ESTABLISHMENT OF PASTURE ENCLOSURES RESTORES SOIL ORGANIC CARBON AND INCREASES GREENHOUSE GAS EMISSIONS IN A SEMI-ARID RANGELAND, KENYA

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DECLARATION

This thesis is my original work and has not been presented for a degree in any other university.

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DEDICATION

I dedicate this thesis to my wife Violet Manyasi; daughter, Cecile Atieno; and parents, John Oduor and Peres Athiambo.

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ACRONYMS AND ABBREVIATIONS

A.S.L	Above Sea Level
AEZ	Agro-ecological zone
ANOVA	Analysis of Variance
ANOVA	Analysis of variance
ASALs	Arid and Semi-Arid Lands
AU-IBAR	African Union - Interafrican Bureau for Animal Resources
BD	Bulk density
CEC	Cation exchange capacity
CGE	Contractual Grazing Enclosure
CGE	Contractual grazing enclosure
CGWP	County Government of West Pokot
DM	Dry matter
EC	Electrical conductivity
FAO	Food and Agricultural Organization of the United Nations
GDE	Grazing Dominated Enclosure
GDE	Grazing dominated enclosure
H“	Shannon-Weiner Index of diversity
LSD	Least Significance Difference
MEA	Millennium Ecosystem Assessment
MAP	Mean annual precipitation
MBC	Microbial Biomass Carbon
MBC	Microbial biomass carbon

MBN	Microbial Biomass Nitrogen
MBN	Microbial biomass nitrogen
OGR	Open Grazing Rangeland
POC	Particulate Organic Carbon
SLU	Swedish University of Agricultural Sciences
SOC	Soil organic carbon
SPSS	Statistical Package for the Social Sciences
SSA	Sub-Saharan Africa
TN	Total nitrogen
UN	United Nations
UNCCD	United Nations Convention to Combat Desertification
UNDDD	United Nations Decade for Deserts and Fight Against Desertification
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
Vi-AF	Vi Agroforestry

ABSTRACT

Communal grazing system can cause degradation of soil and vegetation in arid and semi-arid rangelands thereby impacting soil organic carbon (SOC) stocks and greenhouse gas (GHG) emissions. A field study was conducted to assess the influence of pasture grazing enclosures on pasture rehabilitation, SOC and emission of GHGs. The study was carried out in the semi-arid rangeland of Chepareria Ward in West Pokot County, Kenya. The objectives of the study were to determine the influence of pasture enclosure and its age on: (1) herbaceous vegetation cover, species composition and diversity, and aboveground biomass production; (2) total soil organic carbon, particulate organic carbon, and microbial biomass; and (3) flux rates of soil CO_2, CH_4 and N_2O. Completely randomized design was used for this study. Two types of pasture enclosures, namely contractual grazing enclosures (CGE) and grazing dominated enclosure (GDE) with differing grazing utilization strategies, were assessed based on years since establishment, hereby referred to as age class, as follows: 3-10, 11-20 and >20 years since establishment. Three enclosures were selected in each age/enclosure type combination (n = 3). The adjacent open grazing rangeland (OGR) was used as control. Herbaceous vegetation cover, species diversity as well as above ground, biomass differed significantly among grazing management systems and were consistently higher in enclosures and lower in the OGR. On average, herbaceous species diversity, vegetation cover, and aboveground biomass were 1.47, 1.83 and 7.25 times higher in the enclosures compared to OGR, respectively. Perennial grass cover, species diversity and aboveground biomass were considerably higher in the middle age enclosures (10-20 years) than in the newly established (3-10 years) and older (>20 years) enclosures. The SOC significantly increased from (mean ± SD) 4.72 ± 0.73 g kg^{-1} in OGR to 5.88 ± 0.72 and 6.12±1.00 g kg^{-1} in CGE and GDE respectively ($P < 0.001$), with age exhibiting non significant influence ($P > 0.05$). Significantly higher POC and MBC were observed in GDE than in CGE ($P < 0.05$). The concentration of MBC ranged from 32.05 ± 7.25 to 96.63 ± 5.31 µg C/g of soil in all grazing systems. Total SOC exhibited significant ($P < 0.001$) positive correlation with POC and MBC, suggesting that POC and MBC would account for the dynamics of soil organic carbon and soil biological status in the area. Soil CO_2 emission rate was higher in GDE (224.4 mg C m^{-2} h^{-1}) and CGE (239.9 mg C m^{-2} h^{-1}) relative to OGR (102.4 mg C m^{-2} h^{-1}) ($P < 0.001$). Soil moisture was significantly and positively correlated with CO_2, CH_4, and N_2O flux rates ($P < 0.001$); with peak emission rates observed at soil moisture content between 15 and 25% (v/v). This suggested that soil moisture is the critical factor that influences the emission of CO_2, CH_4, and N_2O from soil in the semi-arid rangeland. These results demonstrated that pasture enclosures were important to restore the degraded communal grazing lands in terms of herbaceous vegetation cover, diversity, aboveground biomass, and soil organic carbon and microbial biomass contents. The higher emission of soil CO_2 in the enclosures was as a result of the improved soil and vegetation conditions which enhanced microbial respiration Overall, the higher soil-atmosphere CO_2 emission in enclosure systems could be offset by the higher aboveground biomass and the SOC sequestered in the soil. Future research in the area should focus on carrying capacity of the enclosures, and include landscapes such as water points and settlements to assess the ecosystem SOC dynamics and GHG emissions.

CHAPTER ONE

Introduction

1.1 Background of the Study

Arid and semiarid lands (ASALs) or rangelands cover about 30% of the World's land area, 43% in Africa and more than 83% of Kenya's land surface (Loveland *et al.*, 2000; Millennium Ecosystem Assessment, 2005; NEMA, 2015). ASALs host more than one-third of the total human population worldwide (Reynolds *et al.*, 2007), 40% and 10% of the total population in Africa and Kenya respectively (Kibirde and Grahn, 2008; AU-IBAR, 2012). Over 70% of Kenya's livestock population and 75% of the wildlife are found in ASALs (Milton *et al.*, 1994; GoK, 2002). Traditionally, rangelands in Kenya are managed through common property tenure systems (McDermott *et al.*, 2010). The tragedy of the commons (Hardin, 1968) is a regular characteristic of rangelands and is responsible for land degradation due to mismanagement of resources through overstocking and overgrazing in the communal grazing lands (Ayantunde *et al.*, 2011). Beside the vastness and importance of ASALs, degradation and desertification remain a major ecological problem in Sub-Saharan Africa (SSA) (Steinfeld *et al.*, 2006; Mcsherry and Ritchie, 2013).

Globally, approximately 20-25% of rangelands are degrading with the degradation being more severe in SSA (Lal, 1988; UNCCD, 2012). An estimated 75% of Africa's drylands are affected by moderate to high degree of degradation (Eswaran *et al.*, 2001). In Kenya, a large proportion of the arid and semi-arid rangelands is highly degraded and another 2% being completely desertified (Keya, 1991). Accelerated land degradation in West Pokot to County has been attributed overgrazing and poor management of grazing resources (Nyberg, 2015). The loss of ecosystem functions and services due to land degradation is undermining the sustainability of rangeland ecosystems. This endangers the rural livelihoods and the general well-being of the population at large (Goldman, 2006). Additionally, degradation may have strong impact on climate change due to the effect on soil organic carbon sequestration and greenhouse gas emission (Lal, 2004b). As natural resources are the primary source of livelihood support for pastoral communities, efforts to rehabilitate them could become a valuable strategy for livelihood improvement.

In response to rehabilitate/restore the degraded grazing lands, communities in Western Kenya started to establish pasture enclosures about three decades ago. Enclosures are private grazing areas fenced-off from the interference by the rest of the community and livestock with the aim of promoting natural regeneration of pasture and reducing land degradation of formerly degraded communal grazing lands. The adjudication/fragmentation/privatization of communal grazing land for the establishment of rangeland enclosures is believed to foster a more responsible use of the land. The enclosures are usually established in degraded areas that have been used for communal grazing in the past (Keene, 2008). In West Pokot County, Kenya, the restoration efforts using enclosures were started in the mid-1980s by a non-governmental organization (NGO) called Vi-Agroforestry (Vi- Agroforestry, 2007). Priority areas for establishing enclosures are jointly identified by the local communities, government organizations, and NGOs (Vi- Agroforestry, 2007). Fencing is done either by the use of thorny tree cuttings or planting sisal (*Agave sisalana*) or a combination of both.

Examples of successful rangeland restoration initiatives using private and communal enclosures in Kenya are found in the Counties of West Pokot (Nyberg, 2015), Baringo (Mureithi *et al.*, 2010b; Mureithi *et al.*, 2015a), Kajiado (Macharia and Ekaya, 2005; Opiyo *et al.*, 2011), and Turkana (Kigomo and Muturi, 2013). As reported by Wairore *et al.* (2015), grazing dominated enclosure (GDE) and contractual grazing enclosure (CGE) are the common types of enclosure management systems in Chepareria Ward, West Pokot County, Kenya. The enclosures are privately owned and utilized, with an average size of 5 hectares (Wairore et al., 2015). Contractual grazing represents a grazing arrangement where a farmer owning few animals leases the enclosure to households with relatively more livestock. On the other hand, GDE is where the livestock utilizing the enclosure is purely owned by the farmer/land owner. The stocking rate of the enclosures in the area ranges between 1 and 42 animals with a mean of 7 animals (Wairore et al., 2015). Livestock management in both CGE and GDE enclosure systems is via the free-range system the animals graze freely within the enclosure.

Several studies have reported the socio-economic benefits of rehabilitating degraded rangelands via enclosures in Kenya due to the positive impact on vegetation and biodiversity (Wasonga, 2009; Mureithi *et al.*, 2010a; Nyberg and Öborn, 2013; Mureithi *et al.*, 2014a; Svanlund, 2014b; Mureithi *et al.*, 2015b; Nyberg *et al.*, 2015; Wairore, 2015). Enclosures are also reported to enhance soil quality by improving soil organic carbon (SOC) (Beukes *et al.*, 2002; Lal, 2004b; Verdoodt *et al.*, 2010b; Mureithi *et al.*, 2015a). According to Hongo *et al.* (1995) and IPPC (2000), enclosures reduce the intensity of wind and water erosion and increase water infiltration and availability. This suggests that the rehabilitation of degraded rangeland can help achieve sustainability of drylands and enhances the livelihoods of the local population. However, these studies were conducted in communal enclosures and have not explicitly explored private enclosure management and the impacts of enclosure management systems on rangeland rehabilitation. With increasing adoption and adaptation of private enclosures in rangeland rehabilitation in Chepareria, Kenya, understanding the management systems is not only essential to management and utilization but is also important to determine the impacts of the existing management pathways. Information gained from this study could determine the emergent trends and issues that may inform the management strategy which is adaptable to various localities in SSA.

1.2 Problem Statement

Whereas enclosures have been reported to increase forage production and soil organic carbon stocks (He *et al.*, 2008; Mekuria and Veldkamp, 2012; Bikila *et al.*, 2016). These studies were conducted in exlosures that received periodic and or special management such as such as tillage, fertilization, weed control and irrigation, and livestock feed via the cut-and-curry system. On the other hand, the enclosures in Chepareria do not receive any special treatment as plants were allowed regenerate naturally for a period of three years (Nyberg, 2015), after which livestock was allowed to graze freely within the fenced area. The labile fractions of SOC are known to quickly respond to changes in management strategies than the total SOC. Nevertheless, the particulate organic carbon (POC) and microbial biomass have nevertheless received little attention in arid and semi-arid rangelands of Kenya. As changes in SOC content may alter soil-atmosphere flux rates of GHGs (Liu *et al.*, 2007; Liu and Greaver, 2009; Tang *et al.*, 2017), the general effect of restoration-induced changes soil-atmosphere emission of CO_2, CH_4 and N_2O, and the underlying controls in tropical rangelands remain unclear.

1.3 Justification

Understanding the composition of SOC in rehabilitated rangelands is essential for the development of stable soil resources and the adoption of appropriate land management practices in the ASALs of Kenya. Determination of the grazing system that increases microbial biomass is necessary to improve biological soil fertility and curb land degradation. Quantification of GHG flux rates under different grazing systems is important to understand the feedback of land rehabilitation on climate change mitigation.

1.4 Objectives

1.4.1 Main Objective

To assess the contribution of pasture enclosure systems in restoration of degraded soils for sustainable soil productivity and enhanced climate change mitigation Chepareria in West Pokot County, Kenya.

1.4.2 Specific Objectives

The specific objectives of the study were:

1. To characterize the enclosure management systems in terms of their herbaceous vegetation cover, species composition and diversity, and biomass.
2. To determine the effect of enclosure management systems and their age of enclosure on soil organic carbon, particulate organic carbon, and microbial biomass.
3. To assess the influence of enclosure systems and age of enclosure on greenhouse gas emissions (CO_2, CH_4 and N_2O).

1.5 Hypotheses

The study was based on the hypotheses that:

1. Pasture enclosure systems have no influence on herbaceous vegetation cover, species richness and diversity, and aboveground biomass production.
2. Pasture enclosure systems and their ages have no effect on soil organic carbon, particulate organic carbon, and microbial biomass.
3. Enclosure system and their ages have no influence on soil-atmosphere emission of CO_2, CH_4 and N_2O.

1.6 Description of the study site

1.6.1 Location

This study was conducted in Ywalateke Location in Chepareria Ward in West Pokot County. The area lies between latitude 1° 18' and 1° 19' N; longitude 35° 14' and 35° 15' E (Figure 1). The area has gently undulating plains with an altitude ranging from 1600 to 1800 meters above sea level. The area is located on the lower slopes of Kamatira hills where agroforestry and enclosures have been extensively promoted by NGO Vi-Agroforestry from the year 1987 to 2000 (Nyberg and Öborn, 2013).

1.6.2 Climate

The area experiences a highly variable seasonal climate which is a characteristic of semi-arid regions in SSA. The county has a bimodal type of rainfall. The mean annual precipitation is about 850 mm (CGWP, 2013b). The wet season runs between March – May (long rains) and August – November (short rains). Temperature varies with altitude and ranges between 24^0C to 30^0C (CGWP, 2013b).

1.6.3 Soils and water resources

The soils in the area are rocky, moderately shallow, well drained and highly rich in ferromagnesian minerals (Touber, 1991), and vary significantly across the study area (Sposito, 2013). Generally, the soils are well drained, moderately deep, dark reddish brown to dark red, friable to firm, sandy clay in many places (Jaetzold and Schmidt, 1983), and are classified as Haplic Lixisols (Hiederer and Köchy, 2011). The main sources of water in the study area are rivers Muruny, Weiwei and Suam and seasonal streams (Makokha *et al.*, 1999a).

1.6.4 Vegetation

The vegetation is dominated by grasslands (*Themeda triandra*, *Eragrostis superba*, *Cymbopogon validus, Cenchrus ciliaris and Cynodon dactylon)* with scattered native and exotic tree species. Common native tree species include *Acacia spp*., *Balanites aegyptiaca*, *Kigelia africana* and *Terminalia brownii* while the exotic tree species are *Croton spp*., *Ficus spp*., *Grevillea robusta* and *Azadirachta indica* (Makokha *et al.*, 1999a).

1.6.5 Land-use and livelihood

The main land-use and source of livelihood in Chepareria is livestock keeping (Svanlund, 2014b). Though livestock serve as a measure of wealth, subsistence crop production, particularly maize production, is also undertaken in the arable areas (Makokha *et al.*, 1999a; CGWP, 2013b). Other cultivated crops include beans, millet and sorghum (Vi- Agroforestry, 2007). Fruits farming, contractual grazing and pasture production are other land–use practices that are slowly gaining popularity in the study area (Vi- Agroforestry, 2007).

