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Centre for the study of Environmental Change and Sustainability

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Understanding the savanna dynamics in relation to rangeland management systems and environmental conditions in semi-arid Botswana

Olaotswe Ernest Kgosikoma

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HERBACEOUS VEGETATION RESPONSES TO RANGELAND MANAGEMENT SYSTEMS AND ENVIRONMENTAL VARIABLES ACROSS SEMI-ARID SAVANNAS, BOTSWANA.. 58

Declaration

I declare that this thesis was composed by myself and its contents are all my own work.

Name: Olaotswe Ernest Kgosikoma Date: 19th July 2011

Signature:

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Abstract

This thesis investigates the effects of rangeland management systems on savanna ecosystems under different environmental conditions in Botswana, Southern Africa. The soil sampling and vegetation assessment were conducted in twenty–three transects located in communal and ranching grazing lands across three sites. Pastoralists were also interviewed on vegetation change on their respective grazing lands.

Soil texture, pH, bulk density and soil organic carbon differed between sites, but not between communal and ranching lands. Soil organic carbon was positively affected by soil clay content. The herbaceous vegetation composition showed heterogeneity between sites, but Matlolakgang and Xanagas rangelands were in poor condition, particularly the communal grazing land that had high cover of increasers II species. Higher herbaceous biomass was observed in ranches than communal lands and biomass also increased with increasing soil organic carbon.

Bush encroachment was observed in communal and ranching grazing lands at Matlolakgang and Xanagas, but not Goodhope. Woody plant cover, density and diversity increased with decline in soil clay content, but not linearly. Pastoralists indicated that herbaceous vegetation compositions had changed in some areas, with increase of unpalatable grass species and bush encroachment especially in communal land. Pastoralists considered bush encroachment a problem as it suppress herbaceous vegetation productivity, but they also considered woody vegetation as a valuable grazing resource.

The long-term indicators of ecosystem degradation (soil and woody cover) showed that communal and ranching grazing did not affect the savanna ecosystem differently. This is contrary to assumptions of Tribal Grazing Land Policy of Botswana, which promote ranching as a more sustainable management system. The results revealed that rainfall and soil clay strongly influence rangeland condition. These results have implications for the management policies of communal grazing lands throughout sub-Saharan Africa and in similar arid habitats across the world.

Chapter 1

The savanna ecosystem dynamics and rangeland management systems

1.1 General introduction

Savanna ecosystems are globally important and cover about 20 % of the Earth's land surface area (Bond and Midgley, 2000; Sankaran *et al.*, 2005) and 50 % of Africa (Wang *et al.*, 2010). The savanna biome is characterised by scattered trees and continuous layer of herbaceous plants (Knoop and Walker, 1985; Sankaran *et al.*, 2005; Wiegand *et al.*, 2006). In recent decades, savanna ecosystems have been experiencing environmental degradation, characterised by vegetation composition shift towards bush encroached ecosystems (Archer *et al.*, 1995; Sankaran *et al.*, 2005; Yanoff and Muldavin, 2008). Rangeland degradation also consists of loss of biodiversity and palatable grass species for livestock (Abule *et al.*, 2005; Metzger *et al.*, 2005) and increased cover of unpalatable plants such as forbs (Makhabu and Marotsi, 2011). The change in composition of savanna vegetation is attributed to human disturbance (e.g., fire and grazing) and abiotic factors (e.g., rainfall and soil fertility) (Scholes and Archer, 1997; Treydte *et al.*, 2007).

The drivers of changes in savanna ecosystems are highly debated (Vetter, 2005) especially the impact of land use and rainfall variability. Livestock grazing is the dominant land use in savanna ecosystems globally (Ringrose and Matheson, 1991;

Dorrough and Scroggie, 2008; Masike and Urich, 2008) and subsequently vegetation changes have mainly been attributed to poor livestock management systems (Brown and Archer, 1999). According to the conventional range management concept, overgrazing causes loss of palatable herbaceous plant species and bush encroachment in the savanna (Abule *et al.*, 2005). This argument is supported by land degradation (Rohde *et al.*, 2006) and “tragedy of commons” narratives (Hardin, 1968). The pastoralists, especially those using communal grazing land, are criticised for poor livestock management that degrades the environment (Hardin, 1968; Allsopp *et al.*, 2007). Yet, pastoralists’ perception and ecological knowledge is often not considered during rangeland condition assessment despite their knowledge of the local environment (Berkes *et al.*, 2000; Angassa and Oba, 2008b; Roba and Oba, 2009). Pastoralists’ management practices are ecologically adapted to local environmental conditions (Fernandez-Gimenez, 2000) and the local community’s participation in rangeland assessment and monitoring is essential for understanding changes in their environment (Quinn *et al.*, 2007).

Improved knowledge on savanna ecosystem dynamics under different soil types, landscapes, rainfall patterns (Tefera *et al.*, 2008b) and land use is essential to guide sustainable management. This thesis therefore investigates the changes in savanna ecosystem in relation to rangeland management systems and environmental conditions across the savanna ecosystems of Botswana. The ecological data on soil and vegetation dynamics across savanna ecosystems are complemented by pastoral ecological knowledge, which provide long-term knowledge on savanna ecosystem dynamics and implications of ecosystem change to pastoralists’ livelihood.

1.1.1 Justification

Globally, managed grazing lands comprise the largest land use (Liebig *et al.*, 2006) estimated to cover about 25 % of Earth's land surface (Asner *et al.*, 2004). This makes them an essential resource for both maintaining environmental services like biodiversity conservation (O' Connor, 2005) and as a source of livelihood, especially for rural communities (Eriksen and Watson, 2009a; Muhumuza and Byarugaba, 2009). In Botswana, approximately 76 % of total land surface area is used for grazing (Asner *et al.*, 2004) and those grazed by domestic animals, especially communal rangelands, are being degraded and their sustainability threatened by overgrazing (Vetter, 2005; Darkoh, 2009). The degraded savannas are dominated by unpalatable and annual herbaceous plant species and encroached by bushy vegetation (van Vegten, 1984; Abule *et al.*, 2005). The changes in savanna compositions influence the sustainability of livestock production, and ecosystem functions such as carbon cycles (Sankaran *et al.*, 2005). Given the extent of rangelands in Botswana, their degradation is likely to have a disproportionately high impact on the livelihood of pastoralists and biodiversity conservation.

In response to rangeland degradation, the ranching system was introduced in Botswana in 1975 as a better option that would promote conservation of natural ecosystems and sustainability of savanna ecosystems (Botswana Government, 1975; Dougill *et al.*, 1999). The appropriateness of the ranching system and its underlying assumptions in dry savannas has recently been debated (Ellis and Swift, 1988;

Dahlberg, 2000b; Rohde *et al.*, 2006). Some authors (Ellis and Swift, 1988; Westoby *et al.*, 1989) suggest that savanna ecosystems are primarily influenced by rainfall variability and traditional practices in communal lands are more suitable for exploiting heterogeneous arid ecosystems. Contentious theories (e.g. equilibrium vs non-equilibrium) on savanna dynamics have led to uncertainty about appropriate management systems for these ecosystems. Yet, few studies have undertaken comparative analysis of the effects of management systems on savanna ecosystems dynamics (Dahlberg, 2000b; Smet and Ward, 2005; Tefera *et al.*, 2010) and are mostly site specific (Asner *et al.*, 2004), despite the high variability of the savanna ecosystem. A broad-scale multivariate analysis of relationships between diversity, environmental variables and management systems would improve our understanding of savanna ecosystem dynamics (van der Heijden and Phillips, 2009). By considering the interactions between natural factors (e.g. rainfall & soil types) and anthropogenic drivers (Scholes and Archer, 1997), comparative analysis improves knowledge of how a particular factor influences vegetation conditions (Groffman *et al.*, 2007). Proper understanding of the effects of rangeland management systems on ecosystem dynamics is essential for maintaining productivity and biodiversity in rangelands (Sternberg *et al.*, 2000; Mohammed and Bekele, 2010). This in turn will enhance the sustainability of savannas and the livelihood of pastoral communities and ecosystem functions (e.g. carbon sequestration).

1.1.2 Aim of study

The aim of this study was to investigate the effect of rangeland management systems on savanna ecosystem dynamics in semi-arid conditions of Botswana. The research goal was to use both ecological research and pastoral ecological knowledge to understand the savanna dynamics in relation to rangeland management systems. The specific objectives were;

a. To investigate soil characteristics in relation to rangeland management systems and environmental variability.
b. To compare the effect of rangeland management systems on herbaceous vegetation.
c. To compare bush encroachment under communal grazing and ranching systems.
d. To use pastoral ecological knowledge to understand the savanna ecosystem dynamics in relation to rangeland management systems.

1.1.3 Thesis structure

This thesis consists of six chapters that investigate the effects of rangeland management systems and environmental conditions on the soil properties and vegetation structure of savanna ecosystem.

Chapter 1: The present chapter provides a review of literature on savanna ecosystem dynamics in relation to livestock production systems. It discusses the structure of savanna ecosystem, ecological models, management systems and policy relevant to grazing management in savanna ecosystems. The objectives, rational are introduced in the context of existing knowledge and sites and methods described.

Chapter 2: This chapter focuses on soil properties variations in response to management systems and environmental conditions.

Chapter 3: It investigates the herbaceous vegetation dynamics in association with grazing management systems and environmental conditions.

Chapter 4: It characterise bush encroachment in grazing management and its relationships with different environmental conditions across savanna ecosystems.

Chapter 5: Characterise vegetation in grazing lands as perceived by pastoral communities.

Chapter 6: This chapter summaries and discuss all results and their implication to management and policy. It integrates the entire body of work.

1.2 Literature review

Globally, the savanna ecosystems are generally well-researched (Skarpe, 1992; Dougill *et al.*, 1999; Smet and Ward, 2005) and knowledge of interactions between vegetation, soil and herbivores has been well documented (Fensham *et al.*, 2005; Metzger *et al.*, 2005; Vetter, 2005). However, savanna ecosystems are diverse (Smith and Smith, 2001) due to habitat response to both biotic and abiotic factors (Skarpe, 1992) and therefore results from various studies do not always agree. In turn, this has resulted in controversy over the effect of grazing management and rainfall on arid ecosystems (Sankaran et al., 2005) and how to best manage the ecosystems.

1.2.1 The structure and determinants of savanna ecosystem dynamics

The savannas are characterised by the coexistence of a continuous layer of herbaceous vegetation and a patchy cover of woody vegetation in the landscape (Skarpe, 1992). The stable association of trees and grasses is attributed to limited soil moisture (Walker *et al.*, 1981), disturbance (e.g. grazing and fire), and poor soil fertility, which limits the establishment of woody vegetation (Sternberg *et al.*, 2000; Scholes *et al.*, 2002; van Langevelde *et al.*, 2003). In arid environments, soil moisture is considered

the major limiting factor to plant growth due to rainfall variability (Walker *et al.*, 1981). Subsequently, Walter's two layer hypothesis, which uses the competition for soil moisture between woody and herbaceous vegetation, is often used to explain the structure of savannas vegetation (Walker *et al.*, 1981; Knoop and Walker, 1985). The woody and herbaceous plants coexist by utilizing soil moisture at different soil layers, with herbaceous plants as superior competitors for moisture in the top soil layers, while the woody plants have exclusive use of moisture in the deeper soil horizon (Walker *et al.,* 1981). Disturbance such as grazing offsets the dominance of herbaceous species by reducing biomass and as a result the establishment of woody vegetation is enhanced as more water becomes available (Knoop and Walker, 1985; Ward, 2005).

The two-layer theory assumes that woody species are all deep rooted, but some woody plant species such as *Combretum apiculatum* and *Terminalia sericea* have shallow lateral and adventitious roots that compete for moisture in the topsoil layer after rains (Cole and Brown, 1976; Cole, 1986). These shallow rooted woody plant species have the potential to increase in population size in response to increased soil moisture availability especially when the habitat is overgrazed and the grass biomass reduced (Skarpe 1990b, Scholes and Archer 1997). In addition, the roots of tree seedlings compete directly with grass roots for moisture in the surface soil layer (Kraaij and Ward, 2006) and also some soils are too shallow for the establishment of separate rooting niches (Wiegand *et al.*, 2005). These arguments highlight the limitations of the two-layer mechanism to explain the savanna vegetation dynamics and hence alternative theories such as patch dynamics have been proposed (Higgins *et al.*, 2000; Meyer *et al.*, 2009).

The patch dynamics theory is suggested as a unifying mechanism to several theories that explain the coexistence of woody and herbaceous vegetation in savanna (Wiegand *et al.*, 2005; Meyer *et al.*, 2007; Meyer *et al.*, 2009). The theory characterises the savanna ecosystem as a mosaic of open and bush encroached patches reflecting different stages of an asynchronously cyclical succession (Wiegand *et al.*, 2006). The shifting-mosaic in the savanna is a natural process driven by inter-tree competition and environmental conditions (e.g. rainfall) (Wiegand *et al.*, 2005). The recurrence of high rainfall events promotes the establishment of woody plants (Wiegand *et al.*, 2005; Joubert *et al.*, 2008). For example, three successive years of good rainfall are required for successful establishment of *Acacia mellifera* (Joubert *et al.*, 2008). Bush encroachment is therefore a natural transient stage in the savanna ecosystem's cyclical succession (Meyer *et al.*, 2009).

1.2.2 Ecological theories and rangeland management

The rangelands dynamics in dry environments are explained by several conceptual models (Belsky, 1990), but the debate on equilibrium and non-equilibrium theories has been foremost (Westoby *et al.*, 1989; Vetter, 2005; Tefera *et al.*, 2010). The key debate thus far has been the importance of biotic and abiotic factors in regulating both primary and secondary production in rangelands (Vetter, 2005). The shift in ecological theory has implications for rangeland condition assessment and causal factors of changes in rangeland (Ellis and Swift, 1988; Dougill *et al.*, 1999; Vetter, 2005). The appropriate conceptual model of vegetation dynamics is therefore essential for sustainable rangeland management (Joubert *et al.*, 2008).

The equilibrium theory suggests that rangeland ecosystems are in a state of dynamic equilibrium with the environment, especially when undisturbed by anthropogenic activities (Cole, 1986; Ellis and Swift, 1988) and that a particular rangeland has a single climax state, which occurs in the absence of grazing (Westoby *et al.*, 1989). The climax state is characterised by dominance of palatable and perennial grasses such as *Brachiara nigropedata* (van Oudtshoorn, 2002). However, there is strong interdependency between herbivores and the natural vegetation (Dickhoefer *et al.*, 2010) such that grazing pressure constantly pushes the vegetation away from climax state (Westoby *et al.*, 1989). The changes caused by grazing could be reversed by reducing the grazing pressure and management goals should be to establish a long-term balance between the grazing pressure and the successional tendency (Ellis and Swift, 1988; Westoby *et al.*, 1989). Subsequently, proper management of rangelands requires knowledge of carrying capacity of the land and stocking rates (Mphinyane *et al.*, 2008).

In contrast, the non-equilibrium concept suggests that climate variability, particularly rainfall, is the key driving force in dry rangeland ecosystems (Ellis and Swift, 1988). The impact of grazing on vegetation is minimal, therefore overshadowed by the effect of rainfall variability (Ellis and Swift, 1988; Westoby *et al.*, 1989). The vegetation changes in non-equilibrium ecosystems are discontinuous and a particular rangeland could exist at different "states" of vegetation or transition states (Westoby *et al.*, 1989). The ecosystems in drylands are recognized as unstable and transitions between states are facilitated by natural events (e.g. fire, drought) and management actions (e.g. stocking rates change). The management of arid ecosystems should be flexible and opportunistic as illustrated by Turkana pastoralists in Kenya who use mobility to respond to vegetation heterogeneity (Ellis and Swift, 1988). The non-equilibrium and patch dynamics models both emphasise the role of abiotic factors and describe arid terrestrial ecosystems as heterogeneous patches of vegetation, though in the non-equilibrium model the vegetation change is not necessarily cyclic as suggested in patch dynamics. Lately, it has also been proposed that rangeland ecosystem dynamics are better explained as a continuum between equilibrium and non-equilibrium (Derry and Boone, 2010).

Despite contrasting theories and causal factors of savanna dynamics as illustrated by above discussions, land use is considered a dominant driver of environmental change in savanna ecosystems (Scholes and Breemen 1997). Livestock grazing is a major land use and plays a key role in shaping the structure and dynamics in savannas (Skarpe, 1991) through its effects on soil properties, herbaceous and woody vegetation.

1.2.3 The effect of livestock grazing on savanna soils

Grazing management systems may degrade the savanna ecosystem through their impact on soil attributes (Liebig *et al.*, 2006). The removal of aboveground plant biomass by livestock grazing reduces soil quality (Walker and Desanker, 2004). High grazing intensities reduce vegetation cover, which limits organic matter added to the soil, and subsequently contribute to reduced soil structure stability, resistance to rainfall impact, infiltration rate and fauna activities (Roose and Barthes, 2001; Snyman and du Preez, 2005). Domestic grazing animals also change plant species communities (Klumpp *et al.*, 2009), which in turn affect soil fertility (Scholes, 1990) because of changes in root biomass (Klumpp *et al.*, 2009) and quality of organic matter. The soil nutrient depletion eventually causes declines in the primary production (Girmay *et al.*, 2008) and carrying capacity of rangelands.

In addition, livestock grazing causes compaction of soil (Geissen *et al.*, 2009) by exerting pressure on the soil, but this is dependent on animal size, the stocking density, soil texture, soil moisture and vegetative cover (Bilotta *et al.*, 2007). The livestock's hoof action compacts soil through reduction in pore space (Drewry, 2006), which increases bulk density, especially on the surface soil (Liebig *et al.*, 2006). Soil compaction then leads to reduced water infiltration rate, increased runoff (Asner *et al.*, 2004) and soil erosion (Fatunbi and Dube, 2008) during rainfall.

