Quantitative and qualitative herbage yield, and carbon sequestration in subtropical grasslands subjected to different precipitation, grazing management and land use type

By

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Declaration

I, Deribe Gemiyo Talore, declare that the thesis, which I hereby submit for the degree of Doctor of Philosophy in Animal Science at the University of Pretoria, is my own work and has not previously been submitted by me for the degree at this or any other tertiary institution.

Date---

Signature---

Preface

This dissertation is based on the following chapters, which have been published or are to be submitted for publication.

1. Talore DG, Hassen A, Tesfamariam EH (2015) Influences of land-use types on soil organic carbon, total nitrogen and related soil properties in semi-arid area, Pretoria (submitted to Journal of Plant, Soil and Environment)
2. Talore DG, Tesfamariam EH, Hassen A, Du Toit JCO, Klumpp K, Soussana JF (2015) Long-term impacts of stocking rate on soil carbon sequestration and related soil properties in the arid Eastern Cape, South Africa. DOI:10.1002/jsfa.7302 (published in Journal of the Science of Food and Agriculture)
3. Talore DG, Tesfamariam EH, Hassen A, Du Toit JCO, Klumpp K, Soussana JF (2015) Long-term impacts of season of grazing on soil carbon sequestration and related soil properties in the arid Eastern Cape, South Africa. DOI:10.1007/s11104-015-2625-z (published in Plant and Soil)
4. Talore DG, Hassen A, Tesfamariam EH (2015) Yield and nutritive quality of dominant forage species in response to simulated drought in subtropical grassland (to be submitted to Pakistan Journal of Agricultural Sciences)
5. Talore DG, Hassen A, Tesfamariam EH (2015) Soil water content, herbage yield and RUE of subtropical grassland subjected to different levels of precipitation and defoliation interval (to be submitted to Applied Ecology and Environmental Research)

Conference presentations

1. Talore DG, Hassen A, Tesfamariam EH, Du Toit JCO, Katja K, Soussana JF (2013) Long-term impacts of grazing season on soil carbon sequestration in arid areas of South Africa. In: Advances in Animal Biosciences Proceedings of the 5th Greenhouse Gases and Animal Agriculture (GGAA 2013), Dublin, Ireland, 23-26 June 2013, p 431
2. Talore DG, Tesfamariam EH, Hassen A, Soussana JF (2013) Long-term impacts of stocking rate on soil carbon sequestration in arid areas of South Africa. In: David L Michalk, Geoffrey D Millar, Warwick B Badgery, Kim M Broadfoot (eds.). Revitalizing grasslands to sustain our communities, Proceedings of 22nd International Grassland Congress 15–19 September 2013, Sydney, Australia, pp 1237–1238

3. Gemiyo D, Hassen A and Tesfamariam EH (2014) Effects of simulated drought on species composition and biomass yield of semi-arid grassland, South Africa. Livestock, Climate Change and Food Security Conference, 19–23 May 2014, Madrid, Spain
4. Talore DG, Hassen A, Tesfamariam EH (2015) Nutritive quality of dominant forage species in response to simulated drought in subtropical native pasture. Climate Smart Agriculture, Third Global Science Conference, 16–18 March 2015, Le Corum, Montpellier, France

The major objective of this PhD research was to investigate the influences of changes in precipitation, grazing (stocking rate, season of use and defoliation interval) and land-use types on carbon sequestration, herbage yield and nutritive value, and soil-water content (SWC) in the subtropical grasslands of South Africa. To achieve these aims, a series of studies and experiments were undertaken at Grootfontein Agricultural Research Institute (GADI) and the University of Pretoria. To identify research gaps, the databases were reviewed in CHAPTER 1. The influence of land-use types on carbon and nitrogen storage and related soil properties was investigated in CHAPTER 2. In CHAPTER 3, long-term impacts of grazing at contrasting stocking rates were assessed on soil carbon sequestration and related soil properties by collecting soil samples from the long-term trial site at GADI. Similarly, long-term impacts of season of grazing were investigated on soil carbon sequestration and selected soil properties in CHAPTER 4 at the same site, following a similar procedure to that described in CHAPTER 3. In chapter 5, a rainout shelter was established using non-ultraviolet acrylic sheets to investigate various levels of precipitation and defoliation interval on SWC, herbage yield and rain-use efficiency under the acrylic structure. In CHAPTER 6, dry matter (DM) yield and nutritive quality of subtropical grassland were evaluated in different precipitation situations under the acrylic structure. Finally, CHAPTER 7 presented general conclusions and critical evaluations, highlighting technical and policy implications of the results of this study with recommendations for future research. The citations in the text and reference section in this dissertation were prepared in accordance with the guidelines set up for authors for publication of manuscripts in Plant and Soil.

Acknowledgements

Several people and institutions have contributed to the successful completion of this study.

- I would like to express my sincere gratitude to my supervisors, Dr Abubeker Hassen and Dr Eyob Habte Tesfamariam, for their invaluable input, professional guidance and dedication. I wish to express my deep gratitude to them for their follow-up and encouragement throughout my study period. I am also grateful to my supervisor, Dr Abubeker Hassen, for giving me the opportunity to participate in international conferences on two important occasions.
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- Last, my beloved wife, Titina Mathewos, and my baby boy, Tikikil Deribe. Thank you for your patience, support, and uninterruptible love and prayer.

Great [is] the LORD, greatly to be praised; and his greatness [is] unsearchable (Psalm 145:3)

In loving memory of my father,

Dedicated to

My mother, Hezete Ersumo,

My beloved wife, Titina Mathewos, and

My baby boy, Tikikil (Wushu) Deribe

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List of acronyms and initialisms

ADF: acid detergent fibre

ADL: acid detergent lignin

ANPP: aboveground net primary productivity

AOAC: Association of Official Analytical Chemists

b: insoluble but slowly fermentable fraction of fibre

BD: bulk density

c: rate of fermentation of b

C: carbon

Ca: calcium

CEC: cation exchange capacity

C : N: carbon to nitrogen ratio

CP: crude protein

DM: dry matter

DI: defoliation interval

EC: electrical conductivity

ET: evapo-transpiration

GADI: Grootfontein Agricultural Research Institute

g: gram

GHG: greenhouse gases

GLM: general linear model

GP: gas production

IVOMD: *in vitro* organic matter digestibility

K: potassium

kg: kilogram

Mg: magnesium

mL: millilitre

N : nitrogen

Na: sodium

NH_4-N: ammonium nitrogen

NH4AOC: ammonium carbonate

NLIN: Non-linear regression model

NO_3-N: nitrate nitrogen

NP: neutron probe

NDF: neutral detergent fibre

OM: organic matter

P: phosphorus

PET: precipitation

PD: potential degradability

RI: rain interception

RUE: rain-use efficiency

SAS: Statistical Analyses Systems

SEM: standard error mean

SOC: soil organic carbon

SOM: soil organic matter

SWC: soil-water content

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Summary

Rainfall variability in and between seasons, lower soil organic matter content, land degradation, and the associated low biomass yield and poor quality of forage are major constraints to livestock production in the subtropical grasslands of South Africa. In this study, the influences of changes in precipitation, grazing management and other land-use types on carbon (C) and nitrogen (N) sequestration potential, herbage yield and nutritive quality were investigated by collecting data from i) the long-term grazing trial site at Grootfontein Agricultural Development Institute (GADI), an arid grassland in Middleburg, Eastern Cape; ii) various land-use types in a semi-arid area, Pretoria; and iii) short-term experimental plots set up in semi-arid grassland in Pretoria, South Africa.

In the first study, various land-use types (*Leucaena* sp. plots, exclosures, cultivated pasture and croplands) were compared for soil carbon (C) and nitrogen (N) storage. *Leucaena* spp. plots and exclosured lands improved C and N stocks, due mainly to high biomass returns to the soil, while frequently cultivated pasture and croplands had generally lower C and N stocks. Although soil bulk density and soil chemical parameters such as C, N, calcium (Ca), potassium (K), magnesium (Mg), sodium (Na), and pH were affected by land-use type, it was only the C and N stocks that showed land-use by soil depth interaction. This indicates that C and N are more

sensitive to frequent soil disturbances and cultivation, compared with other soil properties, which were less affected by soil depth.

In the second study, the long-term impacts of grazing management (namely stocking rate and season of use) on selected soil properties were investigated by collecting soil samples from the long-term grazing experimental site at GADI, Middelburg, Eastern Cape, South Africa. *Aristida congesta*, *A. diffusa* and *Eragrostis lehmanniana* were dominant grass species while *Pymaspermum parvifolium*, *Felicia murcata*, *Salsola Caluna*, *Walafrida geniculata*, *Diospyros austroafricana, Diospyros lycioides* were dominant shrub species in the study area (Du Toit et al. 2011). Findings from this study indicated that in this arid region of the country, grazing at contrasting stocking rates (light , 0.78 small stock unt $ha^{-1}$and heavy, 1.18 small stock unit ha^{-1}) resulted in lower soil organic C and total N concentration compared with the non-grazed control (exclosure). Similarly, both spring and summer grazing resulted in reduced soil C and N stocks, lower water infiltration rate, and higher soil compaction compared with the non-grazed control, due mainly to plant removal from animals feeding on the forage, and animal treading and trampling. Generally, animal exclusion improved C and N stocks and, among others, resulted in higher water infiltration and lower soil compaction. Improved land and livestock management through the establishment of multipurpose tree species and strategic animal exclusion would improve C and N storage in vast potentially restorable grazing areas of South Africa.

The last experiment was designed to study the effects of different levels of precipitation (simulated drought) and defoliation intervals on soil-water content (SWC), herbage yield and quality. The study was conducted for two years at Hatfield Experimental Farm, University of Pretoria, using rainout shelters to intercept 0%, 15%, 30% and 60% of ambient rainfall as the main plot treatment and two defoliation intervals (45-day and 60-day) as sub-plot treatments. Both rain interception (RI) treatments and defoliation intervals influenced SWC, herbage yield and quality. Rainfall amount and distribution varied within and between years. This variation was confirmed by the negative climatic water balance (precipitation minus evapotranspiration, PET-ET) in most months during the study period, suggesting that the trial site was moisture stressed mainly during the active plant-growing period owing to high atmospheric water demand. In 2013/14, SWC was reduced in the 60% RI treatment, while there were no differences between

RI treatments in 2014/15. Generally, SWC of 40–60 cm was higher, compared with the top (0–20 cm) soil layer, mainly because of high surface evapotranspiration and water uptake by plants in top soil layers. The positive linear relationship between herbage yield and precipitation implies that the amount of precipitation influences grassland biomass production. In this study, the DM yield of dominant species showed differences within seasons as well as between seasons. Generally, grass species contributed a significantly (67%) higher proportion of the total plant biomass and maintained community stability in the sub-climax to climax stages. Forbs and shrubs also played a major role by occupying early niches and protecting soil from wind and water erosion across the RI plots. Thus, maintaining important forbs and shrubs is vital for the revitalization of grassland to restore biodiversity. The simulated severe drought of 60% RI caused detrimental herbage yield decline. However, it resulted in improvements in rain-use efficiency (RUE) and some feed quality attributes (namely crude protein (CP) and *in vitro* organic matter digestibility (IVOMD). Mid-season (January and February) harvests showed high herbage yield, and the recorded yield was proportional to the amount of rainfall. The study also showed that in dry events, when the rainfall of the site decreased by 30% and 60% of the ambient rainfall, defoliating plants at longer intervals (60-day) improved grassland biomass production. The variations found between and within species in yield and nutritive value under different precipitation and defoliation conditions would possibly help to identify adaptable species in future climate change scenarios and management conditions in moisture-stressed areas of South Africa.

GENERAL INTRODUCTION

Livestock agriculture in sub-Saharan Africa is based primarily on grasslands. Valuable products associated with grasslands include sources of forage for domestic and wild animals, habitats for a wide range of animal and plant species, better C sequestration potential than cultivated lands, places for recreation, and a way of life for more than 800 million rangeland-dependent rural communities (Steinfeld et al. 2006; Briske et al. 2008; FAO 2010). The grassland biome is one of the major biomes in South Africa, which supports over 150,000 head of cattle and 1.5 million sheep, and is the foundation of the country's dairy, beef and wool industries (Kotzé et al. 2013; Meissner et al. 2013). It is a centre of diversity, adjoining and extending into most of the major biomes (Nama-Karoo, savanna biome, etc). Almost 80% of the land available for agriculture in South Africa is predominantly arid and semi-arid, characterized by low and unpredictable rainfall patterns. In these arid zones, more than 87% of the land is used for livestock production and game farming (Smet & Ward 2006). This signifies that the animal production sector is mainly natural resource dependent, with veld (rangeland) resources as its main source of fodder.

Land and livestock management practices affect belowground plant nutrients (Kotzé et al. 2013; Wang et al. 2014a) as well as aboveground net primary productivity (ANPP) and forage quality (Briske et al. 2008; Wang et al. 2014a). Livestock grazing management, especially effects of stocking rates on rangeland condition and animal performance, has been widely investigated and documented for the Karoo and grassland biomes of South Africa (Van den Berg & Du Toit 2011; Du Preez et al. 2011a). However, land use and grazing management effects on soil nutrients have not been given adequate attention (Evans et al. 2012; Kotzé et al. 2013). Land-use type is one of the key drivers of C and N dynamics in the ecosystem (Watson et al. 2000). Many studies have shown that land-use changes from arable lands to grassland and vice versa had significant positive or negative impacts on soil organic matter content, thereby affecting soil organic carbon (SOC) and N (Conant et al. 2001; Celik 2005). According to Chen et al. (2011), human alteration of natural ecosystems to agriculture is an important cause of disturbance and may alter plant community cover and the function of the ecosystem. However, the exact mechanisms associated with the impact of land use on C and N dynamics in the ecosystem are poorly understood (Soussana et al. 2004; Lal 2008). According to Milchunas et al. (1988), vegetation and livestock integration is important for sustainable ecosystem services and functions (Figure 1). Removal of biomass by grazing, the trampling of soil and grass, and the inputs of nutrients through defeacation are some of the impacts of livestock on physical environment (Reid et al. 2010).

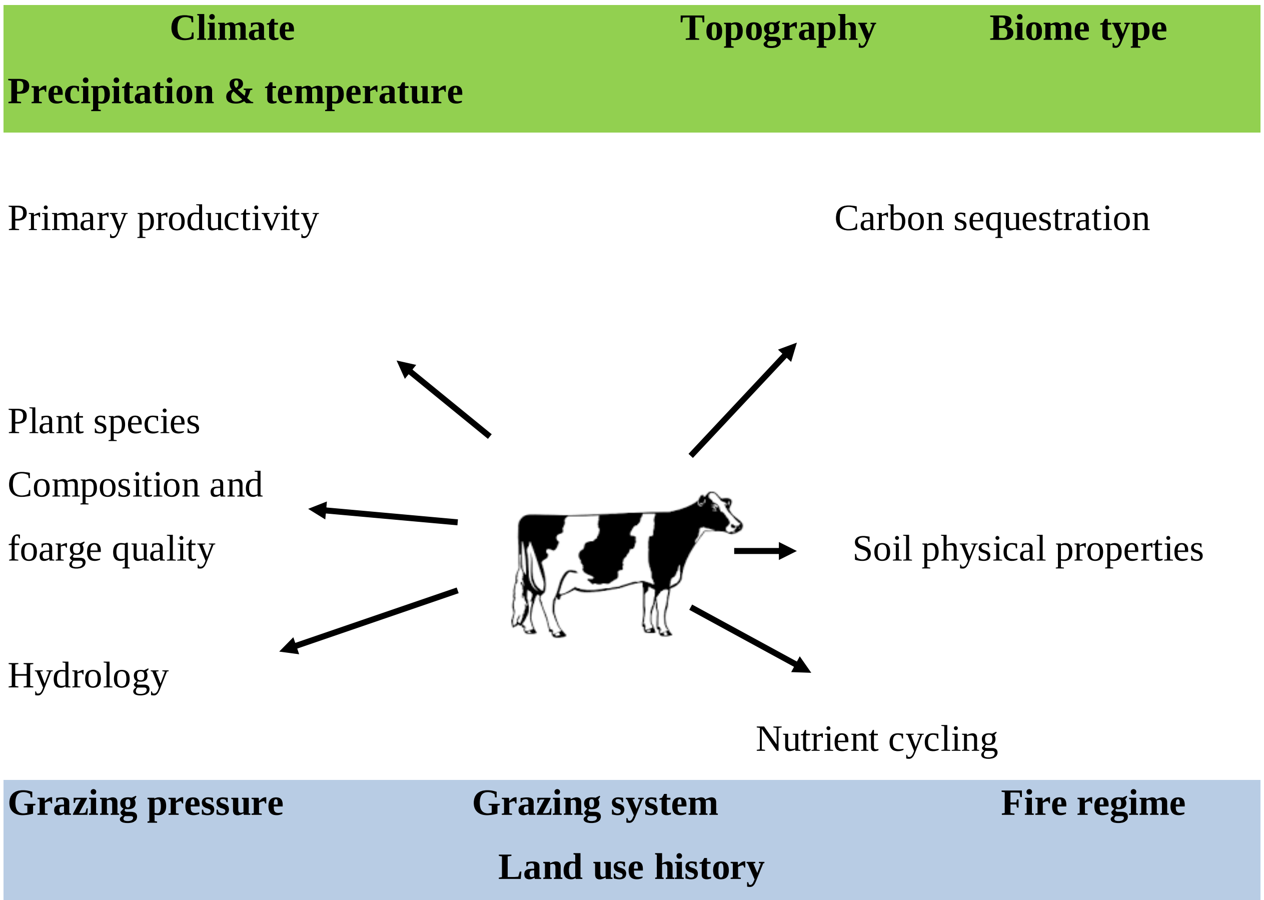


Figure 1 Relationships and/or influences of grazers on ecosystem processes and properties of grasslands. Boxes show natural (green box) and anthropologic (blue box) variables that affect how grazing alters ecological processes in rangelands (Adapted from Josephine Smit, **Source**: https://uchunguziblog.files.wordpress.com/2016/01/rangeland-science.pdf)

Soil organic carbon (SOC) plays a major role in maintaining a balanced global carbon cycle (Lal 2004), and provides ecosystem benefits by linking ecosystem processes and functions. Soil organic C has various benefits: i) it improves soil physical properties by enhancing soil structural stability, erosion resistance, water-holding capacity, and aeration (Holm et al. 2003; Du Preez et al. 2011b); ii) it improves soil chemistry, such as enhanced availability of micronutrients, nutrient cycling, soil buffer capacity and nutrient supply (Scholes et al. 2003; Lal 2009; Du Preez et al. 2011b); and iii) biologically, it stimulates activity and diversity (enhanced faunal activity) of microorganisms in soil (FAO 1995; Moussa et al. 2007; Kotzé et al. 2013). According to Lal (2004, 2008), soil C and N sequestration has climate-change mitigation potential by restoring degraded soils and increasing productivity of grassland ecosystem. High C and N stocks are usually needed to maintain consistent yields through improvements of plant nutrients, water- and nutrient-holding capacity and soil structure (Lal 2004). However, information about soil organic matter (SOM) and the associated C and N storage across a range of environmental and grazing management conditions is not consistent (Lal 2004; Moussa et al. 2007; Steffens et al. 2008; Du Preez et al. 2011a; Witt et al. 2011). Usually, the differences observed in soil C and N within and across landscapes are due to variations in climate, soil type and landscape management (Lal 2002), and type of plant community (Ampleman et al. 2014). Recently, the carbon and water footprint of livestock has raised further questions of environmental impacts (Steinfeld et al. 2006; Scholtz et al. 2013b). Several reports have been published on the effects of grazing on vegetation and biomass production (Hoffman & Ashwel 2003; Du Toit et al. 2011; Wang et al. 2014a). However, assessing vegetation and species composition once in their lifecycle gives only a snapshot of the current situation. Soil indicators (soil C and N stocks) are more reliable for determining long-term impacts of rainfall and grazing management practices (Evans et al. 2012; Kotzé et al. 2013). Nonetheless, soil properties have largely been overlooked in most grassland monitoring programmes (Evans et al. 2012; Kotzé et al. 2013). In particular, information about soil C and N sequestration under different management systems is limited, especially reports on the long-term effects of grazing.

With regard to aboveground net primary productivity (ANPP) and herbage production, there is considerable knowledge about the ecology, ecosystem functioning and productivity of grasslands in South Africa. Yet, most of the researchers have focused on management strategies aimed at maintaining palatable grasses. Species differ in traits that influence their performance under various environmental conditions (Wright & Westobey 2002; Hatier et al. 2014). Concurrently, the species-specific characteristics influence the physical and biological environment (Díaz et al. 2006; Hooper et al. 2005). To a large extent, this has excluded the broader importance of diversity in grassland biomes, drought tolerance of species, and their level of extinction (Low &

Rebelo 1996; Díaz et al. 2006; Hooper et al. 2005) owing to current and predicted future climate change. Vegetative studies historically based on spot sampling are often used to assess the relative importance of species during the growing season (Abule et al. 2007a; Matias et al. 2012). However, such studies have their own drawbacks, as they do not account for the diversity and drought resistance of species that occupy early season niches from seed banks (Tessema et al. 2012). Recurrent drought and rainfall variability may influence the resilience capacity of some plant species; and it may be difficult to fully capture responses of different species and the associated yield and quality to shifts to inter and intra-annual variations of rainfall.

Shortage of rainfall influences SWC, which determines herbage yield particularly in arid lands, which account for 65–75% of the South African land surface (Snyman 1999; Smet & Ward 2006). Understanding SWC under changing precipitation patterns and seasonality is important (Snyder & Tartowski 2006; Swemmer et al. 2007; Yao et al. 2013). Little is known about the potential consequences of changes in precipitation within a growing season, which may affect vegetation structure and function (Knapp et al. 2006; Craine et al. 2012; Yao et al. 2013). The timing of short-duration grazing and defoliation intervals may exacerbate the impact of precipitation changes, possibly due to its influence on water infiltration rate, SWC and herbage yield. It is therefore important to understand the responses of native plant species to the combined effects of drought and defoliation stresses when considering mechanisms of persistence in important forage species (Boschma et al. 2003; Swemmer et al. 2007).

It is thus vital to improve understanding of the impacts of drought and livestock grazing and defoliation intervals on herbage yield and quality, and soil properties in an effort to develop and implement sound management practices aimed at maintaining grassland biodiversity, improving soil ecological functions, and sustaining livestock productivity (Lal 2000, 2004; Moussa et al. 2007; Khumalo et al. 2012). Furthermore, the variations in C and N concentration across locations require better understanding of the relationships of these soil properties in the context of rangeland health and productivity and sustainable livestock production under the prevailing environmental conditions. In this research, long-term studies and short-duration experiments were undertaken to improve the understanding of potential effects of changes in land-use types, precipitation and grazing/defoliation interval on herbage yield, soil moisture content and nutritive quality of dominant species. It was also essential to expand the limited database on soil

C and N sequestration that is associated with different management practices and might be used as an indicator for land rehabilitation and vegetation restoration for optimizing livestock production in arid lands of South Africa.

This research was conducted with these overall objectives: to investigate species composition and yield, nutritive value and soil moisture under different precipitation and defoliation intervals; and to improve understanding of the long-term impacts of grazing and/or other land-use types on soil carbon sequestration and related soil properties in arid and semi-arid areas of South Africa.

CHAPTER 1

1. LITERATURE REVIEW

1.1. Overview of grassland ecosystem in tropical Africa

Grasslands represent about 26% and 70% of world land and agricultural areas, respectively. Grasslands also contain about 20% of the world's soil C and N stocks (Lal 2004; FAO 2009). The grassland and Karoo biomes cover more than 50% of the land surface in South Africa (Smet & Ward 2006). These ecosystems are important for maintaining biological diversity, supporting livestock productivity, and providing key resources for native and domestic ungulate grazers in arid lands of the country (Williams 1996; Steinfeld et al. 2006). Milk, beef, wool, mohair and other valuable products are usually associated with grasslands, and are a way of life for more than 800 million grassland-dependent rural communities (Steinfeld et al. 2006; Briske et al. 2008; FAO 2010). Parts of South Africa are located in a subtropical region, with temperatures modified by altitude. The interior, where the bulk of grasslands are found, is semi-arid to arid, with rainfall decreasing westwards. Apart from soil formation differences, the interface between grasslands and other biomes in South Africa contributes substantially to their floristic and faunal diversity and to the important role they play in the agricultural economy (Hoffman & Ashwell 2001).

1.2. Significances of grasslands in tropical Africa

1.2.1. Food security

Significant portions of world milk (27%) and beef (23%) production occur on grasslands managed solely for these products (Conant et al. 2001). Occupying about 87% of the land surface, the grazing industry has been an important contributor to the overall economic growth of South Africa. Many livestock products (beef, mutton, fleece and hides) are derived directly from grasslands (Schulze et al. 1997; Du Preez et al. 2011a), contributing substantially to the livelihoods of farmers, ranchers and commercial farm owners (Tainton & Hardy 1999; Kotzé et al. 2013).

Land-based livelihood strategies such as livestock farming, crop production, and harvesting of wild natural resources play important roles in rural society in the communal areas of South

Africa (Du Preez et al. 2011a). When reliable growing days drop below the number that are necessary for maize production in eastern and southern Africa, grassland ecosystems become more and more important sources of feed for livestock (Thornton & Jones 2009). But the challenge is that 65% of grasslands of South Africa are affected by desertification (UNEP 1997; Hoffman & Ashwell 2001), and are vulnerable to climate change (FAO 2010). In countries such as South Africa, with such a high level of aridity and environmental fragility, the consequences of desertification threaten the livelihoods of pastoral and agro-pastoral communities (Hoffman & Ashwell 2001; FAO 2006; Steinfeld et al. 2006), who have based their lives on grasslands. The implication is that revitalizing grasslands is vital to improve the livelihoods of the poorest communities of the continent.

1.2.2. Biodiversity

Diversity comprises a broad spectrum of biotic scales and can generally be described as the number of entities, the evenness of their distribution, the differences in functional traits, and their interactions (Diaz et al. 2006). Besides the numbers of species, the numbers of functional groups are important in biodiversity. Species diversity, the number and composition of plant species present at a site, is the most frequently considered aspect of biodiversity. Increased species composition has been shown to increase primary productivity, improve plant allocation patterns (Diaz et al. 2006; Hatier et al. 2014), and reduce invasibility by unsown species that change herbage composition (Kirwan et al. 2007). For most plant species, biodiversity declines with increasing aridity (Wang et al. 2014b).

Diverse plant species in grasslands support a long-term stable ecosystem because they exhibit complementary functionality. Species diversity has two basic components: richness or number of species in a given area, and evenness or how relative abundance or biomass is distributed among species. Increased species richness reduces the vulnerability of grassland ecosystems to drought (FAO 2007; Wang et al. 2014b), increases the carbon source strength (Diaz et al. 2006), and may increase N fixation and protect against nitrogen leaching (Oenema et al. 2005). South Africa's grasslands are rich in species diversity, comprising an estimated 3788 plant species (Gibbs Russell 1987). However, losses of biodiversity have been reported due to shifts in species composition in response to heavy grazing (Moussa et al. 2007; Khumalo et al. 2012). Changes in growing season rainfall have been reported to be associated with declining richness in grass species (Wilkes 2008). This perturbation in the long-term droughts most notably affected the dominant species, which play a significant role in maintaining community structure, and are replaced with less palatable species and ruderals (Schuman et al. 1999; Du Toit et al. 2008; Evans et al. 2011). The implication is that any influences that affect the dominant species have large effects on the biodiversity of grassland ecosystems (Olff & Ritchie 1998; Evans et al. 2011; Khumalo et al. 2012).

1.2.3. Sources and sinks of greenhouse gases

Grasslands face a wide range of challenges from climate change, including the effects of elevated atmospheric carbon dioxide, higher temperatures, changes in the precipitation regime, and increasing concentrations of ground-level ozone (FAO 2007, 2010; IPCC 2007). In spite of notable impacts of climate change on grasslands (Schulze 1997), properly managed soils can store relatively stable and significant proportion of C. The global C stock in grasslands is estimated at 343 Gt C, which is about 50% more than the amount stored in forests globally (Lal 2004). Food and Agricuture Organization of the United Nations, FAO (2010) estimated that the SOC sequestration potential of the world's grasslands is 0.01-0.3 Giga tone (Gt) C yr^{-1}. This amount is 3 times more than the size of atmosphere (770 Pg) and 3.8 times more than the size of biotic pools (610 Pg) (Lal 2000, 2002). Permanent pastures could offset up to 4% of global greenhouse gas (GHG) emissions through carbon sequestration (Soussana et al. 2004), whereas under poor management they become GHG sources (Wilson et al. 2012).

Climate change, along with poor grazing management, is expected to reinforce GHG effects (Karl et al. 2009; Fay et al. 2011), resulting in a decrease of C and N stocks (Wu et al. 2010; Kotzé et al. 2013). However, grasses are generally more resilient to perturbations and regain their functioning faster after severe drought (FAO 2009; Hatier et al. 2014). On the other hand, biodiversity is expected to increase the resilience and productivity of grasslands (Hooper et al. 2005), and hence may play a role in GHG mitigation (Diaz et al. 2006). Of the total global mitigation potential of 5.5-6 Gt CO_2 equivalent yr, almost 1.5 Gt is related to grazing land management and pasture improvement (Smith et al. 2009), while land use and land-use changes contribute 2.5 Gt CO_2 equivalent yr to GHG emissions (Conant et al. 2001).

Grasslands are also sources of GHG, including methane (which has global warming potential of 25 times that of CO_2) by ruminants, and ammonia and nitrous oxide through ammonia volatilization and denitrification (Oenema et al. 2005). Grazers return significant proportions of ingested N (40–95%) to grazing lands through their faeces and urine (Oenema et al. 2005). Ammonia losses increase at higher soil pH and conditions that favour evaporation. However, in most cases, faeces and urine may localize to water points and animal entry areas (Du Toit et al. 2008). From surface N deposition of 5.0–7.4 kg ha^{-1} yr^{-1} (Scholes et al. 2003), a significant proportion of N could be lost through ammonia volatilization and denitrification, which gives nitrous oxide, another potent GHG that contributes to global warming.

1.3. Drivers of grassland ecosystem

The dynamics of grasslands in arid lands depend on two main factors, that is, variations in rainfall and the level of grazing intensity and frequency (Du Toit et al. 2011). Arid land ecosystems are thought to conform to persistent non-equilibrium models in which rainfall variability is the primary driver of vegetation dynamics. In these lands, marked changes in community composition due to drought can be exacerbated by grazing (O’Connor & Bredenkamp 1997; Briske et al. 2003). On the other hand, in the vegetation dynamics model, the importance of grazing is emphasized as a feedback mechanism that influences the competitive interactions of plants and is a key driver of ecosystem dynamics (Ritchie & Olff 1999; Tainton & Hardy 1999; Briske et al. 2003). A threshold model could be applied when a disturbance pushes a system across the brink into a new stable state, which persists after the disturbance has been removed (Briske et al. 2003). In South Africa, examples of threshold models have been found in tall grasslands dominated by *Hyparrhenia hirta*, in which the primary palatable species, *Themeda triandra*, did not return after the removal of grazing, providing evidence that a threshold had been crossed (O’Connor & Bredenkamp 1997). Thus, on localized spatial scales, extinctions may arise from small perturbations, but stability may still exist at the broader community level (Illius & O’Connor 1999).

1.3.1. Impacts of precipitation on grassland ecosystem

i. Species composition

Seasonal water limitation due to both intra- and inter-annual variability in rainfall drives the patterns of diversity and species composition in grass ecosystems (Hoekstra et al. 2005; Knapp et al. 2006; Derner et al. 2011). Research reports on adaptation or sensitivity of native and introduced plant species to experimental manipulation of rainfall (Yahdjian & Sala 2002; Heilser-White et al. 2008; Fay et al. 2011) indicated that responses of different species to moisture stress varied considerably. This could be due to the complex interactions between temperature, soil water availability, and the temperature and moisture sensitivities of key plant processes (Fay et al. 2011).

Altered rainfall timing increases rainfall variability, resulting in a reduction in rates of ecosystem processes (Dai 2011; Du Preez et al. 2011a; Fay et al. 2011) and losses of biodiversity (Khumalo et al. 2012). Early spring green-up and flowering (Cleland et al. 2006) are generally the most commonly found responses of species to warming (Rustad et al. 2001). However, varying responses have been reported in grasslands (Zhou et al. 2006; Almagro et al. 2009; Liu et al. 2009). Certain species characterize different succession stages during grassland retrogression and can be used as indicators of rangeland condition (Malan & Van Niekerk 2005). Variability in precipitation and increases in temperature could lead to changes in ecosystem and biome function such as woody species encroachment and alien species invasion (Scholtz et al. 2013b).

ii. Aboveground net primary productivity

Aboveground net primary productivity (ANPP) in many terrestrial biomes is affected by water availability, which is the major driver of ecosystem dynamics that would be strongly altered by future climate change (Altesor et al. 2006; Heilser-White et al. 2008). On a regional to continental scale, plant production and decomposition rates increase with a rise in mean annual precipitation (Epstein et al. 2002), leading eventually to changes in grassland biodiversity (DEAT 2007; Khumalo et al. 2012). Aboveground net primary productivity is usually lower in ecosystems limited by soil moisture, as a result of low rainfall and high evaporation (Moussa et al. 2007; Du Preez et al. 2011a). Seasonal water limitation due to intra- and inter-annual variability in rainfall influences the patterns of diversity (Hoekstra et al. 2005; Knapp et al. 2006; Derner et al. 2011), which ultimately determine ANPP.

Primary production in low precipitation ecosystems is controlled mainly by rainfall inputs, as evidenced by strong correlations ($r^2 \geq 90$) between ANPP and annual precipitation in space on a regional and global scale (Sala et al. 1988; Jobbagy & Jackson 2000; Sala & Austin 2000). Several studies have shown twofold or more inter-annual variation in ANPP to inter-annual variability in precipitation (Briggs & Knapp 1995; Knapp & Smith 2001; Smith & Knapp 2003). Many studies, observational and experimental manipulations, suggest that ANPP is sensitive to the timing and size of precipitation inputs (Fay et al. 2003; Heisler-White et al. 2009). Berdanier and Julia (2011) observed consistent higher responses of ANPP to soil moisture across space, time and vegetative communities. In contrast, few reports showed that ANPP is less sensitive to altered rainfall amount and longer dry spells (Fay et al. 2011).

On more local scales, however, a considerable amount of variability in ANPP remains unexplained by annual precipitation alone (Knapp & Smith 2001). This variability in ANPP (increase, decrease and no response) in response to warming in grassland areas is likely to reflect the complex interactions among rainfall, temperature and soil water availability with plant physiological processes (Fay et al. 2011). Water stress retards plant development through reductions in photosynthesis, and this could occur at one stage of development rather than another. According to Flombaum and Sala (2007), ANPP in arid lands ranged between 40 g m^{-2} yr^{-1} and 152 g m^{-2} yr^{-1} from several harvests during the plant-growing season. However, there are great variations in the amount of ANPP between ecological areas, depending on drought severity, temperature, soil type, and land management practices (Berdanier & Julia 2011; Fay et al. 2011).

iii. Forage quality

Forage quality determines the amount of nutrients that herbivores can acquire from ingested forage, and varies depending on plant characteristics (Briske et al. 2008). Tissue age, tissue type, functional plant group, and anti-quality agents (for example lignin, cellulose, and secondary compounds) represent more widely recognized characteristics (Van Soest 1982; Evans et al.

