RESPONSE OF SOIL PROPERTIES TO RANGELAND USE IN GRASSLAND AND SAVANNA BIOMES OF SOUTH AFRICA

by

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TABLE OF CONTENTS

DECLARATION

I declare that the thesis hereby submitted for the degree Philosophiae Doctor at the University of the Free State, is my own independent work and has not been submitted to any other University.

I also agree that the University of the Free State has the sole right to publication of this thesis.

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ABSTRACT

RESPONSE OF SOIL PROPERTIES TO RANGELAND USE IN GRASSLAND AND SAVANNA BIOMES OF SOUTH AFRICA

A significant portion of grassland and savanna ecosystems is over-utilized by livestock, due to inappropriate rangeland management. South Africa´s rangelands are increasingly threatened by overgrazing, followed by altered grassland composition and loss of vegetation cover in the grassland ecosystem, and by bush encroachment in the savanna ecosystem. Although not all land is overgrazed, there are some parts where signs of degradation can be found. Overgrazing has detrimental effects on soil and vegetation, but these changes can be reversed or prevented by proper rangeland management practices. The causes of and the processes involved in these changes and human interactions with them are poorly understood. Literature has indicated that rangelands can recover if managed accordingly, however scientists still have much to learn about how grazing affects soil properties. Sustainable utilization of the rangeland ecosystem is based on the appropriate application of rangeland management principles that will safeguard long-term productivity and profitability of the production system at the lowest possible risk.

The main aim of this study was to investigate how soil chemical, physical and microbiological properties responded to different management systems in a clayey grassland and sandy savanna ecosystem of South Africa. For this purpose we sampled rangeland management systems under communal (continuous grazing), commercial (rotational grazing) and land reform (mixture of grazing systems mentioned) farming. Within each of these systems a grazing gradient was identified with increasing grazing pressure, indicated by indicator grass species for the purpose of rangeland condition assessment. Different grass species exist in the clayey grassland and sandy savanna ecosystems, with *Acacia* shrub and tree species being dominant in the savanna ecosystem. Rangeland condition ranged from poor, moderate to good grazing conditions. The results revealed that soils in both ecosystems responded differently to increased rangeland degradation. In the grassland ecosystem bare patches and soil crusts lead to a degradation of the soils, whereas in the savanna ecosystem bush encroachment lead to a temporary improvement of the soil quality.

As a consequence of management, soil degradation in the piosphere of continuous grazed rangeland of the clayey grassland ecosystem is driven by the deterioration of aggregates and associated SOM losses in the poor and moderate rangeland condition, as well as nutrient losses caused by lower plant cover and litter input in the sacrifice area of the piosphere. Rotational grazed camps, in contrast, showed little evidence of soil degradation, but they exhibited an early deterioration of the aggregate structures nearby the water points. Furthermore, aggregate fractionation is a sensitive indicator for detecting the beginning of soil degradation in this ecosystem. Soil degradation was less pronounced under rotational than under continuous grazing systems. Hence, soil analyses confirm that fences and appropriate grazing periods are needed to manage these rangelands sustainably.

In the sandy savanna ecosystem, results also revealed that communal farms were affected negatively by continuous grazing, which exhausted most plant nutrients especially close to the water points, when compared to rotational grazing in commercial farms. In contrast, the communal farms had more plant nutrients than commercial farms when moving away from the water points, which coincided with an increase in *Acacia* species. Only near the water points, high grazing pressure had overridden the positive effects of *Acacia* species. Hence, and in contrast to the results from the grassland ecosystem, rangeland degradation in communal farms of the savanna ecosystem improved soil quality due to bush encroachment, but at the cost of palatable grass area.

Our data also demonstrated that in both ecosystems a decrease in grazing pressure on a rangeland, such as by commercial farmers practicing rotational grazing, could stimulate microbial activity. There was a positive feedback between microbial mediated nutrient mineralization and plant growth, as all microbial biomass and activity as well as grass cover and biomass were elevated when grazing pressure changed. Results further showed that in the long-term, the sandy soils seem to be more resilient to soil degradation, indicated by less significant differences in all measured parameters between the rotational and continuous grazing systems. In the short-term, however, it were the clayey soils in the grassland ecosystem that showed evidence of resilience, as the resting times in the rotational grazing systems was obviously able to compensate or restore disturbances from high grazing pressure, which was not possible under continuous grazing management.

Keywords: bush encroachment, continuous grazing, overgrazing, rangeland management, rotational grazing, soil aggregation, soil degradation, soil microbiology, soil organic matter, soil resilience.

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CHAPTER 1

INTRODUCTION

1.1 Motivation

Degradation of rangeland ecosystems severely threatens livelihoods of African societies and economies, and the functioning of sustainable animal production in the ecosystem. South Africa's semi-arid rangelands, which are essential for livestock production, are increasingly changing and threatened by mismanagement. However, the causes of and the processes involved in these changes are poorly understood. Overgrazing is considered as the most important cause of rangeland degradation in southern Africa (Van der Westhuizen *et al*., 2005). When the production potential of rangelands is over-estimated the subsequent overgrazing will cause a decrease in palatable perennial plants in favour of less palatable undesirable vegetation. This situation leads to altered rangeland botanical composition, specifically a loss of vegetation cover in the grassland ecosystem (Han *et al*., 2008; Ho and Azadi, 2010), and invasion of woody plants causing bush encroachment in the savanna ecosystem (Kraaij and Ward, 2006; Ward *et al*., 2014). This severely threatens the economic viability of pastoralism in both ecosystems. The degree of these changes has largely been determined by management systems (Snyman, 1998, Tefera *et al*., 2010), which differ in land ownership (private or communal) and grazing intensity (rotational vs. continuous grazing).

The changes in basal cover, productivity and litter accumulation, indirectly due to livestock grazing, can negatively affect soil properties. The most obvious change that occurs, especially in the topsoil, is an increase in soil compaction and surface temperatures, a decrease in aggregate stability and infiltration rates, and a reduction in soil organic matter and thus soil-inherent nutrient supply (Snyman and Du Preez, 2005; Snyman 2006). Two main mechanisms are responsible for this: firstly, trampling of the animals compact the soil and increase bulk density and secondly, grazing reduces and alters plant cover and botanical composition and therewith the biogeochemical cycling of nutrients. Increasing bulk densities, as a consequence of animal trampling are well documented in literature. Stavi *et al.* (2008) reported, for instance, significant higher bulk densities along a trampling route in a semi-arid region of Israel, where shrub patches and inter-shrub areas had lower bulk density values. Similar effects of grazing on soil compaction have been found in Inner Mongolia (Steffens *et al*., 2008); in the semi-arid grassland of the Northern Loess Plateau (Zhou *et al*., 2010); and in a tallgrass prairie experiment with varying grazing intensities in Texas (Teague *et al*., 2011). Wiesmeier *et al*. (2012) additionally reported on the effect of reduced grazing on increased soil aggregation in semi-arid grasslands in China. Animal grazing can also alter nutrient cycling within ecosystems by the interactions between plants and the soil (Wei *et al*., 2011). The further effect of animal trampling on soil chemical, physical and microbiological properties, and the interaction between soil nutrients and vegetation are profound and needs to be understood (Tessema *et al*., 2011). However, few studies have investigated the effects of rangeland degradation on soil properties and nutrient cycling, especially in southern Africa. In fact, the crucial role of soil is often disregarded. There are also not many recent studies comparing soil properties in different rangeland conditions in response to management within different climatic regions.

1.2 Aim

This study fits into a subproject called “Vulnerability and resilience of soils under different rangeland use”. This subproject which is one of eight, functions within the research unit: Resilience, collapse and reorganization in social-ecological systems of east- and south Africa’s Savannahs (Research Unit FOR 1501), which is funded by the Deutsche Forschungsgemeinschaft (DFG) - German Science Foundation.

This study further aims to explain how sensitive and to which extent soil properties respond to different rangeland management practices (rotational vs. continuous grazing systems as practised within commercial, communal and land reform farms) in the grassland and savanna ecosystems of semi-arid South Africa, and to establish to which degree changes of the ecosystems are perceived and caused by farmers' decisions.

1.3 Objectives and hypotheses

The main objective of this study was:

- to evaluate the impact of different rangeland management systems on soil degradation, in a clayey grassland and sandy savanna ecosystem,
- to establish whether there is a difference in soil physical, chemical and microbiological properties across degradation gradients due to grazing within these different management practices, as well as between the two different ecosystems.

It is hypothesised that rangeland degradation, which ultimately leads to a loss of grazing land in both the grassland and savanna ecosystems, is controlled by decisions about land use patterns, and that both ecosystems respond differently to rangeland degradation. In the savanna ecosystem bush encroachment leads to an improvement of soil quality, whereas in grassland ecosystems degradation of the soil, which are visible as bare patches and soil crusts, proceeds with intensified management. A further hypothesis is that soil properties decline along degradation gradients, and that this effect will be pronounced most in communal farms with continuous grazing and least in commercial farms with rotational grazing. Additionally, it is also hypothesised that soil microbiological parameters might be more sensitive to soil degradation, compared to the soil physical and chemical parameters that is typically analysed for soil quality indicators, and therefore could be used more efficiently to predict and evaluate the response of soils to degradation caused by rangeland management.

In order to achieve the objectives of the study, the following outcomes are envisioned, and will be summarised within the following chapters.

- Chapter 2 provides an extensive literature review about the challenges involved in rangeland management focusing on savanna and grassland ecosystems, and the effect thereof on soil physical, chemical and microbiological properties.
- Chapter 3 will give a characterization of the research area and how the collection of the data was done.
- Chapter 4 aims to describe the effect of rangeland management in the clayey grassland biome, situated in the Thaba Nchu area on soil properties, with the focus on soil aggregates and associated C content. This chapter has been published as an original research article with the title “Rangeland management effects on the properties of clayey soils along degradation gradients in the semi-arid grassland biome of South Africa”, with authors Kotzé, E., Sandhage-Hofmann, A., Meinel, J., Du Preez, C.C. and Amelung, W. in Journal of Arid Environments 97 (2013) 220-229.
- Chapter 5 focuses on the effect of rangeland management in the sandy savanna biome, situated in the Kuruman area on selected soil properties. This chapter has been accepted as an original research article for publication in Journal of Arid Environments 120 (2015) 14-25, with the title “Rangeland management effects on soil properties in the savanna biome, South Africa: A case study along grazing gradients in communal and commercial farms” with authors Sandhage-Hofmann, A., Kotzé, E., Van Delden, L., Dominiak, M., Fouché, H.J., Van der Westhuizen, H.C., Oomen, R.J., Du Preez, C.C. and Amelung, W.
- Chapter 6 describes selected soil microbiological properties within the clayey grassland and sandy savanna ecosystems. This chapter will be submitted as an original research article for publication with the title “Soil microbiological indicators for soil resilience in different rangeland management systems in a sandy savanna and clayey grassland ecosystem, South Africa” in Soil Biology and Biochemistry, with authors Kotzé, E., Sandhage-Hofmann, A., Du Preez, C.C. and Amelung, W.

- Chapter 7 provides the thesis conclusion and includes a summary and synthesis of the observed findings, as well as recommendations for future research and limitations of the study.

CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

The world is shared by many species of plants and animals, and consequently the significance of all different biomes in the world should not be underestimated. The animals and plants that occur in these biomes are in a very delicate balance, which can easily be disturbed or destroyed. Of particular concern are the savanna and grassland biomes in arid and semi-arid areas, which are some of the most threatened and poorly maintained biomes in the world due to exploitation by urban and agricultural development, as well as the effect of global climate change (Vetter *et al*., 2006; Harris, 2010). Land degradation has been researched by various scientists in recent years, and can be defined as a decrease in either or both the biological productivity and usefulness of a particular area due to human interference (Levia, 1999). It is also described by Ayyad (2003) as the process by which habitat quality for a given species is diminished. The UNCCD (1994) further defines land degradation as the "reduction or loss, in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity and complexity of rain-fed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns such as soil erosion; deterioration of the physical, chemical and biological or economic properties of soil; and long-term loss of natural vegetation". Hoffman *et al*. (1999) has also worked extensively on land degradation in arid and semi-arid areas of southern Africa. Scholes and Biggs (2005) further define land degradation as land-uses that lead to a persistent loss in ecosystem productivity, and often result in a decline in biodiversity, especially when heavy livestock grazing takes place. The impact of overgrazing on vegetation composition and basal cover is considered a major environmental problem in many parts of the world, where overgrazing accounts for more than 30% of all forms of degradation (Wang and Batkhishig, 2014). There is an increasing demand to quantify the effects of various land-uses, like grazing on the physical, chemical and microbiological properties of soil. This is important to ensure that these land-uses sustain soil quality and maximise profitability to farmers. Van der Westhuizen and Snyman (2014) have shown that it is possible to improve land degradation by proper grazing management approaches. More specifically for this study the savanna and grassland biomes are of concern.

2.2 Savanna and grassland biomes

The FAO (2013) currently estimates that 26% of the world land area and 70% of the world agricultural area are covered by grassland ecosystems, and is home to about one billion people around the world (Egoh *et al*., 2011). Furthermore, about 20% of the global and 60% of the African land surface is covered by savanna ecosystems (Scholes and Hall, 1996). The savanna and grassland biomes are the largest and second largest biomes in southern Africa, occupying 34 and 28% of the area of South Africa, respectively (Bredenkamp *et al.,* 1996). The grassland ecosystem is particularly rich in plant species and the biodiversity and the ecosystem services that this biome produces are under significant pressure. There is an enormous challenge in conserving the rich diversity that can be found in this biome, especially with an aridity index varying between 20 and 40% (Mucina and Rutherford, 2006).

The savanna biome can be described as areas that are characterized by the coexistence of 'carbon-rich' woody and 'carbon-poor' herbaceous plants, dominated by grasses (Beringer *et al.,* 2007; Forseth, 2012). The trees in savanna ecosystems are usually drought deciduous. Several savanna types can be distinguished by their relative abundance of trees and grass, which is associated with differing rainfall patterns, height of the water table and soil depth (White *et al.,* 2000; Egoh *et al.,* 2012). Savannas are broadly categorised into two general forms: dry savanna ecosystems (also called arid or semi-arid) with mean annual rainfalls of 400-1000 mm yr^{-1}; and wet savanna ecosystems (also called moist), with mean annual rainfalls of 800-2000 mm yr^{-1} (Staver *et al*., 2011; Donzelli *et al*., 2013). The tree-grass coexistence in semi-arid savanna has been ascribed to the scarcity of resources (Sankaran *et al*., 2005; Donzelli *et al*., 2013). Specifically, water is the increasingly limiting factor, due to decreases in mean annual rainfall, while nutrient availability plays a smaller role (Sankaran *et al*., 2008). According to D'Odorico *et al*. (2006) the balance between trees and grasses in savanna ecosystems are strongly influenced by fire, where tree and shrub populations usually increase with long periods between fire episodes. Fires play a role in releasing nutrients that are tied up in dead plant litter (Beringer *et al*., 2007). The soil also provides a good thermal insulator, so that seeds and belowground rhizomes of grasses are usually protected from fire damage. The net primary productivity of the savanna ecosystem range from 4000-6000 kg ha^{-1} yr^{-1} but depends on conditions such as soil depth (Buis *et al*., 2009; Forseth, 2012). Decomposition usually occurs year-round and is quite rapid, where the annual turnover rate of leaf material can be 60-80%. This turnover is aided by the rich diversity of large herbivores normally found in savanna ecosystems, where about 60% of the biomass can be consumed in a given year (Forseth, 2012).

The grassland biome can be roughly defined as areas dominated by grasses (usually members of the family Gramineae excluding bamboos) or grass-like plants with few woody plants (FAO, 2013). Natural grassland ecosystems are also categorized by hot summers and cold winters, leading to large seasonal temperature and precipitation variations. The variation in precipitation leads to the productivity and type of grassland community that will develop (Forseth, 2012). Higher precipitation normally leads to tall grassland with a high biodiversity of grasses and forbs, where lower precipitation leads to short and arid grasslands. According to Guo *et al*. (2006), the net primary productivity in dry grassland ecosystems can be 4000 kg ha^{-1} yr^{-1}, while higher precipitation may support up to 10000 kg ha^{-1} yr^{-1}. On the other hand, Snyman (2005a) found that semi-arid grassland ecosystems (annual rainfall of 530 mm) had a variation in net primary productivity of 70-889 kg ha^{-1} yr^{-1} in a poor rangeland condition, 200-1968 kg ha^{-1} yr^{-1} in a moderate rangeland condition, and 813-2678 kg ha^{-1} yr^{-1} in a good rangeland condition.

Three selective forces control the development of vegetation in grassland ecosystems: recurring fire (Snyman, 2005b), periodic drought (Snyman, 1998), grazing by large herbivores (Kahmen and Poschlod, 2008), as well as small stock (Van der Westhuizen, 2003). These factors may have led to the dominance of hemicryptophytes in grassland with perennating organs located at or below the soil surface, which enables the plants to survive one growing season to the next. These grasses have belowground rhizomes connecting with the aboveground shoots or tillers (Raunkiaer, 1934). The grass lamina then grows upwards, with dividing meristems at the base of the leaf sheath. This means that when animals graze the grass lamina, the meristem continues to divide and the lamina can therefore continue to grow (Forseth, 2012). Grasses in this biome are often resistant against decomposition, and periodic cool, fast moving surface fires usually originating from lightning at the end of summer, contribute to nutrient cycling (Snyman, 2002). These fires may in some cases stimulate productivity as well as the germination of fire resistant seeds (Snyman, 2005b). Grazing animals also accelerate plant decomposition, whereby their manure creates nutrient hotspots that can alter the plant species composition.

Together with other grassland ecosystems of the world, savanna ecosystems account for 30-35% of the global net primary production, and are *inter alia* important as a feed source for livestock production, habitat for wildlife, provider for environmental protection, storage of carbon (C) and water, and *in situ* conservation of plant genetic resources (Field *et al*., 1998). Therefore, grassland as well as savanna ecosystems have an important effect on global element and energy cycles and together they are the basis for the livelihoods of millions of people in Africa alone. In arid to semi-arid environments even more than 75% of the land is used for livestock production (Tainton, 1999; Smet and Ward, 2006), with about 40% of grassland ecosystems in South Africa. Globally, grassland ecosystems alone house various important species and include 15% of the world’s Centres of Plant Endemism, 11% of Endemic Bird Areas and 29% of eco-regions with outstanding distinctiveness (White *et al*., 2000; Egoh *et al*., 2011). Grassland ecosystems in South Africa specifically are very rich in biodiversity (O'Conner and Bredenkamp, 1997). The rapid increase in population, combined with the effects of climate change, has heightened pressure on the world's grassland and savanna ecosystems, particularly in arid and semi-arid environments, and portions of these ecosystems on every continent are suffering from rangeland degradation.

2.3 Rangeland degradation

Rangeland degradation is a global concern, which affects not only pastoralists relying on healthy rangelands for their survival, but also those who are affected by the subsequent droughts, dust storms and commodity scarcities (Harris, 2010). Rangeland degradation can be caused by either natural climatic conditions leading to drought, or human induced factors caused by the overuse of natural resources (IFAD, 2013). Extremely heavy grazing from livestock production has often been given as one of the reasons for degradation and a subsequent decline in biodiversity in arid and semi-arid areas (O'Conner and Bredenkamp, 1997; Van der Westhuizen, 2003; Rutherford and Powrie, 2011). Degradation further results in declining functional capacity, increased poverty, as well as food insecurity. Major changes in rangeland above- and belowground morphology and soil characteristics have an added drastic effect on the primary productivity of the rangeland ecosystem, and in turn on livestock production (Lesoli, 2011).

Rangeland degradation often leads to changes in the botanical composition of grass communities, which may differ between different ecosystems. In the grassland biome, a decline in the palatable perennial plants takes place, in favour of less palatable, undesirable grasses and herbs (Van der Westhuizen *et al*., 1999; O´Connor, 2005; Snyman, 2005a), whereas in the savanna biome, degradation leads to invasion by woody plants in areas where degradation has taken place, also known as bush encroachment. This encroachment has already taken over whole areas in some landscapes and is threatening in others, thereby putting pressure on the sustainability of both subsistence and commercial livestock farming (Rappole *et al*., 1986, Noble 1997, Archer *et al*., 2001). On a global scale, encroachment may adversely influence about 20% of the world's population (Turner *et al*., 1990). In South Africa alone, this alteration of savanna and grassland biomes affects 10-20 million hectares of rangeland (Ward, 2005).

Despite this concern, most research done on this degradation problem has only focused on the effects of plant communities on grass productivity, as well as on the development of methods to reduce the abundance of sour grasses in the grassland or that of trees and shrubs in the savanna biomes. However, the response of soil ecosystems to grazing is less documented (Milchunas and Lauenroth, 1993; Emmerich and Heitschmidt, 2002) and lags behind that of the aboveground systems (Allsopp, 1999; Neary *et al.*, 1999; Snyman 2004). Recent studies have shown that research about the effects of rangeland degradation on soil properties and nutrient cycling are being neglected, with more emphasis on the aboveground effects (Archer *et al*., 2001). In addition, a great part of current literature dealing with different models explaining changes of the rangeland in South Africa and the grazing capacities, e.g. the disequilibrium theory, predominantly regard the balance or imbalance between livestock numbers in grazing lands and vegetation (Gillson and Hoffman, 2007). While there exists some agreement that changes in vegetation cover are reversible (Behnke *et al*., 1993; Vetter, 2004), almost nothing is known about the resilience of soils. The impact of soil and its interaction to vegetation is often complicated and are consequently disregarded.

2.4 Grazing pressure

Furthermore, the impact of grazing animals on ecological parameters in rangelands is spatially patterned, where the effects of grazing by domestic animals differ somewhat from those of wildlife (Butt & Turner, 2012). The domesticated herds move slower and do not stray great distances from the artificial or natural water points. This causes pressure on vegetation around water points, while the areas further away remain almost undisturbed. The result is concentric annular vegetation around water points, with increasingly degraded vegetation as one approaches the centre. Around the water point itself, where the animals remain the longest, there is often no vegetation at all, and the soil is over-fertilised by animal excrement. The negative effect of this over-fertilisation remains noticeable for many years after the watering places are abandoned (Snyman *et al*., 2013). This phenomenon that develops around water points, are called “piospheres”, with a so-called sacrifice area close to the water point (Andrew, 1988; Thrash and Derby, 1999). The grazing intensity and associated rangeland degradation generally increase with increasing proximity to the water points (Du Preez and Snyman, 1993; Lin *et al*., 2010). Many studies have used grazing gradients to investigate the effects on vegetation (e.g. Thrash 2000; Riginos and Hoffman, 2003) and relate degradation with vegetation composition change or biomass production loss. Generally, vegetation is highly influenced by variations in rainfall (Illius and O´Connor, 1999) and changes in botanical composition or basal cover are usually reversible (Abel, 1997). Recording vegetation therefore only gives a glimpse of the current situation, while soil indicators are more reliable for determining a long-term situation.

For the savanna ecosystem it has been well established that the spatio-temporal variability of water, nutrients and seed distribution is one of the key factors that drive the functioning of these ecosystems (Jeltsch *et al*., 2000). Even at larger scales, patch-dynamic processes control the co-existence of grasses and trees and, upon disturbances the invasion of bush takes place (Wiegand *et al*. 2006; Meyer *et al*. 2009). The specific role of soil heterogeneity for tree recruitment is not well understood, even though Britz and Ward (2007) found a strong relationship between soil texture and bush encroachment. They suggested that soils with lower soil-water content and nutrient availability like sandy soils and clay-pan sites were to a certain extent resistant against bush encroachment. According to Wiegand *et al*. (2006) and Moustakas (2006), bush encroachment in many arid and semi-arid environments is an integral part of savanna dynamics. Any disturbances like fire or grazing, create space, which makes water and nutrients available for tree germination. With enough rainfall available, a bush encroachment patch may develop. In turn, inter-tree competition may result in the reformation of grassy patches within an area encroached by bush. As a result, vegetation changes may be reversible. Yet, the reversibility of such processes implies that soils are resilient to both changes in vegetation type and to disturbances that have induced the vegetation change (Ward *et al*., 2014).

Once woody plants are established, they may alter the soil and microclimate in their surroundings, forming so-called “islands of fertility” (Schlesinger *et al*., 1990; Scholes and Archer, 1997; Archer *et al*., 2001; Hong *et al.*, 2005). Many processes account for this: (1) woody plants pump nutrients into their canopy and redeposit these nutrients in the upper soil layers via litter-fall and canopy leaching, (2) tree canopies with their leaves, scavenge nutrient-rich atmospheric dust, (3) birds and mammals searching for shade and food concentrate their excrements close to the trees and (4) trees in arid environments commonly overcome nitrogen (N) limitations by symbiotic N_2 fixation, and up to 10 times more C is now stored in the woody plants relative to pristine grassland, accompanied by rising root and litter mass (Archer *et al*., 2001; Liu *et al*., 2005). Hence, soil resources are elevated in sub-canopy rather than in inter-canopy spaces (Miller, 2004). Biological soil crusts can additionally stabilise the soil surface and increase nutrient retention in sub-canopy spaces (Dougill and Thomas, 2004; Berkely *et al*., 2005; Veste *et al*., 2006). As a result, the spatial variability of soil nutrients increases with increasing bush encroachment (Hagos and Smit, 2005), where mineral adsorption and nutrient cycling in the surface layers explain much of the resilience of soil to additional chemical changes (Dougill and Thomas, 2003). As tree abundance and density continues to increase, these “islands of fertility” may finally grow together, resulting in a homogenisation of soil properties, but possibly in an even better nutrient supply than the native grassland had possessed.

Farmers normally combat shrub invasion, in order to save their grassland. Fires alone are often inadequate to convert a dense shrub stand back to grassland. Chemical or mechanical management of woody plant regrowth to promote subsequent herbaceous growth must accompany such a process. Fires may stimulate grass growth in bare soil patches, whereas chaining promotes herbaceous production in existing vegetation patches (Ansley *et al*., 2006). Rangeland management has thus become an important co-driver for savanna and grassland ecosystem maintenance. However, it may prevent the aggregation process and thus the necessary restoration of soil properties. In locations with heavy bush encroachment, farmers have shifted to charcoal production, but bush removal can be so severe that bare soil patches remain. Hence, soil degradation rather than soil restoration proceeds. Little is known on the early indicators, rates and threshold values that characterise changes in soil properties.

It is also important to note that animal grazing patterns can exert both positive and negative effects on vegetation. Positive effects comprise: (1) a slight loosening of the soil surface; (2) animal hooves press seeds into the soil and promotes seed setting by plants; and (3) animal manure fertilizes the soil (Savory and Butterfield, 1999). These positive effects are however only evident with an intermediate grazing intensity. With overgrazing, the soil can be loosened too much and is exposed to wind erosion, whereby valuable topsoil is lost. With undergrazing, the loosening of the soil surface is insufficient and unwanted vegetation, like mosses can invade the area (Van der Wal & Brooker, 2004). Similarly, the pressing of seeds to the optimal depth occurs only with a moderate stocking rate. This intermediate disturbance model is discussed in detail by Huston (1994). Under rational use with moderate occupancy, optimal utilization could be sustained without damaging plant cover of the vegetation. Short grazing during each season most closely resembles the original use by the wild herds, which constantly change the location and never remain for very long at any one place (Savory and Butterfield, 1999).

2.5 Rangeland management systems

Most of the dramatic changes mentioned in the ecosystems of concern are consequently driven by rangeland management practices. Especially changes in land use rights, accessibility and thus the patterns of rangeland use, frequently go along with different decisions on the intensity of grazing and the motivation and feasibility to monitor and control ecosystem dynamics. This leads to different rangeland management systems being developed in the world, whereby both natural conditions (climate, soil conditions, topography) as well as social factors (sociocultural values, political belief, level of technological development, population trend, changes in the cost/price/relationships) play a role (Kuhnen, 1982).

According to Snyman (personal communication)[1] in general, there are two rangeland management systems in the world, apart from game farming (Smet and Ward, 2006; Tefera *et al*., 2010): commercial and communal livestock farming. These systems differ mainly in the management of grazing resources, ownership as well as their outputs. The commercial farming sector is well developed, capital-intensive and largely export orientated, with commercial areas being divided into fenced farms, owned by individuals, and then further subdivided into a number of camps where rotational grazing is usually practiced. Stocking rates tend to be more conservative and are adjusted by the farmer to ensure sustainable production. Communal farming differs distinctly from the commercial areas in their production systems, objectives and property rights (Smet and Ward, 2006). The communal production systems are based on pastoralism and members of a community share the grazing areas. There are often unclear boundaries, with continuous grazing being practiced. The outputs and objectives of livestock ownership are diverse, and include draught power, milk, meat, dung, cash income and capital storage, as well as socio-cultural factors. Higher stocking rates in communal areas are common (FAO, 2005). This communal system is under criticism in terms of exceeding the grazing capacity of the land and risking rangeland degradation (Palmer *et al*., 1999; McGranahan and Kirkman, 2013).

In addition, a third rangeland management system can be found in South Africa on farms that were allocated within the scope of the post-Apartheid land reform, (e.g., in Malawi, Zimbabwe) (Adams and Howell, 2001; Walker, 2002). These farms have been obtained by

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either an individual or a group of people, through political redistribution programs. Stocking rates are normally lower than on communal farms and no clear management systems are present. Rotational grazing as well as continuous grazing systems can co-exist, depending on the specific community's way of thinking at a particular time (Lohmann *et al*., 2014).

The different rangeland management systems of South Africa's grassland and savanna biomes have undergone various changes over the past century (Palmer *et al*., 1999). After indigenous users had been dispossessed and displaced in the 19th century, extensive livestock farming on white-owned ranches and intensified land use in crowded African reserves shaped human-environment relations in the 20th century. From the 1920s onwards, white-owned farms were modernized, implementing for example fences, boreholes and instituting rotational grazing systems (Archer *et al*. 1995). Modernization of white-owned farms accelerated after 1948 in the course of governmental programs and led to mechanization and intensification. At the same time, agricultural production in African reserves and later homelands was affected by rapidly increasing population densities (due to population growth and the resettlement of farm workers to these homelands). The present situation is characterised by highly diverse land use changes: commercial farmers experiment with new income generating strategies (e.g. tourism and game farming but also charcoal production), communal farmers seek to combine marketable production with subsistence production and a number of land reform projects (organized in common property associations) attempts to initiate some form of agricultural production on their newly obtained farms. All these changes can exert pressure on the soil and this can ultimately lead to soil degradation. In both the grassland and savanna biomes, the further ecosystem development cannot be understood independently from the socio-economic conditions. Nevertheless, almost nothing is known on the effect of possible degradation of different rangeland management systems (e.g. pasture held in common property, municipal commonages, commercial farms, nature reserve areas and resettlement farms). Neither do we possess knowledge on the rate at which soil properties change when rangeland utilization intensified within these systems, nor on the reversibility of these changes.

2.6 Soil degradation

Studies in South Africa’s rangelands indicated that soil degrades with declining rangeland conditions: soil compaction and surface temperatures increased, aggregate stability and infiltration rates decreased, and soil organic matter (SOM) and thus soil-inherent nutrient supply can be reduced by up to 22% in the topsoil (Snyman and Du Preez, 2005; Snyman 2006). The rate constants of such changes are not well explored yet and the functional relationships to gradients of different stocking intensities require further investigation. Nevertheless, the gross effects coincide with studies from other rangeland regions on the effect of overgrazing on soil properties (e.g. Bauer *et al*., 1987; Milchunas and Lauenroth, 1993; Hibbard, 1995; Archer *et al*., 2001; Russel *et al*., 2001; Mills and Fey, 2003; Savadogo *et al*., 2007). When cattle remove the herbaceous cover, the aboveground phytomass that protects the soil against splash erosion is also removed (Thurow *et al*., 1986), and new C inputs from litter and roots decrease (Mills and Fey, 2003; Snyman and Du Preez, 2005). The resulting losses of SOM are crucial in this semi-arid environment, because the SOM additionally prevents soil erosion through aggregate stabilization (Thurow *et al*., 1986; Feller and Beare, 1997; Six *et al*., 2000). In addition it provides the majority of nutrients for plant growth (Archer *et al*., 2001) and consequently results in an increased potential of soils for physical crusting (Mills and Fey, 2004). The crusts form after an initial breakdown of soil aggregates under the influence of rainfall and a subsequent hardening phase during drying (Fox *et al*., 2004). These physical soil crusts hinder infiltration, increase erosion and impede vegetation establishment (Mills and Fey, 2004) and hence the spatial and temporal patterns of soil properties and vegetation resettlement change. Due to the high variability in these patterns and because of numerous use demands, rangeland soils represent an exceptional challenge for soil quality assessment (Manley *et al*., 1995).

As the sustainability of rangeland resources become an increasing concern for rangeland managers, the response of soil to overgrazing as well as rangeland degradation need to be quantified to develop suitable grazing practices. According to Van der Westhuizen *et al*. (1999) and Snyman (2005a) overgrazing is often caused by the over-estimation of the production potential of a rangeland, which then leads to a decline in the palatable perennial plants in favour of less palatable, undesirable vegetation. These changes in the rangeland condition usually have negative consequences, like increased soil compaction (Warren *et al*., 1986a, 1986b; Thurow *et al*., 1988; Chanasyk and Naeth, 1995), reduced soil aggregate stability (Warren *et al*., 1986b; Russel *et al*., 2001; Lal and Elliot, 1994), decreased soil fertility (Dormaar and Willms, 1998; Ingram, 2002) and lower SOM content (Du Preez and Snyman, 1993, 2003; Whitford, 1996; Snyman, 1999). The latter is indeed one of the most important factors influencing rangeland ecosystem functioning, since SOM improves soil structure (Thurow *et al*., 1986) and this in turn increases water infiltration (Smith *et al*., 1990) and reduces soil erosion through aggregate stabilization (Chevallier *et al*., 2004). This leads to better water-use efficiency by the rangeland ecosystem, which is crucial in semi-arid regions (Reicosky *et al*., 1995; Williams *et al*., 1998; Okatan and Reis, 1999; Snyman, 2005a). Soil organic matter also plays a huge role in soil fertility (Teague *et al*., 1999; Whitford, 1996). In fact, a few studies have evaluated the effects of grazing or rangeland degradation on SOM and its relationship to water and nutrient cycling and related plant productivity (Milchunas and Lauenroth, 1993; Manley *et al*., 1995; Dormaar and Willms, 1998; Schuman *et al*., 1999; Emmerich and Heitschmidt, 2002). Since an important part of the overall ecosystem sustainability occurs belowground, recovery is linked to the soils physical, chemical and biological functions and processes (Singh and Coleman, 1973; Neary *et al*., 1999).

2.7 Effect of overgrazing on soil properties

Several studies investigated by Pei *et al*. (2008) have shown that overgrazing of rangelands leads to an overall decline in soil chemical, physical as well as biological properties, often resulting in dramatic changes in vegetation and modifications in nutrient cycling (Lavado *et al*., 1996; Chaneton and Lavado, 1996; Zhou *et al*., 2010). This then in turn causes permanent degradation of land productivity and destruction of the ecosystem (Su *et al*., 2004; Pei *et al*., 2008). In arid and semi-arid ecosystems the limited amount of water leads to root growth limitations, and the low rates of net primary productivity are most often due to the low availability of nutrients (Milchunas and Lauenroth, 1993; Ingram, 2002; Schenk and Jackson, 2002; Snyman, 2005a). Most nutrients taken up from the soil by plants in arid ecosystems come from nutrient cycling instead of from parent material (Charley and Cowling, 1968) and this is closely related to water availability. Typically during wet periods, these systems can be described as producing ‘‘pulses’’ or ‘‘flushes’’ of nutrients from mineralization (Singh and Coleman, 1973; Sparling and Ross, 1988; West *et al*., 1989). Nutrient cycling is affected by the same factors that are responsible for decomposition of plant material and consequently primary productivity (Ekaya and Kinyamario, 2001).