Good soil quality, both physical and chemical, is essential to sustain terrestrial productivity and the processes that maintain environmental quality (Moussa *et al.*, 2008) and sustainable production systems are those that result in soil quality improvements (Hopmans *et al.*, 2005; Liebig *et al.*, 2006). Soil nutrient balance is therefore a good indicator of the extent of land degradation (Cobo *et al.*, 2010) especially in rangelands and other natural ecosystems where there are minimal external inputs of nutrients. Soil attributes such as soil carbon reflect the long-term productivity and sustainability of land use system because they are relatively stable compared to above ground vegetation (Walker and Desanker, 2004). Rangeland degradation is therefore associated with increased soil compaction , decreased soil water content and decreased soil fertility (Snyman and du Preez, 2005). The assessment of soil quality therefore provides insights into its capacity to support the primary production and sustainability of current management systems.

1.2.4 The effects of livestock grazing on savanna vegetation

Savanna vegetation, particularly in developing countries, is largely exploited through livestock grazing (Scholes and Archer, 1997; Bilotta *et al.*, 2007) and the intensity of grazing influences the defoliation rate and sustainability of the ecosystem (Mphinyane *et al.*, 2008). Most grasses in savannas are fairly tolerant to grazing, but prolonged high grazing intensities will eventually lead to change in species composition (Skarpe, 1992) and reduction in grass biomass especially when soil resources are depleted (van Auken, 2009). Overgrazing affects botanical composition and species diversity by depressing the vigour and presence of dominant species, which then enables colonization of less competitive, but grazing tolerant plant species (Sternberg *et al.*, 2000). Livestock also selectively graze palatable herbaceous plants, which are then replaced by annuals and unpalatable herbaceous plants and bushes (Skarpe, 1992) resulting in the decline of species richness (Fensham *et al.*, 2010). Heavily grazed rangelands are therefore dominated by Increaser II species such as *Aristida congesta* (Trollope *et al.*, 1989) and are indicators of rangeland degradation (Fatunbi and Dube, 2008). The lightly grazed rangelands are dominated by decreaser species that tend to decrease under heavy grazing, while Increaser I species are those that increase in underutilized rangelands (du Plessis *et al.*, 1998). In addition, grazing removes grass biomass and reduces the fuel load, which in turn reduces fire occurrence and intensity and as a result creates a conducive environment for bush encroachment (van Langevelde *et al.*, 2003) .

Bush encroachment is an indicator of environmental degradation (van Vegten, 1984; van Auken, 2009) and is a concern in many parts of Africa and elsewhere in the world (Moleele and Perkins, 1998; Roques *et al.*, 2001; Moleele *et al.*, 2002; Angassa and Oba, 2008a; Sankaran, 2009). A shift from grass dominated savanna vegetation to bushy dominated vegetation has been attributed to overgrazing (Moleele and Perkins, 1998) and lack of fire (Heinl *et al.*, 2008; Lehmann *et al.*, 2008) and is exacerbated by frequent droughts (Cole and Brown, 1976; Scholes and Archer, 1997; Smith and Smith, 2001) and climate change (Fensham *et al.*, 2005). Increased cover of bushy vegetation threatens livestock production because it causes reduction in herbaceous diversity and production (Scholes and Archer, 1997) through competition for soil moisture and nutrients. On the other hand, leguminous woody plants such as *Acacia mellifera* enrich nutrient poor sandy soils in dry savannas through nitrogen fixing (Hagos and Smit, 2005). Soils under the canopy of *A. mellifera* were found to have higher levels of total nitrogen, percent organic matter and calcium than soils some distance away from trees (Hagos and Smit, 2005). Some woody plants are also an important fodder resource especially during dry periods (Moleele, 1998). Therefore one management aim could be selective thinning of woody vegetation to reduce the grass-tree competition whilst retaining the beneficial effects of soil enrichment from leguminous tree and shrub species (Hagos and Smit, 2005).

1.2.5 Pastoral ecological knowledge and its application in rangeland management

Rangeland developments in Africa have failed to contribute towards improved biodiversity conservation and livestock production (Rohde *et al.*, 2006; Solomon *et al.*, 2007; Angassa and Oba, 2008b). This has been attributed to poor understanding of ecological ecosystems and traditional practices by policy makers (Tefera *et al.*, 2007). The participation of local communities and use of their ecological knowledge could therefore help policy makers and researchers to better understand the ecosystems and contribute to sustainable ecosystem management (Verlinden and Dayot, 2005; Reed *et al.*, 2008).

Traditional ecological knowledge (TEK) refers to "biophysical observations, skills, and technologies, as well as social relationships, such as norms and institutions, that structure human-environmental interactions" (Fernandez-Gimenez, 2000). This knowledge is accumulated through regular interaction with the environment through land use such as herding and is passed from generation to generation (Berkes *et al.*, 2000; Oba and Kotile, 2001). Thus, TEK also known as local ecological knowledge (LEK) has a long and close relationship with ecosystems (Brook and McLachlan, 2008) and can improve our understanding of ecosystems, especially where there is lack of long-term scientific data (Knapp and Fernandez-Gimenez, 2008).

The TEK is essential for assessment and monitoring of landscape conditions (Roba and Oba, 2009) because pastoralists use it to assess the state of rangeland and changes in ecosystems (Oba and Kaitira, 2006). The pastoralists' perception and knowledge of their environment influences management practices adopted to exploit a particular ecosystem (Ellis and Swift, 1988; Allsopp *et al.*, 2007; Wei *et al.*, 2009). They understand the heterogeneity of arid environments and use appropriate management practices such as mobility and mixed stocking to adapt to such environments (Thomas and Twyman, 2004; Angassa and Oba, 2008b). Hence, it is important to know how pastoral farmers perceive changes in the ecosystem and the implications to their livelihood (Wigley *et al.*, 2009). The Borana pastoralists in Ethiopia identify proliferation of woody vegetation as a threat to livestock production (Angassa and Oba, 2008b). In contrary, the OvaHerero pastoralists in Namibia considered bush encroachment as beneficial because it is a forage resource especially during dry periods (Katjiua and Ward, 2007). Proper understanding and integration of pastoralists' knowledge on local ecosystems with scientific knowledge can therefore contribute towards developing sustainable management systems.

1.2.6 Rangeland Management Systems and relevant policy

Rangeland management systems refer to all production systems used to exploit the rangeland through grazing. In this thesis, rangeland management systems will be limited to livestock grazing management systems although other authors (Smet and Ward, 2005) have used the term broadly to also include game farming. There are two rangeland management systems referred to as communal (traditional) grazing and ranching (commercial) system in Botswana (Rohde *et al.*, 2006; Masike and Urich, 2008). The communal grazing system involves shared use of rangeland in tribal lands by pastoral communities (Rennie *et al.*, 1977; Rohde *et al.*, 2006; Masike and Urich, 2008). Historically, the communal production system was mainly for subsistence, but it has been gradually shifting towards a cash economy (Wigley *et al.*, 2010) since the independence of Botswana in1966. Communal grazing management is mainly influenced by local ecological knowledge (Smet and Ward, 2005).

The traditional practices of communal grazing system include mixed stocking, with cattle and goats as dominant livestock species (Wigley *et al.*, 2010). Traditionally, herding was a key part of management and contributed towards distributing grazing pressure across the landscape. However, herding is currently not commonly practiced in Botswana (Reed *et al.*, 2008) and livestock are rather allowed to continuously and selectively graze according to their preference (Parsons *et al.*, 1997) in communal rangelands around the water-source (e.g. borehole). There is limited surface water especially during the dry season and as a result, animal distribution is strongly dependent on water availability (Tefera *et al.*, 2007). Thus, borehole rights provided by government institutions give the borehole owners control of grazing resources around the water point. The communal grazing lands are grazed throughout the year (Oba *et al.*, 2001) and no rent is paid for rangeland use (Weimer, 1977).

The communal rangelands are generally considered degraded and poorly managed (Ellis and Swift, 1988; Abel, 1997; Dougill *et al.*, 1999; Hendricks *et al.*, 2007). As illustrated by "Tragedy of Commons" (Hardin, 1968), it is argued that it is profitable for each pastoralist to increase their herd size in communal rangeland. Subsequently, the livestock density would increase until the land carrying capacity is exceeded and eventually degradation occurs because of overgrazing (van Vegten, 1984). In response to perceived rangeland degradation due to poor management, the government of Botswana introduced the ranching system through the Tribal Grazing Land Policy (TGLP) of 1975. The communal grazing rangelands are demarcated into private ranches owned by either individuals or groups who pay a rent for exclusive use of particular fenced areas of rangeland (Dougill *et al.*, 1999). As stipulated in the literature (Botswana Goverment, 1975; Tsimako, 1991) , the objectives of TGLP are:

a. To control overgrazing and halt rangeland degradation through improved management (e.g. rotational grazing & optimal stocking rates) in commercial ranches and shifting large herds of livestock out of already overstocked communal lands.
b. To improve livestock productivity and farmers' income through better management practices (e.g. controlled breeding, & early weaning).

c. To secure interest of the poor (social equity) by reserving the communal grazing land for small scale farmers and have reserve land for future generations.

Ranching management is dependent on a range-succession model, whereby managerial goal is to match the stocking rate to rangeland carrying capacity (Mphinyane *et al.*, 2008). The ranching system is also characterised by rotational grazing, which consists of alternating periods of use and rest, to promote vegetation growth. Hence, ranches are divided into a number of paddocks to allow rotational grazing and spread grazing pressure uniformly across the rangeland. This production system is commercially oriented and normally focuses on a single livestock species, particularly cattle (Smet and Ward, 2005). Despite being promoted as a sustainable management system by grazing policies in Southern Africa (Rohde *et al.,* 2006), ranching has high development costs consisting of fencing, drilling and water reticulation costs (Motlopi, 2006). Thus, it is a production option for members of society that have considerable financial resources resulting in the poorer members of pastoral communities being marginalized as communal grazing areas shrink due to the establishment of ranches (Eriksen and Watson, 2009b). This is further exacerbated by dual grazing rights which allow those farmers allocated ranches to continue having access to communal rangelands (Thomas and Sporton, 1997).

Rangeland use reform through privatisation of communal rangelands has been strongly criticised (Abel, 1997; Rohde *et al.,* 2006). The Tribal Grazing Land Policy

(TGLP) of 1975 was meant to reduce environmental degradation by reducing grazing pressure in communal rangelands (Thomas and Sporton, 1997; Dougill *et al.*, 1999). Yet, it is suggested that TGLP is based on incorrect assumptions (Abel, 1997) and has therefore failed to improve environmental conditions (Dahlberg, 2000b). In addition, the cattle productivity and sustainability of communal rangelands are rather underestimated (Abel, 1997). As shown in Figure 1.1, cattle calving rate in communal rangelands has been consistent over the years and is now slightly higher than in ranching lands. The traditional system is considered more appropriate for dry environments because it adapts to rainfall variability and spatial heterogeneity through opportunistic management such as mobility (Westoby *et al.*, 1989). Despite the TGLP having provision for protecting the interests of the poor through reserve areas and removal of farmers with large herd from the commons (Weimer, 1977), it is not clear how this is achieved during the implementation phase, especially since the communal grazing continues to shrink as more ranches are demarcated. This has considerable implications on issues of equity and access to grazing resources in Botswana.

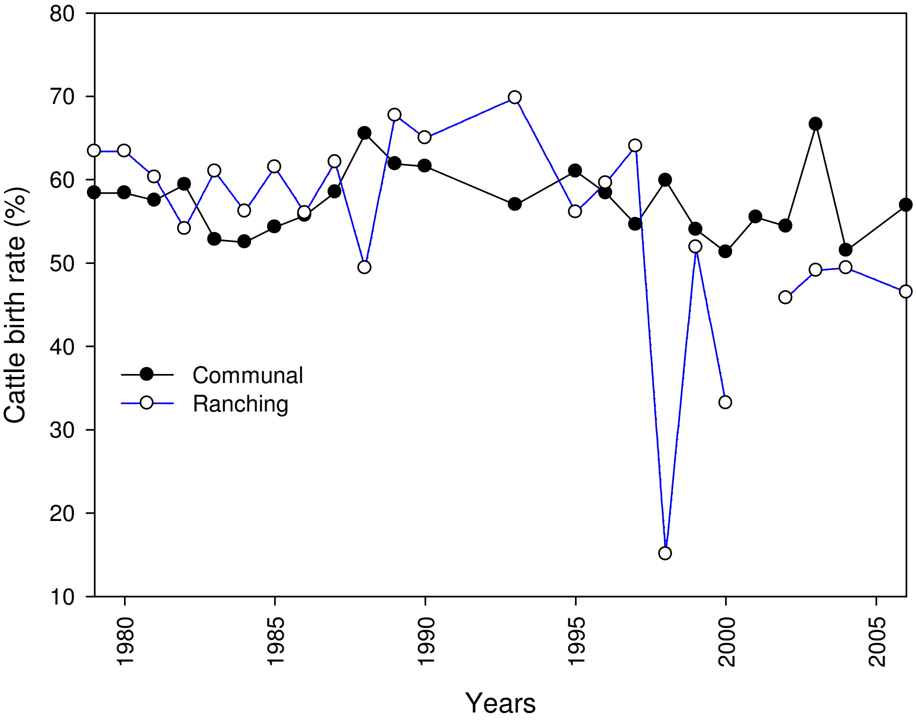

Figure 1.1: Cattle calving rates (%) in communal and ranching lands of Botswana (data: CSO website)

1.2.7 The effect of rangeland management systems on savanna dynamics

There are few studies comparing the effects of communal and ranching management on savanna dynamics (Smet and Ward, 2005; Tefera *et al.*, 2008b) and the results of those studies differ. This variation could be partially attributed to each research being site specific (Sankaran *et al.*, 2005). The savannas are highly heterogeneous (Smith and Smith, 2001) and therefore variation in local environments, such as soils and rainfall, and scales of observations may influence the results (Vetter, 2005). The differences in livestock management practices like stocking rates may also contribute to the lack of consensus especially in a study conducted by Dalhberg (2000) where rotational grazing was not practiced in the ranch despite being a key attribute of the ranching system. It is therefore essential to carry out further comparative research at broader scales and under a range of environmental conditions. The vegetation response to grazing in drylands is mediated by rainfall variability and soil fertility (Anderson *et al.*, 2007; Dorrough and Scroggie, 2008) and therefore, should be taken into account when investigating grazing impact. Vegetation response to grazing under different environmental conditions could be very different as reflected in the literature (Fensham *et al.*, 2010) and application of the results from a few site specific researches on rangeland management could be problematical when making broad management recommendations and policy formulation (Hayes and Holl, 2003; Boyd and Svejcar, 2009).

1.3 Materials and method

1.3.1 Study sites

The study was conducted in Botswana located in Southern Africa. The climate is generally semi- arid with mean annual rainfall ranging from 650mm in the northeast to as low as 250mm in the south-west (Burgess, 2005). The country covers 581 730 km² of land area with 77.2 % classified as Kalahari sandveld and 22.8 % as part of eastern hardveld (Central Statistics Office, 2008b). The vegetation type of Botswana changes from tree savanna and woodland in the northeast to shrub savanna in the southwest (Ministry of Agriculture, 1991; Burgess, 2005). The land tenure system consists of communal land (54.8%), freehold land (3.4 %) and state land (41.8 %) (Central Statistics Office, 2008b). The country is also divided into six agricultural regions: Gaborone, Southern, Maun, Central, Francistown and Western (Central Statistics Office, 2008a) and present study sites are located in the Western, Gaborone and Southern regions.

1.3.2 Site selection and their description

The savannas of Botswana are characterized by variation in the physical environment and composition of the vegetation. Therefore, three sites were selected from a list of range monitoring sites that take into account the ecological variation in the country. At each site, communal grazing land and government ranches located close to each other were selected. Government ranches were preferred to private ranches because their management is uniform and representative of "appropriate" ranch management practices. Given that ranches and communal rangelands are close to each other, it was assumed that both sites are exposed to similar environmental conditions, and soil types. Any localized vegetation changes cannot be attributed to climate such as rainfall, which operates over relatively large areas (Fensham *et al.*, 2005) and is therefore not expected to have any significant effect on vegetation and soil conditions between communal and ranching systems at each site (Wigley *et al.*, 2010). Rather, the management systems are likely to be key drivers to changes in savanna soil and vegetation at local level, while other factors such as soil type and rainfall variability are more relevant at regional level (between the three sites).

Study site 1 is the Goodhope rangelands (Figure 1.2) but is inclusive of villages surrounding Goodhope such as Sheepfarm and Metlojane. It is part of the eastern hardveld and soils are non- calcareous loam soils with low amount of calcium carbonate (APRU 1979). Arable farming is an alternative source of livelihood but livestock production is still dominant. The mean annual precipitation (MAP) is 459.2 mm at this site, the highest of the three sites in present study (Figure 1.3a) and can range from 186 mm to 734 mm, which reflects high inter-annual rainfall variability (CV = 35 %). Rain falls mainly between October and March (Figure 2a). The vegetation is generally characterised as *Acacia giraffae* Tree Savanna (APRU 1979) which indicates the abundance of *Acacia giraffae* (*Acacia erioloba*).

The Matlolakgang (site 2) is located between Malwelwe and Ngware villages in Kweneng district. The livelihoods of these communities include both livestock and arable farming, but most land is used for grazing. This site is part of the Kalahari sandvelt ecosystem and dominated by non-calcareous sandy soils. The mean annual rainfall is 395 mm and ranges between 134 and 658 mm with high CV of 33.9 %. It rains mainly between October and March (Figure 1.3b). Its vegetation is described as Central Kalahari Bush Savanna (Weare and Yalala, 1971) dominated by *Megaloprotachne albescens* (APRU, 1979).