2011) that affect feed quality. Feed quality is also affected by the ratio of soluble to structural cellular components, which decreases as mean tissue age increases with advances in the growing season (Briske et al. 2008).

Most grazing lands in the developing world are not well managed, and thus are low in feed quality (Briske et al. 2008; Wilson et al. 2012). The reasons include erratic rainfall and higher temperatures, which speed up the plant lignification process (Küchenmeister et al. 2013). However, there are great spatial and temporal variations in the quality of grassland species based on temperature, precipitation, soil type and maturity level (Evans et al. 2011). Forage quality is generally highest in spring and declines in summer (Liebig et al. 2014). High summer temperatures increase the rate of plant maturation and cell wall lignification, and result in lower forage quality, which is characterized by reduced CP content and digestibility (Marais 2005; Küchenmeister et al. 2013). This may happen because of long days, which can increase the stem number per plant, diameter of the stem, and internode length, thereby reducing the leaf-to-stem ratio (Buxton 1995; Liebig et al. 2014). The effects of short-term drought, however, have been reported to be less evident on feed quality (Marais 2005; Knapp et al. 2006; Küchenmeister et al. 2013) compared with the differences associated with inherent plant species characteristics and differences in functional groups.

iv. Soil carbon and nitrogen

The amount and distribution of rainfall indirectly affect soil C and N cycling across various climates (Fay et al. 2011) through their influence on aboveground and belowground productivity (Wang et al. 2014a). Wu et al. (2010) reported that altering precipitation regimes leads to chronic resource alterations (Smith et al. 2009), threshold exceedance, and disturbances (Briske et al. 2006; Knapp et al. 2008). The amount of C sequestered in SOM can enhance rainfall effectiveness through increased water-holding capacity and water-source replenishment to better withstand times of drought. These could be achieved by increased net primary productivity, which covers soil and facilitates the water infiltration rate (Abule et al. 2007b; Wang et al. 2014a).

Changes in SOC can occur in response to a wide range of management and environmental factors (Schuman et al. 2002; Briske et al. 2008) with a net loss of soil carbon pools through disturbances (Li et al. 2007). In arid lands, C and N stocks are expected to be low because rainfall limits biomass production (Du Preez et al. 2011a). In addition, lower rainfall, accompanied by higher decomposition rates due to increased temperature, could contribute to losses of C through oxidation and N through ammonia volatilization (Kotzé et al. 2013). On the other hand, under heavy rainfall conditions, soil C and N and other plant nutrients could be exposed to erosion and leaching to deeper soil profiles, depending on soil texture, and thus affects the distribution of soil nutrients in plant-soil system (Schuman et al. 1999). Temperature-dependent processes affecting nutrient mineralization and plant growth lead to seasonal dynamics in soil chemical properties (Liebig et al. 2006). Nitrification over the growing season would probably lead to a drawdown of soil NH_4-N in the grazing lands with a corresponding peak in soil NO_3-N in the summer season (Liebig et al. 2014).

1.3.2. Impacts of grazing management on grassland ecosystem

Grazing management systems (for example stocking rate and grazing season), through their effects on plant species diversity and the relative abundance of plant growth forms or functional groups influence grassland productivity and sustainable animal production. This in turn might influence grassland community stability and ecosystem function (Hickman et al. 2014). Stocking rates should fulfil both economic and ecological benefits from the rangeland, while injudicious and unrealistic stocking rates would ultimately have adverse effects on long-term productivity of the rangeland and on animal productivity. Livestock grazing at various stocking rate and grazing season as well as defoliation frequency have effects (positive or negative) on plant species composition, aboveground primary productivity, forage quality, and soil properties.

I. Species composition

Historically, grazing ecosystems in all continents have been severely impacted by human land use in the forms of overstocking, over-grazing and utilization of resources beyond the carrying capacity of the area or season. The types and numbers of animals utilizing the rangeland, among others as a management tool, are important aspects of livestock production in the natural rangeland of South Africa (Hardy et al. 1999; Du Toit et al. 2011). Grazing modifies species richness and vegetation composition, structure and function of ecosystems (Altesor et al. 2006; Yahdjian & Sala 2006; Sebastia et al. 2008).

Livestock grazing at a heavy stocking rate has often resulted in overgrazing, and consequently in vegetation degradation (Fynn & O'Conner 2000; Conant & Paustian 2002; Pellant et al. 2005; Meissener et al. 2013). Overgrazing increases the proportion of pioneers, creeping and annual grasses, which may result in bush encroachment in communal grazing lands (Solomon et al. 2007; Kotzé et al. 2013) because a land resting period is vital for herbaceous species to regenerate and for soils to be restored (Moussa et al. 2007). According to Du Toit et al. (2011), a high stocking rate (over 16 small stock units (SSU) ha^{-1}) has resulted in the occurrence of unpalatable plant species. After unpalatable and grazing resistant species have dominated a grazing area, it is difficult to retrogress to a state in which palatable species dominate, even when grazing is removed (Hardy et al. 1999). The influence of herbivores is induced primarily because of their effect on the competitive abilities of the plants within a grassland community, favouring certain species and reducing the competitiveness of others (Tainton & Hardy 1999; Altesor et al. 2006). The long-term effects of grazing pressure change plant community (Sisay & Baars 2002). Frequent removal of highly desirable species due to continuous grazing pressure leads to grassland species deterioration (Malan & Van Niekerk 2005; Kotzé et al. 2013), and reduces competitive relationships among species (Abule et al. 2005; Malan & Van Niekerk 2005).

Grazers also affect rangeland plants through selective grazing, which ultimately affects species composition. For instance, the impact of cattle grazing on sourveld is less pronounced than that of sheep, as cattle graze more evenly than sheep (Hardy et al. 1999). Generally, sheep grazing is highly selective and has been shown to result in a reduction of palatable species (Hardy et al. 1999). Higher numbers of non-graminoid species were found in grazed paddocks at low stocking rate, but the reverse was true of graminoid species. Less palatable species such as *Sporobolus africanus, S. pyramidalis*, *Eragrostis plana* and *E. curvula* increased in response to greater grazing intensity (Du Toit et al. 2011). *Senecio retrorsus*, which is known to increase in grassland that is overutilized, and *Tolpis capensis*, which is often found in disturbed places, increased in response to heavy grazing pressure, while *Themeda triandra*, *Helichrysum nudifolium* and *Vernonia natalensis* declined in response to a heavy stocking rate (Du Toit 2008, 2011). In contrast, Hickman et al. (2014) found the greatest plant diversity with grazing pressure in tall prairie.

II. Aboveground net primary productivity

Grazing management practices have considerable effects on ANPP in a wide range of climatic, biotic and abiotic environments (Snyman 2005; Wu et al. 2010; Du Toit et al. 2011; Witt et al. 2011). In general, removal of plant biomass by grazing animals reduces ANPP. Grazing may decrease ANPP by removing large portions of aboveground biomass and decrease leaf area and light interception (Wu et al. 2010; Wang et al. 2014a). Herbivores remove or breakdown standing dead biomass that shades green leaves and consume relatively old plant tissues that have low radiation use efficiency (Pineiro et al. 2010). The net direct effect of defoliation on ANPP would result from the balance of these two opposing mechanisms.

On the other hand, grazers may change ANPP indirectly by altering species composition or soil nutrients and by decreasing water availability (Pineiro et al. 2010; Wu et al. 2010). Heavy stocking rate reduces ANPP through formation of patches and ruderals under selective grazing (Abule et al. 2005; Malan & Van Niekerk 2005). Grazing at a heavy stocking rate removes photosynthetic tissue (Wang et al. 2014a) and decreases peak standing crop (PSC) of aboveground live phytomas (Schuman et al. 1999; Wang et al. 2014b). Grazers also reduce ANPP through treading and trampling in wet seasons, affecting photosynthetic capacity and the survival of rangeland plants. As a result, ANPP varies greatly, depending on grazing and defoliation management practices and moisture conditions. Aboveground net primary productivity usually ranges between 40 g m^{-2} and 80 g m^{-2} in arid lands. Wang et al. (2014b) reported that ANPP (from a peak time harvest) ranged between 64.1 g m^{-2} in grazed management conditions and 116.4 g m^{-2} in exclosure in arid areas. Under circumstances in which water and temperature are not limiting factors, ANPP increases as growing season advances (Berdainer & Julia 2000). However, an absolute increase in plant growth is seldom documented under grazing conditions. ANPP generally decreases with increasing grazing severity (Milchunas & Lauenroth 1993) because plants do not have sufficient time for recovery after defoliation (Abule et al. 2005; Wang et al. 2014b).

III. Forage quality

A long rest period or light grazing pressure (less frequent defoliation) allows plant tissues to mature and forage quality to decrease compared with more frequent grazing/defoliation intervals (Briske et al. 2008). On the other hand, heavy grazing management practices affect feed quality indirectly through their effect on species composition. For instance, heavy grazing favours annual grasses and creeping forbs, which are less preferred compared with the perennial counterparts (Arzani et al. 2006). Grazing at heavy stocking rate also results in an increase of unacceptable plant species (Du Toit et al. 2011), with higher fibre content in spite of relatively high energy and protein contents (Mbatha & Ward 2010). The availability of good quality grasses and legumes under light to moderate grazing conditions usually results in better nutritive value of forages (Abule et al. 2007a). When grasslands are properly managed, they normally have higher biomass yield with more of the acceptable species (Sisay & Baars 2002), ultimately resulting in better intake by animals due to higher rumen fermentation characteristics of the forage.

Grazing in the plant-growing period, that is, the spring and summer seasons, is found to be important to improve forage quality (Laughlin & Abella 2007). It is well documented that plant species have higher CP content in spring to mid-summer under controlled grazing conditions (Du et al. 2011; Evans et al. 2011) owing to forage re-growth in the growing season. In contrast, Mbatha and Ward (2010) reported that grazed treatments had even higher CP content in the late growing season in semi-arid environments. In the grazed treatments, this could happen because, under frequent removal of forages, plants might not fully mature until the late growing season, resulting in high CP and low cell wall contents, and thus improved digestibility.

IV. Soil properties

a) Soil carbon and nitrogen storage

Interventions such as grazing (season of grazing and stocking rate) and stock exclusion are factors that influence the speed and direction of C and N dynamics (Tainton & Hardy 1999; Briske et al. 2003). Soil organic carbon has been adopted as an indicator of soil fertility level based on the rationale that it contributes to soil physical, chemical and biological properties (Du Preez et al. 2011b; Kotzé et al. 2013) and is strongly influenced by grazing managements

(Conant et al. 2001; Moussa et al. 2007). Apart from inherent soil characteristics, grazing at a heavy stocking rate is one of the major factors reported to influence C and N concentration of soils, resulting in low C and N stocks (Du Preez et al. 2011a; Kotzé et al. 2013). Similar to other arid regions, South African soils generally have low soil C and N concentration. For instance, about 58% of South African soils contain less than 0.5% organic C, 38% of soils contain 0.5%–2% organic C and only 4% contain more than 2% organic C (Du Preez et al. 2011a). Communal grazing and frequently cultivated lands generally had lower SOC and N (Moussa et al. 2007; Du Preez et al. 2011b).

Grazing has shown to affect soil C, total nitrogen (TN), and other ecosystem functioning (Franzluebbers 2005), but the effects of grazing on C and N dynamics were not consistent. Some authors reported that grazing management practices could increase soil C (Reeder et al. 2004; Li al. 2011) and N concentrations (Liu et al. 2012), while others reported no effects (Barger et al. 2004) or a decrease in soil C (Gill 2007) and N levels (Steffens et al. 2008). Similar reports are available on the pattern of distribution of C and N across soil layers. However, it appears that grazing reduced soil C and N in the top soil layers (Witt et al. 2011; Kotzé et al. 2013). On the contrary, in exclosure, due to higher fine root production and greater root turnover, C and N storage is usually high at the top soil layers (Derner et al. 2006; Mekuria & Aynekulu 2011). This great variability is due to dissimilarities in climate, soil type and landscape management (Lal 2002), and types of plant community (Ampleman et al. 2014). Moreover, differences in functional group composition and the source and magnitude of C and N inputs contributed to this inconsistency in ecosystem C and N storage in response to grazing (Derner et al. 2006). However, other researchers (He et al. 2011; Witt et al. 2011; Wang et al. 2014b) reported that SOC storage is usually higher under light grazing and exclosure scenarios, while N storage showed great variations. Beukes and Cowling (2003), on the other hand, noted that short-frequency intensive herding could have positive effects on soil C and N. However, intensive herding reduces photosynthesis, further reducing total energy and carbon source for plant growth (Briske et al. 2008). In mixed-grass rangeland, grazing at light and heavy stocking rates did not affect C and N, but may affect the distribution of C and N among the system components (Schuman et al. 1999). However, for grasslands dominated by C_4 grasses, grazer effects on C shifted from slightly negative at light grazing to positive at moderate and heavy grazing

(McSherry & Ritchie 2013). In grassland soils, at least 60% of total C is stored between 0.2 m and 1 m depth (Jobbagy & Jackson 2000). However, as decomposition rates are higher in the top soil layer, C and N are vulnerable to losses (Lal 2004). Conant and Paustian (2002) estimated that a transition from heavy to moderate grazing could sequester 0.21, 0.09 and 0.05 Mg C ha^{-1} $year^{-1}$ in Africa, Australia/Pacific and Eurasia, respectively. The C and N concentration under free grazing at various intensities shows great variations in arid to semi-arid regions (Schuman et al. 1999; Liu et al. 2011; Dong et al. 2012; Evans et al. 2012; Wang et al. 2014b). Tables 1 and 2 summarize soil C and N concentration and sequestration rates under various management conditions.

The mechanisms of C sequestration in the soil are influenced by two major activities, aboveground litter quality and decomposition (Sun et al. 2011; Liu et al. 2012), and belowground root activity (Issac & Nair 2006). In systems with high plant diversity, there is a possibility of longer residence of C due to slower decomposition that is probably associated with different degrees of chemical resistance (Sala et al. 1998). Total nitrogen (TN) concentration follows patterns of soil C in grassland soils (Conant & Paustian 2002; Pineiro et al. 2010). The relationship of soil C in landscape indicates that N, along with soil C, is a good indicator of rangeland health (Rezeai & Gilkes 2005). The C and N concentration or stocks could vary among seasons. However, research reports are limited as to how the season of grazing affects soil C and N stocks. Mbatha and Ward (2010) reported that grazed treatments had higher N in the late wet season. Winter grazing and exclosure can increase C storage (Donkor et al. 2002; Witt et al. 2011; Kioko et al. 2012) due to better vegetation cover and less soil compaction. However, this requires a balance between C and N stocks and feed quality for livestock production. According to Evans et al. (2012), however, summer grazing increased C concentration compared with spring grazing, while N concentration was not affected.

Table 1.1 C and N concentration in grasslands under various soil depths, rainfall and grazing management, namely light stocking rate; moderate stocking rate; heavy stocking rate and exclosure conditions

Soil depth (cm)	Rainfall (mm)	Country	Location	C (g kg^{-1}) Grazing*			C (g kg^{-1}) Exclosure	N (g kg^{-1}) Grazing			N (g kg^{-1}) Exclosure	References
				LSR	MSR	HSR		LSR	MSR	HSR		
0-5	590	China	Qinghai-Tibetan	75.8	68.9	59.9	90.6	6.21	5.93	5.41	7.02	Dong et al. 2012
10-20	590	China	Qinghai-Tibetan	34.5	31.9	26.6	42.3	3.23	3.05	2.62	3.61	Dong et al. 2012
0-20	590	China	Qinghai-Tibetan	35-76	32-69	27-60		3.2-6.2	3.1-5.9	2.6-5.4		Dong et al. 2012
0-5	553	South Africa	Free State	23.4	25.8	11.5	24.3	1.93	2.18	0.92	2.06	Kotzé et al. 2013
5-10	553	South Africa	Free State	14.3	17.4	11.2	17.9	1.12	1.50	0.94	1.52	Kotzé et al. 2013
0-5	280	China	Inner Mongolia	14.62	13.69	14.27	15.09	1.66	1.64	1.78	1.61	Liu et al. 2012
0-5	373.3	China	Gansu province	-	6.7	-	8	-	0.9	-	0.85	Wang et al. 2014a
0-15	384	USA	Wyoming	35	-	36	28	31		28	23	Schuman et al. 1999
5-10	373.3	China	Gansu province	-	6.2	-	6.5	-	0.9	-	0.87	Wang et al. 2014a
10-20	280	China	Inner Mongolia	13.23	13.11	12.71	14.22	1.56	1.48	1.57	1.52	Liu et al. 2012
20-30	280	China	Inner Mongolia	11.54	11.52	11.21	13.84	1.40	1.34	1.39	1.42	Liu et al. 2012
0-80	620	China	Gansu province	43	51	60		2.5	2.7	3.2		Li et al. 2011
0-60	384	USA	Wyoming	92	-	101	88	83	-	78	77	Schuman et al. 1999
0-7.5	270	Canada	Lac Du Bois		1.90-2.40		2.09		0.17-0.20		0.18	Evans et al. 2012
15-30	270	Canada	Lac Du Bois		2.44-2.66		2.71		0.24		0.25	Evans et al. 2012

*LSR: light stocking rate; MSR: moderate stocking rate; HSR: heavy stocking rate. Stocking rate (number of animals per unit area) varies from site to site depending on amount of rainfall, soil type, and forage availability

Table 1.2 Management effects on soil organic carbon (SOC) sequestration rates in various ecosystems

Management practice/ecosystem	Country	Location	SOC sequestration	References
Grazing				
Short-grass prairie	USA	Colorado	0.12 Mg C ha^{-1} yr^{-1}	Derner et al. 1997
Northern mixed-grass prairie	USA	North Dakota	0.29 Mg C ha^{-1} yr^{-1}	Reeder & Schuman 2002
Arid to semi-arid rangelands		Various	0.02-0.12 Mg C ha^{-1} yr^{-1}	Lal 2000
Semi-arid grazing lands	Australia	Malga	0.92-1.1 $C0_2$-e $ha^{-1}yr^{-1}$	Witt et al. 2011
Semi-arid lands	World	World average	0-2 Mg C ha^{-1} yr^{-1}	Lal 2009
Temperate grassland	France	France	1-6 Mg C ha^{-1} yr^{-1}	Sousanna et al. 2004
Legume inter-seeding				
Northern mixed-grass prairie	USA	South Dakota	0.33-1.56 Mg C ha^{-1} yr^{-1}	Mortenson et al. 2005
Restoration				
Southern mixed-grass prairie	South Sudan	South Sudan	Restored soil C to 80% of native rangeland in 100 yr	Olsson & Ardö 2002
Mined lands	USA	Wyoming	400% increase over 30 years	Stahl et al. 2004
Southern mixed-grass prairie	USA	Oklahoma	No difference 0-10 cm with moderate grazing but 65% decrease with heavy grazing	Fuhlendorf et al. 2002
Citrus plantations	Spain	Eastern Spain	10 Mg C ha^{-1} yr^{-1}	Iglesias et al. 2013
Nitrogen fertilization				
Tall grass prairie	USA	Kansas	1.6 Mg C ha^{-1} yr^{-1}	Rice 2000

b) Bulk density and soil compaction

The impact of grazing management on soil physical characteristics such as bulk density and compaction has been well documented (Derner et al. 2006; Kotzé et al. 2013). BD affects porosity, and the water and aeration status of the soil, as well as root penetration and development. It is one of the important soil physical properties to be monitored on grazing lands in line with soil C and N stocks as they have inverse relationships with BD (Da Silva et al. 1997; Gifford & Roderick 2003). Bulk density and C and N stock interaction can become important to the effectiveness of grazing systems (Soussana et al. 2004). Grazing-induced compaction affects soil BD at different rates, depending on the amount of available plant residue on the ground (Rodd et al. 1999; da Silva et al. 1997). Higher BDs increase compaction and reduce water infiltration (Evans et al. 2012), which in turn reduces root growth and density. Increased BD under heavy grazing showed greater soil penetration resistance (Bouwman & Arts 2000; Donkor et al. 2002). Other reports indicated that soil BD is non-responsive to grazing effects (Soussana et al. 2004; Liebig et al. 2014). On other hand, winter grazing and livestock exclusion reduces BD in a range of environments, from arid to sub-humid (Bouwman & Arts 2000; Beukes & Cowling 2003; Altesor et al. 2006; Witt et al. 2011; Kioko et al. 2012; McSherry & Ritchie 2013). The freeze-thaw cycles contributes to reduced compaction by livestock grazing during winter (Donkor et al. 2002).

c) Soil texture

Soil texture affects water-holding capacity and nutrient retention in soils (Sala et al. 1988). In drier areas receiving less than 370 mm, annual precipitation barely penetrates the shallow layer and the major path for water loss is bare soil evaporation. Soil texture, predominantly the clay content, affects soil aggregate stability, which is a good indicator of SOM (Tan et al. 2004; Li et al. 2007). The clay content exerts a major control on the amount of organic carbon in the soil (Telles et al. 2003; Lal 2009) because C concentration is usually high in macro-aggregates of clayey soils (Six et al. 2000; Kotzé et al. 2013). Disturbances of soil texture and structure by drought and grazing decrease soil infiltration and water retention, and accelerate soil erosion (Evans et al. 2012). However, generally, soil texture is one of the soil properties that are barely affected by management.

d) Soil chemistry

Drought and livestock grazing influence soil chemistry. For instance, inorganic nutrients and pH could enrich under moisture-stressed poor grassland condition (Kotzé et al. 2013). According to Mbatha and Ward (2010) dry season grazing can increase P content by about 53% in grazed plots compared to non-grazed control, while livestock exclusion generally reduces the P content. Erosion and low nutrient inputs of plants are also possible causes of losses of plant nutrients such as K. Reports indicated that higher concentration of soil P and Na have been found around water points, probably because of supplementary feeds being given to animals (Kotzé et al. 2013).

On the other hand, grazers affect soil chemistry by returning ingested N to soil through urine and faeces (Scholes 2003; Oemema et al. 2005). Continuous grazing management systems are responsible for nutrient losses by depleting plant cover and litter input (Moussa et al. 2007). Kioko et al. (2012) reported that continuous grazing had the lowest exchangeable Ca and lower pH, while areas excluded from grazing had higher pH value, Ca, Mg and K. Similarly, Chen et al. (2012) reported higher Ca and cation exchange capacity (CEC) in exclosure. In contrast, soil pH was lower in exclosure (Pineiro et al. 2010; Kotzé et al. 2013). Frequent disruption of aggregates by animals in the wet seasons, spring and summer, may lead to soil nutrient loss (Six et al. 2000; Kotzé et al. 2013). Residual turnover and nutrient cycling in the presence of higher Ca lead to higher macro-porosity and water-holding capacity of soils in exclosure (Donkor et al. 2002; Witt et al. 2011). Generally, the magnitude and distribution of fine root mass (mainly at top soil profile) is a principal driver mediating plant nutrient cycling and biogeochemistry of grassland ecosystems (Derner et al. 2006; Witt et al. 2011).

Chemical properties of the soil showed great variations in plant growing seasons (Kioko et al. 2012; Kotzé et al. 2013). This variation in soil chemistry between seasons is due mainly to differences in soil moisture and vegetation cover. In the wet season, nutrient losses are usually high under heavy grazing and in erosion-prone environments. Under less severe grazing in winter, vegetation cover can improve, and so do plant nutrients (Enrique & Lavado 1996; Wang et al. 2014a). Spring and summer grazing may result in loss of soil nutrients through runoff, animal treading and trampling (Donkor et al. 2002; Evans et al. 2012). According to Liebig et al. (2014), pH and ammonium nitrogen (NH_3-N) were found to be higher in spring than summer in the top soil layer, while electrical conductivity was the lowest in spring under grazing conditions. Higher pH in heavily grazed areas contributes to concentrations of basic cations in spring (for example exchangeable Ca^{+2} and Mg^{+2}) (Liebig et al. 2006; Li et al. 2007). However, grazing early in the growing season usually affects soil nutrients.

e) Soil-water content and infiltration rate

Optimum SWC and better infiltration are required to maintain soil health. In dry arid regions, precipitation regimes are expected to have less frequent, but more intense rainfall events. According to Fay et al. (2011), both small but frequent and more intense but infrequent rainfall events affect SWCs. For example, root growth decreases by 20–25% with intense, but less frequent rainfall events. Soil organic matter can enhance rainfall effectiveness through increased infiltration, water-holding capacity and water retention (Lal 2000; Henry 2010) that helps plants to better withstand times of drought. These could be achieved through increased net primary productivity (Donkor et al. 2002; Abule et al. 2007a; Wang et al. 2014a).

The water infiltration rate could also be affected by livestock grazing due to soil compaction. Grazing affects water infiltration and nutrient recycling through soil trampling and soil surface sealing. Heavy grazing increases soil compaction and reduces infiltration rate that affects SWC, aeration, and temperature of the plant-soil system (Brevik et al. 2002; Gill 2007). Increase in soil compaction due to grazing at heavy stocking reduces water infiltration (Hiernaux et al. 1999; Evans et al. 2012), affects root growth and density (Donkor et al. 2002), and creates less pore space, which can limit gas exchanges (Kotzé et al. 2013), and ultimately reduce plant water-use efficiency (Snyman 2005). In contrast, Beukes and Cowling (2003) indicated that water infiltration increased under heavy grazing condition due mainly to more active soil biota. On the other hand, livestock exclusion enhances SWC and water-retention capacity by increasing water infiltration (Chen et al.

2011; Witt et al. 2011). Altesor et al. (2006) noted that SWC (measured in mid winter) at top soil layers in an ungrazed treatment was 10% higher than grazed treatments.

1.3.3. Impacts of land-use types on soil carbon and nitrogen storage

Increased human population has an impact on land-use change, which affects the C cycle and vegetation composition (Canadell et al. 2007), resulting in a decline of species diversity and shifts in functional groups. Land-use change is one of the drivers of ecosystem C and N storage (Watsol et al. 2000; Biazin & Sterk 2013). According to Chen et al. (2011), SOC and total nitrogen (TN) concentration in top soil layers were highly related, and closely linked to land-use types. Soil organic matter and TN concentration increased considerably after farmland had been abandoned to grassland for ten years, while C : N ratio was not affected (Chen et al. 2011). The land-use change from cropland to grassland (Conant et al. 2001) and crop to pasture (Lal 2002) increases C stocks, while the reverse reduces C and N stocks (Jiao et al. 2012). It has been noted that land-use change had a larger effect on C stocks than climate change (1400 g m^{-2} vs 200 g m^{-2} C losses) (Burke & Grime 1996). Reports indicated that a loss of C in the first 10 to 20 years due to land-use changes is enormous (Evans et al. 2012; Wilson et al. 2012). For example, North America lost as high as 1400 g C m^{-2} in hundred years due to land-use changes from grassland to arable lands (Burke et al. 1996). Intensive land-use for crop cultivation in extremely fragile environments of arid areas often results in serious soil loss by water and wind erosion (Chen et al. 2012), resulting in reduced C and N stocks (Lal 2009; Kotzé et al. 2013). The rate of change in C, N and other related soil nutrients may be gradual, but its effect could differ remarkably based on ecosystem productivity and functioning.

1.4. Conclusion and hypotheses formulation

Unpredictable rainfall, along with poor grazing management practices influence rangeland vegetation, forage quality, soil properties, and ultimately affect livestock production in arid and semi-arid areas of the continent, mostly in sub-Saharan Africa. Grasslands may become more vulnerable to climate change in these areas as precipitation reduces, and events are expected to be less frequent, but more intense (Fay et al. 2008). Less frequent, but more intense rainfall events affect aboveground phytomass and rooting structures of vegetation, and may ultimately result in shifts in species composition and function. Simulated drought or changed precipitation study could provide information on the direction of shifts in species composition in the predicted climate change scenarios, and may help to identify drought resilient species in their natural habitat. Besides, the identification of direction of shifts in species composition, stability, that is, those species that adapt and give reasonable yield and quality under altered precipitation conditions need to be known (Wu et al. 2010; Fay et al. 2008).

Overgrazing and its associated effects (vegetation losses, land degradation, soil erosion, etc) are not uncommon in the communal grasslands of South Africa. Grazing management practices, stocking rate and time of forage utilization are important, because in the long term these practices could result in shifts in vegetation and modification of soil properties. These ultimately affect soil ecological function, rangeland productivity and thereby sustainable animal production. Nonetheless, the long-term effects of these practices in arid area are not well documented. In Africa, lack of adequate scientific information and understanding concerning SOM has limited the potential use of grasslands in mitigation against climate change by sequestrating carbon in soil (Lal 2001, 2002). Besides C, N is one of the most limiting soil nutrients in grasslands, and along with C, their dynamics need to be understood. However, results from different parts of the world were found to be inconsistent regarding soil C and N stocks, even under similar rainfall and grazing management conditions. This variability signifies the complex relationships between these and other soil nutrients, depending on landscape position and soil geomorphology, as well as grazing management practices. Site-specific differences of these important plant nutrients need to be investigated in line with grazing management practices. It is, therefore, important to improve understanding of impacts of drought and grazing and defoliation on plant and soil properties. This helps to develop and implement sound management practices aimed at maintaining grassland biodiversity, sustaining soil ecological functions, and improving forage availability and quality. Reliable information is required to understand processes and establish the cause-effect relationships, and to develop appropriate methods of land and livestock management.

Therefore, it is hypothesized that in low rainfall arid regions, grazing at a light stocking rate would have minimal effects on soil properties compared with heavy grazing. Similarly, in these regions, the effect of grazing on soil C, N and other selected physical and chemical properties is minimal under winter grazing, during which soil-water content is minimal. It is expected that long-term studies would have more time to reveal variation in soil properties and would thus show a positive effect of grazing treatments on C, N and other soil parameters. It was also hypothesized that changes in precipitation would not result in shifts in species composition, yield and changes in nutritive value due to the higher resilient ability of dominant plant species in this environment. In this regard, any observed variability among key grassland species in response to reduced precipitation might help to identify suitable forage species that could produce reasonable yield without significant decrease in forage quality under climate change conditions for the region.

1.5. Specific objectives

This research was undertaken with the following specific objectives:

- To investigate effects of land-use types on carbon and nitrogen storage and related soil properties. It is expected that as a key driver of ecosystem function, land-use changes could affect (negatively or positively) C and N stocks, among others.
- To assess long-term impacts of stocking rate on soil carbon sequestration and other soil properties. It is expected that grazing pressure due to heavy stocking rate could have negative impact on C, N and other soil properties under continuous grazing, where there was no rest period for vegetation to be regenerated and soils to be restored.
- To assess long-term impacts of season of grazing on soil carbon sequestration and other soil properties. It is expected that soil C sequestration and other soil properties could be affected among seasons, due to the fact that soil could be sealed by animals, infiltration could be affected, and aboveground biomass could be limited by the amount and distribution of rainfall.
- To investigate influence of reduced precipitation (simulated drought) and defoliation interval on SWC, and herbage yield and rain-use efficiency (RUE). It is expected that differences between species responses in terms of soil moisture and herbage yield would occur due to the influence of different levels of rain interception and defoliation intervals.
- To determine yield and nutritive quality of dominant forage species under reduced precipitation (simulated drought) conditions. It is expected that changes in precipitation could shift species composition and so does yield and nutritive quality.

CHAPTER 2

Influences of land-use types on soil organic carbon, total nitrogen and related soil properties in semi-arid area, Pretoria

2.1. Abstract

Carbon (C) and nitrogen (N) sequestration potential of different land-use types varies according to management practices, but such information is limited in sub-Saharan Africa. This study aimed at quantifying soil C, N and selected soil properties under various land-use types: cropland (CL); cultivated pasture land (CPL); *Leucaena* sp. plot (LP); and exclosure around Pretoria, which represents the semi-arid agro-ecological zone of South Africa. Soil samples were collected at two soil layers (0–15 cm and 15–30 cm) in each land-use type, using four to five transects. Generally, all soil parameters considered in this study differed significantly between land-use types. The exclosure had higher C, N, and C : N ratios, but lower bulk density (BD) and pH than the LP, CPL and CL. The major cations (Ca, Mg, K and Na) showed wide variations among land-use types. The top soil layer (0–15 cm) demonstrated lower BD (P<0.01) and pH (P<0.05), but had higher C (P<0.001), N (P<0.001), Na (P<0.05) and CEC (P<0.05) than the lower 15-30 cm. There was interaction effects (P<0.05) between land-use type and soil depth only for C and N concentration. The top 0–15 cm soil layer had generally higher C and N than the lower 15–30 cm layer, signifying their sensitivity to land-use types and soil depths. When potential C and N stock was estimated, the C sequestration rate was 1.41, 0.73, 0.40 and 0.33 Mg C ha^{-1} yr^{-1} in the LP, exclosure, CL and CPL while the corresponding values for N sequestration were 0.11, 0.05, 0.04 and 0.03 Mg N ha^{-1} yr^{-1}, respectively. Land-use management practices geared towards the use of multipurpose tree species through identification of suitable niches and animal exclusion from fragile soils would potentially improve C and N storage in semi-arid areas of South Africa.

Key words: cultivated land, exclosure, carbon stock, nitrogen stock, plantation, soil property

2.2. Introduction

Soils store the largest terrestrial organic C stock, which is about twice as large as that in the atmosphere and about three times that in vegetation (Lal 2008) and have substantial potential to act as C sinks (Lal 2009). Besides soil C, N is one of the limiting nutrients for plant growth and rangeland productivity. As a result, soil C and N storage have received increasing attention worldwide in recent years, largely because their emissions contribute to global warming (Lal 1998, 2008; Chen et al. 2014). Land-use change is one of the key drivers of ecosystem C and N dynamics (Watson et al. 2000). Many studies have reported that land-use changes had significant impact on soil organic matter content and thereby on soil C and N storage (Conant et al. 2001; Celik 2005; Du Preez et al. 2011b). Human alteration of natural ecosystems for agriculture is an important component of disturbance, and could alter community cover and functions of ecosystems (Chen et al. 2011; Du Preez et al. 2011b). Nevertheless, there is little information about the extent and mechanisms associated with land use on the dynamics of C and N storage in arid and semi-arid areas of South Africa (Du Preez et al. 2011b) and elsewhere in the world (Lal 2008).

Soil C and N vary spatially from within a field to a much larger regional scale, and are influenced by both intrinsic and extrinsic factors (Castrignano et al. 2000). The land-use change from cropland to grassland (Conant et al. 2001) or from crop to pasture (Lal 2002) increases C storage, while the reverse reduces C storage (Jiao et al. 2012; Du Preez et al. 2011b). Frequent cultivation in a fragile soil often results in serious soil loss by water erosion (Moussa et al. 2007; Chen et al. 2011) and decreases C and N storage (Du Preez et al. 2011b; Kotzé et al. 2013). Other than management practices, inherent soil characteristics may affect plant nutrients. Soil texture for example, particularly the clay content, exerts a major control on the amount of C in the soil (Telles et al. 2003; Tan et al. 2004), so that C and N may have been protected by macro aggregates (Six et al. 2000; Sotomayor-Ramırez et al. 2006; Evans et al. 2012). Owing to differences in soil type, landscape position and climate, however, soil C and N storage may be context and site specific, as biomass production, the major C and N inputs, is limited by the amount and distribution of rainfall (Reeder & Schuman 2002). In addition, management practices that are probably affecting soil C and N would influence the concentration of other soil macro-nutrients such as potassium (K) and phosphorus (P) (Jiao et al. 2012). Better understanding of C and N sequestration potential across land-use types under extremely unpredictable rainfall, heterogeneous soils, and management systems is essential (Reeder et al. 2004; De Baets et al. 2013).