2.7.1 Soil physical properties

The effect of grazing animals on specifically soil physical properties is most noticeable at the soil surface, and also at high stocking rates, especially when the soils are wet. The soil physical properties that depend on pore continuity, such as water infiltration and aeration, are the most sensitive to compaction by grazing animals (Greenwood and McKenzie, 2001). The effect of animal trampling can lead to decreased soil permeability of both air and water. According to Heathwaite *et al*. (1990) as well as Pietola *et al*. (2005) the infiltration capacity of the soil is lower and this may result in higher rates of surface runoff during heavy rains. This can lead to more soil erosion and nutrient losses, a problem often related to overgrazing (Wells and Dougherty, 1997; Kurz *et al*., 2005; Zhou *et al*., 2010). Compaction reduces the volume of soil in the plant rooting zone which can store oxygen and water in these pore spaces, thereby limiting the rooting volume of the plants. The remaining pore spaces will as a consequence, have relatively fewer large pores (those which store air) and relatively more small pores (those which hold water). This goes together with soil aggregation, which affects the physical and hydrological functioning of soil. Particularly large macro-aggregates are vulnerable to breakdown when animal trampling takes place. In a few studies, aggregates served as an indicator for the resilience of a soil against degradation (Bossuyt *et al*., 2002; Six *et al.*, 2004; An *et al.,* 2009; Stavi *et al*., 2011). It could thus also serve as an indicator of rangeland resilience against intensified grazing, especially where low rainfall and high evaporation is present.

In line with this, Kurz *et al.* (2006) also found that grazing animals can alter the hydrology and the drainage pathways in a biome by compacting the topsoil, which is indicated by an increased bulk density of 8-17% and decreased macroporosity of 57-83% (Singleton *et al*., 2000). Overall, Kurz *et al*. (2006) concluded that the presence of cattle had a longer lasting effect on the soil hydrological parameters measured than on the nutrient concentrations in the soil. Gifford and Hawkins (1978) wrote a review of the impact of grazing on infiltration, and found that steady state infiltration rates of light to moderately grazed pastures were about 75% of those for ungrazed treatments. They also found that infiltration rates of heavily grazed rangelands were only half of those for ungrazed areas. Snyman and Du Preez (2005) also found that rangeland degradation decreased infiltration rates. Increased runoff linked to decreased infiltration often leads to erosion, which in turn is associated with a loss of nutrients as well as decreased plant available water. The increased runoff from grazed rangelands is caused by the loss of macro-pores open to the soil surface as well as by removal of aboveground vegetation (Greenwood and McKenzie, 2001).

2.7.2 Soil chemical properties

The effect of grazing animals can also transform the features of rangelands as a nutrient source, where overgrazing can cause reduced plant-cover, which is the main source of plant nutrients. These plant nutrients can be mobilized and removed from the grazed rangeland by overland-flow of water by effecting a spatial and chemical re-distribution of nutrients and, sometimes by causing enough physical damage to reduce grass growth (Drewry and Paton, 2000). According to Blank *et al.* (2007), soil nutrient availability is often patchy and both spatially and temporally variable in rangeland soils, and can be explained by a combination of decreased root uptake in relation to mineralization, differences in soil water content with season and elevation, and nutrient release from vegetation and soil. The effects of grazing animals on nutrient losses to water are reported to range from not measurable (Owens *et al*., 1989) to considerable (Heathwaite and Johnes, 1996). This variation is probably due to the great number of variables involved in the nutrient loss process, and to the substantial effect the relative timing of management and weather has. Milchunas and Lauenroth (1993) found, for example, that grazing had no effect on total nutrient stocks, possibly because total nutrient stocks may not be reliable estimates of nutrient availability. Marrs *et al*. (1989) also described inconsistent results in total and available soil nutrient contents in upland peatlands that was grazed. An enrichment of nutrients, especially in the inner parts of the piosphere, due to manure of the animals (Perkins and Thomas, 1993), as well as through a centripetal nutrient flow to the waterholes at natural water points, can also take place. The application of supplementary feeding given to grazing animals, usually at the water points, can furthermore play a role in higher values for specifically phosphorus (P) and calcium (Ca), as well as pH (Smet and Ward, 2006). It is also important to note that the type of plant cover (grass or tree), can lead to increased values for soil nutrients. In bush-encroached areas, the combination of relocation and surface root turnover, as well as shedding of leaves and seeds, will act together as a source of nutrients (Mampholo, 2006).

Contrasting results on the effect of rangeland management on C and N in grassland ecosystems are found in literature (Snyman, 1999). With increasing grazing intensities, decreasing C (Abril and Bucher, 1999; Neff *et al*., 2005; Steffens *et al*., 2008), as well as unchanged (Binkley *et al*., 2003; Barger *et al*., 2004) and increasing C (Reeder and Schuman, 2002; Conant *et al*., 2005) has all been found. Fewer results are available concerning N in grassland ecosystems affected by management, but they also vary. Increasing (Bauer *et al*., 1987) unchanged (Schuman *et al*., 1999; Barger *et al*., 2004) and decreasing (Frank *et al*., 1995) N is described with increasing grazing intensity. Bauer *et al*. (1987) showed a negative correlation, while Abril and Bucher (1999) observed positive correlations between organic C and N in grazed compared to ungrazed areas. It has been generally conceded that high grazing pressure reduces the growth rate and reproductive potential of individual grass plants, which in consequence depletes the nutrient status of the soil (Abule *et al*., 2005), however higher nutrient values can be attributed to the positive influence of animal manure, especially in the piosphere region. According to Snyman (1999) the most important factors which can contribute to a change in organic C and N (with or without grazing) includes: the condition of the rangeland, environmental factors such as soil water and temperature, and the grazing history of the rangeland.

2.7.3 Soil microbiological properties

Soil microbial communities play a fundamental role in rangeland ecosystems by regulating the dynamics of organic matter decomposition and plant nutrient availability, and are of paramount importance for the functioning and stability of ecosystems. These microbial communities can be used as an important measure of sustainable land use and are sensitive to changes in soil chemical as well as physical properties (Bardgett *et al*., 1997; Patra *et al*., 2005; Xue *et al*., 2008). It is clear that rangeland management affect the structure and activities of these microbial communities, since their abundance and activity is strongly related to the quantity and quality of available plant litter, which in turn, is related to animal grazing intensity. Su *et al.* (2004) for example showed that heavy grazing pressures resulted in loss of soil organic C and N, and subsequently in a depletion of soil enzyme activities. Mofidi *et al.* (2012) found similar results in rangelands of Iran, where lower levels of grazing had the highest biological activity. To indicate the effect of manure on soil microbiological properties, Bardgett *et al*. (1997) found in North-Wales that long-term removal of grazing animals from a rangeland resulted in a significant reduction in microbial biomass and activity in the surface soil layer, due to less manure in the soil. According to Degens *et al.* (2000) land-uses that deplete organic C stocks in soils may cause declines in the catabolic diversity of soil microbial communities. Although the implications of this for microbial processes are unknown, maintenance of soil organic C may be important for preservation of microbial diversity. On the other hand, Bardgett *et al.* (1997) concluded in their study that changes in microbial community structure are also likely to have a profound influence on organic matter dynamics and nutrient supply in an ecosystem.

Information on how grazing affects the size and composition of key microbial functional groups is very scarce, and this restricts our understanding of the actual effects of grazing on rangeland functioning. It also affects our ability to predict rangeland response to changes in grazing intensity or management practices, because the composition of microbial communities can determine their resistance and resilience to disturbances (Patra *et al.,* 2005). This was shown in a study done by Griffiths *et al*. (2000), where resilience was proved to be lower in soils with decreasing biodiversity. They showed that soils with impaired biodiversity were not resilient to persistent stress, compared to soils with higher biodiversity. Patra *et al.* (2005) also demonstrated that grazing deeply affects microbial functional groups, and are important for predicting the effects of changed grazing regimes on rangeland ecosystem functioning and response to disturbance. Microbial community biodiversity is an integral part of soil quality and crucial to maintain ecosystem function.

2.8 Conclusion

A significant portion of grassland and savanna ecosystems is over-utilized by livestock, due to inappropriate rangeland management. Although not all land is overgrazed, there are some parts where signs of degradation can be found. Overgrazing has detrimental effects on soil and vegetation, but these changes can be reversed or prevented by proper rangeland management practices. Literature has indicated that rangelands can recover if managed accordingly, however scientists still have much to learn about how grazing affects soil properties. Sustainable utilization of the rangeland ecosystem is based on the appropriate application of rangeland management principles that will safeguard long-term productivity and profitability of the production system at the lowest possible risk. In large parts of southern Africa, the rangeland ecosystem is seen as a national asset with sufficient biological potential for sustainable production. Therefore, research should develop practical guidelines, based on scientific understanding, to improve the management of livestock to minimize the detrimental effects of grazing on rangelands. Responsible or well-managed grazing practices have the potential to enhance the overall soil physical, chemical and biological quality.

CHAPTER 3

CHARACTERIZATION OF RESEARCH AREAS AND COLLECTION OF DATA

3.1 Research areas

In order to achieve the goals of this study, two research areas had been selected (Figure 3.1). The one area was in the grassland biome near Thaba Nchu with clayey soils (hereafter known as grassland ecosystem) and the other area in the savanna biome near Kuruman with sandy soils (hereafter known as savanna ecosystem).

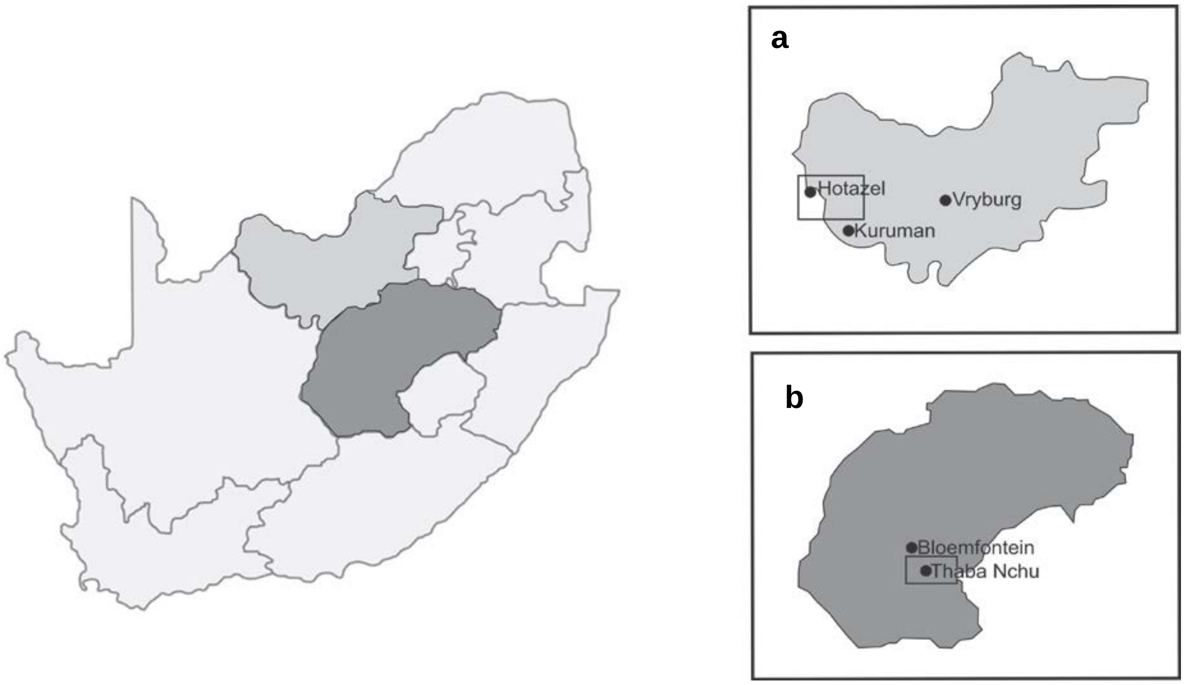


Figure 3.1 A map of South Africa showing the research areas in the savanna biome at Kuruman (a) and grassland biome (b) at Thaba Nchu.

For the grassland ecosystem, the research area was located near Thaba Nchu (latitude 28° - 29° S, longitude 26° - 27° E; with an altitude of 1400 to 1600 m above sea level), in the Free State Province, South Africa (map a in Figure 3.1). The mean annual precipitation was 553 mm (± 190 mm), with about 70% of the rain occurring in summer between January and March. The climate was relatively dry, had a low, variable and unpredictable rainfall and a high evaporation rate of 1832 mm a^{-1} (Basson, 1997). Non-perennial streams existed in the area. The soils were all classified as Lixisols (WRB, 2007), and had more than 15% clay in the A-horizons and more than 25% clay in the B-horizons, with no obvious signs of wind or water erosion. This research area belonged to the “Moist Cool Highveld Grassland Type”, which is part of the grassland biome (Bredenkamp *et al*., 1996). This grassland type is widespread and covers the central eastern part of the Highveld in the Free State. A single layer of perennial C4 bunchgrasses dominated this particular grassland type, and trees were sparse. The amount of grass cover depended beside the low rainfall and evapotranspiration on the degree of grazing. Frost, fire and grazing maintained the grass dominance and prevented establishment of trees. The land was characterized by maize and wheat production in the northwest and stock farming and subsistence farming in the east and south. Rather than soil condition or productivity, these patterns were determined by the existence of former homeland areas of Bophuthatswana, around Botshabelo and Thaba Nchu, with extensive areas of small freehold farmers, either on the former land reform farms around Thaba Nchu, or newly resettled lands around Botshabelo. The recommended animal stocking rate for the area of Thaba Nchu was 6 ha per livestock unit (LSU) (Department of Agriculture and Rural Development, 2003).

For the savanna ecosystem (map b in Figure 3.1), the research area was located near Kuruman at the border of the Northern Cape and North-West Province of South Africa, and was situated on the fringe of the Kalahari (Latitude 27° - 28° S, Longitude 22° - 24° E; with an altitude of 1050 to 1200 m above sea level). The arid climate in Kuruman received rainfall mostly in the summer months of October to March with a mean annual rainfall of 255 mm and temperature of 17.5°C, and a very high evaporation rate of 2050 mm a^{-1}. The soils were deep Arenosols with aeolian origin, underlain by calcrete (WRB, 2007), typically containing less than 10% clay. The vegetation in the area was dominated by the Kalahari thornveld and shrub bushveld (Tainton, 1999) and had been more specifically described as the Kalahari Mixed Thornveld A16 (Mucina and Rutherford, 2006), characterized by a fairly well developed tree stratum with *Acacia erioloba, Acacia mellifera, Acacia haematoxylon* (≤ 2 m height) and some *Boscia albitrunca* as the dominant trees. The shrub layer was dominated by individuals of *Acacia mellifera, Acacia hebeclada, Lycium hirsutum, Grewia flava* and *Acacia haematoxylon*. The grass cover contained species such as *Eragrostis lehmanniana, Schmidtia kalahariensis* and *Stripagrotis uniplumis*. The low precipitation had a great impact on land use in this area. The majority of the Northern Cape Province was used for stock farming including cattle, sheep or goat farming as well as mining whilst only about 4% was reserved for conservation (Hoffman and Cowling, 1990). In this research area, the estimated grazing capacity of the rangelands were 13 ha LSU^{-1} (Department of Agriculture and Rural Development, 2003). Overgrazing was one of the main causes of land degradation, with alien plant invasions posing a threat to the rich flora of the area. This was also one of the worst affected areas in terms of bush encroachment which implies that large areas of grazing land were lost, species diversity was reduced and habitats were transformed (DEAT, 2002). Various land-use activities all contributed to a loss of vegetation cover, soil erosion and ultimately land degradation. Land degradation was thus an important issue to rural communities and farmers that depend on the land for their livelihood.

3.2 Rangeland management

Communal and commercial livestock ranching are the most common rangeland management systems in both the grassland and savanna ecosystems (Smet and Ward, 2006; Tefera *et al*., 2010). The commercial farms are well developed and mainly market-orientated. Commercial farms (about 70% of all land used in the RSA) are typically managed using a rotational grazing system at moderate stocking densities. The commercial farms were surveyed and allocated to individual owners during the nineteenth century. Size and animal type (sheep, cattle) varied with time, but management structure remained more or less constant.

The communal farms belong to the former homeland Bophuthatswana, which was developed as an integral part of the Betterment Villages on land previously owned by commercial farmers (Jacobs, 2003). The communal production systems are based on pastoralism and members of a community share grazing areas. The rangeland is a common pool resource with no restrictions in stocking rates and a continuous grazing system. There are often unclear boundaries, with open access rights to grazing areas. The communal farms underlay several changes of the local land use system, from its very beginning in 1833 to the current state of low agricultural production. Basic shifts were in the 1940s during the betterment schemes, the 1970s and post-apartheid (Naumann, 2014). From 1977 to 1994 subsidization of agriculture in Bophuthatswana, the destined homeland of the Batswana people, resulted in rising stocking rates and overgrazing became a problem in the communal areas. On the other hand the infrastructure of the communal grazing lands, such as boreholes and fences, were maintained and local rangers encouraged rotational grazing management. With the end of apartheid in 1994, Bophuthatswana was reincorporated into South Africa. The work of the local rangers, who cared for infrastructure, was discontinued. This resulted in a complete deterioration of the grazing land, and fences and boreholes are subsequently broken or non-existent. The communal rangeland is currently a common pool resource with no restrictions in stocking rates and no rotational grazing system. Livestock seem to be of little economic importance, either as income source for people (most income is derived from social grants) or as contributing significantly towards nutrition. A decoupling of the social and ecological system seems to take place (Naumann, 2014).

In addition, land reform farms are a third, less important rangeland management system for livestock production. In the context of the post-apartheid land reform programme, farm units situated in the Thaba Nchu and Kuruman areas were allocated to Trusts, Close Cooperations or Communal Property Associations. These farms belong to various numbers of members, who do not necessarily have any experience with pastoralism. The grazing management system is mixed but comparable to communal farms. Domestic stock is not the main income source here.

Furthermore, in both ecosystems a small game reserve was chosen to represent a control area, ungrazed by domestic animals, in order to compare the different management systems to a natural scenario and to see how much these management systems affected the land.

3.3 Soil sampling

Three rangeland management systems as described above, were selected for this study: (a) commercial farms – CF, (b) communal farms – CO, and (c) land reform farms – LF. In the grassland ecosystem in the Thaba Nchu area, we sampled four replicates (hence 4 different farms) in every management system, treating the different farms thus as independent replicates. In the savanna ecosystem in the Kuruman area, we sampled three replicates for the communal and commercial farms, and two replicates for the land reform farms. In addition, a nature reserve – NR in both ecosystems, served as a control area, ungrazed by domestic animals. In the latter pseudo-replication was unavoidable. The management systems differ mainly in ownership, managing of grazing resources and stocking rate. Definition and selection of the land reform farms in the Thaba Nchu area proved to be challenging. After sampling it was realized that two of the farms are not strictly land reform farms as defined politically, because they were bought and not obtained via the resettlement program. On the other hand, the management of these farms clearly resembled that of a typical land reform farm, i.e., it was neither clearly continuous nor rotational grazed, depending on the individual purposes of the owner. We thus equated these farms as land reform farms.

For each farm a representative degradation gradient was selected, starting nearby an artificial water point. In commercial farms the gradient fitted in one single camp. The gradients included 6 single plots, each 10 x 10 m in size (schematic representation of these plots in Figure 3.2). These plots were defined exclusively through grass quality conditions, using a similar technique as Van der Westhuizen *et al*. (2005), independent of bare patches or bush encroachment. Indicator grass species defined on-site by plant experts for the purpose of rangeland condition assessment were identified. Rangeland condition ranged from poor, moderate (poor and moderate = piosphere) to good conditions (good = vegetation plot outside the piosphere).

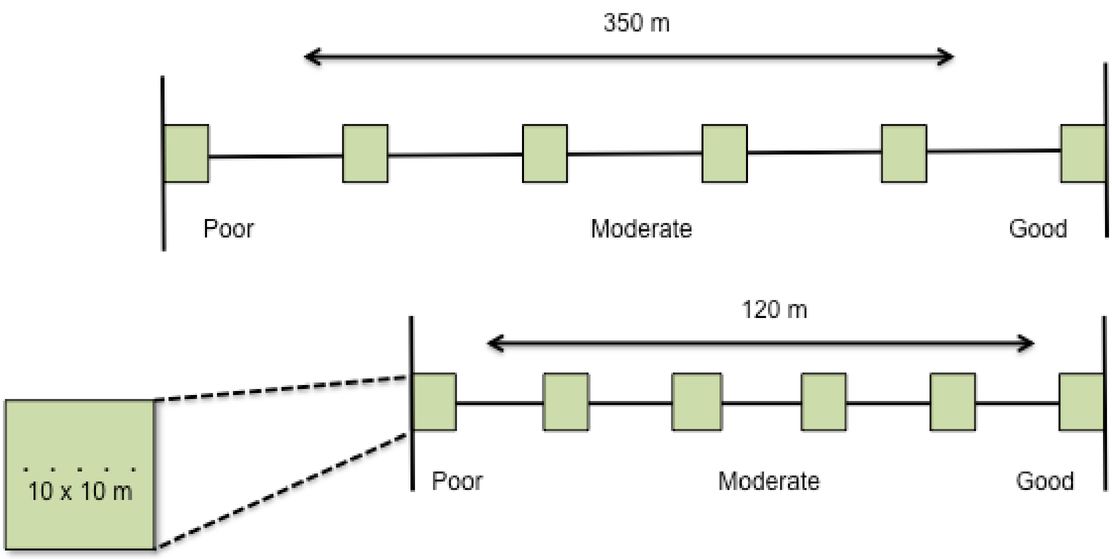


Figure 3.2 Schematic representation indicating the six 10 x 10 m sampling plots, in two different lengths of degradation gradients (keeping the distance between the six sampling plots constant).

In both ecosystems, the technique by Van der Westhuizen *et al.* (2005) was used to identify indicator grass species for the purpose of rangeland condition assessment. In the grassland ecosystem, the undisturbed open grassland ("good" condition) was dominated by the climax grass *Themeda triandra* (Redgrass), and only a few other species occurred. With increasing grazing pressure, *Eragrostis spp.* (Weeping lovegrass) replaced *T. triandra* and became therefore dominant in the subclimax, "moderate" condition stage. In the sacrifice area, under "poor" conditions, pioneer annual grass species such as *Aristida congesta* (Tassel bristlegrass) and *Cynodon dactylon* (Couchgrass) increased in proportion to *Eragrostis spp*. Bare patches were common in poor rangeland conditions, although they also occurred under moderate and good rangeland conditions. In the savanna ecosystem the dominant indicator grass species is *Stipagrostis spp*. (Bushmann grass) for good rangeland condition, *Eragrostis spp.* (Curley leave, Lehmann's love grass) for moderate rangeland condition and

Aristida spp. (Tassel three-awn) and *Schmidtia kalahariensis* (Kalahari sour grass) for poor rangeland condition (Figure 3.3). *Acacia* species are also dominant in this ecosystem with the area being affected by bush encroachment.

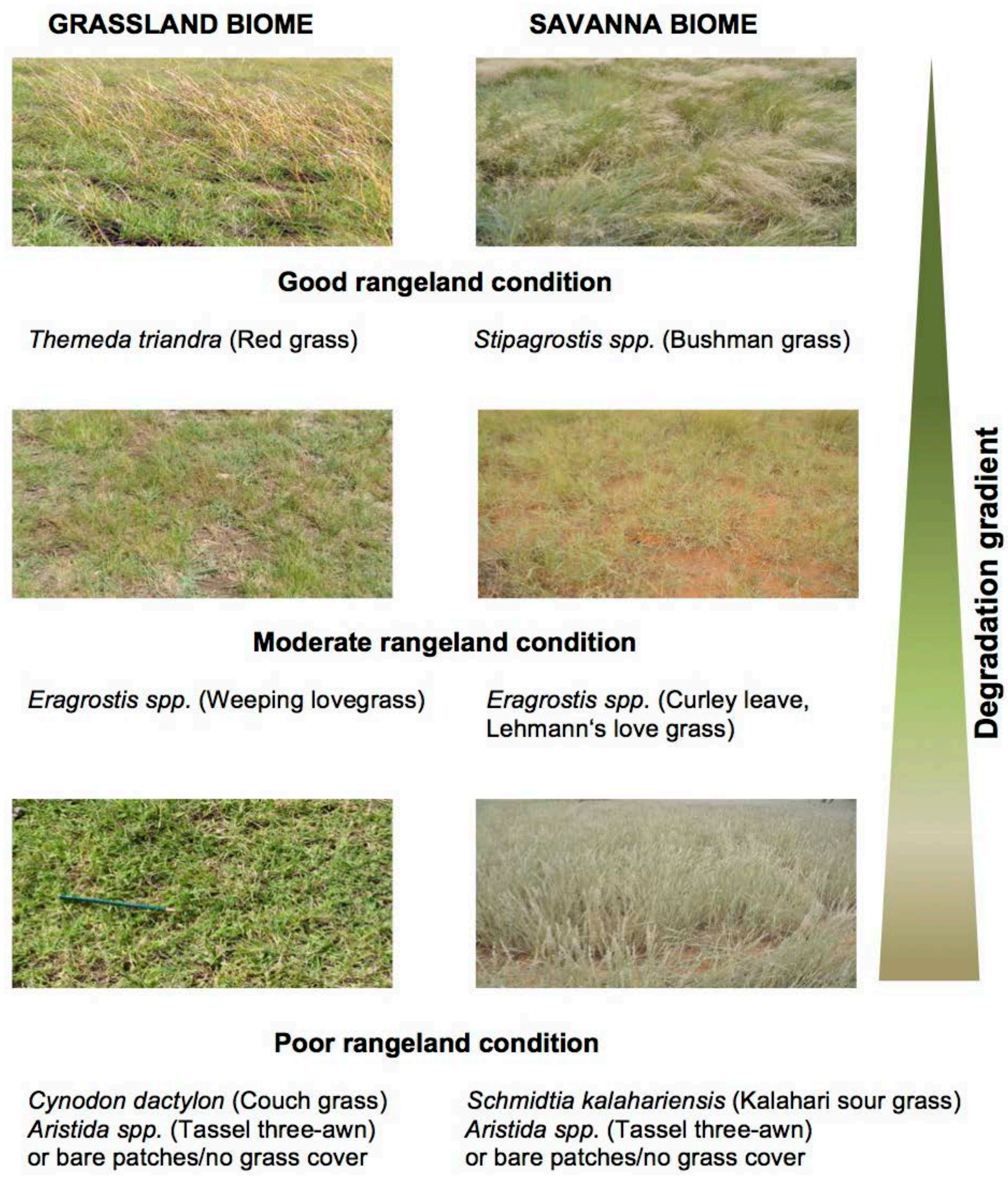


Figure 3.3 Pictures comparing poor, moderate and good rangeland conditions in the grassland (left hand side) and savanna ecosystems (right hand side), using indicator grass species for the purpose of rangeland condition assessment.

The length of the degradation gradients differed between and within the different management systems, but distances between the 6 plots were kept constant within each sampling site (Figure 3.1). Not all three rangeland conditions were necessarily present in the degradation gradient, therefore a variation in replicates exist. Along a centreline, a soil sample was taken every 2 m using a 50 mm diameter self-made hand auger (see Lobe *et al*., 2001 for detailed description) and combined into a composite sample for three depth intervals (0-5, 5-10 and 10-20 cm). The composite samples were air-dried, sieved (<2 mm) and prepared for chemical and physical analyses. Samples for bulk density were taken under all rangeland conditions of all management types assuming that bulk density of the sacrifice zone is similar to that under poor rangeland conditions. In the savanna ecosystem, additional composite samples were taken in direct proximity of the water points, where no vegetation grew (= sacrifice zone), as well as additional soil samples (0-10 cm) outside the gradients under *Acacia mellifera* and grass, to get an understanding of the influence of specific overlying vegetation on soil parameters. Plant experts estimated the percentage of bare ground, shrub, litter, moribund (dying vegetation) and senescent (old vegetation) in the plots of the degradation gradients of commercial (3 replicates) and communal (2 replicates) farms. Soil samples for microbiological parameters were taken only in the 0-5 cm soil layer. Because of the sensitivity of microbial soil samples, samples were carefully sieved (<2 mm) in the field, and stored in a portable freezer (approximately -30°C), before being transported to the laboratory and prepared for the various analyses.

3.4 Soil analyses

Unless otherwise specified, soil physical, chemical and microbiological analyses were done on all samples taken from the rangeland management systems, as well as all grazing conditions within both ecosystems. To avoid repetition full descriptions of all analyses will be given in the following relevant chapters, and will only be briefly mentioned in the next section of this chapter.

3.4.1 Soil physical analyses

Particle size analyses were done by the standard sieve-pipette method (The Non-Affiliated Soil Analysis Work Committee, 1990). Bulk density was determined by gravimetric analysis using 0.1 dm^3 steel cylinder samplers. Due to the higher clay content and associated aggregation in the grassland ecosystem, soil aggregation was only measured here, and not in die sandy savanna ecosystem (described in Section 4.2.4.). Particulate organic matter (POM) was only determined in the savanna ecosystem (described in Section 5.2.4).

3.4.2 Soil chemical analyses

All soil chemical analyses were performed in duplicate, and done according to the following standard methods (The Non-Affiliated Soil Analysis Work Committee, 1990): pH (1:2.5 soil to water suspension), exchangeable Ca, Mg, K and Na (1 mol dm^{-3} NH_4OAc at pH 7), extractable Cu, Fe, Mn and Zn (DTPA solution), and CEC (1 mol dm^{-3} NH_4OAc at pH 7). All of the mentioned elements were determined by atomic absorption. Total C and N were determined by dry combustion using a CHNS analyser (Elementar-Analysensysteme GmbH, Hanau, Germany). There was no detectable inorganic C; therefore total C was equal to organic C (described in Section 4.2.4).

Plant-available P was measured for all the grassland and savanna ecosystem samples using the Olsen method (The Non-Affiliated Soil Analysis Work Committee, 1990) and for samples of the 0-5 cm soil layer, sequential extraction of P fractions was only done for the savanna samples according to the scheme of Hedley *et al.* (1982) and Tiessen and Moir (1993), but slightly modified in that the soil volume for analyses was doubled (described in Section 5.2.4).

Due to the absence of C3 woody vegetation in the grassland ecosystem, $\delta^{13}C$ isotopes were measured in the savanna ecosystem, which contains both C3 woody and C4 grassland vegetation. Bulk soil (0-5 cm) and soil fractions were analysed for $\delta^{13}C$, using an Isotope

Ratio Mass Spectroscope (Delta V Advantage IRMS, Thermo Electron Corporation, Germany) (described in Section 5.2.4).

3.4.3 Soil microbiological analyses

Selected soil biological parameters were measured only in the 0-5 cm soil layers, and sampling took place in spring, after the first rainfall occurred. As mentioned before, due to the sensitivity of microbial activity in soil samples, special care was taken with the sampling procedure. Analyses dealt with below are described in Section 6.2.4.3

Enzymes in soil are considered to be representative of the functional component of soil microorganisms and are important for soil quality evaluation. Specific enzymes (β-glucosidase, urease, acid- and alkaline-phosphatase, dehydrogenase) were chosen because of their importance for nutrient cycling in ecosystems, and were determined colorimetrically using enzyme-specific procedures.

In addition to the enzyme analyses, phospholipid fatty acid (PLFA) analysis provides an insight into the soil microbial community profile. PLFAs are sensitive to changes in land-use overall, and the bacterial and fungal ratios also change with land-use and plant cover type (Hossain and Sugiyama, 2011).

3.5 Statistical Analyses

Statistical analyses were performed on all measured soil properties using analyses of variance in order to determine statistically significant differences between the main experimental factors: rangeland management systems, rangeland conditions and soil depth. For comparison of means we used a post-hoc Tukey-HSD test with a $p < 0.05$ level of significance. To explore variation in soil microbial community composition among study sites, the mole percentages (nmol%) of twelve individual PLFAs were subjected to principal component analysis (PCA) after standardizing to unit variance. All data were tested for normal distribution as well as homogeneity before statistical analyses were performed using

Statistica 9.1 for Windows (StatSoft Inc., 2010), Genstat 16 (NAG Ltd. Oxford, United Kingdom) and SPSS version 12.0 (SPSS Inc. Chicago, USA).

CHAPTER 4

RANGELAND MANAGEMENT IMPACTS ON THE PROPERTIES OF CLAYEY SOILS ALONG GRAZING GRADIENTS IN THE SEMI-ARID GRASSLAND BIOME OF SOUTH AFRICA

Abstract

The grassland biome of South Africa is a major resource for livestock farming; yet the soils of these rangelands are stressed differently by various management systems. The aim of this study was to investigate how basic soil properties respond to different management systems. For this purpose we sampled rangeland management systems under communal (continuous grazing), commercial (rotational grazing) and land reform (mixture of grazing systems) farming. Within each of these systems a grazing gradient was identified with decreasing grazing pressure with increasing distance to the water points. Results showed that communal farms with continuous grazing were generally depleted in the respective nutrient stocks. The depletion increased with rising grazing pressure. Along that line there was a breakdown of macroaggregates with losses of the C and N stored therein. However, the commercial farms also exhibited a decline of macroaggregates and their associated C content nearby the water points. Aggregate fractionation is a sensitive indicator for detecting the beginning of soil degradation in this ecosystem; yet, degradation was less pronounced under the rotational grazing of the commercial farms than under communal property right conditions. Hence, soil analyses confirm that fences and appropriate grazing periods are needed to manage these rangelands sustainably.

4.1 Introduction

Rangelands cover half of the world´s land surface. In the arid to semi-arid environments of South Africa more than 75% of the land is used for livestock production (Smet and Ward, 2006; Tainton, 1999). These rangelands are often subjected to degradation (WOCAT, 2009), mainly driven by poor land management (FAO, 2009).

Two main rangeland management systems are found in southern Africa, apart from game ranching (Smet and Ward, 2006; Tefera *et al*., 2010): commercial livestock ranching and communal livestock ranching. These systems differ mainly in the management of grazing resources, ownership as well as their outputs. The commercial farming sector is well developed, capital-intensive and largely export orientated. Therefore, commercial areas are divided into fenced farms, owned by individuals, and then further subdivided into a number of camps. Rotational grazing is usually practiced. Stocking rates tend to be more conservative and are adjusted by the farmer to ensure sustainable production.

The communal areas are situated mainly in the former homelands in the northern and eastern parts of South Africa, e.g. in Bophutatswana (Suttie *et al*., 2005). They differ distinctly from the commercial areas in their production systems, objectives and property rights. The communal production systems are based on pastoralism and grazing areas are shared by members of a community. There are often unclear boundaries, with open access rights to grazing areas. The outputs and objectives of livestock ownership are diverse, and include draught power, milk, meat, dung, cash income and capital storage, as well as socio-cultural factors. Higher stocking rates in communal areas are common (Suttie *et al*., 2005). This communal system is under criticism in terms of exceeding the carrying capacity of the land and risking rangeland degradation (Smet and Ward, 2006).