The Xanagas (site 3) is located between Karakubis and Charleshill in Gantsi district. This particular rangeland is used by different tribal communities such as Baherero, Bakgalagadi and Basarwa. Cattle and goat production is the dominant activity in the area (Thomas and Twyman, 2004) as arable farming is limited by poor rainfall and soil fertility. The mean annual rainfall (Figure 1.3c) normally ranges from 128 to 647 mm with high inter-annual variability as shown by 42.5 % coefficient of variation (CV). The Xanagas site is part of the Kalahari sandveld dominated by calcareous soils (APRU, 1979). The vegetation is generally classified as Northern Kalahari Tree and Bush Savanna (Weare and Yalala, 1971) and herbaceous vegetation is dominated by *Stipagrostis uniplumis*.

Figure 1.2: The locations of the three study sites in Botswana (Matlolakgang is located between Malwele and Ngware)

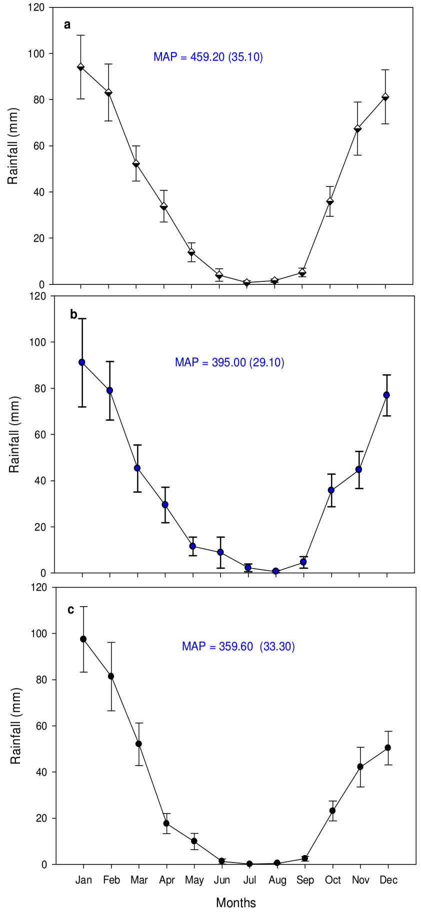

Figure 1.3: Mean annual precipitation (MAP)(SE) and distribution over the year at Goodhope (a), Matlolakgang (b) and Xanagas (c) for the period 1998 to 2008 (collected by Department of Agricultural Research) .

1.3.3 Data collection

This study uses an observational approach (Sagarin and Pauchard, 2010) to seek the patterns on savanna ecosystem dynamics in relation to rangeland management systems. This approach allows wide-scale observation of processes and provides excellent insights into the temporal and spatial scale of any environmental problems (Sagarin and Pauchard, 2010). A manipulative experimental approach was not possible because of time limitation. The current study used multiple methods or indicators including structured interviews on local ecological knowledge to assess rangeland dynamics (Reed *et al.*, 2011), which will be discussed in detail in each chapter. Nevertheless, general information on transects setup and stocking rates are presented below.

To achieve a representative sampling, a stratified sampling scheme was used (e.g., Waite, 2000). Each ranch and communal rangeland was approximately divided into four blocks. In ranches, this was achieved by pairing neighboring paddocks and using a coin toss to select one for our study. In communal land, transects were set close to the ranch in each direction where possible to minimize differences due to change in climatic conditions (e.g. rainfall and soils). In each block, the starting point of each transect was randomly established using a predetermined distance estimated by vehicle odometer and where possible transects run from east to west except for three at Matlolakgang communal lands which were oriented north to south to avoid them being parallel to the road. Transects were well spaced (at least 1 km) to account for spatial variability in vegetation and avoid pseudo-replication (e.g., Waite, 2000). A total of 4 transects each measuring 300 m were setup within each management system at all three sites between April and May 2009. But, the fourth transect in Goodhope communal rangelands was abandoned to avoid possible confounding effect of arable farming. All (23) transects were set at least 50 m perpendicular (away) from fence or road to avoid non-grazing disturbance in both communal and ranching lands. Each transect position was marked by Global Positioning System (GPS) and their locations are as shown in Table 1.1. A transect was adopted for this study because it minimizes the variation caused by topography, vegetation type and soil type at local level (Dube and Pickup, 2001). The rainfall data was collected from records kept by Department of Agricultural Research. Rangeland assessments were carried out in 2009 and 2010 towards the end of the rainy season (March to May).

Table 1.1: The positions of transects located at various study sites

Rangeland area	Management system	Transect 1	Transect 2	Transect 3	Transect 4
Xanagas	Ranch	S 22°09.099’ E022°16.684	S 22°08.987’ E020°17.886	S 22°10.360’ E020°16.668	S 22°10.302’ E020°17.842
	Communal	S 22°15.350’ E020°12.817	S22°14.032’ E020°13.752’	S 22°04.998’ E020°31.131	S 22°06.078’ E020°31.016
Matlolakgang	Ranch	S 23°53.988’ E025°19.650	S 23°53.628’ E 025°18.931	S 23°54.082’ E025°17.580	S 23°54.779’ E025°18.604
	Communal	S 23°57.147’ E025°18.912	S 23°54.867’ E025°22.262	S 23°55.472’ E025°20.610	S 23°55.294’ E025°18.337
Goodhope	Ranch	S25°32.034’ E025°28.557’	S25°31.313’ E025°27.585’	S25°30.840’ E025°28.851	S25°30.299’ E025°27.402’
	Communal	S25°30.656’ E025°29.971’	S25°28.377’ E025°30.412’	S25°29.205’ E025°25.683’	

1.3.4 Stocking rates and herbivore type

The stocking rates for ranches were estimated for 2009 using April animal weights. The animals' weights were converted to Livestock Units (LSU), which is equivalent of 450 kg (Abule *et al.*, 2007). The stocking rate was then calculated by dividing ranch size (ha) by total LSU. Goodhope had mixed stock therefore sheep number was multiplied by factor of 0.2 (Manitoba, 2004) and goat by 0.1. Our estimated stocking rates were 8.17, 9.22 and 16.55 ha/LSU for Goodhope, Matlolakgang and Xanagas ranch respectively. Our estimated stocking rates were comparable to stocking rates for year 1999 to 2001 and were all combined to estimate the mean stocking rates shown in Figure 1.4. It was difficult to estimate stocking rates for communal grazing lands because there was no defined boundary for use by any particular community. The district area used for communal grazing was therefore estimated by subtracting known size area for other land uses (arable farming, wildlife and conservation and ranching) from total area of the district based on Central Statistics Office (CSO) report (Central Statistics Office, 2008a). Livestock numbers from the Annual Agricultural Survey Report of 2006 (latest report) and 1995 were then used to estimate stocking rates. There were no specific weights, so we assumed all cattle were equivalent to 1 LSU, Sheep (0.2), goat (0.1), donkey (0.6) and horse (1.2) (Manitoba, 2004). Our mean estimated stocking rates for communal rangelands were as shown in Figure 1.4 and are generally lower than potential carrying capacity of each site. The potential carrying capacity refers to the maximum possible stocking of herbivores that can be supported by the rangeland on a sustainable basis (FAO, 1988). Matlolakgang and Xanagas have potential carrying capacity of 09-12 ha/LSU, while Goodhope is indicated as 05-08 ha/LSU (See Appendix 1). The ranches used in this study had been grazed for over 30 years as shown by reports (APRU, 1979), and it is assumed communal grazing had been occurring seasonally or regularly for longer periods.

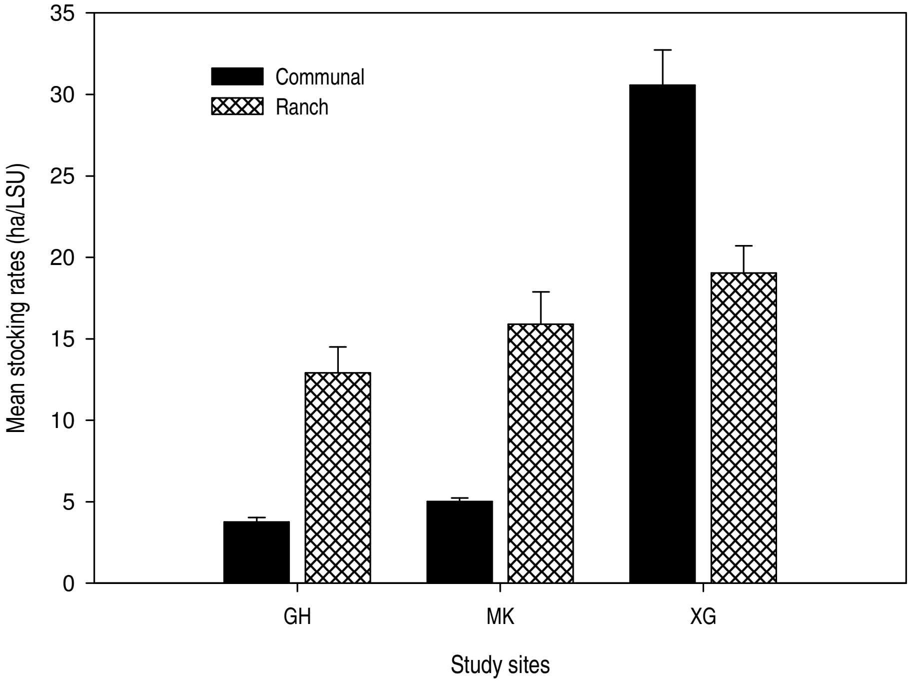

Figure 1.4: Stocking rates in communal and ranching lands at Goodhope (GH), Matlolakgang (K) and Xanagas (XG)

Chapter 2

Soil dynamics in response to rangeland management systems and environmental conditions in semi-arid, Botswana

2.1 Introduction

The savanna ecosystem productivity is dependent on biological processes of soil resources (Anderson and Ingram, 1989; Marzaioli *et al.*, 2010). Soils serve multiple functions in an ecosystem such as sustaining biological productivity, maintaining environmental quality like nutrients recycling and supporting plant and animal health (Schoenholtz *et al.*, 2000; Moussa *et al.*, 2008). The quality of soil is therefore determined by properties and processes that allow it to function effectively as part of a healthy ecosystem (Schoenholtz *et al.*, 2000). Thus, soil fertility is a good indicator of rangeland degradation (Cobo *et al.*, 2010) and a good means of comparing the effect of different land uses on natural ecosystems. Soil degradation is characterized by a decrease in soil quality that is indicated by soil compaction, poor water infiltration rate (Chartier *et al.*, 2011) and lower organic matter content (Snyman and du Preez, 2005). Soil degradation threatens the sustainability of savanna ecosystem and livestock production systems especially in developing countries like Botswana, which are dependent on natural ecosystems for forage supply. Improved knowledge of the effects of disturbance, especially grazing management, on interactions between aboveground and belowground biota is essential to determine the sustainability of land use (Klumpp *et al.*, 2009).

The effects of grazing management on physical and chemical soil properties are mediated by other variables such as rainfall (Mandal *et al.*, 2010) and thus detectable after decades (Hiernaux *et al.*, 1999; Liebig *et al.*, 2006). Savanna ecosystems grazed at different intensities like communal and private (ranch) grazing lands (Parsons *et al.*, 1997) are likely to undergo contrasting shift in vegetation composition, and primary production (Klumpp *et al.*, 2009; Dunne *et al.*, 2011). This eventually leads to change in soil quality (e.g. carbon storage) by altering soil organic matter and nutrient cycling through litter quality and decomposition (Klumpp *et al.*, 2009). High grazing intensities also destroy soil structure and promote soil compaction through trampling and reduced vegetation cover (Castellano and Valone, 2007; Geissen *et al.*, 2009) especially during wet conditions (Fernández *et al.*, 2010). Compacted soils have high bulk density and reduced permeability to water (du Toit *et al.*, 2009), which in turn lead to increased runoff and soil erosion. The increased water runoff eventually limits growth of grass seedlings and leads to changes in vegetation patterns (Chartier *et al.*, 2011).

Grazed land in tropical Mexico has been observed to be acidic with a pH value below 5.5 (Geissen *et al.*, 2009). In South Africa, soil acidification was largely observed in communal grazing lands, particularly around water-points where it could be attributed to urine deposition (Smet and Ward, 2006). High grazing intensities have also been shown to reduce soil carbon content because of decline in plant cover, which leads to reduced organic inputs (Lal, 2004; Rutherford and Powrie, 2010) and also by increasing soil carbon loss through soil erosion. Soil organic carbon responds slowly to changes in land use and is therefore a good indicator of the sustainability of particular land use systems (Walker and Desanker, 2004). As discussed in Chapter 1, the communal grazing lands in Botswana and elsewhere are supposedly overgrazed relative to private ranches, and therefore are expected to have low soil carbon.

Sustainable rangeland use sequesters soil carbon and improves the resilience of the agro-ecosystem (Lipper *et al.*, 2010). Carbon sequestration could therefore increase ecosystem productivity and also contribute towards combating land degradation (Bationo *et al.*, 2007) and climate change adaptation (Lipper *et al.*, 2010). However, the soil organic carbon in savanna ecosystems is highly heterogeneous (Lal, 2004). This is largely attributed to grazing intensity (Elmore and Asner, 2006) and distribution of woody vegetation (Silver *et al.*, 2010), although in drier savannas the contribution of herbaceous vegetation to the carbon pool is greater relative to that in wet savannas (Wang *et al.*, 2009).

In arid environments like Botswana, soil organic carbon (SOC) is lower than 2 % both on surface and subsurface soils and is positively associated with rainfall (Ringrose *et al.*, 1998; Bird *et al.*, 2004; Lal, 2004). Thus, rainfall variability may influence spatial and temporal pattern in carbon storage and loss (Wang *et al.*, 2007; Silver *et al.*, 2010). Soil organic carbon is also influenced by silt and clay particle contents, particularly clay as it stabilizes organic carbon (Walker and Desanker, 2004; Bationo *et al.*, 2007). Despite a growing interest in understanding the carbon stocks (Walker and Desanker, 2004; Schulp and Verburg, 2009), there is limited knowledge of soil organic carbon sequestration on rangelands (Follett and Reed, 2010) especially in relation to grazing management systems. Yet, rangelands cover an extensive land area in the world (Follett and Reed, 2010) and could therefore play a significant role in carbon sequestration (Lal, 2004; Silver *et al.*, 2010) and mitigate against climate change (Rees *et al.*, 2005).

Agricultural potential of soil is dependent on soil fundamental indicators such as pH, organic carbon, and clay content (Gray *et al.*, 2009). Soil indicators of rangeland degradation reflect long-term effects of grazing on ecosystem and are more reliable than vegetation as they are not responsive to rainfall variability (Smet and Ward, 2006). For that reason, it is essential to understand the relationship between key soil properties with both management systems and environmental indicators. Improved knowledge supports proper management systems, which promote soil productivity and ecosystem sustainability (Haghighi *et al.*, 2010). This study was therefore undertaken to evaluate the changes in soil properties as an indicator of rangeland degradation in response to rangeland management systems in savanna ecosystem. Specifically, the objectives were to investigate; (i) soil physical properties and pH in relation to communal and ranching management systems (ii) soil organic carbon between management system (iii) determinants of soil organic carbon and (iv) interrelationship between soil key properties with environmental indicators.

2.2 Methods and materials

2.2.1 Study sites

The study was conducted in Goodhope, Matlolakgang and Xanagas rangelands located in different ecological regions of Botswana. Detailed descriptions of the three sites are presented in Chapter 1. Matlolakgang and Xanagas rangelands are part of the Kalahari sandveld, while Goodhope rangelands are located in the hardveld.

2.2.2 Soil sampling and laboratory analysis

Soil samples were collected in April/May of 2009 and 2010. Along each transect described in Chapter 1, soil samples were collected using a hammer type core sampler close to the centre of 1 x 1 m quadrats positioned systematically at 100 m intervals. The soil samples were labelled and transported to the Department of Agricultural Research’s Soil and Plant Analysis Research Support (SPARS) Laboratory located in Sebele, Botswana. The samples were thereafter air-dried in shallow trays (except bulk density) and soil lumps gently crashed by hand. The soils were then sieved through a 2 mm mesh sieve to remove unwanted materials (e.g. roots and rocks) and homogenize the sample before being analyzed for physical and chemical properties. Each soil sample was given laboratory number for easy identification.

In 2010, 69 soil samples were collected across different savannas for measuring both pH and soil texture. Soil pH was measured to characterize soil acidity (Geissen *et al.*, 2009) and was determined in a 0.01m calcium chloride ($CaCl_2$) solution (Robertson *et al.*, 1999; Aarrestad *et al.*, 2011). From each sample, 20 g of soil was weighed and placed in a 100 ml plastic bottle. 50 ml of calcium chloride solution was then added to each bottle and shaken in a horizontal position for two hours. The mixture was allowed to stand for 30 minutes, before a calibrated pH meter was used to measure soil pH by immersing the electrode in the upper part of the sample suspension.

The soil particle size (texture) analysis (PSA) was carried out by the pipette method (Gee and Bauder, 1987; Marzaioli *et al.*, 2010). The dispersed soil particles take different times to settle in a l L soil suspension according to their size (Medinski *et al.*, 2010) and subsequently Strokes' Law is used to extract sub-samples of the soil samples at given depth after specified settling times for each size fraction (Gee and Bauder, 1987). In this study, a 50 g of dried soil from each soil samples were added to the 200 ml bottles, followed by addition of 100 ml of extracting solution. The mixtures were then shaken horizontally for 30 minutes, and thereafter extracts were transferred into 1 L plastic measuring cylinder, which were then filled to 1 L mark by distilled water. The soil-water suspension was stirred and 5 ml of suspensions were pipetted at 10 cm depth and dried in aluminium cups at 105 °C. The extract in measuring cylinder were allowed to stand for 2 hours and another 5 ml of suspensions were pipette at 10 cm depth and dried at 105 °C.