On the other hand, soil nutrients show great variability across soil layers (Celik 2005; Chen et al. 2011). Conant et al. (2001) noted that C in the top soils is the most sensitive to management or land-use changes. Soil organic C and N concentration in top soil layers are highly related to each other, and are closely linked to land-use types (Chen et al. 2011) owing to the amount and composition of soil organic matter. According to Celik (2005), soil organic matter (SOM), which is a reservoir of C and N, reduced by 48–50% in the top 0–20 cm soil layers in cropland compared with soils planted with pasture over 12 consecutive years. Soil organic C and N levels increase or decrease in the top 30-cm soil layer due to changes in SOM. The extent of these changes is dependent *inter alia* on land use, soil form and environmental conditions (Reeder et al. 2004; Du Preez et al. 2011b). Hence, site-specific information is required on the amount of major soil nutrients under various land-use types.

This study was undertaken to compare and quantify variations in soil C and N storage, as well as selected soil properties (major exchangeable cations, pH, cation exchange capacity (CEC) and bulk density (BD) under various land-use types in Pretoria to provide baseline information for optimizing land-use practices that improve soil ecological function, and sustainable crop and livestock production in semi-arid areas of Pretoria, South Africa.

2.3. Materials and methods

Soil samples were collected from different land-use types at Hatfield Experimental Farm, University of Pretoria. The study site is situated at an elevation of 1372 m above sea level, latitude of 25° 45`S and longitude of 28$^{\circ}$ 16`E. The study site is in a summer rainfall area with long-term mean annual precipitation of 674 mm. Standard deviation and coefficient of variation of long-term mean annual precipitation, temperature, and evapotranspiration of the study site as well as soil sampling time (2014/15) are presented in Table 2.1.

Table 2.1 Soil sampling time (2014/2015) and long-term (1996-2014) mean, standard deviation (SD) and coefficient of variation (CV) of rainfall, temperature and evapotranspiration in Pretoria, South Africa

Year		2014/2015			Long-term (18 years)		
		Mean	SD	CV (%)	Mean	SD	CV (%)
Rainfall (mm)		720.2	7.18	36.5	671.4	6.48	17.0
Temperature (°C)	max	26.3	4.27	16.2	26.0	4.63	17.8
	min	11.41	5.05	44.3	11.92	5.13	43.0
Evapotranspiration (mm)		4.01	1.41	35.3	3.94	1.39	35.3

The soil of the study site is categorized as sandy clay loam, consisting of 55.9% sand, 34.2% clay and 9.9% silt (Soil Classification Working Group 1991), with pH of 6 in a 2.5: 1 soil-water ratio. The soil form is Hutton with weak structure, a homogenous red colour, and is non-calcareous (Non-affiliated Soil Analysis Work Committee 1990).

Four land-use types were considered in the study: cultivated pastureland (CPL), about 60 years, long-term maize nutrition trial cropland (CL), about 60 years, *Leucaena leucocephala* sp. plots (LP), about 16 years, and exclosure, about 60 years. The land-use types were selected based on histories of soil, crop and pasture management, and spatial heterogeneity at Hatfield Experimental Farm. Crop land is a maize *(Zea mays)* trial site, in rotation with wheat *(Triticum aestivum)*. Fertilizers (NPK) were applied at an annual rate of 150 kg N ha^{-1}, 60 kg P ha^{-1} and 100 kg K ha^{-1} before 1984. Since 1984, the rates have been changed to 100 kg N ha^{-1} and no application of P, due to the increased build-up of P in the soil. The exclosure is a fenced paddock (6.6 ha), which has been excluded from livestock grazing for the past 60 years.

During soil sampling, higher plant residues and litter accumulation were observed in the LP and exclosure treatments. Cultivated pasture land is land used for the cultivation of improved pasture, which has been planted with weeping love grass *(Eragrostis curvula)* for the past 60 years. Although *E. curvula* was sown every year, other invaders encroached the

CPL because of poor management. *Leucaena* sp. plot was the land used for *leucaena* seedling development and pruning was undertaken every year.

Six parallel transects for the CPL, five for the exclosure, and four for the LP and CL, , were demarcated for soil sampling. The transects were 50 m apart and perpendicular to the long axis of each plot. Transects were treated as replicates, as they were well distributed among the land-use types. Within each transect, soil samples were collected from ten sampling sub-plots, 20 m apart, and at least 15 m from the edges of the transect to minimize edge effects. At each sampling point, soil samples were collected at two depth intervals (0–15 and 15–30 cm) from each sampling sub-plot along the transects. For each transect, samples were bulked per layer, and mixed to make a single homogenous composite soil sample per layer. To determine BD, undisturbed soil cores (5.8 cm diameter) were collected (spring 2014) from each depth interval (0–15 and 15–30 cm) using core sampler.

Peak season aboveground net primary productivity was measured from each land-use type using the quadrat method (Daget & Poissonet 1971). For the CPL and exclosure treatments, four quadrats of 1 m x 1 m were used along each transect, and the forage/herbage biomass was clipped 5 cm above the ground. For maize treatments, four randomly spaced quadrats of 2 m x 2 m were sampled along each transect. Stover mass, which was left as aftermath and residue on the field, was estimated from grain yield in each plot. In the LP treatment, leaf and twigs were sampled from ten sub-plots of 5 m x 5 m quadrats along transects. The harvested forage/herbage phytomass was pooled for each transect and oven-dried (65 °C for 72 hr) (AOAC 2002).

Bulk density was determined on intact soil cores by drying the soil at 105 °C for 24 hr and dividing the oven dry mass by the inner volume of the core samples. Soil particle sizes were analysed with the wet sieving and sedimentation method. The samples were air dried and ground to pass through a 150-µm screen and analysed for total C and N, using a Carlo Erba NA1500 C/N analyser (Carlo Erba Strumentazione, Milan, Italy). All other chemical analyses were undertaken in duplicate: pH (1:2.5 soil to water suspension), exchangeable Ca, K, Mg and Na (1 mole dm^{-3} NH_4OAC at pH 7) following the methods described by the

Non-Affiliated Soil Analysis Work Committee (1990). Owing to differences in BD because of land-use type, annual C and N sequestration rates were estimated in each land-use type by dividing C or N stock differences by the number of years of land-use management. SOC and N concentration varied along the soil depths at different layers, and thus SOC and TN stocks (kg m^{-2}) were estimated after Bagchi and Mark (2010) as follows:

$$SOC = \sum_{i=1}^{n} D_i \rho_i SOC_i \qquad TN = \sum_{i=1}^{n} D_i \rho_i N_i$$

where SOC is soil organic C stock (Mg ha^{-1}); TN is total N stock (Mg ha^{-1}); i is the soil layer number [i=1(0-15), 2 (15-30)]; D_i is the depth interval (cm); ρi is the bulk density (g cm^{-3}) in the soil layer i; SOC_i is the mean SOC concentration (g kg^{-1}) in soil layer i; N_i is the mean N concentration (g kg^{-1}) in soil layer i.

Linear mixed model analysis, also known as the restricted estimation of maximum likelihood (REML) method (Payne 2014), which is used to analyse unbalanced replications, was applied to the soil property data, averaged over six, five, four and four replicates per transect for CPL, exclosure, LP and CL, respectively, in a split-plot arrangement. The analysis was applied as for an unblocked design with treatments as whole plots, depths as sub-plots, and transects as replication. The fixed effects were specified as depth, treatment and depth-by-treatment interaction, while the random effects were specified as transect-by-treatment interaction and the transect-by-treatment-by-soil-depth interaction. There were no correlations over depths, and Tukey's least significant difference tests were used to separate variance component means at less than or equal to 5% level. Data were analysed using the statistical program GenStat (Payne 2014).

2.4. Results

2.4.1. Soil organic carbon and nitrogen storage

For all land-use types and individual soil cores, C concentration ranged from 0.89% to 2.29% (mean 1.30% and CV 12.77%), whereas N concentration ranged from 0.076% to 0.17% (mean 0.11% and CV 12.67%) (Figure 2.1 and Table 2.2). All soil properties considered in this study were influenced by land-use type. The exclosure showed higher (*P*<0.001) C and N concentrations compared with other land-use types at both soil layers (Figure 2.1). Similarly, the C and N concentrations were (*P*<0.001) higher in the LP than in the CPL. C : N ratios differed (*P*<0.001) across land-use types; the least (9.81) being found in the CL, and the highest (13.44) in the exclosure.

Soil depth had significant effects on C, N, BD, Na, pH and CEC. There were higher (P<0.001) C and N concentrations in the top soil layer, where litter input and turnover is usually high, compared with the sub-soil layer. The top soil layer (0–15 cm) had lower BD (*P*<0.01) and pH (*P*<0.05), but higher Na concentration and CEC (*P*<0.05) than 15–30 soil layers. The C and N concentrations were affected by the interaction effect of land-use types and soil depths (Table 2.2); the LP and CL at the 0–15 cm soil layer having higher C concentration compared with the CPL at the 0–15 cm soil layer, while there was no interaction effect (*P*>0.05) at the 15–30 cm soil layer (Figure 2.1).

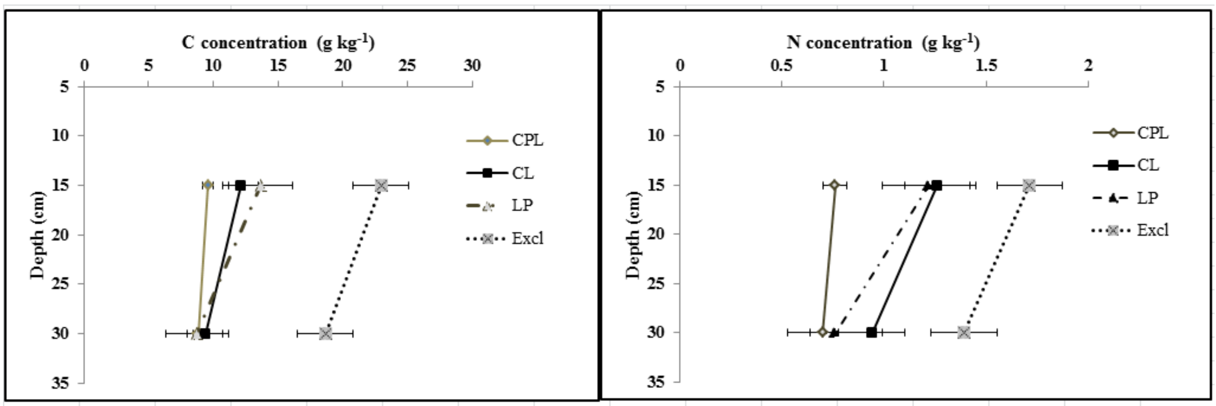


Figure 2.1 C and N concentration as affected by land-use types in Pretoria. Cultivated pasture land, CPL; Cropland, CL; *Leucaena* sp. plots, LP and Exclosure, Excl. *Bars* represent standard error of mean

Table 2.2 Partial ANOVA table showing degrees of freedom and F-values from restricted estimation of maximum likelihood analysis (at α=0.05) for soil organic carbon, total nitrogen concentration, carbon : nitrogen ratios, bulk density and soil chemical properties across land-use types (LU) and soil depths (SD) in Hatfield, Pretoria, South Africa

Parameters		DF	SOC	TN	C : N	BD	Ca	Mg	K	Na	CEC	pH	EC
Effects	LU	1	102.36***	66.77***	52.4***	10.16***	5.79**	9.41*	3.06*	6.28*	4.58*	2.85*	7.42***
	SD	3	33.53***	41.92***	0.76NS	6.78**	0.00NS	0.25NS	0.15NS	1.66*	2.79*	3.27*	0.06NS
	LU x SD	3	3.26*	3.74*	0.18NS	2.71NS	0.33NS	0.13NS	1.45NS	0.15NS	0.29NS	0.52NS	0.08NS

NS, *P*>0.05; **P*<0.05; ***P*<0.01; **P*<0.001; DF, degree of freedom; SOC, soil organic carbon; TN, total nitrogen; BD, bulk density; CEC, cation exchange capacity; EC=electrical conductivity

The C and N stocks and C : N ratios differed between land-use types (Table 2.3). There were higher (*P*<0.001) C stocks in the exclosure compared with the LP, CL and CPL. Likewise, the C stock was higher (*P*<0.01) in CL than CPL, while there was difference between CPL and LP (*P*>0.05). N stock differed for each land-use type, the exclosure having the largest (*P*<0.001) N stocks (0.34 kg m^{-2}), while the CPL stored the smallest (0.16 kg m^{-2}). Similarly, the C : N ratios were largest (13.44) in the exclosure, with the smallest (9.81) in the CL. Regardless of land-use type, however, the top 0–15 cm soil layer had higher C and N stocks than the sub–soil layer.

Table 2.3 Soil organic C and total N stocks, C : N ratios and bulk density (mean ± SEM) for soil samples as affected by land-use type and soil depth in Hatfield, South Africa

Fixed effects	SOC (kg m^{-2})	Total N (kg m^{-2})	C : N	Bulk density (g cm^{-3})
Land-use type				
CPL	2.03 ± 0.11^{c}	0.16 ± 0.01^{d}	12.65 ± 0.19^{b}	1.47 ± 0.016^{a}
CL	2.40 ± 0.13^{b}	0.25 ± 0.01^{b}	9.81 ± 0.23^{d}	1.49 ± 0.020^{a}
LP	2.26 ± 0.13^{bc}	0.20 ± 0.01^{c}	11.36 ± 0.23^{c}	1.35 ± 0.018^{b}
Exclosure	4.35 ± 0.12^{a}	0.32 ± 0.01^{a}	13.44 ± 0.21^{a}	1.40 ± 0.020^{b}
Soil depth (cm)				
0–15	3.04 ± 0.09^{a}	0.26 ± 0.01^{a}	11.72 ± 0.15	1.40 ± 0.013^{b}
15–30	2.48 ± 0.09^{b}	0.21 ± 0.01^{b}	11.91 ± 0.15	1.45 ± 0.013^{a}

Mean values in each column for each effect followed by the different letters are statistically different at *P*<0.05, SOC, soil organic carbon; SEM=standard error mean

The C sequestration rate of the LP was in the range of 1.41 Mg ha^{-1} yr^{-1}, compared with 0.73 Mg ha^{-1} yr^{-1} for exclosure, 0.40 Mg ha^{-1} yr^{-1} for CL, and 0.33 Mg ha^{-1} yr^{-1} for CPL. Likewise, the N sequestration rates in the LP, exclosure, CL and CPL were 0.11, 0.05, 0.04 and 0.03 Mg ha^{-1} yr^{-1}, respectively.

2.4.2. Bulk density

Land-use types had profound effects on BD (Table 2.3). Bulk denisty was lower (*P*<0.001) in the LP and exclosure compared with the CPL and CL, while differences between LP and exclosure and between CPL and CL were minimal. The top soil layer (0–15 cm) had lower BD compared with the sub-soil layer.

2.4.3. Soil chemistry

All soil chemical properties considered in this study were influenced by land-use type (Table 2.4 and 2.5). Soil samples collected from the LP, exclosure and CL showed higher (P<0.01) Ca concentration compared with CPL, while the LP and CL had higher (P<0.05) K concentration compared with the CPL. Likewise, there was higher (P<0.05) Mg concentration in the exclosure compared with the CPL, while the LP and CPL had higher Mg compared with the CL. The Na concentration in the LP and exclosure was higher than the CL and CPL, but there was differences (P>0.05) between CPL and CL.

There was higher (P<0.05) CEC in the exclosure and LP compared with other land-use types (Table 2.5), while there was no CEC differences between the CPL and CL were (P>0.05). In contrast, the CL and LP had higher (P<0.05) pH value compared with the exclosure. Electrical conductivity (EC) was higher in the LP and exclosure than in the CL and CPL. Similarly, the EC of the LP was higher than that of the CPL. Between soil layers, the CEC was higher at the top 0–15 cm soil layer, while the reverse happened for the pH value; the sub-soil layer had higher pH value than the top soil layer.

Table 2.4 Selected soil chemical parameters (mean ± SEM) for soil samples as affected by land-use type and soil depth in Hatfield, Pretoria, South Africa

	Exchangeable cations (cmol kg^{-1})			
Fixed effects	Ca	Mg	K	Na
Land-use type				
CPL	2.91 ± 0.24^{b}	1.43 ± 0.14^{b}	0.17 ± 0.06^{b}	0.06 ± 0.002^{b}
CL	3.97 ± 0.30^{a}	0.91 ± 0.17^{c}	0.41 ± 0.07^{a}	0.06 ± 0.002^{b}
LP	4.14 ± 0.30^{a}	1.69 ± 0.17^{ab}	0.38 ± 0.07^{a}	0.07 ± 0.002^{a}
Exclosure	4.24 ± 0.27^{a}	2.09 ± 0.15^{a}	0.32 ± 0.06^{ab}	0.07 ± 0.002^{a}
Soil depth (cm)				
0-15	3.81 ± 0.20	1.57 ± 0.11	0.33 ± 0.05	0.07 ± 0.002^{a}
15-30	3.82 ± 0.20	1.49 ± 0.11	0.31 ± 0.05	0.06 ± 0.002^{b}

Mean values in each column for each effect followed by the different letters (^{a}, ^{b}) are statistically different at P<0.05, SOC, soil organic carbon; SEM=standard error mean

Table 2.5 Cation exchange capacity, pH (H_2O) and electrical conductivity (mean ± SEM) for soil samples as affected by land-use type and soil depth in Hatfield, Pretoria, South Africa

Fixed effects	CEC (cmol kg^{-1})	pH (H_20)	EC (mS cm^{-1})
Land-use type			
CPL	6.18 ± 0.53^{b}	6.08 ± 0.10^{ab}	4.57 ± 0.33^{c}
CL	7.21 ± 0.65^{ab}	6.19 ± 0.12^{a}	5.35 ± 0.41^{bc}
LP	8.32 ± 0.65^{a}	6.14 ± 0.11^{a}	6.29 ± 0.41^{ab}
Exclosure	8.90 ± 0.58^{a}	5.76 ± 0.11^{b}	6.71 ± 0.36^{a}
Soil depth (cm)			
0-15	8.16 ± 0.43^{a}	5.94 ± 0.08^{b}	5.78 ± 0.27
15-30	7.15 ± 0.43^{b}	6.15 ± 0.08^{a}	5.68 ± 0.27

Mean values in each column for each effect followed by different letters (^{a}, ^{b}) are statistically different at P<0.05. CEC, cation exchange capacity; EC, electrical conductivity; SEM=standard error mean

Peak season aboveground net primary productivity (ANPP) showed great variations between land-use types (Figure 2.2). The LP and exclosure had higher (P<0.05) ANPP compared with the others, while there was difference between CPL and CL (P>0.05).

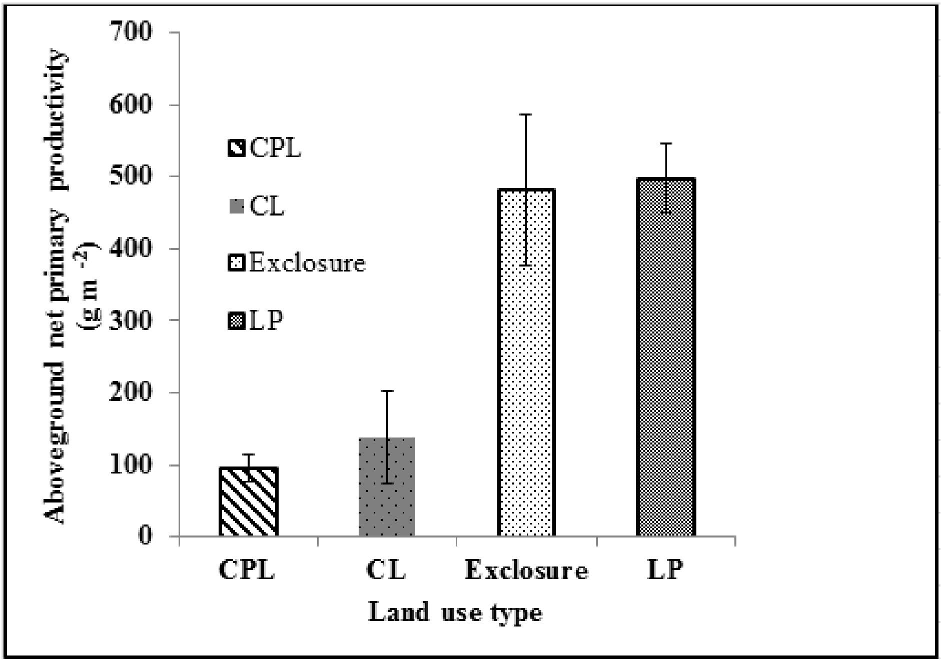


Figure 2.2 Peak season aboveground net primary productivity as affected by land-use type in Pretoria, South Africa. Cultivated pasture land, CPL; Cropland, CL; *Leucaena* plot, LP. SEM, standard error mean. *Bars* represent standard error of mean

2.5. Discussion

2.5.1. Soil organic carbon and nitrogen storage

The results from this study revealed that land-use type influenced soil C and N storage under various land management practices in this semi-arid environment. Climate (rainfall and temperature) and vegetation cover are the major determinants of SOM (Du Preez et al. 2011b), thus influencing soil organic C and N stocks (Allen et al. 2010). The higher C and N stocks in the LP and exclosure are due mainly to high litter input returned to the soil and the absence of disturbance (that is, grazing and cultivation). The CL has had no rest period because of continuous tillage for *Zea mays* production. The lower C and N concentration results obtained for CPL and CL in this study agree with previous reports (Du Preez et al. 2011b), which showed lower C and N concentration because of fast breakdown of organic matter under frequently cultivated soils. A surrogate for potential C and N sequestration rate was estimated by dividing the difference in soil C between land-use types by duration of management. Computing C and N sequestration rate using this procedure showed that the C and N sequestration rate of the LP was almost double (1.41 Mg C ha^{-1} yr^{-1} vs 0.11 Mg N ha^{-1} yr^{-1}) that of the exclosure (0.73 Mg C ha^{-1} yr^{-1} vs 0.05 Mg N ha^{-1} yr^{-1}). The higher C sequestration observed for LP signifies the potential of agro-forestry system to mitigate GHG emission by sequestrating C in the soil. In agreement with the findings of this study, Six et al. (2000) and De Baets et al. (2013) found higher C and N storage in an agro-forestry system due to higher biomass residue, litter composition and quality.

The C stock (0.73 Mg C ha^{-1} yr^{-1}) recorded in the exclosure is similar to values reported elsewhere (Witt et al. 2011; Kotzé et al. 2013), while the amount of C sequestrated in the exclosure (0.73 Mg C ha^{-1} yr^{-1}) in the study is reasonably higher compared with values reported for exclosure from other arid and semi-arid areas (Lal 2000; Du Preez et al. 2011a). Similarly, the N stock found in the exclosure is higher, compared with values reported from arid regions of the continent (Telles et al. 2003; Talore et al. 2015). The contribution of high aboveground plant biomass and associated belowground root density and distribution may have contributed to the increased C and N stocks in the LP and exclosure in the current study, which agrees with other reports (Chen et al. 2011; Wang et al. 2014a). A low decomposition rate due to minimum aggregate disturbance has also contributed to higher C and N stocks in the LP and exclosure. According to Moussa et al. (2007) and Witt et al. (2011), biomass production is one of the predominant limiting factors that affect C and N sequestration in arid lands. Increased soil macro-aggregation and stability due to higher SOM, in the presence of plant residue contributes to increased C and N storage in a no-till system (Six et al. 2000). However, the amount of C fixed by LP in the study is lower than values reported elsewhere in a semi-arid environment for citrus plants (Iglesias et al. 2013). On the other hand, the relationship of N with C plays a key role in building soil fertility and enhancing soil productivity (Franzluebbers 2003; Lal 2008; Franzluebbers & Stuedemann 2009). At landscape level, the availability of N is vital for C sequestration. However, the C : N ratio is variable and depends on climate, soils type, vegetation condition and agricultural management practices (Watson et al. 2000; Lal 2004; Pineiro et al. 2010).

Although a considerable amount of biomass was produced in the CPL and CL, removal of part of the biomass (forage and grain) resulted in low SOM, and associated lower C and N stocks. Moreover, C and N stocks in cultivated lands are lower in part because of the enhanced organic matter decomposition due to physical breakdown of larger particles into smaller particles, which increases the surface area of organic materials exposed to microbial attack (Six et al. 2000; Sotomayor-Ramırez et al. 2006; Du Preez et al. 2011b). The lower C stock in the CL found in this study concurs with the findings of Conant and Paustian (2002), who showed lower soil organic carbon under intensive crop cultivation. In this study, the C and N stocks found in the CPL are the least, which agrees with reports of Wilson et al. (2012), who noted that most cultivated pastures (95%) in the tropics are not well managed, and contribute less to C and N sequestration. The annual C sequestration rate of CL (0.4 Mg C ha^{-1} yr^{-1}) was similar to values reported by Conant et al. (2001) (0.29 Mg C ha^{-1} yr^{-1}) for fields that received inorganic fertilizer.

Usually, soil C and N reduce as soil depth advances. Regardless of land-use type, soil C and N were higher in the top (0–15 cm) soil layers, suggesting that SOM is usually high in the top soil layers. The higher C and N stocks in the top 15-cm soil layer in the study for the LP and exclosure concur with the findings of Conant et al. (2001). Soil organic carbon in the top soils decreases with time in a land-use system that has been converted from permanent pasture to arable land (Conant et al. 2001; Wilson et al. 2012) owing to frequent disturbance through cultivation and removal of plant residue. Celik (2005) noted that the SOM of cropland soils decreased by 48–50% at 0–20 cm soil layers compared with pasture soils over 12 years. The effect of continuous disturbances and breakage of soil macro aggregates on soil C and N stocks has widely been documented (Six et al. 2000; Evans et al. 2012; Wang et al. 2014b). This implies that in most of cases, soil disturbance leads to a reduction of SOC and N stocks.

High clay content is usually important for soil aggregate stability (Six et al. 2000; Kotzé et al. 2013) and reduces C losses from oxidation, and N losses from ammonia volatilization, N mineralization and runoff (Six et al. 2000). In this study there was high clay content in the exclosure, which may have contributed to increased C and N stocks. This is in agreement with other reports (Six et al. 2000; Witt et al. 2011) that recorded increased C and N storage in macro aggregates of clay soils. However, research reports are not consistent about the role of soil texture on responses of soil C and N. For instance, Reeder et al. (1998) found higher C stock in sandy soils than clay soils, while Sousanna et al. (2004) reported insignificant effects of soil texture on C stocks. On the other hand, Chan et al. (2003) noted that light-textured soils (<35% clay content) have inherent problems of low organic matter levels, resulting in reduced C and N stocks.

Unlike soil C and N, other soil properties were less likely to be affected by the soil depth considered in this study. This was probably due to soil mix-up between the top and sub-soil layers by land ploughs during tillage (Kotzé et al. 2013). In addition, in this study, erosion and low plant residue inputs were possible causes for nutrient loss from the top soil layers under low vegetation cover for the CPL and CL. Kotzé et al. (2013) noted loss of plant nutrients by erosion from bare soils. The higher CEC and lower pH level in the top (0–15 cm) soil layers across land-use types in the current study might be partly due to high weathering of the soil (Witt et al. 2011; Wang et al. 2014b). However, concentrations of plant nutrients across soil profiles vary depending on plant residue, soil type, land management practices and climatic conditions (Kotzé et al. 2013; Talore et al. 2015).

2.5.2. Management implications

The higher C and N stocks in the exclosure and LP suggest that appropriate land and livestock management are essential in the extensive arid lands of South Africa. Soils of these regions are fragile, but may have high potential to store atmospheric C and N under proper management. This confirms earlier findings by Beukes and Cowling (2003) of higher C under properly managed grazing conditions. In these large grazing areas of the country, land management through appropriate stock exclusion is of utmost importance for land rehabilitation (Moussa et al. 2007; Du Preez et al. 2011a). Owing to the untapped potential of these lands, special attention needs to be given to micro-climatic zones where soils are potentially under degradation, such as communal grazing areas in the Bophirima districts of North-West province, South Africa (Snyman & Du Preez 2005; Moussa et al. 2007). Livestock exclusion in these lands had been recommended since communal lands have been reported to be inefficient in land and livestock management (Moussa et al. 2007; Kotzé et al. 2013). This is due mainly to vegetation removal, resource overutilization, and the absence of rest periods for adequate vegetation recovery and restoration of soils. An appropriate environmental policy that supports livestock exclusion from fragile soils and, where possible, promotion of agro-forestry systems (for example using multipurpose trees such as *Leucaena*) by identifying drought-tolerant grasses for niches (marginal areas, eroded lands, etc) would help to rehabilitate nutrient depleted soils in extremely degraded semi-arid areas.

2.6. Conclusion

This study examined the influences of land-use type on soil C, N and selected soil properties in semi-arid environments, where rainfall is extremely variable. The results suggest that the LP and exclosure contributed to increased C and N stocks. When the numbers of years of land management were taken into consideration, the SOC and N stocks increased more than double in the LP, compared with exclosure, and more than fivefold compared with other land-use types, largely because of higher plant residues and litter accumulation. The great variability in C and N storage across soil layers indicates that C and N are more sensitive to frequent cultivation and soil disturbance than other soil nutrients. The results also suggest that inorganic fertilizers applied for the CPL and CL did not result in a substantial build-up of organic matter in the long run, nor do C and N stocks. In the absence of appropriate intervention strategies, losses of C and N concentration induced by poor pasture management and frequent crop cultivation would be more rapid owing to less vegetation cover and litter inputs. Such information is important for land-use and ecosystem management, and sustainable crop and livestock production in semi-arid environments. Because there are vast native pasture areas in South Africa, there is substantial potential for C and N storage in semi-arid areas through strategic animal exclusion and identification of suitable niches to establish multipurpose trees. To optimize productivity in terms of livestock production and plant nutrient replenishment, further study is needed to determine the C and N sequestration potential of native grazing lands. More focus should be given to commonly practised land-use types in grassland ecosystems and the effects of grazing management practices such as stocking rate and time of forage utilization on C and N sequestration potential and status of other soil nutrients.

CHAPTER 3

Long-term impacts of stocking rate on soil carbon sequestration and selected soil properties in the arid Middleburg, Eastern Cape, South Africa

3.1. Abstract

Little is known about the way in which basic soil properties respond to contrasting stocking rates in the Karoo biome, South Africa. The aim of this study was to investigate impacts of long-term (>75 years) grazing at heavy (1.18 small stock unit, SSU ha^{-1}) and light (0.78 SSU ha^{-1}) stocking rate and exclosure (non-grazed control) on selected soil properties. Soil samples were collected to a depth of 60 cm from the long-term experimental site of Grootfontein Agricultural Development Institute (GADI), Eastern Cape, South Africa. The samples were analysed for carbon (C), nitrogen (N), bulk density (BD), and infiltration rate, among others. Generally, heavy and light grazing reduced soil N storage by 27.5% and 22.6%, respectively, compared with the exclosure. Animal exclusion improved the water infiltration rate and C stocks (P<0.05), which were 0.128, 0.097, and 0.093 Mg ha^{-1} yr^{-1} for exclosure, light and heavy grazing, respectively. Soil penetration resistance was higher for grazing treatments in the top 3–7 cm soil layer but for exclosure it was higher at the top 1 cm soil surface. Although livestock exclusion could improve C sequestration, sufficient resting periods for 1–2 years followed by three consecutive grazing years at a light stocking rate would be ideal for sustainable livestock production in the Eastern upper karro areas of South Africa.

Key words: arid lands, carbon, continuous grazing, exclosure, nitrogen, soil properties

3.2. Introduction

The grassland and Karoo (grassy dwarf shrubland) biomes of South Africa are major feed resources for livestock farming (Mucina & Rutherford 2006). In these arid environments of the country, majority of the lands is used for livestock grazing (Smet & Ward 2006). However, most of these biomes have been subjected to loss of nutrients and biodiversity changes, soil organic matter (SOM) and land deterioration owing to vegetation removal by stock and burning, and climate variability (Du Preez et al. 2011a). Grazing management practices affect the magnitude, distribution and cycling of carbon (C) and nitrogen (N) in the Karoo rangeland ecosystems (Pineiro et al. 2010; Kotzé et al. 2013). Improving soil organic carbon (SOC) storage in dryland soils through management of livestock is one of the techniques advocated to mitigate against and adapt to greenhouse gas emission (GHG) (Kotzé et al. 2013; McSherry & Ritchie 2013). However, understanding of grazing effects on C dynamics in Karoo rangeland (i.e field) remains limited, particularly in sub-tropical Africa, where rainfall varies remarkably (McSherry & Ritchie 2013).

The effects of grazing on C and N stock have been variously documented in the literature. For example, increasing grazing intensity enhances soil C (Reeder et al. 2004; Li et al. 2011) and N concentrations (Liu et al. 2011); has no effects (Barger et al. 2004); and decreases soil C (Gill 2007) and N levels (Steffens et al. 2008). These variations in C and N stock are a reflection of disparities in climate, soil type, landscape position, plant community type, and management practices (Milchunas & Lauenroth 1993; Reeder & Schuman 2002; Li et al. 2011; McSherry & Ritchie 2013).

The availability of N can control C and N accumulation, because it constrains both inputs and outputs of C and N (Pineiro et al. 2010). It increases primary productivity, raising C inputs to the soil, and may decrease soil respiration, lowering C outputs from the soil (Pineiro et al. 2010; Cheng et al. 2011). Grazers can alter N stocks by increasing or decreasing N inputs and outputs. They may decrease N inputs by lowering legume biomass or cover, because most grasslands experience some level of N limitation (Lal 2004). Heavy grazing can negatively influence vegetation by destroying and disrupting the soil structure, enhancing organic matter oxidation (Frank et al. 1997) and resulting in changes in soil C and N storage. Grazing-induced changes in C and N balance modify the concentration of other plant nutrients in the soil (Marriott et al. 2010; Evans et al. 2012) and may affect soil compaction (Evans et al. 2012).

Previous studies evaluated grazing (light and heavy) effects on vegetation composition, rangeland conditions and organic matter on the clayey soils of South Africa (Du Preez et al. 2011a; Kotzé et al. 2013). Only a few studies documented the effects of land management systems on soil properties in the country (Kotzé et al. 2013). However, the long-term effects of sheep stocking rate on C and N stocks and other soil properties have not been studied in arid regions. In drier and arid ecological regions, there might have trade-offs between managing lands for soil C and N and animal production (Marriott et al. 2010). Context-specific information is essential to advocate land management practices that increase carbon sequestration (Derner et al. 2007).

The aim of this study was to assess long-term impacts of grazing treatments on soil C, N and other soil properties, as well as soil compaction and water infiltration. It was hypothesized that in low rainfall arid regions, grazing at a light stocking rate would have insignificant negative effects on soil properties compared with heavy grazing.

3.3. Materials and methods

This study was conducted at the long-term experimental site of GADI, Eastern Cape, South Africa. The study site is 122.4 ha and located at latitude 31° 22`E, longitude 24° 45`N, with an altitude of 1900 m above sea level. The long-term annual mean and median rainfall are 370.2 and 359 mm, respectively. Rainfall distribution (seasons of southern hemisphere) is 15% in spring (September–October), 30% in summer (November–March), 50% in autumn (April–May) and 5% in winter (June–August). The average January temperature is 20.9 °C, while the average July temperature is 7.9 °C (Figure 3.1) (Du Toit 2010). Treatment site, relative position and experimental layout of the study site are indicated in Figure 3.2 and Figure 3.3.