1.7 Description of enclosure systems

An enclosure refers to a fenced pasture for specified duration of time, usually three years, to allow natural regeneration of vegetation (Behnke, 1986). Makokha *et al.* (1999) noted that before the establishment of enclosures in Chepareria by Vi-Agroforestry in the mid-1980s, the communal grazing lands were severely degraded and supporting sparse undesirable vegetation with little grazing value. Hence, the enclosures were established in the area to foster land rehabilitation and improve livestock production.

The selection of enclosure management regimes was based on the classification by Wairore *et al.* (2015c). Contractual grazing enclosure (CGE) and grazing dominated enclosure (GDE) were the main livestock-based in Chepareria (Wairore *et al*., 2015c). The enclosures are privately owned with an average size of 5 ha. Under GDE the famers keep their livestock within own enclosure while CGE is where the enclosure is rented for grazing. The grazing intensity and frequency of utilization being higher in CGE compared to GDE (Wairore *et al.*, 2015c). Open grazing rangeland (OGR) is characterized by year-round grazing with higher grazing intensity than the enclosure systems.

1.8 Structure of the thesis

This thesis is divided into three parts. The **first part** describe**s** the background of the study and defines the scope of the study (objectives and research hypotheses) in *Chapter 1*. *Chapter 2* reviews previous studies that used enclosures as a tool for the management of degraded rangelands. The **second part** report results of the rehabilitative impacts of enclosures on degraded rangelands and is comprised of three ongoing papers (*chapters 3*, *4* and *5*) that have their own literature review and methodology sections. Hence, some sections of the papers may sound in duplicity with chapters 1 and 2, but were all considered relevant for a better understanding of this entire research. *Chapter 3* assesses the impact of age of enclosure and management on characteristics of herbaceous vegetation and aboveground biomass. *Chapter 4* highlights how the establishment of enclosures restores the quality of degraded soils through increased organic carbon and microbial biomass, while *Chapter 5* quantifies soil greenhouse gases emission in the enclosures and open rangeland in the semi-arid rangelands of Chepareria. The **third part**, representing *Chapter 6***,** covers the general discussion and conclusions and recommendations of the thesis.

CHAPTER TWO

Literature review

2.1 Degradation of tropical rangelands

Several factors contribute to the degradation of rangelands, including natural factors, such as, long-term drought, wind erosion and sand storms (Schlesinger *et al.*, 1990). Among the human induced causes, overgrazing and land use change are common worldwide (Manzano and Navar, 2000; Zhao *et al.*, 2005). Overgrazing is the primary cause of grassland degradation in Africa due to the predominance of common grazing areas where grazing intensity is high (FAO, 2001; Symeonakis and Drake, 2004; Al-Rowaily *et al.*, 2015). The ever-increasing human population in ASALs who depend on agriculture for livelihood has contributed to the recent acceleration of anthropogenic land degradation (IFAD, 2010; Van Pham and Smith, 2014). The National Environment Management Authority (2015) reported that the 85% of Kenyan ASALs have undergone massive degradation. According to Leon and Osorio (2014), land degradation in the tropics has decreased soil ecosystem services by up to 60% between 1950 and 2010, thus suppressing agronomic and economic development for the rural population (Scherr, 2001).

Overgrazing in rangelands usually reduces total plant cover, aboveground biomass and changes species composition (Li *et al.*, 2014; Mekuria *et al.*, 2015). Repeated trampling by animals disintegrates plant residues and litter on the soil surface exposing them decomposition, compacts the soil and reduces infiltration of rain water penetrability. This increases the risk of runoff and causes loss of soil nutrients (Jeddi and Chaieb, 2010b). Studies indicate that the restoration of degraded rangelands depends on the local climatic conditions, disturbance history and soil texture (Suding *et al.*, 2004; Wang *et al.*, 2013).

2.2 Use of enclosures to restore degraded rangelands

Among the factors that influence the restoration of degraded rangelands, local intervention through the change in management practice is important. Reducing grazing pressure by fencing has proven to be an effective method to rehabilitate degraded rangelands in ASALs (Jeddi and Chaieb, 2010b; Mureithi *et al.*, 2010c; Verdoodt *et al.*, 2010b; Mekuria and Aynekulu, 2011a). Globally, enclosures have been widely adopted to allow natural regeneration of vegetation and recuperation of soil properties (Pei *et al.*, 2008; Schoenbach *et al.*, 2011; Mekuria and Veldkamp, 2012; Mureithi *et al.*, 2015a).

The rates of rehabilitation of vegetation and soil may differ over time. Generally, vegetation recovers quickly than soil (Pywell *et al.*, 2002; McDermot and Elavarthi, 2014). Hence, the benefit of restoration may vary in different grasslands and among ecosystem components, such as plants and soil. Vegetation analysis in West Pokot County in Kenya showed that enclosures increased plant cover by 86% between 2001 and 2014 (Nyberg, 2015). Mekuria and Aynekulu (2011) attributed a higher total soil nitrogen (N), available phosphorus (P), and cation exchange capacity (CEC) inside enclosures than in communal grazing lands, as well as higher biomass production and vegetation cover. Similarly, Mureithi *et al.* (2014) observed a significant decrease in soil bulk density, and increase in the soil organic carbon, total nitrogen, and microbial biomass contents in the enclosures as compared with degraded open rangeland.

2.3 Effect of enclosure management on herbaceous vegetation

Grazing management has variable effects on plant community structure in different rangeland ecosystems (Guo, 2007; Marriott *et al.*, 2009; Mayer and Erschbamer, 2017). Mekuria and Yami (2013) concluded that some vegetation indicators, such as species composition, vegetation cover and forage production, reflect the condition of rangeland. For example, a reduction in vegetation cover and palatable species, as well as increases in unpalatable plant species, indicates a decline in rangeland condition (Mekuria and Yami, 2013). Compared to short-term light grazing, frequent grazing reduces plant cover and the proportion of perennial grasses, increases the proportions of annual grasses and reduces biomass production (Mekuria and Veldkamp, 2012; Porensky *et al.*, 2017). Perennial grasses have better survival adaptations than other plants as propagation is both sexual and *via* rhizome (Tessema *et al.*, 2016). When grazing pressure increases, propagation of perennial grass is reduced due to frequent aboveground disturbance while exclusion of grazing can quickly prompt the population perennial grasses (Li *et al.*, 2014).

2.4 Soil organic carbon in rangelands

The extensive distribution of rangelands makes them important soil C reservoirs that could off-set fossil fuel emission and mitigate climate change (McDermot and Elavarthi, 2014). Rangelands store about 27% of the global SOC reserves (MEA, 2005), and have the potential to sequester 198 million tons of CO_2 from the atmosphere (Schuman *et al.*, 2002; Lal, 2004b). In Africa, approximately 59% of the total C stock is held in ASALs (Campbell *et al.*, 2008; UNEP 2008). However, organic C sequestration in drylands is severely constrained by the xeric nature of soils that limit plant productivity (Balogh *et al.*, 2005). Additionally, overgrazing may deplete SOC pools and change the soils from being C sinks to emission sources (Balogh *et al.*, 2005; Lal, 2009).

Adoption of practices which can reverse land degradation in rangelands can stimulate the increase in ecosystem C stocks and sequestration of atmospheric CO_2 in soils. Some studies reported an increased concentration of SOC under heavy grazing in the surface 30 cm of soil compared to the adjacent non-grazed enclosures (Derner and Schuman, 2007; Mekuria and Aynekulu, 2011a). Other studies have shown that rotational-deferred grazing and continuous grazing at heavy stocking rates did not affect carbon sequestration in a mixed-grass prairie (Teague and Barnes, 2017). These variations in SOC sequestration hints to the complexity of the interaction of management and environmental conditions.

Increasing SOC concentration in rangelands can improve plant production, soil aggregation, prevent erosion and increase ecosystem diversity (Bronick and Lal, 2005). The soil C sequestration rate is determined by the balance between C inputs and outputs, which are affected by management, rate of soil organic matter (SOM) input and decomposition of organic matter by the soil microorganisms, whereas SOC loss as CO_2 can be reduced by biochemical recalcitrance, chemical stabilization and physical protection of SOM by soil aggregates (Jastrow and Miller, 1997; Lal *et al.*, 2015).

2.5 Soil organic carbon fractions

Partitioning SOC into functional fractions is important to better understand its dynamics and roles in ecosystems (Cambardella and Elliott, 1992). The physically protected and recalcitrant or stable fractions are the commonly recognized pools (Herrick and Wander, 1997). The various fractions of organic C have different susceptibility to land management strategies (Kapkiyai *et al.*, 1999). Physical fractionation procedures based on differential have been used to separate coarse fractions from fine fractions. The coarse fractions ranging between 53 and 250 μm may provide an accurate estimate of the labile SOC while fractions <53μm may provide an accurate estimate of the stable pool (Cambardella and Elliott, 1992).

Each fraction of SOC plays a particular role in release of soil nutrients, CEC and soil aggregation (Awale et al., 2017). Fractions with rapid turnover rate are assumed to have an important role in nitrogen availability because SOC dynamics and N cycling are closely linked through the processes of N mineralization and immobilization (McGill and Cole, 1981). The stable fractions play an important role in cation exchange reactions in sandy soils and are important in soil aggregation (Christensen, 2001).

2.5.1 Litter

Levels of SOC are linked to herbaceous plant production, plant litter inputs into the soil system and rates of decomposition (Bikila *et al.*, 2016). Though many studies exclude litter in organic matter definitions, fresh plant residues are considered an important component of the labile SOC (Haynes, 2005). Litter quality is equated with the rate at which organic substrates are decomposed and protected against soil erosion (Christensen, 2001).

2.5.2 Microbial biomass

Microbial biomass is the biologically active and most dynamic pool of organic C. Microbial biomass is comprised of the soil biota actively involved in the mineralization of organic residues and soil nutrient supply (Dalal *et al.*, 1991; Myrold, 1998). It gives a quick indication of soil biological status, plays a vital role in soil aggregation and therefore important for improving soil quality (Bationo *et al.*, 2007). This fraction is estimated by fumigation extraction and gives a good general measure of active soil biota (Paul, 2014).

2.6 Influence of grazing on microbial biomass

Generally, grazing significantly decreases microbial biomass C and N pools in grassland ecosystems (Zhou *et al.*, 2017). Reports show that heavy grazing reduces soil biota compared to ungrazed areas by reducing the labile C available to soil microbes (Northup *et al.*, 1999; Stark and Kytöviita, 2006; Eldridge and Delgado-Baquerizo, 2017). Sarathchandra *et al.* (2001)

attributed the low MBC in grazed than ungrazed land to lower SOC, which is the energy source for soil microorganisms. High vegetation cover and organic matter reduces the evaporative water loss and increase soil water retention (Wasonga and Nyariki, 2009), sequentially increasing microbial biomass in less disturbed grazing areas (Mureithi *et al.* 2014).

2.7 Soil greenhouse gas emissions in the rangelands

Studies on CH soil CO_2, N_2O and CH_4 flux rates have been conducted in mesic environments with soil moisture levels typically above optimum, but little is known about responses in drier systems with sub-optimal soil water. Livestock grazing impacts the emission of soil CO_2, N_2O and CH_4 due to the effect on plant biomass, soil moisture, SOC and soil porosity, that directly influence soil microbiological processes (Smith *et al.*, 2003). Because the main source of methane emission from soil in rangelands is from the deposited animal manure (Samal *et al.*, 2015). In this study, animal excreta and effluents were indirectly included as the measurement of GHGs was conducted under natural field conditions. The GHG emission rates depend on soil management, soil type, and climate (Jones *et al.*, 2005; Skiba *et al.*, 2012). Soil CO_2 account for 60-90% of the total CO_2 flux within terrestrial ecosystems (Buchmann, 2000), and small changes in the magnitude of soil respiration could have a great impact on atmospheric concentration of CO_2 (Peng *et al.*, 2009). Carbon dioxide released from the soil surface during respiration is largely from microbial decomposition of plant litter and readily decomposable organic C (Janzen, 2004).

In a study carried out by Tang *et al.* 2017, heavy grazing reduced soil CO_2 emission by 11%, CH_4 uptake (net capture from the atmosphere) by 19% and N_2O emission by 28%. The reduction in soil CO_2 emission rate under heavy grazing relative to the moderately grazed grassland is ascribed to lower plant cover that suppresses soil respiration due to limited soil moisture (Liu *et al.*, 2009b). Rangeland soils may also act as carbon sinks under increased SOC stock (Soussana *et al.*, 2010; Valentini *et al.*, 2014). The reduction in N_2O emission with increasing grazing intensity is due to decline in soil total N (Gelfand *et al.*, 2013). The inhibitory effect of grazing on methanotrophs by water stress is responsible for the decreased CH_4 uptake in tropical rangelands (Tang *et al.*, 2017).

2.8 Factors contributing to GHG emissions from soil in rangelands

In pastoral ecosystems, animal manure directly or indirectly affect GHG emissions via the involvement of microbial processes and modification of soil characteristics (Thangarajan *et al.*, 2013). In addition, livestock, as a vector of organic matter, can result in spatial heterogeneities in soil properties, available nutrients and biomasses, most likely leading to intensification of GHG emissions via enhanced microbiological processes (Smith *et al.*, 2003).

Many studies have reported the effects of environmental variables such as soil temperature, soil water status, and microbial and root biomasses on temporal and spatial variation in soil CO_2 flux (Ussiri and Lal, 2013; Giardina *et al.*, 2014; Zelikova *et al.*, 2015; Grand *et al.*, 2016). However, soil moisture status and availability of SOC are considered the key controlling factors influencing soil CO_2 and N_2O flux rates in tropical rangelands (Kuzyakov and Gavrichkova, 2010; Yemadje *et al.*, 2016). Soils under natural vegetation account for 60% of the total N_2O emission from natural sources (IPCC, 2013). Soil N_2O flux rates in rangelands vary with vegetation type, soil properties and land management practice (Brummer *et al.*, 2008).

Although CH_4 emissions mostly occur only in hydromorphic conditions, CH_4 emission may also occur in aerobic soils when significant CH_4 production occur in anaerobic microsites (Le Mer and Roger, 2001). Soil CH_4 uptake in rangelands is sensitive to land management practices (Moiser *et al.*, 1996). According to Wang *et al.* (2012), light grazing may increase CH_4 uptake from the atmosphere or have no impact on soil CH_4 consumption. On the other hand, heavy grazing has been reported to reduce annual soil CH_4 consumption by 24-31% (Smith *et al.*, 2000).

CHAPTER THREE

Impacts of pasture enclosure and time since establishment on characteristics of the natural vegetation

Abstract

Establishment of grazing enclosures has become an important rangeland rehabilitation strategy in semi-arid regions. This study assessed the impact of enclosure and time since establishment on the characteristics of herbaceous vegetation in the semi-arid rangeland of Chepareria, West Pokot County, Kenya. Two enclosure systems with differing grazing utilization strategies, namely, grazing dominated (GDE) and contractual grazing (CGE) were selected based on three age classes (3-10, 11-20 and >20 years since establishment), with three replications in each age class. Point-to-line transect and quadrat-based methods were used to measure herbaceous plant cover, relative abundance, diversity index and aboveground standing crop biomass inside the enclosures and in the adjacent open grazing rangeland (OGR) as reference. The enclosure systems promoted the recovery of preferred perennial grass species like *Cynodon dactylon* while species like *Brachiaria deflexa* which is tolerant to high grazing pressure was observed in OGR only. Herbaceous plant cover (mean ± standard deviation) was significantly (P<0.001) lower in OGR (20.73 ± 2.64%) compared to CGE (34.42 ± 5.97%) and GDE (40.22 ± 5.48%). The relative abundance of perennial grasses was significantly higher in GDE (0.33) followed by CGE (0.22) and OGR (0.14). Similarly, species diversity index and aboveground biomass were considerably higher in GDE and CGE compared to OGR (P<0.001). Significantly higher plant cover, diversity index and aboveground biomass were exhibited in the middle aged enclosures (11-20 years) than in the newly established (3-10 years) and older (>20 years) enclosures (P<0.05). The results suggest that the vegetation in the previously overgrazed rangelands recovered following the establishment of pasture enclosures, with the recovery increasing with the age of enclosure until 20 years, and later decreased in the enclosures >20 years. Therefore, GDE and enclosures which are less than 20 years are vital to restore vegetation in previously degraded communal grazing lands in Chepareria.