In addition, a third rangeland management system can be found in South Africa on farms that were allocated within the scope of the post-Apartheid land reform, (e.g., in Malawi, Zimbabwe) (Adams and Howell, 2001; Walker, 2002). These farms have been obtained by political redistribution programs by either an individual or a group of people. Stocking rates are normally lower than on communal farms and no clear management systems are present. There are weak forms of rotational grazing as well as continuous grazing systems.

The impact of grazing animals on ecological parameters in rangelands is spatially patterned. Animal activities are concentrated around the artificial or natural water points. According to Andrew (1988) piospheres develop around these water points, with a so-called sacrifice area close to the water point (Thrash and Derby, 1999). The grazing intensity and associated rangeland degradation generally increase with increasing proximity to the water points (Du Preez and Snyman, 1993; Lin *et al*., 2010). Many studies have used grazing gradients to investigate the effects on vegetation (e.g. Thrash 2000; Riginos and Hoffman, 2003) and relate degradation with plant composition change or biomass production. But generally, vegetation is highly influenced by variations in rainfall (e.g. Illius and O´Connor, 1999) and changes in plant composition or ground cover are usually, not always, reversible (Abel, 1997). Recording vegetation therefore only gives a snapshot of the current situation, where soil indicators are more reliable for determining a long-term situation.

Soil organic matter (SOM) is specifically a main factor in rangeland ecosystem functioning, as it improves soil structure, enhances water infiltration and thus prevents erosion through aggregate stabilization. Only a few studies compared the effects of continuous and rotational livestock ranching on SOM properties (Southorn, 2002; Smet and Ward, 2006; Sanjari *et al*., 2008; Medina-Roldán, 2012), and the results of these studies were inconsistent. Some show a loss (Neff *et al*., 2005; Snyman and Du Preez, 2005; Han *et al*., 2008), some no change (Mathews *et al*., 1996; Barger *et al*., 2004), others even a gain (Reeder and Schuman, 2002; Conant *et al*., 2005) in SOM under different grazing intensities. In other land-use systems, losses of soil organic C (SOC) are frequently related to a decay of soil aggregates, which then makes the stored C accessible for microbial degradation (Blanko-Canqui and Lal, 2004; Lobe *et al*., 2011). Particularly large macro-aggregates are vulnerable to breakdown, and served, therefore, as an indicator for the resilience of a soil against degradation (e.g., Bossuyt *et al*., 2002; Six *et al.*, 2004; An *et al.*, 2009). It could thus also serve as an indicator of rangeland resilience against intensified grazing, especially where low rainfall and high evaporation is present.

The objective of this study was to evaluate the impact of different rangeland management systems on soil degradation. We hypothesized that soil properties like SOM content and aggregate stability decline along gradients of increasing grazing intensity, and that this effect will be pronounced most in communal and least in commercial farms. To test these hypotheses, we selected grazing gradients in the rangeland management systems that are practised in the grassland biome, on clayey soils in the Thaba Nchu area, near Bloemfontein, South Africa, Here due to the resettlement programs in former Bophutatswana communal and land reform farms exist adjacent to commercial farms. The Maria Moroka nature reserve nearby was stocked with game, and served as control area ungrazed by domestic stock.

4.2 Materials and methods

4.2.1 Study area

The study sites were located on clayey soils (See Section 4.3.3 for details) with no obvious signs of wind or water erosion, in the Thaba Nchu area (between 28.95 "S, 26.46" E 29.41 "S, 27.00" E), of the Free State Province, South Africa. They belong to the "Moist Cool Highveld Grassland Type", which is part of the Grassland Biome (Bredenkamp *et al*., 1996). This grassland type was widespread and covered the central eastern part of the Highveld in the Free State with an altitude of 1400 to 1600 m. Non-perennial streams existed in the area. The mean annual precipitation was 553 mm (±190 mm), with about 70% of the rain occurring in summer between January and March. The climate was relatively dry, has low, variable and unpredictable rainfall and a high evaporation rate of 1832 mm a^{-1} (Basson, 1997). A single layer of perennial C_4 bunchgrasses dominated this particular grassland type. The amount of cover depended beside the low rainfall and evapotranspiration on the degree of grazing. Frost, fire and grazing maintained the grass dominance and prevented establishment of trees.

4.2.2 Rangeland management

Two main rangeland management systems dominate the Thaba Nchu area: communal and commercial farming. In addition, land reform farms are a third, less important management system for livestock production.

The communal farms of Thaba Nchu underlay several changes of the local land use system, from its very beginning in 1833 to the current state of low agricultural production. Basic shifts were in the 1940s during the betterment schemes, the 1970s and post-apartheid (for more detail see Naumann, 2014). From 1977 to 1994 subsidization of agriculture in Bophuthatswana, the destined homeland of the Batswana people, resulted in rising stocking rates and overgrazing became a problem in the communal areas. On the other hand the infrastructure of the communal grazing lands, such as boreholes and fences, were maintained and local rangers encouraged rotational grazing management. With the end of apartheid in 1994, Bophuthatswana was reincorporated into South Africa. The work of the local rangers, who cared for infrastructure, was discontinued. This resulted in a complete deterioration of the grazing land. Fences and boreholes are subsequently broken or non-existant. The communal rangeland is currently a common pool resource with no restrictions in stocking rates and no rotational grazing system. Livestock seem to be of little economic importance, either as income source for people in the Thaba Nchu area (most income is derived from social grants) or as contributing significantly towards nutrition. A decoupling of the social and ecological system seems to take place (Kuhn, personal communication).

The commercial farms surrounding the former homeland were surveyed and allocated to individual owners during the nineteenth century. Size and animal type (sheep, cattle) varied with time, but management structure remained more or less constant. Rotational grazing in fenced camps is common. The main purpose of commercial farmers is to produce high quality animals that can be marketed.

In the context of the post-apartheid land reform programme, farm units situated in the Thaba Nchu area were allocated to Trusts, Close Cooperations (CC) or Communal Property Associations (CPA). These farms belong to various numbers of members, who do not necessarily have any experience with pastoralism. The grazing management system is mixed but comparable to communal farms. Domestic stock is not the main income source here.

4.2.3 Sampling sites

Three rangeland management systems were selected for this study (Table 4.1): (a) commercial farms – CF, (b) communal farms – CO, and (c) land reform farms – LF. We sampled four replicates (hence 4 different farms) in every management system, treating the different farms thus as independent replicates. In additional, a nature reserve – NR, served as a control area, ungrazed by domestic stock. In the latter pseudo-replication was unavoidable. The management systems differ mainly in ownership, managing of grazing resources and stocking rate. Definition and selection of the land reform farms in the Thaba Nchu area proved to be challenging. After sampling it was realized that two of the farms are not strictly land reform farms as defined politically, because they were bought and not obtained via the resettlement program. On the other hand, the management of these farms clearly resembled that of a typical land reform farm, i.e, it was neither clearly continuous nor rotational grazed, depending on the individual purposes of the owner. We thus equated these farms as land reform farms.

Table 4.1 Description of the different rangeland management systems sampled in the Thaba Nchu area

Farm	Management system (number of farms)	Ownership	Stocking rate (ha LSU^{-1})	Rangeland conditions (no of samples)			Lenght of piosphere Std.dev.	Soil type	Clay (%)
				Poor	Moderate	Good			
Commercial (CF)	Rotational grazing (4)	One owner	6.4	4	4	4	60 (±21)	Lixisol	35
Communal (CO)	Continuous grazing (4)	Several households	4.3	4	4	3	73 (±77)	Lixisol	35
Land reform (LF)	Combination: continuous / rotational (4)	1-20 members	9.5	4	4	3	62 (±67)	Lixisol	28
Nature reserve (NR)	Fenced off grazing (1)	Government owned	10.4*	2	3	1	60 (-)	Lixisol	34

* (data from E. Schultze: personal communication)

At each sampling site grazing gradients were selected, starting at the artificial water points. The length of the grazing gradients (piosphere) differed between and within the different management systems. Therefore, according to Van der Westhuizen *et al*. (2005), rangeland condition ranging from poor – nearby the water points (= sacrifice zone), to moderate and to good conditions (Figure 4.1) serve as parameter for statistical analysis (see also discussion). Here, good rangeland condition lies outside the piosphere. Indicator species are used to define the rangeland condition. The undisturbed open grassland ("good" condition), was dominated entirely by the climax grass Redgrass (*Themeda triandra),* and only a few other species occurred (Figure 4.1). With increasing grazing pressure, Weeping Lovegrass (*Eragrostis spec.)* replaced Redgrass and became therefore dominant in the subclimax, "moderate" condition stage. In the sacrifice area, under "poor" conditions, pioneer annual grass species such as Tassel Bristlegrass (*Aristida congesta*) and Couchgrass (*Cynodon dactylon)* increased in proportion to Weeping Lovegrass. Not all three rangeland conditions were necessarily present in the grazing gradient, therefore a variation in replicates was given (Table 4.1). Figure 4.1 shows a spectrum of the visible different rangeland conditions. Bare patches were common in poor rangeland conditions, although they also occurred under moderate and good rangeland conditions.

Good rangeland condition

Dominant plant species:
Themeda triandra (Red grass)
Soil:
pH 6.2
C(%): 2.1
N (%): 0.17

Moderate rangeland condition

Dominant plant species:
Eragrostis curvula (Weeping love grass)
Soil :
pH: 6.3
C (%): 2.1
N (%): 0.16

Poor rangeland condition

Dominant plant species:
e.g. Cynodon dactylon (Couch grass)
Aristida congesta (Tassel three-awn)
or none
Soil:
pH : 6.7
C (%): 2.12
N (%): 0.18

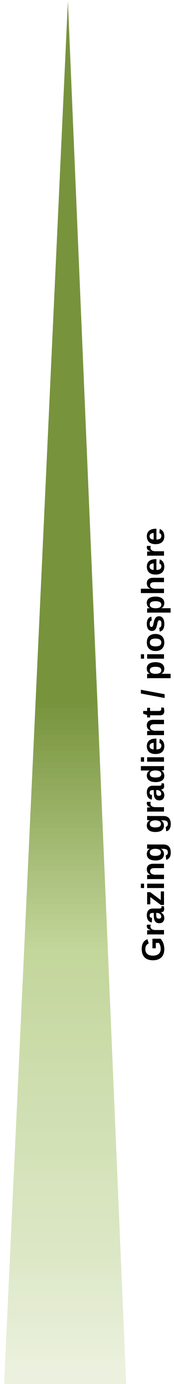


Figure 4.1 Different rangeland conditions in the Thaba Nchu area (two examples per condition: poor, moderate, good); the main plant species and means of main soil properties [0-5 cm] (all management systems).

4.2.4 Soil sampling and analyses

The soils of the study sites were all classified as Lixisols (WRB, 2007), and they had a slight clay difference in the 0-20 cm layer (Table 4.1). Composite soil samples were collected in March 2010 at three soil depths (0-5, 5-10, 10-20 cm) along the grazing gradient of rangeland conditions (poor, moderate, good), using a 50 mm diameter self-made hand auger (see Lobe *et al*., 2001 for detailed description). Each composite sample comprised of 10 randomly taken subsamples that were thoroughly mixed. The composite samples were air-dried, sieved (<2 mm) and prepared for analyses. All chemical analyses were performed in duplicate, and done according to the following standard methods (The Non-Affiliated Soil Analysis Work Committee, 1990): pH (1:2.5 soil to water suspension), exchangeable Ca, Mg, K and Na (1 mol dm^{-3} NH_4OAc at pH 7), extractable P (1 mol dm^{-3} $NaHCO_3$ at pH 8.5) and extractable Mn, Fe, Cu and Zn (DTPA method). Total C and N were determined by dry combustion using a CHNS analyzer (Elementar-Analysensysteme GmbH, Hanau, Germany). Particle size analyses were done by the standard sieve-pipette method (The Non-Affiliated Soil Analysis Work Committee, 1990). Bulk density was determined by gravimetric analysis using 0.1 dm^3 steel cylinder samplers.

For sampling of aggregates from 0-10 cm soil depth, larger aggregated blocks of defined volume (>1 dm^3) had been prepared in the field along the grazing gradient of the different rangeland management systems. These blocks were air-dried and stored until fractionation. For fractionation we used about 100 g of the aggregated blocks, sieved them to 8 mm and rewetted them (for more details, see Lobe *et al*., 2011). The re-wetted sample was then transferred to a 2800 µm sieve placed on a sieve stack and the sample was wet-sieved to obtain six aggregate-size fractions, which are referred to as: >2800 µm (peds), 2800-2000 µm (large macroaggregates), 2000-250 µm (small macroaggregates), 250-53 µm (large microaggregates), 53-20 µm (small microaggregates), and <20 µm (silt-sieved aggregates). Insignificant amounts of the 2000-2800 µm aggregate size fraction (on average 2.5 g 100 g^{-1}) forced us to combine this fraction with the aggregates >2800 µm and renamed this to "large macroaggregates". Each of the size fractions was corrected for sand content, by subtracting the total sand content of each size fraction from the amount of sample retrieved on each size fraction, and the C and N contents within these fractions were then determined as described earlier for bulk soil.

4.2.5 Statistical analyses

Statistical analyses were performed on all measured soil properties using analyses of variance in order to determine statistically significant differences between the main experimental factors: rangeland management systems, rangeland conditions and soil depths. For comparison of means we used the least significant difference (LSD) method with a $p < 0.05$ level of significance. All analyses were conducted using the Statistica 9.1 package for Windows (StatSoft Inc., 2010).

4.3 Results

Statistical analyses revealed that the length of the piosphere as well as the distance of sampling points to water resource had no effect on the measured parameters, because both the length of the piosphere and the distance to the water points did not vary systematically within a given management system and in-between them (Table 4.1). On the other hand, differentiating between poor, moderate and good rangeland condition by means of indicator plant species revealed differences in soil properties as related to the management system.

In the Lixisols investigated, generally bulk density, pH, as well as the contents of Ca, Mg, Na and Cu increased with depth, while those of C, N, P, K, Fe, Mn and Zn decreased correspondingly within the top 20 cm soil (Table 4.2). These soil properties only slightly reflected an influence of the rangeland management systems and grazing intensities ($p > 0.05$). Nevertheless, some consistent differences remained for the bulk densities and selected plant nutrients, whereas soil C and N contents as well as aggregation were significantly different between sites as outlined below.

Table 4.2 Bulk density, pH, plant nutrient concentrations and standard deviation of the rangeland management systems (commercial farms - CF, communal farms - CO, land reform farms - LF and the nature reserve - NR) in different rangeland conditions (p - poor; m - moderate; g - good) for three soil depths (SE in brackets).

Rangeland Management System	Rangeland Condition	Depth (cm)	Bulk Density [g cm-^{3}]	pH [H_2O]	C [g kg^{-1}]	N [g kg^{-1}]	P [mg kg^{-1}]	Ca [mg kg^{-1}]	Mg ([mg kg^{-1}]	K [mg kg^{-1}]	Na ([mg kg^{-1}]	Cu [mg kg^{-1}]	Fe [mg kg^{-1}]	Mn [mg kg^{-1}]	Zn [mg kg^{-1}]
CF	**p**	0-5	1.36 ±0.05	6.6 ±0.5	23.4 ±3.4	1.93 ±0.27	16.0 ±7.9	2362 ±962	529 ±70	456 ±82	43 ±11	1.6 ±0.2	76.8 ±25.8	18.6 ±8.2	2.0 ±0.6
		5-10	1.45 ±0.05	7.1 ±0.5	14.3 ±1.0	1.12 ±0.13	7.3 ±4.1	2718 ±1141	650 ±139	315 ±69	113 ±46	1.6 ±0.2	35.2 ±19.2	17.5 ±8.6	0.5 ±0.1
		10-20	1.47 ±0.12	7.7 ±0.5	11.2 ±1.7	0.85 ±0.16	4.0 ±1.7	3567 ±1249	930 ±190	285 ±132	265 ±73	1.5 ±0.3	10.3 ±6.1	12.1 ±4.8	0.2 ±0.0
	m	0-5	1.25 ±0.05	6.4 ±0.3	20.3 ±1.8	1.58 ±0.10	4.2 ±0.7	1516 ±378	579 ±78	337 ±63	48 ±13	1.8 ±0.2	50.1 ±12.3	27.6 ±3.2	1.2 ±0.1
		5-10	1.47 ±0.08	6.6 ±0.3	13.6 ±1.2	1.08 ±0.08	2.9 ±0.3	2111 ±674	710 ±121	224 ±70	113 ±35	1.9 ±0.2	17.3 ±5.1	18.0 ±3.3	0.3 ±0.0
		10-20	1.42 ±0.09	7.4 ±0.5	11.3 ±1.0	0.90 ±0.10	2.3 ±0.3	3166 ±898	1015 ±195	216 ±87	279 ±78	1.8 ±0.2	7.5 ±2.9	12.1 ±1.8	0.3 ±0.0
	g	0-5	1.38 ±0.03	6.4 ±0.2	20.7 ±1.6	1.56 ±0.14	10.5 ±6.9	1463 ±278	519 ±18	305 ±50	37 ±7	1.9 ±0.2	51.5 ±11.9	27.4 ±4.6	1.1 ±0.2
		5-10	1.41 ±0.08	6.4 ±0.1	14.5 ±0.7	1.10 ±0.07	2.3 ±0.3	1779 ±251	600 ±65	191 ±52	89 ±33	2.1 ±0.2	24.7 ±9.2	26.0 ±2.7	0.3 ±0.1
		10-20	1.49 ±0.09	7.2 ±0.2	11.3 ±0.8	0.90 ±0.06	2.6 ±0.8	2765 ±493	875 ±161	205 ±85	200 ±81	1.8 ±0.2	6.0 ±2.3	10.8 ±1.8	0.3 ±0.0
CO	**p**	0-5	1.46 ±0.06	7.1 ±0.6	11.5 ±1.2	0.92 ±0.09	9.9 ±2.7	1989 ±879	513 ±160	290 ±74	56 ±32	1.5 ±0.3	23.8 ±14.2	19.5 ±7.8	1.2 ±0.6
		5-10	1.53 ±0.09	7.1 ±0.7	11.2 ±0.3	0.94 ±0.05	7.0 ±1.9	2300 ±751	600 ±160	197 ±34	114 ±62	1.7 ±0.3	15.2 ±8.3	12.9 ±4.4	0.6 ±0.3
		10-20	1.52 ±0.08	7.4 ±0.7	10.6 ±0.3	0.88 ±0.03	5.8 ±1.3	2484 ±822	693 ±204	134 ±37	206 ±92	1.7 ±0.3	14.0 ±6.5	12.7 ±4.4	0.4 ±0.1
	m	0-5	1.42 ±0.02	6.4 ±0.3	15.3 ±1.4	1.21 ±0.13	5.6 ±0.2	1535 ±586	429 ±94	313 ±6	36 ±13	1.6 ±0.2	47.1 ±13.8	25.6 ±6.3	1.1 ±0.2
		5-10	1.44 ±0.08	6.6 ±0.4	11.6 ±1.1	0.93 ±0.07	3.7 ±0.2	1668 ±725	484 ±139	255 ±7	84 ±31	1.8 ±0.1	35.2 ±10.8	28.2 ±6.3	0.3 ±0.0
		10-20	1.47 ±0.07	6.9 ±0.5	9.9 ±1.4	0.80 ±0.06	3.6 ±0.5	2068 ±735	560 ±140	214 ±8	193 ±58	1.8 ±0.2	18.3 ±6.2	21.8 ±6.7	0.3 ±0.0
	g	0-5	1.36 ±0.02	6.1 ±0.1	16.9 ±1.1	1.26 ±0.15	5.0 ±0.2	1120 ±337	418 ±82	286 ±16	46 ±11	1.8 ±0.2	59.4 ±6.3	32.6 ±4.7	1.0 ±0.0
		5-10	1.40 ±0.05	6.3 ±0.2	12.8 ±0.4	0.96 ±0.12	3.7 ±0.4	1434 ±646	428 ±146	215 ±15	87 ±41	2.1 ±0.1	57.3 ±17.3	33.9 ±2.2	0.3 ±0.0
		10-20	1.45 ±0.06	6.6 ±0.4	10.8 ±0.7	0.88 ±0.06	3.5 ±0.2	1752 ±812	620 ±164	143 ±22	181 ±55	2.1 ±0.3	34.6 ±19.5	22.9 ±4.6	0.5 ±0.2

Table 4.2 (cont.)

Rangeland Management System	Rangeland Condition	Depth (cm)	Bulk Density [g cm^{-3}]	pH [H_2O]	C [g kg^{-1}]	N [g kg^{-1}]	P [mg kg^{-1}]	Ca [mg kg^{-1}]	Mg [mg kg^{-1}]	K [mg kg^{-1}]	Na [mg kg^{-1}]	Cu [mg kg^{-1}]	Fe [mg kg^{-1}]	Mn [mg kg^{-1}]	Zn [mg kg^{-1}]
LF	**p**	0-5	1.39	6.4	25.8	2.15	16.8	1661	509	428	20	1.6	16.1	10.9	0.4
			±0.08	±0.1	±5.3	±0.47	±6.9	±304	±72	±114	±3	±0.6	±5.6	±2.0	±0.1
		5-10	1.42	6.5	17.4	1.50	7.2	1897	539	281	31	2.2	26.7	17.1	1.5
			±0.07	±0.3	±3.9	±0.33	±1.1	±231	±131	±108	±5	±0.6	±15.2	±6.8	±0.7
		10-20	1.42	6.6	12.4	1.07	4.1	1865	544	151	38	2.2	35.6	20.3	1.0
			±0.12	±0.3	±1.9	±0.10	±0.7	±294	±150	±63	±3	±0.7	±21.8	±6.5	±0.4
	m	0-5	1.46	6.2	24.0	1.88	8.7	1466	506	358	27	2.4	58.5	26.5	2.4
			±0.31	±0.1	±4.4	±0.33	±2.8	±167	±121	±83	±5	±0.7	±30.8	±5.5	±1.8
		5-10	1.44	6.2	15.3	1.23	6.0	1560	544	186	45	2.6	44.4	22.1	1.0
			±0.17	±0.1	±2.0	±0.09	±0.7	±203	±154	±67	±13	±0.8	±22.5	±6.7	±0.3
		10-20	1.40	6.7	13.2	1.04	12.6	2109	880	121	100	2.5	33.2	20.7	0.6
			±0.08	±0.1	±1.6	±0.03	±6.9	±248	±308	±44	±38	±0.8	±13.2	±5.5	±0.1
	g	0-5	1.31	6.3	13.2	1.20	5.9	1118	379	242	18	2.3	23.7	19.1	0.7
			±0.2	±0.3	±0.3	±0.01	±1.6	±152	±83	±62	±2	±0.8	±2.1	±6.6	±0.1
		5-10	1.50	6.3	10.2	0.94	2.6	1318	400	131	25	1.5	41.1	11.8	8.3
			±0.15	±0.3	±1.6	±0.11	±0.1	±85	±65	±38	±3	±0.1	±19.5	±0.8	±6.6
		10-20	1.35	6.7	9.2	0.88	5.3	1931	520	82	49	2.6	31.4	11.9	0.8
			±0.16	±0.1	±0.5	±0.01	±2.2	±11	±65	±25	±13	±1.1	±14.7	±1.8	±0.5
NR	**p**	0-5	1.29	6.7	24.3	2.06	12.8	1953	909	446	24	1.1	4.0	10.5	1.8
			±0.12	±0.6	±1.11	±0.88	±3.8	±372	±130	±58	±8	±0.6	±0.7	±4.8	±1.2
		5-10	1.43	6.8	17.9	1.52	5.8	2393	1069	281	36	2.4	2.7	9.6	0.7
			±0.06	±0.5	±8.4	±0.61	±0.7	±477	±130	±7	±12	±0.75	±0.4	±5.3	±0.4
		10-20	1.54	7.3	15.4	1.16	6.5	3015	1219	162	40	2.3	4.7	10.4	0.4
			±0.05	±0.8	±7.4	±0.43	±0.9	±198	±200	±10	±5	±0.88	±3.6	±6.4	±0.2
	m	0-5	1.18	6.2	24.7	1.88	4.4	1350	599	332	16	2.0	44.7	20.4	1.5
			±0.17	±0.1	±5.9	±0.48	±1.1	±474	±168	±89	±5	±0.33	±32.2	±3.5	±0.5
		5-10	1.38	6.3	17.6	1.40	4.8	1378	519	209	21	2.5	41.1	16.0	0.4
			±0.13	±0.2	±2.7	±0.25	±1.4	±561	±172	±52	±5	±0.36	±30.1	±7.7	±0.3
		10-20	1.46	6.4	15.4	1.17	4.0	1614	606	142	65	1.9	22.2	22.2	0.4
			±0.08	±0.2	±1.9	±0.14	±0.9	±601	±144	±76	±46	±0.72	±20.7	±5.72	±0.1
	g	0-5	1.37	6.2	39.9	2.59	4.6	2650	919	474	20	2.5	0.5	28.2	2.3
		5-10	1.50	6.3	20.6	1.49	4.4	2146	819	168	16	2.7	0.5	17.3	0.7
		10-20	1.44	6.6	16.7	1.10	5.0	2673	959	82	20	2.3	1.1	11.0	0.4

4.3.1 Bulk density

In the very surface soil, the rangeland management systems showed lowest bulk densities in the nature reserve and highest values in the communal areas. Along the grazing gradient, higher bulk density values in communal farms occurred in the sacrifice zone of the piosphere (Table 4.2).

4.3.2 Plant nutrients

Inorganic soil nutrients as well as pH were enriched in the zone of highest animal activities (poor). This was more pronounced in the communal farms. For example, Ca and P contents under poor rangeland conditions were almost twice as high as those under good rangeland conditions. On the other hand extractable Mg, and Cu, and Na contents consistently showed the lowest concentrations under the poor rangeland condition. Differences between the rangeland management systems throughout the respective soil layers were small and insignificant (Table 4.2). But, in general, all exchangeable cations (Ca, Mg, K, Na) as well as P had the highest values in commercial and land reform farm management, along the whole grazing gradient. Lower nutrient contents were found not only in the piospheres of communal farms but also beyond them (good condition).

4.3.3 Carbon and nitrogen

The contents and stocks of C and N clearly differed significantly between the rangeland management systems ($p < 0.05$), with the lowest values being found in the communal gradients. In contrast, the C stocks in the commercial and land reform farms were elevated by a factor of 1.4 and 1.5 respectively, whereas in the nature reserve they even increased by a factor of 1.9 (Figure 4.2). The same pattern was repeated for the contents of total soil N, which correlated closely with those of soil C ($r = 0.97$; see Table 4.2 for details).

The depletion of C and N stocks in the communal farms was most pronounced under the poor and moderate rangeland condition of the top soil layer (Figure 4.3). Beyond the grazing gradient (good), the C stocks of the communal managed systems were only slightly lower than in the commercial farms. The differences along the grazing gradient in the latter were even smaller. The land reform farms showed an overall unclear distribution in response to grazing intensity, where less C was found outside the piosphere compared to moderate and poor grazing conditions. On the other hand, exceptionally high C stocks were measured under good rangeland conditions of the nature reserve, where the low number of samples (n=1; Table 4.1) probably influenced the value extraordinary. The differences between the management systems were also evident in the 5-10 cm and 10-20 cm soil layers (Table 4.2).

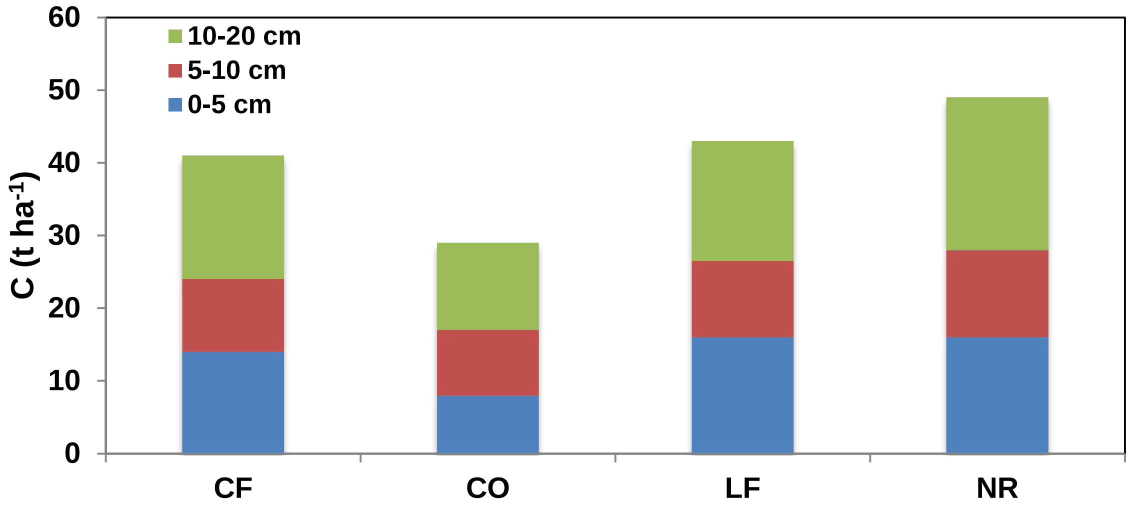


Figure 4.2 Carbon stocks [t ha^{-1}, 0-20 cm] of different management systems (commercial farms - CF, communal farms - CO, land reform farms - LF, nature reserve - NR).

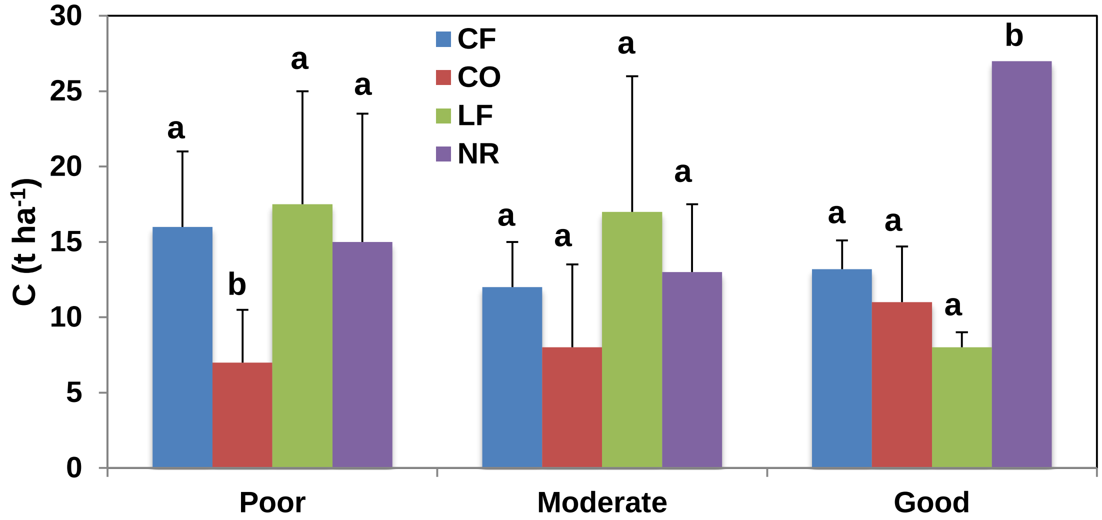


Figure 4.3 Carbon stocks [t ha^{-1}, 0-5 cm] of different rangeland conditions and management systems (commercial farms - CF, communal farms - CO, land reform farms – LF, nature reserve – NR). The standard error is shown in the error bars; statistical difference is indicated by letters.

4.3.4 Aggregate-size distribution, and associated carbon and nitrogen

The different rangeland management systems varied in the distribution of their aggregate size fractions. On average, after sand correction, large macroaggregates (>2000 µm) accounted for the greatest part (89.2%) of all sampled sites, which is typical for grassland soils. Commercial farms (90.7%), land reform farms (89.2%) and the nature reserve (91.9%) exhibited a similar amount of macroaggregates.

In contrast, the soils of the communal farms differed in structural stability. They were less aggregated and therefore accounted for lower proportions of macroaggregates (85.2%) and larger proportions of the small macroaggregates, large microaggregates, and small microaggregates. Additional, the number of large macroaggregates decreased with increasing grazing intensity in the order good > moderate > poor rangeland conditions (Figure 4.4). The opposite was therefore true for the number of small macroaggregates. The other rangeland management systems did not reveal any distinct trends. Nevertheless, we also found a decline of macroaggregates close to the water points of the commercial farms.

When reducing the effect of cattle management on soil structure to a single parameter, e.g., by calculating the mean weight diameter (MWD), the above-mentioned results were confirmed. The communal farms showed least aggregation in the poor (MWD=4.32) and a better aggregation beyond the gradient (MWD=4.57). The other farms exhibited a similar MWD at all rangeland conditions and, therefore, a higher MWD for the poor rangeland in the commercial than in the communal farms (CF: MWD=4.55, LF; MWD=4.53, NR: MWD=4.74).

Due to their high proportions, most of the soil C was also located in the macroaggregates (76-88%), followed by the small microaggregates (6-12%). The sand-corrected C concentration within the different aggregate-size fractions varied widely. In all land-use systems, C concentrations were highest in the large microaggregates (data not shown). The

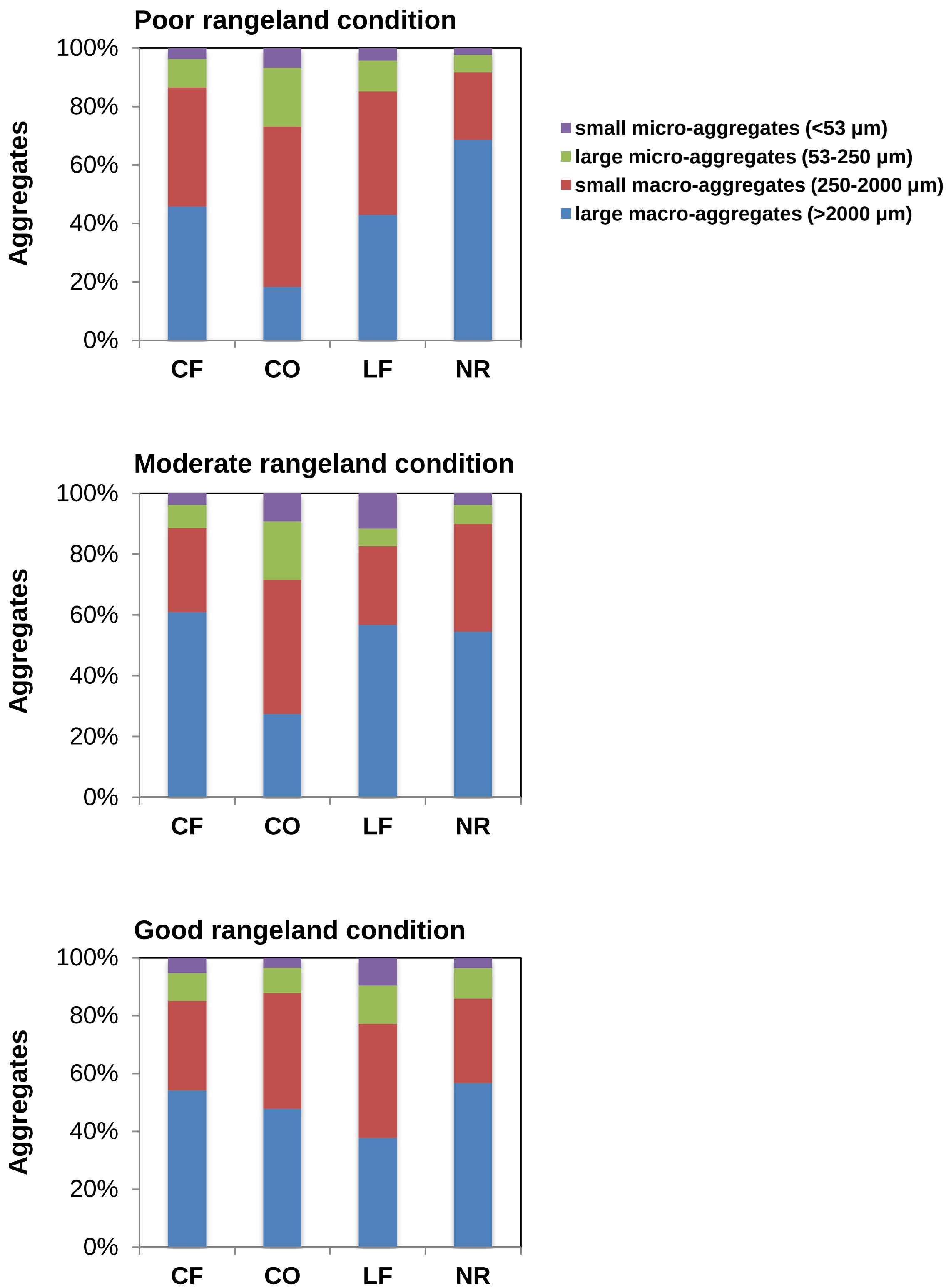


Figure 4.4 Aggregate distribution (0-10 cm) in a) poor, b) moderate, c) good rangeland conditions and different management systems (commercial farms - CF, communal farms - CO, land reform farms - LF, nature reserve - NR).

distribution of N across the aggregate fractions again followed the same trend as that of C. Comparing the management systems, the communal farms showed the lowest C contents in all defined aggregate sizes, especially being prominent close to the water points, where the C stocks in the macroaggregates (Figure 4.5) and the large microaggregates differed significantly from those in the commercial farms and the nature reserve. The loss of C under communal farms correlated with the loss of macroaggregates and the organic matter stored therein. Notably, we also detected a depletion of C in the macroaggregates of the commercial farms nearby the water points, which was not indicated by mere bulk soil C analyses. Hence, combining aggregate fractionation with element analyses sensitively reflected changes in soil properties as induced by different ownership and thus grazing systems.