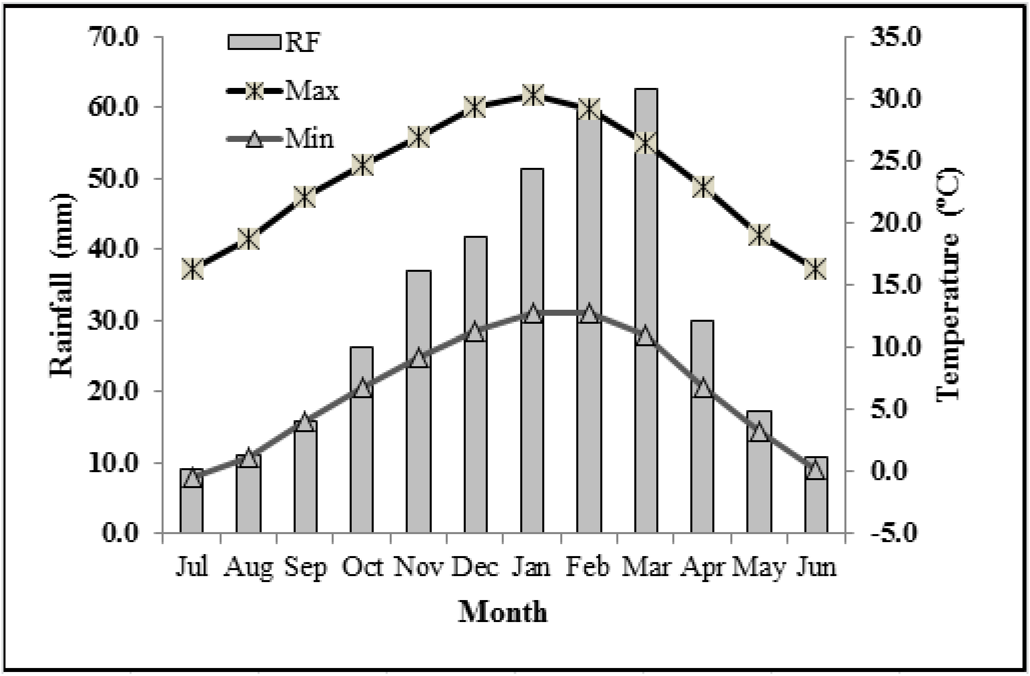


Figure 3.1 Long-term average monthly rainfall (1889–2014) and temperature (1916–2008) in Middleburg, Eastern Cape, South Africa

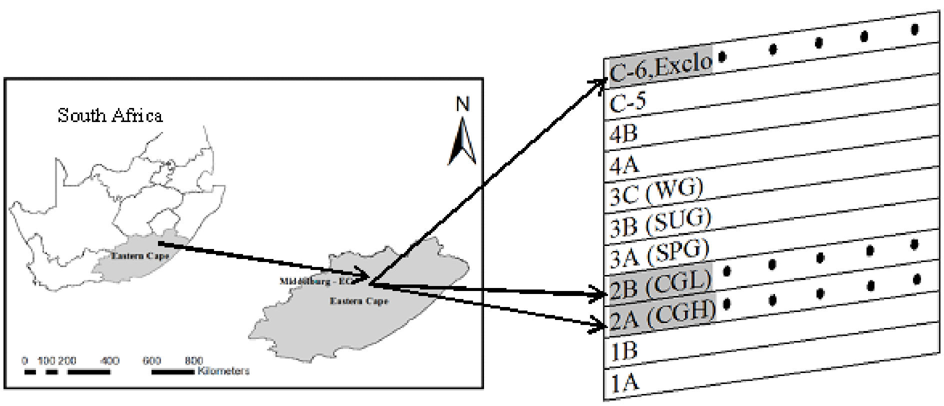


Figure 3.2 Treatment sites, their relative positions and experimental layout of the long-term grazing study site in Middileburg, Eastern Cape, South Africa

Closed small circles are sampling points: C-6, Exclo (exclosure); CGL (stocking at light grazing); CGH (stocking at heavy grazing). WG, winter grazing; SUG, summer grazing; SPG, spring grazing

Figure 3.3 Long-term (>75 years) experimental trial site (left, water point; right, exclosure) at Grootfontein Agricutrual Development Institute (GADI), Middleburg, Eastern Cape, South Africa

The Karoo biome is grassy dwarf shrubland, the third largest biome in South Africa, covering about 20.5% of the land, (Mucina & Rutherford 2006; Du Toit et al. 2011), and is ideal for sheep and goat production. The underlying geology is varied because distribution of the biome is determined primarily by rainfall. Climatologically and biologically, the Karoo is a heterogeneous and ecotonal region. It is an ideal biome for sheep and goat production. In the main river valleys, people farm olives, citrus and deciduous fruit. According to Du Toit et al. (2011), the dominant grass species in continuous grazing at heavy and light stocking rate treatments include *Aristida congesta (*three-awn grass*), A. diffusa* and *Eragrostis lehmanniana* (Lehman's love grass). On the other hand, grass species in the Eastern upper karoo areas such as *Themeda triandra* (red grass), *Sporobolus fimbriatus* (drop seed grass), *Eragrostis curvula* (weeping lovegrass), *Heteropogon contortus* (spear grass), *Hyparrhenia hirta* (thatch grass) and karoo bushes (shrubs) such as *Pymaspermum parvifolium, Felicia murcata, Salsola Caluna, Walafrida geniculata, Diospyros austroafricana* (jackal-berry), *Diospyros lycioides* (star apple) *and Searsia undulate* are commonly found in the exclosure (Du Toit et al.2011). Most of the grasses are of the C-4 type and, like the shrubs, are deciduous in response to rainfall events. Grazing increased the relative abundance of dwarf shrubs rapidly over the study years (Du Toit

2010). The predominant soil type, over 80% of the Karoo area, is a lime-rich weakly developed shallow (<30 cm) soil over rock. Generally, the soils of the study site are of the Shigalo series aridisols (Soil Survey 1999). Although less than 5% of rain reaches the rivers, the high erodibility of soils poses a major problem where overgrazing occurs. The soil colour of the study site was identified as 7.5YR, 5YR and 10YR in heavy, light grazing and exclosure, respectively. The soil texture of the site was analyzed and, on average, the silt, clay and sand contents of the study site were 13.52, 19.9 and 64.7%, respectively.

Merino wethers at a stocking rate of 0.78 and 1.18 small stock unit (SSU) ha^{-1} were used for light and heavy grazing treatments, respectively, with individuals being replaced with young animals after three to four years. Animals were allowed free grazing on rangeland in day hours (9:00 to 17:00) and were not supplemented with other feeds throughout the study period, because the initial objective of the study was to evaluate wool yield and animal performance of growing sheep under rangeland conditions. Except for minimal urine and faeces returning to the soil in grazing treatments, inorganic fertilizers were not applied to the rangeland during the study period. The study started in 1934 and terminated in 2010/2011. The study site has consisted of grazing treatments, namely heavy grazing, light grazing and exclosure (non-grazed control) for more than 75 years. The plots were laid out in a parallel rectangular strips (width: length ratio of approximately 1 : 10) of 8.5 ha (heavy grazing), 8.5 ha (light grazing) and 3.4 ha (exclosure) along the gentle sloping Karoo rangeland. The strips or paddocks were fenced with barbed wires.

Five parallel transects (100 m wide), 100 m apart and perpendicular to the long axis of each plot were demarcated for soil sampling (Table 3.1). Transects were treated as replicates along the treatment strips, as the size of this long-term experiment made it impossible to replicate the treatments in the traditional way. Soil samples, at least 15 m from the edge of the plots, were collected every 20 m along each transect, at five sampling sub-plots for grazing treatments and three sampling sub-plots for exclosure. To analyse C and N and soil chemistry, soil was sampled up to 60 cm depth (0–10 cm, 10–20 cm, 20–30 cm and 30–60 cm) from each sampling sub-plot along transects. To determine bulk density, undisturbed soil cores (5.8 cm in diameter) were collected using a core sampler with the same sampling procedure as on the transects. Soil penetration resistance was measured (May 2012, early winter) at 1 cm intervals to a depth of 60 cm with a Rimik™ CP20. A total of 2744 readings were taken: 1161 from the heavy grazing, 1168 from the light grazing, and 415 from the exclosure. Infiltration rate readings (n = 90) were collected every ten minutes for an hour, using a mini double ring infiltrometer (15 cm inner diameter and 30 cm outer) until it reached a steady state .

Bulk density was determined on intact soil core by drying the soil at 105 $^{\circ}$C for 24 hr and dividing the oven dry mass by the volume of core sampler. Soil particle size distribution was analysed using the wet sieving and sedimentation method. The samples were air dried and ground to pass through a 150-µm screen and analysed for total C and N using a Carlo Erba NA1500 C/N analyser (Carlo Erba Strumentazione, Milan, Italy). All other chemical analyses were undertaken in duplicate: pH (1:2.5 soil to water suspension), and exchangeable Ca, K, Mg and Na (1 mole dm^3 NH_4OAC at pH 7). SOC content varied along the soil depths at various layers, and thus SOC stock (kg m^{-2}) and total N (TN) stock (kg m^{-2}) was estimated after Bagchi and Mark (2010) as follows:

$$SOC = \sum_{i=1}^{n} D_i \rho_i SOC_i \qquad TN = \sum_{i=1}^{n} D_i \rho_i N_i$$

where SOC is soil organic C stock (Mg ha^{-1}); TN, is total N stock (Mg ha^{-1}); i is the number of soil layers (i = 1, 2, 3 and 4); D_i is the depth interval (cm); ρ_i is the BD (g cm^{-3}) in soil layer i; SOC_i is the mean SOC concentration (g kg^{-1}) in soil layer i; N_i is the mean N concentration (g kg^{-1}) in soil layer i.

Linear mixed model analysis, also known as restricted estimation of maximum likelihood (REML) analysis (Payne 2014), was applied to the soil property data, averaged over each transect (five replicates for grazing and exclosure). The analysis was applied as for an unblocked design (a split-plot arrangement) with treatments (heavy grazing, light grazing and exclosure) as whole plots, depths (0–10, 10–20, 20–30 and 30–60 cm) as the sub-plots and transects as the replication. The fixed effects were specified as depth, treatment and the depth-by-treatment interaction, while the random effects were specified as the transect-by-treatment interaction and the transect by treatment-by-soil depth interaction. The REML method was used because observations were unbalanced between treatments at sampling sub-plots (five for grazing and three for exclosure) and at some soil layers (30–60 cm). There were no correlations over depths and Tukey's least significant differences test was used to separate REML estimates of variance components means at the 5% level. Data were analysed using the statistical program GenStat® (Payne 2014).

3.4. Results

3.4.1. Carbon and nitrogen storage

For all treatments and individual soil core, C concentration ranged from 2 g kg^{-1} to 14.7 g kg^{-1} (mean: 5.1), while N concentration ranged from 0.3 to 1.1 g kg^{-1} (mean: 0.6) and C : N ratios from 9.15 to 18.34. Heavy and light grazing practised in the study were lower than the arid region average (livestock unit, LSU ha^{-1}). However, the long-term cumulative grazing modified C and N concentration, and C : N ratios among treatments and across soil layers (Tables 3.1). For the treatment, that was maintained (that is, 75 years' livestock exclusion), the C and N concentration and C : N ratios improved. As a result, the exclosure attained higher C ($P<0.01$) and N ($P<0.001$) concentration in comparison with the heavy and light grazing. The exclosure had 29.6% and 36.4% more C concentration and 27.5% and 22.6% more N concentration compared with the light and heavy grazing, respectively. With regard to soil layers, the soil C and N concentration and C : N ratios varied considerably. The C and N concentration and C : N ratios in the Karoo biomes decreased with soil depth, exhibiting more C and N concentration, and C : N ratios at the uppermost soil layers (0–30) (see supplementary data published online).

Owing to differences in BD because of grazing, C and N stocks showed great variability across treatments (Table 3.2 and Figure 3.3). Accordingly, estimating the annual C storage since last soil sampling (C difference divided by 75 years), the CGH, CGL and exclosure treatments stored 0.093, 0.097 and 0.128 Mg C ha^{-1} yr^{-1}, respectively. Likewise, the heavy, light and exclosure stored 0.078, 0.082 and 0.096 Mg N ha^{-1} yr^{-1}, respectively. Except in the topmost soil layers, the C stock in the exclosure was consistently higher in all depths compared with the heavy and light grazing. The N stock showed significant treatment-by-depth interactions (Figure 3.3); the exclosure at 20–30 cm soil layer having higher (P<0.01) N stock (0.11 kg m^{-2}) compared to other treatment-by-depth combinations. Heavy grazing at 10-20 cm soil layer stored the least N (0.074 kg m^{-2}) than other treatment-by-depth combinations.

Table 3.1 Partial ANOVA table showing degrees of freedom and P-values from restricted estimation of maximum likelihood analysis (at α=0.05) for soil organic carbon and total nitrogen concentration, C : N ratios, bulk density and soil chemical properties across grazing management (GM) and soil depth (SD) in the arid Eastern Cape, South Africa

Effects	DF	SOC	TN	C : N	BD	pH	Ca	K	Mg	Na	CEC
GM	2	<0.01**	<0.001***	3.02NS	0.05NS	<0.001***	<0.001***	0.05NS	0.05NS	0.85NS	<0.001***
SD	3	0.05NS	<0.01**	2.76NS	<0.001***	<0.001***	<0.001***	1.70NS	<0.001***	<0.001***	<0.001***
GM x SD	6	0.29NS	0.23NS	0.88NS	<0.001***	<0.001***	1.18NS	0.54NS	0.05NS	0.88NS	<0.001***

NS, non-significant at $P>0.05$; *$P\leq0.05$; **$P<0.01$; ***$P<0.001$.

DF, degree of freedom; SOC, soil organic carbon; TN, total nitrogen, BD, bulk density; CEC, cation exchange capacity

3.4.2. Bulk density

Grazing and soil depth both influenced BD (Table 3.1). Generally, BD was lower in the uppermost soil layer (0–20 cm). The exclosure treatment had lower (*P*<0.05) BD at 0–10 cm soil layer compared to the light and heavy, while the reverse occurred at 10–20 cm soil layers; the heavy and light had higher (*P*<0.001) BDs compared to the exclosure (Figure 3.3). There was interaction (*P*<0.001) effects of treatment by depth, the exclosure at 0–10 cm soil layer having lower BD compared with the heavy and light at 10–20 cm soil layers, while no interaction effects of other treatment-by-depth combinations (*P*>0.05) on BD (Figure 3.3).

3.4.3. Soil chemistry

Continuous grazing managements were shown to influence all chemical properties of the soil considered in this study, except for Na concentration (Table 3.3). Calcium concentration was higher (*P*<0.001) in the exclosure compared with heavy and light grazing, while almost the reverse situation was observed for Mg concentration, where heavy and light grazing attained higher (*P*<0.01) Mg concentration. Similarly, heavy and light grazing had higher (*P*<0.001) K concentration compared with the exclosure. There were treatment-by-depth interaction (*P*<0.001) effects on CEC, the exclosure having higher CEC at all depths compared with the heavy and light grazing at depths considered in this study, while the CEC in the light grazing at 30-60 cm soil layer was higher (*P*<0.001) compared with the light grazing at 0-10 soil layer (Figure 3.3). The pH in the light at 30-60 cm soil layers was the highest (7.34), while it was the least (5.85) in the exclosure at 0–10 cm soil layers. The heavy and light grazing at 30–60 cm soil layers had higher pH compared to exclosure at 0–10 and 30–60 cm soil layers (Figure 3.3).

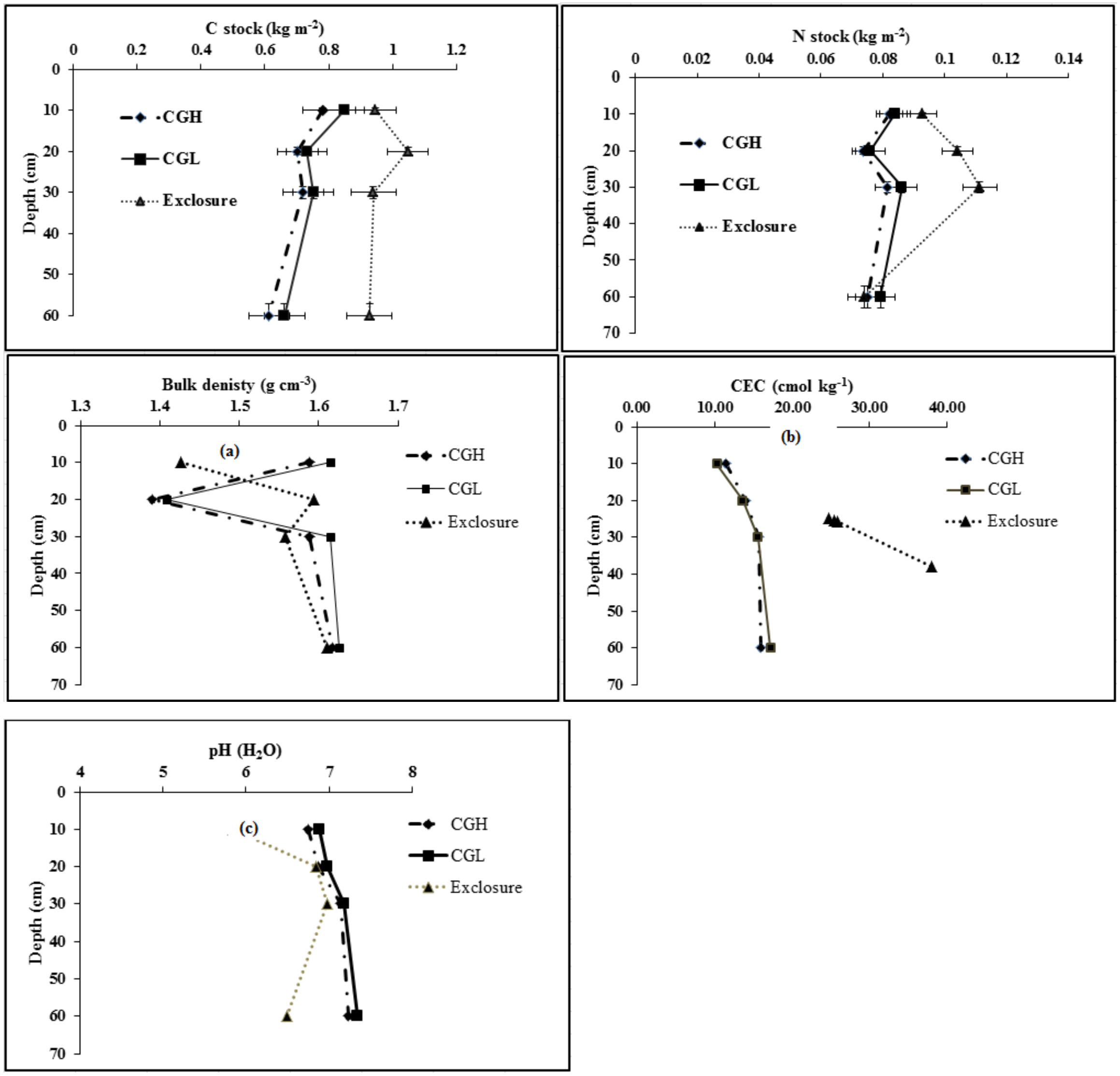


Figure 3.4 Carbon (C) and N stocks, bulk density~~,~~ cation exchange capacity, and pH (H_2O) as affected by grazing at heavy and light stocking rates and exclosure in the arid Eastern Cape, South Africa. CEC, cation exchange capacity; BD, bulk density; CGH, continuous grazing at heavy rate; CGL, continuous grazing at light rate

Bars represent standard error of mean

Table 3.2 Soil organic carbon (SOC) and total N concentration and C : N ratios (mean ± SEM) as affected by grazing management (GM) and soil depth (SD) in the arid Eastern Cape, South Africa

	Soil parameters		
Fixed effects	SOC (g kg^{-1})	Total N (g kg^{-1})	C : N
GM			
CGH	4.64 ± 0.31^{b}	0.51 ± 0.02^{b}	8.93 ± 0.34
CGL	4.88 ± 0.31^{b}	0.53 ± 0.02^{b}	8.92 ± 0.34
Exclosure	6.33 ± 0.32^{a}	0.65 ± 0.02^{a}	9.93 ± 0.35
Soil depth (cm)			
0-10	5.61 ± 0.26^{a}	0.57 ± 0.018^{a}	9.83 ± 0.35^{a}
10-20	5.62 ± 0.26^{a}	0.58 ± 0.018^{a}	9.66 ± 0.35^{b}
20-30	5.29 ± 0.27^{ab}	0.61 ± 0.019^{a}	8.73 ± 0.36^{c}
30-60	4.62 ± 0.27^{b}	0.51 ± 0.019^{b}	8.84 ± 0.36^{c}

Mean values followed by the same letter (a-b) within a column are not statistically different at $P<0.05$. SOC, soil organic carbon; N, nitrogen; SEM =standard error of the mean

Table 3.3 Selected soil chemical properties (Mean ± SEM) as affected by grazing management (GM) and soil depth (SD) in the arid Eastern Cape, South Africa

Fixed effects	Exchangeable cations[§] (mg kg^{-1})					
	Ca	K	Mg	Na	pH (H_2O)	CEC(cmol kg^{-1})
Grazing management						
CGH	895 ± 67.4^{b}	223 ± 23.6^{ab}	600 ± 53.7^{ab}	18.4 ± 2.50	7.00 ± 0.08^{a}	14.26 ± 0.82^{b}
CGL	766 ± 67.4^{b}	272 ± 23.6^{a}	681 ± 53.5^{a}	15.5 ± 2.50	7.10 ± 0.08^{a}	14.17 ± 0.82^{b}
Exclosure	1341 ± 71.8^{a}	171 ± 23.9^{b}	469 ± 54.2^{b}	20.3 ± 2.58	6.54 ±0.09^{b}	28.51 ± 0.84^{a}
Soil depth (cm)						
0-10	757 ± 60.1^{c}	215 ± 15.9	337 ± 36.8^{d}	13.40 ± 2.01^{b}	6.49 ± 0.07^{b}	15.73 ± 0.70^{c}
10-20	971 ± 60.1^{b}	230 ± 15.9	518 ± 36.1^{c}	16.07 ± 2.01^{b}	6.89 ± 0.07^{a}	17.43 ± 0.70^{bc}
20-30	1060 ± 65.9^{ab}	233 ± 16.2	685 ± 36.9^{b}	17.01 ± 2.08^{b}	7.10 ± 0.07^{a}	19.01 ± 0.73^{b}
30-60	1216 ± 62.4^{a}	208 ± 16.2	794 ± 36.9^{a}	25.73 ± 2.08^{a}	7.02 ± 0.07^{a}	23.76 ± 0.73^{a}

[§]Ammonium acetate solution. Mean values in each column followed by the same letters are not statistically different at $P<0.05$, SEM=standard error mean; CEC, cation exchange capacity

3.4.4. Soil compaction and water infiltration

After 75 years of grazing exclusion, generally higher soil penetration resistance was observed in the heavy than the light and exclosure (Figure 3.4). As a result, soil compaction under continuous grazing at heavy stocking rate was higher ($P<0.01$) than the exclosure in the top 11-cm soil layer (Figure 3.4a, b). The only exception, was the upper 1-cm depth (0–1 cm), where the exclosure had higher ($P<0.001$) compaction. At soil depths of 11–18 cm, soil compaction of the exclosure treatment was higher ($P<0.001$) compared with the heavy and light. Compaction readings below 20 cm (Figure 3.4c, d), however, reflected the effect of parent material rather than the treatments considered for this study (see supplementary data published online).

The water infiltration rates of the heavy and exclosure treatments were higher ($P<0.05$) compared with the light for the first ten minutes (Figure 3.5). Similarly, there was higher ($P<0.01$) water infiltration rate in the exclosure treatment in the interval of 10–20 minutes. At a steady state, water infiltration rate of the exclosure was higher ($P<0.05$) than the light grazing. However, the difference between the heavy and light was not significant at a steady state, which was reached in about one hour.

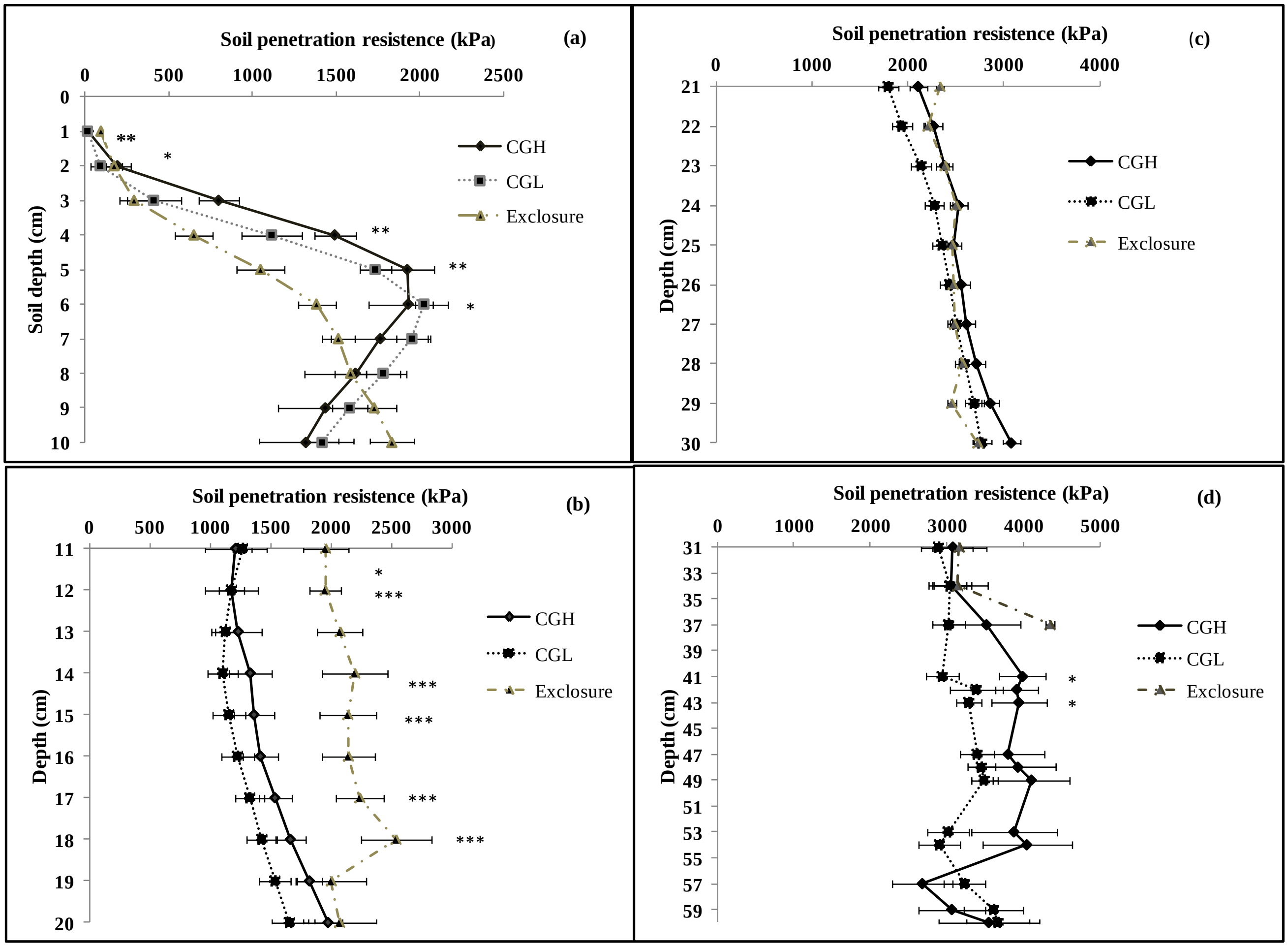


Figure 3.5 Soil penetration resistance in long-term (75 years) as affected by grazing at heavy and light stocking rate and exclosure at 0–10 cm (a), 10–20 cm (b), 20–30 cm (c) and 30–60 cm (d) soil layers in the Middleburg, Eastern Cape, South Africa

CGH, continuous grazing at heavy stocking rate; CGL, continuous grazing at light stocking rate

Bars represent the standard error of the mean

**P*<0.05

***P*<0.01

****P*<0.001

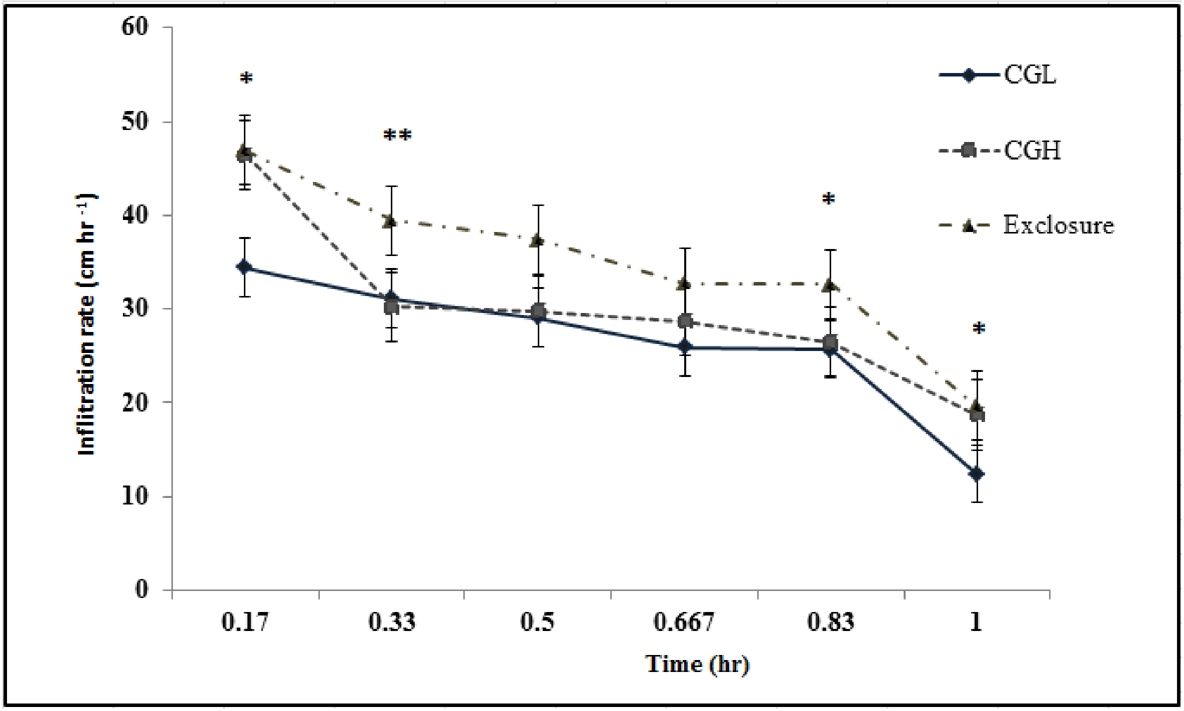


Figure 3.6 Infiltration rate as affected by grazing at heavy (CGH) and light (CGL) stocking rate and Exclosure in the Middleburg, Eastern Cape, South Africa.

CGH, continuous grazing at heavy stocking rate; CGL, continuous grazing at light grazing rate. *Bars* represent the standard error of the mean (n=90)

*P<0.05

**P<0.01

3.5. Discussion

3.5.1. Impacts of grazing on soil carbon and nitrogen storage

The stocking rates practised in the study were generally lower than the carrying capacity norm (small stock unit (SSUha^{-1}) for the grassland-Karoo mixed-rangeland region of South Africa (Kotzé et al. 2013) and other arid regions (Henry 2010). However, the long-term cumulative effects of continuous grazing at light and heavy stocking rates had negative effects on C and N storage. The C storage (0.09 to 0.13 Mg ha^{-1} yr^{-1}) obtained in the study is within the range reported for similar management situations under rangeland conditions (Schuman et al. 2002; Henry 2010). The C concentration of the exclosure in the current study is consistent with other reports (Witt et al. 2011). The reduced C and N storage could be because in dry and arid regions, biomass production is limited by climatic constraints. Continuous grazing puts additional pressure on ecosystems of such fragile soils with low vegetation cover. On the other hand, exclosure as a method of land restoration is said to increase SOC sequestration through higher aboveground and belowground biomass and species richness (Cheng et al. 2011; Wang et al. 2014a).

The C stock and C : N ratios obtained from the exclosure (that is, 75 years of livestock exclusion) in this study concur with the reports of Witt et al. (2011). On the other hand, long-term continuous grazing reduced C and N stocks through modifying species composition and richness, because some native plant species become locally rare under heavy grazing pressure (Du Toit et al. 2011). For instance, palatable Karoo bushes such as *Felicia muricata*, *Salsola calluna* and *Walafrida geniculata* decreased, and in some cases disappeared, under heavy grazing pressure (Mucina & Rutherford 2006; Du Toit et al. 2011). Continuous grazing at a heavy stocking rate, influences SOC storage by redistributing soil C (Yong-Zhang et al. 2005; Chen et al. 2012) and modifying the ecosystem structure and function through its influences on input and output pathways of C and N (Pineiro et al. 2010). However, the impact of grazing on C and N was not the same and was context specific. That is, it varied depending on climate (precipitation, drought), soil type and management practices (McSherry & Ritchie 2013).

In the study, the absence of rest periods, continuous removal of vegetation before plant establishment, and trampling by animals during the early growing and wet seasons resulted in lower SOC and N stocks. Heavy and moderate grazing reduced the peak standing by up to 40% (Derner & Schuman 2007), and created fewer plant residues returning to the soil in sandy soils (Du Toit et al. 2011). On other hand, soil compaction (higher BD) reduces the oxygen content and affects decomposition (Brevik 2002; Barger et al. 2004). Reports indicated that reduced root growth in the presence of grazing pressure might affect the rooting biomass contribution to the SOM pool (Reeder et al. 2004; Li et al. 2011). Reduced root allocation usually influences soil carbon inputs and decreases nitrogen retention within the soil (Li et al. 2011). The results of the study confirm previous reports of positive effects of grazing exclusion on soil nutrients (Witt et al. 2011; Evans et al. 2012).

The higher N in the exclosure was attributed to the higher rate of return of crop litter to the soil. On the other hand, the N input from faeces of animals was small, and its contribution to N concentration was insignificant owing to the very low stocking rate practised in the study area. The findings agree with other reports (Stewart & Frank 2008; Witt et al. 2011), which found greater soil C and N stocks in the exclosure in the arid regions (Wu et al. 2010). Nitrogen availability in arid regions is a critical indirect control of C and N, due to the strong relationship between these two soil nutrients, by increasing and decreasing cumulative N inputs and output pathways (Pineiro et al. 2010). However, these pathways are controlled by a number of factors and their interactions, validating tight and complex relationships between soil C and N stocks of the grassland ecosystem.