3.1 Introduction

Selective removal of palatable species in common grazing areas, which are subjected to year round grazing by livestock, often results in altered vegetation composition and productivity (Boly *et al.*, 2011). According to Millennium Ecosystem Assessment (2005), land degradation due to overgrazing is very pronounced in Sub-Saharan Africa. Arid and semi-arid lands (ASALs), which host the largest vegetation resource (Mengistu *et al.*, 2005a), occupy 85% of the land area in Kenya (NEMA, 2015). A large proportion of Kenyan ASAL rangelands is highly degraded and another 2% has been completely desertified (Keya, 1991). Overgrazing in communal grazing areas lowers both the productivity and resilience of plant species (Kairis *et al.*, 2015; Wiesmair *et al.*, 2017), reduce vegetation cover (Mekuria and Veldkamp, 2012) and changes species diversity (Al-Rowaily *et al.*, 2015). It also alters the soil structure and compactness (Mekuria *et al.*, 2007b; Mekuria and Aynekulu, 2011a; Pizzio *et al.*, 2016). Studies have shown that the replacement of palatable by unpalatable plants decreases rangeland productivity and plant diversity (Hobbie, 1992; Cingolani *et al.*, 2005).

Pasture enclosures are an effective management technique for the rehabilitation of degraded rangelands in Eastern Africa (Box, 1971; Barrow and Mlenge, 2003; Mekuria and Aynekulu, 2011a). They can be used to maintain species diversity and rangelands productivity (Liu *et al*., 2009a; Mekuria and Veldkamp, 2012). Furthermore, open grazing systems have negative impact on soil hydrological properties, thus hindering plant growth in arid and semi-arid climates where water is a scarce resource (Jeddi and Chaieb, 2010a). The process of vegetation recovery in the enclosures starts with the rapid recovery of herbaceous species, mainly grasses, then after 3 to 5 years, shrub and tree species gain importance (Yayneshet *et al.*, 2009).

Communities in Chepareria started to establish pasture enclosures by fencing the communal grazing areas30 years ago with an aim of encouraging regeneration of plants and pastures (Mekuria and Aynekulu, 2011a; Wairore *et al.*, 2015d). Grazing dominated enclosure (GDE) and contractual grazing enclosure (CGE) are the common forms of grazing enclosures in Chepareria (Wairore *et al.*, 2015b). The enclosures are privately owned with an average size of 5 ha. Under GDE system, the famers keep their livestock within own enclosure while CGE system, the enclosure is leased for grazing to other farmers, with the grazing intensity and frequency of utilization being higher in CGE than in GDE (Wairore et al., 2015b). Studies show that enclosure management systems enhance species diversity and biomass production (Verdoodt *et al.*, 2009; Verdoodt *et al.*, 2010a; Mureithi *et al.*, 2015a; Wairore *et al.*, 2015a). However, a study that investigates the dynamics of herbaceous vegetation restoration in the grazing enclosure systems in West Pokot County has not been conducted.

The purpose of this study was to assess the enclosure management systems in Chepareria, as a contribution to help reduce land degradation in North-western Kenya and other rangelands in Sub-Saharan Africa. The objectives of the study were: (1) to determine the herbaceous vegetation cover, species diversity, and aboveground biomass production in CGE and GDE enclosures, and in the open grazing rangeland (OGR) as control; and (2) to understand the effect of age of enclosures on herbaceous vegetation cover, species diversity, and aboveground biomass.

3.2 Materials and methods

3.2.1 Site description

The study was conducted in Chepareria Ward (01° 18' 16.84" - 01° 19' 40.94" N; 035° 14' 15.57" - 035° 15' 49.06" E) of West Pokot County, located in Northwestern Kenya at an altitude of altitude of 1680 – 1800 m above sea level. The area receives bimodal rainfall with long rains occurring from March to May and the short rains occurring in October to November. The total annual ranges from 600 – 850 mm. The average minimum and maximum annual temperature is 16^{o}C and 38^{o}C respectively (CGWP, 2013a). The community is agro-pastoral and livestock-based enclosure regimes account for 78.3% while crop-grazing account for 28.7% of the enclosures (Wairore *et al.*, 2015c). Soils are shallow and well drained, with predominantly sandy clay to loamy sand texture and are classified as Haplic Lixisols (Hiederer and Köchy, 2011). Vegetation is predominantly grassland with scattered native Acacia-species including *Acacia spp*., *Balanites aegyptiaca*, and *Kigelia africana*.

3.2.2 Selection of the enclosures

The selection of enclosures involved discussions with the local community elders and officials from Vi-Agroforestry. Enclosures were grouped into three age classes: 3-10 years, 11–20 years, and >20 years. Three enclosures were randomly selected along each age in chronological sequence (n = 3). Nine adjacent open grazing sites (OGR) were selected as controls.

3.2.3 Vegetation sampling

Point- to- line transect method was used to assess the herbaceous vegetation over a period of two wet seasons; November 2016 and June 2017. Randomized design with split plot arrangement was used in this study. The enclosure system was considered to be the main plot while the age of enclosure was the sub-plot. Within each enclosure and in the adjacent open grazing areas, three 50 m transects were laid in a Z-shaped orientation, at least 10 m from the edge to avoid edge effects. Transects were assessed using point quadrat method as described by Daget and Poissonet (1971) and Floret (1988). A long metallic wire that was sharpened on one end to make point was descended from the line transect to the ground at 50 cm intervals along the transect. A total of 100 points were made per transect. At each of the 100 points, the species, vegetation type (i.e., grass, forb, or shrub), or ground cover (bare ground) that intersects the point is recorded as a "hit." Variables measured included ground cover, species composition and relative abundance (RA). The proportion of vegetation cover (VC) was estimated using Equation 1 and relative abundance using Equation 2. As shown in Equation 3, Shannon diversity index (Shannon, 1948) was determined by calculating the frequency of each plant species. Plant species richness (number of species sampled per transect) was also determined. Herbaceous biomass was estimated using 0.25 m^{-2} quadrats by clipping grass and forb materials at 2 cm above the ground level. A total of 135 quadrats were sampled: 45 for GDE, 45 for CGE and 45 for OGR. Samples were oven-dried at 70^{o}C for 48 hours in the laboratory to constant weight. Aboveground biomass production was expressed in kg/ha on dry matter basis.

$$\mathrm{VC} = (\mathrm{n}/\mathrm{N}) * 100 \qquad \text{Equation 1}$$

Where:

VC = vegetation cove

n = the number of hits of all plant species or type of ground touched

N = the total number of hits (100 hits in this case)

$$\mathrm{RA} = \left(\frac{\mathrm{A}}{\mathrm{T}}\right) \qquad \text{Equation 2}$$

Where:

RA = relative abundance of plant species

A = number of hits of functional group A

T = total number of hits for all plant species

$$H' = -\left(\sum pi \; ln \, (pi)\right) \qquad \text{Equation 3}$$

Where:

H’ = Shannon diversity index

pi = number of individual species in each transect at which species *i* was recorded

ln = the natural log of the number.

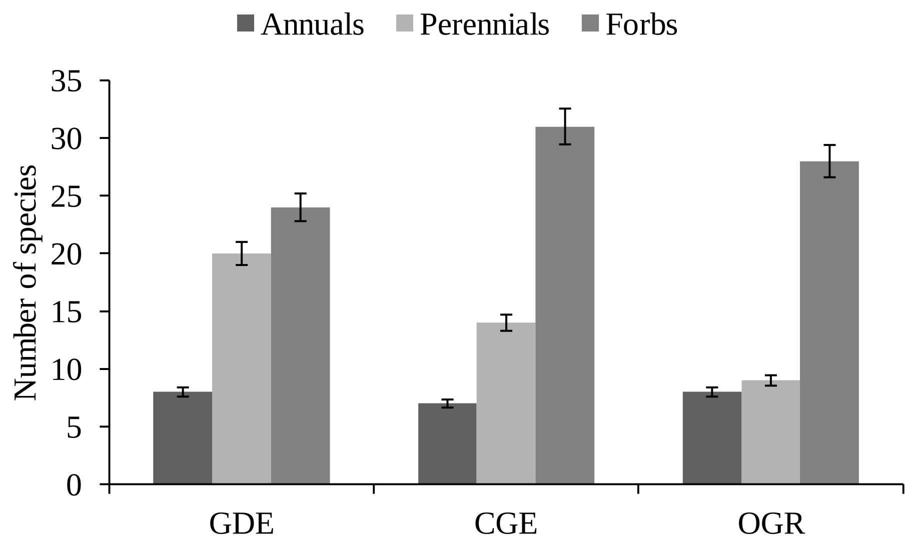
3.2.4 Statistical Analysis

One-way analysis of variance (ANOVA) was used to test for differences in vegetation cover, species diversity, and productivity between CGE, GDE and OGR. Effects of enclosure system and age of enclosure on cover, species diversity, and productivity were analyzed by two-way ANOVA. Means were separated using Fischer’s least significant difference (LSD) test at $P \leq 0.05$. All statistical analysis were conducted using Genstat 15th edition (VSN International, 2012).

3.3 Results

3.3.1 Species composition

A total of 83 grass and forb species were identified in the study area, where 40.0 and 60.0% represented grasses and forbs respectively. Of the grass species, 38.2% were annuals and 61.8% were perennials with over three quarters of the perennials (88.0%) observed in the enclosures (Figure 1). *Sporobolus pyramidalis*, *Digitaria macroblephala* and *Cynodon dactylon* were among the perennial grass species present only in GDE. *Brachiaria deflexa* grass and forbs, such as *Craterostigma hirsutum* and *Justicia exigua,* were recorded in OGR only (Table A1).



Vertical bars enclosure represents standard deviation (SD) of the mean. OGR – open grazing rangeland; GDE – grazing dominated enclosure; and CGE – Contractual grazing.

Figure 1. Composition of grass and forb species in enclosures and open grazing rangeland, in Chepareria, Kenya.

3.3.2 Herbaceous cover, relative abundance and species diversity index

Herbaceous cover was significantly higher in the enclosures than in continuously grazed areas and varied between 20.73 % in OGR and 40.22% in GDE (Table 1). Perennial grass cover dominated in GDE whereas annual grasses and forbs cover were high in OGR and CGE respectively (Table 1). The relative abundance of perennial grasses was considerably higher in GDE and lower in OGR (P<0.001, Table 1). Herbaceous species diversity index was also higher in the enclosures than in the adjacent open grazing areas with GDE and CGE exhibiting significant differences (Table 1). The herbaceous biomass in the enclosures was 7 times higher than in the open-access grazing areas (Table .1).

The age of enclosures showed significant effect on total herbaceous vegetation cover, perennial grass cover, species diversity index, and biomass production (P<0.05, Table 2). These values were consistently higher in the medium aged enclosures (11-20 years) than in the recently established (3-10 years) and old (>20 years) enclosures (Table 2).

Table 1. Vegetation cover, diversity and aboveground biomass of the three grazing systems in Chepareria.

	Grazing system			*P-value*	*L.S.D*
	OGR	CGE	GDE		
Cover					
Perennial grasses (%)	2.89 ± 1.48 c	7.84 ± 4.49 b	13.44 ± 3.57 a	<0.001	1.37
Annual grasses (%)	14.44 ± 2.45 a	7.71 ± 1.67 c	11.91 ± 2.75 b	<0.001	0.98
Forbs (%)	3.40 ± 2.21 c	18.87 ± 2.96 a	14.87 ± 17.05 b	<0.001	1.17
Total plant cover (%)	20.73 ± 2.64 c	34.42 ± 5.97 b	40.22 ± 5.48 a	<0.001	2.01
Relative abundance					
Perennial grasses	0.14 ± 0.07 c	0.22 ± 0.10 b	0.33 ± 0.07 a	<0.001	0.03
Annual grasses	0.70 ± 0.12 a	0.23 ± 0.05 b	0.30 ± 0.06 c	<0.001	0.04
Forbs	0.16 ± 0.10 c	0.56 ± 0.09 b	0.37 ± 0.06 c	<0.001	0.03
Species diversity index	0.69 ± 0.33 b	0.95 ± 0.11b	1.08 ± 0.02 a	<0.001	0.08
Herbaceous aboveground biomass (kg DM ha^{-1})	72.0 ± 24.7 c	483.1 ±70.0 b	560.4 ± 93.1 a	<0.001	61.1

Note: Values are means ± standard deviation (SD). Different lowercase letters indicate significant differences between grazing systems ($P < 0.05$). OGR – open grazing rangeland; CGE – contractual grazing enclosure; GDE, - grazing dominated enclosure.

Table 2. Effect of enclosure age on herbaceous vegetation cover and aboveground biomass in Chepareria, Kenya.

Age of enclosure (years)	Perennial grass	Annual grass cover	Forbs	Total vegetation cover	Diversity index	Aboveground biomass
			%			kg DM ha^{-1}
3- 10	9.10 ± 5.00 b	10.00 ± 2.78 a	17.17 ± 3.70 a	35.30 ± 7.00 b	0.99 ± 0.12 b	440.7 ± 167.0 b
11- 20	12.13 ± 4.33 a	9.26 ± 3.45 a	16.93 ± 3.66 a	38.80 ± 5.02 a	1.00 ± 0.07 a	566.7 ± 170.9 a
> 20	10.70 ± 5.09 ab	10.17 ± 3.05 a	16.50 ± 3.54 a	37.87 ± 6.68 ab	1.02 ± 0.10 ab	558.0 ± 194.3 a
P-value	0.01	0.28	0.69	0.05	0.03	0.01
$LSD_{0.05}$	1.642	0.962	1.281	2.358	0.032	73.9

Note: Values are means ± SE. Different lowercase letters within the same column indicate significant differences between means at $P \leq 0.05$. OGR – open grazing rangeland; CGE – contractual grazing enclosure; GDE, - grazing dominated enclosure.

4.4 Discussion

Regeneration of forbs relative to grasses may have been due to the presence of quality seeds in the soil (Hutchings and Booth, 1996). It has been suggested that the recovery of vegetation in rangelands depends on several factors including the availability of viable seed in the soil (Kinucan and Smeins, 1992). The prevalence of high perennial grass species in the enclosures compared to open grazing rangeland indicated that perennial grasses such as *Sporobolus pyramidalis*, *Digitaria macroblephala* and *Cynodon dactylon* would ultimately take over. The absence of grass species like *Cynodon dactylon* in the OGR implies that overgrazing in Chepareria results to the local extinction of highly desirable grasses. Studies have shown that overgrazing can lead extinction of preferred grass species in grazing lands (O'Connor and Roux, 1995; Haftay *et al.*, 2013; Lalampaa, 2016). Thus the grass species such as *Brachiaria deflexa* and forbs such as *Craterostigma hirsutum* and *Justicia exigua* were predominant in OGR system.

Intensive grazing caused degradation of vegetation cover in Chepareria. The low relative abundance of the perennial grass species and high abundance of annual grass species in the OGR suggest that there was a preferential consumption of perennial grasses that were subsequently outcompeted by the less-grazed ephemerals grass species, as also observed by (Westoby, 1979; Skarpe, 1991; Rooyen *et al.*, 2015). This also indicates that controlled grazing in the enclosure systems can improve the natural regeneration perennial grass species. Similar results were reported in arid and semi-arid rangelands of southern Tunisia and southern Ethiopia by Angassa and Oba (2010) and Gamoun *et al.* (2010), where vegetation cover and abundance of perennial grasses increased as following the establishment of enclosures.