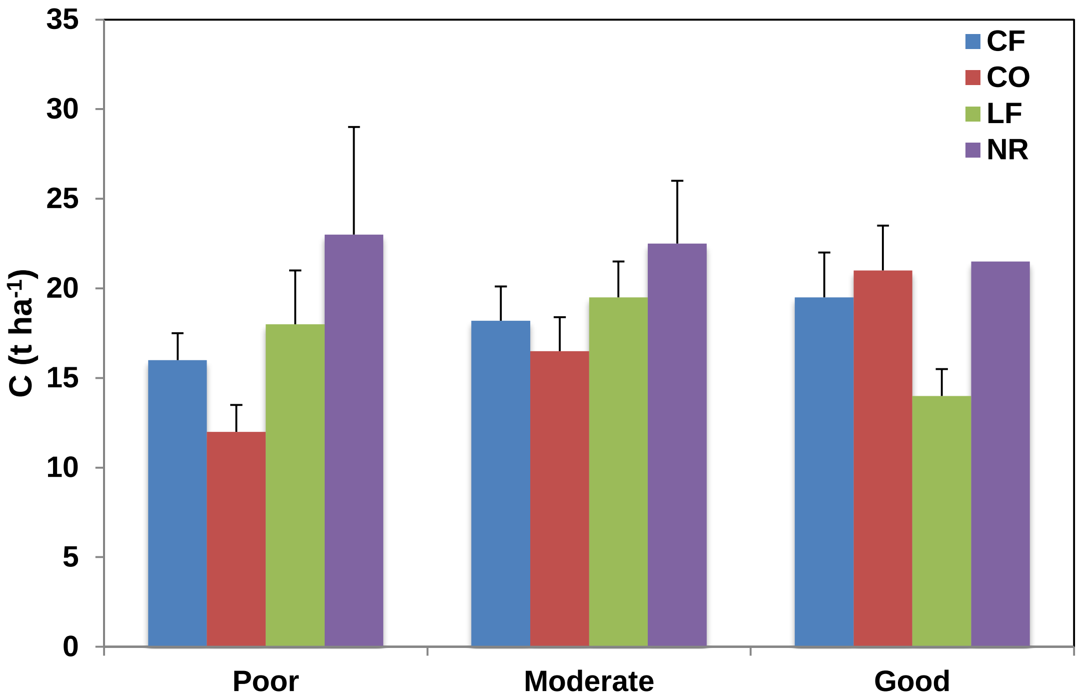


Figure 4.5 Carbon stocks [t ha^{-1}] in large macroaggregates of different rangeland conditions and management systems (commercial farms - CF, communal farms - CO, land reform farms – LF, nature reserve – NR).

4.4 Discussion

The results from this study showed distinct effects of grazing intensity on the main soil properties within the piosphere (poor and moderate rangeland condition) of clayey soils in the grassland ecosystem. Different rangeland management systems also influence the magnitude of the impact.

4.4.1 Bulk density

Animal trampling may have a severe effect on soil compaction and bulk densities (Willat and Pullar, 1984; Sanjari *et al.*, 2008; Medina-Roldán, 2012). The continuous grazed rangelands of the communal farms in Thaba Nchu area were most compacted, especially close to the water points. Similar effects have been observed for other grassland ecosystems (Daniel *et al*., 2002; Savadogo *et al*., 2007; Stavi *et al*., 2008, 2011), and also Steffens *et al*. (2008) monitored significantly higher bulk densities in a semi-arid steppe in heavily grazed plots compared with ungrazed or only winter-grazed plots. In our study, higher bulk densities of communal areas were restricted to the sacrifice area of the piosphere, where animal activities were the highest. The impact diminished beyond the gradient (good condition). This was also observed by Andrew (1988), where compaction was restricted to an area of 50 m around the water points. Intriguingly, differences in bulk densities were not so clear visible in the piospheres of the rotational grazing system in the commercial farms. Obviously, short periods of intensive grazing followed by rest periods of several months, protects the soil from hoof damage to a much higher degree than does continuous grazing.

4.4.2 Plant nutrients

Two contrasting pathways influence the nutrient contents in grazing gradients. (i) nutrient losses caused by high grazing pressure and its accompanied depletion of plant cover and litter input, as well as additional trampling and resulting mixing of surface soil with subsoil or erosion, and (ii) enrichment of nutrients in the inner parts of the piosphere due to urine and dung of the animals (Perkins and Thomas, 1993), as well as through a centripetal nutrient flow to the water holes at natural water points. In our study with exclusive artificial water points, inorganic soil nutrients as well as soil pH values were enriched in the sacrifice area of the piosphere (poor rangeland condition). This enrichment was more pronounced for the continuous grazing in the communal farms. In contrast, values for K were low along all rangeland conditions in the communal farms. This depletion of K was in line with findings of Tefera *et al*. (2010) and Snyman and Du Preez (2005), who stated that erosion or low nutrient input of plants are possible causes for the nutrient losses. According to Snyman (2005b), nutrient losses by erosion usually account for only a small proportion of total nutrient losses in the Thaba Nchu area (e.g., 5.1% of N) and are therefore possible but unlikely. Despite the enrichment of nutrients in the sacrifice zone of the piosphere, communal farms always have the lowest nutrient levels within the gradients and beyond of them when compared with the other rangeland management systems. We conclude that overgrazing in continuous grazed management systems is responsible for nutrient losses by the depletion of plant cover and litter input (see also Section 4.3.2).

A different process may affect the contents of soil P and Na; high concentrations at the commercial farms nearby the water points were likely due to supplementary feeding given to the cattle (personal communication by the farmers; see also: Smet and Ward, 2006). The land reform farms with no clear grazing system showed various results with a high standard deviation, which reflects the ambiguous type of management in these sampled farms.

4.4.3 Carbon and nitrogen

In contrast to the slight variations of inorganic nutrient stocks, the organic C and N contents changed significantly. The surface soil layer (0-5 cm) was most vulnerable to SOM losses. Many other studies also found smaller soil C and N contents in heavily grazed plots compared with light or moderate grazed ones (e.g., Xie and Wittig, 2004; Shourkaie *et al*., 2007; Han *et al*., 2008). Snyman and Du Preez (2005) observed in the South African grassland biome that degradation of the rangeland from good to poor condition resulted after

5 years in a decline of SOM, which equated to 25% of the soil C and to 16% of the soil N content. The main reason for rangeland degradation in the piosphere in this grassland ecosystem was a lower litter production at poor rangeland condition (Snyman, 2005a). This is different to overgrazing for very short periods, a practice known for holistic farming to refresh the production of grass in large camps. Under the heavy and/or continuous grazing, plant growth is being suppressed and more grass residues are removed by grazing. Different stocking rates and resting times of rangelands thus contribute to the SOM losses in the piosphere. It should be noted that the processes described here might only be valid for this type of a grassland ecosystem (sweet grasses dominating; bushes mostly absent due to cold partly temperatures).

The recommended stocking rate for the area of Thaba Nchu is 6 ha per livestock unit (LSU) (Department of Agriculture and Rural Development, 2003). This rate is usually exceeded on the communal farms, amounting to 4.3 ha LSU^{-1} (Table 4.1). There, overstocking accompanied the subsidization from 1977 to 1994, and the collapse of institutions after the end of apartheid severe impacts on rangeland around water points in the Thaba Nchu area. The situation is comparable to Hardin's concept of the "tragedy of the commons" (Hardin, 1968), stating that the incentive to have one more animal may be greater for the individual than the incentive to save common productivity by protecting the rangeland from overgrazing at reduced LSU. Yet, the situation is different to the situation described by Hardin (1968) in that only a part of the total LSU income is used for saving the livelihood of the community, and the open access rights to the land certainly strengthened this problem. Indeed, we observed a lack of vegetation cover and an increased appearance of bare patches close to the water points particularly on communal farms (Figure 4.1).

It has been estimated for two of the communal farms that 60 and 78% bare patches exist under poor rangeland conditions, whereas these areas in the commercial farms only occupied between 20 and 40% of the same rangeland condition (A. Linstädter, unpublished data). Probably, occurrence and permanence of bare patches are good indicators for soil degradation outside of the piospheres. Additionally, the plant composition also changed. The rangeland outside the piosphere was dominated by perennial climax species *Themeda triandra* (Redgrass) and had the highest basal cover. Perennial bunchgrasses of *Eragrostis* species dominate in moderate rangeland condition. The poor rangeland condition was dominated by perennial *Cynodon dactylon* and short-lived perennial bunchgrass *Aristida congesta,* which had the lowest basal cover. Therefore, the changes in plant composition have been accompanied by reduced below- and aboveground biomass production. The change in plant composition occurred in all management systems due to higher grazing intensities at the water points. Resting time in the rotational systems allowed the rangeland in the piosphere to regenerate. Even if mostly *Cynodon dactylon* occurred at the water points with lower biomass production than for example red grass, the input of organic matter and likely also the supplemental fodder improve soil conditions in commercial farms. Hence, the lack of severe SOM losses in the commercial farms is mainly related to the shift in plant-species composition, which was still able to more or less maintain SOM stocks relative to the communal land, where trampling was too severe to allow the continued growth of *Cynodon* plants.

An impact of stocking rates on vegetation and SOM was also obvious in the land reform farms, which had very diverse grazing systems. Continuous as well as simple rotational systems were common there. However, the numbers of animals of the members of these farms were low, and as a consequence the stocking rate was on average 9.6 ha LSU^{-1} (min 6 ha LSU^{-1}; max 13 ha LSU^{-1}). Therefore, even if these farms did not practice clear rotational grazing, the pressure on the soil was much lower than in the communal farms, allowing the soils to regenerate as well. As a result these farms exhibited intermediate SOM contents with high standard deviation, due to the diversity of grazing practices.

In general, the differences declined with decreasing grazing intensities as well as depth. Obviously, key changes and losses are restricted to the sacrifice area where animal activity is the highest. The stock values of C and N under good rangeland condition, sampled at the far end of the piosphere, indicated only insignificant differences for rotational as well as continuous grazing systems (Figure 4.3). These findings were confirmed by average C and N data of sampled grids (100 x 100 m), established in every management system also outside of the measured piosphere (data not shown here).

4.4.4 Aggregate-size distribution, and associated carbon and nitrogen

Aggregation is an integrative indicator for the soil's overall quality (Boix-Fayos *et al*., 2001). This property affects the physical and hydrological functioning of soil, is associated with its capability in sequestrating organic C, and it impacts primary production (Bird *et al*., 2007). Animal trampling generally resulted in a loss of the proportion of stable aggregates, as well as infiltration rate (Stavi *et al*., 2008), and our results confirm that this process is also valid for the different grazing systems in the order of the property rights they correlate with, i.e., it increases in the order commercial farming (rotational continuous grazing) < land reform farming (mostly continuous grazing, low stocking rates) < communal farming (continuous grazing, high stocking rates). As a result, also the SOM stored and likely physically protected within the aggregates was also lost.

The processes of aggregate disruption and associated SOM decomposition were again most evident for the poor rangeland conditions in the continuous grazing systems. With increasing distance from the water point, soil aggregate structure stabilized in all rangeland management systems and it was similar outside of the piospheres. The proportion of macroaggregates indicated a diminishing influence of the trampling animals' impact and the C contents stored therein (Figure 4.5). However, intriguingly also a loss of macroaggegrates and of their associated C occurred in the commercial farms under poor rangeland conditions, despite these sites being managed with a resting time (Figure 4.4). Apparently, losses in macroaggregate C may serve as an early warning indicator for future ecosystem development; and even at any given farming practice under rotational grazing, these farmers have to handle their camps with care.

4.5 Conclusions

Rangeland management in the semi-arid grassland biome of the Thaba Nchu area is affected by different property rights and these then had effects on the grazing system and therewith on local soil properties nearby the water points.

The intention of commercial farmers is firstly to be economically viable by selling animals, which results in a more foresighted management of their rangeland. A rotational grazing system in fenced camps gives soil and vegetation resting times for recovering, accompanied by adapted stocking rates. In the communal farms in Thaba Nchu poor institutional maintenance lead to a deterioration of the infrastructure (fences, water points) accompanied by open access rights to the rangeland. Continuous grazing is practiced and overstocking is common in the communal areas. Land reform farms all have better infrastructure than communal farms, however, there is less foresighted management of the rangeland, but low stocking rates avoid degradation.

As a consequence of management, soil degradation in the piosphere of continuous grazed rangeland of the grassland ecosystem is driven by two processes: (i) deterioration of aggregates and associated SOC losses in the poor and moderate rangeland condition, and (ii) nutrient losses caused by lower plant cover and litter input in the sacrifice area of the piosphere. Rotational grazed camps, in contrast, showed little evidence of soil degradation, but they exhibited an early deterioration of the aggregate structures nearby the water points. As a consequence climate-adapted stocking rate and duration of the resting time, will remain necessary in future, even for recommended management systems.

CHAPTER 5

RANGELAND MANAGEMENT EFFECTS ON SOIL PROPERTIES IN THE SAVANNA BIOME, SOUTH AFRICA: A CASE STUDY ALONG GRAZING GRADIENTS IN COMMUNAL AND COMMERCIAL FARMS

Abstract

Rangeland degradation in savanna ecosystems is often accompanied by bush encroachment, however, little is known about the influence of different rangeland management systems on soil properties in such biomes. For this purpose we sampled soils under communal (continuous) and commercial (rotational) grazing systems along a gradient of decreasing grazing pressure away from water points. The results revealed that communal systems with continuous grazing were depleted in most nutrients nearby the water points compared to those of commercial systems with rotational grazing. In contrast, with increasing distance from the water points nutrient stocks of these communal systems were higher than those of the commercial systems. These nutrient enrichments were accompanied by increasing proportions of bush cover (up to 25%). Specific analyses (phosphorus fractions, particulate organic carbon and isotopic composition) confirmed that soils of the communal grazing systems, apart from the water points, were strongly influenced by the overlying woody *Acacia* vegetation. Only near the water points, high grazing pressure had overridden the effects of the woody plants. Hence, and in contrast to previous reports from grasslands, rangeland degradation in communal farms of the savanna ecosystem improved soil quality due to bushes, but at the cost of palatable grass area.

5.1 Introduction

Rangeland degradation in savannas is often accompanied by bush encroachment, which has become a widespread phenomenon in arid and semi-arid environments (V undergone encroachment of woody plants (Eldridge *et al*., 2011). This includes large areas of rangelands in South Africa, especially in the arid sandy savanna biome of the Northern Cape and North-West Provinces (Jacobs, 2003; Wigley *et al*., 2010). The driving factors of bush encroachment are not completely understood, but beside background variables like long-term vegetation cycles (Wiegand *et al*., 2006), and climate change or increased rainfall (Van Auken, 2009), land-use practices like high stocking rates and overgrazing, as well as altered burning techniques, e.g. fire suppression, are local drivers for bush encroachment (Throop and Archer, 2008).

Two main rangeland management systems with different land-use practices exist in South Africa: commercial and communal livestock ranching. The privately owned commercial farms are generally considered as well adopted management systems. Rotational grazing in fenced camps and moderate stocking rates are common and the management system consists of alternating periods of use and rest, to promote vegetation growth. In communal rangeland, mostly located in former homelands, fences are broken or even lacking, and livestock is allowed to continuously and selectively graze without any control around water sources such as artificial water points. Stocking rates are also usually high. This management system is under criticism for its observed rangeland degradation (Smet and Ward, 2006), which in savannas goes along with encroachment of woody species and changes in plant composition (Todd, 2006; Kioko *et al*., 2012). Yet, little attention has been given to the impact of such vegetation changes under different rangeland management systems on nutrient dynamics and properties of the savanna soils.

Water points are the area of highest animal activities and, the impact of grazing on ecological parameters is spatially patterned with highest impacts nearby the water and decreasing influence of grazing with increasing distances from the water. Piospheres develop with a sacrifice zone nearby the water points (Lange, 1969) and with zones, defined as poor, moderate and good depending on grassy vegetation status (Van der Westhuizen *et al*., 2005). Studying grazing effects on vegetation and soil properties along transects with decreasing grazing intensity is a widely used approach (Washington-Allen *et al.*, 2004; Tefera *et al*., 2007) and results have revealed that in the vicinity of water points, deposition of animal excreta and/or feed supplements increased carbon (C), nitrogen (N) and phosphorus (P) contents markedly (Fernandez-Giminez and Allen-Diaz, 2001; Smet and Ward, 2006; Shariary *et al*., 2012). In finer textured sites changes in soil properties by grazing can be linked to a deterioration of soil structure by trampling, as e.g. reported for the grassland ecosystem in South Africa, where soils degraded at communal farms under high trampling pressure (Kotzé *et al*., 2013). Yet, changes could be different in the savanna ecosystem, where increased bush encroachment may partly replace the formation of bare patches.

Bushes promote nutrient inputs via wildlife by providing shelter to birds and animals, and protecting soil from organic matter degradation and erosion. Monitoring organic matter and major nutrients provide a first clue to detect any soil changes. Among the latter, particularly P may be of interest, because it is usually a limiting nutrient in tropical soils due to fixation processes (Zhao *et al* therefore a tool for making direct assessments of vegetation shifts of C4 grasses ($\delta^{13}C$ - 13‰) to C3 woody plants ($\delta^{13}C$ - 27‰) on the fate of SOC (Liao *et al*., 2006b), promising insights into patterns and processes of recent and past woodland development (Boutton *et al*., 1998; Bai *et al*., 2009; 2012).

In grassland biomes free of bushes, rangeland degradation usually results in a deterioration of soil properties (e.g. Kotzé *et al*., 2013). Degradation processes, however, are likely different when bushes invade. This study was therefore designed to test the hypothesis that rangeland degradation by bush encroachment in the savanna ecosystem also alters soil properties, but not necessarily negatively. In fact, nutrient reallocation and soil protection under the shrubs could improve soil fertility, thus offer chances for future soil management at unadjusted rangeland use. To test this hypothesis we evaluated the status of soil chemical and physical characteristics in relation to land-use systems along a gradient from water points in an arid sandy savanna ecosystem of South Africa. The analyses included physical (texture, bulk density) and chemical (organic matter, nutrients, pH) soil parameters with special interest on isotope composition in particulate organic matter and P fractions.

5.2 Materials and methods

5.2.1 Study area

The study sites were located near Kuruman at the border of the Northern Cape and North-West Province of South Africa, and were situated on the fringe of the Kalahari (Latitude 27° - 28° S, Longitude 22° - 24° E; with an altitude of 1047 m to 1161 m above sea level) (Figure 5.1). Mean annual rainfall and temperature of Kuruman varied between 266 mm and 478 mm, and 17.5°C and 17.7°C, respectively. The soils were deep Arenosols with aeolian origin, underlain by calcrete (WRB, 2007).

The vegetation in the area had been classified as the savanna biome, dominated by the Kalahari thornveld and shrub bushveld vegetation (Tainton, 1999). It had been described as the Kalahari Mixed Thornveld A16 (Mucina and Rutherford, 2006), characterized by a fairly well developed tree stratum with *Acacia erioloba, Acacia mellifera, Acacia haematoxylon* (≤ 2 m height) and some *Boscia albitrunca* as the dominant trees. The shrub layer was dominated by individuals of *Acacia mellifera, Acacia hebeclada, Lycium hirsutum, Grewia flava* and *Acacia haematoxylon*. The grass cover contained species such as *Eragrostis lehmanniana*, *Schmidtia kalahariensis* and *Stripagrotis uniplumis*.

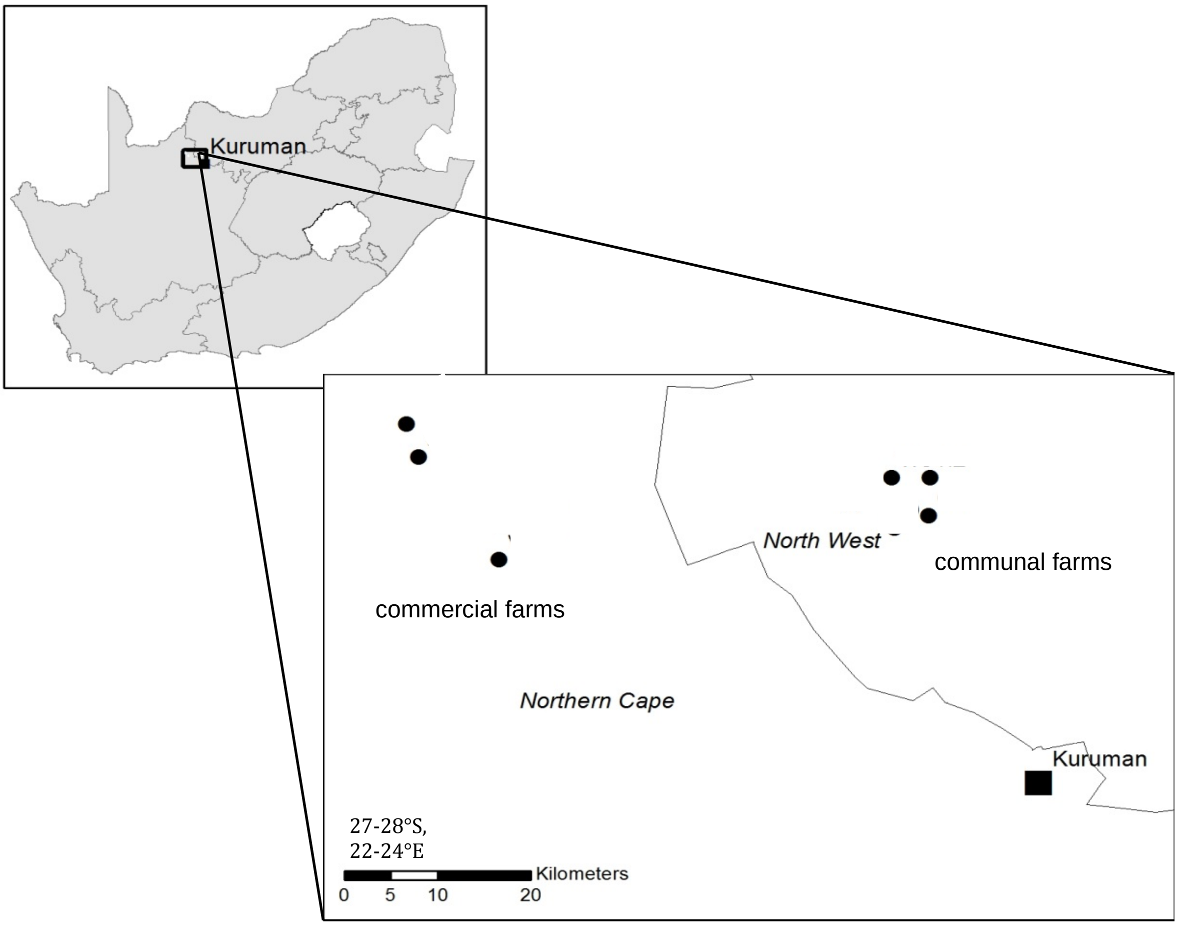


Figure 5.1 Location of study sites comprising of commercial and communal farms near Kuruman at the border of the Northern Cape and North-West Provinces in South Africa.

5.2.2 Rangeland management

Communal and commercial livestock ranching were the most common rangeland management systems in the Kuruman area (Smet and Ward, 2006; Tefera *et al*., 2010). The commercial farms were well developed and mainly market-orientated. They were situated west of the former Bophuthatswana in the Northern Cape Province. Commercial farms (about 70% of all land used in the RSA) were typically managed using a rotational grazing system at moderate stocking densities. The communal farms belonged to the former homeland Bophuthatswana, North-West Province (Figure 5.1), where most of them were developed as an integral part of the Betterment Villages on land previously owned by commercial farmers (Jacobs, 2003). The communal production systems were based on pastoralism and grazing areas were shared by members of a community. The rangeland is a common pool resource with no restrictions in stocking rates and a continuous grazing system. There were often unclear boundaries, with open access rights to grazing areas.

5.2.3 Fieldwork

Fieldwork was conducted during March and April 2011. Communal (continuous grazing) (CO) and commercial (rotational grazing) (CF) rangeland management systems were selected for this study. We sampled three replicates per tenure system, hence three communal and three commercial farms. For each farm a representative grazing gradient was selected, starting nearby an artificial water point, where first vegetation appeared. In commercial farms the gradient belonged to one single camp. The gradients included 6 single plots, each 10 m x 10 m, in size. These plots were defined exclusively through grass quality conditions, according to Van der Westhuizen *et al*. (2005), independent of bush encroachment. Rangeland condition ranged from poor, moderate (poor and moderate = piosphere) to good conditions (good = pasture plot outside the piosphere) defined on-site by plant experts. The length of the grazing gradients differed between and within the different management systems (Table 5.1), but distances between the 6 plots were kept constant within each sampling site. Along a centerline, a soil sample was taken every 2 m with an auger and combined into a composite sample for three depth intervals (0-5, 5-10 and 10-20 cm). Additionally, composite samples were taken in direct proximity to the water points, where no vegetation grew (= sacrifice zone). Samples for bulk density were taken under all rangeland conditions of both management types assuming that bulk density of the sacrifice zone is similar to that under poor rangeland conditions Additional soil samples (0-10 cm) were taken outside the gradients under *Acacia mellifera* and grass, to get an understanding of the influence of specific overlying vegetation on soil parameters.

Plant experts estimated the percentage of bare ground, shrub, litter, moribund (dying vegetation) and senescent (old vegetation) in the plots of the grazing gradients of commercial (3 replicates) and communal (2 replicates) farms.

Table 5.1 Characteristics of commercial and communal farms selected for the study in the Kuruman area

Management system (n=3)	**Length of grazing gradient** [m]	**Length of sacrifice zone** [m]	**Stocking rate** [ha LSU $^{-1}$]	**Rangeland condition**	**Bare soil** (%)*	**Grass cover** (%)*	**Shrub coverage** (%)*	**Sand** (%) 0-5 cm soil depth	**Silt** (%) 0-5 cm soil depth	**Clay** (%) 0-5 cm soil depth
Commercial farms	256 (±18)	24 (±3)	14.2 (±4)	Sacrifice zone	100	-	-	95.6 (±1.2)	1.8 (±0.5)	2.6 (±0.5)
				Poor	56 (±10)	42	4.6 (±1.2)	97.1 (±0.1)	1.1 (±0.3)	2.6 (±0.3)
				Moderate	47 (±5)	46	3.1 (±0.6)	96.6 (±0.4)	1.1 (±0.04)	2.95 (±0.4)
				Good	25 (±7)	71	3.6 (±1.6)	94.7 (±1.3)	1.8 (±0.8)	3.9 (±0.3)
Communal farms	613 (±58)	45 (±5)	13.1 (±3.8)	Sacrifice zone	100	-	-	98 (±0.4)	0.5 (±0.1)	1.9 (±0.2)
				Poor	68 (±12)	27	9.1 (±0,5)	98.2 (±1.5)	0.8 (±0.3)	3.15 (±0.3)
				Moderate	68 (±4)	27	24.7 (±0.8)	96.7 (±0.05)	1.4 (±0.2)	2.0 (±0.5)
				Good	60 (±3)	38	5.9 (±0.9)	96.6 (±0.3)	1.4 ±0.4)	2.7 (±0.4)

5.2.4 Analyses

The composite soil samples were air-dried, sieved (< 2 mm) and prepared for analyses. All chemical analyses were performed in duplicate, and done according to the following standard methods (The Non-Affiliated Soil Analysis Work Committee, 1990): pH (1:2.5 soil to water suspension), exchangeable Ca, Mg, K and Na (1 mol dm^{-3} NH_4OAc at pH 7), extractable Cu, Fe, Mn and Zn (DTPA solution), and CEC (1 mol dm^{-3} NH_4OAc at pH 7). All of the mentioned elements were determined by atomic absorption. Total C and N were determined by dry combustion using a CHNS analyzer (Elementar-Analysensysteme GmbH, Hanau, Germany). There was no detectable inorganic C, so total C was equal to organic C, further called SOC.

Plant-available P was measured using the Olsen method (The Non-Affiliated Soil Analyses Work Committee, 1990) for all samples and for samples of 0-5 cm soil depth, sequential extraction of P fractions was done according to the scheme of Hedley *et al*. (1982) and Tiessen and Moir (1993), but slightly modified in that the soil volume for analyses was doubled. Preliminary tests had shown that this did not affect the overall P fraction yields (own unpublished data). The method is generally based on the extractability of P fractions. Briefly, first, the easy-extractable (labile) P forms were extracted in a batch, where for resin P anion exchange strips were used and 0.5 M $NaHCO_3$ than extracted the bioavailable P (P_o and P_i). The following three fractions – bicarbonate P, Al and Fe bound P, and Ca bound P – were extracted through the same procedure but with different reagents (0.5 M $NaHCO_3$ buffered at pH 8.5; 0.1 M NaOH; 1 M HCl). For the very stable phosphorus pool we took concentrated HCl (cHCl) and at least the residual P pools were achieved with an aqua regia digestion.

Particle size analyses were done by the standard sieve-pipette method (The Non-Affiliated Soil Analysis Work Committee, 1990). Bulk density was determined by gravimetric analysis using 0.1 dm^3 steel cylinder samplers.

A two-step particle size fractionation was conducted by an ultrasonic dispersion method of Amelung and Zech (1999). Briefly, samples were gently sonicated (60 J ml^{-1}) so that microaggregates were preserved from disruption. The coarse fraction (cPOM: 2000–250 µm) was separated by wet sieving, and the filtered remnant was sonicated a second time at 440 J ml^{-1}. Intermediate (mPOM: 250–53 µm) and fine (fPOM: 53–20 µm) fractions were also gained by wet sieving and all fractions were dried at 40°C prior to elemental analysis. Sieving was supported by gentle agitating using small rubber spatulas. Mineral associated matter (MOM) were calculated from data of POM.

Bulk soil (0-5 cm) and soil fractions were analyzed for $\delta^{13}C$, using an Isotope Ratio Mass Spectroscope (Delta V Advantage IRMS, Thermo Electron Corporation, Germany) with respect to the Peedee Belemnite standard according to Equation 5.1:

$$\delta^{13}C=[(R(sample) / R(standard)) - 1] \times 1000, \qquad (5.1)$$

where R(sample) is the $^{13}C/^{12}C$ isotope ratio of the sample and R(standard) is the $^{13}C/^{12}C$ isotope ratio of the international Peedee Belemite standard.

The relative proportions of SOC derived from the original C4 grassland vegetation (FC) vs. the more recent C3 woody vegetation was estimated by mass balance (Liao *et al*., 2006b) according to Equation 5.2:

$$Fc= (\delta P - \delta A) / (\delta G - \delta A) \qquad (5.2)$$

where δP is the $\delta^{13}C$ value of the SOC in a POM fraction, δG is the average $\delta^{13}C$ value of SOC in that same fraction from remnant grasslands, and δA is the average $\delta^{13}C$ value of woody plant material from *Acacia* vegetation (Table 5.2). $\delta^{13}C$ of remnant grassland in Kuruman was defined as -17,1‰, according to our analyses of $\delta^{13}C$ under grass vegetation outside the piospheres (Table 5.2). The $\delta^{13}C$ woodland SOC at equilibrium is not known, because there is no stable forest in this area, i.e. the $\delta^{13}C$ of soil in wooded areas is still changing. Hence, we assumed that woodland soils would ultimately achieve a $\delta^{13}C$ value similar to that of *Acacia* plant inputs (-26.96‰). This ignores that during SOM formation there may be an isotopic enrichment of 1-3 delta units; yet, only surface soils were analyzed here with likely very slow organic matter degradation due to dryness.

Table 5.2 Carbon [g kg^{-1}] and $\delta^{13}C$ [‰] concentrations in bulk soil and fractions of particulate organic matter (0-5 cm) in grazing gradients, of commercial and communal farms, in bulk soil (0-10 cm) under *Acacia* and grass vegetation, and in plant material of *Acacia* and grass vegetation (Standard deviation in parentheses)

	Bulk soil $\delta^{13}C$ ‰	**C g kg^{-1}**	**$\delta^{13}C$ ‰**	**C g kg^{-1}**	**$\delta^{13}C$ ‰**	**C g kg^{-1}**	**$\delta^{13}C$ ‰**
		cPOM (250-2000µm)		**mPOM** (53-250µm)		**fPOM** (20-53µm)	
Commercial farm							
poor	-18.6 (±1.2)	15.8 (±1.2)	-19.3 (±1.6)	8.7 (±0.7)	-20.8 (±1.1)	4.5 (±0.3)	-20.1 (±1.6)
moderate	-18.1 (±0.6)	1.4 (±0.1)	-18.9 (±1.0)	3.5 (±0.4)	-21.8 (±1.3)	1.4 (±0.05)	-20.9 (±0.6)
good	-19.4 (±2.2)	6.2 (±0.1)	-20.5 (±1.3)	4.1 (±0.4)	-21.5 (±0.9)	2.0 (±0.08)	-21.8 (±1.0)
Communal farm							
poor	-18.4 (±2.5)	5.1 (±0.3)	-22.4 (±2.7)	2.0 (±0.2)	-21.1 (±1.6)	1.4 (±0.06)	-22.5 (±2.2)
moderate	-19.9 (±0.7)	8.7 (±0.1)	-23.9 (±2.4)	2.6 (±0.4)	-22.4 (±0.6)	2.3 (±0.1)	-21.7 (±0.8)
good	-19.6 (±1.6)	7.3 (±0.5)	-23.0 (±2.5)	4.2 (±0.2)	-22.0 (±1.3)	2.0 (±0.08)	-21.5 (±1.0)
			$\delta^{13}C$ ‰ plants				
under *Acacia*	-21.4		-26.9				
under grass	-17.1		-13.6				

5.2.5 Statistical Analyses

Statistical analyses were performed on all measured soil properties using analyses of variance in order to determine statistically significant differences between the main experimental factors: rangeland management systems, rangeland conditions and soil depths. Regression analyses was used to test if the measured soil properties could be used to predict differences in rangeland management systems, as well as rangeland conditions. For comparison of means we used the least significant difference (LSD) method with a $p < 0.05$ level of significance. All analyses were conducted using the Statistica 9.1 package for Windows (StatSoft Inc., 2010).