3.5.2. Grazing effects on soil properties

Soil health in terms of physical and chemical properties depends on plant and soil management. As physical soil qualities, BD and soil compaction are good indicators, because of the strong correlation with soil processes (Evans et al. 2012). In the grazing exclosure, lower BDs were explained partly by the higher rate of biomass recycling back to the soil, leading to better macro porosity, and thus water-holding capacity (Gill 2007; Witt et al. 2011). BD and particle size fractions control the dynamics and turnover of soil organic matter. These dynamics influence C and N storage and recycling (Du Toit et al. 2011; Kotzé et al. 2013; Materechera 2014). Evans et al. (2012) showed that continuously grazed sandy soils have poor vegetation cover because they are prone to destruction of the topsoil structure, leading to soil coarseness and higher BD. The absence of trampling by livestock leads to the formation of macro-aggregates in which SOC can be stabilized and sequestrated in the long term (Li et al. 2011). According to Kotzé et al. (2013), losses of soil C correlated with losses of macro aggregates and the organic matter stored in them after heavy grazing.

SOC is thus more likely to be accumulated and stabilized under minimal disturbance by animals, which increases nutrient retention and recycling compared with grazed treatments. This was confirmed by a positive relationship among soil parameters, namely CEC, C, N and Ca contents in the exclosure, which is consistent with other studies (Wu et al. 2010). Soil pH was lower in the exclosure, due to high organic matter, which concurs with other studies (Du Preez et al. 2011a; Kotzé et al. 2013), which reported lower pH in the exclosure compared with continuously grazed lands.

3.5.3. Grazing effects on soil compaction and water infiltration

The higher compaction and lower water infiltration in the continuous grazing treatments are attributed mainly to long-term animal impacts, which is in agreement with studies in the dry savanna and Karoo regions of South Africa (Kotzé et al. 2013) and other grassland ecosystems (Evans et al. 2012). However, in the current study, the higher compaction in the exclosure as depth progresses (11–20 cm) was attributed mainly to inherent parent material of the soil. Soil compaction influences soil quality and processes, water-holding capacity, air circulation and biological activities (Gill 2007; Evans et al. 2012).

Compaction usually increases soil erosion under grazing, and decreases the water available to plants. Less pore space can limit gas exchange and reduce root growth. The heavy grazing treatment in the study reached the critical limit (2000 kPa) for root growth described by Donkor et al. (2002) at about 11 cm. That means continuous heavy grazing enhanced soil compaction and reduced the infiltration rate, which affects soil-water content, aeration, and the temperature of the soil system (Brevik et al. 2002; Gill 2007), further influencing nutrient retention and recycling. The higher water infiltration rate of the exclosure treatment on the top soil could be explained by lower soil compaction associated with higher organic matter residues in the absence of grazing. On the other hand, livestock exclusion enhances soil water infiltration and water retention capacity by increasing soil moisture (Gill 2007; Chen et al. 2011; Witt et al. 2011). This finding is in agreement with previous studies in the arid region of South Africa (Kotzé et al. 2013; Materechera 2014; Metzger et al. 2014) and other regions (Michunas & Laueroth 1993; Donkor et al. 2002). The soil nutrient improvement in the exclosure treatment might have resulted in higher soil C stocks, and is related mostly to higher plant biomass and litter accumulation on the soil surface, which improved water-holding capacity of soil, aeration, and retention and recycling of soil nutrients.

3.6. Conclusion

The study examined long-term impacts of grazing (contrasting stocking rate) on C and N sequestration and other soil properties, including soil penetration resistance and water infiltration rate in the Karoo biome of the arid Eastern Cape, where rainfall is extremely seasonal and variable. The results showed that continuous grazing demonstrated lower C and N concentration, and C : N ratios, compared with exclosure. Grazing had greater soil BD and compaction in the top soil layer and changed the Ca, K, CEC and pH of the soil. The results suggest that livestock exclusion in such sandy soils of arid regions is a feasible conservation-oriented management option for carbon sequestration and land rehabilitation through improved plant-soil system. However, from the perspective of resource utilization, exclosure (1–2 years' rest), followed by three years' consecutive grazing at light stocking would improve C and N, and optimize returns in terms of livestock products, ecosystem services and functions. Investigation of time of forage utilization is required to strengthen these results to provide a notion for environmental policy makers with adequate information on strategies of managing vulnerable soil across seasons in fragile, Eastern upper karoo sandy soils of South Africa.

CHAPTER 4

Long-term impacts of season of grazing on soil carbon sequestration and selected soil properties in the arid Eastern Cape, South Africa

4.1. Abstract

The grassland and Karoo biomes of South Africa are major feed resources for livestock farming, yet soil nutrient depletion and degradation are major problems. The objective of this study was to assess the impacts of long-term (>75 years) grazing on rangeland in spring (SPG), summer (SUG), winter (WG), and exclosure (non-grazed control) treatments on soil nutrients, penetration resistance and infiltration tests. Soil samples were collected to a depth of 60 cm to analyse bulk density (BD), soil physical and chemical parameters as well as soil compaction and infiltration. Generally, grazing treatments reduced soil organic carbon (SOC) stocks and C : N ratios, and modified soil properties. There was higher SOC stock (0.128 Mg ha^{-1} yr^{-1}) in the exclosure than in the SPG (0.096 Mg ha^{-1} yr^{-1}), SUG (0.099 Mg ha^{-1} yr^{-1}) and WG (0.105 Mg ha^{-1} yr^{-1}). The C : N ratios exhibited similar patterns to that of C. Among grazing treatments, the WG demonstrated 7% to 10% additional SOC stock over the SPG and SUG, respectively. The SOC stock was found to be higher by 20–23% at 0–30 cm than the 30–60 cm soil layer. Short-period animal exclusion could be an option to improve plant nutrients for conservation-oriented rangeland management in the Eastern upper karoo sandy soils of South Africa. However, this may require a policy environment that supports stock exclusion from areas vulnerable to land degradation, nutrient and soil C losses by grazing-induced vegetation and landscape changes.

Keywords: arid ecosystem, exclosure, organic matter, soil organic carbon, total nitrogen, soil properties

4.2. Introduction

The Karoo biomes of South Africa are major feed resources for livestock farming. In the arid to semi-arid areas of the country, more than 75% of the land is used for livestock production (Smet & Ward 2006), and is the foundation of the country's dairy, beef and wool industries (Du Toit et al. 2011). South African soils are generally characterized by low SOC contents due to land deterioration (Kotzé et al. 2013), vegetation removal by stock and burning, variability in soil types and extreme climate variability (Du Preez et al. 2011a). In such arid regions, rangeland management and protection from degradation is of utmost importance to improve soil organic matter, and thereby SOC storage.

In many ecosystems, livestock grazing has been shown to affect plant diversity, soil C, N, and ecosystems functioning (Franzluebbers 2005). However, the effect of grazing on C and N stocks has been variously documented in the literature. Grazing management practices have been shown to increase soil C (Reeder et al. 2004) and N concentrations (Liu et al. 2012), have no effects (Barger et al. 2004), and decrease soil C (McSherry & Ritchie 2013) and N levels (Steffens et al. 2008). This variation is due to the disparities in climate, soil type and landscape management (Lal 2002) and type of plant community (Ampleman et al. 2014).

In the vast arid grassland and Karoo biomes of South Africa, like most African countries, little is known about the impacts of long-term grazing strategies on soil C dynamics, as grazing practices would shed light on soil C dynamics in these ecosystems, resulting in alteration of the physical and chemical characteristics of soils. There is limited information on factors controlling the spatial and temporal variability of SOC stocks through C decomposition and dynamics, and other soil properties under different grazing managements (Kotzé et al. 2013). Although season and frequency of grazing are manipulated to achieve desired management goals in grazing lands (Evans et al. 2012), little has been documented about the long-term effects of seasonal grazing management on C and N dynamics in the arid Karoo biome. Moreover, due to multifaceted interactions between C and N in the Karoo ecosystem, grazer effect on SOC is highly context specific

(McSherry & Ritchie 2013) and site-specific information is essential for various management scenarios (Derner & Schuman 2007; Du Preez et al. 2011a).

The aim of this study was to assess the impacts of long-term seasonal grazing management and exclosure (non-grazed control) on soil C, N and other soil parameters. It was hypothesized that long-term seasonal grazing managements would not have a significant effect on soil C and N contents and other soil properties compared with exclosure, but would have additional benefits in terms of sustainable animal production and ecosystem services.

4.3. Materials and methods

This study was conducted at the long-term experimental site of Grootfontein Agricultural Development Institute (GADI), Eastern Cape, South Africa. The study site is 122.4 ha and located at latitude 31^{o} 22`E, longitude 24$^{o}$ 45`N, with an altitude of 1260 m above sea level. The area lies within 100 mm to 500 mm rainfall regions (arid zone), with higher coefficients of variation (>40%) as rainfall decreases. The long-term mean annual and median rainfall are 370.2 and 359 mm, respectively. The rainfall (southern hemisphere seasons) distribution is 15% spring (September–October), 30% summer (October–March), 50% autumn (April–May), and 5% winter (June–August). The average January temperature is 20.9 ^{o}C, while the average July temperature is 7.9 ^{o}C (see Figure 3.1) (Du Toit et al. 2010).

The Karoo biome is grassy shrubland, the third largest biome in South Africa, covering about 20.5% of the land area (Acocks 1998; Mucina and Rutherford 2006; Du Toit et al. 2011). The underlying geology is varied because distribution of the biome is determined primarily by rainfall. Climatologically and biologically, the Karoo is a heterogeneous and ecotonal region. It is an ideal biome for sheep and goat production. In the main river valleys, people farm olives, citrus and deciduous fruit. Grass species in the Eastern upper karoo such as *Aristida diffusa* (iron grass), *Eragrostis curvula* (weeping lovegrass) and *Hyparrhenia hirta* (common thatch grass) increased in the SPG and SUG treatments.

Summer grazing rapidly increased the relative abundance of dwarf shrubs over the study years. Most of the grasses are of the C-4 type and, like the shrubs, are deciduous in response to rainfall events. On the other hand, grass species such as *Themeda triandra* (red grass), *Sporobolus fimbriatus* (drop seed grass), *Digitaria eriantha* (finger grass) and *Cymbopogon pospischilli* (turpentine grass) were commonly found in the WG and exclosure treatments (Du Toit 2010; Du Toit et al. 2011). Karoo bushes (shrubs) such as *Pymaspermum parvifolium, Felicia murcata, Salsola Caluna, Walafrida geniculata were dominant in the study area.*

The predominant soil type, over 80% of the karoo area, is a lime-rich weakly developed shallow (<30 cm) soil over rock. Generally, the soils of the study site are of the Shigalo series aridisols (Soil Survey Staff 1999), categorized under textural class sandy-clay-loam. Although less than 5% rain reaches the rivers, the high erodibility of soils poses a major problem where overgrazing occurs. The soil colour of the study site was identified as 7.5YR, 5YR, 10YR and 5YR in the SPG, SUG, WG and exclosure, respectively.

Merino wethers at a stocking rate of 2.35 SSU ha^{-1} were used for the season of grazing treatments, with individuals being replaced with young animals after three to four years. During day hours (9:00 to 17:00) animals were allowed free grazing on rangeland, and were not supplemented with other feeds throughout the study period, because the initial objective of the study was to evaluate wool yield and animal performance of growing sheep under veld conditions. Except minimal and localized urine and faeces (estimated to be less than 3 kg N ha^{-1} yr^{-1}) (Oenema et al. 2005) returned to the soil in grazing treatments, inorganic fertilizers were not applied on the rangeland during the study period. The study started in 1934 and terminated in 2010/2011. Soil samples were taken between May and November 2012.

The treatment consisted of grazing by Merino sheep at various grazing seasons, including i) SPG (spring grazing by sheep from August–December), ii) SUG (summer grazing by sheep which is from December–April), iii) WG (winter grazing by sheep from April–August) (grazing for about 75 years), and iv) an exclosure, which has not been grazed for more than 75 years. The plots were laid out in parallel rectangular strips (width/ length ratio of approximately 1:10) of 25.5 ha (seasonal treatments) and 3.4 ha (exclosure) along the gentle sloping Karoo rangeland. Five parallel transects, 100 m apart and perpendicular to the long axis of each plot, were delineated. Soil samples, at least 15 m from the edge of the plots, were collected every 20 m along each transect (refer chapter 3 figure 3.2 and 3.3).

To determine C, N, soil texture and soil chemistry, soil was sampled until 60 cm depth in 10 cm steps (0–10 cm, 10–20 cm, 20–30 cm), and 30–60 cm (pooled because of bedrock below 30 cm depth in some plots). Five samples were taken from each sampling sub-plot for grazing treatments and three sampling sub-plot for exclosure along transects, and 496 soil samples were collected. For each transect, samples were pooled per layer and mixed to make a single homogenous soil sample per layer. To determine BD, undisturbed soil cores (5.8 cm diameter) were collected using a core sampler with the same sampling procedure as on the transects. Soil water contents were measured from BD soil core samples. Soil penetration resistance was measured at 1 cm intervals down to a depth of 60 cm using a Rimik™ CP20, with a total of 2800 readings being taken: 747 from SPG, 776 from the SUG, 862 from the WG and 415 from the exclosure. Infiltration rate readings (n = 150) were collected every ten minutes for an hour, using a mini double ring infiltrometer (15 cm inner and 30 cm outer diameters) until it reached a steady state.

BD was determined on intact soil cores by drying the soil at 105 $^{\circ}$C for 24 hr and dividing the oven dry mass by the volume of core sampler. Soil particle sizes were analysed using the wet sieving and sedimentation method, following the standard procedure of the Non-Affiliated Soil Analysis Work Committee (1990). The samples were air dried and ground to pass through a 150-µm screen and analysed for total C and N using a Carlo Erba NA1500 C/N analyser (Carlo Erba Strumentazione, Milan, Italy). All other chemical analyses were undertaken in duplicate following the procedure of the Non-Affiliated Soil Analysis Work Committee (1990): pH (1:2.5 soil to water suspension), exchangeable Calcium (Ca), Potassium (K), Magnesium (Mg) and Sodium (Na) (1mole dm^{-3} NH_4OAC at pH 7). Since the SOC content varied along the soil depths, due to differences in BD at various soil profiles, SOC stock (kg m^{-2}) and TN stock (kg m^{-2}) were estimated after Bagchi and Mark (2010) as follows:

$$SOC = \sum_{i=1}^{n} D_i \rho_i SOC_i \quad , \qquad TN = \sum_{i=1}^{n} D_i \rho_i TN_i$$

where SOC is soil organic C stock (Mg ha^{-1}) and TN is total N stock (Mg ha^{-1}); i is the soil layer number,(i=1, 2, 3 and 4); D_i is the depth interval (cm); ρ_i is the bulk density (g cm^{-3}) in soil layer i; SOC_i is the mean SOC concentration (g kg^{-1}) in the soil layer i; N_i is the mean N concentration (g kg^{-1}) in the soil layer i.

All analyses were tested for normality using the UNIVARIATE procedure (SAS Institute 2008). The effects of season of grazing, soil depth and their interactions on soil C, N, C : N ratio and other soil parameters were examined using analysis of variance with the MIXED model (PROC MIXED SAS Institute Inc. 2008). Season of grazing, soil depth, and their interactions were fixed effects, while transect and transect by grazing management were random effects. Where necessary, the data were transformed using logarithmic transformation. When treatment effects were significant (P<0.05), the means were separated using Tukey's test. Pearson correlations were used to analyse the trends of soil parameters (PROC CORR SAS Institute Inc. 2008).

4.4. Results

4.4.1. Season of grazing on soil carbon and nitrogen storage

The results indicated that the long-term (about 75 years) cumulative effects of season of grazing on soil C and N concentration varied considerably (Table 4.1). For all treatments and individual soil cores, C concentration ranged from 1.9 to 11.2 g kg^{-1} (mean: 5.1), while N concentration ranged from 0.2 to 1.4 g kg^{-1} (mean: 0.6) and C : N ratios from 3.70 to 17.16. Season of grazing influenced soil C concentration and C : N ratios, with the lowest values being found in SPG. There was higher (P<0.001) C concentration in the exclosure than in the grazing treatments, while the differences for C : N ratios were only marginally significant (P=0.053). There was interaction effect of season of grazing by soil depth on C contents (P=0.048).

Table 4.1 Soil organic carbon and total nitrogen concentration and carbon : nitrogen ratios (mean ± SEM) as influenced by grazing management (GM) and soil depth (SD) in the Middleburg, Eastern Cape, South Africa

Fixed effects		Soil parameters		
		SOC (g kg^{-1})	Total N (g kg^{-1})	C : N
GM		***	NS	*
	SPG	4.8 ± 0.15^{c}	0.57 ± 0.20	8.50 ± 0.22^{b}
	SUG	4.7 ± 0.15^{c}	0.58 ± 0.20	8.57 ± 0.23^{b}
	WG	5.2 ± 0.15^{b}	0.60 ± 0.20	8.71 ± 0.24^{b}
	Exclosure	6.2 ± 0.20^{a}	0.64 ± 0.30	9.83 ± 0.30^{a}
	F-test	9.826	1.850	2.600
	P-value	<0.001	0.139	0.053
SD (cm)		***	**	***
	0-10	5.34 ± 0.13^{a}	0.54 ± 0.02^{b}	10.05 ± 0.21^{a}
	10-20	5.40 ± 0.13^{a}	0.58 ± 0.02^{b}	7.76 ± 0.22^{c}
	20-30	5.38 ± 0.14^{a}	0.73 ± 0.02^{a}	8.32 ± 0.22^{b}
	30-60	4.15 ± 0.15^{b}	0.50 ± 0.02^{c}	8.67 ± 0.40^{b}
	0-60	5.29 ± 0.26^{a}	0.61 ± 0.03^{b}	8.67 ± 0.40^{b}
	F-test	11.261	20.566	15.795
	P-value	<0.001	<0.001	<0.001
GM *SD		*	NS	NS
	F-test	0.877	2.108	1.515
	P-value	0.048	0.107	0.08

Mean values in each column for each parameter followed by different letters are statistically different at $P \leq 0.05$; NS, P>0.05; *P<0.05; **P<0.01; ***P<0.001. SOC, soil organic carbon; N, nitrogen; SEM=standard error mean; SPG, spring grazing; SUG, summer grazing, WG, winter grazing

The C and N stocks and C : N ratios demonstrated great variations, depending on season of grazing and soil depths (Figure 4.1). There were higher C stock (P<0.001) and C : N ratios

(P<0.001) in the exclosure than in the season of grazing, suggesting that C storage in dry lands could be improved by livestock exclusion. Among the grazing treatments, the C stock was 7% and 10% higher in the WG than in the SPG and SUG, respectively. The N stock differences between the season of grazing and exclosure was in the range of 6%. Generally, the WG treatment resulted in higher C stocks as vegetation were resistant to grazing pressure and plant residues left on the soil. In addition, there was minimum trampling by sheep due to low moisture in the soil.

4.4.2. Distribution of carbon and nitrogen storage in different soil layers

Seasonal grazing demonstrated visible and consistent effects on C and N concentration along the soil layers (Table 4.1). Generally, C and N concentrations and C : N ratio were higher in the top 0–30 cm soil layer but tended to decrease with depth. The C content in the top soil layers (0–10 cm, 10–20 cm, and 20–30 cm) was higher (P<0.001) than in the 30–60 cm layer. Similarly, N concentration in the 20–30 cm soil layers was higher (P<0.001) than in the other soil layers. At the same time, N concentration at 0–10 cm and 10–20 cm was higher than the 30–60 cm soil layer. There were higher proportions of C (63.1%) and N (78.7%) in the top 0–30 cm, where litter turnover is usually high, which signifies that C and N in the Karoo biome are vulnerable to losses through C oxidation, ammonia volatilization, N mineralization and runoff. There were higher C : N ratios (P<0.001) in the top 0–10 cm than in the 30–60 cm soil layer, but they were lowest (7.76) at 10–20 cm.

Soil C and N stocks, and C : N ratios were generally higher in the top 0–30 cm layers, where SOM is usually high, owing to plant residue and litter cover (Figure 4.1). There was increased (P<0.001) C stock in the top soil layers, which was higher by 24.1, 26.1 and 28.7% at 0–10, 10–20 and 20–30 cm, respectively, compared with the 30–60 cm soil layer. N stock (P<0.001) and C : N ratios (P<0.001) differed at each soil layer, suggesting great variations according to soil layer. The highest N stock (0.12 kg m^{-2}) was found at 20–30 cm, with the least (0.08 kg m^{-2}) at 30–60 cm. Likewise, the highest (10.1) C : N ratios was found at 0–10 cm, with the least (7.7) at 20–30 cm. Grazing management-by-soil depth interaction was significant in N stock (P<0.01) and C : N ratios (P<0.05). Livestock exclusion increased N stock and C : N ratios, but decreased at soil layers >30 cm depths (Figure 4.1).

4.4.3. Soil physical parameters

Grazing management practices were shown to influence the clay content ($P<0.001$), while they had non-significant ($P>0.05$) effects on silt and sand contents (Table 4.2). Clay content was 48% lower in the exclosure than in SPG, but it was 37% higher in the SUG than in the exclosure. There was no evidence for differences in soil texture in various soil layers. However, the BD was lower ($P<0.05$) in the exclosure and WG than in the SUG and SPG (Table 4.2). Higher BDs were found when the paddocks were grazed early in the growing season (spring) and summer. In the surface soil, lower ($P<0.05$) BD was found compared with 10–20 cm and 20–30 cm soil layers. Likewise, the BD was lower ($P<0.01$) at 10–20 cm and 20–30 cm than 30–60 cm. There was no significant effect of grazing season-by-soil depth interactions on soil physical parameters except for BD ($P<0.01$) (Figure 4.1). BD decreased in the WG and exclosure, but it increased at soil layers >10 cm.

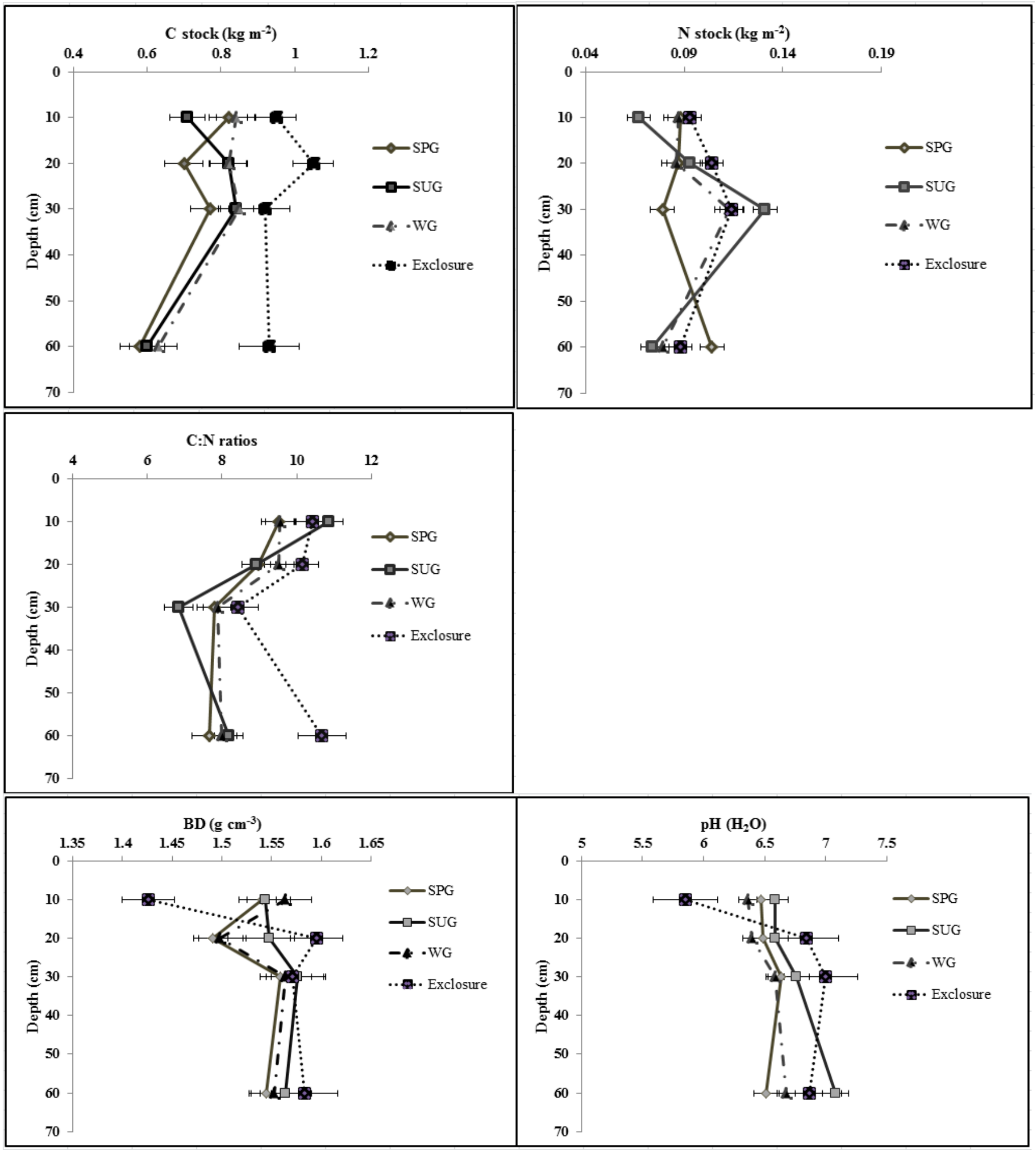


Figure 4.1 Carbon (C) and N stocks, C : N ratios, bulk density and pH (H_2O) as affected by grazing in spring, summer and winter and exclosure in the arid Eastern Cape, South Africa

SPG, spring grazing; SUG, summer grazing; WG, winter grazing

Bars represent the standard error of the mean

Table 4.2 Soil texture and bulk density (mean ± SEM) as influenced by grazing management (GM) and soil depths (SD) in the Middleburg, arid Eastern Cape, South Africa

Fixed effects		Soil texture		Bulk density
	Silt (%)	Clay (%)	Sand (%)	(g cm^{-3})
(GM)	NS	***	NS	*
SPG	10.8 ± 1.72	28.3 ± 1.65^{a}	60.9 ± 1.80	1.55 ± 0.01^{a}
SUG	10.1 ± 1.59	26.1 ± 1.53^{a}	64.3 ± 1.66	1.56 ± 0.01^{a}
WG	13.9 ± 1.69	20.1 ± 1.53^{b}	66.0 ± 1.66	1.52 ± 0.01^{b}
Exclosure	13.8 ± 2.10	19.1 ± 2.02^{b}	64.6 ± 2.20	1.51 ± 0.02^{b}
F-test	1.339	6.945	1.472	1.413
P-value	0.271	<0.001	0.232	0.025
SD (cm)	NS	NS	NS	**
0–10	10.8 ± 1.59	22.8 ± 1.53	66.5 ± 1.66	1.50 ± 0.01^{c}
10–20	14.0 ± 1.59	22.3 ± 1.52	63.8 ± 1.66	1.53 ± 0.01^{b}
20–30	9.50 ± 1.86	25.7 ± 1.79	64.9 ± 1.95	1.55 ± 0.02^{b}
30–60	14.4 ± 1.97	22.9 ± 1.89	60.7 ± 2.06	1.58 ± 0.02^{a}
F-test	1.798	0.772	1.655	3.025
P-value	0.158	0.514	0.187	0.005
GM* SD	NS	NS	NS	**
F-test	1.390	0.370	1.532	3.025
P-value	0.215	0.945	0.160	0.005

Mean values in each column for each parameter followed by different letters are statistically different at $P \leq 0.05$; NS, $P>0.05$; *$P<0.05$; **$P<0.01$; ***$P<0.001$. SEM=standard error mean; SPG, spring grazing; SUG, summer grazing, WG, winter grazing

4.4.4. Soil chemistry

Effects of grazing management were evident in all soil chemical properties considered in this study (exchangeable Ca, K, and Na, CEC and pH) except for the Mg concentration, which was only marginally significant (P=0.055) (Table 4.3). The exchangeable Ca, Mg, and pH generally increased along soil depth until 30 cm with Mg concentration increment extending to 60 cm depth. The Ca concentration was higher ($P<0.001$) in the exclosure compared with the seasonal grazing. Conversely, the K concentration was lower ($P<0.05$)

in the exclosure and WG than in the SPG and SUG. Grazing in the SUG contributed to an increased (26 mg kg^{-1}) Na concentration compared to the SPG and exclosure while the Na concentration in the WG was the least (13 mg kg^{-1}). Likewise, the pH value (6.75) was higher (P=0.01) in the SUG than other seasonal grazing, being least (6.32) in the exclosure. The highest (P<0.001) CEC was found in the SUG and the lowest in the SPG (15.3 cmol kg^{-1}). Except for the K and Na, which showed no variations (P>0.05), soil chemical parameters exhibited great variations along the soil layers (Table 4.3). In general, K concentration showed a decreasing trend, while Na increased as soil depth advanced. There were lower Ca (P<0.01) and Mg (P<0.01) at 0–10 cm and 10–20 cm than 20–30 cm and 30–60 cm soil layers. Conversely, the CEC was higher (P<0.05) at 30–60 cm than 0–10 cm, 10–20 cm and 20–30 cm soil layers. The pH showed an increasing trend as depth advanced. However, statistically it was higher (P<0.01) only at 20–30 cm than at the 0–10 cm soil layer. There was significant grazing-by-soil depth interaction (P<0.001) on pH value (Table 4.3 and Figure 4.1). Soil pH increased in seasonal grazing, but decreased at depth <10 cm.

4.4.5. Association between soil properties pooled along soil layers

Despite the differences in grazing management, soil C was strongly positively correlated (r=0.944, P<0.01) with N stock values along soil layers (Table 4.4), suggesting that factors influencing N stock could strongly affect the C stock as well. However, the C stock value was moderately correlated with the CEC (r = 0.323, P<0.05) and silt (r = 0.250, P<0.05) contents. There was positive, but weak correlation of C stock with the Ca, Mg and BD. On the other hand, the N stock was positively correlated with Mg (r = 0.235, P<0.05), CEC (r=0.297, P<0.05) and BD (r=0.235, P<0.05).

Table 4.3 Selected soil chemical properties (mean ± SEM) as influenced by grazing management (GM) and soil depths (SD) in the Middleburg, Eastern Cape, South Africa

Fixed effects	Exchangeable cations (mg kg $^{-1}$)				pH (H_2O) [a]	CEC(cmol kg^{-1})
	Ca	K	Mg	Na		
GM	***	*	NS	*	**	***
SPG	906± 80[b]	224 ± 15[a]	554 ± 49	17.3 ± 3.0[b]	6.53 ± 0.09[b]	15.3 ± 1.7[c]
SUG	858± 74[b]	234 ± 14[a]	659 ± 45	25.7 ± 2.8[a]	6.75 ± 0.08[a]	32.7 ± 1.6[a]
WG	728 ± 74[b]	197 ± 14[b]	527 ± 45	13.0 ± 2.8[c]	6.50 ± 0.08[b]	25.6 ± 1.6[b]
Exclosure	1496 ± 92[a]	172 ± 18[b]	468 ± 56	20.6 ± 3.4[b]	6.32 ± 0.10[c]	28.4 ± 1.96[b]
F-test	15.432	2.967	2.683	3.737	4.153	19.902
P-value	<0.001	0.039	0.055	0.016	0.01	<0.001
SD (cm)	**	NS	**	NS	**	*
0-10	771 ± 74[b]	222 ± 14	417 ± 45[c]	16.9 ± 2.8	6.32 ± 0.08[b]	25.7 ± 1.6[b]
10-20	952 ± 74[b]	214 ± 14	525 ± 45[b]	18.7 ± 2.8	6.58 ± 0.08[ab]	22.4 ± 1.6[b]
20-30	1159 ± 80[a]	207 ± 15	628 ± 49[a]	18.5 ± 3.0	6.76 ±0.09[a]	25.1 ± 1.7[b]
30-60	1106 ± 92[a]	184 ± 18	640 ± 56[a]	22.6 ± 3.4	6.46 ± 0.10[ab]	29.7 ± 1.96[a]
F-test	5.032	1.018	4.633	0.559	5.052	2.871
P-value	<0.004	0.391	0.006	0.664	0.004	0.044
GM*SD	NS	NS	NS	NS	***	NS
F-test	1.164	0.603	0.305	0.463	4.317	0.680
P-value	0.335	0.789	0.970	0.893	<0.001	0.723

[a]pH in water solution. Mean values in each column for each parameter followed by different letters ([a], [b], [c]) are statistically different at $P \leq 0.05$; NS,$P>0.05$; *$P<0.05$;**$P<0.01$;***$P<0.001$. SPG, spring grazing; SUG, summer grazing; WG, winter grazing; SEM=standard error mean; CEC, cation exchange capacity

Table 4.4 Correlation coefficients (pooled across soil layers and grazing management) of soil parameters in the Middleburg, Eastern Cape, South Africa

	N	Ca	K	Mg	Na	CEC	pH	Silt	clay	Sand	BD
C	0.944**	0.197	−0.222	0.175	−0.181	0.323*	−0.048	0.250*	−0.189	−0.139	0.226
N		0.095	−0.216	0.235*	0.107	0.297*	0.007	0.162	−0.079	−0.151	0.235*
Ca			−0.111	0.268*	0.239*	0.082	0.178	0.109	−0.066	−0.013	−0.136
K				0.237*	−0.020	−0.109	0.033	−0.239*	0.170	0.108	0.142
Mg					0.635**	0.248*	0.337*	−0.193	0.285*	−0.081	0.102
Na						0.239*	0.219	−0.149	0.192	−0.050	−0.078
CEC							0.113	0.005	−0.119	0.089	0.163
pH								−0.131	0.083	0.031	0.232*
Silt									−0.492**	−0.538**	−0.083
Clay										−0.409**	0.083
Sand											−0.047

BD, bulk density; CEC, cation exchange capacity

** Correlation is significant at the 0.01 level

* Correlation is significant at the 0.05 level

4.4.6. Soil penetration resistance

Grazing management practice caused variations in terms of soil compaction measured as penetration resistance of the soil at 1 (P<0.05), 4 (P<0.05), 5 (P<0.01), 6 (P<0.01), 7, 8, 9, and 10 cm depths (P<0.05) (Figure 4.2a). Generally, at the top 0–3 cm and 4–11 cm interval, SUG had higher soil penetration resistance compared with other treatments. At 10–20 cm depth, there was consistently higher soil penetration resistance in the SUG treatment than in the SPG, although these differences were marginally significant (P<0.10) (Figure 4.2b). The differences in terms of soil penetration disappeared at greater than 14 cm soil depth and resumed at depth 50–55 cm (Figure 4.2d). However, owing to the presence of rock outcrops and in some cases shallow parent material, measurements were not taken below 38 cm and sometimes 50 cm for the exclosure and WG treatments, respectively (Figures 4.2c, d).

4.4.7. Water infiltration rate and volumetric water content

Excessive soil moisture or rainfall can lead to nutrient loss through enhanced leaching or runoff. However, an optimal level of soil moisture is required to facilitate nutrient recycling processes in soil. The soil volumetric water content and water infiltration rate varied considerably in the soil layers (Figure 4.3). The exclosure and winter grazing increased the infiltration rate, while generally early wet season (spring) and main rainy season (summer) grazing reduced water infiltration rate (Fig 4.3a). The SWC of the top soils in the WG (20.62%) and exclosure (23.98%) treatments was higher (P<0.001) compared with the SPG (19.34%) and SUG (19.91%) (Figure 4.3b). Below 20 cm depth, the exclosure had consistently higher SWC than grazing treatments. At the steady state, the exclosure and WG demonstrated marginally higher (P=0.051) infiltration rate compared with the SUG and SPG treatments.