The enclosures promoted herbaceous species richness and diversity and biomass, consistent with previous findings in the area by Wairore *et al.* (2015) and in the semi-arid rangelands of Tigray in Ethiopia by Mekuria and Veldkamp (2012). In the OGR, the low species diversity index was as a result of repeated grazing and human interference that negatively affected the regeneration and growth of plant species. Besides, the decline in species diversity and biomass production in the communal grazing land could be a result of the loss of seedling of some species unable to establish at early stage of development, and selective defoliation and trampling by grazing herbivores (Belaynesh, 2006; Abesha, 2014). Hence, the higher herbaceous species diversity in CGE and GDE relative to OGR suggested that the reduced disturbances caused by livestock favored the establishment of pioneer species in the enclosures. According to Milchunas *et al*. (1988), results from a grazing model showed that semiarid grasslands with long histories of grazing displayed a decline in species diversity with increasing grazing intensity due to compensatory growth. The observed results on species diversity do not support this concept as higher species diversity was found when the grazing pressure was reduced. The high aboveground herbaceous biomass in the enclosure systems demonstrated the importance of rangeland enclosures in promoting range productivity. The increase in biomass in enclosures could be due to the improved soil conditions which subsequently lead to the increased plant growth and accumulation of aboveground biomass.

The results showed that herbaceous cover, perennial grass cover, species diversity and aboveground biomass were higher in the middle age enclosures (10-20 years) than in the young and older enclosures, corroborating the observations in the grazing lands in Tigray in northern Ethiopia (Abesha, 2014). According to Oba *et al.* (2001) and Haftay *et al.* (2013), species richness is lower in heavily grazed grassland where biomass production is low and the two exhibited positive correlation. Contrary observations were reported by Zhang *et al.* (2005) and Angassa and Oba (2010), who found that herbaceous species richness, herbaceous biomass and grass basal cover declined with an increase in age of the enclosures. Zhang *et al.* (2005)

attributed the low species diversity, basal cover and biomass in the 6- and 10-year enclosures relative to the 18-year enclosure to the greater density of woody species in the younger enclosures.

4.5 Conclusion

The influence of pasture enclosure management on species diversity and aboveground biomass was significant. Perennial grasses dominated the GDE where species diversity was high and more biomass was accumulated. This was attributed to the reduced grazing pressure and disturbance in the enclosures compared to the open grazing rangeland. The enclosures not only improved basal vegetation cover but also increased herbaceous biomass production. The medium aged enclosures (11-20 years) were important to improve vegetation recovery over the younger and older enclosures in terms of perennial grass cover, basal cover, diversity and biomass. This may probably be as a result of grazing which influenced vegetation recovery and acted as a diversifying factor. It was evident that GDE emerged a better management strategy for improving aboveground vegetation cover and biomass production in the degraded semi-arid rangelands in West Pokot County.

CHAPTER FOUR

Enhancing soil organic carbon, particulate organic carbon and microbial biomass in semi-arid rangeland using pasture enclosures

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Abstract

Rehabilitation of degraded rangelands through the establishment of pasture enclosures (fencing grazing lands) is believed to improve soil quality and livelihoods, and enhance the sustainability of rangelands. Grazing dominated enclosure (GDE) and contractual grazing enclosure (CGE) are the common enclosure management systems in Chepareria,West Pokot County, Kenya. Under CGE, a farmer owning few animals leases the enclosure to households with relatively more livestock, while GDE is where the livestock utilizing the enclosure are purely owned by the farmer. Livestock management in both systems is via the free-range system. This study evaluated the effect of enclosure management on total soil organic carbon (SOC), particulate organic carbon (POC) and microbial biomass carbon (MBC) and nitrogen (MBN) as key indicators of soil degradation at 0 to 40 cm depth. The two enclosure systems were selected based on three age classes (3-10, 11-20 and >20 years since establishment) (n =3). The adjacent open grazing area (OGR) was used as a reference (n = 9). Relative to OGR, the pasture enclosures significantly decreased soil bulk density and increased the concentrations of total organic C, POC, MBC and MBN compared to the degraded OGR ($P < 0.001$). Significantly higher concentrations of POC and MBC was recorded in GDE than CGE ($P = 0.01$). The POC accounted for 24.5 – 29.5% of the total SOC. MBC concentrations ranged from 32.05 ± 7.25 to 96.63 ± 5.31 µg C g^{-1} of soil in all grazing systems, and was positively correlated with total SOC and POC ($P < 0.001$). The proportional increase in POC (55.6%) and MBC (30.5%) was higher in GDE compared to CGE (39.2 and 13.9% for POC and MBC respectively). This study demonstrated that controlling livestock grazing through the establishment of pasture enclosures is the key strategy to enhance total SOC, POC, MBC, and MBN in degraded rangelands; a precondition for improving soil quality. Therefore, the establishment of enclosures is an effective restoration approach to restore degraded soils in semi-arid rangelands.

4.1. Introduction

Human-induced soil degradation is a major concern globally (Oldeman, 1992; Oldeman et al., 2017), and has contributed to the decline in net primary productivity in arid and semi-arid lands (Zika and Erb, 2009). Overgrazing in rangelands has altered the natural ecosystem, causing disturbances in biotic and abiotic components and livelihood of the community. Among the negative impacts of overgrazing is the loss of soil organic carbon (SOC), a scenario that occurs both in the temperate(Holt, 1997; Hafner *et al.*, 2012; Wu *et al.*, 2014) and tropical rangelands (Mekuria et al., 2011; Mussa et al., 2017). Soil organic carbon is the basis for soil fertility, the source of energy for soil microorganisms and regulates climate and biodiversity (Lal, 2004a; Li *et al.*, 2013; Lal, 2014; Dou *et al.*, 2016). Restoration degraded rangelands have therefore attracted considerable attention in the recent past. The restoration of degraded grazing land may be important to improve the accumulation of SOC soil. The establishment of pasture enclosures by fencing degraded communal grazing areas has been reported to reduce the negative impacts of continuous grazing by preserving soil resources, leading to accumulation of SOC that was previously lost (Wang *et al.*, 2011; Li *et al.*, 2013; Mussa *et al.*, 2017). Understanding the dynamics and potential of soil to store organic carbon is not only essential for improving soil quality and enhancing the sustainability of rangelands in Sub-Saharan Africa, but also mitigate climate change by offsetting CO_2 emissions (Savadogo *et al.*, 2007; Hafner *et al.*, 2012).

Soil organic carbon is regarded as an indicator of soil quality and by extension, the state of soil degradation as it determines soil structure, nutrient retention and supports biological diversity (Giller *et al.*, 1997; Gil-Sotres *et al.*, 2005; Chazdon, 2008). The reduction or loss of SOC could, therefore, lower soil fertility and consequently, lead to land degradation (Rounsevell et al., 1999). According to (Wang et al., 2011), the establishment of enclosure in a degraded rangeland resulted to a 34% increase in total SOC content in the upper 40cm layer of soil. Besides, (Mureithi *et al.*, 2014b) recounted that degraded soils in semi-arid rangeland with low levels of organic carbon may be functionally improved by establishing pasture enclosures. However, (Savadogo et al., 2007) and (Chazdon, 2008) acknowledged that changes in total SOC require several years to detect. The labile fractions of total SOC include particulate organic carbon (POC) and microbial biomass carbon (MBC) (Weil et al., 2003). These fractions may be more sensitive to land management than the total SOC. The POC acts as a substrate for soil microorganisms and influences soil nutrient cycles and biological properties of soil (Weil and Magdoff, 2004).

Livestock enclosures have gained cognizance as a successful tool for controlling heavy grazing and land degradation in Eastern Africa (Makokha *et al.*, 1999b; Mekuria *et al.*, 2007a; Mwilawa *et al.*, 2008a; Mureithi *et al.*, 2014b). In the arid and semi-arid rangelands of Western Kenya, efforts to restore degraded grazing lands through the establishment of pasture enclosures started in the mid-1980s (Vi- Agroforestry, 2007). As indicated by (Wairore *et al.*, 2015c), grazing dominated enclosure (GDE) and contractual grazing enclosure (CGE) are the common types of enclosure management systems in West Pokot County, Kenya. The enclosures are privately owned and utilized, with an average size of 5 hectares (Wairore *et al.*, 2015c). Contractual grazing represents a grazing arrangement where a farmer owning few animals leases the enclosure to households with relatively more livestock. On the other hand, GDE is where the livestock utilizing the enclosure is purely owned by the farmer. The stocking rate of the enclosures in the area ranges between 1 and 42 animals with a mean of 7 animals (Wairore *et al.*, 2015c). Livestock management in both CGE and GDE systems is via the free-range system. Previous studies in semi-arid rangelands show that POC and MBC concentrations increase after enclosing degraded grazing lands (Silveira *et al.*, 2013; Mureithi *et al.*, 2014b; Wu *et al.*, 2014), while others reported that grazing management has no significant impact on the dynamics of labile fractions of carbon (Pringle et al., 2014; Stavi et al., 2015). These variations were attributed to differences in soils (Bruun et al., 2015) and vegetation characteristics such as litter quantity and quality (Castellano et al., 2015; Yé et al., 2017; Yu et al., 2017). However, pasture management in the former studies was via cut-and-curry where livestock was not allowed to graze (excluded) in the enclosures.

Despite the fact that the practice of enclosures has existed in West Pokot County for over three decades, data on the effectiveness of these enclosures to restore degraded soils in terms of organic C in the area is lacking. Understanding the effect of enclosure management system and their age on SOC is crucial to offer the most effective carbon management options in rangelands. Based on the hypothesis that GDE enclosures are more effective to restore degraded soils than CGE enclosures by improving the content of soil organic carbon and microbial biomass, this study was carried out to determine the concentrations of total SOC, POC and MBC in CGE and GDE under three age-classes (3-10, 10-20, and >20 years since effective protection) with the similar quantifications in the adjacent open grazing areas as the baseline.

4.2. Methods

4.2.1. Study site

The study was conducted in Chepareria Ward (01° 18' 17" - 01° 19' 41" N and 035° 14' 16" - 035° 15' 49" E, 1680 m. a. s. l) in West Pokot County, Northwestern Kenya. The area is classified as semi-arid; receiving an average rainfall of 280 mm of rainfall for the short rains which occur between mid-October and January and 570 mm for the long rains which occur from mid-March to July (County Government of West Pokot, 2013). The annual average daily air temperature ranges between 16 and 30 ^{o}C (County Government of West Pokot, 2013). The soils are predominantly sandy clay to loamy sand and are classified as Haplic Lixisols (Hiederer and Köchy, 2011). Vegetation is predominantly grassland (*Themeda triandra, Eragrostis superba, Cymbopogon validus, Cenchrus ciliaris* and *Cynodon dactylon*) (Oduor, 2018), with scattered native *(Acacia spp*., *Balanites aegyptiaca*, and *Kigelia africana*) and exotic (*Grevillea robusta*) tree species [22]. The average herbaceous vegetation cover range between 20.7% in open grazing rangeland and 40.2% in enclosure systems, with 72.0 kg dry matter (DM) ha^{-1} and 521.8 kg DM ha^{-1} of herbaceous above-ground biomass in open grazing rangeland and enclosure respectively (Oduor, 2018). The traditional open grazing areas are characterized by free-range grazing of livestock with a stocking rate that exceeds the upper limit of the enclosure systems. The open grazing areas had a history of severe land degradation prior to the establishment of enclosures in mid-1980s by Vi-Agroforestry (Vi- Agroforestry, 2007).

4.2.2. Soil sampling

Soil sampling was carried out during the short rain season, November 2016. In consultation with the local leaders and Vi-Agroforestry officials, the CGE and GDE enclosures were grouped into three age classes: 3-10 years, 11–20 years, and >20 years, and three enclosures were randomly selected from each enclosure type/age class combination. A total of 18 enclosures were selected for sampling. Nine open grazing sites (OGR) were selected as controls (n = 9). This gave a total of 27 sampling sites. Within each enclosure/age class and in the adjacent open grazing areas, three 50-m transects were laid out in a Z-shaped orientation, at least 10 m from the edge to avoid edge effects. Along each transect, five sampling points were laid at 10 m apart and soil samples collected using a soil auger at 0-10, 10-20, and 20-40 cm depths. The five soil samples at each depth and within each transect were mixed to form a composite sample, producing three composite samples (one for each depth) for each transect and a total of nine composite samples (3 depths x 3 transects) from each enclosure and open grazing site. A total of 243 soil samples were obtained (27 sampling sites × 9 composite samples). About 0.5-kg sub-sample was placed in air tight plastic bags for soil moisture determination, extraction of microbial biomass carbon (MBC) and microbial biomass nitrogen (MBN). The remainder of the soil was air-dried, sieved through a 2-mm mesh and stored at 4 ^{o}C in a refrigerator for physical and chemical analyses. Steel cylinders of 98.2 cm^{-3} were used to obtain undisturbed soil samples for soil bulk density determinations, using the same sampling design. Within each transect, a 40 cm profile pit was dug in and one core sample taken in each depth, making a total of three core samples per transect.

4.2.3. Soil analysis in the laboratory

Soil water content was determined gravimetrically by oven-drying 100 g soil sub-sample at 105 ^{o}C to constant weight for 48 hours (Reynolds, 1970). Soil pH and electrical conductivity (EC) were determined in soil-water suspension in the ratio1:2.5 (weight/volume). Soil pH was measured using a glass electrode pH meter (model: HI 2211, Hanna instruments), while EC was measured using a conductivity meter (model: HI 9812, Hanna Instruments). Soil bulk density (BD) was determined using core ring method by oven-drying core samples at 105 ^{o}C for 48 hours (Blake, 1965), and particle size distribution using the hydrometer method (Bouyoucos, 1962). Total soil organic carbon (SOC) was determined using the wet oxidation method (Nelson and Sommers, 1996), total nitrogen (TN) was determined using the Kjeldahl method (Kjeldahl, 1883) and cation exchange capacity (CEC) was determined by the ammonium acetate (NH_4OAc) method as described by (Chapman, 1965).

Physical fractionation was used to determine particulate organic carbon content, associated with the sand fraction (2000 – 53 μm), following procedures by Cambardella and Elliott (Cambardella and Elliott, 1992). Approximately 20-g of sieved (<2.0 mm) air-dried soil sub-sample was dispersed with 70 ml of 5-g L^{-1} sodium hexametaphosphate solution and the suspension was passed through a 53 μm sieve using a jet distilled water. The material retained in the sieve was dried at 45 ^{o}C for 48 hours in a forced air oven. The oven-dried material was ground and analyzed for organic carbon by the wet oxidation method (Nelson and Sommers, 1996) and TN using the Kjeldahl method (Kjeldahl, 1883).

Chloroform fumigation-extraction method was used to determine MBC and MBN contents in soil (Vance et al., 1987). Ethanol-free chloroform was used to fumigate 10 g of field-moist soil samples for 24 h in a vacuum desiccator at room temperature. Another set of the same soil samples were not fumigated. The soluble C from the fumigated and non-fumigated samples was extracted with 50 ml of 0.5-M K_2SO_4 solution. The extracted soil MBC was measured spectrophotometrically at 600 nm. The difference between the extracted C in the fumigated and non-fumigated soils represented the microbial biomass C (Nunan et al., 1998). MBN was determined by digesting 20 ml of the soil extract using Kjeldahl digestion and the digest analyzed for total N. Correction factors (kc) of 0.45 and 0.54 were used for MBC and MBN respectively (Brookes *et al.*, 1985; Beck *et al.*, 1997).

