophysical Research: Atmospheres, 109*(D17).

Scholes, M. C., Matrai, P. A., Andreae, M. O., Smith, K. A., Manning, M. R., Artaxo, P., . . . Ciccioli, P. (2003). Biosphere-atmosphere interactions. *Atmospheric chemistry in a changing world*, 19-71.

Schönbach, P., Wan, H., Gierus, M., Bai, Y., Müller, K., Lin, L., . . . Taube, F. (2011). Grassland responses to grazing: effects of grazing intensity and management system in an Inner Mongolian steppe ecosystem. *Plant and Soil, 340*(1-2), 103-115.

Schuman, G., Janzen, H., and Herrick, J. (2002). Soil carbon dynamics and potential carbon sequestration by rangelands. *Environmental pollution, 116*(3), 391-396.

Scurlock, J., and Hall, D. (1998). The global carbon sink: a grassland perspective. *Global change biology, 4*(2), 229-233.

Six, J., Conant, R., Paul, E. A., and Paustian, K. (2002). Stabilization mechanisms of soil organic matter: implications for C-saturation of soils. *Plant and Soil, 241*(2), 155-176.

Six, J., Elliott, E., and Paustian, K. (1999). Aggregate and soil organic matter dynamics under conventional and no-tillage systems. *Soil Science Society of America Journal, 63*(5), 1350-1358.

Smith, P. (2004). Carbon sequestration in croplands: the potential in Europe and the global context. *European journal of agronomy, 20*(3), 229-236.

Smith, P. (2008). Land use change and soil organic carbon dynamics. *Nutrient Cycling in Agroecosystems, 81*(2), 169-178.

Smith, P., Bhogal, A., Edgington, P., Black, H., Lilly, A., Barraclough, D., . . . Merrington, G. (2010). Consequences of feasible future agricultural land-use change on soil organic carbon stocks and greenhouse gas emissions in Great Britain. *Soil use and management, 26*(4), 381-398.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., . . . Rice, C. (2008). Greenhouse gas mitigation in agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences, 363*(1492), 789-813.

Smithson, P. A. (2002). IPCC, 2001: climate change 2001: the scientific basis. Contribution of Working Group 1 to the Third Assessment Report of the Intergovernmental Panel on Climate Change, edited by JT Houghton, Y. Ding, DJ Griggs, M. Noguer, PJ van der Linden, X. Dai, K. Maskell and CA Johnson (eds). Cambridge University Press, Cambridge, UK, and New York, USA, 2001. No. of pages: 881. Price£ 34.95, US $49.95, ISBN 0-521-01495-6 (paperback).£ 90.00, US $130.00, ISBN 0-521-80767-0 (hardback). *International Journal of Climatology, 22*(9), 1144-1144.

Snyman, H., and Du Preez, C. (2005). Rangeland degradation in a semi-arid South Africa—II: influence on soil quality. *Journal of Arid Environments, 60*(3), 483-507.

Soussana, J.-F., Tallec, T., and Blanfort, V. (2010). Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. *Animal, 4*(03), 334-350.

Soussana, J., Allard, V., Pilegaard, K., Ambus, P., Amman, C., Campbell, C., . . . Domingues, R. (2007). Full accounting of the greenhouse gas (CO 2, N 2 O, CH 4) budget of nine European grassland sites. *Agriculture, Ecosystems and Environment, 121*(1), 121-134.

Srivastava, D. S., and Jefferies, R. (1996). A positive feedback: herbivory, plant growth, salinity, and the desertification of an Arctic salt-marsh. *Journal of Ecology*, 31-42.

Stavi, I., Ungar, E. D., Lavee, H., and Sarah, P. (2008). Grazing-induced spatial variability of soil bulk density and content of moisture, organic carbon and calcium carbonate in a semi-arid rangeland. *Catena, 75*(3), 288-296.

Steffens, M., Kölbl, A., Totsche, K. U., and Kögel-Knabner, I. (2008). Grazing effects on soil chemical and physical properties in a semiarid steppe of Inner Mongolia (PR China). *Geoderma, 143*(1), 63-72.

Su, Y. Z., and lin Zhao, H. (2003). Soil properties and plant species in an age sequence of Caragana microphylla plantations in the Horqin Sandy Land, north China. *Ecological Engineering, 20*(3), 223-235.

Svejcar, T., Angell, R., Bradford, J. A., Dugas, W., Emmerich, W., Frank, A. B., . . . Mayeux, H. (2008). Carbon fluxes on North American rangelands. *Rangeland Ecology and Management, 61*(5), 465-474.

Svensson, J. R., Lindegarth, M., Jonsson, P. R., and Pavia, H. (2012). *Disturbance–diversity models: what do they really predict and how are they tested?* Paper presented at the Proc. R. Soc. B.