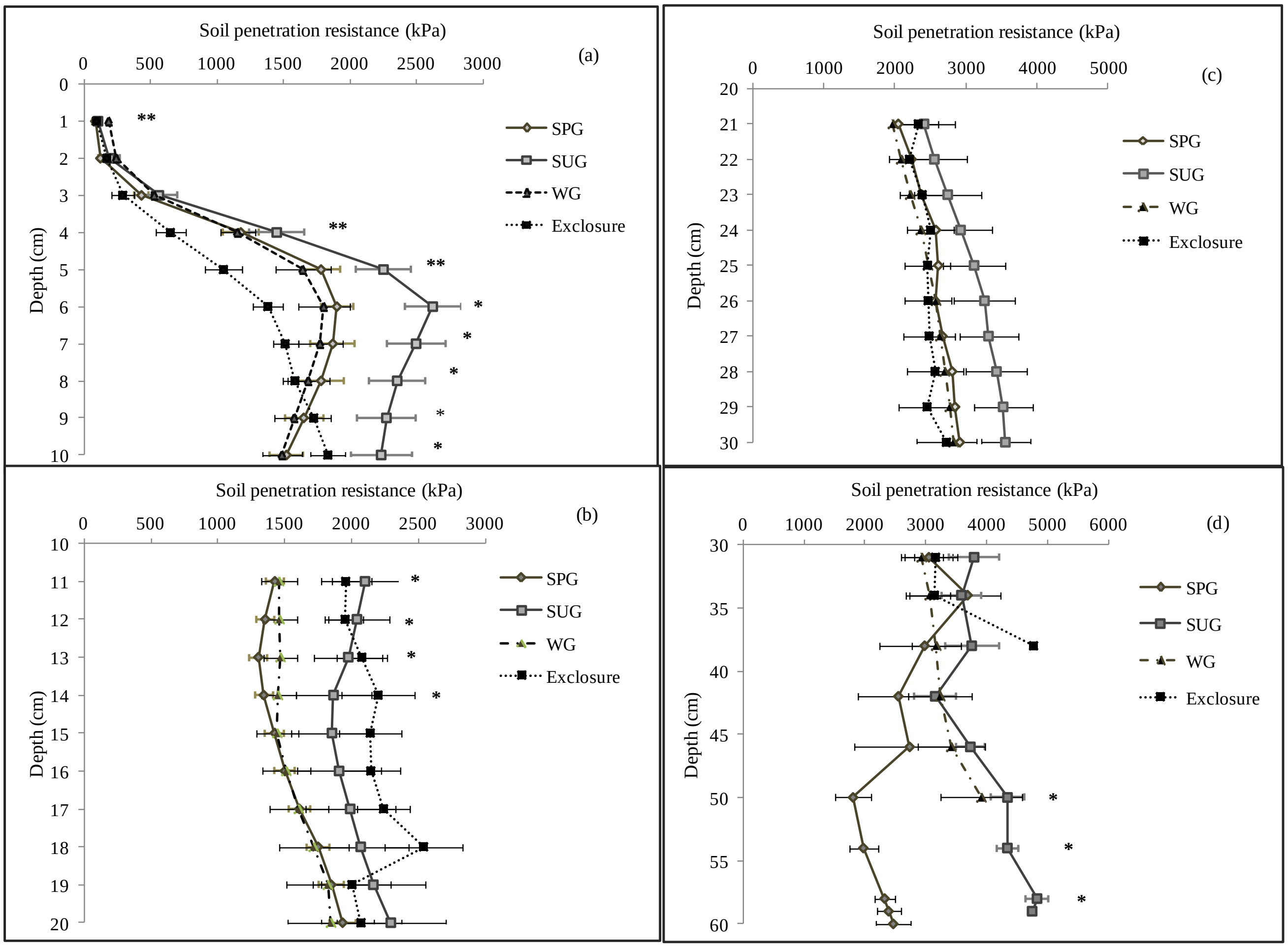


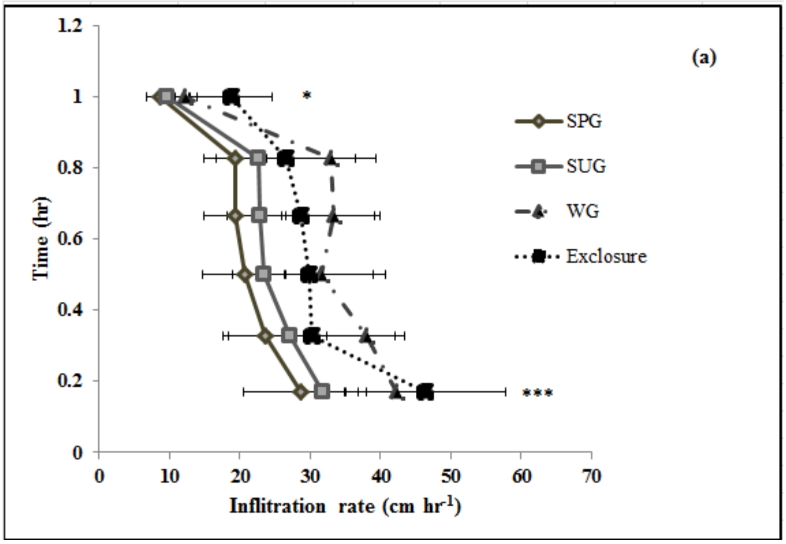
Figure 4.2 Soil penetration resistance (kPa) as affected by spring, summer, and winter grazing and exclosure at a) 0–10 cm; b) 11–20 cm; c) 21–30 cm; and d) 31–60 cm depths in the Middleburg, Eastern Cape, South Africa.

SPG, spring grazing; SUG, summer grazing; WG, winter grazing

Bars represent the standard error of mean

*P<0.05

**P<0.01



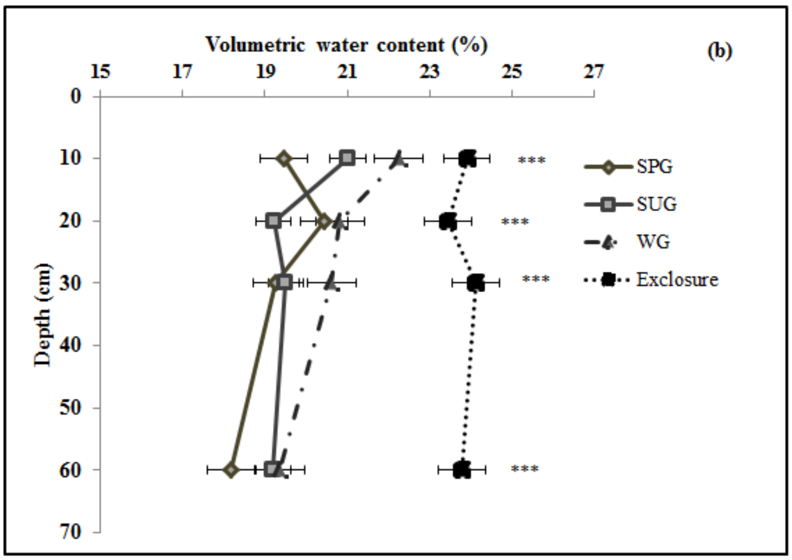


Figure 4.3 a) Water infiltration rate and b) soil volumetric water content as affected by spring (SPG), summer (SUG) and winter grazing (WG), and exclosure treatments in the Middleburg, Eastern Cape, South Africa

Bars represent the standard error of the mean

$^{*}P<0.05$

$^{***}P<0.001$

4.5. Discussion

4.5.1. Influences of season of grazing on soil carbon and nitrogen storage

The results of the study indicated that long-term cumulative effects of seasonal grazing practices considerably influenced soil C and N stocks. The soil C and C : N ratios were consistently higher in the exclosure. Thus with no disturbances in the long term, C would be expected to increase until it reached equilibration (Derner & Schuman 2007; Wilson et al. 2012). Consequently, more C (29.3–33.3%) was stored in the exclosure than in seasonal grazing, which was at a rate of 0.096–0.128 Mg ha^{-1} yr^{-1}, on average, for the entire period. The results are in the range (0.02–0.12 Mg ha^{-1} yr^{-1}) reported for arid rangelands (Lal 2000). The C and N stock obtained in this study was closer to the average for Africa (Beukes & Cowling 2003; Kotzé et al. 2013), but smaller than reports elsewhere in the world (Evans et al. 2012; Liu et al. 2012).

In terms of grazing time, the results show that WG (0.79 kg C m^{-2} compared with 0.96 kg C m^{-2} of the exclosure) seemed to be a good compromise with tangentially higher soil C stocks than the SPG (0.74 kg C m^{-2}) and SUG (0.72 kg C m^{-2}). This is partly associated with better soil conditions and a soil surface covered by vegetation in winter, which provides root density and depth, and increases tolerance to drought (Cheng et al. 2011). First, the reduced C in the SPG and SUG treatments could be due to removal of aboveground plant biomass by animals early in the growing season. Second, reduced root growth in the presence of grazing, might affect the rooting biomass contribution to the SOM pool (Reeder et al. 2004; Briske et al. 2008; Liu et al. 2012). Reduced root allocation usually influences soil C inputs and decreases N retention in the soil (Derner et al. 2006). These results confirm the findings of Evans et al. (2012), who reported reduced SOC when lands were grazed during the plant growing season (spring to summer), due partly to increased soil compaction. On the other hand, in the WG treatment, vegetation was usually grazed after it had fully matured and, as a result, plants were resistant to grazing pressure (Du Toit et al. 2011). The higher SOC concentration in the top soil layers was attributed mainly to the magnitude and proportion of fine root mass (Lal 2000; Derner et al. 2006) and litter accumulation (Liu et al. 2012). On the other hand, Beukes and Cowling (2003) reported reduced SOC in the top soils (0–10 cm) in non-selective grazing at a heavy stocking rate. Estimating an annual C storage since the last soil sampling (C difference divided by 75 years), the C stock rate of the exclosure was in the range of 0.128 Mg C ha^{-1} yr^{-1} compared to 0.105 for WG, 0.099 for SUG and 0.096 Mg C ha^{-1} yr^{-1} for SPG, respectively. Livestock exclosure, as a method of land restoration, increases C sequestration through higher aboveground and belowground phytomass and species richness (Liu et al. 2012). The results agree with other reports (Cheng et al. 2011; Witt et al. 2011) that higher residues return from higher primary productivity contribute to increased soil C and N stocks. In terms of grazing, SOC storage varies, depending on whether the grazing pressure exceeds the carrying capacity of the area and season beyond a given threshold (Cheng et al. 2011). This threshold depends mainly on vegetation type, as C_4-dominated grasslands differ so strongly in their response of SOC to grazing (McSherry & Ritchie 2013).

The N stock in the exclosure was higher by 9.9–16.3% than seasonal grazing, and most (70%) of this stock was in the top soil layers. This could be due to a function of root distribution and hence senescence for a long period. Grazers return significant proportions of ingested N through their faeces and urine (Oenema et al. 2005), which may increase N inputs under heavy grazing conditions. However, because of light grazing pressure in the study, the N differences associated with animal (sheep) faeces and urine leftovers were small (estimated to be less than 3 kg ha^{-1} yr^{-1}). According to Steffens et al. (2008), the higher N in the livestock-excluded site is due to the presence of accumulated aboveground and belowground biomass, plant litter and crop residues, which may have contributed to the higher soil organic matter, a reservoir of soil C and N (Pineiro et al. 2010). Nevertheless, these results confirm that in arid regions, a slightly higher C : N relationship can be maintained through livestock exclusion. Improvement in N provides the potential for C sequestration (Pineiro et al. 2010), land rehabilitation and restoration (Derner & Schuman 2007). Although grazing affected N through removal of aboveground biomass, regardless of these practices, there was higher N stock in the uppermost (0–30 cm) soil layers, implying that the amount and turnover of N in the top soils is usually high (Derner et al. 2006), and at the same time, vulnerable to losses through ammonia volatilization and run-off.

4.5.2. Season of grazing effects on soil properties

Soil health in terms of physical and chemical properties depends mainly on plant and soil management practices. Bulk density and soil compaction are good indicators of soil quality because of strong correlation with soil processes (Mikhailova et al. 2000; Murphy et al. 2004; Evans et al. 2012) that influence aggregate stability, microbial process, nutrient retention and/ or nutrient recycling (Neff et al. 2005) and C sequestration. The higher clay content in the SPG and SUG did not show a corresponding higher C and N stocks, which may be due to light textured clay (<35%) content. Such soils have inherent physical problems such as hard setting, sodicity and low organic carbon levels (Chan et al. 2003). These may be due to a frequent disruption of aggregates by animals because soils were wet during SPG and SUG grazing (Six et al. 2000). Contrary to these findings, McSherry and Ritchie (2013) reported positive effects of grazing on SOC at lower precipitation and finer soils with higher clay (>35%) content. For exclosure, higher Ca and lower BDs were explained partly by a higher rate of biomass recycling back to the soil, leading to better macro porosity, and thus water-holding capacity (Donkor et al. 2002; Witt et al. 2011). The absence of trampling by livestock leads to the formation of steady macro aggregates in which SOC has been stabilized and sequestrated in the long term (Six et al. 2000; Kotzé et al. 2013). Indeed, exclosure showed higher CECs and nutrient retention compared with grazed treatments.

The increasing trend in Ca, Mg and CEC as soil depth advanced could be due to leaching of nutrients to lower layers owing to sandy soils. In arid environments, because of limited biomass, livestock production affects SOC owing to the amount and composition of litter inputs to the soil system, and in turn affects soil moisture and physical properties (Witt et al. 2011). Long-term seasonal grazing and exclosure could modify soil nutrients in the top soil layers. According to Derner et al. (2006), the magnitude and distribution of fine root mass in the top soil profile is a principal driver that mediates the effect of community composition in the biogeochemistry of grassland ecosystems.

4.5.3. Season of grazing on soil compaction and water infiltration

For seasonally grazed treatments, severe grazing on immature vegetation is often associated with higher soil compaction through animal trampling. For the Karoo region, Beukes and Cowling (2003) reported that unlike grazing early in the plant-growing season (spring), winter grazing soils are less sealed and compacted, as lands are covered by mature vegetation, similar to the current findings. Others (Bouwman & Arts 2000; Donkor et al. 2002; Evans et al. 2012; Kotzé et al. 2013) reported similar effects. However, the extent of compaction is dependent on soil moisture, inherent soil particle distribution, soil type, stock class and stocking rate (Beukes & Cowling 2003). Factors such as spatial arrangement of grazing related to the distribution of water points (Kotzé et al. 2013), landscape position and variability of inherent soil parent material may contribute to the compaction of the soil.

Of the seasonal grazing treatments, WG showed the lowest soil compaction due mainly to reduced soil moisture content in winter, because soil compaction increases under optimal soil water conditions (Hamza & Anderson 2005). The freeze-thaw cycles in winter might have contributed to lower compaction for grazing (Donkor et al. 2002). The higher water infiltration rate and soil moisture content in WG is often associated with low surface run-off, probably owing to an increase in SOM, which enhances porosity and water retention. Under WG and long-term grazing exclusion, soil moisture is usually higher because of better vegetation cover (Witt et al. 2011). The implication is that land restoration through livestock exclusion is of the utmost importance because stock farming is practised in most of South Africa (Du Preez et al. 2011a; Kotzé et al. 2013).

4.6. Conclusion

Findings from this study indicated that grazing reduced soil C stocks and C : N ratios, and altered the magnitude and allocation of soil nutrients. The SPG and SUG increased soil compaction, while it reduced the water infiltration rate, nutrient and water retention, and nutrient recycling. Long-term accumulation of SOM in the exclosure, however, generally resulted in improved soil C and C : N ratios, as well as other soil nutrients. Exclosure and WG showed higher volumetric water content and infiltration rates. Among the treatments, WG is a promising alternative management option to optimize livestock production, while managing soil nutrients and other ecosystem services and functions. The results based on season of grazing suggest that stock exclusion (soil and vegetation resting time of 3–5 years) would be a viable management option to improve SOM and critical nutrients in the sandy soils of Eastern upper karro areas of South Africa. However, this may require environmental policy that supports stock exclusion from such areas vulnerable to land degradation, nutrient and soil C losses through grazing-induced vegetation and landscape changes. Grazing management, along with changes in precipitation, influences not only belowground soil nutrients but it could primarily affect aboveground plant biomass. Therefore, investigating aboveground plant biomass would reinforce the current findings because it has strong relationship with belowground plant nutrients. To achieve the stated objective, effects of change in precipitation and defoliation interval on herbage yield, soil-water content and rain-use efficiency were studied further.

CHAPTER 5

Soil -water content, herbage yield and rain-use efficiency of semi-arid native pasture subjected to different levels of precipitation and defoliation interval

5.1. Abstract

Precipitation variability influences native pasture productivity, and its effect is expected to be exacerbated by defoliation interval, although little is known about the exact relationship between the two, especially in semi-arid areas. The objective of this study was to determine responses of soil-water content (SWC), herbage yield and rain-use efficiency (RUE) to changes in rainfall and defoliation interval during the plant growing season. A field experiment was set up in an area of 47 m × 43 m using a split-plot design to test four levels of rain interception (RI): 0% (control), 15%, 30%, and 60% RI as main plot treatment and two levels of defoliation interval (45-day and 60-day interval) as sub-plot treatment. The rainfall was intercepted with rainout shelters. Data were collected for two years, during the 2013/14 and 2014/15 plant-growing seasons. The results indicated that annual average herbage yield of the control (206 g m^{-2}) was 89% higher than that of the 60% RI (109 g m^{-2}) and 36% higher than the 30% RI (151 g m^{-2}) treatments. It was also apparent that the SWC of the top 0–20 cm layer was 18% and 32% lower than the 20–40 cm and 40–60 cm soil layers, respectively, due to evaporation losses from the soil surface. Defoliation of the herbage biomass at 60-day intervals provided plants with sufficient recovery time, higher water use, and therefore higher biomass production. At the end of the growing season, the profile SWC of the 60-day defoliation treatment was 41% lower than the 45-day defoliation interval. A significant ($P<0.001$) positive linear relationship was found between herbage yield and amount of precipitation. RUE was higher ($P<0.01$) in the 60% RI plot (4.39 kg DM ha^{-1} mm^{-1}) compared with the control (3.66 kg DM ha^{-1} mm^{-1}), 15% RI (3.26 kg DM ha^{-1} mm^{-1}) and 30% RI (3.41 kg DM ha^{-1} mm^{-1}) treatments. The study indicates that 60% reduction in precipitation had a detrimental effect on herbage yield. Defoliating plants at longer interval (60-day) provided the plant with sufficient recovery time, which would boost drought resilience through improved SWC and RUE, and herbage yield per unit area.

Key words: cutting interval, herbage, moisture stress, phytomas, rain interception

5.2. Introduction

Eighty-two percent (68 million ha) of South Africa's land surface is rangeland (native pasture), which can be utilized effectively only by grazing ruminants (Snyman 1998; Meissner et al. 2013). Rainfall is one of the most limiting environmental factors that influence pasture productivity in arid and semi-arid areas (Snyman 2005), which account for 65–75% of the country (Snyman 1999; Smet & Ward 2006). In South Africa, large variations in grassland production may occur at any site, due primarily to differences in amount and distribution of rainfall between years (Snyman 1999). In these moisture-stressed regions, where droughts are common, the productivity of rangeland depends mainly on availability of water and efficient utilization of water resources (Snyman 1994; Bennie & Hensley 2001; Hassen et al. 2007). On the other hand, defoliation management is vital for optimal grassland management and sustainable animal production (Snyman 1999; Donkor et al. 2003). However, information is limited about the interaction between short-duration climate variability and the impact of plant defoliation management (defoliation interval, frequency, etc) on basic ecosystem processes such as SWC and primary production.

Drought reduces plant production mainly by reducing plant water availability (Harper et al. 2005), reducing litter residue returns to the soil (Luo & Zhou 2006) and restricting nutrient cycling (Snyder & Tartowski 2006). Because most plants in herbaceous-dominated systems are shallow rooted, they exhibit greater inter-annual variability in yield in response to drought than other systems (Sala et al. 1998). However, within-season variation in herbage yield determines forage availability and constrains herbivore carrying capacity (Yahjdin & Sala 2006) and grazer performance (De Waal 1990; Craine et al. 2012). Major factors such as precipitation, which controls the primary production of grasslands on a regional scale, appear to be different from those that control at a single location over time (Yahdjian & Sala 2006). Site-specific information is required to describe the impact of temporal changes in precipitation on herbage yield and livestock carrying capacity of semi-arid grasslands (Meissner et al. 2013). Understanding how variability in amount and timing of precipitation influences herbage yield through the direct effect on soil moisture provides critical information about the potential consequences of changes in precipitation within a growing season on native pasture productivity (Craine et al. 2012; Yao et al. 2013). Thus, sustainable utilization of grassland ecosystem in the arid and semi-arid regions must emphasize the capture and efficient utilization of water resources (Snyman 1999).

A number of research reports record how precipitation variability and defoliation management influence the RUE of cultivated forages in South Africa (Snyman 1994, 1999, 2005; Marais 2005; Hassen et al. 2007) and elsewhere (Yahdjian & Yala 2002; Evans et al. 2011). However, information is limited on RUE of mixed species of natural pasture in tropical Africa (Snyman 1994), except for a few reports on major grass species, namely *Themeda-Cymbopogon* natural grassland and savanna mixed plant communities of arid regions (Snyman 1993, 2000).

This study investigated the SWC, herbage yield responses and RUE of mixed species of grassland to varying amounts of rainfall and defoliation intervals. It was hypothesized that changes in the amount of rainfall and length of defoliation interval, as it occurs in semi-arid areas owing to moisture stress, affects SWC, herbage yield and RUE.

5.3. Materials and methods

The study was conducted at Hatfield Experimental Farm, University of Pretoria. The site is situated at 25° 45'S and 28° 16'E, 1372 metres above sea level. The area receives rainfall in spring (September to November) and summer (December to March) and the long-term mean annual precipitation is 674 mm. Climatic water balance (PPT-ET) was calculated for plant growing period in both years (Table 5.1). The area has a slope of 0.5–1% with a north-southern aspect. The soil of the study site is non-calcareous, sandy loam with 20–35% clay and is of the Hutton form (Soil Classification Working Group 1991) with weak structure and homogenous red colour. Table 5.2 shows selected soil physical and chemical characteristics of the study site.

The rainout shelters were established on natural pasture of various plant species as described by Mucina and Rutherford (2006) that had been protected from livestock grazing for about 60 years. Three levels of rainfall interception (RI) (15%, 30% and 60%) and control (0%) were imposed using rainout shelters as the main plot treatment and two cutting intervals (45-day and 60-day) as sub-plot treatments in a split plot design with five replications. The rain exclusion shelters were erected as described by Yahdjian and Sala (2002) with slight modifications in terms of plot size, roof shape and inclination. Each plot was covered by a 7 m x 7 m metal frame, supporting v-shaped clear acrylic bands without ultraviolet filter (Figure 5.1). A 25-cm deep ditch was excavated along the entire structures to protect runoff water. The acrylic bands or panels were constructed with a longitudinal plait of 120^{o}, with a mean height of 1 m at the lowest side of the shelter. Water collected from the acrylic material was channelled via gutters into tanks, which were closed at the top, except for the sleeve connected to the gutter, to avoid water loss through evaporation. In each rainfall event, the volume of water in the tanks was measured, and rain gauge readings were taken. Photo-synthetically active radiation (PAR) was measured (10:00 to 12:00) above and below the roofs of the rainout shelters using Ceptometer (LP-80 AccurPAR Decagon Devices Inc. USA).

Figure 5.1 Rainout shelter structure (rainout shelter, left; Jojo tanks for water collection, right) at Hatfield Experimental farm, Univeristy of Pretoria

Table 5.1 Mean growing season (2013/2014 and 2014/2015) and long term (16 years) rainfall and temperature in Hatfield, Pretoria, South Africa

Parameters				Growing season					
				Oct	Nov	Dec	Jan	Feb	Mar
Rainfall (mm) (PET)									
2013/2014			Total	97.7	102.8	143.3	41.30	156.3	253.3
2014/2015			Total	49.8	103.7	265.4	147.2	23.0	43.8
Temperature (°C)									
	2013/2014	Max	Mean	28.11	29.24	27.19	30.21	29.38	25.68
		Min	Mean	12.77	14.94	15.98	16.89	16.97	15.79
	2014/2015	Max	Mean	29.21	26.95	28.39	26.57	27.23	29.49
		Min	Mean	12.90	14.29	16.25	19.26	18.66	15.93
Evapotranspiration (ET)			Mean	159.6	152.6	147.9	154.6	95.61	98.20
Climatic water balance(PET-ET)				-85.87	-49.34	56.44	-91.34	-12.51	0.36
Long-term (16 years)									
	Rainfall		Total	58.75	86.28	142.14	102.5	77.72	94.89
	Temperature	Max	Mean	28.18	27.85	28.56	28.54	28.67	27.67
		Min	Mean	13.87	14.98	16.35	7.15	17.05	15.40

ET, evapotranspiration; PET, precipitation

Table 5.2 Selected soil physical and chemical characteristics at top soil layers (0–30 cm) of Hatfield, Pretoria, South Africa

Parameters	Mean	Maximum	Minimum	SD
Soil texture (%)				
Sand	52.8	58	50	3.01
Silt	12.4	16	8	2.63
Clay	34.8	42	30	3.79
C (g kg $^{-1}$)	20.80	25.66	17.25	3.00
N (g kg^{-1})	1.55	1.84	1.35	0.22
pH (H_2O)	5.76	6.54	5.50	0.30
BD (g cm^3)	1.40	1.48	1.26	0.08

BD, bulk density; SD, standard deviation

SWC was measured at three soil depths (0–20 cm, 20–40 cm and 40–60 cm) between 10h00 and 12h00, immediately before harvesting the herbage during the plant growing season (27 November to 19 May 2013/2014 and 2014/2015) using neutron water meter (Model 503 DR CPN Hydroprobe; Campbell Pacific Nuclear, Ca) calibrated for the study site. Profile SWC was determined as the sum of the SWC of the 0–20 cm, 20–40 cm, and 40–60 cm soil layers. Aluminium neutron probe (SMNP) access tubes were installed in the middle of each sub-plot to a depth of 60 cm. Additional access tubes were installed at 1 m from the borders of the 30 and 60% RI main plots to determine edge effects.

Green regrowth (phytomas) samples were collected from each plot at 45- and 60- day defoliation intervals, simulating different resting period after defoliation. The samples were taken from Nov 26 to May 20 (2013/2014 and 2014/2015 plant growing season) for two years. Defoliation regimens were first initiated on Nov 26, 2013/2014, and were subsequently taken at 45-day intervals (November, January, February, March and May) and 60-day intervals (November, January, March and May). The samples were harvested from four 0.25 m x 0.25 m quadrats at each sub-plot. The collected samples were pooled by sub-plot and oven dried at 65 °C for 72 hr to estimate the annual herbage DM yield (AOAC 2002).

Rain-use efficiency (RUE) is defined as the plant DM produced per unit of water evapotranspired (Snyman 1999). In forage plants, where leaf yield or edible phytomass is the main economic trait, RUE can be calculated as a given level of edible biomass or leaf yield per unit of water used by plants (Hassen et al. 2007). In this study, RUE (kg ha^{-1} mm^{-1}) was calculated for each treatment as a ratio of annual total herbage (green, edible leaves) DM yield to the mean amount of precipitation that was received for each treatment during plant growing season. Runoff was assumed to be negligible, due to better basal cover, while deep infiltration was assumed to be minimal, due to a relatively lower cumulative weekly rainfall compared with evapotranspiration and soil water-holding capacity.

The data were analysed using SAS version 9.2, with level of significance for all statistical tests set at $P \leq 0.05$. Regression equations were used to test for a relationship between growing season mean precipitation and herbage yield. A one-way analysis of variance (ANOVA) was used to test rain harvest efficiency, micro-climatic effects associated with the shelters, and photosynthetic active radiation (PAR) above and below all rainout shelter roofs. Herbage DM data were analysed using mixed linear model (Gerry and Michael 2002; SAS 2008); applying RCBD design (a split-plot arrangement) with rain interception treatment (0%, 15%, 30% and 60%) as main plots and defoliation interval (at every 45 and 60 days) as sub-plots. The fixed effects were specified as rain interception, defoliation interval and year (2013/14 & 2014/15) and the rain interception-by-defoliation interval interaction, while the random effects were specified as replication-by-rain interception and replication-by-rain interception-by-defoliation interval interaction.

The model is:

$Y_{ikjl}=\mu + a_i + df_{j(i)} + g_k + ag_{ik} + ag_{ik} + dfg_{j(i)k} + adfg_{ijk} + e_{ijkl}$

where y, is an observation variable [(herbage yield and RUE) and SWC (year excluded and replaced with soil layers, depth)];

μ, overall mean;

a_i, the effect of rain precipitation, RI (i=0%, 15%, 30% and 60%);

$df_{j(i)}$, effect of defoliation interval within RI [j (i)=1, 2)];

g_k, effect of year (k=1, 2);

ag_{ik}, interaction between RI level and year;

$dfg_{j(i)k}$ interaction between defoliation interval and year;

$adfg_{ijk}$, interaction between RI, defoliation interval and year;

e_{ijkl}, a random or unexplained error

5.4. Results

5.4.1. Light transmittance and rain-harvest efficiency

The plots with rainout shelter had an average light transmittance value of 91.1% (Table 5.3). Generally, percentage light transmittance (PAR) did not differ (*P*>0.05) between treatments and years. The 15%, 30% and 60% RI plots diverted 17.8%, 31.5% and 59.1% of the ambient rainfall, respectively, averaged over two years (Table 5.3). There was no effect of year (*P*>0.05) on light transmittance.

Table 5.3 Partial ANOVA and mean (± SEM) values showing effects of rainout shelters and year (2013/14 and 2014/15) on light transmittance in Hatfield, Pretoria, South Africa

Effects	Degree of freedom	Light transmittance (%)	Rain interception (%)
Rain interception (RI)	2	0.5693NS	1458.41***
15%		91.8 (± 0.01)	17.8 (± 0.55)
30%		90.8 (± 0.01)	31.5 (± 0.55)
60%		90.5 (± 0.01)	59.1 (± 0.55)
Year	1	0.4854NS	-
1		91.4 (± 0.01)	na
2		90.7 (± 0.01)	na
RI*Year	2	0.8377NS	na

NS, non-significant (*P*>0.05); ****P*<0.001; na=not applicable; SEM=standard error mean

5.4.2. Soil-water content

There were interaction effects of all possible combinations (*P*>0.05) on SWC in both years. Changes in precipitation amount (RI) have influenced volumetric SWC (Table 5.4). In 2013/14, the 60% RI treatment had lower (*P*<0.01) SWC compared with other RI treatments, while there were no differences (*P*>0.05) between RI treatments in 2014/15. At the same time, in 2013/14, there was lower (*P*<0.001) SWC in the 0–20 cm soil layer compared with 20–40 cm soil layer. Likewise, the SWC in the 20–40 cm soil layer lower (*P*<0.001) than 40–60 cm soil layer. In 2014/15, the SWC of 40–60 cm soil layer was more by 29.2% and 24% than the top (0–20 cm) and sub (20–40 cm) soil layers, respectively, but the differences was minima*l* (*P*>0.05). Roof edge did not affect SWC.

Table 5.4 Partial analysis of variance (ANOVA) and mean (± SEM) values showing effects of rainfall interception, defoliation interval and year (2013/14 and 2014/15) on soil-water content (SWC) during plant growing season in Hatfield, Pretoria, South Africa

Source of variation	Degree of freedom	SWC (m^3 m^{-3}) Year1 (2013/14)	Year 2 (2014/15)
Rain interception (RI)	3	5.87**	0.42NS
Control (0%)		0.29 (± 0.01)[a]	0.25 (± 0.08)
15%		0.27 (± 0.01)[b]	0.26 (± 0.08)
30%		0.29 (± 0.01)[a]	0.21 (± 0.08)
60%		0.25 (± 0.01)[b]	0.35 (± 0.08)
Defoliation interval (DI)	1	2.97NS	1.20NS
45 days		0.28 (± 0.01)	0.22 (± 0.06)
60 days		0.27 (± 0.01)	0.31 (± 0.06)
Depth (cm)	2	51.55***	0.28NS
0-20		0.22 (± 0.01)[c]	0.25 (± 0.07)
20-40		0.27 (± 0.01)[b]	0.24 (± 0.07)
40-60		0.33 (± 0.01)[a]	0.31 (± 0.07)
RI*Depth	6	2.20NS	0.84NS
RI*DI	3	1.81NS	0.94NS
DI*depth	2	0.71NS	0.81NS
RI*DI*Depth	6	1.18NS	0.98NS

Mean values in each column for each effect with different letters are statistically different at $P<0.05$, NS, non-significant ($P>0.05$); **$P<0.01$. SEM=standard error mean

5.4.3. Herbage yield and rain-use efficiency

Generally, irrespective of defoliation interval, herbage yield declined as RI level increased from 0% to 60% level. Herbage yield was higher ($P<0.001$) in the control (206 g m^{-2}) treatment compared with other RI treatments: 15% (162 g m^{-2}), 30% (151 g m^{-2}) and 60% (109 g m^{-2}). Likewise, the herbage yield in the 15% and 30% RI treatments was higher ($P<0.001$) compared with the 60% RI treatment. Intercepting 60%, 30% and 15% of the incoming precipitation resulted in the reduction of herbage yield by 89.0%, 36.4% and 28.0%, respectively, compared with the control treatment. There was no effect of defoliation interval ($P>0.05$) on herbage yield. Regardless of the defoliation interval, 60%

RI treatment had lower herbage yield compared with other treatments for both years. There was significant year-to-year variation in herbage biomass: a higher (*P*<0.001) yield was recorded in 2014/15 than in 2013/14.

Table 5.5 Partial ANOVA and mean (± SEM) values showing effects of rainfall interception, defoliation interval and year (2013/14 and 2014/15) on annual herbage dry matter yield and rain-use efficiency during plant growing season in Hatfield, Pretoria, South Africa

Source of variation	Degree of freedom	Herbage yield (g m^{-2})	RUE (kg ha^{-1} mm^{-1})
Rain interception (RI)	3	33.45***	6.37**
Control (0%)		206 (± 6.90)[a]	3.41 (± 0.20)[b]
15%		162 (± 6.90)[b]	3.26 (± 0.20)[b]
30%		151 (± 6.90)[b]	3.66 (± 0.20)[b]
60%		109 (± 6.90)[c]	4.39 (± 0.20)[a]
Defoliation interval(DI)	1	0.50NS	15.53**
45 days		154 (± 4.88)	3.29 (± 0.14)[b]
60 days		159 (± 4.88)	4.07 (± 0.14)[a]
Year	1	18.99***	3.01NS
2013/14		142 (± 4.88)[b]	3.51 (± 0.14)
2014/15		172 (± 4.88)[a]	3.86 (± 0.14)
RI*DI	3	0.07NS	0.35NS
RI*Year	3	1.24NS	0.16NS
DI*Year	1	3.20NS	21.50***
RI*DI*year	6	0.37NS	0.07NS

Mean values in each column for each parameter with different letters are statistically different at *P*<0.05, NS, non-significant (*P*>0.05); ***P*<0.01; ****P*<0.001. RUE, rain use efficiency; RI, rain interception; DI, defoliation interval; SEM=standard error mean

Except for defoliation interval-by-year interaction, all other interaction combinations, including RI*DI*year were not important (*P*>0.05) on RUE (Table 5.5 and Figure 5.1). Among the rain interception treatments the highest annual RUE (4.39 kg ha^{-1} mm^{-1}) was noticed in the 60% RI treatment compared with 15% (3.26 kg ha^{-1} mm^{-1}) and 30% RI treatments (3.66 kg ha^{-1} mm^{-1}) as well as the control (3.41 kg ha^{-1} mm^{-1}). Defoliating herbage biomass at 60-day intervals in 2013/14 resulted in higher ($P<0.01$) RUE compared with 2014/15, while the effect of year differences had influence ($P>0.05$) on RUE when herbages were defoliated at 45-day intervals (Figure 5.1).