The soil bulk density was determined in both 2009 (n = 46) and 2010 (n=46). On each transect soil samples were collected from two quadrats located at 100m and 300 m, using a hammer type core sampler (metal cylinder) gently driven into the soil until the top was level with soil surface (du Toit *et al.*, 2009). The core sampler was removed carefully to ensure that there was no soil compaction and loss (Elliott *et al.*, 1999). In 2009 water was carefully poured on each area selected for sampling because the Kalahari soil is sandy and loose when dry. This was not necessary in 2010 because the soil was wet enough from rain. The soil cores were transferred to labelled plastic bags or kept on core samplers before being taken to the laboratory. The soil samples were then transferred into tins, oven-dried at 105 °C for 24 hours and weighed. The bulk density (g/cm^3) was calculated as the dry soil mass divided by the volume of 100 cm^3 (Anderson and Ingram, 1989; Elliott *et al.*, 1999).

Separate soil organic samples from depths of 0-100 mm (n = 46) and 110-300 mm (n = 46) were collected using an auger in both 2009 and 2010. For each transect, soil samples of the same sampling depth were bulked and kept in labelled (transect name & soil depth) plastic bags to avoid contamination. The soil organic carbon was determined using the dichromate oxidation method of Walkey & Black (Anderson and Ingram, 1989; Meersmans *et al.*, 2009). 2.00 g of soil was weighed from each of the samples then placed into a labelled (e.g. sample number) 500 ml conical flask. Control flasks were also prepared without soils to provide a standard curve by pippetting 0, 0.5, 0.75, 1.0, 1.5, 2.0, 2.0 and 3.0 ml of glucose standard solution. 10 ml of potassium dichromate solution was then added to all flasks, both samples and standards, followed by 20 ml of concentrated sulphuric acid (H_2SO_4). Finally, the flasks were gently shaken on an asbestos board under a fume hood. After the samples had settled for 1 hour, 200 ml of superflock solution was added to each flask. This was swirled to mix thoroughly and then left overnight to allow particles to settle. Finally, the light absorbance of the clear solutions in all flasks, both samples and standards, were read with a spectrophotometer at 620 nm wavelength. This method is commonly used to determine SOC (Ringrose *et al.*, 1998) though it is known to underestimate the soil organic carbon due to incomplete oxidation of the recalcitrant fraction of organic carbon (Meersmans *et al.*, 2009). In our case, it was the only available method for determining soil carbon at Sebele laboratory.

2.2.3 Statistical analysis

The data were analysed at different spatial scales (large-scale vs local scale). Large-scale analysis refers to the use of all available data regardless of site to provide a general picture of rangeland conditions in Botswana. The local scale refers to site-specific analysis to determine variations influenced by local condition (Detsis, 2010). There was more replication for variables that were measured in 2010 only to account for variability across the landscape. However, during analysis each transect was used as a sample unit such that means for samples collected in each transect was used for analysis. A General Linear Model (GLM) was applied to data to investigate the relationship between soil properties and the multiple variables such as management systems, sites, and rainfall at a large scale. At local level, one-way analysis of variance (ANOVA) was used for comparative analysis of soil organic carbon, pH, soil particles, and bulk density in relation to rangeland management systems. Stepwise multiple regression analyses were used to identify the “best” subset of predictors that explain soil organic variation. The threshold during variables selection on stepwise regression was set at p-value of 0.05 such that all variables above that were eliminated.

Principal components analysis (PCA) was used to investigate the relationships between several environmental variables such as rainfall, soil properties and vegetation conditions in different savanna ecosystems. The PCA was based on a correlation matrix and only components that had eigen-value greater than one were retained (Quinn and Keough, 2002). Correlation coefficient was further used to test the interrelationships between key soil properties with environmental indicators. Fitted regression was used to illustrate the relationship between soil organic carbon and clay content. For each test, normality of data was tested through histogram and normal probability plot (Hair *et al.*, 1998). All statistical analyses were performed using Minitab (15) statistical package.

2.3 Results

2.3.1 Soil physical properties and pH in relation to communal and ranching management systems

Soil particle sizes differed significantly between sites ($P<0.001$), but not rangeland management systems ($P>0.05$). Goodhope soils had higher clay ($F=64.2$, $R^2=0.66$) and silt ($F=31.6$, $R^2=0.49$) content than other sites ($P<0.001$), while Matlolakgang and Xanagas were dominated by sandy soils ($F=66.7$, $R^2=0.67$, $P<0.001$). At local level, the Goodhope rangelands showed marked differences in sand ($F=4.89$, $R^2=0.20$) and silt ($F=5.56$, $R^2=0.23$) content between communal and ranching management systems ($P<0.05$), but not clay content ($F=1.41$, $R^2=0.07$, $P>0.05$) (Table 2.1). The soil particle sizes at Matlolakgang and Xanagas rangelands were not significantly different between management systems ($P>0.05$).

The soils at all sites were acidic and their pH narrowly ranged between 4.5 and 5.5 (Table 2.1). The soil pH was considerably influenced by site effects ($F=12.6$, $P<0.001$), but not management system ($F = 0.05$, $P>0.05$). The site effects accounted for 28% of soil pH variation, and Matlolakgang rangelands had the lowest pH (4.64 ±0.08) compared to the Goodhope (5.28 ± 0.18) and Xanagas (5.33 ± 0.06) soils ($P<0.001$). Comparative analysis at local scale also failed to show significant pH differences between communal and ranching systems ($P > 0.05$).

Soil bulk densities were stable across all savanna ecosystems as shown by low coefficient of variation (CV = 2.0 - 8.5 %). Our results (general linear model) indicated that soil bulk density differed between management systems ($F=6.19$, $P<0.05$), but not sites ($F=1.08$, $P>0.05$). The management system and site effects were responsible for 9.21 % (R^2) of bulk density variability. The communal grazing land (1.57 gm^{-3}) generally had higher bulk density than ranching land (1.53 gm^{-3}) ($F=6.65$, $R^2= 0.07$, $P<0.05$). However, comparative analysis of soil density between communal and ranching lands at local scale failed to show any significant differences ($P>0.05$) at all sites (Table 2.1).

Table 2.1: Average soil bulk density, particle sizes and pH by management systems at different sites

Site	Management system	Particle size distribution (%)			pH	Bulk density (g cm^3)
		Sand	Clay	Silt		
Goodhope	communal	82.93 ± 1.13 *	9.33 ± 0.59	7.73 ± 0.74 *	5.02 ± 0.27	1.56 ± 0.03
	ranch	79.12 ± 1.23 *	10.58 ± 0.81	10.30 ± 0.76 *	5.45 ± 0.23	1.50 ± 0.03
Matlolakgang	communal	88.50 ± 0.63	5.75 ± 0.07	5.75 ± 0.66	4.75 ± 0.13	1.57 ± 0.03
	ranch	88.66 ± 0.45	5.78 ± 0.07	5.56 ± 0.47	4.52 ± 0.06	1.54 ± 0.02
Xanagas	communal	90.03 ± 0.44	6.00 ± 0.10	3.97 ± 0.42	5.43 ± 0.08	1.58 ± 0.01
	ranch	89.88 ± 0.67	6.13 ± 0.16	3.98 ± 0.72	5.23 ± 0.08	1.55 ± 0.01

*indicates significant differences ($P<0.05$) between communal and ranch land

2.3.2 Soil organic carbon in relation to communal and ranching management system

Soil organic carbon in both surface and subsurface soils was highly variable as indicated by coefficients of variation ranging between 17.0 and 63.4 %. The soil organic carbon (SOC) in surface soil (< 10 cm) was not significantly different between years (F=0.32, P>0.05) or rangelands management systems (F=3.29, P>0.05). Rather, surface soil organic carbon strongly varied between sites (F= 1.16, P<0.001), with Goodhope rangelands having the highest SOC (Figure 2.1a). At local level, soil organic carbon content (%) was significantly higher under the ranching system than communal grazing system at Goodhope (F= 7.21, R^2=0.38, P<0.05), but no statistical significant differences were observed between management systems at other sites (P> 0.05). The soil organic carbon was also not significantly different in relation to soil depth (upper and lower layer) (P > 0.05). Subsurface soil organic carbon varied most strongly between sites (F = 25.38, P<0.001), but not management systems (F=1.38, P>0.05) and year (F=0.00, P>0.05). Comparative analysis at local level failed to show any significant variations (P>0.05) of subsurface soil organic carbon between communal and ranching rangelands (Figure 2.1b).

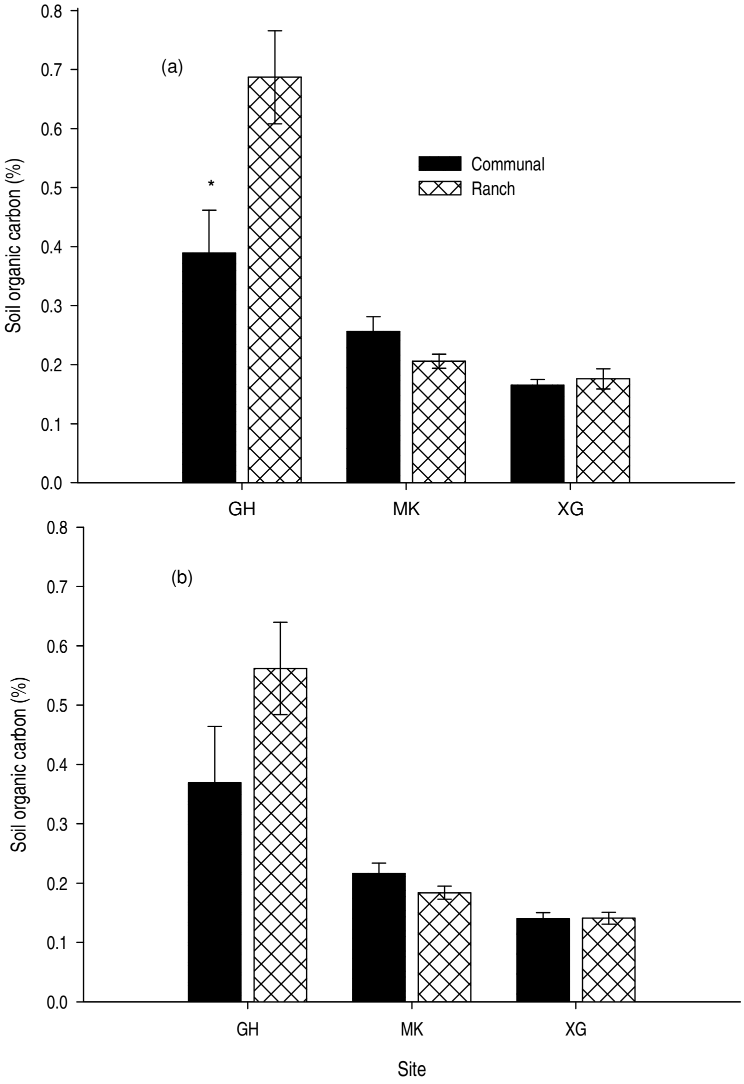

Figure 2.1: Soil organic carbon in the upper (a) (0-10 cm) and lower (b) (11-30) soil layers under different management systems at Goodhope (GH), Matlolakgang (MK) and Xanagas (XG). (* $P < 0.05$)

2.3.3 Determinants of soil organic carbon

Simple regression indicated that rainfall had a substantial relationship with soil organic carbon ($P<0.001$) and accounted for 44 % (R^2) of the variability in soil organic carbon. However, stepwise regression with seven possible predictors (including rainfall, species richness, stocking rates, herbaceous biomass, litter biomass, soil clay content and soil pH) indicated that soil organic carbon was strongly related with clay content and herbaceous aboveground biomass. The combined effects of clay (%) and above ground biomass explained 83.96 % of the variation in soil organic carbon. However, the main source of SOC variation was associated with clay fraction, which accounted for 80.35 % of SOC variability ($P<0.001$) (see Figure 2.2), while herbaceous biomass accounted for only 3.61 % of SOC variability ($P<0.05$). Soil organic carbon increased with increasing soil clay fraction, and there was higher soil carbon and clay variability observed at Goodhope (Figure 2.2) than at Matlolakgang or Xanagas.

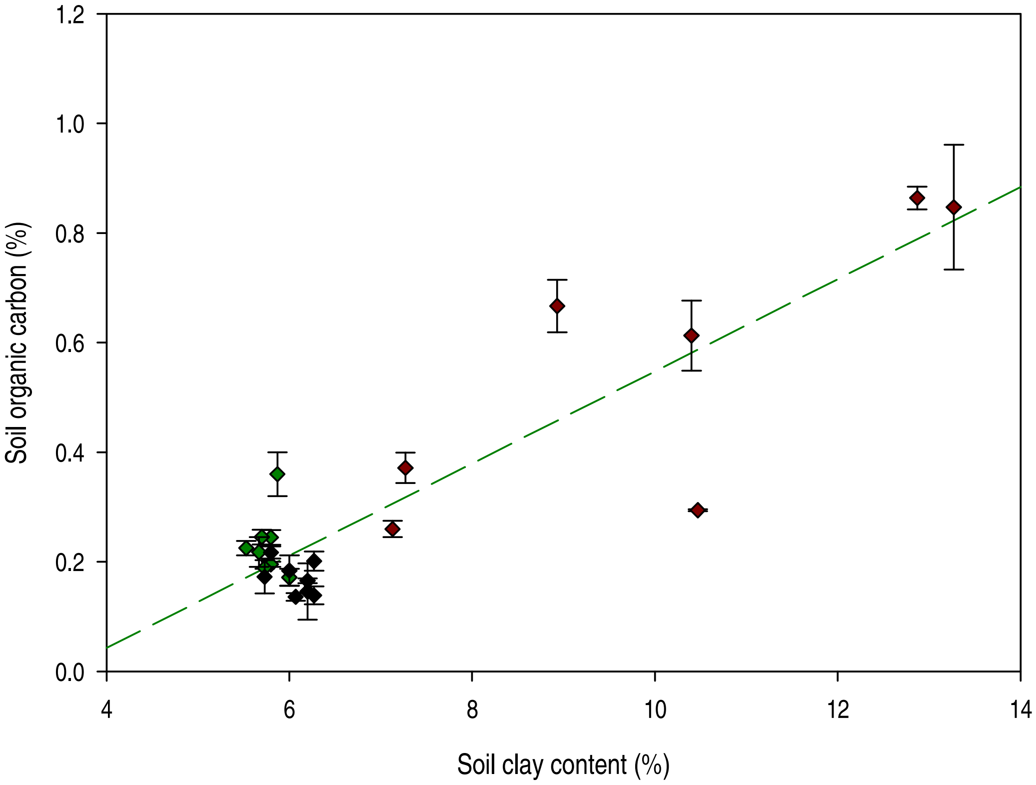

Figure 2.2: Relationship between soil clay content and soil organic carbon in rangelands grazed by livestock in Botswana (♦ Goodhope, ♦ Matlolakgang, ♦ Xanagas)

2.3.4 Interrelationships between soil properties with environmental indicators

Principal component analysis (PCA) including soil properties (texture, soil carbon, and pH), herbaceous vegetation (species richness, biomass), woody vegetation (cover, species richness) and rainfall variability indicated that the first three principal components (PC) accounted for 83.2 % of variability within the savanna ecosystems. PC1, which thereafter is referred to as rangeland health condition, explained 58.4 % of ecosystem variability (Figure 2.3). Soil characteristics particularly soil organic carbon (0.38), clay (0.38) and silt (0.37) content improved the rangeland health condition (PC1), while sand fractions contributed negatively (Figure 2.3). Rainfall and herbaceous biomass also contributed positively to rangeland health condition (PC1). PC2 explained 14.5 % of savanna ecosystem variability of which soil pH had the highest loading value (0.51). The third component accounted for 10.3 % of ecosystem variability.

The interrelationships between environment variables that characterize savanna ecosystems are illustrated in the loading plot of PCA (Figure 2.3) and correlation matrix (Appendix 2). Environmental variables that contributed positively to good rangeland condition (PC1) were positively correlated with each other (Figure 2.3). Rainfall was positively correlated to several variables such as soil organic carbon ($r = 0.66$, $P < 0.001$), silt ($r = 0.77$, $P<0.001$), herbaceous aboveground biomass ($r = 0.46$, $P<0.05$) and soil clay fractions ($r = 0.61$, $P<0.005$). Similarly, other environmental variables such as sand fraction, vegetation species richness and woody cover that contributed negatively to rangeland condition (PC1) were also closely correlated. Soil sand fraction had strong negative correlation with both soil organic carbon ($r = 0.93$, $P<0.001$) and clay fractions ($r = 0.93$, $P<0.001$). Herbaceous species richness had a negative association with SOC content ($r = 0.29$, $P<0.05$). The aboveground herbaceous biomass was independently influenced by different factors such as rainfall and soil particle size.