Tarhouni, M., Belgacem, A. O., Neffati, M., and Henchi, B. (2007). Qualification of rangeland degradation using plant life history strategies around watering points in southern Tunisia. *Pakistan journal of biological sciences: PJBS, 10*(8), 1229-1235.

Tastad, A. C. (2013). *The relative effects of grazing by bison and cattle on plant community heterogeneity in northern mixed prairie.* University of Manitoba.

Thomas, A. D., and Rosenstock, T. S. (2016). Greenhouse gas emissions from natural ecosystems and agricultural lands in sub-Saharan Africa: synthesis of available data and suggestions for further research. *Biogeosciences, 13*(16), 4789.

Thompson, R. L., Ishijima, K., Saikawa, E., Corazza, M., Karstens, U., Patra, P. K., . . . Prinn, R. G. (2014). TransCom N 2 O model inter-comparison–Part 2: Atmospheric inversion estimates of N 2 O emissions. *Atmospheric Chemistry and Physics, 14*(12), 6177-6194.

Tian, H., Xu, X., Liu, M., Ren, W., Zhang, C., Chen, G., and Lu, C. (2010). Spatial and temporal patterns of CH4 and N2O fluxes in terrestrial ecosystems of North America during 1979-2008: application of a global biogeochemistry model. *Biogeosciences, 7*(9), 2673.

Tom, N., Raman, A., Hodgkins, D. S., and Nicol, H. (2006). Populations of soil organisms under continuous set stocked and high intensity-short duration rotational grazing practices in the central tablelands of New South Wales (Australia). *New Zealand Journal of Agricultural Research, 49*(3), 261-272.

Tubiello, F. N., Salvatore, M., Ferrara, A. F., House, J., Federici, S., Rossi, S., . . . Flammini, A. (2015). The contribution of agriculture, forestry and other land use activities to global warming, 1990–2012. *Global change biology, 21*(7), 2655-2660.

Tubiello, F. N., Salvatore, M., Rossi, S., Ferrara, A., Fitton, N., and Smith, P. (2013). The FAOSTAT database of greenhouse gas emissions from agriculture. *Environmental Research Letters, 8*(1), 015009.

Tuffour, H. O., Bonsu, M., and Khalid, A. A. (2014). Assessment of soil degradation due to compaction resulting from cattle grazing using infiltration parameters. *IJSRES, 2*, 139-149.

Valentini, R., Arneth, A., Bombelli, A., Castaldi, S., Cazzolla Gatti, R., Chevallier, F., . . . Henry, M. (2014). A full greenhouse gases budget of Africa: synthesis, uncertainties, and vulnerabilities. *Biogeosciences*, *11*, 381-407.

Van Auken, O. (2009). Causes and consequences of woody plant encroachment into western North American grasslands. *Journal of Environmental Management, 90*(10), 2931-2942.

Veldkamp, E., Becker, A., Schwendenmann, L., Clark, D. A., and Schulte-Bisping, H. (2003). Substantial labile carbon stocks and microbial activity in deeply weathered soils below a tropical wet forest. *Global change biology, 9*(8), 1171-1184.

Wang, G., Huang, Y., Wang, E., Yu, Y., and Zhang, W. (2013). Modeling soil organic carbon change across Australian wheat growing areas, 1960–2010. *PloS one, 8*(5), e63324.

Wang, Q., and Batkhishig, O. (2014). Impact of overgrazing on semiarid ecosystem soil properties: A Case study of the Eastern Hovsgol Lake Area, Mongolia. *Journal of Ecosystem and Ecography, 4*(1), 1.

Wang, Y., Li, Y., Ye, X., Chu, Y., and Wang, X. (2010). Profile storage of organic/inorganic carbon in soil: From forest to desert. *Science of the Total Environment, 408*(8), 1925-1931.

Wang, Z.-P., Song, Y., Gulledge, J., Yu, Q., Liu, H.-S., and Han, X.-G. (2009). China's grazed temperate grasslands are a net source of atmospheric methane. *Atmospheric Environment, 43*(13), 2148-2153.

Wei, L., Hai-Zhou, H., Zhi-Nan, Z., and Gao-Lin, W. (2011). Effects of grazing on the soil properties and C and N storage in relation to biomass allocation in an alpine meadow. *Journal of soil science and plant nutrition, 11*(4), 27-39.

Welker, J. M., Fahnestock, J. T., Povirk, K. L., Bilbrough, C. J., and Piper, R. E. (2004). Alpine grassland CO2 exchange and nitrogen cycling: grazing history effects, Medicine Bow Range, Wyoming, USA. *Arctic, antarctic, and alpine research, 36*(1), 11-20.