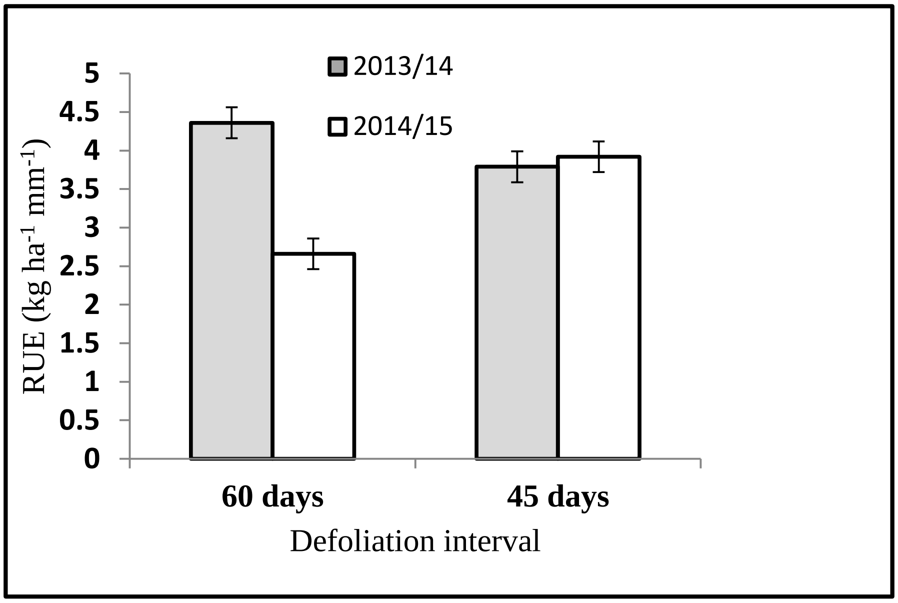
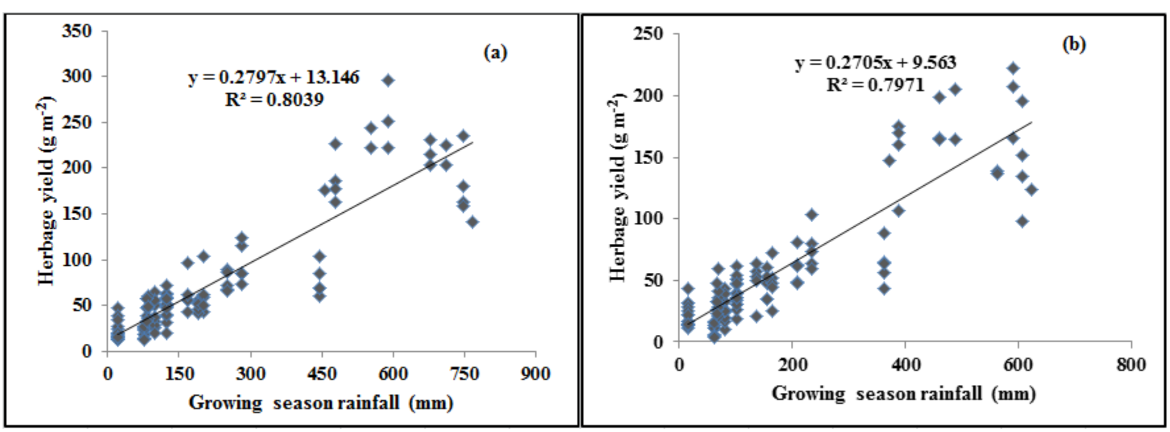


Figure 5.2 Rain use efficiency as affected by the interaction effect of defoliation interval by year in Hatfield, Pretoria. RUE, rain use efficiency

The overall relationship between herbage yield and growing season precipitation is shown in Figure 5.2. Most of the variations in herbage yield were explained by the amount of precipitation under normal and simulated drought scenarios. This was evidently shown by the positive ($P<0.001$) linear relationship that was found across treatments with a coefficient of determination (r^2) that ranged from 74% in the 30% RI plot to 80% in the control plot.



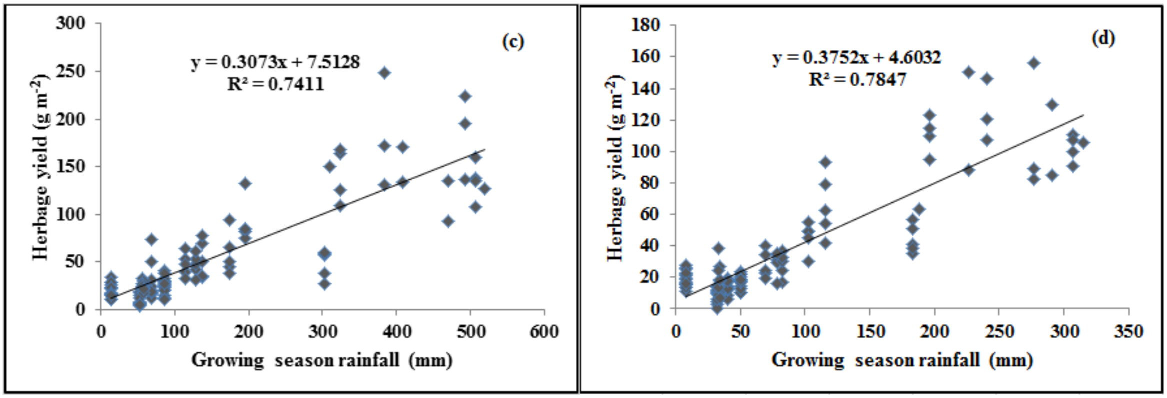


Figure 5.3 Relationships between herbage yield and growing season precipitation in the control (a), 15% (b), 30% (c), and 60% (d) rain interception plots in Hatfield, Pretoria, South Africa

5.5. Discussion

5.5.1. Soil water content

Rainfall data during the study period showed that there was a great inter- and intra-season variation in the amount of precipitation in the study site. In general, the amount of evapotranspiration (mm) exceeded the amount of rainfall (mm) by 17.2% in 2013/2014 and by 43.3% in 2014/2015. This influenced the volumetric SWC of all treatments. As a result, the 60% RI treatment had lower SWC in 2013/14, while no differences were observed between the RI treatments in 2014/15. This may be because plants utilized and depleted soil water of the 60% RI in 2013/14, while in 2014/15 some did not survive, resulting in increased SWC in the 60% RI treatments. According to Snyman (1994, 2005), SWC exhibits temporal and spatial variation owing to various influences, biotic and abiotic factors. In particular, soil water variation was greatly influenced by higher temperatures and the associated increase in atmospheric water demand. In 2013/14, the total rainfall (747 mm) was higher than the area’s normal long-term mean (674 mm), while in 2014/15 (589 mm) it was below normal rainfall. Under better basal cover conditions (as observed for the 15% RI), soil water was depleted partly owing to higher water uptake by plants and partly through increased evapotranspiration, while in the 60% RI treatment water losses were high, mainly through evaporation. Because of surface evaporation and evapotranspiration (ET), the top soil layers (0–20 cm) had less SWC due to evaporation compared with the 20–40 and 40–60 cm soil layers. However, the higher SWC in the control and 30% RI treatments might partly be due to better basal cover and the presence of mixed plant species, which can use water from different root zones (Donkor et al. 2003). The higher soil water found in the sub-soil layers in the study concurs with other studies (Snyman 1999; Donkor et al. 2003), which reported high SWC at lower (30–60 cm) soil layers. Snyman (1994) noted that climax plants of grassland in good condition (as observed in the 15% RI in the study) could withdraw more water from the top soil layer owing to their well-distributed root systems.

Soil texture is one of the major factors that influence SWC and water-holding capacity. According to Snyman (1999, 2005), water could easily be evaporated and be less likely to infiltrate to the lower (>80 cm) soil depths in sandy soils of arid lands. However, high proportion of sand (52.8%) in the study site may have contributed to more water in the lower soil layers. Similarly, Bennie and Hensley (2001) noted that sandy soils could conserve as much as 20% annual precipitation in arid lands owing to deep drainage. This implies that local edaphic factors, including soil formation and geomorphology, affect SWC, modify the magnitude of water infiltration to soil, and affect water utilization by forage species (Swemmer et al. 2007).

5.5.2. Responses of herbage yield and rain-use efficiency to rain interception and defoliation intervals

The defoliation interval considered in this study affected the overall herbage DM yield between RI treatments and across years. Generally, defoliation of herbage at 60-day intervals gave longer rest periods for plants to recover from the physiological shock associated to defoliation, resulting in more herbage yield. However, the differences observed for defoliation intervals are partly associated to variation in the amount of the rainfall in and between years, which variably influenced annual biomass production. The negative climatic water balance (precipitation minus evapotranspiration, PET-ET) in mid plant growing period (January, February and March) indicate that there were dry spells in 2014/15, while the opposite was noticed in 2013/14. The PET-ET demonstrated that the early to mid plant growing period determines biomass production due to high temperature and increased ET when precipitation is not sufficient. However, increased amounts of rainfall during the early plant growing season (December, particularly in 2014/15) contributed to improved plant growth in January–February due to carry-over effects of soil moisture. Severe moisture stress (60% RI) reduced herbage yield, regardless of the defoliation interval.

There were minimal differences in herbage yield between defoliation intervals when precipitation was lower than the average (or in the lowest ranges); however, generally long (60-day) recovery periods maximized total DM of the herbage. Plant defoliation at 45-day intervals reduced the photosynthetic capacity of native pasture species, probably because of lack of time for plants to recover, produce new leaves, and store sufficient reserves. Leaf production is critical to maximizing canopy size and subsequently enhancing the photosynthetic process and biomass accumulation (Hassen et al. 2007). Severe drought reduces herbage yield by its direct effect on leaf production, that is, by decreasing leaf number and leaf area expansion rate (Hassen et al. 2007). Severe drought reduces stomatal conductance and leaf area through increasing the soil water deficit, and reducing substrate availability, which reduces plant respiration (Luo & Zhou 2006). The reduction in herbage yield at 45-day defoliation intervals is due partly to reduced DM allocation to new tillers, leaf area and leaf area ratio (Hassen et al. 2007) and depressed sexual reproduction (Archer & Tieszen 1983). Even under normal rainfall conditions (control), a frequent defoliation regime (short cutting intervals) has been reported to reduce total herbage yield by about 25% (Donkor et al. 2003), which agrees with the findings of the present study in the control treatment. In the active plant-growing season, frequent defoliation reduced root biomass by up to 75% (Archer & Tieszen 1983), which lowered aboveground plant biomass. In contrast, in some plant species, root DM yield increased as an adaptive mechanism under moisture-deficit conditions (Snyman 1999). According to Snyman (2005), extensive root growth occurs when aboveground production is dormant (March to April), and defoliation does not occur. Frequent plant defoliation reduces aerial and basal cover, increases water loss through evaporation, and enhances water and nutrient losses through runoff (Snyman 1999). The timing of precipitation in the growing season is also important (Craine et al. 2012) because reproduction of different grass species responds differently to precipitation at different times of the year (Snyman 1994) and this may affect grazer performance (Craine et al. 2012).

The ability of plants to cope with moisture stress is an important physiological characteristic to conserve water and improve RUE (Snyman 1999; Hassen et al. 2007), particularly in moisture-stressed regions. According to Bennie and Hensley (2001), annual water loss through evaporation from the soil surface (50 to 75%) is one of the major factors that reduce RUE in arid grassland ecosystem. The results indicated that severe moisture-stressed plots (60% RI) demonstrated higher RUE, suggesting that under stress conditions, species might use soil water efficiently. Increased RUE in the 60% RI could also be due to reduced water loss through runoff. According to field species assessment and causal observation, new forb and shrub species emerged and colonized the severely moisture-stressed treatment, 60% RI. The increased RUE in the 60% plot in the study was partly due to the higher number of mixed grass-legume species and their differential water use ability from various soil profiles, probably due to differences in their root systems, and niche separation. This finding is in agreement with previous reports (Snyman 1993, 2000; Hassen et al. 2007) that species that use less water per unit DM had the highest water-use efficiency and could survive under moisture-stressed conditions. As well as total rainfall, evenly distributed rainfall over the plant-growing period may have contributed to better RUE in 2013/14. There is limited literature on RUE with which to compare the results of the study because the only water-balance and RUE studies in natural pasture in these regions were undertaken on dry *Themeda-Cymbopogon* natural grassland (Snyman 1993) and on savannah plant communities (Moore et al. 1988; Snyman 2000). The highest RUE (4.39 kg ha^{-1} mm^{-1}) was obtained in the study in the 60% RI treatment, which was dominated by *Eragrostis lehmanniana* and *Ipomoea crassipes* mixed pasture, was lower than values reported by Snyman (1994) who reported high RUE (7.2 kg ha^{-1} mm^{-1}) for cultivated forage species such as *Cenchrus ciliaris, Chloris gayana, Digitaria eriantha subsp. eriantha, Eragrostis curvula* and *Panicum maximum.*

5.6. Conclusion

Findings from this study showed that rainfall variation affected herbage yield and RUE through its effects on SWC. Consequently, increased water loss resulted in moisture scarcity mainly in the top soil layers (0–20 cm), where most grassroots are concentrated.

Intercepting 15% and 30% of the incoming rain resulted in 28% and 36.4% reductions in herbage yield, respectively. Compared with the control, the 60% RI treatment reduction in rainfall caused 89% decline in herbage yield, which is detrimental to forage production in this environment. The RUE of the 60% RI treatment was higher than the other RI treatments, implying that mixed species (particularly grass-legume mixtures) could better utilize soil water efficiently under moisture-stress conditions. The positive linear relationship between precipitation and herbage yield under normal and reduced precipitation scenarios indicates that precipitation is a major factor that affects herbage production in semi-arid areas. The results also suggest that defoliating plants at longer intervals (60-day defoliation) provided an adequate recovery period that could improve herbage yield and RUE, and sustainable native pasture and animal production in semi-arid areas. However, further study is required to understand yield and quality of dominant forage species with the aim of identifying drought-resilient species that have nutritional importance and are capable of maintaining plant community structure and stability under future climate change circumstances.

CHAPTER 6

Yield and nutritive quality of dominant forage species subjected to different levels of precipitation in subtropical native rangeland, Pretoria, South Africa

6.1. Abstract

Understanding how future drought might influence nutritive quality in semi-arid native grassland is vital to determining appropriate adaptation options. An experiment was conducted at Hatfield Experimental Farm, University of Pretoria, South Africa by intercepting the ambient rainfall at different levels to simulate various levels of drought. The treatments were control (0%), 15%, 30%, and 60% rain interception (RI) using rainout shelters. Responses of natural pasture – composed of *Digitaria eriantha L. (Common Finger grass)*, *Setaria sphacelata var. Torta* (Pigeon grass), *Eragrostis lehmannaina var. Lehmanniana (*Lehman lovegrass), *Heteropogon controtus (*spear grass), *Elephantorrhiza elephantina*, shrub (elephant's root, shrub) and *Ipomoea crassipes (*morning glory, forb) – to moisture stress were examined by monitoring basal cover, DM yield, chemical composition, *in-vitro* gas production (2, 4, 8, 12, 24, 32 and 48 hrs) and fermentation characteristics. On average, the DM yield of grasses ranged between 42 and 258 g m^{-2}, dwarf shrub between 64 g m^{-2} and 90 g m^{-2} and forb between 155 and 411 g m^{-2}. Generally, severe moisture stress (60% reduction in precipitation) resulted in shifts in species composition, which influenced the chemical composition and *in vitro* organic matter digestibility (IVOMD) of the dominant grass, forb and shrub species. Crude protein (CP) values and total gas production of the forage samples were higher in the 30% and 60% RI treatments, while acid detergent fibre (ADF), neutral detergent fibre (NDF), and acid detergent lignin (ADL) varied without showing a trend. In general, the fractional rate of fermentation and potential degradability were greater in the severe drought treatments (30% and 60% RI). This study revealed that severe moisture stress (60% RI) reduced yield, but improved certain quality attributes such as CP and IVOMD.

Key words: chemical composition, dry matter yield, forb, gas production, grass, shrub

6.2. Introduction

Water availability is the primary constraint to plant productivity in many terrestrial biomes (Heisler-White et al. 2008), particularly in arid and semi-arid ecosystems (Knapp et al. 2006). Rainfall determines biomass yield and, according to IPCC (2007) projection, it is one of the main variables that will be influenced by future climate change in South Africa, where rainfall distribution, frequency and variation remain uncertain (Gaughan & Waylen 2012).

Besides effects on yield (Küchenmeister et al. 2013), drought may influence future assemblages of grassland plant community on a local and national scale (Evans et al. 2011; Gaughan & Waylen 2012; Matias et al. 2014). Drought may also exert an influence on nutritive quality by altering leaf/stem ratios, and causes other morphological modifications that affect the cell wall fraction and thus the nutritive values of plant parts. However, reports on the effects of drought on nutritive values are limited and inconsistent (Hassen et al. 2006; Küchenmeister et al. 2013). Understanding plant performance and persistence during and after drought is important for sustainable farming systems, where forage-based feed underpins animal production (Hatier et al. 2014). In the prolonged dry season, animals may experience feed shortage and nutrient deficiency because forage quantity and quality deteriorate considerably. Despite this, there are huge uncertainties in terms of elaborating how plant species respond to drought in their natural habitat. While simulated moisture deficit effects on yield, drought resilience and nutritive quality are not new elsewhere (Yahdjian & Sala 2002; Evans et al. 2011), such data are scarcely reported from tropical Africa, where rainfall varies remarkably. Moreover, the sensitivity or resilience of grassland species to extreme and sustained dry periods, such as those that may occur in the future, need to be tested.

Recently, Gemeda and Hassen (2014) reported that native plant species collected from the Kalahari Desert demonstrated great variability in terms of chemical composition and *in-vitro* fermentation characteristics under normal conditions. However, the nutritive value of native species under drought situations and their tolerance to moisture stress have not been studied. This study was aimed at evaluating the effects of moisture stress (simulated drought) on yield and nutritive quality of dominant forage species in their natural habitat and at identifying drought-resilient species for future community assemblages under predicted climate-change states.

6.3. Materials and methods

The study was conducted at Hatfield Experimental Farm, University of Pretoria, Pretoria, South Africa. The site is situated at 25° 45´S and 28° 16´E, 1372 metres above sea level. The area receives rainfall in spring (September to November) and summer (December to March), and the long-term mean annual precipitation is 674 mm. The soil is categorized as sandy loam, with 20–35% clay content. The soil form is of Hutton with weak structure and a homogenous red colour, and is non-calcareous. A rangeland of 6.6 ha that has been protected from livestock grazing since 1960 was used for the experiment (Mucina and Rutherford 2006). Based on plot coverage and feed values (Tainton & Hardy 1999), six dominant species were selected as indicators from a diverse successional range, encompassing pioneer dwarf shrub, forb and grasses. These included *Elephantorrhiza elephantina* (dwarf shrub), *Ipomoea crassipes* (creeping forb), *Eragrostis lehmannania, Setaria sphacelata var. torta*, *Digitaria eriantha* and *Heteropogon controtus* (grasses).

Four rain interception levels were imposed using rainout shelters (see figure 5.1) by intercepting 0%, 15%, 30% and 60% of the ambient rainfall in a completely randomized block design with five replications. The rain exclusion shelters were erected as described by Yahdjian and Sala (2002) with slight modifications in terms of plot size, roof shape and inclination. Each plot was covered by a 7 m x 7 m metal frame supporting v-shaped clear acrylic bands without ultra-violet filter. The acrylic bands or panels were constructed with a longitudinal plait of 120^{o}, with a mean height of 1 m on the lowest side of the shelter. A 25-cm deep ditch was excavated two metres from the border of the exterior plots along the entire structures to protect runoff water. Water collected from the acrylic material was channelled via gutters into tanks, which were closed at the top, except for the sleeve connected to the gutter, to avoid water loss through evaporation. In each rainfall event, the volume of water in the tanks was measured, and rain gauge reading was taken. Photo-synthetically active radiation (PAR) was measured in full sunny conditions (10:00 to 12:00) using Ceptometer (LP-80 AccuPAR Decagon Devices Inc. USA).

Aboveground net primary productivity (ANPP), which is described as the sum of grass, shrub and forb biomasses (Wang et al. 2014a). Data on plant density, floristic composition, and individual species cover were measured using the quadrat method (Daget & Poissonet 1971), at the peak plant growth period (optimum time for yield and quality) in mid February 2014 and 2015. Plant species samples were pressed between newspaper and taken to Herbarium of department of Plant Sciences, University of Pretoria for identification. The percentage cover of species was measured following description of Abule et al. (2007a). This time was selected based on previous reports (Marais 2006). The size of quadrat was 1 m by 1 m and all species inside the quadrat were assessed. Four quadrats were used at each plot, and species were clipped 5 cm above the ground from each quadrat and average for each plot for DM determination and analysis of chemical composition. The harvested forage of each species was oven dried (65 °C for 72 hr) (AOAC 2002). For all chemical composition analyses and *in vitro* studies, the oven-dried samples were ground to pass through a 1-mm sieve in a Willey mill (Arthur H. Thomas, Philadelphia, Pa, USA). Crude protein (CP) was measured on a LECO FP-428 nitrogen and protein analyser (LECO Corporation, St Joseph, Mich, USA). NDF and ADF concentration were determined using an ANKOM200/220 fibre analyser (ANKOM Technology, Fairport, NY, USA) based on the methods described by Van Soest et al. (1991). Sodium sulphite and heat-stable amylase were used in the analysis of NDF. Acid detergent lignin (ADL) was determined by solubilisation of cellulose with sulphuric acid in the ADF residue (Van Soest et al. 1991).

To measure gas production, rumen-fluid inoculums were collected before the morning feeding from two rumen cannulated sheep fed on *ad libitum* alfalfa hay. The inoculum was mixed, strained and transferred to pre-heated (39 ºC) thermos flasks, while being purged with CO_2 continuously to maintain anaerobic conditions. The preparation then continued as described by Goering and Van Soest (1970). The buffer solution, macro and micro solution, was prepared as suggested by Goering and Van Soest (1970) with slight modifications as described by Mould et al. (2005). The inoculum was added to the buffer solution: 15 mL of rumen fluid and 25 mL of buffer solution. A semi-automated system was used to measure gas production through *in vitro* incubation at 39 ºC, according to Theodero et al. (1994). Gas pressure was measured at 2, 4, 8, 12, 24, 32 and 48 hr with a pressure transducer (PT) connected to a data tracking system. Three replicates of the same feed were incubated per run and three runs were executed for every dwarf shrub, forb, and grass species. *In vitro* organic matter digestibility (IVOMD) was determined in two digestion phases according to the method of Tilley & Terry (1963), as modified by Engels and Van der Merwe (1967).

The rate and extent of gas production was determined for each species by fitting gas production data to the non-linear equation (Ørskov & McDonald 1979):

$$y = b\,(1-e^{-ct})$$

where y is gas production at time t; b is the slowly fermentable fraction (g kg^{-1} DM); and c is the rate (% h^{-1}) of fermentation of fraction b. The *in vitro* incubation times were used to fit non-linear regression models using the NLIN procedure (SAS version 9.2).

The data were statistically analysed using the GLM procedure of SAS (Statistical Analysis System, version 9.2).

The model used was: $Y_{ijk} = \mu + d_i + \varepsilon_{ik}$

where Y_{ijk} is the observation, μ is the overall mean, d_i is the drought effect (i=0%, 15%, 30%, 60%), and ε_{ijk} is the residual error. Where F test declares significances at $P < 0.05$, differences between means were separated using Tukey's test. Linear regression equations were used to analyse the relationship between different levels of rain interception and major feed quality attributes (PROC REG SAS, version 9.2).

6.4. Results

Generally, percentage of light transmittance was not different among the treatments. The treatments with the rainout shelter had an average light transmittance value of 91.1%. The 15%, 30% and 60% RI treatments diverted 17.8 ± 0.55%, 31.5 ± 0.55% and 59.1 ± 0.55% of the ambient rainfall events, respectively. In the plant growing season the experimental site received 88% of the rainfall in summer (December to February), while about 12% rainfall was received in spring (September to November).

6.4.1. Major plant species

Over 40 plant species were identified, of which the major species are described in Table 6.1. Generally, dwarf shrubs and forbs were dominant in the treatment plots during the pre-rain and early rainy seasons, while grasses were dominant during the main rainy season. The dominant grass species identified were Common Finger grass *(Digitaria eriantha)*, pigeon grass *(Setaria sphacelata var. Torta)*, Lehman love grass *(Eragrostis lehmanniana var. lehmanniana)*, Spear grass *(Heteropogon controtus)*, while the dominant forb and shrub species were Elephant's root *(Elephantorrhiza elephantina)* and Morning glory *(Ipomoea crassipes)*, respectively.

Table 6.1 List of dominant plant species identified in sub-tropical native pasture in Hatfield, Pretoria, South Africa

Common name	Scientific name	Adaptation (agro-ecological areas)	Ecological status	Palatability	Season of occurrence	Strategy
Common Finger grass	*Digitaria eriantha L.*	500 mm+	De	HD	RS	LS
Pigeon grass	*Setaria sphacelata L.*	700-1700 mm	De	HD	RS	LS
Lehman lovegrass	*Eragrostis lehmanniana L.*	250-500 mm	Inc II	DE	RS	LS
Hairy love grass	*Eragrostis barbinods L.*	300-600 mm	Inc II	DE	RS	LS
Spear grass	*Heteropogon contortus L.*	Wider adaptation	Inc I	DE	RS	LS
Elephant's root (dwarf shrub)	*Elephantorrhiza elephantina*	Wider adaptation	Inc III	LD	PR	P
Morning glory (forb)	*Ipomoea crassipes L.*	Wider adaptation	Inc II	LD	ER	MS
Buffalo grass	*Aristida congesta subsp.congesta*	300-1200 mm	Inc III	LD	RS	LS
-	*Peucedanum magalismontanum*	500 mm+	Inc II	LD	PR	P
Chandelier plant	*Bryophyllum delagoense*	500 mm+	Inc I	LD	PR	P
Moss verbena	*Verbana tenuisecta L.*	500-1000 mm	De	LD	PR	P
African potato	*Hypoxis hemorocallida L.*	500-2000	De	LD	PR	P
Dollar leaf	*Rhynchosia monophylla L.*	650 mm+	De	DE	ER	MS
Bi-coloured-leafed vernonia	*Vernonia oligocephala L.*	600 mm+	Inc I	LD	ER	P

Ecological status (De: decreaser, Inc I: increaser I; Inc II: increaser II; Inc III: increaser III); palatability (HD: highly desirable; DE: desirable, LD: less desirable); season of occurrence (RS: main rainy; ER: early rain; PR: pre-rain seasons); strategy (P: pioneer; MS: mid-successional and LS: late successional) (O'Connor & Bredenkamp 1997; Acocks, 1998; Mucina & Rutherford 2006)

6.4.2. Aboveground primary productivity

Aboveground net primary productivity is shown in Figure 6.1a and b. Generally, the contribution of grasses to the annual total yield was higher than forbs and shrubs regardless of RI over the two-year period. In year 1, the 15% and 30% RI treatments had significantly (P<0.05) higher proportion of grasses compared with the 0% and 60% RI treatments, while in year 2, the 0%, 15% and 30% had higher proportions of grass biomass than the 60% RI treatment. In year 1, a significantly (P<0.01) higher proportion of forbs was found in the 0% RI plot, while in year 2 the difference was insignificant (P>0.05). The control (0%) and 30% RI treatment had generally higher total ANPP in year 1, while in year 2 ANPP decreased consistently in drought-induced plots (15%, 30% and 60% RI) compared with plots that received ambient rainfall.

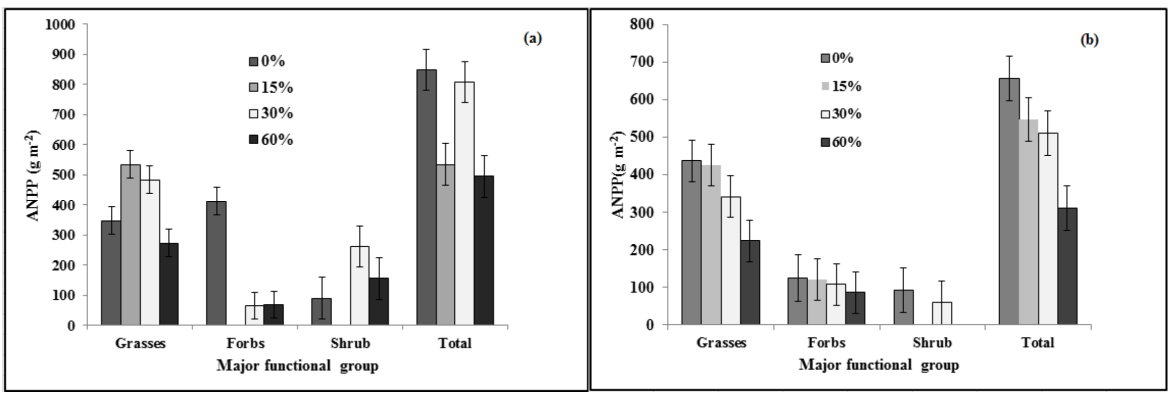


Figure 6.1 Aboveground primary productivity of major functional groups of grasses, forbs and shrubs as affected by various levels of rain interception (0%, 15%, 30% and 60%) in 2013/14 (a) and 2014/15 (b) in Hatfield, Pretoria, South Africa. *Bars* represent standard error of mean'

6.4.3. Yield and yield components

Different rain interception levels influenced yield and yield components of the dominant species considered in this study (Table 6.2). Basal cover and DM yield of *Digitaria eriantha* were significantly lower (P<0.05) in the 60% RI. The year effect was significant (P<0.01) only in plant height, where higher plant height was recorded in year 2 compared with year 1. *Digitaria eriantha* had lower (P<0.05) plant height and density in the 60% treatment than in the 0%, 15% and 30% RI treatments. Similarly, all the RI levels introduced in this study influenced the DM yield of *Setaria sphacelata* through its effect on plant density and height. The DM yield of Setaria sphacelata was about four times lower in the 60% than in the 15% RI treatment. Although the basal cover and plant density of *Eragrostis lehmanniana* tended to be lower in the 0% plots than in the 60% RI, the differences were not significant (P>0.05) in terms of DM and basal cover. On average, plant height of Heteropogon controtus was higher (P<0.01) in the 30% RI compared with other treatments. *Elephantorrhiza elephantina* and *Ipomoea crassipes* covered 3% to 19% and 20% to 41%, respectively, while their DM exhibited wide variations (155 to 411 g m^{-2}). On average, the DM yield of grasses ranged between 42 g m^{-2} for *Heteropogon controtus* in the 60% and 258 g m^{-2} for Setaria sphacelata in the 15% RI treatment.

6.4.4. Chemical composition

Except for the OM and CP concentrations, where the RI effect was not consistent, generally the RI treatment significantly affected other chemical composition parameters of the dominant forages considered in this study (Table 6.3 and 6.4). It appears that the greatest CP concentration was recorded for *Eragrostis lehmanniana* (65 g kg^{-1} DM), *Heteropogon controtus* (61.5 g kg^{-1} DM) and Heteropogn controtus (62.4 g kg^{-1} DM) in the 60% RI treatment, while a lower CP concentration was recorded for these grasses in the 15% and 30% RI plots (Table 3). The fibre constituents exhibited great variations under the various drought conditions. The lowest ADF and NDF concentration were recorded for grasses in the 15% RI treatment compared with the control treatment. However, except for Setaria sphacelata, it appears that fibre components increased as drought level rose to 30% RI.

Regardless of species difference, there were mild relationships between RI and forage quality attributes for dominant grass species with a coefficient of determination (R^2) 0.4040 for CP and 0.3032 for ADF (Figure 6.2). However, there was no relationship between RI and NDF or IVOMD.