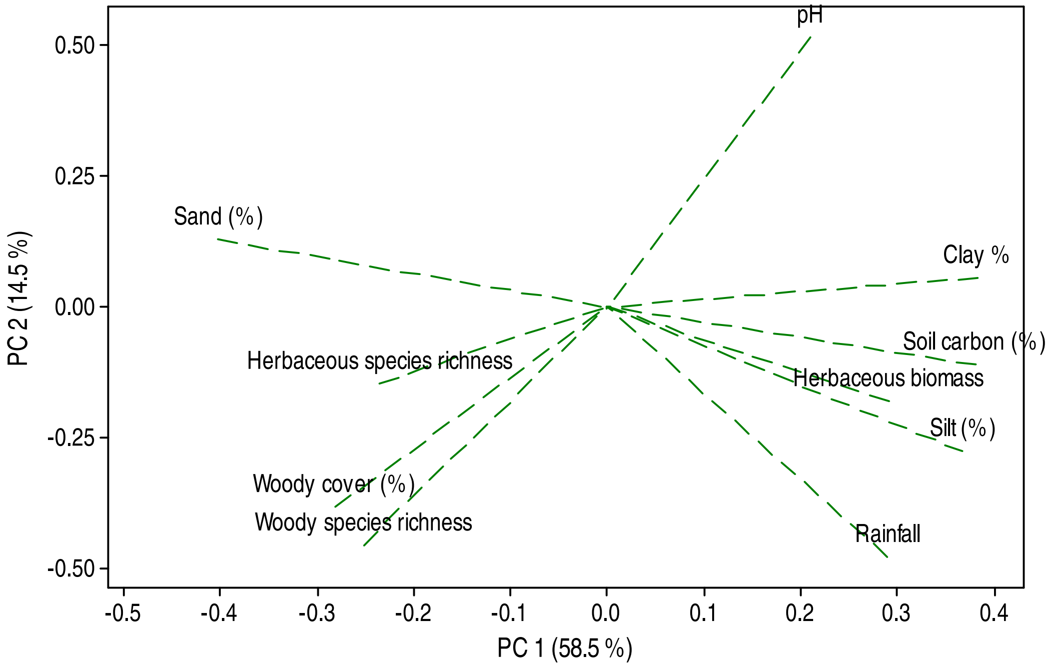

Figure 2.3: PCA showing interactions between different environmental elements (e.g. soils, vegetation and rainfall characteristics) across savanna ecosystems of Botswana

2.4 Discussion

2.4.1 Soil physical properties and pH in relation to communal and ranching management systems

The soil particle sizes differed between study sites (large scale), but not rangeland management systems (local scale). Based on the textural triangle for soil textural analysis (Gee and Bauder, 1987), Goodhope rangelands soils are loamy sand due to high clay content while Xanagas and Matlolakgang rangelands are characterised as sandy soils. High sand fractions observed in Xanagas and Matlolakgang soils are partly from Kalahari sand and bedrock (Cole and Brown, 1976) though sand contents observed in this study were slightly lower than the > 95 % sand recorded in another recent study in the Kalahari (Wang *et al.*, 2007). The results of the present study suggest that livestock grazing in communal and ranching have limited influence on soil textural properties, especially the clay content which is influential on stabilizing of organic molecules and micro-organisms (Amato and Ladd, 1992), despite variations in stocking rates between the two management systems (chapter 1). This is consistent with a study by du Toit *et al.*, (2009), which also found that stocking intensity did not influence the soil particle size distribution in South African rangelands. Contrary to our finding, other authors (Tessema *et al.*, 2011) have observed that grazing pressure in semi-arid savanna of Ethiopia had an influence on soil texture. The differences in effects of grazing management on soil properties could be due to variations in animal type (e.g. size) or climate between study sites.

The soils in all grazing rangelands investigated in current study in Botswana were acidic ranging between 4.5 and 5.5, which is consistent with previous studies in Botswana (Wang *et al.*, 2007) and other tropical soils (Geissen *et al.*, 2009). Matlolakgang rangelands were more acidic than other rangelands, which could be attributed to the proximity of bedrock to the surface at this site compared with the other sites (Wang *et al.*, 2007) or higher livestock grazing intensity which leads to high urine deposition (Smet and Ward, 2006). However, despite varying between sites, soil pH was not statistically different between management systems. This suggested that differences in grazing intensities between communal and ranching grazing lands had no or similar impact on soil acidity especially that no significant differences had been observed on soil pH between grazed and ungrazed rangelands in South African (Moussa *et al.*, 2008). But, the results of this research contradict observation that soil pH responded positively to increased grazing pressure reported in other studies (Sasaki *et al.*, 2008; Zemmrich *et al.*, 2010) and differences could be due to variations on climatic conditions and stocking rates between study sites.

The range of soil bulk density (1.4 to 1.62 gm^3) observed in this study is in agreement with levels reported in other studies in Botswana (Bird *et al.*, 2004; Wang *et al.*, 2007). Large-scale analyses indicated that livestock grazing had different impacts on soil bulk density between communal and ranching management system. This observation is consistent with other studies that reported increased soil bulk density and compaction in grazing lands due to increased stocking rates (Steffens *et al.*, 2008; du Toit *et al.*, 2009) caused by hoof traffic of grazing animals (Walker and Desanker, 2004). However, comparison at each site did not show strong management effects on soil bulk density, except at Xanagas where high bulk density was observed in communal rangelands than ranching land in 2010, but not 2009. Yet, Xanagas communal rangelands had lower grazing intensity than the ranch, and therefore it is possible that bulk density variation between managed grazing lands could have been temporarily due to moisture content in the soil, which can lead to soil shrinking or swelling (Elliott *et al.*, 1999). Soil compaction, reflected by high bulk density, is associated with rangeland degradation (Snyman and du Preez, 2005) in situations when the soil is not well protected by above-ground production and there is considerable trampling by animals. In this study, it is not apparent that any one management system has more degrading effect on savanna ecosystems as indicated by lack of significant differences on soil bulk density between management systems.

The lack of significant impact of different grazing intensities between communal and ranching systems in this study could be due to dominance of sandy soils (Hiernaux *et al.*, 1999) or short wet period.

2.4.2 Soil organic carbon in relation to communal and ranching management system

The amount of soil organic carbon observed in the present study was low particularly in rangelands located in the Kalahari sandveld region, but highly variable spatially especially between Goodhope and other sites. Other studies (Schulp *et al.*, 2008; Lal, 2009; Wang *et al.*, 2009) also reported high spatial SOC heterogeneity in drier savanna ecosystems, which supports our findings. The amount of organic carbon in soils at Xanagas and Matlolakgang are comparable with those of other studies within Botswana (Ringrose *et al.*, 1998; Wang *et al.*, 2007) and nutrient-poor savannas (Scholes, 1990). However, the Walkey & Black method used to determine soil organic carbon in the present study is known to underestimate soil carbon, and therefore could have partly contributed to low observed values (Sollins *et al.*, 1999; Meersmans *et al.*, 2009). Despite low soil carbon concentration, the savanna ecosystems could still contribute significantly towards carbon cycle and sequestration if managed properly because they cover a large land area (Silver *et al.*, 2010).

There were no statistically significant SOC differences between top and surface soils even in well drained soils at Xanagas and Matlolakgang, which is consistent with reports by other authors (Hopmans *et al.*, 2005), but contradict observation of others (Bird *et al.*, 2004; Shrestha *et al.*, 2004; Silver *et al.*, 2010). Rather, there was a strong site effect on soil organic carbon. The Goodhope sites exhibited higher soil organic carbon than other sites and the variation could be attributed to several factors such as rainfall, and clay content (Girmay *et al.*, 2008), whereas differences at local scale could be attributed to management systems (Schulp *et al.*, 2008).

In general, soil organic carbon did not differ significantly between communal and ranching management systems. This was further evident at Matlolakgang and Xanagas rangelands where no pronounced differences in SOC were observed between communal and ranching grazing lands. Other studies in Southern Africa (Tefera *et al.*, 2010) and Ethiopia (Tefera *et al.*, 2007) also found no significant organic carbon differences between livestock grazing management lands, which is consistent with our observation especially in sandveld area. Additionally, other authors (Gass and Binkley, 2011) found no significant differences in soil carbon between grazed and ungrazed plots in Colorado (USA), which confirms lack of disparity between grazing management systems characterized by different grazing intensities.

Conversely, soil organic carbon was significantly higher in ranching lands than communal grazing lands at Goodhope in the current study, which could reflect the impact of grazing intensity. The low grazing intensity observed in Goodhope ranch leads to lower rates of vegetation defoliation causing biomass build up and increases in soil carbon (John *et al.*, 2006; Savadogo *et al.*, 2007; Steffens *et al.*, 2008). Subsequently, it could be expected that impact of different grazing intensities under different management systems will be indirectly reflected in SOC content, especially at local scale, by regulating the vegetation composition (Girmay *et al.*, 2008) and organic matter input and output to the soil (Lawrence, 2005; PiÑEiro *et al.*, 2006; Bagchi and Ritchie, 2010). In this study, this was particularly observed only at Goodhope, but not other two sites in Kalahari sandveld. The variations on carbon response to grazing between the study sites could suggest that local environments such as soil type (Aarrestad *et al.*, 2011) could be playing a key role in regulating soil carbon and not management systems. The communal and ranching management systems either have minimal or similar effects on savanna ecosystems, which then nullifies the assumption of Tribal Grazing Land Policy that ranching will reduce rangeland degradation (Botswana Goverment, 1975).

2.4.3 Determinants of soil organic carbon

The SOC variability is influenced by biotic and abiotic factors like local climate, fauna diversity, land use and management (Walker and Desanker, 2004). In this research, soil carbon in topsoil substantially increased with increasing rainfall across the savanna ecosystems. Goodhope savannas exhibited the highest SOC and mean annual rainfall than other sites. Rainfall alone accounted for moderate (44 %)

variation in soil carbon, which is supported by results from other studies in Botswana (Ringrose *et al.*, 1998; Bird *et al.*, 2004) and elsewhere (Adler *et al.*, 2005; Wheeler *et al.*, 2007). However, multivariate analyses indicated that SOC variation in topsoil was largely explained by soil texture, particularly clay fraction as suggested by other authors (Bird *et al.*, 2004). This is because the cohesive properties of clay particles protect and stabilize carbon and therefore increase the amount of carbon stored in the soil (Walker and Desanker, 2004). The clay content also contributes towards soil water retention capacity (Zemmrich *et al.*, 2010), which improves savanna terrestrial ecosystem production (John *et al.*, 2006) and indirectly improves soil carbon through increased organic matter in the form of litter input to the soil.

Goodhope rangelands in the clayey hardvelt generally had high biomass production, surface litter and organic carbon, which could highlight the importance of clay content in regulating the soil carbon. The increase in soil carbon inputs is dependent on increase in above ground biomass and organic matter recycled to the soil (Follett and Reed, 2010) explaining why Goodhope had higher SOC than other sites. The results of this research concurs with observation in other studies (Bationo *et al.*, 2007). In this study, we could not separate the impact of rainfall and clay content on soil organic carbon because they were highly correlated. Small fluctuations in the textural properties of topsoil are known to have a large effect on SOC (Bationo *et al.*, 2007), supporting our observation of higher organic carbon retention with increased clay content, but variation in rainfall influences the input of carbon to the soil from increased biomass and therefore increased litter.

2.4.4 Interrelationships between soil key properties with environmental indicators

The analysis of results in this study showed many interactions between the soil properties and other environmental variables. This increases risk of collinearity and makes it difficult to separate the effect of individual environmental variables on ecosystem productivity (Quinn and Keough, 2002). Meanwhile, the PCA results showed that soil properties play a significant role on variability across the savannas and that soil organic carbon, texture and pH are interrelated with herbaceous plant diversity and productivity. Soil organic carbon is strongly correlated to clay content, but has a negative relationship with species richness. High soil fertility, indicated by soil with high carbon and clay fraction (Scholes, 1990; Harrison and Bardgett, 2010), enhances high growth of competitive herbaceous species resulting in the loss of less dominant species. As a result, plant diversity (species number) tends to decrease with increasing productivity (Gross *et al.*, 2009; Partel *et al.*, 2010), which could explain why Goodhope rangelands had lower plant diversity than other sites.

In this study, sandy environments favoured high plant diversity, which is consistent with observations by other authors (Devineau and Fournier, 2007). The negative relationship between herbaceous plant diversity (species richness) and soil carbon sequestration provide a conservation challenge because management goals could either be diversity conservation or carbon sequestration and not both. This presents a management challenge for effective diversity conservation and mitigation of climate change through carbon sequestration though other studies have reported positive species richness and carbon sequestration (Midgley *et al.*, 2010).

The aboveground herbaceous biomass was influenced by both mean annual rainfall and soil properties such as SOC, and soil size particles. The positive relation between rainfall and aboveground herbaceous biomass was expected because soil moisture is known as the limiting factor in arid and semi-arid environments (Walker *et al.*, 1981; van Langevelde *et al.*, 2003). The increase in rainfall would in turn enhance herbaceous plant growth as confirmed in this study and others (Zhou *et al.*, 2009). Increased herbaceous biomass lead to increased plant litter, which contributed to organic matter in the soil and improved soil organic carbon especially in arid ecosystems (Wang *et al.*, 2009). Improved soil fertility indicated by soil organic carbon in turn promoted high growth rate of herbaceous vegetation. In the present study, herbaceous biomass responded positively to increasing organic carbon or *vice versa* as suggested by other authors (John *et al.*, 2006), which suggested that management systems that promote carbon sequestration could also improve livestock productivity and economic return for pastoral farmers. This was evident at Goodhope where the rangeland under ranching management was characterized by higher organic carbon and herbaceous biomass than communal rangeland. It is suggested that rainfall control the duration of grass production in dry savannas, whereas soil fertility modulates the growth rate during the productive season (Scholes, 1990) which supports our findings that both rainfall and soil organic carbon affect herbaceous biomass positively.

2.5 Conclusion

Soil texture and pH differed significantly between sites, but not between communal and ranching lands. This indicated that livestock grazing in communal and ranching lands either have no or similar impact on soil properties. The soil organic carbon was also not affected by management systems, rather it was strongly affected by environmental factors particularly soil clay fractions. Herbaceous biomass and rainfall also had a minor influence on soil organic carbon. PCA results indicated that health condition of savanna ecosystems is dependent on interactions between several environmental factors. However, that soil properties are strongly associated with variability on savanna ecosystems as illustrated by its moderate relationship with herbaceous biomass. The relationships between soil properties and herbaceous vegetation and bush encroachment will be further investigated in chapter 3 and 4 respectively.

Chapter 3

Herbaceous vegetation responses to rangeland management systems and environmental variables across semi-arid savannas, Botswana

3.1 Introduction

The vegetation composition in savanna ecosystems is a function of land use, management and environmental suitability determined by rainfall and soil attributes (O' Connor, 1994; Muhumuza and Byarugaba, 2009). Yet, rangeland degradation and loss of plant diversity is a common problem in savanna ecosystems (Scholes and Biggs, 2005). Vegetation diversity is essential for rural livelihood and its loss could make the rural economy vulnerable to poverty (Sallu *et al.*, 2009) because loss of diversity causes decline in the ecosystem's ability to maintain its natural production capacity (Jones-Walters, 2008). Herbaceous plant species, especially perennial grasses, are primary fodder for cattle and their loss reduces the carrying capacity of any particular rangeland. This in turn threatens livestock production, especially of cattle (Wigley *et al.*, 2009), which is a dominant economic activity in savanna ecosystems of Botswana (Masike and Urich, 2008).

In the context of a range-succession model, degradation of savanna ecosystems is attributed to high livestock stocking rates particularly in communal grazing lands (Vetter, 2005). Overgrazing leads to high defoliation of palatable herbaceous plants

(Mphinyane *et al.*, 2008; Tefera *et al.*, 2010) and weakens their competitive ability against other species (Abule *et al.*, 2005). This eventually leads to loss of palatable herbaceous plant species, which are replaced by unpalatable plant species (Adler *et al.*, 2005) and a high frequency of bare ground (Diaz *et al.*, 2007). Bare grounds are susceptible to erosion (Fatunbi and Dube, 2008) and compaction due to trampling by animals (Batey, 2009) which further enhances the recruitment of unpalatable plants species like annual grasses, forbs (Hayes and Holl, 2003) and woody vegetation (Kraaij and Milton, 2006). Annual grasses and forbs have a poor and erratic herbage yield in response to rainfall variability and their increase is particularly associated with rangeland degradation (Smet and Ward, 2005). Bush encroachment also reduces herbaceous productivity through shading and competition for resources (Hagos and Smit, 2005). These effects of livestock grazing on vegetation dynamics in arid environments are challenged by the non-equilibrium concept, which suggests that rainfall is the main driver of rangeland dynamics (Westoby *et al.*, 1989). It is proposed that livestock grazing in drylands have minimal effect on vegetation dynamics (Ellis and Swift, 1988), especially if the grazing occurs after seed production.

Rangeland degradation and loss of vegetation diversity are not always directly related (Detsis, 2010). The loss of vegetation diversity in rangelands is usually blamed on overgrazing by livestock (Oba *et al.*, 2001), although grazing lands in Ethiopia support high plant diversity (Mohammed and Bekele, 2010). Moderate livestock grazing pressure has been shown to promote high plant species diversity in arid rangelands by maintaining biomass below 400-500 g m^{-2} which is considered to be optimal for plant diversity conservation (Oba *et al.*, 2001). The impact of livestock grazing on plant diversity is also regulated by resource availability, such that grazing increases plant richness in areas of high primary productivity (Osem *et al.*, 2002). In arid environments, plant growth and diversity is limited by soil moisture and nutrients (Olff and Ritchie, 1998; Osem *et al.*, 2002) and individual plant species responses to grazing are different (Peper *et al.*, 2011). The extent of savanna ecosystems in Botswana provides an opportunity for plant diversity conservation, but requires proper understanding of diversity patterns and their determinants especially in dominant land uses (Chazdon *et al.*, 2009).