4.2.4. Statistical analysis

Effects of grazing systems and soil depths, and enclosure type and age on total SOC, SOC fractions, microbial biomass, and the interactions were analyzed by two-way analysis of variance (ANOVA) using Genstat 15th edition (VSN International, 2012). Means were separated using Fischer's protected least significant difference (LSD) test at $P \leq 0.05$. Pearson correlation analyses were conducted to establish the relationship between soil organic carbon fractions and soil texture and microbial biomass carbon using SPSS 20th version (SPSS, 2011).

4.3. Results

4.3.1. Soil physical and chemical characteristics

The sand, silt and clay contents were similar for all the grazing systems (Table 3). Soil bulk density in the 0-10 cm was lowered significantly from 1.49 g cm^{-3} in the OGR to 1.42 and 1.39 g cm^{-3} in CGE and GDE enclosures respectively ($P < 0.001$, Table 3). Soil moisture content was generally higher in the enclosures relative to the OGR and increased with depth. The enclosure system did not significantly alter soil pH and CEC (Table 3).

Table 3. Soil physical and chemical properties under different grazing management systems in Chepareria, Kenya.

Grazing system	Soil depth (cm)	pH	CEC	Sand	Silt	Clay	Moisture Content	BD
			Cmol kg^{-1}	%				g cm^{-3}
GDE	0-10	6.1 ± 0.55	8.0 ± 1.03	78.7 ± 2.61	5.4 ± 1.62	13.6 ± 1.17	6.79 ± 2.27 bc	1.39 ± 0.10 bc
	10-20	6.1 ± 0.30	8.3 ± 0.93	77.8 ± 2.52	5.7 ± 2.37	14.2 ± 1.09	7.28 ± 2.29 abc	1.37 ± 0.05 c
	20-40	6.0 ± 0.34	9.1 ± 0.78	78.2 ± 2.52	6.0 ± 2.00	14.0 ± 1.15	8.16 ± 2.23 ab	1.36 ± 0.06 c
CGE	0-10	6.2 ± 0.22	8.2 ± 0.75	81.3 ± 1.29	7.8 ± 1.60	13.4 ± 1.21	6.32 ± 2.76 c	1.42 ± 0.10 abc
	10-20	6.0 ± 0.61	8.7 ± 0.95	80.6 ± 1.57	8.0 ± 2.23	13.7 ± 1.16	6.83 ± 2.68 bc	1.46 ± 0.10 ab
	20-40	6.2 ± 0.24	8.6 ± 1.16	80.6 ± 1.60	7.7 ± 2.88	13.4 ± 1.18	8.51 ± 2.44 a	1.45 ± 0.05 ab
OGR	0-10	6.3 ± 0.27	9.0 ± 0.92	79.5 ± 1.61	6.8 ± 1.88	13.8 ± 1.29	5.85 ± 2.51 c	1.49 ± 0.05 a
	10-20	5.2 ± 0.56	8.6 ± 0.95	78. 9 ±1.57	7.3 ± 2.09	13.7 ± 1.30	6.38 ± 2.55 c	1.47 ± 0.06 a
	20-40	5.0 ± 0.24	8.7 ± 0.90	78.7 ± 1.48	7.2 ± 1.75	13.9 ± 1.17	6.78 ±2.22 bc	1.47 ± 0.06 a
P-value		0.13	0.56	0.16	0.08	0.168	0.01	<0.001
$LSD_{0.05}$		NS	NS	NS	NS	NS	1.13	0.07
cv%		6.6	10.7	2.4	31.8	8.7	15	5.2

Note: Values are means ± SD (n = 9). Different lowercase letters within the same column indicate significant differences between means at $P \leq 0.05$. NS – not significant. OGR – open grazing rangeland; CGE – contractual grazing enclosure; GDE, - grazing dominated enclosure; BD- bulk density; cv% - coefficient of variation.

4.3.2. Total soil organic carbon and nitrogen

Grazing system and soil depth had significant (P < 0.001) effect on total SOC concentration. The proportion total SOC in the enclosures was 27.1% higher compared to OGR and the concentration decreased with depth (Table 4). However, the difference in SOC content between CGE and GDE was not significant. On the other hand, the values of total N content in CGE and GDE were similar but highly significant (P < 0.001) compared to total N content in OGR (Table 4). Within the enclosure systems, the age of enclosure had no effect on total SOC and TN concentrations (P = 0.52).

Table 4. Soil organic carbon and total nitrogen concentrations at three depths under different grazing management systems.

	Total soil organic carbon (g kg^{-1})			Total nitrogen (g kg^{-1})		
Depth (cm)	OGR	CGE	GDE	OGR	CGE	GDE
0-10	4.93±0.69 Ba	6.22±0.78 Aa	6.61±0.89 Aa	0.53±0.07 Ba	0.63±0.08Aa	0.65±0.08 Aa
10-20	4.88±0.65 Ba	5.86±0.67 Aa	6.28±0.99 Aa	0.58±0.11 Ba	0.63±0.08Aa	0.61±0.07 ABa
20-40	4.36±0.74 Bb	5.57±0.57 Ab	5.47±0.77 Ab	0.52±0.10 Bb	0.61±0.08Aa	0.59±0.07 Ab
Pooled mean	4.72±0.73 B	5.88 ±0.72 A	6.12±1.00 A	0.54±0.09 B	0.62±0.08 A	0.62±0.08 A

Values are mean ± standard deviation (SD) (n = 9). Values with different uppercase letters across the rows (grazing systems) and the lowercase letters within columns (soil depths) are significantly different at $P < 0.05$. OGR – open grazing rangeland; CGE – contractual grazing enclosure; GDE, - grazing dominated enclosure.

4.3.3. Particulate organic carbon

Grazing management significantly affected the concentration of POC (Table 5). The concentration of POC in the 0-10 cm increased significantly ($P < 0.001$) from 1.40±0.21 in OGR to 2.01±0.26 in CGE and 2.28±0.34 g kg^{-1} in GDE (Table 5). Unlike total SOC, the difference in POC content between CGE and GDE was significant ($P = 0.01$), but exhibited no significant variations among the age classes ($P = 0.71$). Relative to OGR, the proportion of POC in CGE and GDE was high by 38.8 and 55.2% respectively. In general, POC accounted for 24.5, 27.1 and 29.5% of the total SOC in OGR, CGE and GDE respectively.

Table 5. Distribution of particulate organic carbon with depth in three grazing systems in Chepareria, Kenya.

	Particulate organic carbon (g kg^{-1})			Particulate organic nitrogen (g kg^{-1})		
Depth (cm)	OGR	CGE	GDE	OGR	CGE	GDE
0-10	1.40 ± 0.21 Ca	2.01 ± 0.26 Ba	2.28±0.34 Aa	0.19 ± 0.12 Aa	0.16 ± 0.04 Aa	0.16 ± 0.03 A b
10-20	1.20 ± 0.24 Cb	1.52 ± 0.26 Bb	1.80±0.25 Ab	0.17 ± 0.07 Aa	0.18 ± 0.02 Aa	0.18 ± 0.04 Aab
20-40	0.88 ± 0.15 Bc	1.31 ± 0.16 Ac	1.32±0.19 Ac	0.18 ± 0.04 Aa	0.18 ± 0.05 Aa	0.20 ± 0.03 Aa
Pooled mean	1.16 ± 0.30 C	1.61 ± 0.37 B	1.80 ± 0.50A	0.18 ± 0.07 A	0.17 ± 0.05 A	0.18 ± 0.04 A

Note: Values represent mean ± standard deviation (SD) (n = 9). Values with different uppercase letters across the rows and lowercase letters within columns are representing significant differences between grazing systems and soil depths respectively, at $P < 0.05$.
OGR – open grazing range; GDE - grazing dominated enclosure; and CGE - contractual grazing enclosure.

4.3.4. Microbial biomass carbon and nitrogen

Enclosures significantly increased MBC and MBN, with higher concentrations observed in the 0-10 cm depth in all the grazing systems (P < 0.001, Table 6). Compared to the mean MBC recorded in OGR, the MBC contents in CGE and GDE significantly increased by 13.9% and 30.5% (P < 0.001). Within the enclosures, significantly higher concentration of MBC was observed in GDE relative to CGE (P = 0.01). However, MBC and MBN concentrations were similar across the enclosure age classes (P = 0.63 and 0.97 for MBC and MBN respectively).

Table 6. Distribution of microbial biomass carbon and nitrogen with depth in three grazing systems in Chepareria, Kenya.

	Microbial biomass carbon (μg g^{-1})			Microbial biomass nitrogen (μg g^{-1})		
Depth (cm)	OGR	CGE	GDE	OGR	CGE	GDE
0-10	77.08 ± 5.25 Ca	88.22 ± 6.16 Ba	96.63 ± 5.31 Aa	37.57 ± 2.01 Ba	38.44 ± 2.26 Ba	40.90 ± 5.68 Aa
10-20	73.67 ± 4.27 Cb	81.05 ± 3.74 Bb	94.10 ± 5.55 Aa	36.24 ± 2.50 Aa	37.57 ± 3.45 Ab	37.89 ± 3.30 Ab
20-40	32.05 ± 7.25 Cc	38.94 ± 10.42 Bc	47.77 ± 6.04 Ab	18.01 ± 3.71 Cb	22.09 ± 3.04 Ac	21.64 ± 3.34 Ac
Pooled mean	60.93 ± 21.36 C	69.40 ± 23.04 B	79.50 ± 23.28 A	31.97 ± 7.49 B	31.34 ± 10.00 B	33.48 ± 9.49 A

Note: Values represent mean ± SD (n = 9). Values with different uppercase letters across the rows and lowercase letters within columns are representing significant differences between grazing systems and soil depths respectively, at $P < 0.05$.
GDE - grazing dominated enclosure, CGE - contractual grazing enclosure, and OGR – open grazing rangeland.

4.3.5. Relationship between SOC, TN, POC and Microbial biomass

Total SOC exhibited significant ($P < 0.001$) positive correlation with TN, POC and MBC at all soil depths, but was only significant with PN at 10-20cm depth (Table 7). Total nitrogen showed significant relationship with POC at all soil depths and with MBC at the surface 0-10cm only. The POC positively associated with MBC at all soil depths with the relationship being stronger at the surface 0-10cm ($r = 0.63$) compared to 10-20 and 20-40cm depths ($r = 0.57$ and 0.41 respectively) (Table 7).

Table 7. Linear correlation analysis of SOC, TN, POC, PON, MBC and MBN in the three soil depths (n = 81).

Depth (cm)		SOC	TN	POC	PN	MBC	MBN
0-10	SOC	-					
	TN	0.71**	-				
	POC	0.86**	0.70**	-			
	PN	0.06	0.10	0.04	-		
	MBC	0.57**	0.46**	0.63**	0.18	-	
	MBN	0.32**	0.10	0.38**	0.01	0.21*	-
10-20	SOC	-					
	TN	0.54**	-				
	POC	0.81**	0.42**	-			
	PN	0.29**	0.14	0.25*	-		
	MBC	0.40**	0.06	0.57**	0.15	-	
	MBN	0.04	0.10	0.03	0.16	0.18	-
20-40	SOC	-					
	TN	0.66**	-				
	POC	0.91**	0.65**	-			
	PN	0.16	0.19	0.14	-		
	MBC	0.30**	0.09	0.41**	0.10	-	
	MBN	0.17	0.14	0.17	0.00	0.05	-

Values are correlation coefficient, *r*. SOC – total soil organic carbon; TN – total nitrogen; POC – particulate organic carbon; PON – particulate organic nitrogen; MBC – microbial biomass carbon and MBN – microbial biomass nitrogen.
*Denotes significant correlation at the 0.05 level
** Denotes significant correlation at the 0.01 level: others are not significant.

4.4. Discussion

Similarities in soil pH, texture and CEC indicated that areas inside enclosures were comparable to the communal grazing lands prior to the establishment of enclosures and that differences in the measured variables among the studied sites were caused by land use change and not by inherent site variability. Low CEC indicated the deficiency of significant amounts of exchangeable cations such as Ca^{2+}, Mg^{2+}, and K^{+} (McKenzie et al., 2004). Despite the fact that the top-soil bulk density in all the grazing systems were generally below the root-restricting value of 1.80 g cm^{-3} for loamy sand soils (NRCS, 2001), the lower bulk density under GDE and CGE indicated the potential of enclosures to improve soil physical properties such as compaction that hamper critical soil functions, like the capture, storage and supply of water for plants (Kinyua et al., 2010). This result agreed with (Yong-Zhong et al., 2005) who showed that grazing exclusion sites reduced soil bulk density compared to the adjacent continuous grazing sites in the sandy grassland of Inner Mongolia, China. Higher soil moisture content in CGE and GDE could perhaps be as a result of the improved soil physicochemical properties. The reduced soil bulk density in the enclosed systems may have increased the rate of water infiltration in the soil due to high pore space. As indicated by (Castellano and Valone, 2007), low water infiltration rates in degraded grasslands relative to enclosed sites were due to the high soil compaction induced by the grazing livestock. On the other hand, higher SOC in the enclosures increased the capacity of the soil to retain moisture (Hudson, 1994). Increase in moisture with depth may be due to high evaporative loss at the soil surface than in the deep soil horizons.

Irrespective of land use, the amounts of SOC and TN in soil are determined by the balance between organic matter inputs and losses (Benbi et al., 2015). The significantly higher level of SOC and TN in the enclosures compared to the open grazing land was probably because of the reduced soil disturbance by grazing animals. This prompted the production of aboveground biomass (Oduor, 2018), thereby facilitating the accumulation and storage of C into the soil and its mineralization releasing nitrogen. According to (Yong-Zhong *et al.*, 2005; Jeddi and Chaieb, 2010b; Oduor, 2018), high removal of forage by the grazing animals in open grazing lands reduces herbaceous vegetation cover and accumulation of aboveground biomass. Consequently, this reduced the amount of C incorporated into the soil in open grazing lands. In addition, the high bulk density in the surface 0-10 cm and low soil moisture content in OGR could have reduced the input of soil organic matter by hampering storage and supply of water for plant growth (Castellano and Valone, 2007; Kinyua et al., 2010). The reduction in SOC with increasing soil depth in all grazing systems suggests that organic matter accumulation in the surface 0-10 cm was higher than in the 10-20 cm and 20-40 cm depths. Higher SOC in the 10-20 cm and 20-40 cm in CGE and GDE relative to OGR could be as a result of the reduced grazing activities, which promoted root growth and accumulation of root biomass (Yu et al., 2017). This facilitated the incorporation organic C in the subsoil. The reduction in SOC content with increasing soil depth is consistent with previous research in semi-arid rangelands in Tigray, Ethiopia and Inner Mongolia in China (Mekuria and Aynekulu, 2011b; Mussa *et al.*, 2017). These results corroborate with studies conducted in semiarid grasslands in Northern and Eastern Ethiopia and in Northwestern Kenya where higher soil organic C in enclosures was attributed to increased biomass production and reduced trampling by the grazing livestock (Mekuria *et al.*, 2011; Mureithi *et al.*, 2014b; Mussa *et al.*, 2017). Age of enclosure did not influence SOC levels because enclosures are continuously used for periodic grazing. This agrees with other studies in the area (Svanlund, 2014a; Ituika, 2016). Furthermore, the ~30 years of existence of enclosures in the area could be a short time to detect the changes in total organic carbon (Roldan et al., 2005).