5.3 Results

The visual grazing gradients were significantly longer in the communal (610 m) than in the commercial farms (256 m) (Table 5.1). Additionally, the sacrifice zone around the water points (bare soil) was significantly larger on communal farms (45 m) compared with the commercial farms (24 m). These findings were underlined by visual estimations of bare ground surface in the gradients. Bare ground was highest in the sacrifice zone (100%) near the water points and reached 68% in poor and moderate rangeland conditions of communal farms. Under good rangeland condition outside the piosphere, still 60% of the land surface remained as bare ground. The commercial farms also exhibited 100% of the land surface as bare ground in the sacrifice zone, but values declined to 56% for poor, 47% for moderate and 26% for the good rangeland conditions, respectively (Table 5.1). Additionally, the area influenced by shrubs (including beneath canopy) varied within the gradients. In communal farms the space influenced by shrubs amounted to 9% under poor, 25% under moderate and 10% under good rangeland condition, respectively. In contrast, in the commercial farms shrub cover was much lower and reached on average, only 3.8% of the area of the surface soil (Table 5.1 and Figure 5.2). Thus, the rotational grazing at the commercial farms and measures taken against bush encroachment did not prevent that only bare soil remained nearby the water points. However, the overall area of rangeland degradation was significantly reduced, as indicated by the lengths of the grazing gradients, the areal extension of bare ground as well as by the presence of bushes.

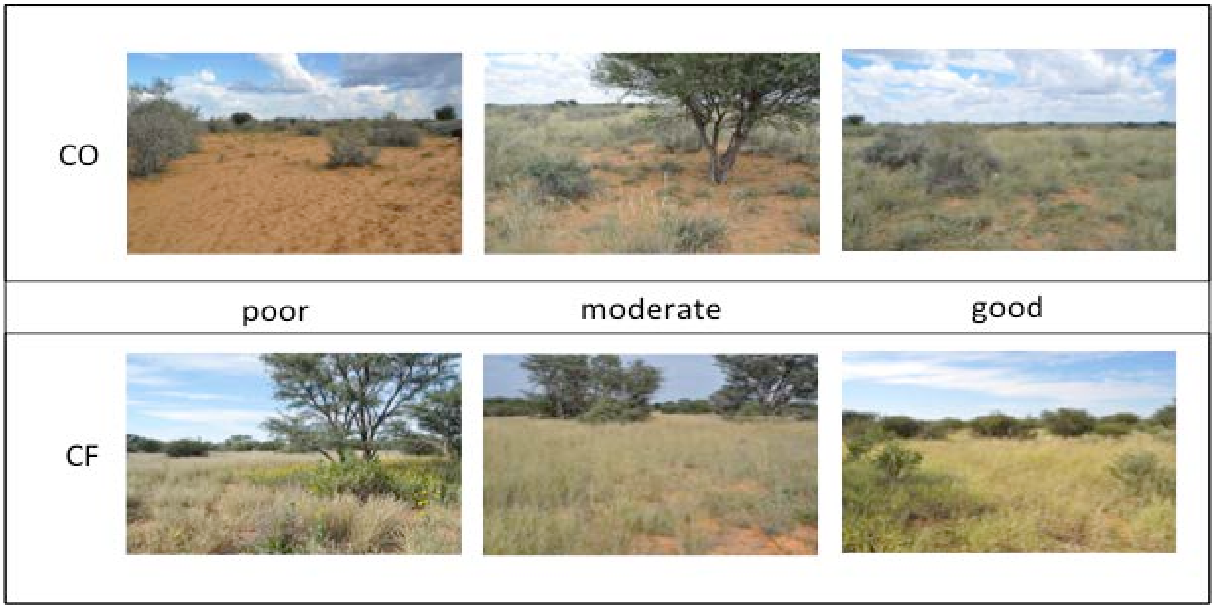


Figure 5.2 Rangeland in poor, moderate and good condition sampled on communal (CO) and commercial (CF) farms near Kuruman.

5.3.1 Physical soil properties

Increased animal trampling did not alter bulk densities (Table 5.3) within the grazing gradients of both management systems, nor could we detect any clear management effect on bulk densities with increasing soil depth. Hence, for understanding changes in soil chemical properties (see below), it was sufficient to consider changes in element concentrations, because at constant bulk densities they paralleled changes in element stocks.

The lack of soil compaction corresponded to the soils' sandy texture and coinciding absence of aggregate formation. Specifically the Arenosols of the studied sites contained between 95 and 98% sand, where communal farms had the highest sand contents in the sacrifice zones (97.9%) and under poor rangeland conditions (98%) in the 0-5 cm soil layer. Differences were small, but the topsoils (0-5 cm) of the commercial farms had more silt ($p < 0.05$) in the sacrifice zones (1.8%) and under poor rangeland conditions (1.1%) compared with the communal farms (0.5% and 0.8%, respectively). Under moderate rangeland conditions, the silt content of communal farms in the 0-5 cm soil layer increased to 1.4% and in the 0-10 cm soil layer even to 2.1%, and exceeded values of commercial sites with 1% silt in the same soil depth (data not shown). In an absolute sense these differences were small; yet, they may already affect the result of POM fractionation.

Fractionation of particulate organic matter clearly separated the soil properties of the different farm types: POM fractions, expressed as percent of bulk soil, reflected the slight texture differences in rangeland management. The relationship between coarse sand and cPOM was linear ($r = 0.94$; $p < 0.001$) as well as that of fine sand and mPOM ($r = 0.96$, $p < 0.001$). As a result, communal farms with higher contents of fine sand showed significantly higher values for mPOM (81%) and lower levels for cPOM (12%) than the commercial farms (63% mPOM and 29% for cPOM), respectively.

5.3.2 Chemical soil properties

5.3.2.1 Nutrients

The Arenosols typically showed poor nutrient status with an average CEC (based on adsorbed Na) of 500 mg kg^{-1} soil. Significant differences between management systems occurred particularly for the soils of the sacrifice zone nearby the artificial water points (Table 5.3), where the CEC of the commercial farms reached extraordinary high values of 1068 mg kg^{-1} in the topsoil (data not shown). This was almost threefold higher than the values of the communal farms (377 mg kg^{-1} soil). The pH values as well as the contents of Ca, Mg, K, and Zn were also elevated nearby the water points of both management systems, with significant higher contents for Cu and Zn under poor rangeland condition in the commercial farms with rotational grazing (Table 5.3). The concentration of most nutrients decreased slightly with increasing soil depth. Notably, the two management systems showed different patterns for nutrients within the grazing gradients. In the communal farms, the topsoils (0-10 cm) under the moderate and good rangeland conditions were frequently enriched with nutrients relative to those under poor rangeland conditions and of the sacrifice zones. In contrast the commercial farms were depleted of most nutrients in the topsoils under moderate and good rangeland conditions relative to the poor rangeland conditions and sacrifice zones (Table 5.3).

Table 5.3 Bulk density, pH, plant nutrient concentrations and standard deviation of the rangeland management systems (commercial farms - CF, communal farms - CO, in different rangeland conditions (sa – sacrifice zone; p – poor; m- moderate; g- good) in three soil depths

Tenure System	**Rangeland Condition**	**Depth** (cm)	**Bulk Density** [g cm-^{3}]	**pH** [H_2O]	**C** [g kg^{-1}]	**N** [g kg^{-1}]	**P** [mg kg^{-1}]	**Ca** [mg kg^{-1}]	**Mg** ([mg kg^{-1}]	**K** [mg kg^{-1}]	**Na** ([mg kg^{-1}]	**Cu** [mg kg^{-1}]	**Fe** [mg kg^{-1}]	**Mn** [mg kg^{-1}]	**Zn** [mg kg^{-1}]
CF	**sa**	0-5		8.0	13.7	1.2	96.6	1164	140	404	26	0.4	12.4	6.1	3.1
				±1.0	±0.5	±0.04	±44	±465	±31	±264	±36	±0.09	±6	±2.8	±0.5
		5-10		8.1	11.3	1.0	99	1381	169	521	34	0.4	19.7	7.6	3.5
				±1.2	±0.3	±0.03	±57	±441	±84	±367	±51	±0.2	±11	±5.3	±0.9
		10-20		8.1	4.2	0.4	82	652	87	281	10	0.4	24.5	5.4	2.9
				±0.9	±0.1	±0.0	±70	±223	±35	±209	±12	±0.2	±25	±1.6	±1.2
	p	0-5	1.54	6.9	4.8	0.38	8.7	331	53	80	3.7	0.2	6.6	5.2	1.2
			±0.03	±0.5	±0.16	±0.01	±1.5	±45	±13	±14	±1.1	±0.02	±1.6	2.4	±0.4
		5-10	1.58	6.6	2.9	0.25	6.4	326	42	77	3.7	0.25	9.0	4.8	0.92
			±0.0	±0.6	±0.1	±0.01	±3.6	±84	±11	±17	±1.2	±0.0	±1.4	±1.7	±0.4
		10-20	1.58	6.6	1.4	0.07	12.1	255	37	91	4.1	0.2	7.6	4.1	0.3
			±0.04	±0.5	±0.0	±0.0	±8.6	±21	±4.7	±8.1	±0.8	±0.0	±1.5	±1.4	±0.1
	m	0-5	1.56	6.2	1.9	0.1	2.2	209	35	51	3.4	0.2	7.2	4.9	0.2
			±0.02	±0.4	±0.05	±0.0	±0.2	±45	±15	±13	±0.6	±0.0	±1.6	±0.5	±0.0
		5-10	1.55	6.3	1.5	0.1	3.4	219	36	58	3.3	0.2	7.5	4.9	0.1
			±0.09	±0.4	±1.2	±0.0	±3.8	±31	±14	13	±0.2	±0.0	±3.0	±0.6	±0.0
		10-20	1.54	6.3	1.3	0.05	1.6	242	42	59	3.5	0.2	6.2	3.8	0.1
			±0.09	±0.3	±.0.0	±0	±0.8	±54	±16	±17	±0.3	±0.0	±2.2	±1.1	±0.1
	g	0-5	1.53	6.2	2.7	0.2	2.1	299	50	64	3.3	0.2	8.5	5.6	0.2
			±0.08	±0.4	±0.07	±0.04	±1.1	±108	±25	±17	±0.5	±0.0	±4.1	±1.4	±0.0
		5-10	1.55	6.4	1.9	0.1	1.2	268	45	62	3.6	0.2	7.3	4.1	0.1
			±0.0	±0.5	±0.1	±0.07	±0.7	±94	±18	±17	±0.7	±0.0	±3.8	±0.8	±0.1
		10-20	1.54	6.5	1.6	0.1	1.1	321	59	75	3.2	0.2	6.1	3.4	0.1
			±0.08	±0.4	±0.0	±0.0	±0.4	±140	±34	±32	±0.2	±0.0	±2.7	±0.5	±0.0
CO	**sa**	0-5		8.5	2.4	0.2	5.5	411	41	191	3.4	0.1	3.0	4.5	1.1
				±0.1	±0.03	±0.0	±1.2	±13	±1.4	±10	±0.6	±0.0	±0.0	±0.9	±0.0
		5-10		8.1	1.3	0.2	7.6	370	35	139	3.3	0.2	2.6	2.8	0.7
				±0.1	±0.1	±0.01	±2.0	±14	±7	±9.9	±0.4	±0.0	±0.7	±0.8	±0.1
		10-20		8.1	1.0	0.01	8.8	331	35	135	3.4	0.19	2.9	3.1	0.5
				±0.1	±0.1	±0.0	±2.7	±4.2	±7	±24	±0.5	±0.0	±0.5	±0.7	±0.0
	p	0-5	1.6	6.9	2.2	0.2	2.7	309	31	72	2.7	0.15	4.5	5.6	0.3
			±0.02	±0.8	±0.1	±0.0	±0.4	±84	±12	±35	±0.2	±0.01	±0.8	±1.9	±018
		5-10	1.6	6.7	1.5	0.1	1.9	277	27	67	2.9	0.16	4.3	4.8	0.2
			±0.03	±0.8	±0.03	±0.03	±0.6	±61	±8	±35	±0.2	±0.0	±1.4	±1.7	±0.1
		10-20	1.56	7.1	1.4	0.1	1.5	358	29	79	2.9	0.18	4.1	4.7	0.15
			±0.02	±0.1	±0.1	±0.03	±0.02	±90	±8	±32	±0.2	±0.0	±1.4	±2.7	±0.1
	m	0-5	1.5	7.0	3.4	0.2	1.6	392	29	51	4.5	0.2	7.6	8.7	0.2
			±0.05	±0.8	±0.5	±0.0	±0.3	±218	±3	±8	±0.9	±0.0.	±2.3	±6.2	±0.04
		5-10	1.6	7.0	2.2	0.2	1.1	371	31	54	3.7	0.2	4.9	5.6	0.2
			±0.03	±0.8	±0.1	±0.04	±0.2	±201	±8	±11	±1.1	±0.02	±1.2	±2.9	±0.1
		10-20	1.5	6.4	1.7	0.2	1.1	347	32	52	1.4	0.1	4.6	6.5	0.1
			±0.05	±0.8	±0.1	±0.05	±0.3	±184	±7	±10	±0.2	±0.01	±0.8	±3.8	±0.01
	g	0-5	1.55	6.8	3.0	0.2	1.4	285	32	48	2.7	0.15	5.3	6.0	0.2
			±0.03	±0.6	±1.2	±0.01	±0.5	±41	±3.5	±5.3	±0.2	±0.01	±1.2	±2.2	±0.05
		5-10	1.58	6.9	1.9	0.15	1.1	281	31	54	3.0	0.15	4.3	4.1	0.1
			±0.04	±0.5	±0.2	±0.02	±0.3	±28	±3.1	±5.3	±0	±0.03	±0.4	±1.8	±0.01
		10-20	1.52	6.9	2.0	0.18	1.2	308	34	55	1.5	0.1	3.9	5.1	0.09
			±0.05	±0.5	±0.2	±0.0	±0.32	±60	±4	±7	±0.3	±0.01	±0.39	±2.3	±0.06

The total P content of all farms was low and ranged between 71 mg kg^{-1} P for commercial farms and 52 mg kg^{-1} P for communal farms in the surface soil layer (data not shown). Highest values were again found under poor rangeland condition, with significant higher P contents in soils of the commercial farms. Along the grazing gradient the P contents decreased with increasing distance from the water points, but at all sites P was elevated in the commercial farms relative to the respective rangeland conditions of the communal farms. The proportions of P_i and P_o did not vary much among rangeland conditions. Plant-available P (Table 5.3) followed the same trend as total P in commercial farms, whereas in the communal farms, differences across all rangeland conditions remained low.

In contrast to the assessment of P contents, sequential fractionation of P into different fractions detected variation between the two management systems and, in parts (Figure 5.3; for extraction agents see Section 5.2.4), also along the gradients (data not shown). In the commercial farms, although not significant, the labile P pool (resin and bicarbonate extractable P fractions) was higher in the commercial than in communal farms, with a tendency to decrease along gradients away from the water points of the commercial farms (46.9% under poor, 42.0% under moderate, and 42.5% under good rangeland condition). This trend was reversed for the communal farms (33.7% under poor, 35.6% under moderate, and 36.0% under good rangeland condition). In both management systems, the labile pool was dominated by bicarbonate extractable P_o, though its contents were consistently lower along the gradient of the communal farms than along that of the commercial farms. In contrast, portions of the recalcitrant P were higher under poor rangeland conditions of the communal farms with continuous grazing system, whereas this fraction accounted only for 17% of the recalcitrant pool in commercial farms. Fractionation of the P pools under certain overlying vegetation showed differences between grass and *Acacia* vegetation, i.e., changes in the proportions of these vegetation types likely also affected P pool composition (Figure 5.3).

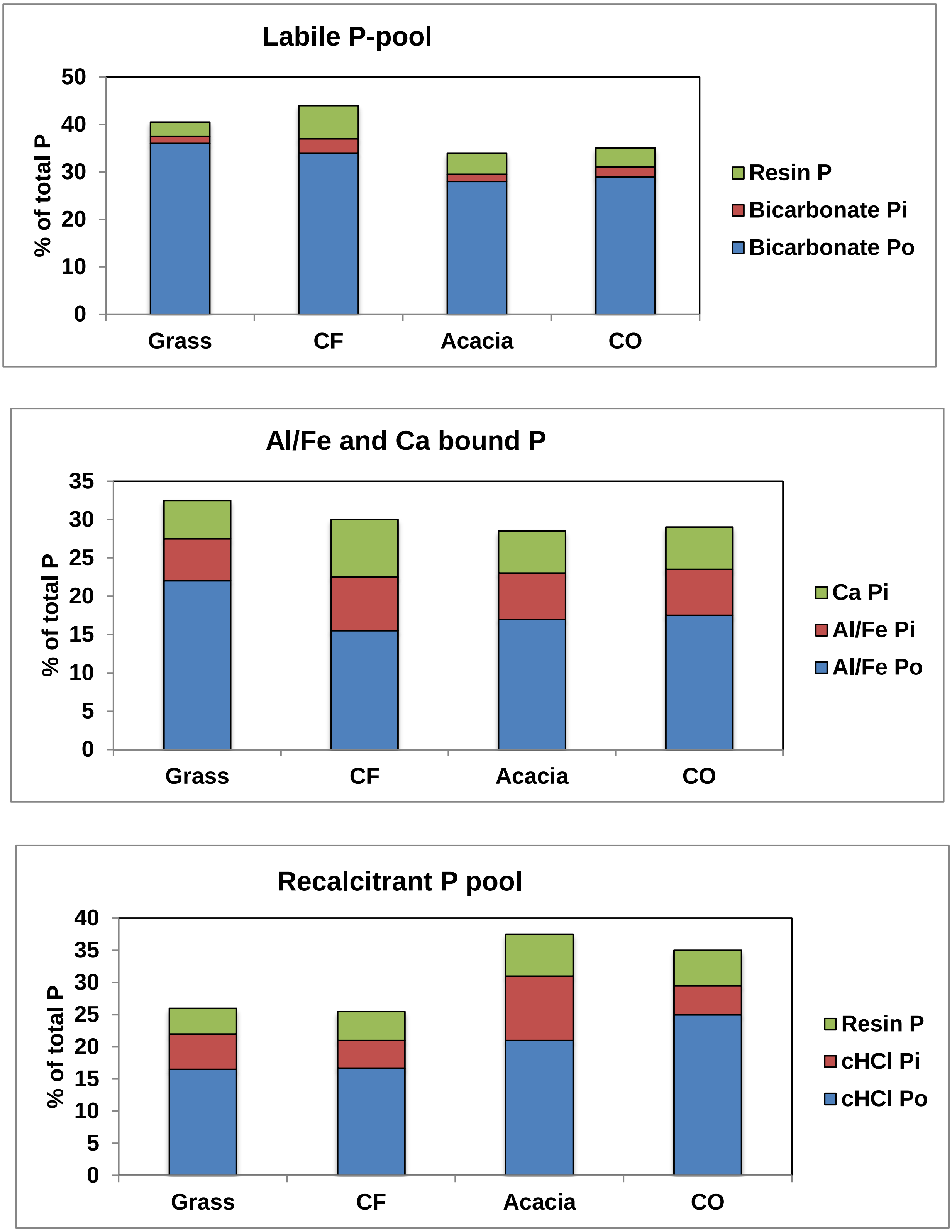


Figure 5.3 Propotions of phosphorus pools and fractions within pools in commercial (CF) and communal (CO) farms, and under overlying grass and *Acacia* vegetation.

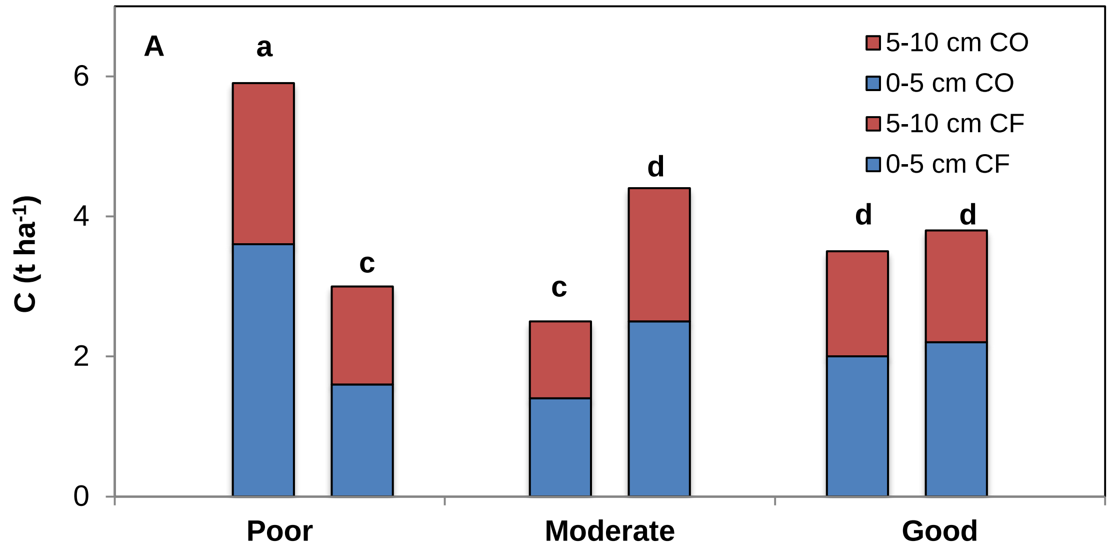
5.3.2.2 Soil organic matter

The stocks of SOC and total N followed the patterns of nutrients, but relative differences among sites were greater. Under poor rangeland conditions, SOC and N stocks of the commercial farms exceeded those of the communal farms, by a factor of 2.1 for the top 5 cm soil layer and by a factor of 2 for the top 10 cm soil layer. In contrast, communal farms with continuous grazing showed high SOC and N stocks in the 0-10 cm soil layer under moderate and good rangeland conditions, that now exceeded the stocks of commercial farms with rotational grazing by almost 2 t SOC ha^{-1} under moderate and 1.3 t SOC ha^{-1} under good rangeland conditions. Notably, the effects were restricted to the 0-10 cm soil layer, whereas in the 10-20 cm soil layer, SOC and N stocks were similar along the whole grazing gradient and between both land-use systems (Table 5.3; Figure 5.4).

The C:N ratios ranged between 11 and 23, although in the topsoil of the sacrifice zone the ratios of commercial and communal farms were similar (11.5 to 11.9). In the 5-10 cm and 10-20 cm soil layers, the C:N ratios in communal farms increased to 15 and 20, respectively, whereas the ratios in commercial farms remained more or less constant. Hence, there was a significant alteration in soil quality in the deeper soil layers.

The different patterns of soil fertility at both management systems were underlined by Pearson correlations (Pearson), revealing close positive relationships between the C (0.81***) and N (0.78***) contents and grazing intensity at the commercial farms. At the communal farms with continuous grazing, strongest rank correlations were detected between C stocks (0.71***) and CEC (0.78***) with shrub cover.

The stocks of the three different POM fractions and the mineral associated fraction (MOM) reflected differences in total SOC. Most SOC was found in the MOM fraction with about 46% of the total SOC and the least SOC were found in fPOM. Under poor rangeland condition,



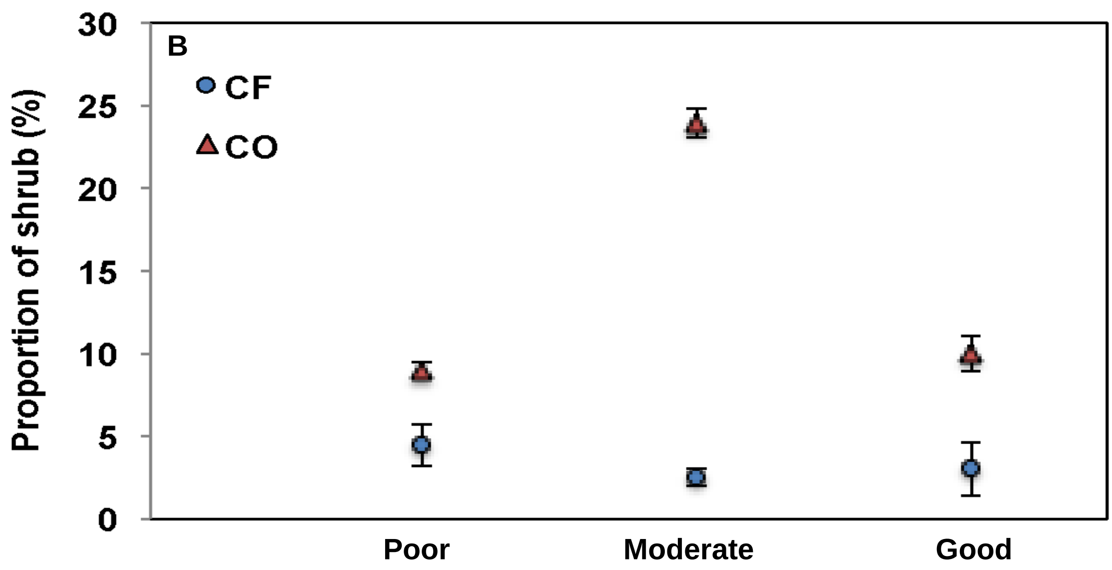


Figure 5.4 Carbon stock in soil (A) and proportion of shrub (B) under poor, moderate and good rangleand conditions of commercial (CF) and communal (CO) farms. Statistical difference is indicated by letters.

however, commercial farms showed again higher SOC and N stocks in all isolated POM fractions, including MOM relative to the communal farms. Differences for cPOM and fPOM were significant. With increasing distance to the water point and under moderate rangeland condition, SOC stocks in POM fractions of communal farms exceeded those of the commercial farms, in particular for cPOM (Figure 5.5). Only outside the piosphere, under good rangeland condition, both farming systems exhibited similar POM-C contents. The N stocks in the POM fractions followed the same trend as SOC, but differences were most pronounced for MOM in moderate and good rangeland condition with higher N stocks in communal farms (data not shown).

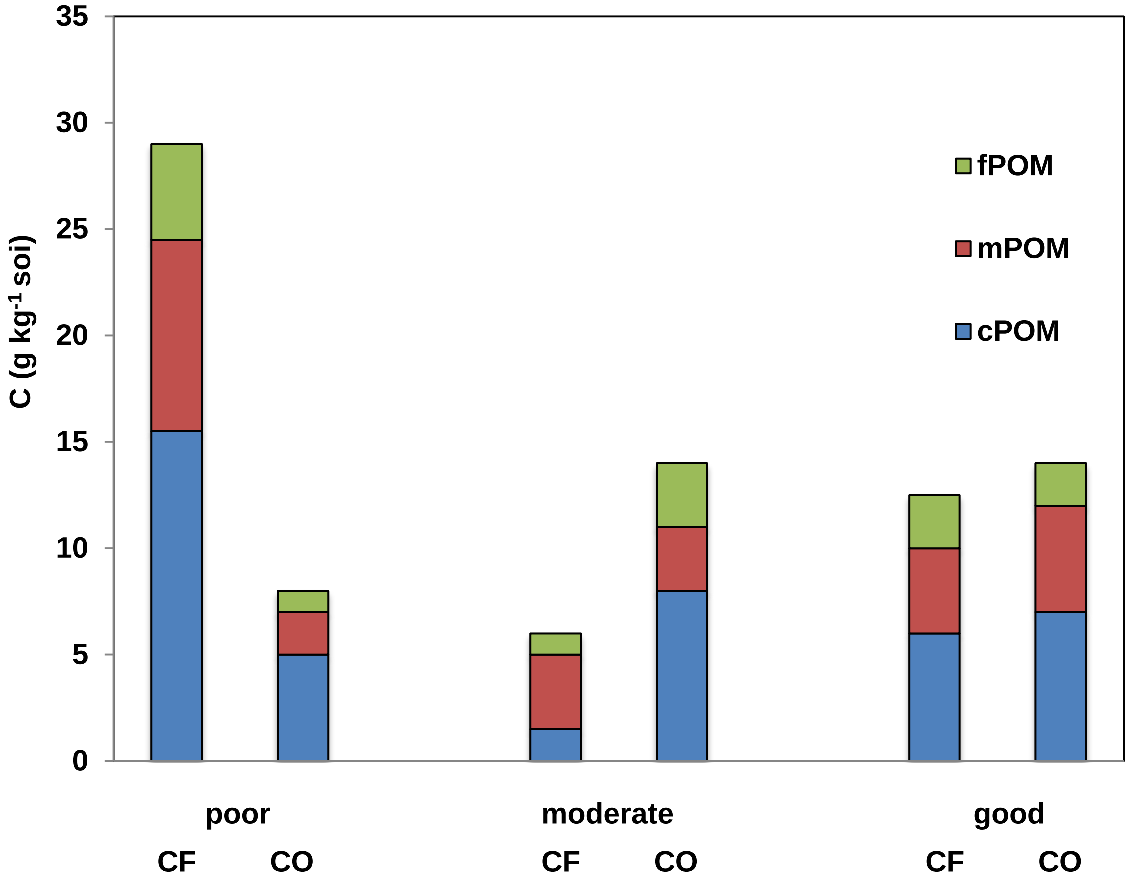


Figure 5.5 Carbon concentrations in fine (fPOM), intermediate (mPOM) and coarse (cPOM) fractions under poor, moderate and good rangeland conditions of commercial (CF) and communal (CO) farms.

The stable C isotopic composition ($\delta^{13}C$) of the plants in Kuruman reflected the differences in photosynthetic pathways between the C4 grasses and the C3 *Acacia* bushes and trees (Table 5.2). In the bulk soil (0-5 cm), the $\delta^{13}C$ values decreased with increasing distance to the water points in both management systems along the grazing gradient with higher levels in commercial farms (Table 5.2). In spite of the small non-significant differences in bulk soil between the two management systems, the $\delta^{13}C$ values differed significantly in the cPOM fraction. In general, cPOM had a $\delta^{13}C$ of -18.9 to -20.5‰ in commercial farms and of -22.4 to -23.9‰ in the communal farms. This difference in isotopic composition was most pronounced under moderate rangeland condition (Table 5.2), and reflects different portions of, e.g., C4-derived C in the POM fraction (Figure 5.6). Using the calculation outlined in Equation 5.2 revealed that under poor and good rangeland condition of the communal farms, more than half of the SOC was derived from C4 grasses. This proportion declined under moderate condition to 33.6% of the total SOC. In contrast, in the commercial farms, 75% of SOC was derived from the C4 grasses, and under moderate rangeland conditions even 90%.

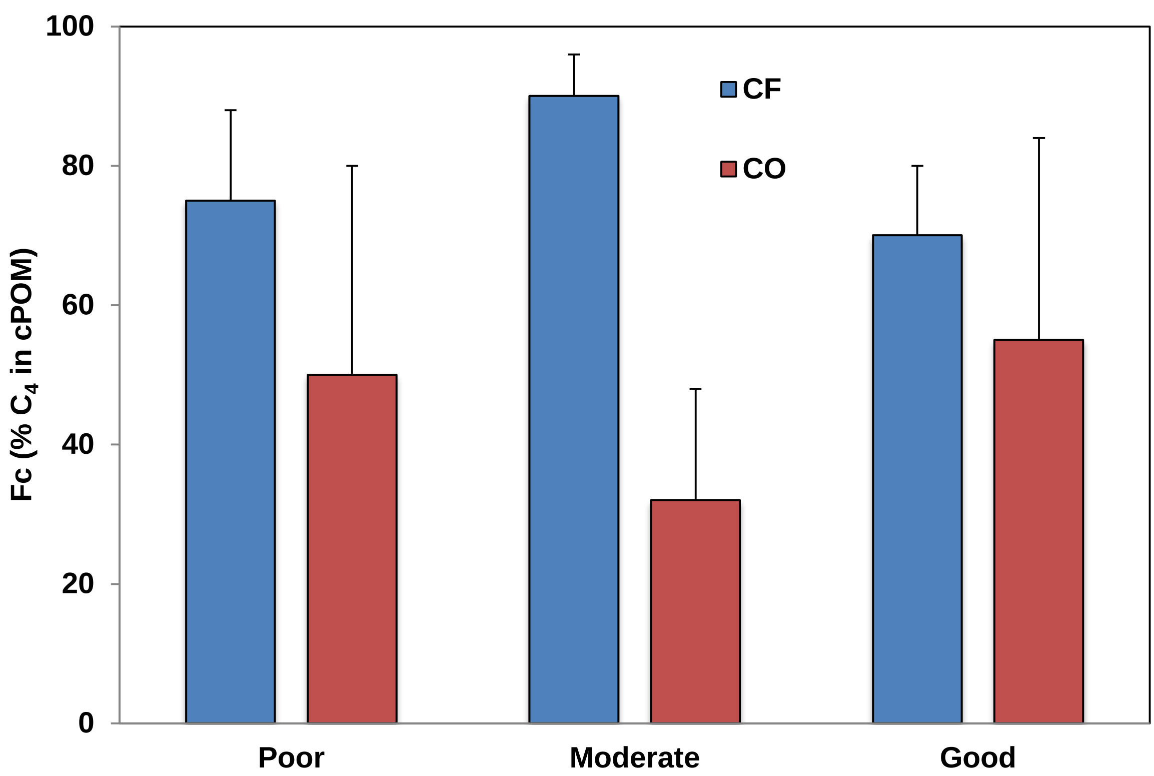


Figure 5.6 Relative amount of C4-derived C in the coarse fraction of particulate organic matter (Fc) under poor, moderate and good rangeland conditions of commercial (CF) and communal (CO) farms.

5.4 Discussion

Since water is the main limiting factor of plant production and animal welfare in this savanna ecosystem, the activities of animals in the rangelands are concentrated around the water points. As a result and as explained in the introduction, piospheres developed in radial gradients nearby the water points. They show a sacrifice zone in intimate vicinity to the drinking points, and impacts of animals attenuate with increasing distance to the water points. Most studies dealing with piospheres measured soil properties or vegetation patterns in defined distances to the water points up to 500 m (e.g. Fernandez-Giminez and Allen-Diaz, 2001; Snyman and du Preez, 2005; Smet and Ward, 2006) or more and tested whether soil or vegetation properties changed with distance (e.g. Shahriary *et al*., 2012). Instead of that, we chose a visual defined grass quality gradient to estimate the total length of the piosphere until averaged pasture condition (= good) were reached to compare the effects of different management systems on soil properties as also done by local rangeland extension services (Van Westhuizen *et al*., 2005). This means, that the expressions “poor, moderate or good” condition, according to the local rangeland classification, were linked to palatable grass composition and grazing intensity but not to the occurrence of bushes. The results showed distinct differences between the grazing systems, with significant enlarged sacrifice zones and enlarged lengths of the piospheres in the continuous grazing system of the communal farms compared with those in the rotational grazing system of the commercial farms.