A detailed knowledge of herbaceous species composition and distribution patterns in relation to different management systems is therefore essential for sustainable management of savanna ecosystems and livestock production (O' Connor, 1994; Mohammed and Bekele, 2010). However, there have been few studies in this area of research (Tefera *et al.*, 2007), and the results are not consistent. Studies conducted in South Africa, Swaziland and Ethiopia suggest that ranching systems promote high abundance of palatable herbaceous species, while unpalatable plants like the grass *Aristida congesta* are largely observed in communal lands (Smet and Ward, 2005; Tefera *et al.*, 2007; Tefera *et al.*, 2008b). In contrast, no significant difference in the cover of annual herbaceous plants was found between communal and ranching rangelands in Botswana (Dahlberg, 2000b). Rather, there was higher abundance of the unpalatable *Aristida congesta* under ranching systems than on communal grazing land (Dahlberg, 2000b). The lack of consistency in these studies could be partly attributed to variations in grazing intensity and the heterogeneity of savannas due to climate variation and soil types. Previous studies are based on small and localised landscapes, which are not reliable for making inferences about large-scale vegetation dynamics in heterogeneous ecosystems (Smet and Ward, 2005; Vetter, 2005). In addition, most studies tend to focus on vegetation change along a grazing gradient, which does not reflect the general range condition. It is essential to understand the herbaceous vegetation dynamics in response to management systems whilst also taking into account the spatial and environmental variability of the rangelands themselves (Tomimatsu and Ohara, 2009).

The aim of this part of the study is to investigate herbaceous vegetation dynamics in relation to dominant rangeland management systems under different environmental conditions in the savanna ecosystem of Botswana. The objectives were to;

i. Characterise the distribution of herbaceous species between communal and ranching grazing systems.
ii. Compare the herbaceous plant biomass between communal and ranching grazing systems.
iii. Compare the herbaceous species diversity between rangeland management systems.
iv. Identify the relationships between herbaceous vegetation attributes and other environmental variables across the savanna ecosystems.

3.2 Methods and material

3.2.1 Study sites

Data were collected in three sites located in different agro-ecological regions of Botswana during 2009 and 2010. The Goodhope, Matlolakgang, Xanagas ranches and their surrounding communal grazing areas were selected. The sites represent variation in biophysical characteristics in terms of rainfall, soil types and dominant vegetation types. A detailed description of these study sites and transects layout is outlined in Chapter 1.

3.2.2 Data collection

Vegetation composition was assessed towards the end of rainy season (April-May) of 2009 and 2010, which coincides with the flowering period of most herbaceous species (Abule *et al.*, 2007) making their identification easy. The Tidmarsh wheel-point method (Everson *et al.*, 1990; Brockett, 2001) was used to make individual point observations, where the point of the spike hit the ground (Bonham, 1989) along each transect (Figure 3.1). The wheel-point method reduces biased aiming by the observer compared with the point-frame method (Bonham, 1989). The wheel had two markers leading to point observations at 1 m intervals and at each point, the nearest herbaceous plant was recorded.

Figure 3.1: The wheel-point used to count herbaceous plant species along a transect

All herbaceous plant species hit with a point along transect were identified and recorded. Individual species were further classified in terms of life-forms (i.e. annuals and perennial), and their palatability (i.e. desirable, intermediate and poor) was determined from literature (Field, 1976; van Oudtshoorn, 2002). All non-graminoid non-woody plants were classified as forbs (Jacobs and Naiman, 2008). Bare ground and litter hits were also recorded giving a total of 300 hits per transect. The individual species counts were then used to calculate percent cover as;

(Number of point intercept/total points) x 100 (Bonham, 1989)

The herbaceous plant species were classed as dominant (≥ 15 %), common (<15-5 %), less common (<5-1 %) and rare (< 1 %) (Tefera *et al.*, 2010) based on their mean percent cover calculated from the transects. The herbaceous species were further categorized as decreaser, increaser I and increaser II in relation to their ecological response to grazing (Dyksterhuis, 1949; Trollope *et al.*, 1989; du Plessis *et al.*, 1998).

The increaser I species are herbaceous plants such as *Schimidtia pappophoroides* that increases in underutilized rangelands, while Increaser II species are those herbaceous species that tend to increase in overgrazed rangelands such as *Aristida congesta* (Trollope *et al.*, 1989; du Plessis *et al.*, 1998). The decreaser species are herbaceous species that tend to decrease when overgrazed such as *Digitaria eriantha* (Trollope *et al.*, 1989).

The herbaceous diversity was measured using species richness (R) and Simpson's Index (D') of diversity in 2009 and 2010. Species richness is the total number of species present in that particular habitat, which could also be referred to α-diversity (Waite, 2000). Simpson's Index of diversity was calculated as follows:

$$(D') = 1\text{-}D, \quad \text{where} \quad D = \sum (p_i)^2 \quad \text{and} \quad p_i = n_i / N$$

n_i is the number of individuals of species i and N is the total of individuals in the sample (Waite, 2000; Lamb *et al.*, 2009). Simpson's Index of diversity has a range of 0-1 where 1 represent maximum diversity.

The herbaceous biomass measurements (n = 69) were carried out in 2010 using the clipping method (Bonham, 1989). The aboveground herbaceous biomass was clipped at soil level (Osem *et al.*, 2002) in 1 m² quadrats located at regular intervals at 100 m, 200 m and 300 m along each transect. The clipped standing herbaceous and surface litter biomass were kept separately in labelled paper bags and transferred to the laboratory for analysis. The standing biomass and litter samples were oven-dried at

60 ºC for 48 hours and then weighed to determine dry-matter biomass (Gross *et al.*, 2009).

3.2.3 Statistical analysis

Each transect (23) was used as the observational unit during the analysis (Bonham, 1989). Pearson correlations between herbaceous plants attributes such as palatability, life form, ground cover and other variables like rangeland management systems were performed to explain the distribution of species. One-way analysis of variance was used to compare various herbaceous vegetation variables (e.g. palatability, life form, and biomass) between communal and ranching management system. The t-test was avoided because multiple t-tests increase possibility of Type I error rate (Hair *et al.*, 1998). Principal components analysis (PCA) was used to investigate the relationships between herbaceous vegetation attributes and environmental variables in different savanna ecosystems. The PCA was based on a correlation matrix and only components that had eigenvalue greater than one were retained (Quinn and Keough, 2002). The general linear model (GLM), multiple regressions and stepwise regression were used to assess the relationship between herbaceous diversity-productivity and other explanatory variables such as environmental factors such rainfall, management systems and soil properties (Grafen and Hails, 2003).

3.3 Results

3.3.1 Characteristics of herbaceous vegetation in communal and ranching rangelands

Herbaceous vegetation compositions differed between sites and grazing management at local scale (see Appendix 3). Rangelands at all sites were dominated by one or two herbaceous species, except Matlolakgang communal land, which had no dominant herbaceous species. There was a strong negative correlation between perennial and annual herbaceous species ($r = 0.874$; $P<0.001$) but rangelands at all sites were dominated by perennial herbaceous plant species with few annuals. The communal and ranching grazing lands in Goodhope both had high (90.7 and 92.1 % respectively) cover of perennial herbaceous vegetation ($P>0.05$). Matlolakgang ranch had higher cover of perennial herbaceous species (84.9 %) than the communal grazing land (66.2%) ($P<0.05$) and the opposite was observed at Xanagas, where communal rangeland had higher (75.3 %) cover of perennial species than the ranch (63.5 %) ($P<0.05$) (Table 3.1). Annual herbaceous cover only differed significantly between Matlolakgang communal rangelands (19.1 %) and the ranch (7.5 %) ($p<0.05$).

Increase in grass cover resulted in decrease in bare ground ($r = 0.779$; $P<0.001$) as shown by high (> 95 %) grass cover recorded at Goodhope communal and ranch lands, which also had low (< 1%) bare ground cover. There was more bare ground under communal rangelands than ranches at Goodhope and Matlolakgang ($P<0.05$) but not at Xanagas ($P>0.05$) (Table 3.1). Rangelands with high bare ground (%) also had high forb cover ($r = 0.494$; $P<0.001$), as evident at Matlolakgang rangelands (Table 3.1). The communal grazing lands at Xanagas had significantly higher percent forbs cover than the ranch ($P<0.005$), but no significant differences in forbs were observed between management systems at Goodhope and Matlolakgang ($P>0.05$). Litter cover was high within all Matlolakgang rangelands, and almost devoid at Goodhope, but a significant difference in litter cover was recorded between communal and ranch lands at Xanagas ($P<0.005$) (Table 3.1).

The percent cover of decreaser species differed significantly between sites ($R^2=0.21$; $P<0.01$). The decreaser species cover was significantly higher under the ranch management system than on communal rangelands at both Matlolakgang and Xanagas ($P<0.05$), but not Goodhope ($P>0.05$) (Table 3.1). Goodhope rangelands had higher cover of increaser I species than other sites ($R^2=0.30$; $P<0.01$), but pronounced differences in cover of increaser I species between management systems were observed only at Xanagas ($P<0.001$). The distribution of increaser II species also differed between sites ($R^2=0.30$; $P<0.001$), and were higher under communal land management than on ranches at Matlolakgang and Xanagas ($P<0.05$).

The general linear model (GLM) analysis showed that distribution of desirable (palatable) herbaceous plant species differed significantly between sites ($F=20.7$; $P<0.001$), but not between rangeland management systems ($F=2.29$; $P>0.05$).

Goodhope communal and ranch lands were both dominated by desirable (very palatable) herbaceous plants ($P>0.05$), but other sites had high cover of unpalatable (poor) species (Figure 3.2). The Xanagas ranch had higher cover of desirable herbaceous species than communal land ($P<0.005$) (Figure 3.2). Communal lands in all three study sites had higher cover of intermediate palatable species than the ranches ($P<0.05$). The cover of unpalatable (poor) herbaceous species was not statistically different between communal and ranching lands at all sites ($P>0.05$) (Figure 3.2).

Table 3.1: Mean (± SE) percent ground cover, lifespan and ecological status of herbaceous layer in communal range and ranches at different sites

	Goodhope		Matlolakgang		Xanagas	
	Communal	Ranch	Communal	Ranch	Communal	Ranch
Ground cover (%)						
Bare ground	0.95 ± 0.36 *	0.17 ± 0.09	10.54 ± 3.32 *	2.58 ± 1.02	3.42 ± 1.62	6.08 ± 1.42
Forbs	1.46 ± 0.72	0.29 ± 0.19	20.96 ± 3.83	16.96 ± 5.41	20.04 ± 3.23 **	7.37 ± 1.28
Grass	97.44 ± 1.04	98.54 ± 0.78	64.33 ± 6.94	75.46 ± 6.61	76.25 ± 4.37	81.50 ± 1.95
Litter	0.17 ± 0.11	1.00 ± 0.53	4.17 ± 1.99	5.00 ± 1.71	0.29 ± 0.13 ***	5.04 ± 0.94
Lifespan (%)						
Annuals	8.22 ± 3.59	6.75 ± 3.77	19.12 ± 3.47 **	7.50 ± 1.51	20.96 ± 3.22	25.33 ± 4.67
Perennials	90.67 ± 3.75	92.08 ± 3.57	66.17 ± 7.23 *	84.92 ± 3.85	75.33 ± 3.05 *	63.54 ± 3.33
Ecological Status (%)						
Decreasers	38.72 ± 3.67	41.00 ± 5.46	14.54 ± 4.15 *	30.75 ± 5.80	28.87 ± 5.19*	43.29 ± 4.19
Increaser I	20.61 ± 6.68	17.62 ± 2.60	6.29 ± 1.91	7.58 ± 1.56	1.17 ± 0.48 ***	14.92 ± 1.76
Increaser II	40.67 ± 8.02	41.38 ± 6.95	79.17 ± 4.62 *	61.67 ± 5.80	69.96 ± 5.38 **	41.79 ± 4.26

Significant differences between communal and ranch management systems indicated by: *<0.05 , ** <0.01 ,***<0.001

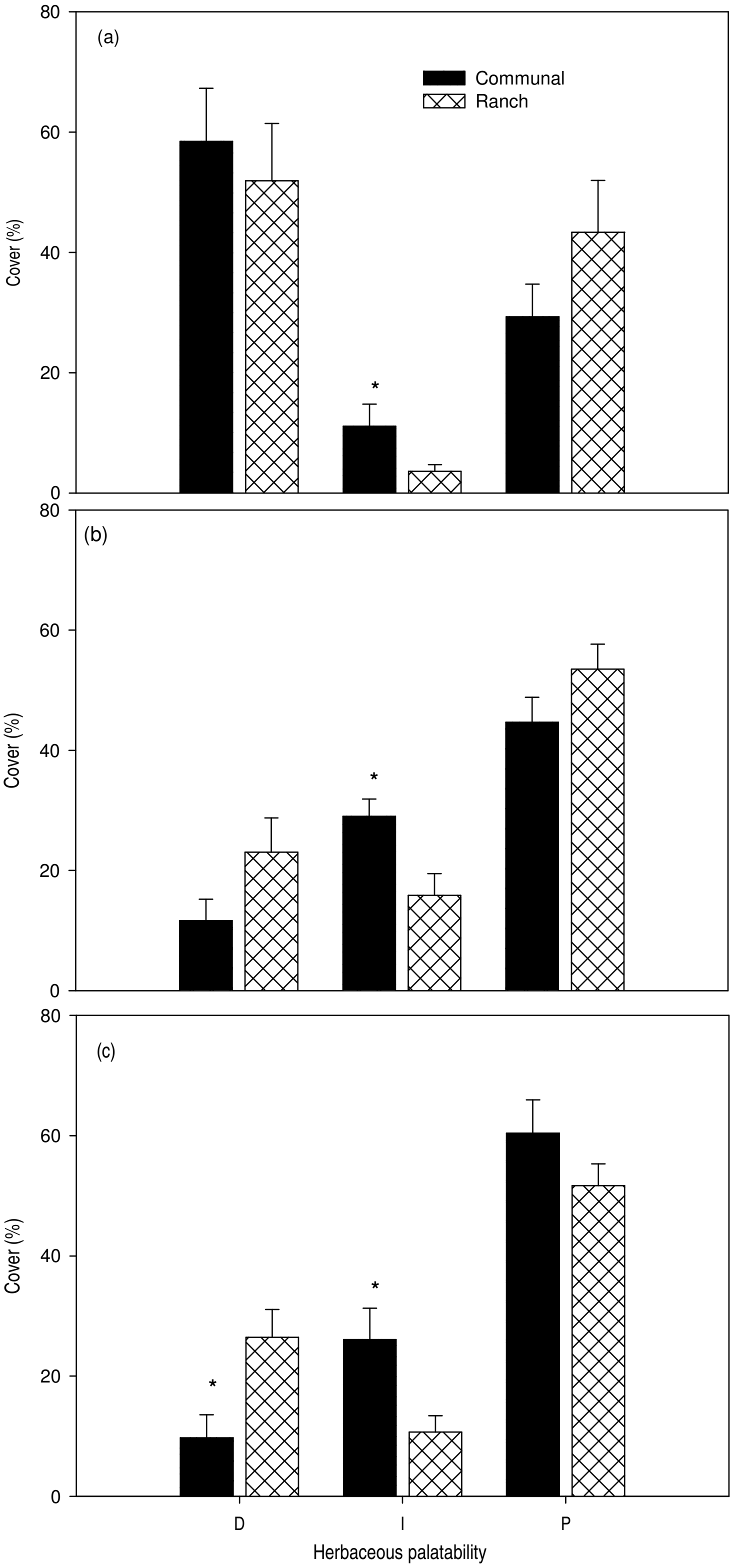

Figure 3.2: Comparison of mean percent cover (±SE) of desirable (D), intermediate (I), and poor (P) herbaceous vegetation on communal and ranch land at Goodhope (a), Matlolakgang (b) and Xanagas (c). *P <0.05 between communal and ranching management systems

3.3.2 Herbaceous biomass in relation to rangeland management systems

Herbaceous standing biomass was highly variable as shown by coefficient of variation (CV) ranging between 27 and 67 %. General linear model showed that herbaceous biomass was significantly different between sites (F=11.4; $P<0.001$) and rangeland management systems (F=14.4; $P<0.001$). Goodhope rangelands had higher herbaceous biomass (126 gm^{-2}) than both Matlolakgang (65.3 gm^{-2}) and Xanagas (65.3 gm^{-2}) rangelands (F=10.6; R^2=24.3; $P<0.001$). Higher herbaceous biomass was generally observed under ranching systems (106.2 gm^{-2}) than communal rangelands (59.9 gm^{-2}) (F=12.9; R^2=16.2; $P<0.01$). The ranches at Goodhope (F=25.0; R^2=0.568; $P<0.001$) and Matlolakgang (F=7.0; R^2=0.24; $P<0.05$) had significantly higher herbaceous biomass than their respective communal grazing lands (Figure 3.3a). However, the herbaceous biomass recorded in communal and ranching systems at Xanagas did not differ significantly (F=0.23; $R^2 = 0.01$; $P>0.05$).

The variability in surface litter biomass was weakly ($R^2 = 0.43$) accounted for by site effects (F=4.0; $P< 0.05$) and management systems (F =40.1; $P<0.001$). Surface litter biomass was significantly higher in ranches (31.29 ± 3.64gm^{-2}) than communal grazing lands (6.76 ± 1.28 gm^{-2}) (F=37.7; R^2=0.36; $P<0.001$) across all sites. Comparative analysis between management systems at local scale also confirmed that surface litter mass was higher under ranches than communal land at all sites ($P<0.05$) (Figure 3.3b).