The higher concentration of POC in the enclosures suggested that the accumulation of organic matter was higher in the fenced areas than in the open grazing areas. Compared to total SOC, the considerably higher POC content in GDE than in CGE implied that POC is more sensitive to changes in grazing management. The results were consistent with (Conant et al., 2003; Plaza-Bonilla et al., 2014) who reported that particulate organic carbon responds to changes in grazing management compared to total SOC. Higher concentration of POC in GDE relative to CGE and OGR may be due to the lower grazing pressure which reduced soil disturbance. The reduced soil disturbance permitted the protection of soil organic matter from decomposition. According to (Burke et al., 1995; Goebel et al., 2009), trampling by livestock disintegrates soil macro-aggregates thus exposing soil organic matter to decomposition. The incorporation and stabilization of particulate organic matter into soil aggregates is a dominant factor for protecting organic carbon in grazing lands (Six *et al.*, 1998; Gale *et al.*, 2000; Yost *et al.*, 2016). In addition, the higher herbaceous vegetation cover observed in CGE and GDE compared to OGR (Oduor, 2018), greatly contributed to the conservation of POC in the enclosures by reducing erosion. Higher levels of POC in the surface 0-10 cm soil compared to 10-20 and 20-40 depths suggest that plant roots supplied more organic matter in the surface soil compared to the subsoil. The sandy nature of soils in the study area implies that the POC have low colloidal protection, and consists mainly of partially humified plant residues. The proportion of POC of the total SOC in this study (24.5 – 29.5%) was within the reported ranges of between 2 and >50% in semiarid grasslands (Chan, 1997; Gill et al., 1999; Kaye et al., 2002).

Similar to the trends observed with POC, the significantly higher contents of MBC and MBN in the GDE and CGE compared to OGR was attributed to the increased concentration of POC in the enclosures which acted as a source of energy for soil microbiota. This was supported by the significant positive correlation exhibited between MBC and POC in all soil depths. Moreover, the significant decrease in MBC and MBN content with depth in all the grazing systems indicated a higher potential for organic matter inputs from root exudates and plant litter in the surface soil relative to the deeper soils (Liu et al., 2012). These results were consistent with studies by Wu (Mureithi *et al.*, 2014b; Wu *et al.*, 2014) in a semi-arid rangeland in North-Western Kenya and Hulunbuir grassland of Inner Mongolia where higher microbial biomass C and N contents were recorded in enclosed areas than in the open grazing lands. The range of microbial biomasses C recorded in this study (32.1 to 96.6 µg g^{-1} soil) was relatively low compared to those recorded in Baringo County in Kenya (73 to 156 µg g^{-1} soil) (Mureithi *et al.*, 2014b). This could be attributed to the differences in soil type and management strategies in the two areas. However, it has been recognized that microbial biomass recovers slowly in sandy soils in semiarid climates (Burke et al., 1995; Weber et al., 2016). Nonsignificant variations in POC and microbial biomass levels among the enclosure age classes could be the short residence time soluble fractions of organic C (Buyanovsky et al., 1994; Schlesinger and Andrews, 2000).

4.5 Conclusions

This study showed that the soils in the semi-arid rangelands of Chepareria are very fragile. Relative to the enclosure systems, continuous grazing in the open grazing land caused a considerable increase soil bulk density and additional loss of total SOC, total N, POC, and microbial biomass contents. The observed variations in all these parameters indicated that the communal grazing lands were in a degraded state. This may portray serious consequence for soil quality, plant growth and loss of livelihood in tropical rangelands where grazing is the major land-use. Restoration of the degraded grazing land via the establishment of pasture enclosures increased the contents total SOC and total N and reduced soil bulk density. The concentrations of POC, MBC and MBN were considerably higher in GDE than in CGE. The results supported the hypothesis that GDE enclosures are more effective to restore degraded soils than CGE enclosures. This indicates that the degraded soils in the open grazing land can indeed recover following the establishment of enclosure. The POC and MBC were more sensitive to grazing management than total SOC and can be used as indicators of the soil C dynamics in semi-arid rangelands. Therefore, this study demonstrated that controlling livestock grazing through the establishment of enclosures is integral to increase SOC stocks or reduce its losses; a precondition for improving soil quality and climate change mitigation. Future research should focus on enclosures carrying capacity and seasonal ecosystem dynamics of carbon and nitrogen to better understand the ecology of this fragile ecosystem.

CHAPTER FIVE

Pasture enclosures increase soil carbon dioxide flux rate in semi-arid rangeland, Kenya

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Abstract

Grazing management may influence **s**hort-term soil greenhouse gas (GHG) emission that account for a considerable portion of atmospheric carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). The effects of restoration of degraded rangeland on soil CO_2, CH_4 and N_2O flux rate and in east Africa are poorly understood. A field experiment was conducted in a semiarid rangeland in Chepareria, Kenya, to determine the rates of emission of soil CO_2, CH_4 and N_2O using static opaque chambers. Two enclosure systems namely; grazing dominated enclosure (GDE) and contractual grazing enclosure (CGE) were selected for this study. The open grazing rangeland (OGR) adjacent to the enclosure was used as the control. The mean emission rates were 18.6 μg N m^{-2} h^{-1}, 50.1 μg C m^{-2} h^{-1} and 199.7 mg C m^{-2} h^{-1} for N_2O, CH_4 and CO_2 respectively. The soil CO_2 was higher in GDE (224.4 mg C m^{-2} h^{-1}) and CGE (239.9 mg C m^{-2} h^{-1}) systems, relative to OGR (102.4 mg C m^{-2} h^{-1}) ($P < 0.001$). Similarly, higher CH_4 and N_2O emissions were observed in GDE and CGE than in OGR, however the differences were not significant ($P = 0.33$ and 0.53 for CH_4 and N_2O, respectively). Generally, the flux rates of CO_2, CH_4 and N_2O exhibited significant ($P < 0.001$) positive relationship with soil moisture content. This study suggested soil CO_2 fluxes in the three grazing systems exhibit obviously spatial and temporal variation, and that soil moisture is the major factor affecting soil GHG fluxes in Chepareria.

5.1 Introduction

The atmospheric concentration of greenhouse gases (GHG) has increased over the last century due to anthropogenic activity, and is highly associated with increased mean global temperatures (Karl and Trenberth, 2003). Globally, land use change and forestry, and agriculture accounts for about 10.0% and 11.2% of total anthropogenic GHG emissions, respectively (Tubiello *et al.*, 2015). In Kenya, land use change and agriculture sectors contribute at 38% and 41% of total anthropogenic GHG emissions, respectively (NEMA, 2015). Approximately 85% of Kenya's land area is classified as arid and semi-arid (ASAL) (NEMA, 2015), where grazing is the dominant land use. Agricultural systems account for low amounts of GHG per unit land (Zhuang and Li, 2017); however, the vast area covered by the agricultural systems may mean that these lands contribute a large percentage to the national GHG inventories. Whereas open grazing management have caused severe deterioration of soil and vegetation properties (Sandhage-Hofmann *et al.*, 2015), fencing of communal grazing land is a restoration technique commonly practiced in drylands (Mwilawa *et al.*, 2008b; Shang *et al.*, 2014; Mekuria *et al.*, 2015).

Unlike exclosure systems where pasture management is via cut-and-carry, livestock-based enclosure systems where animal grazing is allowed are common in Chepareria, Kenya. According to Wairore *et al.* (2015), grazing dominated enclosure (GDE) and contractual grazing enclosure (CGE) are the common forms of grazing enclosures in the area. The enclosures are privately owned with an average size of five hectares and are managed through annual deferred grazing where livestock graze in the open rangeland (OGR) during the rainy season, which is also the season of vegetative growth, and then allowed in the enclosed areas during the dry season (Wairore *et al.*, 2015). Contractual grazing enclosure represents a grazing arrangement where enclosure owner lease the land to household with relatively more animals. Grazing dominated enclosure is where the animals grazing the enclosure belong to the owner of the enclosure. Grazing intensity and frequency of utilization of enclosure follow the order of OGR > CGE > GDE. Research in northern Ethiopia suggests that soil and vegetation properties within enclosures improve with the age of enclosures (Mengistu *et al.*, 2005b; Abebe *et al.*, 2006). According to Mengistu *et al.* (2005), considerable species diversity and soil organic matter were observed inside enclosures compared to open grazing areas.

Degraded soils often have low GHG emission rates (Pelster *et al.*, 2017), and restoration of these soils may cause increases in the GHG emissions (Zhuang and Li, 2017). The increased GHG emissions from restored rangelands are thought to be related to the increased vegetation cover and biomass production (Mekuria *et al.*, 2015; Yan *et al.*, 2015), soil organic carbon (SOC) content (Shang *et al.*, 2014), improved soil moisture content (Mekuria *et al.*, 2015), and the reduced soil compaction (Han *et al.*, 2008). Vegetation contributes to soil organic matter which may increase the rate of soil respiration and organic matter mineralization, emitting CO_2 to atmosphere (Davidson *et al.*, 1998; Hu *et al.*, 2016). Raich and Schlesinger (1992) observed that soil CO_2 flux is a result of root respiration and decomposition of organic matter. In turn, mineralization of soil organic matter also leads to accumulation of ammonium and nitrates thereby stimulating nitrification and denitrification (Hanan *et al.*, 2016), which contribute up to 70% of the global N_2O emissions (Syakila and Kroeze, 2011). Dung from grazing animal is the main source of CH_4 in rangelands (Samal *et al.*, 2015; Assouma *et al.*, 2017), not to mention the livestock CH_4 emissions from enteric fermentation. The effect of grazing on bio-chemical processes that determine GHG emissions may vary with type of grazing management practice (Herman *et al.*, 1995). For example, high concentrations of nutrients and microorganisms in vegetated sites may increase GHG emission compared to bare soil, with soil moisture strongly regulating the fluxes (Otieno *et al.*, 2010; Liu *et al.*, 2014; Li *et al.*, 2016a). Unger *et al.* (2010) reported that drying and wetting cycles stimulate microbial respiration rate, though respiration declined naturally by 40% within few hours after wetting. Microbial respiration is considered the largest source of atmospheric CO_2 in the carbon cycle (Hashimoto *et al.*, 2015).

Information on the effect of restoration of degraded rangeland on herbaceous vegetation cover and biomass production, and soil organic carbon (SOC) in relation to soil CO_2, CH_4 and N_2O flux rate and in the semi-arid rangelands of east Africa is scanty and remains little understood (Lal, 2008). Hence, the evaluation of the effect of pasture enclosures on land rehabilitation on SOC and GHG fluxes in semiarid rangelands is important for determining the effect of restoration on climate change mitigation. This study was based on the hypothesis that higher SOC and GHG flux rate was expected to occur in enclosure management systems than in the open grazing rangeland.

5.2 Materials and methods

5.2.1 Site description

The study was conducted in Chepareria in West Pokot County, Kenya, during the dry season and long rainy season of 2017. Chepareria Ward is located on the lower slopes of Kamatira hills (between latitude 1° 18' - 1° 19' N and longitude 35° 14' - 35° 15' E) at an altitude of 1560 meters above mean sea level. The area is classified as semi-arid (Agroecological zone IV); receiving on average 280 mm of rainfall for the short rains, which occurs between mid-October and January, and 570 mm for the long rains, which occur from mid-March to July (CGWP, 2013b). The maximum and minimum temperatures occur in the months of February and July, respectively, ranging between 16 ^{o}C and 30 ^{o}C. The soils are predominantly sandy clay and are classified as Haplic Lixisols (Hiederer and Köchy, 2011). Soil physicochemical characteristics of the study site are given in Oduor *et al*. (2018). The main land-use and source of livelihood in the area is predominantly agro-pastoralism (Svanlund, 2014b). The area had a history of severe land degradation prior to establishment of enclosures (Nyberg, 2015).

5.2.2 Enclosure selection and sampling design

In consultation with local leaders, 18 enclosures were selected from CGE and GDE based on three age classes (3 - 10, 11 - 20, and > 20 years, since establishment) with three replications in each age class (n = 18). The adjacent open grazing rangeland (OGR) was considered as a control (n = 9), giving a total of 27 sampling plots.

5.2.3 Gas sampling and laboratory analysis

Field gas measurements were conducted between 29 January and 28 February 2017 for the dry season and between 13 April and 13 May 2017 for the wet season, between 9.00 – 15.00 hours. At each sampling point, 3 static opaque frames measuring 27 cm × 37.2 cm × 10 cm were installed 5 cm deep two months prior to the first sampling, and remained in place throughout the study period. Sampling was conducted once a week during the dry season and twice a week during the wet season, making a total of eight sampling dates. On each sampling date, a lid (27×37.2×12.5 cm) fitted with a reflecting tape at the top, a rubber sealing, a fan, a 50 cm non-forced vent, a thermometer and a sampling port, was fitted to the frame using metal clamps for 30 minutes. Four gas samples were taken at 10 min intervals (0, 10, 20, and 30 min). A 20 ml sample was drawn from each of the three chambers using a 60 ml syringe at each time interval, mixed and then the pooled sample was transferred into 20 ml pre-evacuated glass vial to achieve over-pressure (Arias-Navarro *et al.*, 2013). The CO_2, CH_4 and N_2O were determined at the Mazingira Centre at the International Livestock Research Institute (ILRI) in Nairobi, Kenya, using a gas chromatograph (8610C; SRI, Santa Monica, CA) equipped with a flame ionization detector for CH_4 and CO_2 (after being methanized) and a ^{63}Ni electron capture detector for N_2O. The CO_2, CH_4 and N_2O concentrations in the samples were calculated based on the peak areas measured by the gas chromatograph relative to the peak areas measured from calibration gases. The GHG flux rates were calculated using linear regression of gas concentrations versus chamber closure time and corrected for temperature and moisture using Equation 4 (as outlined in Qui *et al.*, 2006);

$$F = \frac{P}{Po} * \frac{M}{Vo} * \frac{dc}{dt} * \frac{To}{T} * H \qquad \text{Equation 4}$$

Where:

F is the flux rate in mg C m^{-2} h^{-1} for CO_2, μg C m^{-2} h^{-1} for CH_4 and μg N m^{-2} h^{-1} for N_2O;

P is the atmospheric pressure of the sampling site (Pa); *M* is the gas mass (g mol^{-1});

dc/*dt* is the rate of concentration change;

T is the absolute chamber temperature at sampling time (^{o}C);

Vo, *Po*, and *To* are the molar volume, atmospheric pressure, and absolute chamber temperature, respectively (mL, Pa, and °C), under standard conditions; and

H is the chamber height over the soil surface (cm).

Air temperatures (TA) at 1.5 m above the ground and inside the chamber (TC) were measured during gas sampling using digital probe thermometer (Einstich - TFA). Soil moisture content (SM, %v/v) and soil temperature (T_S) were measured at 5 cm depth using soil moisture and temperature sensor model 5MT, Decagon Devices Inc. Soil moisture was converted to water-filled pore space (WFPS) using the soil bulk density through Equation 5 (Zhang *et al.*, 2012);

$$\text{WFPS} = \left(\frac{\text{volumetri moisture content (\%)}}{(1 - \left(\frac{\text{BD}}{2.65}\right))} \right) \qquad \text{Equation 5}$$

Where:

WFPS is the water filled pore space;

BD is soil bulk density (g cm^{-3}) and 2.65 is the assumed soil particle density (g cm^{-3}).

5.2.4 Statistical analysis

Shapiro-Wilkes test for normality was performed on CO_2, CH_4 and N_2O flux rates at $P \leq 0.05$. The effects of different grazing systems and enclosure type and age on GHG flux rates were analyzed by two-way ANOVA. Means were separated using Fischer's protected least significant difference (LSD) test using GenStat, 14th edition (VSN International, 2012), with differences considered significant at $P \leq 0.05$. Stata version 12.0 was used to conduct linear regression analysis to determine the soil and vegetation properties that influence CO_2, CH_4 and N_2O flux rates. Pearson correlation analysis was performed using SPSS 20th edition test the association between the soil and vegetation parameters with CO_2, CH_4 and N_2O flux rates.