The estimated carrying capacity of the rangelands under study is 13 ha LSU^{-1} (Department of Agriculture and Rural Development, 2003), and the stocking number of commercial farms fitted well with this carrying capacity (averaged stocking rates of about 14 ha LSU^{-1}). The stocking rates of continuous grazed management systems were rough estimates, but surprisingly, an estimation of the headmen of the communal farms brought results of averaged, in fact widely varying stocking rates of 13 ha LSU^{-1} similar to that of the commercial farms. Therefore, there must have been other reasons than stocking rates for enlarged piospheres under the continuous grazing system.

A fundamental driver for animals' movement is the number of intact water points and the establishment of fences. After incorporation of Bophuthatswana under South African government in 1994, financial support from the state for fences and water points ran dry and the communal farms were not able to repair them. Animals were forced to enlarge tracks to reach water, resulting probably in an overall extension of the piosphere. These arguments are supported by a recent study of Graz *et al*. (2012), who modeled grazing patterns after closure of water points and removal of fences, simultaneously. As a consequence, an extension of the grazing area developed, concentrated to the single remaining water points. In contrast to communal farms, commercial farms are divided into camps with artificial water points in every camp. They are therefore able to manage the movement of the animals and the grazing in a given camp, which allows soil restoration.

Interestingly, the effects of the two management systems on soil properties and rangeland conditions studied in the savanna ecosystem in the Northern Cape and North-West Provinces were more pronounced than in a grassland ecosystem in the Free State Province (between 28.95"S, 26.46"E and 29.41"S, 27.00"E), where no significant differences between extension of piospheres of continuous and rotational grazed rangelands were observed, but where the soils clearly degraded as evidenced from increasing SOM loss and deterioration of aggregates (Kotzé *et al*., 2013). Both processes were absent here.

The effects of the different management systems on bulk density in this sandy savanna ecosystem were also low. In contrast, many studies detected soil compaction as a consequence of increased grazing pressure (e.g. Du Toit *et al.*, 2009; Kotzé *et al.*, 2013), but most of these studies were not conducted on such sandy soils, lacking the formation and thus also the destruction of aggregates. The sand content in Kuruman area exceeded 93% on both farming systems, and Van Haveren *et al.* (1983) observed strong correlations between texture and bulk density under different grazing pressure with no effects of heavy grazing on coarse textured soils. Hierneux *et al*. (1999) could also not find any higher bulk density under increasing grazing of sandy soils. Hence soil compaction does not seem to be an issue for rangeland degradation in this savanna ecosystem.

Nevertheless, due to continuous grazing and frequent trampling by sheep and cattle, the ground surface become bare and exposed to wind erosion. Yong-Zhong *et al*. (2005) observed in China that decreased vegetation cover and litter accumulation resulted in soil coarsening and higher sand contents. In our study, soil coarsening and a loss of fine soil fractions in communal farms were found as well, but these findings were restricted to the sacrifice zone and the poor rangeland condition. Here, sand content increased by 1.7%. A lower increase of 1.1% was observed in the commercial farms. Simultaneously, there was a loss of fine fractions near the water point and under poor rangeland condition of communal farms. Especially fine silt was lost, which resulted in less than half of the quantities we analyzed in moderate and good rangeland conditions. The vegetation cover at the water point and under poor rangeland condition reached 0% and 27%, respectively. Hence, the soils were indeed prone to erosion. Evidence for wind erosion was also visible in the field, where roots of, e.g., *Acacia eriloba* were blown free off soil under poor management condition at the communal farms. The rangelands under moderate and good condition showed less bare soil and more vegetation cover, which, however, comprised especially shrubs in the communal farms. Hence, erosion risk diminished with increasing distance to the water points, and higher silt contents prevailed. In this regard, the increasing bush encroachment increasingly protected the soils while overall rangeland quality declined.

5.4.1 Soil fertility changes

At the commercial farms, our study confirmed earlier findings that most changes in soil properties and nutrients occurred in direct vicinity of the water points (Tolsma *et al*., 1987; Fernandez-Giminez and Allen-Diaz, 2001). The high levels of nutrients found can mostly be attributed to the excreta of the animals around the water points (e.g. Smet and Ward, 2006). Especially at the commercial farms we found an enrichment of P from feed additives, which improved the overall low P status of the soils compared with other sandy soils in these or other regions (Wang *et al*., 2009;

Phosphorus contents did not follow the general trend of nutrient enrichment under moderate rangeland conditions of the communal farms. However, the vegetation influenced the P fractions in our soils. Studies under the canopy and around the stem of *Acacia mellifera*, for instance, revealed higher P contents than in the open areas (Hagos and Smit, 2005). Differences in especially inorganic P fractions were found by Turrión *et al*. (2007) under different trees and shrubs. Conversely, soils under grass and pasture may promote the buildup of a large labile, probably microbially derived P_o pool due to the arbuscular mycorrhiza activity. As a result the proportion of P_o under grass and pasture exceeded that of forests (Negassa and Leinweber, 2009). Our results also pointed to lower P_o levels of 55.4% under *Acacia* vegetation relative to elevated proportions of 63.8% under grass. Additionally, the P fraction yields under grass fitted well to the P fractions of the commercial farms. In contrast, the P fractions of the communal farms represented those found under *Acacia* vegetation. We surmise therefore that the feed supplements mainly influenced amounts of total P, especially under poor rangeland conditions, but that plant stands and grazing controlled the P status and bonding forms of the soils under moderate and good rangeland condition.

5.4.2 Origin of soil organic matter

Vegetation controls on soil properties should particularly affect the properties of soil organic matter. In the past decades, tree-shrub-grass composition of study sites in the Kuruman area changed (Jacobs, 2003). Communal farms with continuous grazing and an insufficient number of water points seemed to be more affected than commercial farms with a rotational grazing system in camps. The subtropical vegetation mainly comprises grasses with a C4 photosynthetic cycle, as well as trees and shrubs with a C3 photosynthetic cycle. As both photosynthetic pathways exhibit different ^{13}C discrimination, isotope analyses provided insight into the origin of SOC. The overall $\delta^{13}C$ values were similar to that reported for other C4 grasslands in the world (Boutton *et al*., 1998). Increasing woody encroachment usually changed the isotopic signature in direction of the C3 plants (Bai *et al*., 2012; Throop *et al*., 2013). Our analyses from distinct overlying vegetation showed an average $\delta^{13}C$ value of -21.4‰ for soils under *Acacia* vegetation and a $\delta^{13}C$ value of -17.1‰ for soils under grass, reflecting the increased input of C4 debris in the latter. The $\delta^{13}C$ values of the bulk soil within the gradients were intermediate, suggesting that their SOC comprised a mixture of C4 grasses and C3 plants like shrubs, herbs, and trees, the latter inputs being more pronounced in the communal than in commercial farms.

As the $\delta^{13}C$ signal of bulk soils may integrate across time scales exceeding the age of the water points, we also analyzed the POM fractions for isotopes to better identify residues of the C3 and C4 plants. The analyses revealed that $\delta^{13}C$ values of cPOM in communal farms indeed approached that of the savanna trees and shrubs, supporting the idea that cPOM consists of plant fragments with little if any degree of microbial degradation (Amelung *et al*., 1998). In contrast, cPOM in commercial farms had higher $\delta^{13}C$ values across the whole gradient, which were comparable to cPOM values we analyzed under grass. The low $\delta^{13}C$ values of the cPOM fraction are in line with results from the Rio Grande Plains in Texas. In a chronosequence approach, Liao *et al*. (2006b) studied the $\delta^{13}C$ signature in soil fractions of various ages of woody stands (10-130 years) and remnant grassland. They also observed that beside the macro and micro free POM, the coarse cPOM (>250 µm) adapted most rapidly to the $\delta^{13}C$ value of the woody stands, i.e., they confirmed the rapid turnover and recent origin of this fraction. In this regard, cPOM at the communal farms reflected the shift of an erstwhile grass dominated system to an increasing bush dominated system (Figure 5.5), whereas at commercial farms it reflected the dominance of grass. Hence, monitoring the $\delta^{13}C$ signature of cPOM may be a valuable proxy for monitoring grazing effects on soil properties in this ecosystem.

In summary, our results showed that the drier and more sandy rangelands of the savanna ecosystem, also allowing for the encroachment of bushes, are obviously more vulnerable to management systems with continuous grazing than the systems of the grassland ecosystem, where climate restricts the occurrence of bush and where the loamy soils better preserved organic matter.

5.5 Conclusions

Rangeland management systems in the savanna ecosystem of the Kuruman area are determined by different property rights and intentions of management. Commercial farmers with a rotational grazing system practice a more foresighted management system in fenced camps with the aim of selling animals. Several months a year, vegetation and soil have the possibility to restore. On account of supplement feeding and a high deposition of animal excreta at the water points, the topsoils under poor rangeland conditions are even elevated in most nutrients relative to those under moderate and good rangeland conditions and relative to those of the communal farms. Communal farms, in contrast, have an insufficient infrastructure with broken water points and missing fences and, continuous grazing stresses vegetation and soils over the entire year. The topsoils under poor rangeland conditions were thus depleted in most nutrients compared to those under moderate and good rangeland conditions and relative to those of the commercial farms. Hence, and similar to other ecosystems (Kotzé *et al*., 2013.), less adapted management results in a degradation of soil quality nearby the water points, and the degraded area is also larger than at commercial farms.

The moderate and good rangeland studied here exhibit higher proportions of woody species in communal farms. They favored the accumulation of particulate organic C. The measurements of the isotopic composition of the different physical fractions ascertained its woody origin in the communal farms. Additionally, other nutrients co-accumulated.

In summary, rangeland degradation by bush encroachment at grazing gradients of the communal farms resulted in an improvement of the soils and, probably, a buildup of useable nutrient reserves for future generations. Thus, the challenge for future land management options will be how to utilize these improved soil conditions best, particularly in view of the limited water in these arid regions and this warrants further research.

CHAPTER 6

SOIL MICROBIOLOGICAL INDICATORS FOR SOIL RESILIENCE IN DIFFERENT RANGELAND MANAGEMENT SYSTEMS IN A SANDY SAVANNA AND CLAYEY GRASSLAND ECOSYSTEM, SOUTH AFRICA

Abstract

The long-term sustainability of rangeland management systems is *inter alia* dependent on the maintenance of soil microbiological properties. In this study we investigated the difference in soil microbiological properties and associated parameters in rangeland management systems within two different ecosystems with different climate, vegetation and soil. For this purpose, soils were sampled under communal (continuous) and commercial (rotational) grazing systems along a gradient with increasing grazing pressure. The results revealed that the clayey grassland ecosystem had higher values for all measured microbiological properties compared with the sandy savanna ecosystem, irrespective of the rangeland management practices, likely because soil texture played a significant role in microbial communities. Clay was nearly absent in the sandy savanna ecosystem, and most organic matter was stored in particulate form, which in itself is sensitive to management. Nevertheless, in the clayey grassland ecosystem enzyme activities as well as PLFAs responded more sensitive to grazing pressure than other chemical or physical soil properties, whereas in the sandy savanna ecosystem this was not the case. Decreasing the grazing pressure on a rangeland, such as commercial farmers practicing rotational grazing, appeared to stimulate microbial-mediated nutrient mineralization with positive consequences on plant growth. In contrast, the C/N ratio and therewith the overlying woody *Acacia* vegetation controlled the microbial properties in the savanna ecosystem. Overall the sandy soil of the savanna ecosystem seemed to be more resilient to degradation over the long-term, and less

over the short-term, whereas the clayey soil of the grassland ecosystem was more resilient over the short-term, and less over the long-term.

6.1 Introduction

Soil microbial communities play a fundamental role in ecosystems by regulating the dynamics of organic matter decomposition and plant nutrient availability; and hence, are of paramount importance for the functioning and stability of ecosystems. According to Denef *et al*. (2009) their habitat in soils can be defined as a dynamic and heterogeneous environment characterized by several abiotic and biotic processes, which can significantly transform due to changes in land-use, management or environmental conditions. These microbial communities are sensitive to changes in soil chemical and physical properties, and can therefore be used as an important measure of sustainable land use (Bardgett *et al*., 1997; Patra *et al*., 2005; Xue *et al*., 2008).

The response of ecosystem structure and function as affected by grazing management are inconstant, however grazing intensity appears to be the most important driver of net primary productivity and composition, especially in semi-arid regions (Briske *et al*., 2008). According to Steffens *et al*. (2008) as well as Mofidi *et al*. (2012) inappropriate grazing pressures can cause a decline in the physical, chemical and biological properties of soils, which can lead to rangeland degradation and desertification. Ecologists are becoming increasingly interested in the important role that soil organisms have in regulating ecosystem processes, such as nutrient cycling and organic matter decomposition, and how these decomposer organisms respond to disturbed regimes. The activity and growth of soil organisms is often limited by soil environmental conditions, such as temperature, water, pore-size distribution as well as nutrient availability, and therefore indirectly by rangeland management practices (Patra *et al*., 2007; 2008). It is clear that rangeland management affects the structure and activities of these microbial communities, since their abundance and activity is strongly related to the quantity and quality of available plant litter, which in turn, is related to grazing intensity. Su *et al.* (2004) for example showed that heavy grazing pressures resulted in loss of soil organic C and N, and subsequently in a depletion of soil enzyme activities. Mofidi *et al.* (2012) found similar results in rangelands of Iran, where lower levels of grazing had the highest biological activity. The effect of grazing management on microbial biomass and composition (indexed by phospholipid fatty acids [PLFAs]) however tend to be inconsistent (Oates *et al*., 2012). This might be due to the negative effects of heavy grazing, but positive effects of the added manure on microbial communities. To indicate the effect of manure on soil microbiological properties, Bardgett *et al*. (1997) found in North-Wales that long-term removal of grazing animals from a rangeland resulted in a significant reduction in microbial biomass and activity in the surface soil layer, due to less manure in the soil. According to Degens *et al.* (2000), land uses that deplete soil organic C may cause declines in the catabolic diversity of soil microbial communities. Although the implications of this for microbial processes are unknown, maintenance of soil organic C may be important for preservation of microbial diversity. On the other hand, Bardgett *et al.* (1997) concluded in their study that changes in microbial community structure are also likely to have a profound influence on organic matter dynamics and nutrient supply in an ecosystem.

Information on how grazing in different ecosystems affects the size and composition of key microbial functional groups is limited, and this restricts our understanding of the actual effects of grazing on rangeland functioning. It also affects our ability to predict rangeland response to changes in grazing intensity or management practices, because the composition of microbial communities can determine their resistance and resilience to disturbances (Griffiths *et al*., 2000; Patra *et al.,* 2005). This was shown in a study done by Griffiths *et al*. (2000), where resilience was proved to be lower in soils with decreasing biodiversity. They showed that soils with impaired biodiversity were not resilient to persistent stress, compared to soils with higher biodiversity. Patra *et al.* (2005) also demonstrated that grazing deeply affects microbial functional groups by enhancing the activity of soil microbial communities and inducing changes in the size and composition of these communities, and are important for predicting the effects of changed grazing regimes on rangeland ecosystem functioning and response to disturbance. Microbial community biodiversity is an integral part of soil quality and crucial to maintain ecosystem function.

In summary, there is still inconsistent knowledge on the effects of grazing and rangeland management on microbial community composition of soil in semi-arid ecosystems. Therefore, it was the aim of this study to assess changes in soil microbial community structure and function along grazing gradients within the clayey grassland and sandy savanna ecosystem, respectively. In both ecosystems, different rangeland systems with either rotational or continuous grazing are found. To answer these questions, we analyzed specific soil enzymes that are considered to be representative of the functional component of soil microorganisms and are important for soil quality evaluation, as well as phospholipid fatty acids (PLFA) that are sensitive to changes in overall land-use and plant cover type (Hossain and Sugiyama, 2011). The enzyme β-glucosidase is a very common enzyme involved in the C-cycle, urease is an important enzyme in the N-cycle, both acid- and alkaline-phosphatase are good indicators of soil fertilty, and the dehydrogenase enzyme is a very good proxy for total microbiological activity in soil. Phospholipid fatty acids analyses supports the enzyme analyses and give an indication of the microbial community present in soil.

6.2. Materials and methods

6.2.1 Research areas

Two research areas have been selected, one in a clayey grassland ecosystem and one in a sandy savanna ecosystem of South Africa (Figure 6.1).

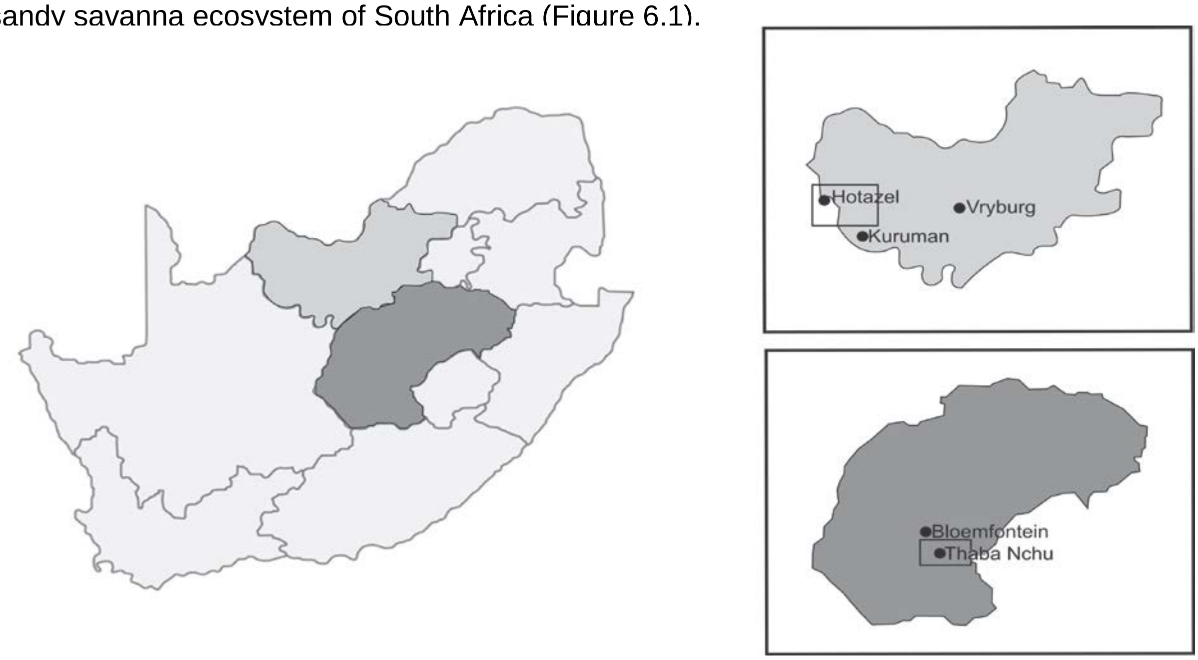


Figure 6.1 Map showing the research areas in the savanna ecosystem near Kuruman (a) and in the grassland ecosystem near Thaba Nchu (b) in South Africa.

The research area for the grassland ecosystem is located near Thaba Nchu, in the Free State Province, South Africa (latitude 28° - 29° S, longitude 26° - 27° E; altitude 1400-1600 m above sea level). The semi-arid climate is relatively dry, has a low, variable and unpredictable rainfall (mean annual precipitation of 553 mm, with about 70% of the rain occurring in summer between January and March) and a high evaporation level of 1832 mm a^{-1} (Basson, 1997). Non-perennial streams exist in the area. The soils are all classified as Lixisols (WRB, 2007), and they have more than 15% clay in the A-horizons and more than 25% clay in the B-horizons, with no obvious signs of wind or water erosion. This research area belongs to the “Moist Cool Highveld Grassland Type”, which is part of the Grassland Biome (Bredenkamp *et al.*, 1996). This grassland type is widespread and covers the central eastern part of the Highveld in the Free State Province. A single layer of perennial C4 bunchgrasses dominated this particular grassland type, and trees are sparse. The amount of grass cover depends beside the low rainfall and evapotranspiration on the degree of grazing. Frost, fire and grazing maintain the grass dominance and prevent establishment of trees. The land is characterised by maize and wheat production in the northwest and stock farming and subsistence farming in the east and south. The recommended animal stocking rate for the area of Thaba Nchu is 6 ha LSU^{-1} (Department of Agriculture and Rural Development, 2003).

For the savanna ecosystem, the research area is located near Kuruman at the border of the Northern Cape and North-West Province of South Africa, and is situated on the fringe of the Kalahari (latitude 27° - 28° S, longitude 22° - 24° E; altitude 1050-1200 m above sea level). The arid climate in Kuruman receives rainfall mostly in the summer months of October to March with a mean annual rainfall of 355 mm and temperature of 17.5°C, and a very high evaporation level of 2050 mm a^{-1}. The soils are deep Arenosols with aeolian origin, underlain by calcrete (WRB, 2007), typically containing less than 10% clay. The vegetation in the area is dominated by the Kalahari thornveld and shrub bushveld (Tainton, 1999) and has been more specifically described as the Kalahari Mixed Thornveld A16 (Mucina and Rutherford, 2006), characterized by a fairly well developed tree stratum with *Acacia erioloba, Acacia mellifera, Acacia haematoxylon* (≤ 2 m height) and some *Boscia albitrunca* as the dominant trees. The shrub layer is dominated by individuals of *Acacia mellifera, Acacia hebeclada, Lycium hirsutum, Grewia flava* and *Acacia haematoxylon*. The grass cover contained species such as *Eragrostis lehmanniana, Schmidtia kalahariensis* and *Stripagrotis uniplumis*. The majority of the Northern Cape Province is used for stock farming including cattle, sheep or goat farming as well as mining whilst only about 4% is reserved for conservation (Hoffman and Cowling, 1990). In this research area, the estimated grazing capacity of the rangelands are 13 ha LSU^{-1} (Department of Agriculture and Rural Development, 2003). Overgrazing is one of the main causes of land degradation in the Northern Cape, with alien plant invasions posing a threat to the rich flora of the area. This is also one of the worst affected areas in terms of bush encroachment which implies that large areas of grazing land are lost, species diversity is reduced and habitats are transformed (DEAT, 2002). Various land-use activities all contribute to a loss of vegetation cover, soil erosion and ultimately land degradation.

6.2.2 Rangeland management

Communal and commercial livestock ranching are the most common rangeland management systems in both the grassland and savanna ecosystems (Smet and Ward, 2006; Tefera *et al*., 2010). These management systems differ mainly in ownership, as well as managing of grazing resources. The commercial farms are well developed and mainly market-orientated. Commercial farms are typically managed using a rotational grazing system at moderate stocking densities. Size and animal type (sheep, cattle) varied with time, but management structure remained more or less constant.

The communal farms in both these ecosystems belong to the former homeland Bophuthatswana, which was developed as an integral part of the Betterment Villages on land previously owned by commercial farmers (Jacobs, 2003). The communal production systems are based on pastoralism and members of a community share grazing areas. The rangeland is a common pool resource with no restrictions in stocking rates and a continuous grazing system. There are often unclear boundaries, with open access rights to grazing areas. Livestock seem to be of little economic importance, either as income source for people (most income is derived from social grants) or as contributing significantly towards nutrition.

6.2.3 Vegetation and soil sampling

Two rangeland management systems as described above, were selected for this study to compare rotational (commercial farms – CF) and continuous grazing systems (communal farms – CO). In the grassland ecosystem, we sampled four replicates (hence 4 different farms) in every management system, treating the different farms thus as independent replicates. In the savanna ecosystem, we sampled three replicates in every management system.

For each farm a representative grazing gradient was selected, along which the rangeland quality was increasingly degraded with increasing grazing pressure. In the grassland ecosystem the gradients started nearby an artificial water point, regardless whether vegetation was present or not. In the savanna ecosystem the gradients also started nearby an artificial water point, but only where vegetation first appeared. Additionally, composite samples were taken here in direct proximity to the water points, where no vegetation grew (=sacrifice zone). In commercial farms the gradient belonged to one single camp. In both ecosystems these plots were defined exclusively through grass quality conditions, using a similar technique as Van der Westhuizen *et al*. (2005) to identify indicator grass species for the purpose of rangeland condition assessment, independent of bare patches or bush encroachment. Indicator grass species defined on-site by plant experts for the purpose of rangeland condition assessment, were identified. Rangeland condition ranged from poor, moderate (poor and moderate = piosphere) to good conditions (good = vegetation plot outside the piosphere).

In the grassland ecosystem, the dominant grass species that was used as indicator species is *Themeda triandra* for good rangeland condition, *Eragrostis spp.* for moderate rangeland condition, and *Aristida spp.* and *Cynodon dactylon* for poor rangeland condition. The dominant grass species that was used as indicator species in the savanna ecosystem was *Stipagrostis spp*. for good rangeland condition, *Eragrostis spp.* for moderate rangeland condition and *Aristida spp.* and *Schmidtia kalahariensis* for poor rangeland condition (Figure 6.2). *Acacia* species were also dominant in this ecosystem with the area being affected by bush encroachment. Bare patches were common in both ecosystems in poor rangeland conditions, although they also occurred under moderate and good rangeland conditions.

The length of the grazing gradients differed between and within the different management systems. Therefore in the savanna ecosystem the gradients included 6 single plots, each 10 x 10 m in size, with distances between these plots being kept constant within each sampling site. Not all three rangeland conditions were necessarily present in the grazing gradient, therefore a variation in replicates exist.

In each plot, two to three experienced observers independently assessed total plant cover, hereafter called grass cover. The grass cover included all species of the grass layer, e.g. grasses and herbs, as well as dwarf shrubs in the grassland ecosystem. In the savanna ecosystem the area covered by large shrubs and trees (e.g. plants belonging to the tree layer) was not included. Biomass was estimated by taking 10 randomly distributed sword height measurements (0.5 m^2) in two strips (0.5 m wide x 10 m long) per plot with a rising plate meter. To calibrate these measurements, the two strips were mowed and the biomass was weighed after drying at 68°C for 48 h.

In the grassland ecosystem, composite soil samples were collected along the grazing gradient using a 50 mm diameter hand auger. Each composite sample comprised of 10 randomly taken subsamples in the poor, moderate and good grazing condtions, that were thoroughly mixed respectively. In the savanna ecosystem, composite soil samples were taken along a centreline in each plot, every 2 m using a 50 mm diameter hand auger, and combined into a composite sample. Soil samples for selected microbiological properties were taken only in the 0-5 cm soil layer. Because of the sensitivity of microbial soil samples, samples were carefully sieved (< 2 mm) in the field, and stored in a portable freezer (approximately -30°C), before being transported to the laboratory and prepared for the various analyses.

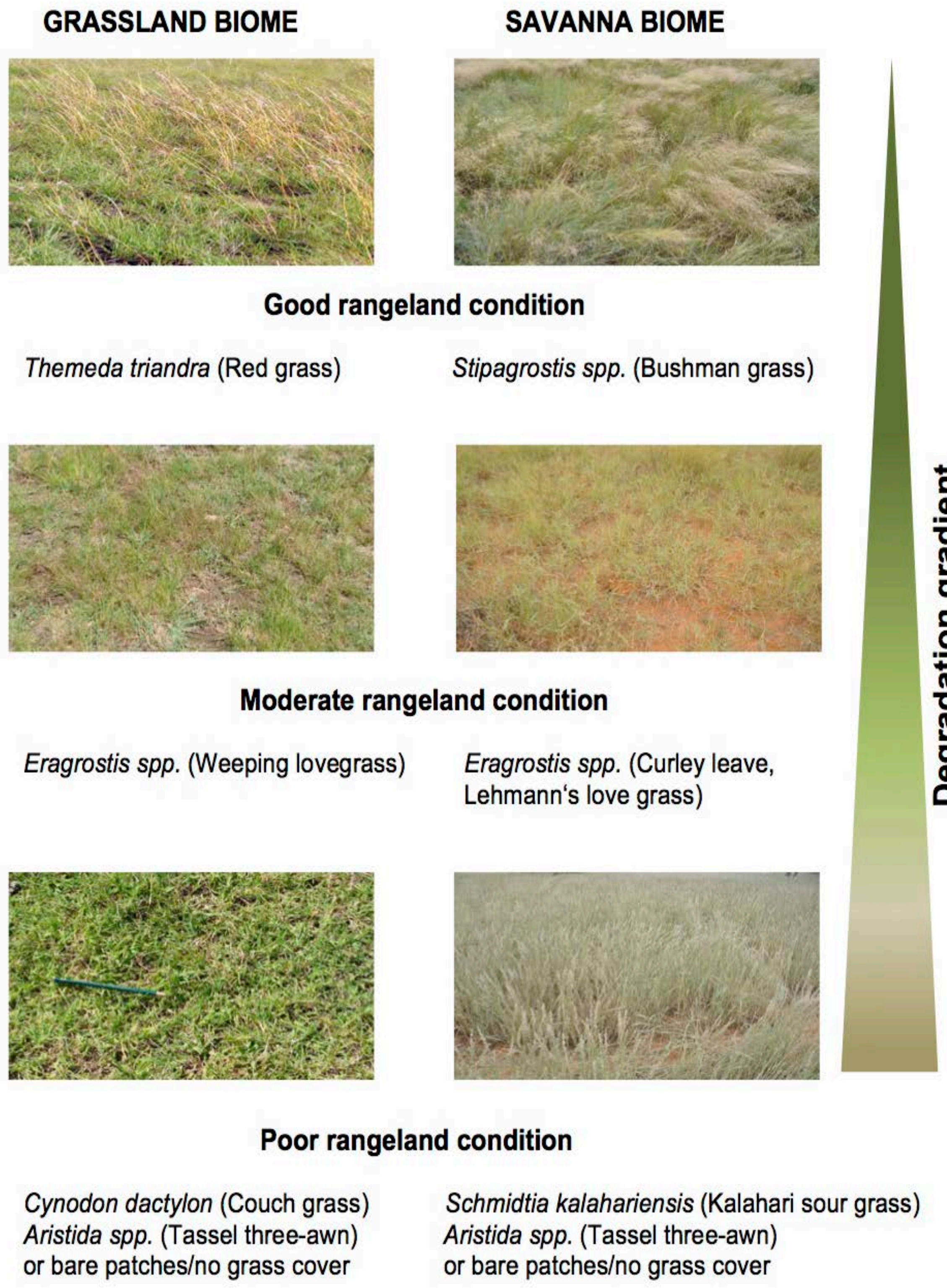


Figure 6.2 Pictures comparing poor, moderate and good rangeland conditions in the grassland (left hand side) and savanna (right hand side) ecosystem, using indicator grass species for the purpose of rangeland condition assessment.

6.2.4 Soil analyses

Unless otherwise specified, soil analyses were done on all samples taken from the rangeland management systems, as well as all rangeland conditions within both ecosystems.

6.2.4.1 Soil physical analyses

Particle size analyses were done by the standard sieve-pipette method (The Non-Affiliated Soil Analysis Work Committee, 1990). Bulk density was determined by gravimetric analysis using 0.1 dm^3 steel cylinder samplers.

Due to the higher clay content and associated aggregation in the grassland ecosystem, soil aggregation was only measured here, and not in the sandy savanna ecosystem (see Section 4.2.4 for a complete description of procedure).

Due to the absence of aggregates - which protects soil organic matter (SOM) - in the sandy savanna ecosystem, particulate organic matter (POM) was investigated to look closer at the fate of organic matter. A complete description of procedure can be found in Section 5.2.4.

6.2.4.2 Soil chemical analyses

All soil chemical analyses were performed in duplicate, and done according to the following standard methods (The Non-Affiliated Soil Analysis Work Committee, 1990): pH (1:2.5 soil to water suspension), exchangeable Ca, Mg, K and Na (1 mol dm^{-3} NH_4OAc at pH 7), extractable Cu, Fe, Mn and Zn (DTPA solution), and CEC (1 mol dm^{-3} NH_4OAc at pH 7). All of the mentioned elements were determined by atomic absorption. Plant-available P was measured using the Olsen method. Total C and N were determined by dry combustion using a CHNS analyser (Elementar-Analysensysteme GmbH, Hanau, Germany). There was no detectable inorganic C; therefore total C was equal to organic C, further called SOC.

6.2.4.3 Soil microbiological analyses

Specific enzymes (β-glucosidase, urease, acid- and alkaline-phosphatase, dehydrogenase) were chosen because of their importance to nutrient cycling in ecosytems; and were determined colorimetrically using enzyme-specific procedures (described below). Once the colour was developed, a spectrophotometer or microplate reader, ELx800 (BioTek Instruments Inc., USA) was used in determining the absorbance at the specific wavelength. Enzyme assays were done in duplicate, with a control for each sample. Dehydrogenase is the only method that required keeping the soil moist for the assay; however, this was done for all the assays for standardization of results (Tabatabai, 1994). Due to this, all data was corrected for water content during the data analysis.

β-glucosidase was determined through an adaption of Dick *et al.* (1996). Briefly, 1 g of field-moist soil was incubated at 37°C for 1 h with toluene, modified universal buffer pH 6.0, and ρ-nitrophenol-β-D-glucosidase (ρNG) then shaken with calcium chloride and tris(hydroxymethyl) aminomethane before filtering through a Whatman no. 2v filter paper. Absorbance of released ρ-nitrophenol (ρNP) was tested with a microplate reader at 405 nm.

Urease was determined by the non-buffered method described by Kandeler and Gerber (1988). Firstly, 5 g of field-moist soil was incubated at 37°C for 2 h with an urea solution and then shaken with potassium chloride. The resulting suspension was filtered through Whatman no. 2v filter paper. Before the absorbance was measured with a microplate reader at 690 nm, the filtrate was prepared with sodium salicylate/sodium hydroxide solution and sodium dichloroisocyanide solution for the colour development.

Both acid- and alkaline-phosphatase enzymes was determined by an assay that was adapted from Tabatabai (1994). Briefly, 1 g of field-moist soil was incubated at 37°C for 1 h with toluene, modified universal buffer (pH 6.5 for acid phosphatase and 11 for alkaline phosphatase), and ρNP. Then, after adding calcium chloride and sodium hydroxide, it was filtered immediately through Whatman no. 2v filter paper. Absorbance, also of ρNP, was measured with a microplate reader at 405 nm.

The assay from Von Mersi and Schinner (1991) was used to determine dehydrogenase, where 1 g of field-moist soil was mixed with THAM and iodonitrotetrazolium violet-formazan (INT) solution. This enzyme assay required incubation at 40°C in the dark for 2 h. The samples were mixed with an extraction solution and kept in the dark for another 30 min. Absorbance of the reaction product INT was tested with a glass cuvette on the spectrophotometer at 464 nm.

A modified version of Marschner (2007) was used to determine PLFA. Briefly, 2 g of frozen field-moist soil was mixed with a citrate buffer, chloroform, methanol, and Bligh and Dyer reagent. The samples were shaken for 2 h, then vortexed and centrifuged at 2500 rpm. The soil-free supernatant was moved to a new tube and the soil was washed with more Bligh and Dyer reagent and once it was vortexed and centrifuged at 2500 rpm again, more supernatant was transferred to the new tube. Chloroform and citrate buffer were added and then the organic phase was removed. The samples were dried under a N_2 stream, conditioned with chloroform, and run through an elution chamber. The phospholipids were collected and dried again under a N_2 stream. An internal standard was added along with methanol:toluol, hexane:chloroform, acetic acid, and deionized water. The organic phase was collected and dried under a N_2 stream. Then the purified PLFAs, which had been methanolyzed into fatty acid methyl esters, were read on a Varian 430-GC gas chromatograph (Agilent Technologies, USA). The identification of individual PLFA markers was performed as described by Reichel *et al*. (2013). A total of thirty-two different PLFAs including saturated, monounsaturated, polyunsaturated, cyclopropyl, and methyl fatty acids were identified. Twelve PLFAs (14:0, i15:0, a15:0, 15:0, 2OH14:0, i16:0, 16:0, i17:0, 17:0, 18:2ω9,12, 18:1ω9c, 18:0) consistently present in the samples were used for data analysis. The fatty acid signatures 14:0, i15:0, a15:0, 15:0, 2OH14:0, i16:0, i17:0, 17:0 and 18:0, which are considered to be of bacterial origin, were used as biomarkers for bacterial biomass. The fatty acids 18:1ω9c and 18:2ω9,12 were used as an indicator for fungal biomass (Frostegard and

Baath, 1996). We used fatty acids i15:0, a15:0, i16:0, and i17:0 to represent Gram-positive bacteria ($Gram^+$), whereas 14:0, 15:0, 2OH14:0 and 17:0 were used to represent Gram-negative bacteria ($Gram^-$) (Djukic *et al*., 2010). The microbial biomass was indicated by total concentration of all PLFA markers (Total PLFA). PLFA-derived ratios of $Gram^+/Gram^-$ and fungi/bacteria were calculated using summed marker concentrations for each microbial group.