Westoby, M., Walker, B., and Noy-Meir, I. (1989). Opportunistic management for rangelands not at equilibrium. *Journal of Range Management*, 266-274.

Wolf, K. (2011). *Effects of High-Density, Short-Duration Planned Livestock Grazing on Soil Carbon Sequestration Potentials in a Coastal California Mixed Grassland.* California Polytechnic State University, San Luis Obispo.

Wu, G.-L., Liu, Z.-H., Zhang, L., Chen, J.-M., and Hu, T.-M. (2010). Long-term fencing improved soil properties and soil organic carbon storage in an alpine swamp meadow of western China. *Plant and Soil, 332*(1-2), 331-337.

Yu, S., and Yao, H. (2003). Simulation and prediction of soil organic carbon dynamics in Jiangsu province based on models and GIS techniques. *Scientia Agriculatura Sinica, 2*, 1007-1013.

APPENDICES

Analysis of Variance for Soil organic carbon and Nitrogen concentrations.

Analysis of variance

Variate: Carbon (0-10)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.7810	0.3905	2.47	0.111
Residual	19	2.9992	0.1579		
Total	21	3.7802			

Variate: Carbon (10-20)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	1.13198	0.56599	6.18	0.009
Residual	19	1.74019	0.09159		
Total	21	2.87217			

Variate: Carbon (20-30)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.61104	0.30552	10.17	<.001
Residual	19	0.57061	0.03003		
Total	21	1.18165			

Variate: Carbon (30-60)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.57829	0.28914	8.93	0.002
Residual	19	0.61501	0.03237		
Total	21	1.19330			

Variate: Carbon (60-90)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.51156	0.25578	6.80	0.006
Residual	19	0.71470	0.03762		
Total	21	1.22626			

Variate: Carbon (90-120)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.39536	0.19768	14.94	<.001
Residual	19	0.25144	0.01323		
Total	21	0.64680			

Nitrogen

Variate: Nitrogen (0-10)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.

Treatment	2	0.0024516	0.0012258	1.33	0.287
Residual	19	0.0174499	0.0009184		
Total	21	0.0199015			

Variate: Nitrogen (10-20)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.0024516	0.0012258	1.33	0.287
Residual	19	0.0174499	0.0009184		
Total	21	0.0199015			

Variate: Nitrogen (20-30)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.00056055	0.00028028	2.91	0.079
Residual	19	0.00182822	0.00009622		
Total	21	0.00238877			

Variate: Nitrogen (30-60)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.0009131	0.0004565	3.50	0.051
Residual	19	0.0024759	0.0001303		
Total	21	0.0033890			

Variate: Nitrogen (90-120)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.01798	0.00899	0.63	0.544
Residual	19	0.27138	0.01428		
Total	21	0.28936			

Analysis of Variance for Soil organic carbon stocks

Variate: $MgCha^{-1}$ (0-10)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	27.07	13.53	0.45	0.646
Residual	19	574.69	30.25		
Total	21	601.75			

Variate: $MgCha^{-1}$ (10-20)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	89.36	44.68	1.67	0.215
Residual	19	509.27	26.80		
Total	21	598.64			

Variate: $MgCha^{-1}$ (20-30)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	44.674	22.337	5.00	0.018

Residual	19	84.928	4.470		
Total	21	129.602			

Variate: $MgCha^{-1}$ (30-60)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	414.05	207.02	4.07	0.035
Residual	18	915.95	50.89		
Total	20	1330.00			

Variate: $MgCha^{-1}$ (60-90)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	438.94	219.47	3.93	0.038
Residual	18	1005.29	55.85		
Total	20	1444.23			

Variate: $MgCha^{-1}$ (90-120)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	362.45	181.22	12.33	<.001
Residual	18	264.53	14.70		
Total	20	626.98			

Soil Bulk Density

Variate: Bulk Density

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.556853	0.278427	28.55	<.001
Residual	41	0.399838	0.009752		
Total	43	0.956691			

Variate: Bulk density (20-30cm)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.0336889	0.0168444	31.58	<.001
Residual	6	0.0032000	0.0005333		
Total	8	0.0368889			

Variate: Bulk density(30-60)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.036822	0.018411	2.43	0.169
Residual	6	0.045467	0.007578		
Total	8	0.082289			

Variate: Bulk density (60-90)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.22249	0.11124	1.95	0.222
Residual	6	0.34173	0.05696		
Total	8	0.56422			

Variate: Bulk density (90-120)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Treatment	2	0.0216222	0.0108111	26.30	0.001
Residual	6	0.0024667	0.0004111		
Total	8	0.0240889			