5.3 Results

5.3.1 Soil moisture, air and soil temperature, and water filled pore space

Air temperature ranged from 25.2 and 28.6^oC while soil temperature varied between 31.5 and 38.1^oC. Soil moisture (SM) ranged between 7.2 and 11.8 (%v/v) during the dry season and 16.8 and 20.9% (v/v) during the wet season in all the grazing systems, and was consistently higher in GDE and CGE than in OGR ($P < 0.001$) (Table 8). The corresponding WFPS was also higher in GDE and CGE than in OGR ($P<0.001$) and varied between 10.2 – 31.9% and 29.0 – 52.1% during the dry and wet seasons, respectively. The minimum SM content corresponded with the maximum soil temperature and vice versa (Table 8).

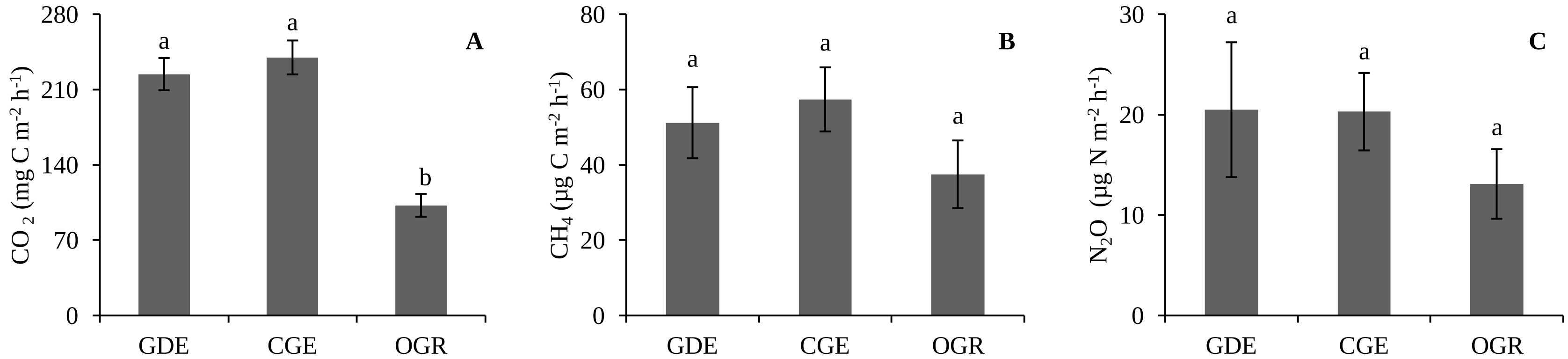
Table 8. Soil moisture, air and soil temperature, and water filled pore space in the three grazing systems during the dry and wet seasons.

	Grazing system	Season	
		Dry	Wet
Air temperature (°C)	GDE	28.55 ± 0.35 a	25.31 ± 0.66 a***
	CGE	28.48 ± 0.36 a	25.31 ± 0.33 a***
	OGR	27.97 ± 0.42 a	25.20 ± 0.77 a***
Soil temperature (°C)	GDE	38.13 ± 0.68 a	31.52 ± 0.90 a***
	CGE	37.06 ± 0.87 a	31.79 ± 0.64 a***
	OGR	35.39 ± 0.90 b	31.67 ± 1.42 a***
Soil moisture (% v/v)	GDE	11.77 ± 1.11 a	20.89 ± 0.64 a***
	CGE	9.78 ± 0.99 ab	19.55 ± 0.56 a***
	OGR	7.16 ± 1.12 b	16.76 ± 0.87 b***
Water filled pore space (%)	GDE	25.87 ± 2.45 a	46.01 ± 1.43a***
	CGE	21.44 ± 2.19 ab	43.07 ± 1.26 ab***
	OGR	16.81 ± 2.73 b	38.39 ± 2.00 b***

Note: Values are seasonal means ± standard deviation (SD) (n = 9). Different lowercase letters indicate significant differences among grazing systems for each parameter (*P* < 0.05). ***Denotes significant difference between seasons (*P* < 0.001). Open grazing rangeland – OGR; Grazing dominated enclosure – GDE; and Contractual grazing enclosure – CGE.

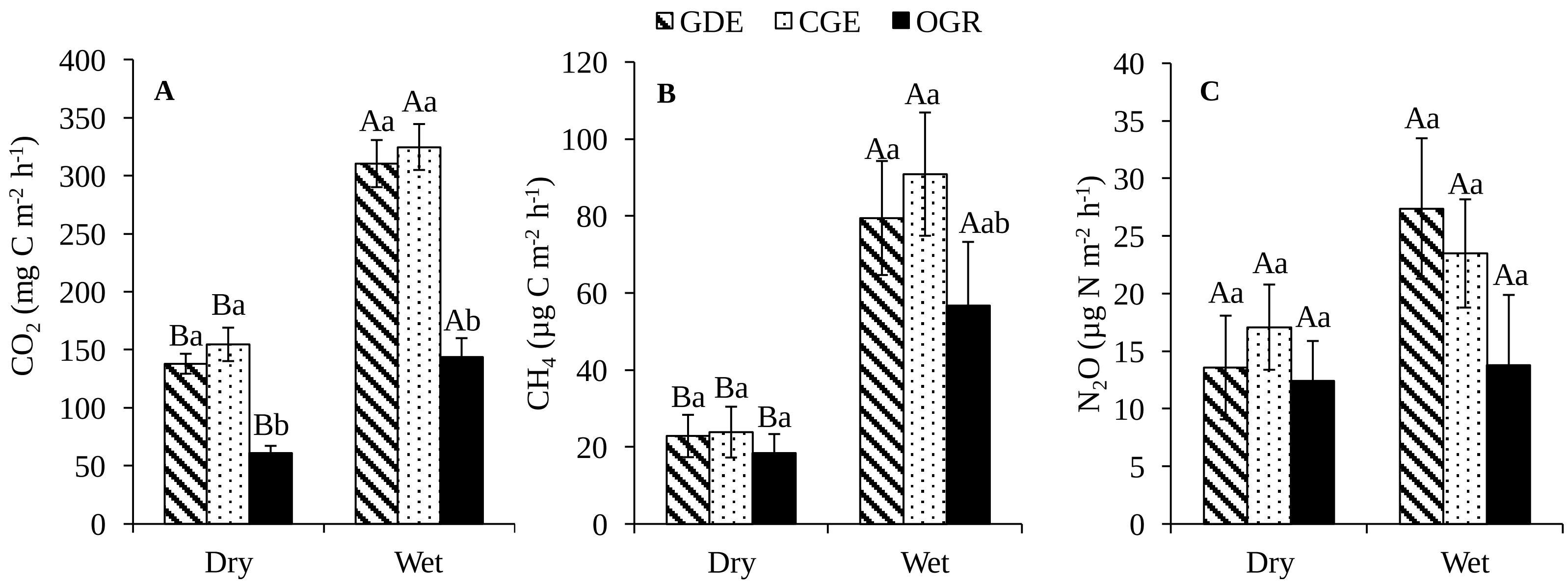
5.3.2 Greenhouse gas emissions

The mean (± SD) soil CO_2 flux rates in CGE (239.9 ± 15.8) and GDE (224.4 ± 15.0) were significantly higher than in OGR (102.4 ± 10.6) (*P*<0.001, Figure 2A). However, the difference in soil CO_2 flux rate between the CGE and GDE was not significant (Figure 2A). In contrast, significant interaction was exhibited between grazing system and season with higher CO_2 emissions observed during the wet season in all the grazing systems (*P* = 0.02, Figure 2). Relative to the minimum and maximum CO_2 emission in the OGR, the minimum and maximum CO_2 emission in CGE and GDE were higher by 186.3 and 32.1% and 298.7 and 41.5% respectively. This implied that GDE substantially increased soil CO_2 emission. Generally, the soil CO_2 emission rate increased with the age of enclosure and was 209.2 ± 17.5, 234.5 ± 18.8 and 252. 7 ± 19.9 mg C m^{-2} h^{-1} in the 3-10, 11-20 and >20 years age classes respectively, although the differences were not significant (*P*=0.27,Table9).



Legend: GDE - grazing dominated enclosure; CGE - contractual grazing enclosure; OGR - open grazing rangeland. Different lowercase letters denote significant differences between the grazing systems. Error bars represent standard error of the mean (SE).

Figure 2. Mean emission of soil CO_2 (A), CH_4 (B), and N_2O (C) in Chepareria, Kenya



Note: GDE - Grazing dominated enclosure; CGE - Contractual grazing enclosure; OGR - open grazing rangeland. Different uppercase and lowercase letters denote differences between seasons and the grazing systems, respectively. Error bars represent standard error of the mean (*n* = 12).

Figure 3. Seasonal emissions of soil CO_2 (A), CH_4 (B) and N_2O (C) in Chepareria, Kenya.

Table 9. Greenhouse gas flux rates in the three enclosure age in Chepareria, Kenya

Enclosure system	Age class (years)	CO_2 mg C m^{-2} h^{-1}	CH_4 µg C m^{-2} h^{-1}	N_2O µg N m^{-2} h^{-1}
GDE	3 - 10	186.0 ± 22.8	34.9 ± 8.2	32.4 ± 18.9
	11 - 20	226.3 ± 21.7	63.1 ± 16.3	9.5 ± 2.4
	> 20	260.9 ± 31.1	55.6 ± 17.9	18.95 ± 6.6
CGE	3 - 10	232.4 ± 26.2	60.8 ±12.9	17.5 ± 5.9
	11 - 20	242.7 ± 31.2	53.3 ± 14.6	26.2 ± 6.9
	> 20	244.6 ± 25.6	58.0 ± 21.1	17.8 ± 7.4
P-value		0.50	0.52	0.25

Note**:** Values are means ± SD (n = 3)**.** Different lowercase letters indicate significant differences among grazing systems (P < 0.05). GDE – grazing dominated enclosure; CGE – contractual grazing enclosure.

Though soil CH_4 and N_2O uptakes (negative fluxes) were recorded in GDE, CGE and OGR, the mean flux rates were positive indicating that the grazing systems acted as net sources for atmospheric CH_4 and N_2O. The CGE and GDE exhibited higher emission rates of CH_4 and N_2O than OGR; but the differences between the grazing systems were not significant (P = 0.29 and 0.58 for CH_4 and N_2O respectively) (Figures 2B and C). Higher CH_4 and N_2O emission rate were observed during the wet season than dry season in all the grazing systems, but this was only significant for CH_4 emission (P < 0.001) (Figures 3A and B). Enclosure age did not influence CH_4 and N_2O flux rates (Table 9).

6.3.3 Relationship between greenhouse gas fluxes and environmental parameters

Soil moisture exhibited significant positive correlation with CO_2, CH_4 and N_2O flux rates (P < 0.001); with peak emission rates were observed at soil moisture content between 15 and 25 % v/v. This relationship was higher for CO_2 than for CH_4 and N_2O (Table 10), with r^2= 0.10, 0.15 and 0.39 for N_2O, CH_4, and CO_2 respectively. In addition, CO_2 emission rate showed significant positive relationship with organic carbon and above-ground biomass (Table 10). Generally, the CH_4 emissions exhibited significant positive correlation with CO_2 fluxes (r = 0.54, P < 0.001).

Table 10. Relationship between GHG flux rates and the environmental parameters under the grazing systems

	CO_2			CH_4			N_2O		
	Coeff.	*Std. Error*	*P-value*	*Coeff.*	*Std. Error*	*P-value*	*Coeff.*	*Std. Error*	*P-value*
Intercept	*275.8*	*235.55*	*0.24*	*14.9*	*161.58*	*0.93*	*166.94*	*96.94*	*0.09*
Soil organic carbon	34.03	16.31	**0.04**	17.47	11.19	0.12	4.13	6.71	0.54
Total nitrogen	-123.1	136.37	0.37	-239.73	93.54	0.06	-80.92	56.12	0.15
Bulk density	137.15	139.82	0.33	113.54	95.91	0.24	110.24	57.54	0.06
Soil temperature	0.39	1.43	0.78	-1.71	0.98	0.08	0.4	0.59	0.49
Soil moisture	10.6	1.16	**<0.001**	3.35	0.8	**<0.001**	1.9	0.48	**<0.001**
Total herbaceous vegetation cover	2.52	2.91	0.39	2.96	2	0.14	0.73	1.2	0.54
Above ground biomass	0.17	0.08	**0.03**	0.07	0.05	0.17	-0.03	0.03	0.38

Coeff = coefficient, CO_2 = carbon dioxide, CH_4= methane, and N_2O = nitrous oxide

5.4 Discussions

5.4.1 Soil moisture and water filled pore space

Higher soil moisture content in the enclosed systems could potentially be as a result of the improved soil physical properties. Lower soil bulk density and higher soil porosity in GDE and CGE than in OGR in the surface 0-10 cm increased the rate of water infiltration in the soil. In addition, the higher vegetation cover in GDE and CGE lowered surface runoff allowing more time for water to infiltrate the soil and protected the soil from direct sunlight lowering evaporation of moisture from the soil surface In contrast, studies have reported that higher vegetation cover usually increase total transpiration and therefore final water balance from evapotranspiration may be higher in areas with high vegetation than in an area with low vegetation (Newman *et al.*, 2006). Yan *et al.* (2016) showed that soil compaction and vegetation cover are the major factors controlling soil moisture holding capacity in grazing rangelands. Also, lower vegetation cover in OGR exposed the surface soil to raindrop impact, that breakdown aggregates, clog soil pores and may create almost an impermeable layer (Vaezi *et al.*, 2017), thus reducing the amount of water infiltrating on the soil surface. The results agreed with studies in Ethiopia which attributed the high soil moisture content in restored systems to high vegetation cover (Mekuria *et al.*, 2015).

5.4.2 Greenhouse gas emissions from soil

The mean C flux rates in GDE (224.4 mg C m^{-2} h^{-1}) and CGE (239.9 mg C m^{-2} h^{-1}) were higher than those recorded in a grazed alpine steppe in China (ranged between 92.7 ± 11.7 and 156.1 ± 19.6 mg C m^{-2} h^{-1}) (Wei *et al.*, 2012). The latter study in China was conducted under temperate and humid conditions characterized by short summers and long cold winters, mean annual temperature ranged from -1.5^{o}C to 2.5^{o}C. The relatively higher temperatures in Chepareria enhanced soil respiration which resulted in higher CO_2 emission. Besides, the sandy nature of soils in Chepareria (Oduor *et al.*, 2018), imply that the soils are well drained and this increased diffusion rate of gases from the soil. The higher soil CO_2 emission in GDE and CGE than in OGR was attributed to the considerably higher concentration of total SOC and its labile fraction in the enclosures (Oduor *et al.*, 2018), which acted as substrate source for soil microorganisms. Similarly, the higher soil moisture contents of in CGE and GDE than in OGR created a favourable climate which increased the autotrophic respiration of plant roots and respiration of soil microbes. These were supported by the positive relationship that was exhibited between CO_2 with SOC and soil moisture, suggesting that availability of soil organic matter substrates and soil moisture status are the major factors controlling soil respiration in the area. These results were consistent with previous studies which showed that soil moisture and SOC are important factors controlling soil CO_2 emission in rangelands (Raich and Schlesinger, 1992; Yiqi and Zhou, 2010; Moyano *et al.*, 2013; Knowles *et al.*, 2015). The observed positive relationship between CO_2 and aboveground biomass implies that aboveground biomass had direct influence on the belowground root biomass which, in turn, influenced autotrophic respiration of plant roots. Previous studies in degraded rangelands reported that restoration reduced or had no impact on soil respiration (Frank *et al.*, 2002; Klumpp *et al.*, 2007; Chen *et al.*, 2015; Sharkhuu *et al.*, 2016). Our observation was consistent with reports which showed that the establishment of enclosures on previously degraded semi-arid grassland increased the emission of CO_2 from soil (Thomas, 2012; Shi *et al.*, 2017). The observed higher emission CO_2 from soil during the wet season in all the grazing systems was attributed to the increased soil moisture content brought about by episodes of rainfall. The higher CO_2 emission rate in the old enclosures (>20 years), could be due to the dominance of perennial grasses (Chapter Four, Table 2) which have greater root biomass than annual grass and forbs. This could have increased the production of root exudates and substrates available in the rhizosphere (Janssens *et al.*, 2001), consequently increasing respiration activities in the soil.