6.2.5 Statistical analyses

Statistical analyses were performed on all measured soil properties using a two-way analyses of variance in order to determine statistically significant differences between as well as within the grassland and savanna ecosystems. Thereafter, specific statistical differences were investigated between the main experimental factors: rangeland management systems and grazing conditions. For comparison of means we used a post-hoc Tukey-HSD test with a $p < 0.05$ level of significance. To explore variation in soil microbial community composition among study sites, the mole percentages (nmol%) of twelve individual PLFAs were subjected to principal component analysis (PCA) after standardizing to unit variance. All data were tested for normal distribution as well as homogeneity before statistical analyses were performed using Genstat 16 (NAG Ltd. Oxford, United Kingdom) and SPSS version 12.0 (SPSS Inc. Chicago, USA).

6.3 Results

Results for soil physical and chemical properties measured in the grassland and savanna ecosystems are summarized in Table 6.1 since they were already reported in Kotzé *et al*. (2013) and Sandhage-Hofmann *et al*. (2015), respectively. These soil properties differed significantly ($p < 0.001$) between the two ecosystems, except for pH ($p = 0.477$). The grassland ecosystem on average had higher values than the savanna ecosystem. These variances could be attributed to three distinctive features in the ecosystems: a difference in climate, type of vegetation as well as soil texture.

Table 6.1 Average measured parameters (vegetation and soil), comparing the grassland and savanna ecosystems (SE in brackets)

Ecosystem	**Grassland**	**Savanna**
Climate	Semi-arid	Arid
Grass cover (%)	62.16 (±4.54)	33.02 (±4.97)
Biomass (g m^{-2})	82.49 (±12.38)	95.58 (±13.57)
Particle size distribution (%)		
- Sand	46.01 (±1.29)	96.39 (±2.99)
- Silt	24.45 (±2.18)	1.29 (±0.13)
- Clay	27.94 (±1.64)	2.88 (±0.19)
Bulk density (g cm^{-3})	1.35 (±0.01)	1.55 (±0.02)
pH (H_2O)	6.49 (±0.14)	6.66 (±0.16)
Exchangeable cations (kg ha^{-1})		
- K	224.68 (±15.49)	47.05 (±3.59)
- Ca	1.24 (±0.12)	0.23 (±0.03)
- Mg	337.14 (±24.56)	29.33 (±2.36)
- Na	30.02 (±2.86)	2.61 (±0.28)
CEC (kmol ha^{-1})	116.45 (±7.99)	14.66 (±1.11)
Extractable nutrients (kg ha^{-1})		
- P	6.56 (±1.08)	2.42 (±0.44)
- Cu	1.14 (±0.05)	0.14 (±0.01)
- Fe	34.24 (±5.22)	5.07 (±0.85)
- Mn	16.86 (±1.85)	4.62 (±0.56)
- Zn	0.85 (±0.12)	0.31 (±0.05)
Organic matter indices		
- C (ton ha^{-1})	12.17 (±1.02)	1.27 (±0.12)
- N (ton ha^{-1})	0.96 (±0.09)	0.22 (±0.02)
- C/N	12.68 (±0.43)	5.77 (±0.21)

* Stocks calculated for 0-5 cm depth

Thus, the focus here will be on all measured soil microbiological and some associated properties, which were sampled at 0-5 cm depth, except for the aggregate-size distribution, which were sampled at 0-10 cm depth. However, to get a better perspective on how these soil properties responded to rangeland management systems, knowledge of the grass cover and biomass is essential.

6.3.1 Grass cover and biomass

In the grassland ecosystem, rotational grazing in CF had significantly higher grass cover and biomass than continuous grazing in CO (grass cover: $F(2,12) = 10.521$, $p = 0.007$, biomass:

F(2,12) = 15.082, p = 0.002) (Table 6.2). Also in the savanna ecosystem, the amount of biomass in rotational grazing systems in CF was significantly higher than in the continuous grazing system in CO: $F(2,9) = 33.660$, $p < 0.001$. Total biomass also significantly increased from poor, to moderate and good grazing conditions in this ecosystem: $F(2,9) = 25.546$, $p < 0.001$, i.e., with increasing distance from the water point (Table 6.2). Interaction between biomass and grass cover was not significant.

Table 6.2 Grass cover and biomass in the grassland (n = 18) and savanna (n = 15) ecosystems, comparing the rotational (CF) and continuous (CO) grazing systems in the poor, moderate and good rangeland conditions (SE in brackets; means followed by different letters either in a column or row, are significantly different).

	Grassland ecosystem				Savanna ecosystem			
	Poor	**Moderate**	**Good**	**Ave**	**Poor**	**Moderate**	**Good**	**Ave**
Grass cover (%)								
CF	70.7 (±13.5)	71.0 (±7.3)	73.2 (±4.0)	71.6 a (±4.8)	27.0 (±7.1)	40.5 (±10.1)	47.0 (±4.0)	38.2 (±4.8)
CO	21.0 (±4.0)	57.5 (±15.5)	51.1 (±8.9)	43.2 b (±8.5)	17.7 (±14.4)	25.0 (±2.1)	33.3 (±8.3)	25.3 (±5.2)
AVE	45.9 (±13.6)	64.3 (±6.7)	62.1 (±5.8)		22.3 (±6.4)	32.7 (±6.7)	40.2 (±4.8)	
Biomass (g m^{-2})								
CF	87.2 (±17.8)	90.6 (±11.9)	129.3 (±20.3)	102.4 a (±10.6)	54.9 (±1.6)	139.4 (±25.8)	177.8 (±7.5)	124.0 a (±19.7)
CO	25.1 (±17.1)	46.9 (±8.7)	56.3 (±6.9)	42.8 b (±7.9)	5.2 (±4.2)	60.2 (±5.9)	93.3 (±14.5)	52.9 b (±16.8)
AVE	56.1 (±17.8)	68.8 (±12.1)	92.8 (±20.1)		30.1 a (±12.3	99.8 b (±24.1)	135.6 c (±21.6)	

*Data kindly provided by R. Oomen. Institute of Crop Science and Resource Conservation, University of Bonn, Katzenburgweg 5, D-53115 Bonn, Germany.

6.3.2. Organic carbon and total nitrogen

Organic C and total N were measured as indices of organic matter, and were described in Kotzé *et al*. (2013) and Sandhage-Hofmann *et al*. (2015). As could be expected these two properties had a strong correlation (r = 0.97), and therefore only organic C and the C/N ratio will be reported here. However, it is important to note that there was about a ten-fold difference on average for organic C as well as total N values measured between the two ecosystems, with the clayey grassland ecosystem showing the highest values (Table 6.1).

In the grassland ecosystem there was not a significant interaction between rangeland management systems and grazing conditions, $F(2,16) = 2.262$, $p = 0.136$. There were however statistically significant main effects for rangeland management systems, where organic C were significantly different between CF and CO [$F(1,16) = 10.437$, $p = 0.005$]. In this ecosystem CF had the lowest value in the moderate grazing condition (12.8 ton C ha^{-1}), followed by the good rangeland condition (14.2 ton C ha^{-1}), and the highest value in the poor grazing condition (15.9 ton C ha^{-1}). For CO the poor grazing condition always had the lowest value (7.7 ton C ha^{-1}), whereas comparable values evolved for moderate (10.9 ton C ha^{-1}) and good grazing conditions (11.6 ton C ha^{-1}).

In the sandy savanna ecosystem no significant interactions realized between rangeland management systems and grazing conditions [$F(2,12) = 0.617$, $p = 0.556$], however some interesting trends were observed. The rotational grazing system followed the same trend as in the grassland ecosystem, with the lowest C stocks also measured in the moderate grazing condition (1.1 ton C ha^{-1}), followed by the good rangeland condition (1.6 ton C ha^{-1}), and the highest value in the poor grazing condition (1.7 ton C ha^{-1}). However, the opposite was true for the continuous grazing system showing the highest values in the moderate grazing condition (1.3 ton C ha^{-1}), followed by the good grazing condition (1.1 ton C ha^{-1}), and the lowest value in the poor grazing condition (1.0 ton C ha^{-1}).

As depicted in Figure 6.3 the C/N ratio in the grassland ecosystem (average C/N = 12.7) showed similar trends for both rangeland management systems, with a slightly increasing C/N ratio as we progress from poor to good grazing condition, however there was not a significant interaction between rangeland management systems and grazing conditions [$F(2.16) = 0.261$, $p = 0.773$]. In contrast, the savanna ecosystem (average C/N = 5.8) had a decreasing C/N ratio from poor to good grazing condition in CF, and a higher C/N ratio in the moderate grazing condition compared to the poor or good grazing conditions of CO. No significant interactions were evident between rangeland management systems and grazing conditions [F(2,12) = 2.229, p = 0.150].

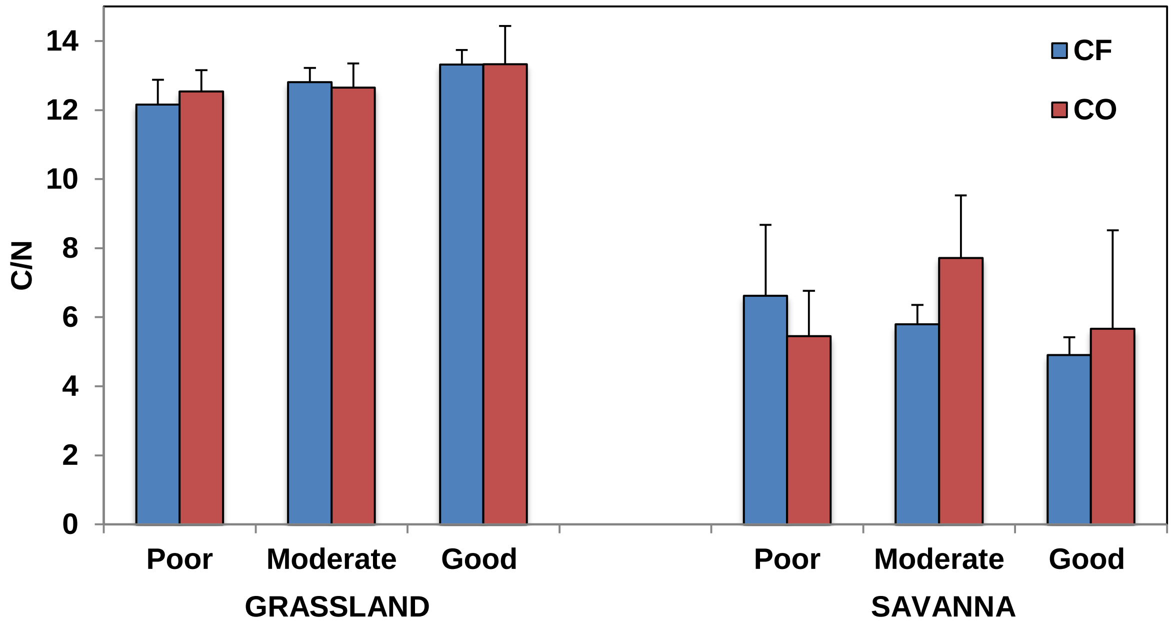


Figure 6.3 The C/N ratio for the grassland and savanna ecosystems, comparing the rotational (CF) and continuous (CO) grazing systems in the poor, moderate and good rangeland conditions.

6.3.3 Aggregate-size distribution, and associated carbon and nitrogen

Aggregate-size distribution was only measured in the grassland ecosystem, due to missing aggregation in the savanna ecosystem. The rangeland management systems varied in the distribution of their aggregate size fractions, with the rotational grazing system (CF) having higher proportions of macroaggregates compared to the continuous grazing system (CO). On average, after sand correction, large macroaggregates (> 2000 μm) accounted for the greatest part (89.2%) of all sampled sites, More macroaggregates were found in the poor and moderate rangeland conditions for CF, compared to those of CO, which were less aggregated and therefore accounted for lower quantities of macroaggregates and larger quantities of microaggregates (Figure 6.4). For more detailed results, see Section 4.3.4 (Kotze *et al.,* 2013).

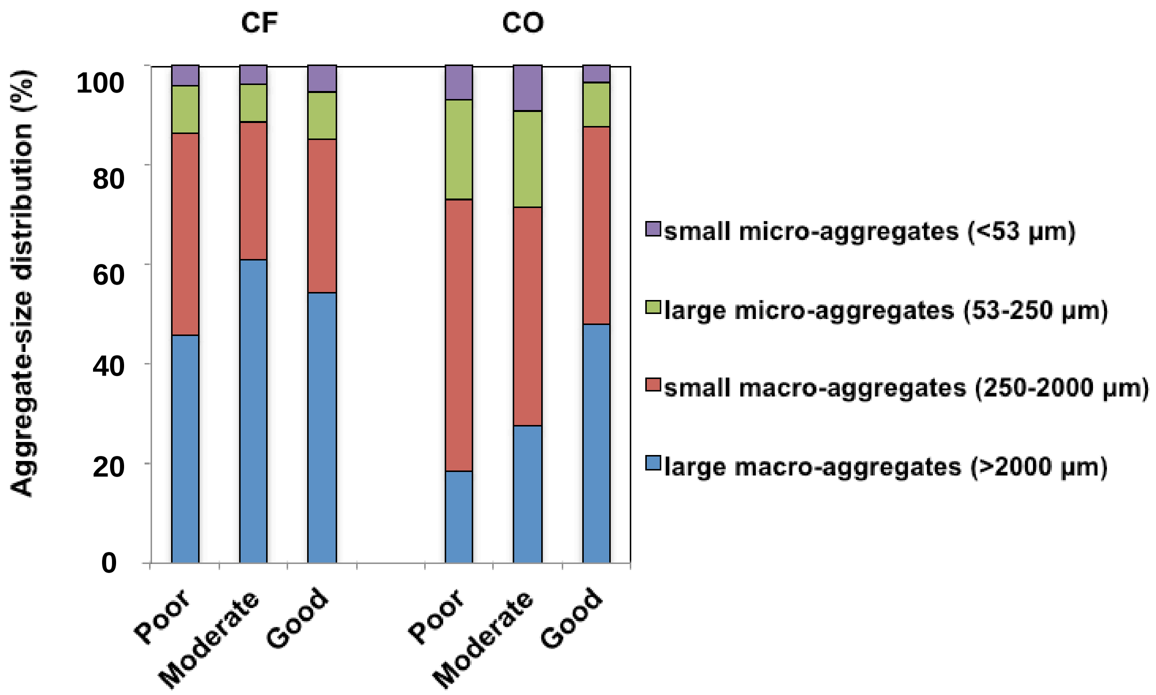


Figure 6.4 Aggregate-size distribution in poor, moderate and good rangeland conditions for rotational grazing (CF) and continuous grazing (CO) systems in the grassland ecosystem.

For both the rotational and continuous grazing systems, most of the soil C stock was located in the macroaggregates (76-88%), followed by the small microaggregates (6-12%), with the distribution of N closely following the same trend as that of C (data not shown). As displayed in Figure 6.5, the lowest C contents in the macroaggregates, were in the poor rangeland conditions, with CO being the lowest. The loss of C correlated with the loss of macroaggregates and the organic matter stored therein. Therefore, combining aggregate fractionation with C analyses sensitively reflected changes in soil properties as induced by different grazing systems.

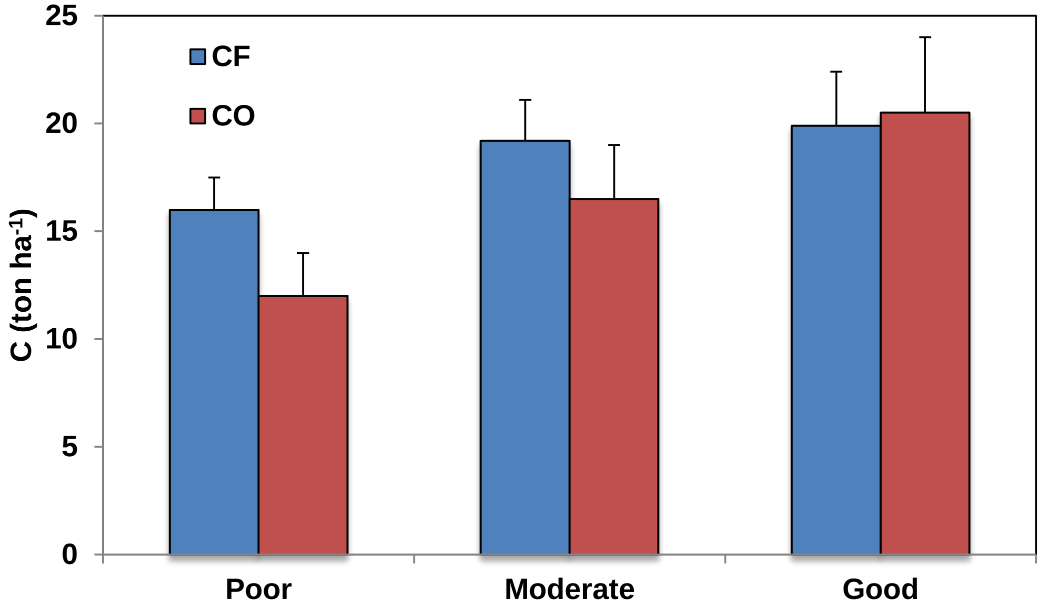


Figure 6.5 Carbon stocks in large macroaggregates comparing rotational (CF) and continuous (CO) grazing systems in poor, moderate and good rangeland conditions for the grassland ecosystem.

6.3.4 Particulate organic matter

Particulate organic matter was only measured in the savanna ecosystem due to the absence of aggregates, which protects organic matter. This parameter was chosen to get better insight in the fate of organic matter in a sandy environment. The stocks of the three different POM fractions and the mineral associated fraction (MOM) reflected differences in total SOC. Most SOC was found in the MOM fraction representing about 46% of total SOC, while least SOC was found in the fPOM fraction, however there was not any significant interactions between rangeland management systems and grazing conditions for the SOC found in any of the POM fractions, except for cPOM [$F(2,12) = 5.049$, $p = 0.026$] with Tukey-HSD = 1.97. In rotational grazing systems (CF) in the poor rangeland condition, higher SOC and N stocks were present in all isolated POM fractions including MOM relative to the continuous grazing systems (CO). With rising distance to the water point and under moderate rangeland conditions, SOC stocks in POM fractions of CO exceeded those of CF, in particular for cPOM (Figure 6.6). Only outside the piosphere, under good rangeland condition, both farm types exhibited similar POM-C contents. The N stocks in the POM fractions followed the same trend as SOC, but differences were most pronounced for MOM in moderate and good rangeland condition with higher N stocks in CO. Results are also described in Section 5.3.2.

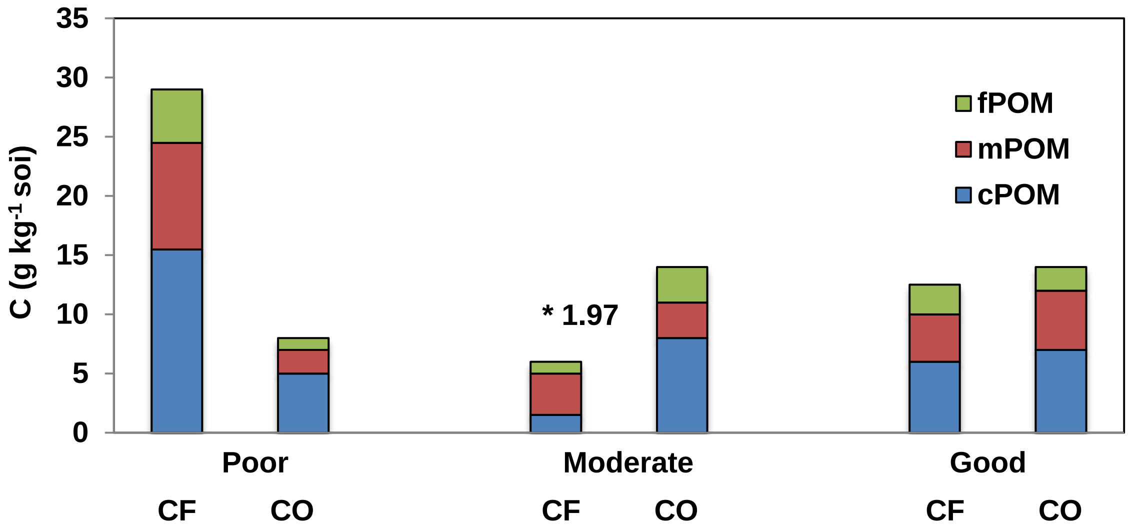


Figure 6.6 Carbon concentrations in fine (fPOM), intermediate (mPOM) and coarse (cPOM) fractions of particulate organic matter, under poor, moderate and good rangeland conditions of commercial (CF) and communal (CO) farms. Significant interaction indicated with *.

6.3.5 Soil enzyme activity

All soil microbial enzyme activities measured were significant higher for the grassland ecosystem compared to the savanna ecosystem (note the difference in scaling on the y-axis), however varied trends evolved for the different rangeland management systems within these ecosystems (Figure 6.7). The activity of the β-glucosidase enzyme showed opposite trends in the respective grazing gradients when comparing the rangeland management systems in the grassland ecosystem, with significant differences (Tukey-HSD = 2803) in both main effects as well as the interaction term between rangeland management systems and grazing conditions, [$F(2,16) = 5.932$, $p = 0.011$]. In the grassland ecosystem the poor grazing condition had the highest activity in the rotational grazing system (CF) and the activity declined towards the good grazing condition, whereas the trend was reversed in the continuous grazing system (CO) (Figure 6.7). Similarly, in the grassland ecosystem, the activity was highest under poor grazing condition in CF, but lowest in the respective poor grazing conditions of CO. In both rangeland management systems of the savanna ecosystem the β-glucosidase activity showed similar trends like the C/N ratio, although not significant (Figure 6.3). Also interesting to note is that in both ecosystems the rotational grazing systems showed similar trends, with the poor grazing condition having the highest activities, followed by the moderate grazing condition and the good grazing condition having the lowest activities.

The activity of the urease enzyme showed no significant interactions in the grassland ecosystem [$F(2,18) = 1.080$, $p = 0.360$], as well as the savanna ecosystem [$F(2,12) = 0.961$, $p = 0.410$]. For CF, urease behaved similarly along the grazing gradient as the β-glucosidase activity (see above), with highest activity under poor conditions. However, for urease, the patterns in the grazing gradients were now also more or less similar in CO, with again highest urease activity in the poor grazing condition, followed by the moderate grazing condition and the lowest activities in the good grazing condition. The urease activity of CO exceeded activities of CF, with the exception of the poor grazing conditions (Figure 6.7).

The activity of the dehydrogenase enzyme again followed similar trends as the β-glucosidase enzyme, with decreasing activities from the poor to the good grazing condition in both CF and CO of the grassland ecosystem. An exception was for the continuous grazing system in the savanna ecosystem. Here the β-glucosidase activity had not differentiated between the grazing conditions, where the dehydrogenase activity also changed in similar direction and declined from good to poor grazing conditions as in CF (Figure 6.7). Thus, with dehydrogenase there was also a significant interaction evident (Tukey-HSD = 232) in the grassland ecosystem in the poor grazing condition, when comparing CF and CO [$F(2,18) = 5.781$, $p = 0.012$]. Dehydrogenase activity was also significantly different when comparing CF and CO [$F(1,18) = 5.570$, $p = 0.03$].

There were no significant interactions between rangeland management systems and grazing conditions for alkaline-phosphatase activities in the grassland [$F(2,18) = 0.376$, $p = 0.692$], or the savanna ecosystem [$F(2,12) = 0.007$, $p = 0.993$]. However, in the grassland ecosystem, significantly higher activity were evident in CF (18907) compared to CO (4846) throughout the grazing gradients for acid-phosphatase [$F(1,18) = 10.921$, $p = 0.004$], as well as for alkaline-phosphatase [$F(1,18) = 9.572$, $p = 0.006$], with CF having a higher activity of 1612 compared to CO with an activity of 540. No clear trend was visible when comparing the poor, moderate and good grazing conditions. For the savanna ecosystem these two enzymes showed no clear pattern or similarities. There was, however, a trend in the grazing gradient for alkaline-phosphatase, where the poor grazing condition showed the highest activity for both CF (3100) and CO (2914) compared to the lower moderate grazing condition, with CF having an activity of 566 and CO an activity of 552. A significant interaction evolved in this savanna ecosystem for acid-phosphatase activity [$F(2,12) = 10.046$, $p = 0.003$], with CF having a decreasing trend and CO having an increasing trend from poor to good grazing condition, with the lowest acid-phosphatase activity (1761) in the poor grazing condition of CO.

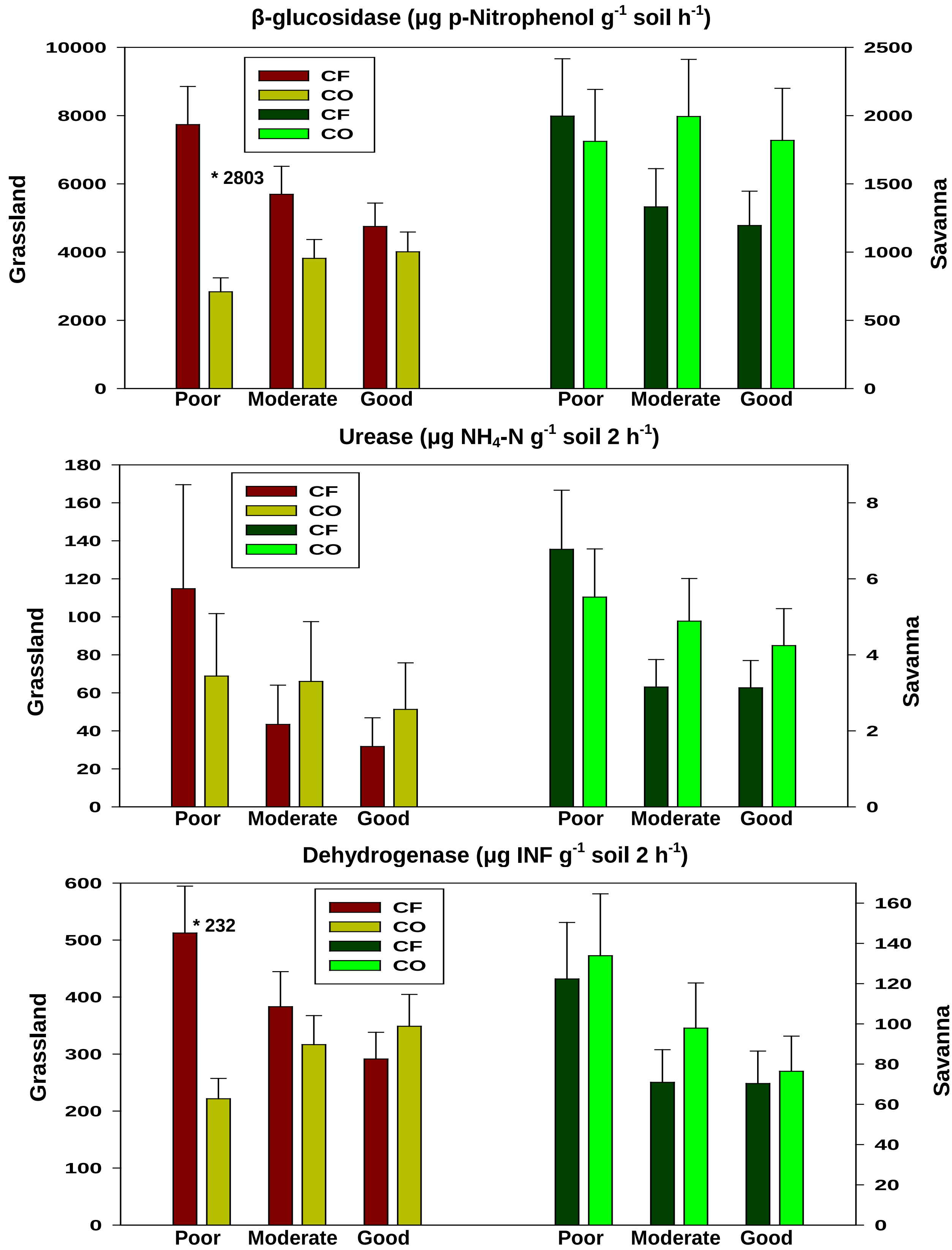


Figure 6.7 A comparison of three soil microbial enzyme activities (β-glucosidase, urease and dehydrogenase) in the grassland (left y-axis) and savanna ecosystems (right y-axis) between the rotational (CF) and continuous (CO) grazing systems within poor, moderate and good rangeland conditions. Significant interactions indicated with *.

6.3.6 Phospholipid fatty acids

Total microbial lipid biomass (PLFA) had significantly higher values in the topsoils of the grassland ecosystem compared to the soils of the savanna ecosystem. Patterns were similar to β-glucosidase and dehydrogenase in the grassland ecosystem, which, similar to the amount of total PLFA, are also proxies for living microbial biomass. There was not a significant interaction between rangeland management systems and grazing gradients, $F(2,18) = 2.572$, $p = 0.104$. However, a significant difference [$F(1,18) = 11.318$, $p = 0.003$] was found when comparing the rangeland management systems showing opposite trends in the grassland ecosystem. For the savanna ecosystem, similar trends between both rangeland systems could be anticipated, except for the good grazing condition of CF, which showed exceptionally high PLFA contents while a similar peak in enzyme activities was lacking. On the other hand, the differences in PLFA were not significant, due to high standard deviations (Figure 6.8).

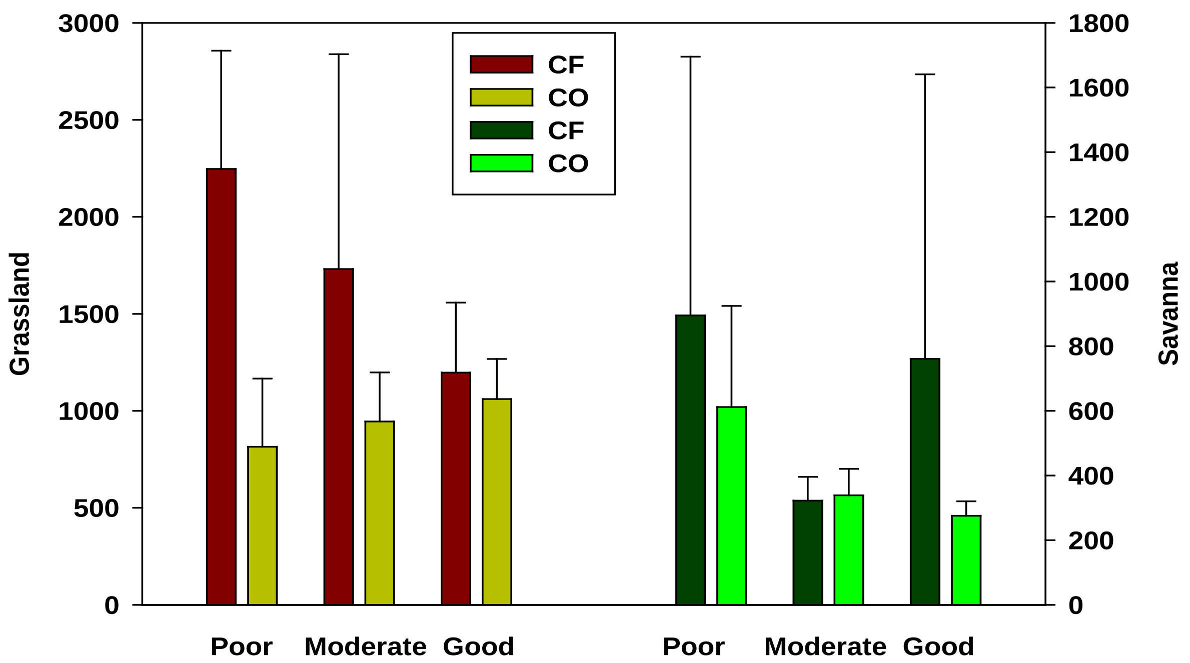


Figure 6.8 Total PLFA (nmol PLFA g^{-1} soil) measured in the grassland and savanna ecosystems, comparing rotational (CF) and continuous (CO) grazing systems within poor, moderate and good rangeland conditions.

Remarkably, the two ecosystems responded contrasting in terms of significant differences evident in the individual PLFAs, where the grassland ecosystem had the most significant differences when comparing CF and CO (see below). In addition, significant interactions were found between rangeland management systems and grazing conditions. On the other hand, the savanna ecosystem had the most significant differences when comparing the grazing conditions (poor, moderate and good) within the rangeland management systems, and not when comparing CF and CO. No significant interactions were thus found in this ecosystem. From a total of thirty-two different PLFAs identified, only twelve PLFAs (14:0, i15:0, a15:0, 15:0, 2OH14:0, i16:0, 16:0, i17:0, 17:0, 18:2ω9, 12, 18:1ω9c, 18:0) were consistently present in the samples. They were thus chosen for principal component analysis. This analysis confirmed substantial differences in soil microbial community composition among study sites (Figure 6.9).

For the grassland ecosystem the first principal component (PC1) explained 66.9% and the second (PC2) 19.8% of the total variance in the PLFAs (Figure 6.9a and c). The PCA plot in Figure 6.9a also showed that all three grazing conditions (poor, moderate, good) are grouped together for CF as well as for CO, i.e. it appears possible to structure the data according to rangeland system. Also interesting to note is that for PC2 the poor grazing condition for both CF and CO had positive loading scores, compared to the other grazing conditions which had negative loading scores. The twelve fatty acids chosen for data analysis also showed clear groupings. For PC1, lipid signatures i15:0, a15:0, 15:0, 2OH14:0, i16:0, 17:0, 18:2ω9,12 and 18:1ω9c had higher positive loading scores while 16:0 and 18:0 had lower negative loading scores (Figure 6.9c). Both fungal biomarkers 18:2ω9,12 and 18:1ω9c had positive loading scores for this ecosystem.

For the savanna ecosystem PC1 explained 54.1% and PC2 30.5% of the total variance in the PLFAs (Figure 6.9b and d). The PC1 plot in Figure 6.9b had a high positive loading score only for the poor grazing condition in CO, whereas all the other sites had lower negative loading scores. For PC2 the poor grazing condition for CF showed a high positive loading score, whereas the moderate grazing condition for CF showed an opposite high negative loading score. Hence, the PCA separates the rangeland management systems less clearly than in the grassland ecosystem, only the poor grazing condition has other factor loadings than the other two grazing conditions. For the savanna ecosystem, the fatty acids shown in Figure 6.9d loads the fungal biomarkers 18:2ω9,12 and 18:1ω9c in the negative score for PC1 and in the positive score for PC2. The common bacteria signatures 16:0 and 18:0 both had positive loading scores for PC1, and negative scores for PC2. On the other hand, the Gram$^+$ bacteria (i15:0, a15:0, i16:0 and i17:0) do not show any clear groupings for the savanna ecosystem in this PCA, while the Gram$^-$ bacteria (14:0, 15:0 and 17:0) grouped together in the top right hand quadrant of the PCA.