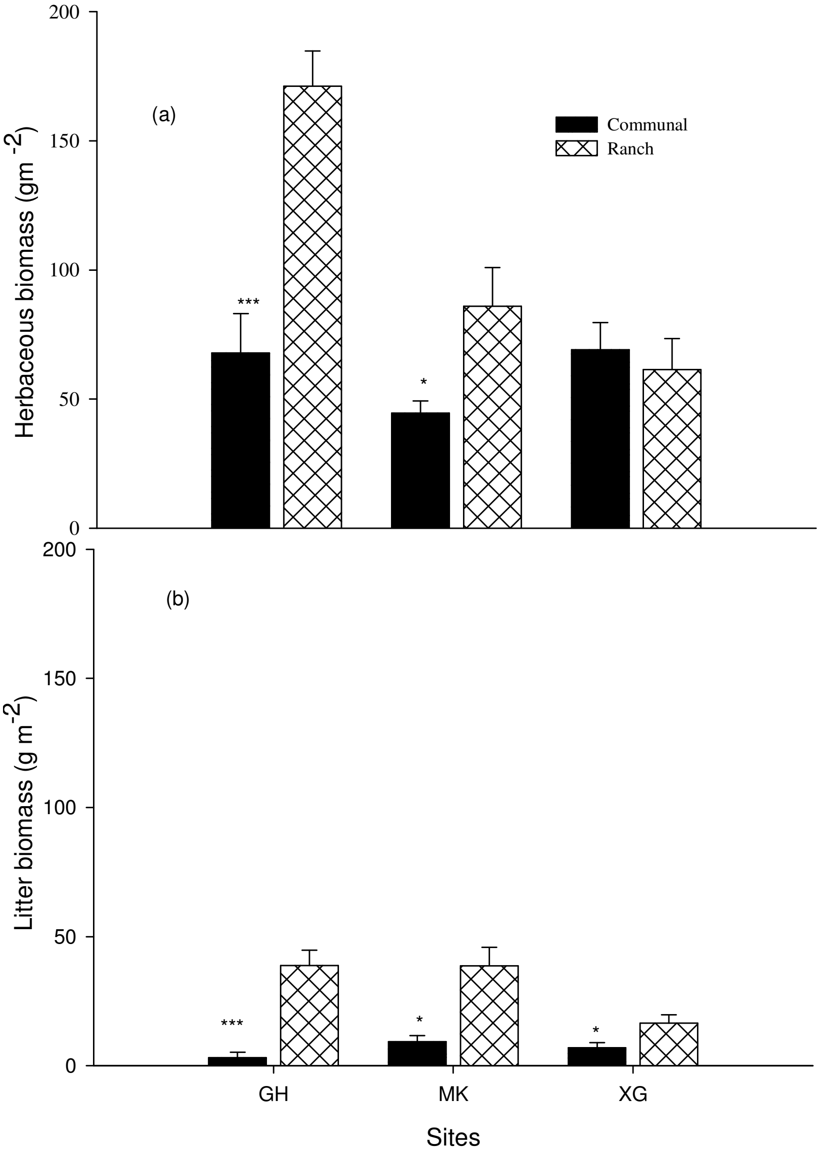

Figure 3.3: Comparison of (a) herbaceous and (b) litter (± SE) biomass between management systems at Goodhope (GH), Matlolakgang (MK) and Xanagas (XG). * p <0.05, *** P < 0.001

3.3.3 Herbaceous plant diversity in relation to management systems

General linear model showed that species richness was significantly different between sites (F=12.2; P<0.001), but not between management systems (F=0.3;

P>0.05). Goodhope rangelands had lower species richness (14) than both Xanagas (18) and Matlolakgang (20) (F=12.3; $R^2=0.364$; P<0.001). One-way ANOVA showed that species richness did not differ between communal and ranching rangelands at all sites (P>0.05) (Figure 3.4). The Simpson's index of diversity suggested that herbaceous plant diversity was generally high (0.7 - 0.87) at all sites and significantly varied between sites (F=13.7; P>0.001), but was not affected by management system (F=0.1; P>0.05). Matlolakgang rangelands had higher herbaceous diversity (0.86 ± 0.01) than other sites (F=14.0; $R^2=0.39$; P<0.001). Simpson's index of diversity showed no significant differences between communal and ranching lands at all sites (P>0.05) (Figure 3.5).

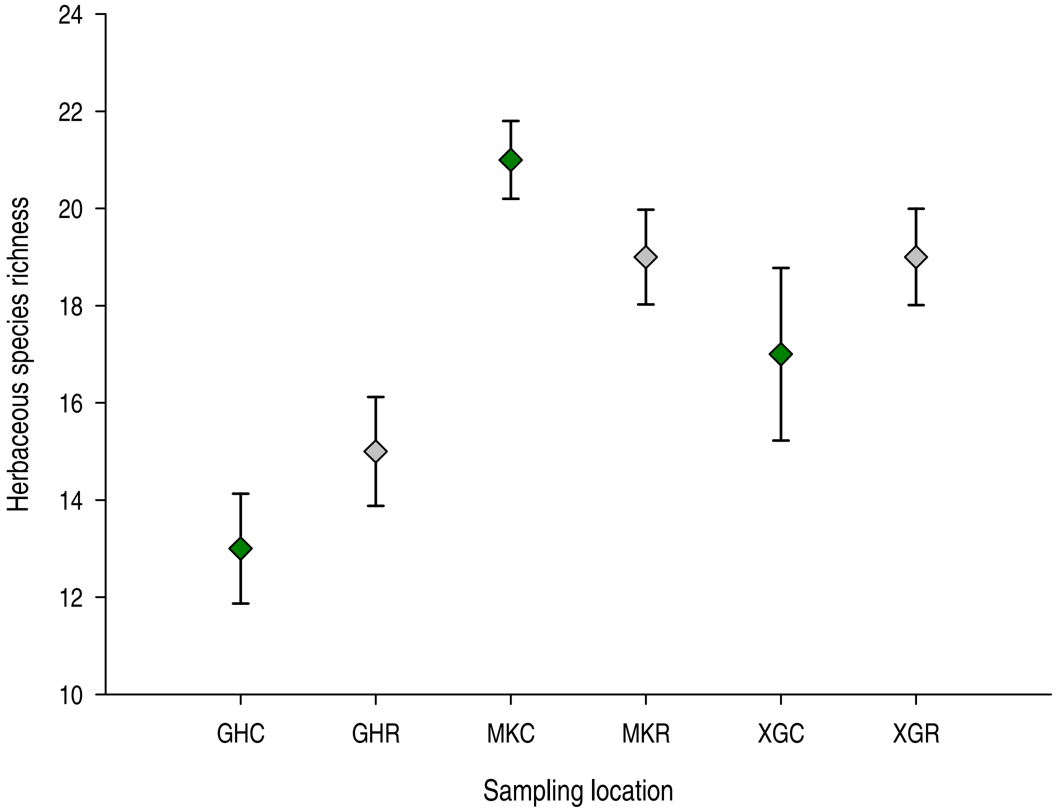

Figure 3.4: Species richness (±SE) in communal (C♦) and ranching(R ♦) grazing lands at Goodhope (GH), Matlolakgang (MK), and Xanagas (XG)

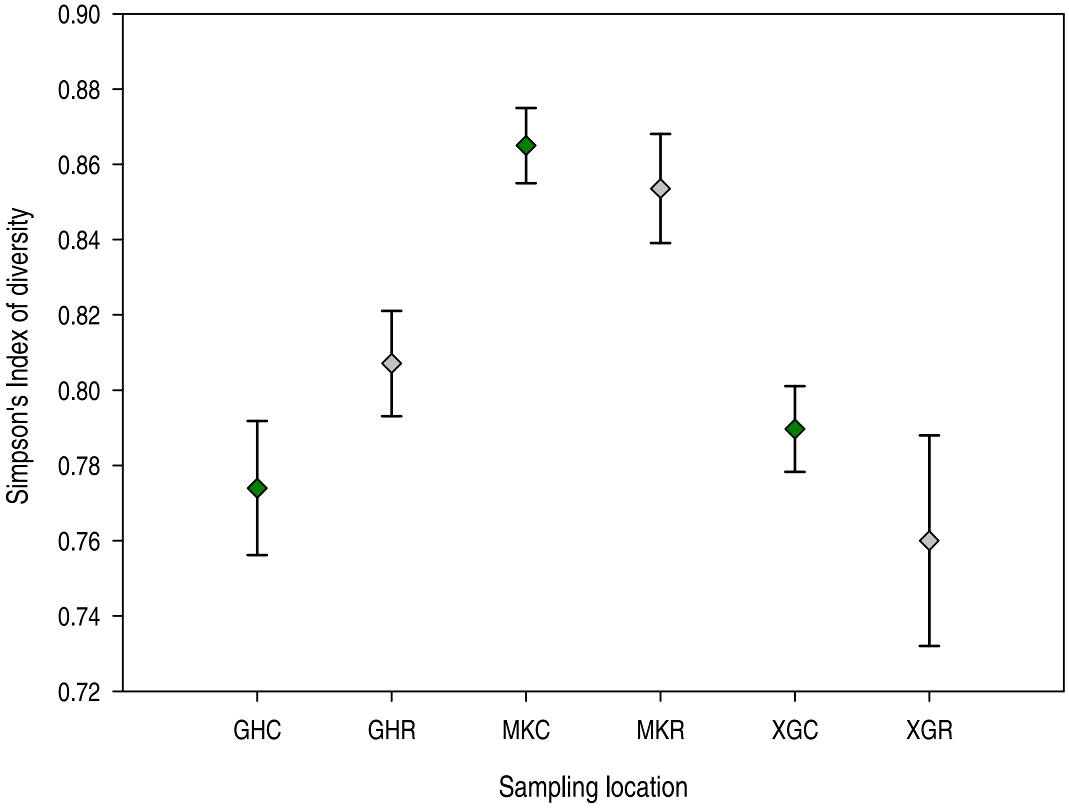

Figure 3.4: Simpson's index of diversity (± SE) in communal (C ♦) and ranching (R♦) grazing lands at Goodhope (GH), Matlolakgang, and Xanagas (XG)

3.3.4 Relationships between herbaceous vegetation attributes and other environmental variables

The principal component analysis including ground cover (grass, bare ground, litter), lifespan (annual, perennial), biomass, palatability, species richness and rainfall showed that variability of herbaceous vegetation compositions across the savannas was largely (81.5 %) explained by four principal components (PC). The first 2 components explained 61.6 % of variability of herbaceous vegetation dynamics (Figure 3.6). Rangeland degradation (PC 1) was largely characterized by increases in bare ground (0.314), forbs (0.335), annual plants (0.298) and species richness

(0.246). Other variables such as increased cover of perennial herbaceous plants, grasses, desirable species, herbaceous biomass and increased rainfall contributed negatively to rangeland degradation (PC1) (Figure 3.6). As shown in Figure 3.6, there were several interactions between herbaceous vegetation attributes. Species richness was closely correlated to forbs and annual herbaceous species, while herbaceous biomass was correlated with perennial herbaceous species and grass presence.

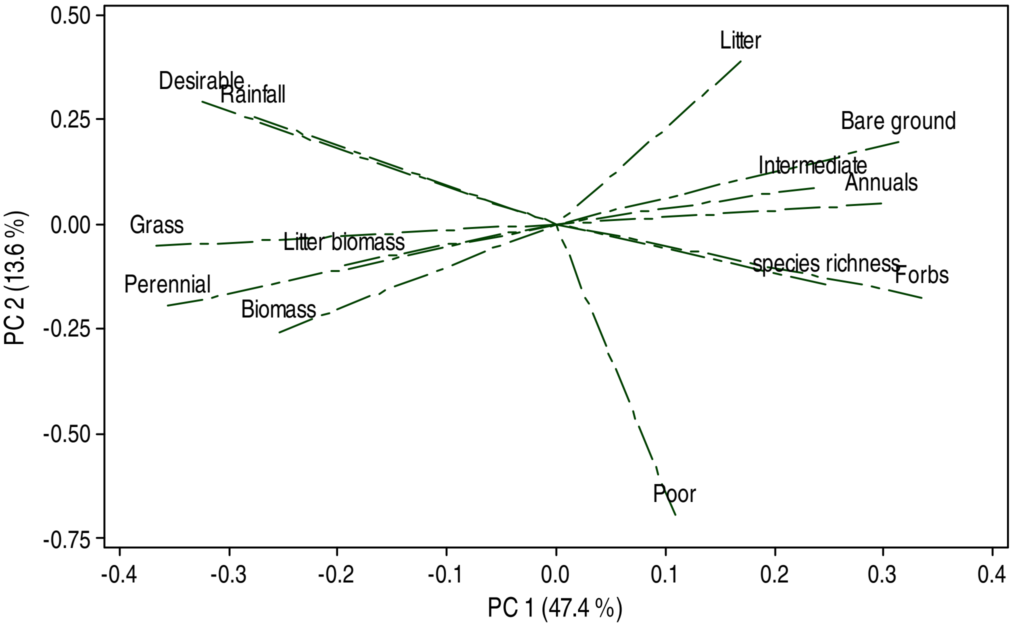

Figure 3.6: PCA loadings showing the relationships between herbaceous vegetation layer attributes and other environmental variables across savanna ecosystems

General linear model showed that herbaceous species richness was mainly explained by combined effects of rainfall and year of data collection, which accounted for 52.6 % of the variation (Table 3.2). Species richness was not significantly affected by grazing management systems and interaction between rainfall and year (P>0.05).

Simpson’s index of diversity also showed significant relationship with mean annual rainfall ($P<0.001$) (Table 3.2), but was not affected by year ($P>0.05$) or management ($P>0.05$).

Table 3.2: The combined effect (General linear model procedure) of factors on herbaceous species diversity

variables	Source	df	F	p-value
Species richness	Year (Y)	1	18.58	<0.001
	Rainfall (MAP)	2	17.15	<0.001
	Management system	1	0.44	>0.05
	Y x MAP	2	0.96	>0.05
Simpson’s index of diversity	Year	1	0.35	>0.05
	Rainfall	2	13.83	<0.001
	Management system	1	0.10	>0.05
	Y x MAP	2	1.50	>0.05

Y = year; MAP = mean annual precipitation

Multiple regression indicated that rainfall ($F=2.52$, $P<0.05$) and management systems ($F=-2.44$, $P<0.05$) had significant influence on aboveground herbaceous biomass ($R^2=0.39$; $P<0.01$). However, stepwise regression with multiple variables (including rainfall, species richness, stocking rates, soil clay, soil organic carbon and soil pH) showed that soil organic carbon was a better predictor of herbaceous biomass explaining 46 % (R^2) of the variance ($P<0.001$) (Figure 3.7). Simple regression indicated that surface litter biomass was largely affected by management systems ($R^2= 0.52$; $P<0.001$). However, multiple regression attributed 61 % of the variation in litter biomass ($P<0.001$) to a combination of management systems ($P<0.005$) and herbaceous biomass ($P<0.05$).

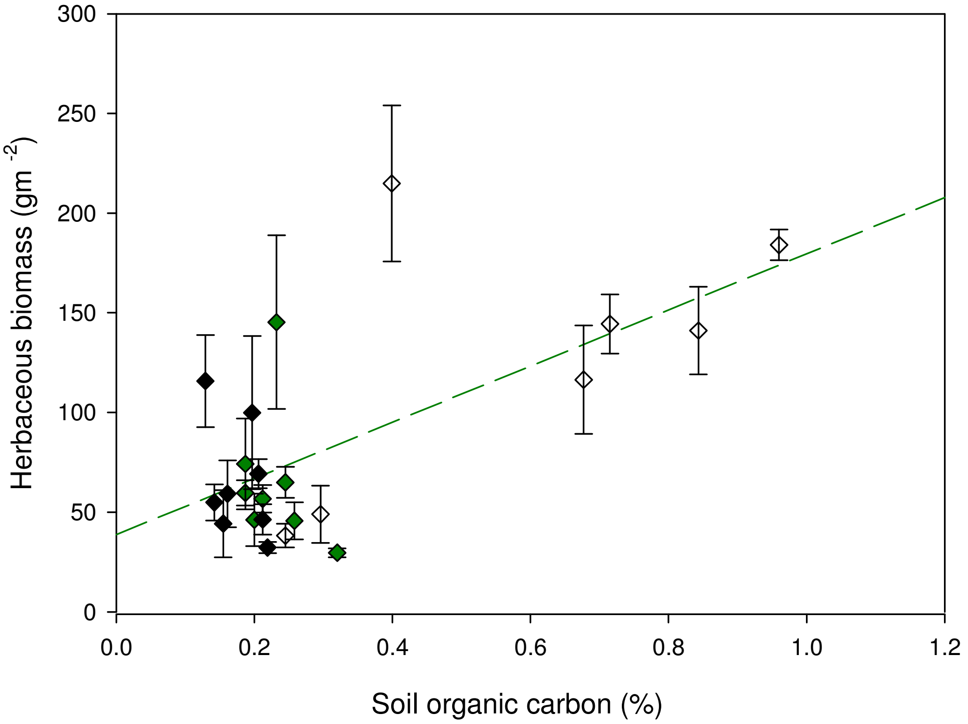

Figure 3.7: Relationship between soil organic carbon and aboveground herbaceous biomass (±SE) across savanna ecosystems (◊ Goodhope, ♦Matlolakgang, ♦ Xanagas)

3.4 Discussion

3.4.1 Characteristics of herbaceous vegetation in communal and ranching rangelands

The composition of herbaceous vegetation varied considerably between sites and management systems even at a local scale, which characterises the heterogeneity of Kalahari savanna ecosystem (Thomas and Twyman, 2004). Spatial variations in the composition of herbaceous species could be attributed to the ability of individual plant species to adapt to the local climatic and edaphic conditions (Carr *et al.*, 2009; Muhumuza and Byarugaba, 2009), which differed between our sites as discussed in Chapter 2. For example, *Stipagrostis uniplumis* a perennial grass is suited to dry sandveld conditions (van Oudtshoorn, 2002) as reflected by its dominance at Xanagas rangelands, and this is consistent with results from previous studies in the region (Cole and Brown, 1976). Heterogeneity of herbaceous vegetation at a local scale could be due to grazing history (Turner, 1999; Diaz *et al.*, 2001) especially since grazing pressure tends to decrease with distance from watering points (Smet and Ward, 2005). There was no dominant herbaceous species observed at Matlolakgang communal grazing land, which could be due to overgrazing depressing the vigour and presence of usually dominant species. This then allows colonization by a range of less competitive, but grazing tolerant plant species (Sternberg *et al.*, 2000).