Table 6.2 ANOVA table showing DM yield and yield components of dominant species as affected by different levels of rainfall interception, year and rain interception by year in subtropical native pasture in Hatfield, Pretoria, South Africa

		Plant growth parameters		Plant stand parameters	
Species	Fixed factors	DM (kg/ha)	Plant height (m)	Basal cover (%)	Plant density
Digitaria eriantha	Rainfall interception (RI)	1.44NS	1.30NS	2.72*	1.53*
	Year (Y)	0.78NS	7.12**	0.93NS	0.58NS
	RI x Y	0.39NS	0.67NS	0.68NS	0.22NS
Setaria sphacelata	Rainfall interception (RI)	5.08*	3.54*	1.88NS	5.74**
	Year (Y)	4.45*	3.36NS	2.46NS	3.55NS
	RI x Y	1.45NS	1.42NS	0.76NS	1.39NS
Eragrostis lehmanniana	Rainfall interception (RI)	1.89NS	0.78NS	0.43NS	0.34NS
	Year (Y)	2.41NS	2.22NS	0.34NS	0.38NS
	RI x Y	0.06NS	0.49NS	0.62NS	0.81NS
Heteropogon controtus	Rainfall interception (RI)	2.01NS	6.64**	0.09NS	1.68NS
	Year (Y)	0.43NS	16.04**	0.90NS	1.50NS
	RI x Y	1.68NS	2.43NS	0.44NS	0.67NS
		Mean	Mean	Mean	Mean
Ipomoea crassipes	Rainfall interception (RI)	220.60	72.23	23.1	90.8
	Year (Y)	215.89	94.91	28.36	39.02
Elephantorrhiza elephantina	Rainfall interception (RI)	30.35	32.30	46.13	20.46
	Year (Y)	52.45	36.06	51.93	25.83

Means within a column for each parameter for grass species with different letters ([a], [b], [c]) are significantly different at *P*<0.05; * *P*<0.05; ***P*<0.01. na=not available. DM, dry matter. [Ψ] include *Hypoxis hemorocallida, Rhynchosia monophylla, E.barbinodis,Verbana tenuisecta, Aristida congesta.*

Table 6.3 Chemical composition (mean ± SEM) of dominant grass species as affected by different levels of rainfall interception in subtropical native pasture in Hatfield, Pretoria, South Africa

Dominant grass species	Rain interception	Chemical composition (g kg^{-1} DM)					
		DM	CP	NDF	ADF	ADL	IVOMD
Digitaria eriantha	0%	952(8.38)	46.1(3.03)	709(6.19)^{a}	419(4.91)^{b}	187(7.49)^{b}	601(7.0)^{b}
	15%	953 (10.2)	40.6(6.66)	663(13.0)^{b}	436(15.1)^{b}	236(8.44)^{a}	582(7.0)^{b}
	30%	954(3.18)	42.6 (0.85)	718(4.22)^{a}	433(8.72)^{b}	127(3.69)^{c}	624(7.0)^{a}
	60%	960(8.82)	56.1(0.00)	704(13.3)^{a}	472(8.82)^{a}	190(3.66)^{b}	618(7.0)^{a}
Setaria sphacelata	0%	938(4.19)	44.0(1.76)^{c}	718(10.2)^{b}	443(12.7)	218(7.8)	586(10.0)
	15%	941(8.52)	40.5(1.66)^{d}	654(19.0)^{c}	449(1.66)	236(9.5)	555(10.0)
	30%	947(4.50)	45.3(0.48)^{b}	740(4.75)^{a}	460(5.92)	218(7.2)	590(12.3)
	60%	946(5.48)	61.5(5.01)^{a}	720(12.7)^{b}	458(11.8)	267(13.1)	588(10.0)
Eragrostis lehmanniana	0%	961(2.46)	50.9(0.91)^{b}	748(5.72)^{a}	415(2.14)^{b}	272(4.06)^{a}	624(11.2)^{a}
	15%	932(4.26)	43.4(1.14)^{d}	660(12.8)^{b}	388(24.6)^{b}	229(11.3)^{c}	567(11.2)^{b}
	30%	967(12.5)	49.7(0.93)^{c}	727(13.3)^{a}	412(5.75)^{b}	190(8.27)^{d}	612(11.2)^{a}
	60%	957(13.3)	64.6(4.09)^{a}	660(13.1)^{b}	469(23.5)^{a}	255(4.97)^{b}	620(11.2)^{a}
Heteropogon controtus	15%	931(0.75)^{b}	47.2(4.11)^{b}	627(4.26)^{c}	459(2.57)^{c}	187(3.79)^{c}	510 (7.35)^{b}
	30%	940(2.00)^{a}	35.7(5.33)^{c}	669(5.01)^{b}	474(1.61)^{b}	215(0.42)^{b}	514(7.35)^{b}
	60%	943(1.15)^{a}	62.4(2.51)^{a}	761(7.47)^{a}	499(3.80)^{a}	238(6.01)^{a}	548(7.35)^{a}

Means within a column for each specie with different letters (^{a}, ^{b}, ^{c}) are significantly different at P<0.05. SEM=standard error mean

DM: dry matter (laboratory based); DM: dry matter; CP: crude protein; ADF: acid detergent fibre; NDF: neutral detergent fibre; ADL: acid detergent lignin; IVOMD: *in vitro* organic matter digestibility

Table 6.4 Chemical composition (mean ± SEM) of dominant forb and dwarf shrub species as affected by different levels of rainfall interception in subtropical native pasture in Hatfield, Pretoria, South Africa

Dominant shrub	Rain interception	Chemical composition (g kg^{-1} DM)					
		DM	CP	NDF	ADF	ADL	IVOMD
E. elephantina	0%	952(2.69)^{b}	106(0.00)^{b}	537(5.77)^{a}	405(5.77)^{b}	177(11.6)^{b}	715(7.72)^{b}
	30%	965(0.00)^{a}	138(0.57)^{a}	482(1.58)^{b}	435(3.11)^{a}	234(7.65)^{a}	737(7.72)^{a}

Means within a column for each specie with different letters (^{a}, ^{b}, ^{c}) are significantly different at P<0.05. SEM=standard error mean

DM: dry matter (laboratory based); DM: dry matter; CP: crude protein; ADF: acid detergent fibre; NDF: neutral detergent fibre; ADL: acid detergent lignin; OMD: *in vitro* organic matter digestibility

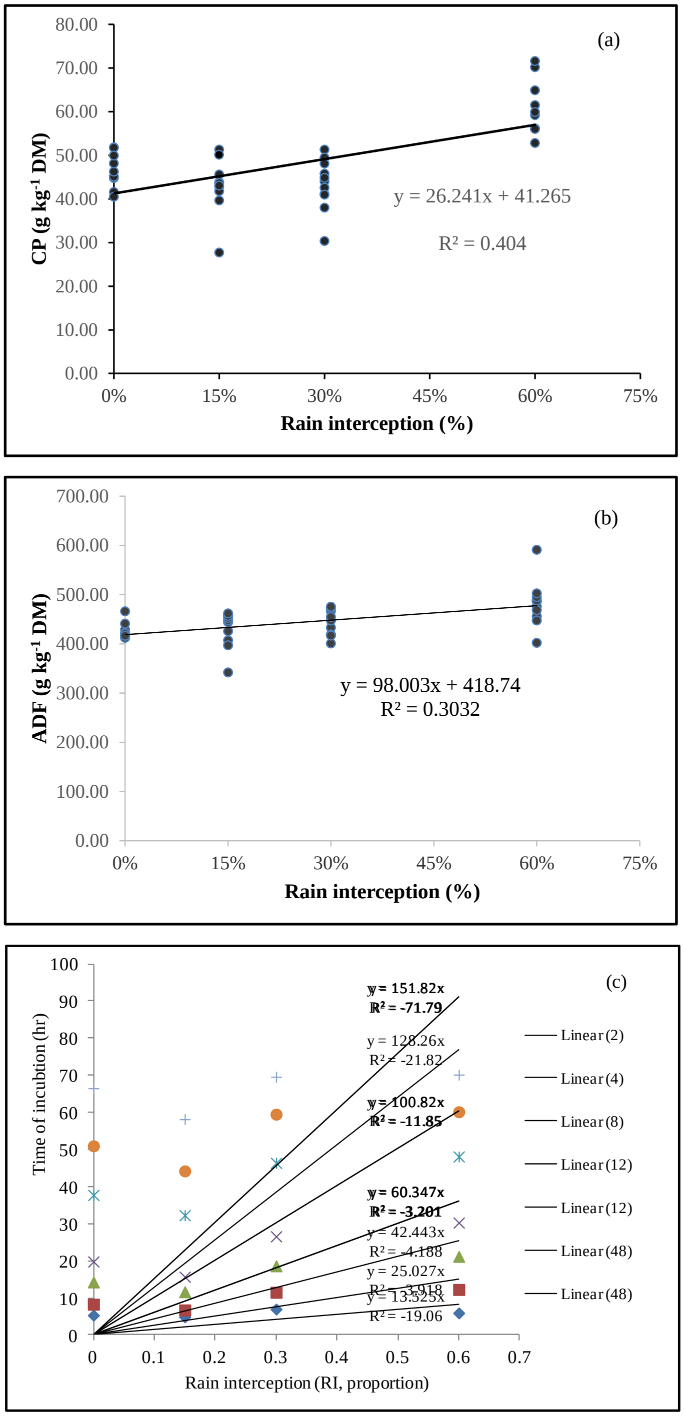


Figure 6.2 Relationship between rain interception (%) and crude protein, CP (a) or acid detergent fibre, ADF (b) and time of incubation, hr (C) for dominant grass species in subtropical native pasture in Hatfield, Pretoria, South Africa

Except for *Setaria sphacelata*, the ADF concentration of grass species was higher (P<0.05) in the 60% RI than other treatments (Table 6.3). Higher (*P*<0.01) ADF, NDF and ADL concentrations were recorded for Heteropogon controtus in the 60% treatment, while the NDF and ADL concentrations were lower in *Eragrostis lehmanniana*. At the same time, the highest (748 g kg^{-1} DM) and lowest (660 g kg^{-1} DM) NDF concentrations were recorded for *Eragrostis lehmanniana*. The IVOMD of dominant grass species were generally higher in the 30% and 60% RI treatments.

There were significant variations in the chemical composition of the shrub and forb species (Table 6.4). CP concentration was higher (*P*<0.001) in the 30% RI for *Elephantorrhiza elephantina*. The highest CP concentration was 138 g kg^{-1} DM, while the lowest was 106 g kg^{-1} DM. Both values were recorded for *Elephantorrhiza elephantina*. Except for NDF, fibre constituents for *Elephantorrhiza elephantina* were higher in the 30% treatment than in the 0%. However, both species showed higher CP concentration, IVOMD in the 30% and 60% RI treatments.

6.4.5. In vitro gas production and fermentation characteristics

The dominant forage species showed different patterns of gas production in response to the RI treatments (Figure 6.3). *Digitaria eriantha* grass had lower (*P*<0.05) total gas production at 15% RI than the control, 30% or 60% RI for 4, 8, 12, 32 and 48 hr periods (Figure 6.3a). Likewise, gas production was higher at 24, 32 and 48 hrs in the 60% RI for *Elephantorrhiza elephantina* (Figure 6.3e).

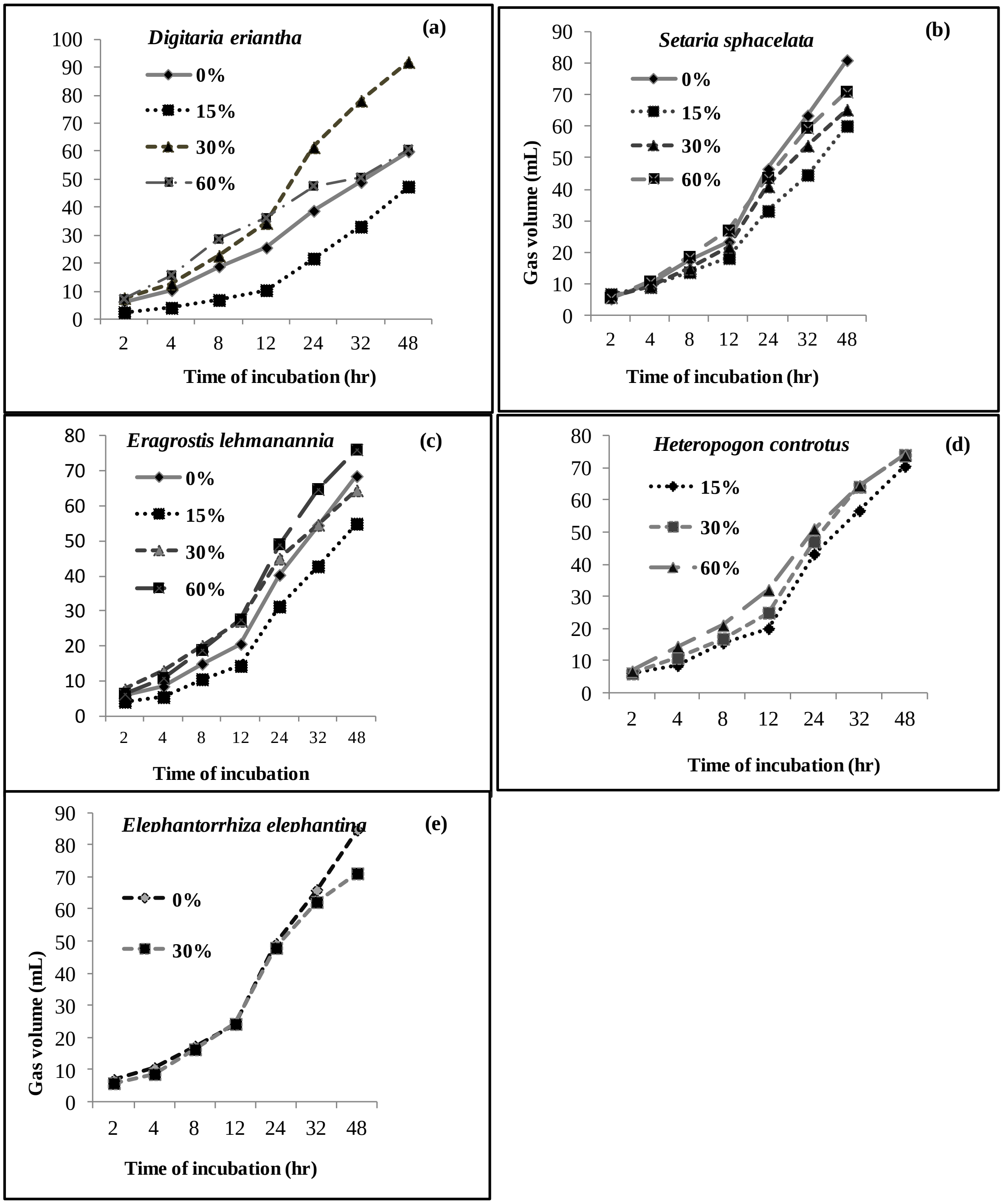


Figure 6.3 Gas production pattern of dominant grass species, (a–d) and dwarf shrub (e) as influenced by rainfall interception (0%, 15%, 30% and 60%) in Hatfield, Pretoria, South Africa

Table 6.5 summarizes gas production parameters of the dominant species. The slowly fermentable fraction (b-value) and potential degradability (PD) values were higher (*P*<0.01) in the 30% RI than other treatments for *Digitaria eriantha* grass. The slowly fermentable fraction in *Setaria sphacelata*, *Eragrostis lehmanniana* and *Heteropogon controtus* was lower (*P*<0.05) in the 15% RI treatment than in the 60%. Similarly, the highest b-value (105 mL 0.4 g^{-1} DM) was recorded for *Digitaria eriantha* grass in the 30% RI, while the lowest b-value (48 mL 0.4 g^{-1} DM) was recorded for the same grass in the 15% RI. Fractional rate of fermentation (c-value) was higher (*P*<0.05) in the 60% RI than in the 0%, 15% and 30% RI for *Digitaria eriantha* grass. Potential degradability was generally higher in the 60% RI for all species considered in this study. However, the potential degradability was significantly higher for *Digitaria eriantha* grass in the 60% RI treatment than in the 0%, higher for *Heteropogon controtus* grass in the 60% RI than 15% RI. For *Digitaria eriantha* grass, potential degradability ranged between 19 and 45 mL 0.4 g^{-1} DM.

Table 6.5 Gas production parameters (mean ± SEM) as influenced by different levels of precipitation in subtropical native pasture in Hatfield, Pretoria, South Africa

Dominant species	Gas production parameters	Rainfall interception levels			
		0%	15%	30%	60%
Grasses					
Digitaria eriantha	b	65.3(3.33)^{b}	47.6(8.07)^{c}	105(2.75)^{a}	59.2(2.62)^{b}
	c	0.045(0.009)^{b}	0.033(0.001)^{c}	0.038(0.001)^{c}	0.075(0.004)^{a}
	PD	30.0(2.63)^{c}	18.8(3.04)^{d}	45.1(0.50)^{a}	35.3(0.83)^{b}
Setaria sphacelata	b	92.7(1.33)^{a}	67.0(15.97)^{b}	82.2(13.53)^{a}	91.6(3.88)^{a}
	c	0.032(0.003)	0.032(0.006)	0.031(0.004)	0.031(0.006)
	PD	36.4(2.44)	26.5(8.93)	30.6(3.88)	34.2(2.95)
Eragrostis lehmanniana	b	80.4(7.91)^{ab}	60.7(18.1)^{b}	83.3(9.62)^{ab}	90.3(5.36)^{a}
	c	80.4(7.91)^{ab}	60.7(18.1)^{b}	83.3(9.62)^{ab}	90.3(5.36)^{a}
	PD	0.032(0.000)	0.033(0.001)	0.039(0.013)	0.035(0.002)
Heteropogon controtus	b	80.7(1.49)^{c}	85.9(0.70)^{b}	91.3(6.10)^{a}	80.7(1.49)^{c}
	c	0.034(0.000)	0.036(0.002)	0.035(0.002)	0.034(0.000)
	PD	32.4(0.40)^{b}	35.7(0.62)^{ab}	37.8(3.71)^{a}	32.4(0.40)^{b}
Dwarf shrub					
Elephantorrhiza elephantina	b	96.3(0.70)^{a}	na	78.5(0.44)^{b}	na
	c	0.032(0.002)^{b}	na	0.040(0.000)^{a}	na
	PD	37.2(1.18)^{a}	na	34.7(0.09)^{b}	na

Units for b (slowly fermentable fraction) and PD (potential) are mL 0.4 g^{-1} DM; units for c (fractional rate of fermentation) are mL h^{-1}. Means within a row for each trait for each specie and each gas production parameter with different letters (^{a}, ^{b,} ^{c}) are significantly different at P<0.05. na = not available. SEM=standard error mean

6.5. Discussion

6.5.1. Basal cover, biomass yield and quality attributes

In this study, drought influenced basal cover, biomass yield, chemical composition and digestibility of the dominant forage species. In particular, moisture stress in the 30% and 60% RI affected species composition and yield by exerting its greatest effect on plant density and basal cover. Early land cover at the onset of the rainy season was achieved through growth of forbs and shrubs such as *Elephantorrhiza elephantina* and *Ipomoea crassipes*, which played an important role in improving basal cover, which protects soil from wind and water erosion. According to Abule et al. (2007a), basal cover has great significance in drought-prone environments, as it protects lands from erosion. However, irrespective of species differences, basal cover decreased consistently as the amount of RI increased to 60%. For example, basal cover of *Digitaria eriantha* decreased by 13–37% in the 60% RI, compared with 0%, suggesting that this grass could be adversely affected in predicted climate-change situations. Conversely, the basal cover of *Eragrostis lehmanniana* showed an increasing trend in the 60% RI treatment, suggesting that this grass is more resilient to the highest drought level introduced in this study.

According to Hatier et al. (2014), susceptibility and less drought resilience in some species could be due to rapid tissue turnovers. On the other hand, a soil seed bank was reported to be an important reservoir for many species in arid areas (Tessema et al. 2014). This might be one of the attributes that help to explain why shrub and forb species, which have greater drought tolerance, emerged and colonized new niches at the early successional stage. In the study, species numbers increased in the 60% RI treatment, contributing to plant diversity. Despite their significance as pioneers, however, most of the forbs (75%) were replaced by grasses as the growing season advanced, and some were less important nutritionally, which agrees with other reports (Abule et al. 2007a). However, the researchers observed that *Elephantorrhiza elephantina* might maintain its nutritive values under severe drought conditions.

In general, the drought level of 60% resulted in an adverse decrease in ANPP of dominant grass species *Digitaria eriantha* and *Setaria sphacelata*, and had an overwhelming negative effect. In this study, ANPP was reduced as the drought level increased. However, a drought level up to 30% did not greatly influence ANPP, suggesting that semi-arid rangeland could be resilient to temporary drought up to 30% rainfall reduction that might be anticipated because of climate change. On the other hand, owing to greater bare soil surface in severe drought (60% RI) plots, ANPP reduced by 58.9% and 50.8% in 2013/14 and 2014/15, respectively, compared with the 0% RI. This result confirms previous findings of Rakefet et al. (2012), which showed that ANPP was not affected by a reduction of 5–35% annual precipitation. However, in the 60% RI treatment, a significant reduction was observed in the ANPP with shifted species composition, dominated by dwarf patchy species. These findings concur with other reports (Harper et al. 2005; Rakefet et al. 2012), who noted a patchier environment under drought conditions. It is well documented (Snyman 1999; Bennie & Hensley 2001) that in arid lands biomass production is limited by climatic constraints, mainly precipitation, suggesting functional relationships between ANPP and precipitation (Milchunas & Lauenroth 1993; Knapp et al. 2006). This suggests that resource overutilization and overgrazing in these areas may result in lost biodiversity (Khumalo et al. 2012) and possible damage to the ecosystem (Meissner et al. 2013).

Similar to the current finding, Hassen et al. (2006) reported a significant influence of moisture stress over 50% on major cultivated pasture forages. However, although yield was significantly reduced in the 60% RI, *Eragrostis lehmanniana* seemed to be more stable. The abundance of genus *Eragrostis* in moisture stressed regions conforms to other reports (Rafay et al. 2013). Mwendia et al. (2013) noted that a reduced water supply resulted in lower DM yield due to reduced leaf area, gas assimilation, stomatal conductance and tissue moisture status in Napier (*Pennisetum purpureum Schumach*.) cultivars (Bana and Atherton). The lower plant height and reduced DM yield observed for grass species in the 60% RI are explained by a number of potential mechanisms. Moisture limitation is a primary factor that determines plant growth (Heisler-White et al. 2008), and retards root development since 50–80% plant growth occurs below the soil surface. Although through niche differentiation, species diversity increases complementarities (Throop et al. 2012) and total resource use, dwarf shrubs have shallow roots and may compete with grasses under moisture stress. Previous reports indicated that differences in morphological characteristics led to variation in the relative yield performance of functional groups (Throop et al. 2012). The role of species diversity on yield improvement has been reported inconsistently. Most reports showed positive effects (Diaz et al. 2006; Hatier et al. 2014), while a few reported negative effects (Wilkes 2008; Throop et al. 2012). This might be owing to competition for limited moisture, sunlight and plant nutrients. This finding agrees with those of Throop et al. (2012), who reported a reduction of yield from heterogeneous pasture under moisture stress. The higher species diversity in the 60% RI is attributed mainly to the emergence and invasion of deep-rooted shrubs and forbs.

The nutritive values of forage species are highly variable and differ according to plant species, maturity stage and environmental factors such as seasonality, sunlight, soil type, temperature and water availability (Buxton 1995; Hatier et al. 2014). In this study, the chemical composition and digestibility of the forage species showed great variations in response to the RI treatments. This variation is attributed mainly to differences in forage species that reached maturity at different times during the plant growing season. The finding in the control treatment is comparable with values reported for South African forages under natural moisture stress conditions (Bredon et al. 1987; Gemeda & Hassen 2014). Generally, dominant grasses were lower in their CP concentration, and most of them had lower CP concentration than the lower threshold level (70 g kg^{-1} DM) that is required for maintenance of ruminants (Van Soest 1994). Low CP concentration in most grasses might also have affected the digestibility of the forages. On the other hand, cell wall concentrations (NDF, ADF and ADL) were high for grasses and moderate for forb and leguminous shrubs in the 30% and 60% RI treatments. Similar to the current findings, Küchenmeister et al. (2013) reported that nutritive values of leguminous forages were less likely affected by temporary moisture deficit since legumes are more drought tolerant than grasses because of their tap root system. Regardless of grass species differences, there was a tendency for CP and ADF to increase with reduction in precipitation (Figure 6.2). The results found in the study agree with previous reports (Buxton 1995; Matias et al. 2013). The exceptional lower CP concentration observed in the 15% RI plots is attributed to higher DM production at 15% due to the dominance of grass species. The improvements in some of the chemical components (such as CP) in the 60% RI plot might be for two reasons. First, sugars produced in the root or crown reserves would be used to support basic plant functions instead of growth during drought (Halim et al. 1989; Hatier et al. 2014). Second, more leguminous dwarf shrubs and forbs were recorded in the 60% RI plots, which may have contributed to increased CP concentration. On the other hand, reports indicated that short-term drought may have more impact on DM than forage quality (Buxton 1995; Knapp et al. 2006; Küchenmeister et al. 2013).

The improvement in CP concentration under drought conditions (30% and 60%) might have resulted in increased total gas production and thus improved organic matter digestibility. Similar to the present findings, Marais (2005) reported reduced fibre concentration and improved gas production and digestibility in cultivated forages under water-stressed conditions. However, digestibility is generally reported to increase in reduced precipitation circumstances (Buxton 1995; Seguin et al. 2002; Halim et al. 1989; Matias et al. 2013; Mwendia et al. 2013). Halim et al. (1989) associated the increased digestibility in Lucerne *(Medicago sativa)* with reduced cellulose concentration and increased hemicellulose in the cell wall of moisture-stressed plants. Increased gas production from forages grown under natural desert conditions were reported previously (Gemeda & Hassen 2014). Other researchers noted that no relationship was found between NDF and moisture stress (Seguin et al. 2002; Matias et al. 2013). This may imply that other than the changed precipitation, temperature may affect this relationship (Matias et al. 2013). This study suggests that there are wide variations in and between species in feed digestibility and gas production in response to different moisture-stress conditions.

6.5.2. Conclusion and management implications

This study revealed that forb and shrub species withstand early drought, which gives them a competitive advantage in easily encroaching drought-affected plots. Hence they can play a significant facilitation role in spring and early summer. However, under severe moisture stress, forbs and shrubs disappeared and *Eragrostis lehmanniana* dominated the drought-affected plots. Prior establishment of nutritionally important pioneer species (drought tolerant) important to create microclimatic conditions for species that dominate later stages in order to promote grass species. The results suggest that in the future climate change scenarios, a 30% to 60% reduction in rainfall would adversely affect basal cover and forage yield, while the reduction in moisture (30 to 60%) would improve CP and IVOMD. Identifying drought-resilient species such as *Eragrostis lehmanniana* is important in order to produce reasonable forage yield without significant decrease in quality under predicted moisture-deficit scenarios in this semi-arid region.

CHAPTER 7

Conclusion, recommendation and critical evaluation

7.1 General conclusion and recommendation

The focus of this research was on reinforcing information in the subtropics on the relationships between selected land-use types, grazing management practices, and expected reduction in rainfall owing to climate change on aboveground herbage biomass and forage quality, soil C and N storage and the status of selected soil nutrients. Significant variations in plant nutrients and herbage yield and quality were observed over a wide array of land-use types, precipitation changes and grazing management practices in grassland ecosystems of these arid and semi-arid areas. Higher soil carbon (C) and nitrogen (N) are usually required to maintain soil aggregate stability, nutrient retention and recycling. The balance between C and N is essential for proper functioning of grassland ecosystems and to sustain rangeland productivity. In no-tillage systems, macro aggregates have a greater proportion of C that is physically protected from microbial attack. Findings from this study showed that no-tillage land-use types such as multipurpose tree plots *(L. leucocephala)* and exclosure had higher soil C (1.41 Mg ha^{-1} yr^{-1} vs 0.73 Mg ha^{-1} yr^{-1}) and N storage rate (0.11 vs 0.05 Mg ha^{-1} yr^{-1}) than in the crop land (CL) and cultivated pasture land (CPL) for C (0.40 vs 0.04 Mg C ha^{-1} yr^{-1}) and N (0.33 vs 0.03 Mg N ha^{-1} yr^{-1}). This is related mainly to fewer disturbances through cultivation and the increased amount of plant biomass residue that returns to the soil under no-tillage land-use farming systems. The strong positive linear relationship that was noted between soil C and N in various land-use and grazing management systems was consistent in semi-arid areas over two sites. Losses of C and N induced by removal of plant biomass due to continuous cultivation (as in cultivated pasture and croplands) in semi-arid areas would be more rapid, and this would be exacerbated by predicted climate change. Conversely, planting multi-purpose trees such as *Leucaena*, and livestock exclusion would probably improve farm agro-forestry and grassland productivity by ameliorating plant nutrients and soil ecological functions, while promoting sustainable crop and animal production.

Research undertaken in the past on livestock grazing effects on soil properties did not reveal conclusive results, because the observed effects of grazing on soil nutrients were not consistent, and were sometimes controversial, depending on management practice, soil type and climate conditions. In the present study, light and heavy grazing showed profound negative effects on C and N storage, as well as on other soil properties. In particular, heavy stocking rates were associated with higher soil penetration resistance and low water infiltration rates. The impact of animals treading and trampling under heavy stocking rates on soil compaction was prominent at the top 8-cm soil layer. Such high compaction in this layer resulted in a lower infiltration rate, which may have led to reduced effective rainfall and therefore low nutrient cycling. Although the differences between light and heavy stocking rates in the study were not high, heavy grazing has more evident and overwhelming effects on vegetation and soil than light grazing. Reduced soil nutrients and water infiltration and increased soil compaction under heavy grazing reduce land productivity, and ultimately affect crop and animal production. These results are important because the stocking rate that is practised in most parts of the continent, including South Africa, is usually higher than the carrying capacity of grazing lands. Resource overutilization through overstocking and overgrazing generally leads to C and N losses, soil degradation and vegetation changes, and ultimately impedes crop and animal productivity, and need to be taken into consideration in grassland management in sub-Saharan Africa.

In addition to grazing at contrasting stocking rates, timing of grazing played a significant role in influencing the physical and chemical characteristics of soils. Findings from this study showed that grazing early in the plant-growing season (spring and summer) reduced C and N, water infiltration rate, nutrient retention and recycling, and increased soil compaction, in comparison with grazing late in the growing season and exclosure treatment. Long-term accumulation of SOM in the exclosure resulted in improvements of soil C, N, and water infiltration rate, but also in reduced soil compaction. The exclosure period is context specific, depending on the needs of society for agricultural land (including grazing), soil type and climactic conditions. Based on the current findings and previous reports, a vegetation rest period of 1–2 years, followed by three consecutive years' herbage utilization would improve soil organic matter, minimize land degradation, and mitigate nutrient and soil C losses from grazing-induced vegetation and landscape changes. Thus, in sweet-pasture areas, where forage quality is less likely to be affected by moisture stress, grazing when vegetation is fully matured – as was observed in the winter grazing treatment – would be a promising management option to optimize livestock production, while minimizing the negative impact on the physical and chemical properties of soils. Most grazing lands in sub-Saharan Africa have been degraded because of poor grazing management, overstocking and overgrazing, soil fragility and climate variability. Rehabilitation of such lands could be achieved through utilization of fodder at full growth stages, which are resilient to grazing pressure and short-term strategic exclusion of livestock in order to improve C and N storage while maintaining natural resources and providing ecosystem services and functions. This is important because of the presence of vast grazing lands in Africa and higher vulnerability and soil fragility of such areas due to poor grazing management and precipitation variability. Exclosure is generally recommended as one of the options to protect soil degradation, and improve plant nutrients (particularly C and N), which ultimately promote sustainable crop and animal production. However, the duration of exclosure may be site-specific, and need to be further investigated to establish the optimum period that ensures sustainable utilization of grazing resources in arid and semi-arid areas.

In addition to belowground plant nutrients, this study examined the effects of changes in precipitation and defoliation intervals on aboveground primary productivity (herbage yield and nutritive value), SWC, and rain-use efficiency (RUE) of herbage in semi-arid grassland. In particular, monitoring species dynamics and the nutritive quality of dominant species under various simulated drought and management conditions would help to identify drought-resilient species with better yield and quality. This might contribute towards improving the productivity of mixed native pastures under predicted climatic conditions for the region. The observed variability within and between species in this study would provide opportunities to select and assemble adaptable species with better yield and quality for future climate change developments. Although grass species contributed significantly

(67%) to the total herbage supply, forbs played a major role by occupying early niches and protecting against surface water evaporation. Thus, conservation of dominant forb species is important for revitalizing grassland through restoring biodiversity. Although mixed rangeland species were found to be resilient to mild (30%) drought conditions, the simulated severe drought condition (60% RI) had detrimental effects on herbage yield. Nonetheless, the degree of sensitivity of species to moisture stress varied a lot. Species such as *Digitaria eriantha* and *Setaria sphacelata* were affected negatively, while *Eragrostis* species were relatively stable in terms of yield and quality under the simulated drought conditions. On the other hand, some nutritive value (CP, IVOMD) aspects that are important to support animal growth improved under moderate to severe drought conditions. That is perhaps why mixed rangeland pastures could stay sweet for season-long grazing in semi-arid areas of South Africa. The observed variations within and between species in response to drought conditions would present an opportunity to select and assemble drought-resilient species with better yield and nutritive quality. For example, further evaluation of the *Eragrostis* species would be essential in selecting adaptable and better yielding species and ecotypes that could be developed into cultivars.

Water is one of the limiting factors for crop and animal production in arid lands because rangeland productivity in terms of soil health and vegetation composition depends mainly on the availability of water resources and efficient utilization. Soil-water content (SWC) was one of the aspects that were investigated in this study. SWC showed a negative climatic water balance (precipitation, PET minus evapotranspiration, ET) over the study years, suggesting that the study site is moisture stress majorly during the active plant growing period because of high atmospheric water demand driven by ET. Hence, irrespective of the drought level introduced, the SWC was consistently lower in the top soil layers (0–20 cm) than the sub-soil layers, implying that high temperature-induced ET is one of the major constraints for efficient water utilization in the study site and generally in arid and semi-arid areas. In this study, intercepting 30% and 60% of the incoming rain resulted in reduction of herbage yield by 89% and 36.4%, respectively. Herbage yield was generally lower in early and late plant-growing seasons. Regardless of defoliation intervals, mid-season harvests (January and February) showed higher herbage yield, and were proportional to the amount of rainfall. The positive linear relationship found between herbage yield and precipitation indicates that precipitation is one of the most limiting factors for plant production in this environment. Improved RUE under severe droughts (60% RI) is an indicator of the presence of dominant species that are capable of surviving under moisture-stressed arid conditions by utilizing limited soil moisture efficiently.

7.2 Critical evaluation

The livestock industry in South Africa is predominately (about 80%) based on grasslands, with the gross value of livestock products having increased by 185% from 1995/2000 to 2006/2010. Despite great potential for livestock production on extensive grasslands, South Africa is a net importer of meat. For example, 231,900 tons of beef were imported from Brazil and other South American countries in 2009 (Meissner et al. 2013). Many livestock products (beef, mutton, fleece and hides) are derived directly from grasslands. In other words, for sustainable animal production and an economically functional industry, sustaining rangeland productivity is vital. This study investigated major plant nutrients that maintain soil ecological functions, including soil C and N stocks, under natural rangeland on a long-term grazing management experimental site. The study was undertaken under various biotic and abiotic factors that could vary in time and space, and might have influenced the rangeland condition. One of the limitations of this study was lack of real replication of treatments owing to the large strip size as the treatment was established 75 years ago. If initial soil sampling and analysis had been conducted for soil carbon and nitrogen, one would have scrutinized before and after treatment effects in order to quantify the rate of C and N sequestration per annum from the grazing management systems.

Because South Africa is located in a semi-arid sub-tropical climate, the country has been frequently threatened by moisture stress, even in main plant-growing seasons. In this study, forage yield and quality showed great variations owing to changes in precipitation. For example, the two most dominant grass species, *D. eriantha* and *E. lehmanniana*, responded differently to the precipitation levels that were introduced in this study. Severe drought, that is, 60% RI, affected *D. eriantha* negatively, while *E. lehmanniana* remained almost stable under reduced precipitation conditions. This variation within and across species is not uncommon in similar regions owing to variations in rainfall amount, event size and distribution, soil type, and management systems. In the present study, nutritive values were conducted once at peak production time. One critical issue regarding chemical composition is that the IVOMD values were on the upper side of digestibility values for the type of forage anlayzed. Some species, for example, *D. eriantha* and *E. lehmanniana* at 30 and 60% drought levels had over 60% IVOMD values. This high digestibility values could be due to drought effects, but should be interpreted with a caution because most grass species had usually lower IVOMD values than what is reported in our study. Another critical issue is the effect of differences in phenology and maturity time of mixed species (grasses, forbs, and dwarf shrubs: leguminous and non-leguminous) of natural pasture and unfortunately this was not captured by analysing forage samples in different seasons. Forbs are pioneers, while grass species become dominant at sub-climax and climax stages, which made it difficult to conduct chemical value analyses planned for forage samples at peak season identification. It was also difficult to compare early season flowering species with peak season maturing or climax stage species, as plants mature at different stages in the growing season. For future studies, chemical composition may be required to be conducted at least in the mid and late plant-growing seasons, perhaps for dominant plant species. Inclusion of parameters that capture plant phenology and physiology for dominant plant species that maintain community structure and stability would also provide more information about drought sensitivity.

Another critical limitation in this experiment, similar to many other studies conducted in short time scales, is the short duration of the study because the responses of plants vary from season to season and year to year whereas ecological responses to precipitation changes over the long term are likely to be more complex. This is particularly true of heterotrophic respiration, which is affected by moisture-deficit-induced reductions in soil nutrients and fertility as a function of lower aboveground net primary productivity. Knowing how ecosystems respond to long-term extreme moisture stress is important, given that climate models predict an increase in the frequency and magnitude of rainfall events in the future, particularly in arid areas of South Africa. There is a need to use such data to parameterize crop and pasture models in order to predict the impact of future climate and management change on biomass production, nutritive value of forages, and soil C and N stocks. Equally, there is a need to maintain the rain manipulation experiment and continue monitoring vegetation and soil by analysing at least once every five years the responses in terms of soil C, N and chemical composition, as well as herbage yield and quality over the long term.

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