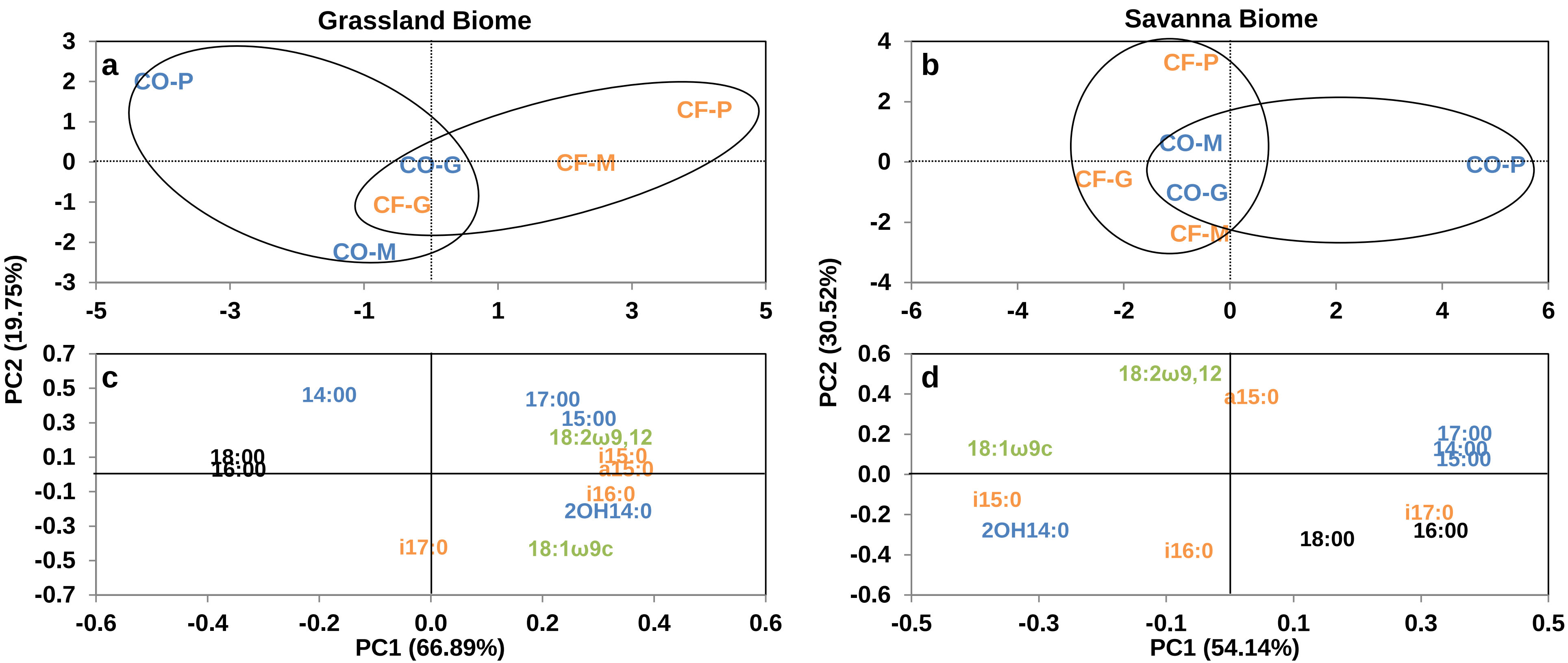


Figure 6.9 Principle component analysis (PCA) dividing the PLFA analyses for the grassland (a and c) and savanna (b and d) ecosystems. For the grassland ecosystem PC1 (66.89%) and PC2 (19.75%) explained 86.64% of the total variance in the data. For the savanna ecosystem PC1 (54.14%) and PC2 (30.52%) accounted for 84.66% of the total variance in the phospholipid fatty acids. In a and b, rotational grazing systems are shown in orange and continuous grazing systems in blue. In c and d, the fungal fatty acid signatures (18:1ω9c and 18:2ω9,12) are indicated in green, Gram$^+$ bacteria (i15:0, a15:0, i16:0, and i17:0) in orange, Gram$^-$ (14:0, 15:0, 2OH14:0 and 17:0) in blue, and common bacteria (16:00 and 18:00) in black.

The absolute abundance of specific microbial groups showed different patterns when comparing the grassland and savanna ecosystems. As displayed in Figure 6.10, all the bacterial, fungal, $Gram^+$ and $Gram^-$ PLFAs showed similar changes along the grazing gradients in the grassland ecosystem, with CF always having higher concentrations than CO. These trends largely explained, therefore, also the changes of bulk PLFA contents as displayed in Figure 6.8. Interestingly, CF always had the highest concentrations of these PFLAs in the poor grazing condition and the lowest in the good grazing condition of the grassland ecosystem, whereas CO had the opposite trend with the lowest concentrations in the poor grazing condition and the highest in the good rangeland condition. Due to the similarity of the behavior of individual PLFAs, the ratios of fungal/bacterial PLFA as well as G^+/G^- ratios showed similar trends for both CF and CO, with not much variance between the ratios. Although not significant, the moderate grazing condition always tended to have the higher ratio for both these indices, irrespective of rangeland management. All microbial groups and ratios did thus also not reflect any significant interactions between rangeland management systems and grazing conditions, except for $Gram^-$, $F(2,18) = 5.768$, $p = 0.012$. There were however statistically significant main effects for rangeland management systems, with clear differences between CF and CO for all the microbial groups, viz. bacteria: $F(1,18) = 8.628$, $p = 0.009$; fungi: $F(1,18) = 12.176$, $p = 0.003$; $Gram^+$: $F(1,18) = 8.621$, $p = 0.009$; $Gram^-$: $F(1,18) = 18.573$, $p < 0.001$; fungi/bacteria: $F(1,18) = 10.219$, $p = 0.005$.

For the savanna ecosystem the absolute abundance of the different microbial groups did not show clear trends or patterns, except that the moderate grazing condition for both CF and CO seemed to have lower concentrations for bacterial, fungal and $Gram^+$ PLFAs, when compared to the poor and good grazing conditions. These minima, however, are not significant due to the large standard deviation during analyses of these sandy sites. The fungi/bacteria ratio for the savanna ecosystem seemed to be influenced differently when comparing CF and CO. This ratio for CF had the lowest value for the moderate grazing condition, whereas CO had the highest value for the moderate grazing condition (note that in CO bush encroachment was most pronounced in the moderate veld). The $Gram^+/Gram^-$ ratio

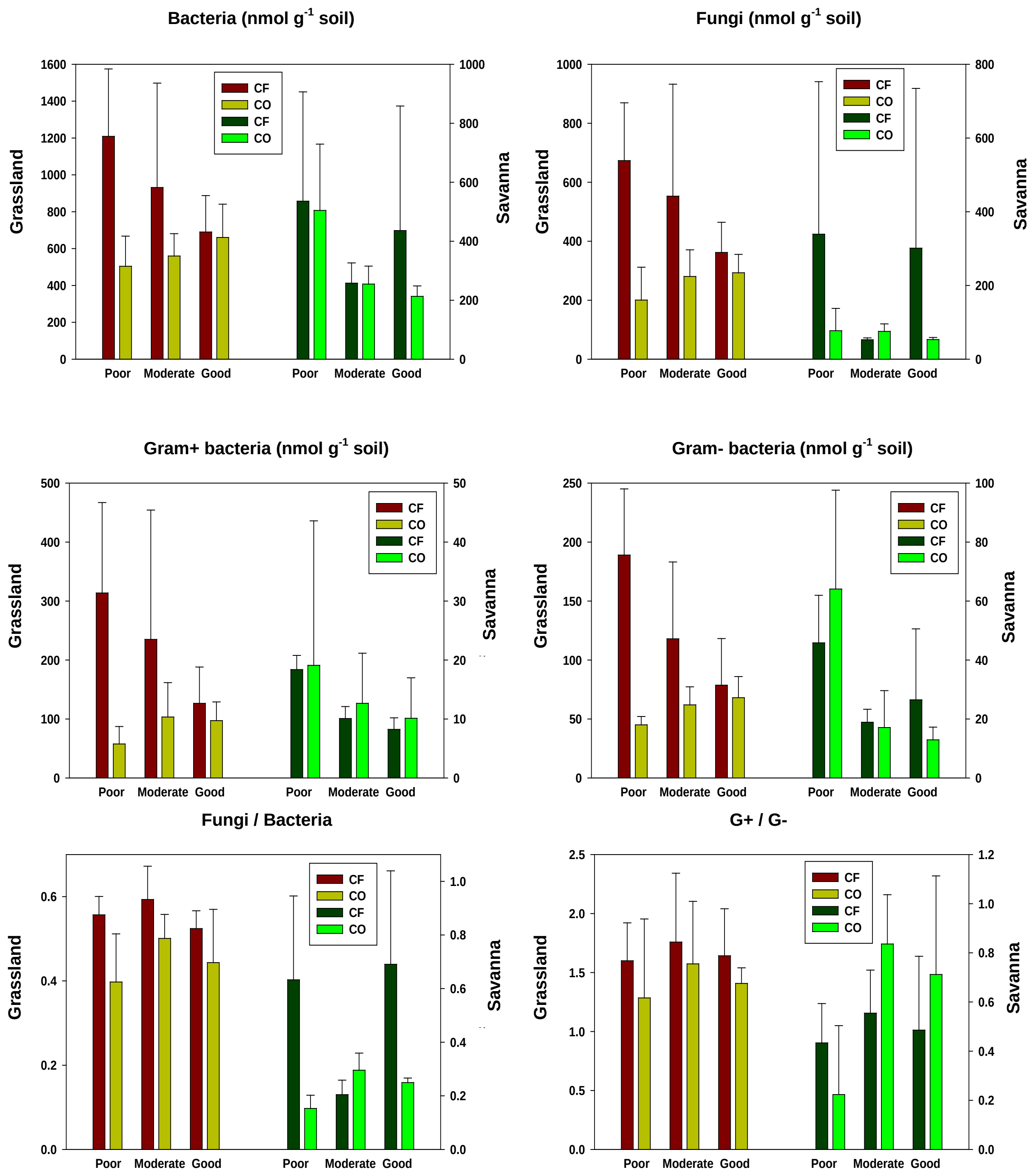


Figure 6.10 Sums and ratios of phospholipid fatty acids (PLFAs) of microbial groups measured in the grassland and savanna ecosystems, comparing rotational (CF) and continuous (CO) grazing systems within poor, moderate and good rangeland conditions.

also tended to have the highest value for the moderate grazing condition for both CF and CO. No significant interactions between rangeland management system and grazing condition existed in this ecosystem for any of the microbial groups or ratios. The only significant main effect manifested for Gram^{-} bacteria in the grazing condition [$F(2,12) = 7.261$, $p = 0.009$], where the poor grazing condition in CO had the most Gram^{-} bacteria.

6.4 Discussion

6.4.1 Grass cover and biomass as well as related changes in basic soil properties

In any rangeland management system, the composition and structure of grass cover and biomass are ultimately influenced by the animals grazing on the land. In both ecosystems the rotational grazing systems had higher values for both grass cover and biomass, compared to the continuous grazing systems, indicating that enough resting time is essential for grass to recover from grazing. This corresponds with results from Tessema *et al.* (2011) and Banerjee *et al*. (2000) who found in continuous grazed pastures reduced vegetation growth, diversity, abundance and biomass compared to rotationally grazed pastures. As could be expected, all grazing gradients in both ecosystems, irrespective of rangeland management system, also showed the highest values for grass cover as well as biomass in the good grazing conditions, where grazing pressure is low, and the lowest values in the poor grazing conditions, where grazing pressure is highest (Rutherford and Powrie, 2011). For the savanna ecosystem the difference between the poor and good grazing conditions were two-fold for grass cover and four times more for biomass, showing that this ecosystem is sensitive for vegetation degradation, especially over a short-term period.

Changes in the contents of soil C and N in the studied grazing gradients, have been described in detail by Kotzé *et al*. (2013) and Sandhage-Hofmann *et al*. (2015). The results are shortly summarized here again since they are relevant for the understanding of soil biological performance. In the grassland ecosystem, both soil C and N contents have been negatively affected by high grazing pressure, accompanied by a depletion of plant cover and litter input; additional trampling and resulting mixing of surface soil with subsoil; or erosion, i.e., the contents declined from good to poor grazing condition in CO, whereas in the poor grazing condition of CF, some enrichment due to urine and dung of the animals was observed (Perkins and Thomas, 1993; Kotzé *et al.*, 2013). The effects are paralleled by changes in the C/N ratio within the grazing gradients for both rangeland management systems. The lower C/N ratio in the poor grazing conditions indicate a higher soil N content, which can be attributed to the concentrated urine and dung in this area due to the water or entry point of the camp. Because the soil C content increases as grass condition move towards the good grazing condition, the C/N ratio also increases. The savanna ecosystem has less rainfall, less grass cover and therefore lower soil C contents than the grassland ecosystem. The trend within a rangeland management system is also dissimilar, because invasion of bush at heavy grazing may enhance C contents, particularly under moderate grazing conditions of the CO management systems (Sandhage-Hoffmann *et al*., 2015). Lower C/N ratios in the continuous grazing systems (CO) compared to the rotational grazing systems (CF) could also be due to the presence of more *Acacia* species, which contributes to higher N fixation in soil (Cramer *et al*., 2007; Isaac *et al*., 2011). The above mentioned changes went along with aggregate breakdown with trampling and soil organic matter loss (grassland ecosystem; Kotzé *et al*., 2013), as well as by accrual of particulate organic matter with enhanced input of plant debris (savanna ecosystem; Sandhage-Hofmann *et al*., 2015) (Detailed results for aggregates can be seen in Section 4.3.4 and for particulate organic matter in Section 5.3.1). All these parameters are well known to co-affect also soil microbial performance. Soil microbial diversity, for instance, varies significantly among soil aggregate classes and particle-size fractions, as well as associated enzyme activities (e.g., Stemmer *et al*., 1998; Kandeler *et al*., 2000; Kihara *et al*., 2012). This must be considered, therefore, when elucidating changes of these parameters with grazing pressure and rangeland management system.

6.4.2 Soil enzyme activity

All the measured microbial enzyme activities were higher in the clayey grassland ecosystem compared to the sandy savanna ecosystem. A clayey soil seems to have properties that favor soil organisms indirectly, such as higher water-holding capacity, increased organic matter levels as well as soil aggregates (Prieto *et al*., 2011). Within the grassland ecosystem all enzymes had elevated values in the poor grazing condition of the rotational grazing systems, whereas the opposite was true for the continuous grazing systems. This could probably again be attributed to the urine and dung of the animals deposited nearer to the centre of the piosphere (Banerjee *et al*., 2000). Under rotational management, there was a resting time of the soil, i.e., the adding of urine and dung nearby the water point went along with a maintenance in vegetation, and as such, also of microbial activity: high enzyme activities were recorded. In contrast, under continuous grazing, the intensive trampling destroyed aggregates (see Section 5.4; Kotzé *et al*., 2013), vegetation, and soil organic matter was lost, likely also by erosion. Hence, even despite enhanced input of urine and dung, microorganisms lack additional C sources and nutrients, and we found lower enzyme activities than under the other grazing conditions.

In the sandy savanna ecosystem, both rangeland management systems had a similar effect on the enzyme activities measured, with the highest values in the poor grazing conditions and comparable values in the moderate and good grazing conditions. These data can again be explained by higher inputs of dung, which also went along with an enrichment of POM (Sandhage-Hoffmann *et al*., 2015). POM fractions are known to host substrates for enlarged microbial and enzyme activities (e.g. Kandeler *et al*., 2000; Moore-Kucera *et al*., 2008; Yan *et al*., 2009), so changes in enzyme activity with grazing coincide with other changes in soil properties. Similar to what Watts *et al*. (2010) found, the dehydrogenase activity, which is a measure of microbial respiration and a reliable index of microbial activity in soil, correlated well with the individual microbial groups (fungi, bacteria, $Gram^+$, $Gram^-$) in both ecosystems.

6.4.3 Phospholipid fatty acids

Total microbial activity as indicated by total PLFA was affected by grazing in a similar manner as organic C, β-glucosidase, dehydrogenase as well as all the various microbial PLFA groups in the clayey grassland ecosystem for both rangeland management systems. Xu *et al*. (2014) found similar results with a study in China, where soil microbial communities also showed a positive relationship with enzyme activities. In the sandy savanna ecosystem, the rotational grazing system had the lowest PLFA values in the moderate grazing condition, corresponding to the reaction of organic C and POM. The bush encroachment in the continuous grazing system did not have any effect on total microbial activity, however it did seem to have an increasing effect on the fungi/bacteria and G^+/G^- bacteria ratios. The higher ratios in the moderate grazing condition indicate that this might be the most productive zone in the grazing gradient. Hence, and as discussed earlier in Section 5.5, the conditions for poor grazing (many bushes in the moderate grazing condition) do not go along with poor soil conditions, but in turn seem to go along with improved soil conditions at advanced stages of rangeland degradation in this ecosystem (Sandhage-Hofmann *et al.*, 2015). Fungi are known to grow on C/N wide substrates (Rousk and Baath, 2007; Strickland and Rousk, 2010) i.e. elevated C/N ratios in the moderate grazing condition, particularly in the continuous grazing systems of CO; Figure 6.3.), support their accrual. Similarly, the abundance of bacteria in all sites also correlates well with a lower C/N, which supports studies done by Bossuyt *et al*. (2001) and Hiltbrunner *et al*. (2012).

The elevated portions of Gram- bacteria in all poor grazing conditions are likely again associated with the better supply of dung-derived organic matter and high nutrient availability from the urine. Hiltbrunner *et al*. (2012) found similar results in a Swiss sub-alpine pasture. Our findings can also be reconciled with Hamer *et al*. (2009) who reported lower proportions of Gram- bacteria in bare soil than under grass cover, as is evident in the poor grazing conditions of particularly the CO farms studied.

The PCA's also showed that substantial differences in soil microbial community composition exist when comparing the grassland and savanna ecosystems. The biggest difference between these two ecosystems was soil texture and climate, which influenced soil water content and subsequently the grass species growing in that specific ecosystem. Both Sessitsch *et al*. (2001) and Xu *et al*. (2014) confirmed in their studies that soil texture has a regulatory role in soil biological processes and thus affected the soil microbial community structure. PLFA analysis exposes the structural characteristics of the living microbial community at the time of sampling and is therefore suitable for sensing rapid changes in microbial communities due to rangeland management. Nevertheless, and despite the differences in ecosystems, soil microbial communities were clearly clustered according to the rangeland management system as well as grazing condition, suggesting that rotational and continuous grazing systems generate different and unique ecological niches in soil microbial communities (Jangrid *et al*., 2011; Vallejo *et al*., 2012), especially in the grassland ecosystem.

6.4.4 Soil quality evaluation

To give an indication of how rangeland management affected soil quality, selected soil microbiological properties were expressed as enzyme or microbial activity per gram soil C or N (β-glucosidase, dehydrogenase, total-PLFA normalized to C; and urease normalized to N) (Table 6.3). In the grassland ecosystem, the trend for both the continuous and rotational grazing systems follow the same decreasing trend, with the highest activities present in the poor grazing conditions and the lowest activities in the good grazing conditions. This corresponds well with the urine and dung deposits close to the water points in the sacrifice area. In the savanna ecosystem, however, there is no clear trend visible. It is nevertheless interesting to note that the lowest enzyme activities for the rotational management system were also in the good grazing conditions. When comparing the two ecosystems, there is more enzyme and microbial activity in the grassland ecosystem compared to the savanna ecosystem, although the amount of enzymes and microbes per g soil C or N is now higher in the savanna ecosystem than in the grassland ecosystem. Trends noticeable in the grassland ecosystem are also clearer than for the savanna ecosystem. A possible reason for the higher and more consistent trends in activities for the grassland ecosystem might be due to the higher clay content that gives a higher buffer capacity to the soil. On the other hand, the low clay content as well as the presence of *Acacia* species throughout the savanna rangelands, especially in the moderate grazing conditions, might affect these results. More in depth research is thus needed to investigate and explain this phenomenon.

Table 6.3 Normalized values for some soil biological properties (SE in brackets)

			β-glucosidase (mg p-Nitrophenol g^{-1} C h^{-1})	Dehydrogenase (mg INF g^{-1} C 2 h^{-1})	Urease (mg NH_4-N g^{-1} N 2 h^{-1})	Total-PLFA (µmol g^{-1} C)
Grassland ecosystem	**CF**	**Poor**	344.6 (±37.7)	23.2 (±3.1)	60.4 (±23.5)	100.2 (±16.1)
		Moderate	294.3 (±69.4)	19.9 (±3.1)	28.1 (±10.1)	91.9 (±35.9)
		Good	232.3 (±19.3)	14.3 (±3.1)	20.9 (±8.8)	58.5 (±9.7)
	CO	**Poor**	296.5 (±35.6)	22.9 (±3.5)	83.6 (±45.6)	80.9 (±14.9)
		Moderate	249.50 (±15.2)	20.4 (±3.1)	54.7 (±3.5)	61.2 (±3.1)
		Good	201.60 (±6.0)	18.3 (±3.5)	23.5 (±15.1)	62.8 (±9.0)
Savanna ecosystem	**CF**	**Poor**	981.2 (±196.9)	57.8 (±4.7)	24.9 (±9.3)	516.4 (±326.7)
		Moderate	1107.6 (±239.2)	60.3 (±13.1)	14.4 (±1.7)	276.9 (±58.9)
		Good	679.4 (±135.4)	45.3 (±12.5)	9.4 (±2.8)	661.8 (±517.7)
	CO	**Poor**	1387.2 (±367.2)	103.1 (±44.9)	21.0 (±5.2)	472.2 (±119.4)
		Moderate	1252.9 (±123.3)	62.3 (±6.9)	24.6 (±4.5)	214.8 (±22.6)
		Good	1461.4 (±335.8)	59.7 (±12.4)	19.0 (±3.9)	214.9 (±36.5)

6.5 Conclusions

In this study two distinctive ecosystems were compared regarding the influence of rangeland management systems on soil microbiological and associated properties. Most other studies in literature have focused on the relation between soil properties and microbes in only one research area. Our data demonstrated that in both ecosystems a decrease in grazing pressure on a rangeland, such as by commercial farmers practicing rotational grazing, could stimulate microbial activity. Likely there was a positive feedback between microbial mediated nutrient mineralization and plant growth, as all microbial biomass and activity as well as grass cover and biomass were elevated when grazing pressure changed. Yet, this study revealed that there are differences depending on whether intensive grazing only reduces vegetation growth by animal trampling (also on CO farms), or whether it also sustains vegetation growth after adequate resting times. In the latter case, urine and dung contributions as well as indirect effects by grass cover promoted microbial performance in the piospheres. This phenomenon was more obvious with the rotational grazing practiced by commercial farmers than with the continuous grazing practiced by communal farmers. Hence, urine and dung of grazing animals, especially in the piosphere, may partially neutralize the negative effect of overgrazing, by the inputs of labile N.

A specific situation occurs when bushes invade: the bush encroachment in the savanna ecosystem, especially evident in the moderate grazing condition of the continuous grazing systems, also can improve soil quality, e.g., by supplying leaf/stem litter, root mass and microbial activity, and thus providing a larger diversity of organic matter. With this it is also interesting to note that in the long-term, the sandy soils seem to be more resilient to soil degradation, indicated by less significant differences in all measured parameters between the rotational and continuous grazing systems. In the short-term, however, it were the clayey soils in the grassland ecosystem that showed evidence of resilience, as the resting times in the rotational grazing systems was obviously able to compensate or restore disturbances from high grazing pressure, which was not possible under continuous grazing management.

CHAPTER 7

SUMMARY, SYNTHESIS AND RECOMMENDATIONS

7.1 Summary

Degradation of rangeland ecosystems severely threatens the livelihoods of African societies and economies. However, the causes of and the processes involved in these changes as well as human interactions with them are poorly understood. This study was set out to explore whether rangeland degradation is controlled by farmers' decisions about land use patterns in a clayey grassland ecosystem and in a sandy savanna ecosystem, and if either ecosystems respond similarly or differently to rangeland degradation. The study has also sought to know whether soil properties decline along degradation gradients, and if this effect will be more pronounced in communal farms where continuous grazing is practiced, and less in commercial farms where rotational grazing occurs. Additionally, this study also investigated whether soil microbiological parameters might be more sensitive to predict and evaluate the response of soils to degradation caused by rangeland management, compared to the more traditional soil physical and chemical parameters that is typically analysed for soil quality indicators. Lastly, this study ultimately intended to see whether the effects of rangeland degradation on soil properties are potentially reversible.

7.2 Synthesis

The main empirical findings were summarized within the respective chapters. This section will synthesize the empirical findings in order to meet the two main objectives of the study.

1. Evaluate the impact of different rangeland management systems on soil degradation, in a clayey grassland and sandy savanna ecosystem**.**

A significant portion of grassland and savanna ecosystems is over-utilized by livestock, due to inappropriate rangeland management. Although not all land is overgrazed, there are some parts where signs of degradation can be found. Overgrazing has detrimental effects on soil and vegetation, but these changes can be reversed or prevented by proper rangeland management practices. Sustainable utilization of the rangeland ecosystem is based on the appropriate application of rangeland management principles that will safeguard long-term productivity and profitability of the production system at the lowest possible risk. Responsible or well-managed grazing practices have the potential to enhance the overall soil physical, chemical and biological quality.

Three rangeland management systems (commercial, communal and land reform) that differ mainly in ownership, managing of grazing resources and stocking rate, were selected for this study to compare different rangeland managing strategies. For each farm a representative degradation gradient (poor, moderate and good grazing condition) was selected and defined by indicator grass species for the purpose of rangeland condition assessment. Different grass species exist in the clayey grassland and sandy savanna ecosystem, with *Acacia* species being dominant in the savanna ecosystem. The soils in both ecosystems responded differently to increased rangeland degradation. In the savanna ecosystem bush encroachment lead to a temporary improvement of the soil quality, whereas in the grassland ecosystem bare patches and soil crusts lead to degradation of the soils.

In the clayey grassland ecosystem, results showed that communal farms with continuous grazing were generally depleted in the respective plant nutrients (e.g. total N, exchangeable Ca, K, and extractable P, Fe and Zn), compared to the commercial farms with rotational grazing. This depletion was amplified when grazing pressure increased, as was evident especially in the poor grazing conditions in the piosphere (especially with amounts of organic C, total N and extractable P being almost double in the commercial farms compared to the communal farms). A breakdown of macroaggregates with losses of associated C and N were noticeable, proving that aggregate fractionation is a sensitive indicator for detecting the beginning of soil degradation in this ecosystem. Soil degradation was less pronounced under the rotational than under continuous grazing systems. The results from soil analyses confirm that fences and appropriate grazing periods are needed to manage these rangelands sustainably.

In the sandy savanna ecosystem, results also revealed that communal farms were affected negatively by continuous grazing, which exhausted most plant nutrients (e.g. total N, exchangeable Ca, Mg, K, Na, and extractable P, Cu, Fe and Zn) especially close to the water points, when compared to rotational grazing in commercial farms. In contrast, the communal farms had more plant nutrients than commercial farms when moving away from the water points (e.g. total N, exchangeable Ca and Na, and extractable Fe and Mn). This enrichment of plant nutrients coincided with an increase in *Acacia* species, where specific soil analyses (P fractions, particulate organic C and isotopic composition of organic matter) confirmed that soils on continuous grazing systems were strongly influenced by the overlying woody *Acacia* vegetation. Only near the water points, high grazing pressure had overridden the positive effects of *Acacia* species. Hence, and in contrast to the results from the grassland ecosystem, rangeland degradation in communal farms of the savanna ecosystem improved soil quality due to bush encroachment, but at the cost of palatable grass area.

2. Establish whether there is a difference in soil physical, chemical and microbiological properties across degradation gradients due to grazing within these different rangeland management practices, as well as between the two ecosystems.

In this study two distinctive ecosystems differing in climate, vegetation and soil were compared regarding the influence of rangeland management practices on soil physical, chemical and microbiological properties. Most other studies in literature have focused on only one research area, and this study provided the ideal opportunity to compare how rangeland management practices influenced soil properties in two different ecosystems. The results revealed that the majority of soil physical, chemical and microbiological properties measured were significantly higher in the clayey grassland ecosystem compared to the sandy savanna ecosystem (in most cases more than double the amount), irrespective of the rangeland management practices, indicating that soil texture plays a significant role in the active processes in these ecosystems. Furthermore, it was clear that the soil properties, which were involved with the soil degradation/improvement processes, differed between these two ecosystems. For the grassland ecosystem, due to the higher clay content (27.9%), soil aggregates were crucial in protecting organic matter and associated microbiology. Nevertheless, even though the clay content was very low in the savanna ecosystem (2.9%), particulate organic matter played a similar important role here. Results further indicated that soil microbiological properties are more sensitive to changes in land use compared to soil chemical or physical properties, and might be better indicators to use when evaluating rangeland degradation. Within the clayey grassland ecosystem soil properties responded more to what was expected due to the buffer capacity of the clayey soil, whereas in the sandy savanna ecosystem, this was not the case. Decreasing the grazing pressure on a rangeland, such as commercial farmers practicing rotational grazing, can stimulate microbial-mediated nutrient mineralization with positive consequences on plant growth. There were positive correlations between soil microbiological properties and organic C and total N content, with the C/N ratio of 6.03 indicating that overlying woody *Acacia* vegetation strongly influenced especially chemical properties. In this study grazing mainly affected soil properties through the direct effect of animal trampling as well as urine and dung contributions, and indirectly through its effect on the perennial grass cover, which suggests that urine and dung of grazing animals may partially neutralize the negative effect of overgrazing, especially in the piosphere region of a rangeland. When comparing soil properties in the sandy savanna ecosystem, most significant differences were observed within the poor, moderate and good grazing conditions, irrespective of the rotational and continuous grazing systems, indicating that this ecosystem seemed to be more resilient to soil degradation over the long-term. Whereas the clayey grassland ecosystem showed signs of being more resilient over the short-term, indicated by significant differences in soil properties when comparing the rotational and continuous grazing systems, irrespective of the grazing conditions.

7.3 Theoretical implications

In order for rangeland management to be effective and sustainable, the response of soil to overgrazing as well as rangeland degradation need to be quantified to develop suitable grazing practices (Bailey, 2004; Sollenberger *et al*., 2012). The effect of grazing management on the response of ecosystem structure and function are nevertheless inconstant, yet grazing intensity appears to be the most important driver of net primary productivity and composition, especially in semi-arid regions (Briske *et al*., 2008). This study contributes towards a better understanding of the causes and the processes involved in rangeland degradation with regard to soil, by comparing a clayey grassland ecosystem and a sandy savanna ecosystem, with different strategies of grazing management.

Rangeland management in South Africa tend to focus more on the effects of management practices on forage production and animal responses, but less attention is given to the impact of grazing on soil properties (Snyman and Du Preez, 2005; Du Toit *et al*., 2009). Land use changes can also apply pressure on the soil, which can ultimately lead to soil degradation. In both the grassland and savanna ecosystems, the further ecosystem development cannot be understood independently from the socio-economic conditions. Nevertheless, almost nothing is known on the extent of soil degradation when comparing different rangeland management systems. Neither do we possess knowledge on the rate at which soil properties change when rangeland utilization intensifies within these systems, nor on the reversibility of these changes (Emmerich and Heitschmidt, 2002). This study indicates that grazing had a direct impact on soil properties (e.g. organic C, total N, extractable P, exchangeable Ca and Mg), and that some systems are more resilient to soil degradation than others. It is also evident that some soil properties (e.g. soil aggregates and POM) are more sensitive to degradation than other soil properties.

The role of soil microorganisms in many ecosystem processes such as organic matter decomposition, nutrient cycling, as well as organic C sequestration is significant. These soil organisms are also essential drivers of plant diversity and productivity in terrestrial ecosystems, and consequently of sustainable land use (Van der Heijden *et al*., 2008; Herold *et al*., 2014). Despite their abundance in soil, the impact of soil microbes on ecosystem processes is still poorly understood, especially their contribution to plant productivity and diversity. Some studies found that soil organisms are more controlled by soil properties than by rangeland management, especially at regional scales (Grayston *et al*., 2001; Herold *et al*., 2014). This study demonstrated that differences in soil properties between two diverse ecosystems with different vegetation had a substantial impact on soil microbiological properties. However, long-term effects of rangeland management on these soil microorganisms are not well known, neither are the effect of different grass species. This type of study can clearly provide answers towards understanding the important role of soil microorganisms in rangeland management.

Furthermore, to effectively guide farmers and farm managers in applying sound grazing management strategies, a holistic approach should be taken with interdisciplinary collaboration (Van der Westhuizen and Snyman, 2014). The ultimate objective should be improvement of the grazing condition over the long-term, which will eventually lead to an increase in grazing capacity. In this study it was evident that many factors played a role in rangeland degradation, and careful consideration should be taken when advising on management strategies. This study will therefore contribute, along with the studies by Snyman *et al*. (2013) and Van der Westhuizen *et al*. (2013) towards successful rangeland management, especially in southern Africa.

7.4 Recommendations for future research

While careful planning occurred in the initial stages of this project a few recommendations can be made to smooth line this type of study for future reference.

- In this study, choosing the study sites (farms) within the research areas to represent the different rangeland management practices (commercial, communal, land reform), proved to be challenging, especially for the land reform farms. After sampling it was realized that two of these farms were not strictly land reform farms as defined politically, because they were bought and not obtained via the resettlement program. Results from soil analyses of these land reform farms were also difficult to interpret. The selection of study sites should therefore be carefully considered in future.

- Soil sampling took place using the idea of a pioshere within a camp to represent a grazing gradient. This is however flawed, since the camp sizes differed, and a better solution might be to choose one camp to represent a specific grazing condition, based on dominant grass species composition in that specific camp. The number of farm replicates used for a specific rangeland manangement system also could have been increased, however, this was not easy to do due to logistical and financial constraints. Using the camp system might solve this problem.
- In order to evaluate true sustainability in any rangeland management system, long-term research is imperative. This should include economic, social and ecological monitoring, which requires interdisciplinary research that involves soil scientists, ecologists, plant scientists, agricultural economists, social scientists, historians and extension officers. Devising practical measures to combat soil degradation in order to extend scientific findings to small and commercial farmers should be an important phase of the research project.
- A focused attempt to increase active collaboration between all relevant disciplines in the research of rangeland management, especially the hard sciences like soil science and vegetation sciences, should be made. This is important to ensure successful guidance to rangeland managers.
- Soil microbiological properties like enzymes and PLFAs proved to be important when evaluating rangeland degradation, however, the question would be if other microbiological methods might be more efficient in providing the answers. Another important aspect could be to sample over seasons, and see whether these

parameters change over a year or over seasons, in order to make long-term predictions.

7.5 Closing remarks

Based on the results and findings of this study, it is evident that the effect of rangeland degradation on soil quality is a cause of concern. However, understanding the function and response of an ecosystem when rangelands are degraded, is important to slow down or even reverse this process. The results obtained in this study can serve as guidelines and strategies for soil and rangeland degradation in the arid and semi-arid grassland and savanna ecosystems of southern Africa, where rainfall is often limited. Better understanding of the effects of rangeland management on soil properties is critical for local and global issues on how to ensure sustainability of rangelands, both for current and future generations.

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