Maximum CO_2 emission rate occurred at WFPS between 25-55%. Below 20% WFPS, soil respiration was inhibited by limited soil moisture whereas above 55% WFPS, respiration was inhibited by the low oxygen availability as most of the soil pores were filled with water. The limited availability of oxygen reduced the decomposition of organic matter and production of CO_2 and its diffusion into the atmosphere (Knowles *et al.*, 2015). These findings corroborated with studies which reported enhanced soil CO_2 emission in vegetated sites compared to bare soil (Arneth *et al.*, 2017; Assouma *et al.*, 2017), and that soil respiration increased with increasing soil moisture and SOC content (Xu *et al.*, 2016; Chen *et al.*, 2017).

The positive CH_4 flux rates in this study imply that soils in GDE, CGE and OGR acted as sources of atmospheric methane, contrary to most agricultural soils in Kenya and Tanzania which act as net sinks for atmospheric methane (Rosenstock *et al.*, 2016; Pelster *et al.*, 2017). Though rangeland soils are widely regarded as sinks for atmospheric CH_4 (Werner *et al.*, 2007; Li *et al.*, 2016b; Pelster *et al.*, 2017), results in this study show that tropical rangeland soils may emit CH_4 to the atmosphere. Samal *et al.* (2015) reported that high soil compaction and limited soil moisture in semiarid ecosystems create anaerobic microsites with low redox potential and reduce the activity of methanotrophs. Higher CH_4 emission was recorded in GDE and CGE than in OGR mainly due to limited soil moisture in OGR; however, the differences between the grazing systems were not significant. Low moisture content inhibited the activity methanogens (Le Mer and Roger, 2001). The observed seasonal variation in CH_4 emission rate can be attributed to the differences in soil moisture content during the dry and wet seasons which affected the activity of soil methanogens. The positive correlation between CH_4 and CO_2 fluxes suggests that respiration is a confounding factor influencing methane production in grazing lands, by creating anaerobic microsites for CH_4 production. Our observation reiterated studies which reported positive CH_4 fluxes in tropical rangeland soils (Topp and Pattey, 1997; Sey *et al.*, 2008; Tang *et al.*, 2017). Strong relationship has been reported between CH_4 emission and soil water content in grassland soils (Steudler *et al.*, 1996; Mosier *et al.*, 1998; Verchot *et al.*, 2000) but not always (Fernandes *et al.*, 2002).

The average N_2O emission rates in this study (18.6 μg N m^{-2} h^{-1}) were lower than those reported by Assouma *et al.* (2017) in a semi-arid rangeland in Senegal (104.2 μg N_2O-N m^{-2} h^{-1}), and comparable to fluxes recorded in smallholder farms in Kisumu County in Kenya (below 20 μg N_2O-N m^{-2} h^{-1}) (Pelster *et al.*, 2017). The grazing systems recorded similar N_2O flux rates under all pasture management practices (P = 0.33) in both seasons, with weak emissions during the dry season, < 20 μg N m^{-2} h^{-1}. High surface-soil compaction under OGR created anaerobic microsites for N_2O production while high soil moisture content under CGE and GDE enhanced N_2O production potential by producing anaerobic microsites for denitrification, thus similar emission rates. Steffens *et al.* (2008) and Chen *et al.* (2017) reported that grazing did not have significant effect on N_2O emission rate. Soil N_2O emissions showed a weak positive relationship with soil moisture (R^2 = 0.10, P<0.001), whereas other studies showed that N_2O emissions were insensitive to soil moisture (Yan *et al.*, 2008). This shows that soil moisture was more critical to determine N_2O flux in semi-arid rangeland soils than the rest of the measured soil and vegetation characteristics by causing a flush of inorganic nitrogen and labile C (Jacinthe and Lal, 2004; Borken and Matzner, 2009). According to Bateman and Baggs (2005), nitrification process dominates at WFPS between 35-60% and above 60% WFPS denitrification processes predominate in semiarid conditions. Generally, WFPS in this study was below 60% suggesting that the soils were aerobic, and that N_2O production was probably through denitrification in aerobic microsites. According to Khalil *et al*. (2004) and Bateman and Baggs (2005), the production of N_2O under aerobic soil conditions occurs through the partial oxidation of NH_4^+ into NO_2^-. The NO_2^- then diffused into anaerobic (or microaerobic) sites within the soil where it is subsequently reduced into N_2O by denitrification.

5.5 Conclusions

Restoration of degraded land through establishment of pasture enclosures improved soil physicochemical properties, vegetation cover and aboveground biomass production. Consequently, the healthy rangelands enhanced the release of soil CO_2-C into the atmosphere, but had no significant impact on the emission of CH_4-C and N_2O-N, with soil moisture content playing the key role in controlling the flux rates in the area. Higher CH_4 emission was observed during the wet season indicating the importance of wetting in increasing CH_4 emissions from the tropical rangeland. These findings indicate that rangeland restoration increases the emission rates of CO_2, CH_4 and N_2O from soil in West Pokot County, Kenya.

CHAPTER SIX

General discussion, conclusions, and recommendations

6.1 General discussion

Enclosures are widely adopted by agro-pastoralists in semi-arid rangelands to rehabilitate degraded communal grazing areas. However, previous work focused on exclosure systems where pasture management was through cut-and-curry system, and there is still limited knowledge on the influence of grazing enclosures on the characteristics of herbaceous vegetation, soil carbon and emission of greenhouse gases from soil. The notably higher vegetation cover, diversity index and aboveground biomass in GDE than in CGE and OGR supports the effect of repeated trampling and consumption of vegetation by livestock. The reduced grazing pressure in enclosures allowed time for natural regeneration of vegetation, and hence improving cover, biomass production and composition of plant species. In turn, this increased the production and incorporation of plant litter and plant root turnover in the soil. The regeneration of certain plant species has influence on the root-to-shoot ratio (Redin *et al.*, 2018), and thereby on soil organic carbon accumulation. Bushchiazo *et al*. (1991) and Redin *et al.* (2018) reported that perennial grass species are efficient in restoring organic C and N in sandy soils compared to annual grass species. Consequently, concentrations of total SOC and its particulate fraction were higher in the restored pastures relative to the communal grazing lands. This could be due to a better efficiency of roots of perennial grasses to produce organic residues as compared to annual grass species. In addition, the encrustation of litter in less-disturbed soils is a major mechanism responsible for protecting organic carbon in rangelands (Chapman and Bolen, 2015). Soil microbial biomass exhibited stronger relationship with POC than with total SOC in the three soil depths. This suggested that the labile fraction of SOC was the main source of energy for the growth of soil microbiota in the area. The higher concentrations of MBC and MBN in GDE and in the surface

0-10 cm of soil where POC was highly concentrated implies that biological activity in rangeland soils is dependent on the availability of decomposable or active organic materials.

The improved soil and vegetation properties in the pasture enclosures created a microclimate that influenced microbial processes that led to the emission of CO_2, N_2O and CH_4 from soil. According to Smith *et al.* (2003), the changes in soil physicochemical properties as a result of grazing influence the emission of CO_2, N_2O and CH_4 from soil. Presence of vegetation has been shown to increase rhizospheric and microbial respiration due to the secretion of root exudates. On the other hand, soil moisture influences the activity of soil microorganisms, whereas bulk density and porosity influence the diffusivity of gases from the soil to the atmosphere. As observed in this study, the higher emission of CO_2 from soil in the enclosure systems than in the OGR was due to the improved vegetation and soil physicochemical properties. The high vegetation cover, soil organic carbon, soil moisture content and low bulk density in the enclosures created favourable condition for soil respiration, and diffusivity (Hassler *et al.*, 2015) of CO_2 in the enclosures.

Though rangeland soils are widely regarded as sinks of atmospheric CH_4 (Werner *et al.*, 2007; Christiansen *et al.*, 2016; Li *et al.*, 2016b; Pelster *et al.*, 2017), contrasting results showing that the soils in grazing lands of Chepareria, Kenya, emit CH_4 into the atmosphere. Relative to other agricultural soils, the high soil bulk density in may have created anaerobic microsites with low redox potential, which in turn accelerated the activity of methanogens and subsequent production of CH_4 (Brewer *et al.*, 2018). In addition, since the measurements of GHGs were conducted under natural field conditions with livestock grazing activities going on, the measured CH_4 could have been released from the traces of dung (or manure) that was deposited within the chambers and in the surrounding. The higher CH_4 emission during the wet season than in the dry season was attributed to the changes in soil moisture content which controls microbial activities.

Nitrification and denitrification process which produce N_2O in soils often occur in close vicinity and occur simultaneously. (Nielsen *et al.*, 1996; Abbasi and Adams, 1998). However, denitrification process is the major producer of N_2O in soils and has been regarded to occur under anaerobic conditions only, but it is now well established that the process can occur in apparently aerobic environments also (Khalil *et al.*, 2004). The distribution of denitrification activity in aerobic soils is heterogeneous, and is to a large extent associated with the amount and location of active organic carbon (Christensen *et al.*, 1990; Kuzyakov and Blagodatskaya, 2015) which promotes the consumption of O_2, thus creating anoxic microsites with low redox potential. Therefore, similarity emission of N_2O within the enclosure systems and in the open grazing rangeland could be as a result of the high soil bulk density and POC in the OGR and enclosures, respectively, which created anaerobic hotspots that produced N_2O in equal proportions.

6.2 Conclusions

The results in this study demonstrated that;

1. Establishment of pasture enclosures in degraded rangelands played a key role in improving vegetation cover, diversity and biomass production, which is important for sustainable utilization of rangelands in arid and semi-arid areas.
2. Restoration of the open grazing rangelands through the establishment of pasture enclosures increased the contents total SOC and reduced soil bulk density, indicating that they were in a degraded state prior to the establishment of enclosures. Moreover, the concentrations of POC and MBC were considerably higher in GDE than in CGE

suggesting that GDE enclosures were effective in improving the soil quality than CGE enclosures.

3. The increased emission of CO_2-C from the soil into the atmosphere in the enclosures was as a result of the improved vegetation and soil physicochemical properties which supported respiration activities in the soil. Soil moisture content plays the key role in controlling the GHGs flux rates in Chepareria.

6.3 Recommendations

1. Establishment of pasture enclosures should be considered as a valuable local intervention that can be out-scaled to other arid and semi-arid lands in Kenya and Sub-Saharan Africa for improved production of pasture and sustainable utilization of rangelands.
2. Future research should focus on the ecosystem carbon balance, carrying capacity of the enclosures and Spatial heterogeneity of soil GHGs emissions in semi-arid rangelands in order to evaluate the coherent patterns in annual CO_2, N_2O, CH_4 fluxes from soils across the pastoral landscape units (i.e. enclosures, forest plantations, water points and settlements).

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APPENDIX

Table 11. Percent (%) cover of individual species in the enclosures and open grazing rangeland.

Species	Life form	OGR	CGE	GDE
Grasses				
Aristida kemensis	A	2.6	-	12.6
Brachiaria deflexa	A	1.9	-	-
Brachiaria eruciforaus	A	1.8	6.2	-
Brachiaria reptans	A	-	-	12.8
Chloris pycnothrix	A	-	9.1	11.5
Digitaria nodosa	A	-	7.6	-
Digitaria velutina	A	4.3	6.2	16
E. Congesta	A	5.6	-	12.1
Eragrostic tuncifolia	A	-	7.1	-
Eragrostis congesta	A	2.1	8.5	16.4
Eragrostis tuneifolia	A	1.9	9.3	12.6
Setaria pallide fusca	A	6.1	-	13.2
Aristida adoerisis	P	-	7.2	13.2
Bothrichloa misculpta	P	-	5.6	9.1
Brachiaria brizantha	P	-	6.1	17.2
Cynodon dactylon	P	-	6.9	18.2
Chloris gayana	P	-	12.6	12.9
Cynodom dactylon	P	-	-	11.2
D. Macroblephara	P	1.0	-	11.3
Digitaria macroblephala	P	-	-	14.5
Digitaria milansiana	P	3.1	8.1	12.1
Digitaria spp	P	-	9.2	16.8
Enteropogon macrostachyus	P	4.9	7.3	15.4
Eragrostis braunii	P	2.3	5.3	16.4
Eragrostis superba	P	2.4	-	16.2
Harpachine schimperi	P	-	-	13.2
Heteropogon contortus	P	4	10.9	-
Hyparrhenia hirfa	P	-	7.4	15.7
Microchloa kunthii	P	3.3	11.9	17.3
Panicum maximum	P	-	2.4	12.6
Paspalum scrobiculatum	P	3	8.9	-
Sporabolus pyramidalis	P	2	-	12.4
Lonchocarpus rogusus	P	-	-	13.4

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Species	Life form	OGR	CGE	GDE
Forbs				
Alyscarpus rogusus	F	7	20.1	11.2
Barleria acanthoide	F	-	-	16.9
Bracharia eruciformis	F	1.6	-	-
Carchorus olitorus	F	1.8	-	-
Cerchorus olitorus	F	-	-	16.1
Commehria benghalensis	F	-	18.9	12.3
Commelina spp	F	-	-	13.4
Corchorus olitorus	F	1	21.3	-
Crabbea velutina	F	5.2	-	-
Craterostigma hisutum	F	1.2	-	-
Crossandra nilotica	F	4.1	-	17.6
Crotolaria spinosa	F	2.8	-	-
Dichondra respens	F	-	-	18.2
Digitaria semipilosum	F	-	18.2	-
Dychoriste radicans	F	3.4	19.7	19.6
Elvolvulus alsinoide	F	-	20.2	17.5
Erlangea cordifolia	F	-	-	18.8
Erlangea calycina	F	-	21.8	-
Erlangea cordifolia	F	-	19.8	22.9
Euphorbia hirfa	F	2.3	17.3	-
Euphorbia inequilatera	F	3.5	17.9	9.7
Fuerstia Africana	F	3.2	19.6	18.9
Fuetrstia africana	F	-	16.8	-
Heliofropium steudneri	F	-	15.7	-
Hupoestes verticillaris	F	6.2	-	-
Hypoeste verticillar	F	4.8	17.8	-
Indigofera brevicelyx	F	-	19.6	-
Indigofera spicata	F	-	16.4	-
Indigostis schimperi	F	-	18.7	-
Indigotera brericalyx	F	-	-	12.9
Indigotera spicata	F	-	15.1	-
Justicia exguea	F	3.4	18.2	-
Kohautia coccinea	F	5.1	15.6	14.6
Leucas martinensis	F	3.6	16.1	11.1

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Species	Life form	OGR	CGE	GDE
Forbs				
Lndigofera spicata	F	-	15.1	-
Lndigotera schimper	F	2.4	-	-
Pentanisia ouranogyne	F	3.6	-	-
Polyghala sphenopfera	F	-	15.4	-
Rhamphicarpa	F	-	17.2	-
Richaedia brasiliensis	F	4.4	17.4	14.4
Ruclia patula	F	2.3	16.3	11.9
Senna mimosoides	F	1.3	19.2	8.8
Sida ovate	F	4.2	-	9.1
Solanum incanum	F	-	15.5	14.7
Sterile material	F	2.4	21.1	-
Stylosanthes frusticosa	F	4.3	-	-
Tridax procambense	F	5.1	-	12.2
Trifolium semipilasum	F	3.2	17.1	17.3
Trifolium sohnstorii	F	2.4	-	-
Triumfetta flavescense	F	-	-	16.8
Zornia glochidiata	F	-	19.2	-

A- Annual grass; P – perennial grass; F – forbs; OGR – open grazing rangeland; CGE – contractual grazing enclosure; and GDE – grazing dominated enclosure