Matlolakgang communal rangelands had a high percentage of bare ground and other increaser II species such as *Aristida congesta*, which are an indicator of overgrazing (Trollope *et al.*, 1989) and rangeland degradation (Pueyo *et al.*, 2006). Percentage bare ground was observed to be greater in communal grazing lands than ranches at Matlolakgang and Goodhope, which is supported by results from other studies (Smet and Ward, 2005; Tefera *et al.*, 2007). In contrast, bare ground recorded on communal and ranch land at Xanagas was not statistically different, which suggests that communal grazing does not always lead to an increase of degraded bare land. The grazing pressures at Xanagas communal and ranch lands were relatively lower than other sites (as shown in Chapter 1), which could explain lack of significant difference on bare ground frequency between management systems at Xanagas. The high cover of forbs in Matlolakgang and Xanagas rangelands, particularly communal lands, also suggested that these particular rangelands are in poor condition (Whitford *et al.*, 1998). That was further confirmed by high abundance of increaser II species , such as the grass *Aristida congesta,* which are associated with high grazing pressure (Trollope *et al.,* 1989; Skarpe, 2000) in communal rangelands at all study sites. The Matlolakgang ranch also had high cover of *Aristida congesta* suggesting that overgrazing is occurring under ranch management as well as on communal rangelands.

The proportion of annual herbaceous plants was high at Matlolakgang rangelands, which suggested that there are heavily grazed (Skarpe, 2000; Diaz *et al.*, 2007). Nevertheless, the distribution of perennial and annual herbaceous species did not show a consistent trend in relation to management systems across the sites. For example, Matlolakgang ranch had high abundance of perennial species in the ranch and high abundance of annuals in the communal lands, but opposite results were observed in Xanagas rangelands. The herbaceous community variability could be due to complex interactions of biotic and abiotic factors (Carr *et al.*, 2009). Vegetation is not necessarily affected by grazing in a linear way (Sasaki *et al.*, 2010) because individual species respond to grazing differently. Perennial grasses such as C*ynodon dactylon* and *Digitaria erianthra* are tolerant of grazing (O' Connor, 1994; van Oudtshoorn, 2002) and that was reflected by their abundance in communal rangelands at both Matlolakgang and Goodhope.

Based on a range condition guide for use with relative frequency of desirable and intermediate species (Mosley *et al.*, 1986), Goodhope communal and ranching rangelands conditions were classified as “fair” because they were dominated by palatable herbaceous vegetation. Meanwhile, Matlolakgang and Xanagas rangelands were considered to be in poor condition as they were dominated by unpalatable herbaceous vegetation. The management systems did not have a clear effect on the distribution of desirable, intermediate and unpalatable herbaceous plants. Their distribution varied between sites, which could indicate that local environmental condition influences how vegetation and individual species respond to grazing (Anderson *et al.*, 2007). The % cover of desirable herbaceous plants did not differ significantly between ranch and communal lands at Goodhope and Matlolakgang, which corroborates results in a similar study in South Africa (Parsons *et al.*, 1997). This does however contradict a study in Swaziland which reported a higher presence of highly palatable species in the ranch than on communal rangeland (Tefera *et al.*, 2008b). The results of this study show that moderate palatable species were more abundant in communal rangelands in all three sites, which is supported by Tefera *et al.,* (2008b), but disagree with the results from the study by Parsons *et al.,* (1997). The differences could partially be attributed to variation in stocking rates, especially since rangeland investigated in this study are lightly stocked compared to those investigated by Parsons *et al.,* (1997).

3.4.2 Herbaceous biomass in relation to rangeland management systems

The biomass of standing herbaceous plants and surface litter varied significantly in relation to management systems and sites. The standing biomass was highly diverse spatially at local (within site) level and also between sites ranging between 30 and 180 gm^{-2}, which is typical for semiarid ecosystems (Osem *et al.*, 2002). At local scale, variation in standing herbaceous biomass could be due to the heterogeneity of livestock grazing (Weber *et al.*, 1998), as herbaceous biomass tends to decline with increasing grazing intensity (Lin *et al.*, 2010; Mbatha and Ward, 2010). Given that livestock grazing is normally not uniformly distributed, it creates patchy vegetation as the animals unevenly reduce grass biomass (Savadogo *et al.*, 2007). The heterogeneity of herbaceous biomass could also be explained by variations in soil fertility because nutrient distribution in savanna ecosystems is also highly patchy (Scholes, 1990). The herbaceous biomass variability could also be affected by species composition as some species have higher productive potential than others (Mphinyane *et al.*, 2008). In all probability, the heterogeneity of herbaceous biomass in this study is influenced by a combination of soil fertility, grazing intensity and species compositions.

The herbaceous vegetation biomass variability between sites may be attributed to annual rainfall, which partially explains the high herbaceous biomass observed at Goodhope. Previous studies in the Kalahari and Southern Botswana have shown that grass biomass is positively associated with rainfall (Scholes *et al.*, 2002; Mphinyane *et al.*, 2008), which is consistent with our results. High herbaceous biomass at Goodhope could also be explained by low woody cover, discussed in Chapter 5 (Scholes *et al.*, 2002), higher soil fertility (Chapter 2) and the water-holding capacity of hardveld (clayey) relative to sandveld as illustrated in the literature (van Langevelde *et al.*, 2003). Despite slightly higher mean annual rainfall at Matlolakgang, herbaceous biomass did not differ to that observed at Xanagas. This could be due to higher utilization of herbaceous vegetation by livestock at Matlolakgang. High grazing intensity has been shown to decrease standing biomass as more biomass is consumed (Mphinyane *et al.*, 2008; Wang *et al.*, 2010) and this is also indicated by high percent bare ground as recorded at Matlolakgang in this study. An alternative explanation could be that Matlolakgang rangelands are relatively more degraded than Xanagas lands as indicated by levels of bush cover (discussed in Chapter 4) and as a result, the herbaceous biomass responds poorly to rainfall due to the dominance of annual species with poor yield.

Goodhope and Matlolakgang ranches generally had higher standing herbaceous biomass than communal rangelands, probably because of moderate stocking rates (> 10 ha/LSU) in ranches relative to communal rangelands. High grazing intensity such as 4 hectares per livestock unit (ha/LSU) observed at Matlolakgang and Goodhope (Chapter 1) tends to reduce herbage yield (Mphinyane *et al.*, 2008) through defoliation and trampling (Savadogo *et al.*, 2007). This could explain low herbage productivity in communal lands at Goodhope and Matlolakgang, and lack of differences in herbaceous biomass between the communal and ranch lands in Xanagas, which had lower grazing intensity than other communal lands. At low grazing intensity (10 ha/LSU), grazing tolerant herbaceous plants are likely to maintain primary production due to compensatory growth (Gruntman and Novoplansky, 2011), provided soil moisture and soil nutrients are not limiting. This could mask any impact of different grazing intensity on herbaceous biomass as observed at Xanagas where there is no significant difference in biomass between management systems. Thus, herbaceous biomass at local scale reflects the grazing intensity and not necessarily management systems. Rotational grazing, which is a key component of ranching management, has been shown to have no effect on vegetation changes (Mashiri *et al.*, 2008) and therefore is not expected to have a significant impact on herbaceous biomass variability between the two management systems in this study.

The surface litter mass was significantly higher in ranches than communal lands at all sites, which slightly differed from the pattern of standing biomass. The surface litter is important because it protects the soils from erosive factors like raindrops (Fatunbi and Dube, 2008) and also contributes to nutrient cycling (Lawrence, 2005). The surface litter biomass could be influenced by anthropogenic factors such as grazing pressure and environmental factors such as rainfall (Lawrence, 2005; Zhou *et al.*, 2009). In the present study, the variation in surface litter mass between sites could be explained by rainfall, while differences within sites are probably due to variations in grazing pressure (Elmore and Asner, 2006). Low levels of surface litter are expected under high grazing intensity (Savadogo *et al.*, 2007) as a result of the high consumption of standing herbaceous biomass (Hiernaux *et al.*, 1999; Ruprecht *et al.*, 2010). This explains surface litter differences between communal and ranching systems at Goodhope and Matlolakgang because at those sites communal grazing lands were more heavily grazed than respective ranches as shown in Chapter 1. However, it is not clear why Xanagas ranches had higher surface litter than communal grazing lands because communal lands had lower stocking rates than the ranch.

3.4.3 Herbaceous vegetation diversity in relation to rangeland management systems

The diversity of species in the herbaceous vegetation varied significantly between sites but not between communal and ranching lands. The variations between sites could be due to the level of ecosystem disturbance from grazing, differences in plant biomass or rainfall variability. The high level of herbaceous vegetation diversity at Matlolakgang could be attributed to heavy grazing pressure especially in communal land where stocking densities are as high as 5 hectares per livestock unit (Chapter 1). Species richness has been shown to increase in response to heavy grazing in Mongolian grasslands (Cheng *et al.*, 2011), mostly due to the establishment of forbs and less dominant grasses, as were observed at Matlolakgang. Continuous grazing also promotes the growth of forb species by reducing standing biomass and leads to a relatively high species-rich community in Mediterranean communities and semi-arid savannas (Sternberg *et al.*, 2000; Jacobs and Naiman, 2008). However, this explanation is challenged in this study by lack of pronounced variation of species richness between communal and ranching lands at each site despite differences in grazing intensities.

Species richness in grazed lands is also known to increase with potential productivity (Osem *et al.*, 2002) until it reaches 500 gm^{-2} before it declines (Oba *et al.*, 2001). Biomass production in all sites in this study were below the optimal productivity level with maximum of 171 gm^{-2} at Goodhope, and therefore herbaceous diversity would be expected to increase with increasing biomass, which is opposite to our observations. Hence there is either no substantial relationship between diversity and herbaceous productivity (Sanderson, 2010) or it varies with no general pattern (Partel *et al.*, 2010). It is possible that climatic conditions, in particular soil properties and rainfall, could be more important in regulating species diversity between sites than the grazing pressure. The diversity did not vary significantly in response to management systems, which could suggest that both communal and ranching management systems have no or similar affects on herbaceous diversity. Thus, the results of this study do not support the perception that heavy grazing causes loss of biodiversity (Oba *et al.*, 2001). However, it has been suggested that grazing has no effect on diversity at local scale (Adler *et al.*, 2005; Sassi *et al.*, 2009) and that could explain lack of pronounced differences on herbaceous vegetation between communal and ranching lands in this study.

3.4.4 Relationships between herbaceous vegetation attributes and other environmental variables

The PCA results illustrated the vegetation community structure response to grazing and environmental conditions recorded in this study. These could be used as indicators of rangeland conditions in semi-arid ecosystems as suggested by other authors (Trodd and Dougill, 1998). The increases of bare ground, annual herbaceous species, and forbs clearly indicate that the rangeland ecosystem in Botswana is experiencing ecological stress, which is consistent with the literature (du Plessis *et al.*, 1998; Skarpe, 2000). It is however essential to have good understanding of the individual factors that influence diversity patterns in savanna vegetation (Sankaran, 2009) and also plant productivity (Wang *et al.*, 2010) to facilitate sustainable management (Sternberg *et al.*, 2000). The herbaceous vegetation diversity presented as Species richness and Simpson's index of diversity declined with increasing mean annual rainfall across all study sites, which is consistent with the observations of other researchers (Aarrestad *et al.*, 2011) in northern Botswana. In arid environments, low and highly variable rainfall limits growth and dominance by a few perennial species (Fensham *et al.*, 2010) and allows establishment of other species, thus increasing diversity. Therefore, our results suggested that rainfall is the main determinant of herbaceous diversity in savanna ecosystems, which is consistent with findings of other authors (Sternberg *et al.*, 2000; Adler *et al.*, 2005). However, others (Adler and Levine, 2007) have observed positive relationship between rainfall and richness, which is opposite our observation. It is acknowledged that interactions between environmental and management factors may be difficult to distinguish clearly due to the short period of study (Oconnor, 1994; Watkinson and Ormerod, 2001). The significant effects of rainfall on herbaceous diversity should not however be considered as clear evidence that savanna ecosystems are unstable and driven by rainfall variability, as proposed by the non-equilibrium model (Ellis and Swift, 1988; Westoby *et al.*, 1989) especially when 47.4 % of this variance is not explained by factors measured in our data.

The combined effects of rainfall and management systems accounted for 39 % of variation on standing herbaceous biomass. Grazing pressure and rainfall distribution have previously been considered responsible for variation plant biomass in the Mediterranean (Sternberg *et al.*, 2000), which supports our findings. Yet, soil organic carbon, which could reflect soil fertility, was considered a better predictor of herbaceous biomass from this study. In dry savannas, soil fertility affects the growth rate during the rainy period (Scholes, 1990) and that could explain our observed relationship between soil organic carbon and herbaceous biomass. According to studies in Australia by Fensham *et al.*(2010), small variations in soil properties could affect floristic composition more than grazing in arid environments, and therefore, this could explain our observations. However, it is also recognized that organic carbon is strongly associated with climate especially rainfall (Gray *et al.*, 2009) and therefore the relationship of SOC with herbaceous biomass could be confounded by the amount of rainfall and its inter-annual variability in the study region. The unaccounted variance of 54 % could be attributed to other environmental factors such as fire (Medinski *et al.*, 2010). The surface litter mass was mainly influenced by management systems, and reflects the impact of variation in grazing intensity and rotational grazing. The grazing management systems have limited influence on species diversity, however do have significant influence on herbaceous vegetation biomass, which agrees with findings of other authors (Sasaki *et al.*, 2008).

3.5 Conclusion

The herbaceous vegetation distribution showed spatial heterogeneity both within sites and between sites, but the functional traits of species did not show any consistent pattern in relation to management systems. Goodhope communal and ranching lands were generally in fair conditions and characterised by moderate cover of both decreaser and increaser II species in the vegetation. The percent covers of increaser II species were slightly higher under communal lands than ranches at Matlolakgang and Xanagas, but all lands were in poor condition compared to Goodhope rangelands. The herbaceous biomass was significantly higher under ranching lands than communal lands at Goodhope and Matlolakgang, but not at Xanagas. However, our results indicated clearly that management systems do not affect herbaceous plants diversity differently despite communal lands often being blamed for loss of diversity in the literature. Soil properties and rainfall distribution were shown to be the main determinants of herbaceous biomass, and diversity in all our study sites.

Chapter 4

Bush encroachment in relation to rangeland management systems and environmental conditions in Kalahari ecosystem of Botswana

4.1 Introduction

Bush encroachment, the increase in woody vegetation density, cover and biomass (Tews *et al.*, 2004; van Auken, 2009), is a common environmental problem in southern Africa (Moleele *et al.*, 2002; Joubert *et al.*, 2008; Wigley *et al.*, 2009). Bush proliferation in savanna ecosystems is associated with increase in woody plants such as *Terminalia sericea, Acacia mellifera* (van Vegten, 1984) and *Grewia flava* (Tews *et al.*, 2004). Their establishment success has been attributed to being perennial and to their opportunistic characteristics (Smith and Smith, 2001; Hipondoka and Versfeld, 2006). However, factors that drive bush encroachment are not well understood (Moleele *et al.*, 2002; Ward, 2005; Wiegand *et al.*, 2005). It is often associated with overgrazing (van Vegten, 1984; Skarpe, 1990a), rainfall (Joubert *et al.*, 2008), fire suppression (Oba et al., 2000; van Langevelde et al., 2003) and soil characteristics (Wilson and Bowman, 1994; Mourik *et al.*, 2007; Sankaran *et al.*, 2008).

In savanna ecosystems, the dominance of herbaceous plants and patchy distribution of woody plants is essential for livestock production and conservation of rangeland

(Bond and Midgley, 2000). Bush encroachment suppresses the productivity of herbaceous plants (Hagos and Smit, 2005; Ward, 2005) and negatively affects both the availability of fodder for livestock and biodiversity conservation (Skarpe, 1990a; Wigley *et al.*, 2010). It also reduces the area covered by grasses (Scholes and Archer, 1997; Hagos and Smit, 2005). This has a negative impact on the carrying capacity of rangelands (Ward, 2005; Wiegand *et al.*, 2006) and threatens livestock production (Kraaij and Ward, 2006; Angassa and Oba, 2008a). Therefore, bush encroachment is an indication of rangeland degradation in savanna ecosystems (Oba *et al.*, 2000).

Bush encroachment in African savanna ecosystems has been attributed to overgrazing by livestock (Skarpe, 1990b; Roques *et al.*, 2001; Coetzee *et al.*, 2007) especially in communal rangelands (van Vegten, 1984; Kraaij and Ward, 2006). According to the two layer hypothesis based on rooting depth niche separation (Walker *et al.*, 1981; Knoop and Walker, 1985) overgrazing removes the grass layer and thus reduces competition for resources for soil moisture, which in turn promotes establishment of woody plants (Harrington, 1991; Dougill *et al.*, 1999; Yanoff and Muldavin, 2008). In addition, grazing animals accelerate bush encroachment through dispersal of the seeds of woody plants (Brown and Archer, 1999; Tews *et al.*, 2004). For example, cattle and goats browse leaves and pods of shrubs and trees such as *Dichrostachys cineria* and *Grewia flava* as part of their diet (Moleele, 1998). This facilitates seed dispersal through faecal deposits, which also improve chances of seed germination and the survival of woody plants (Loth *et al.*, 2005). Heavy grazing also reduces the fuel load, hence reducing the frequency of wildfires and promoting the establishment of woody plants (Archer *et al.*, 1995; van Langevelde *et al.*, 2003).

Other authors (Kraaij and Ward, 2006; Angassa and Oba, 2008a) however have observed that bush encroachment is not mainly caused by overgrazing but is part of a natural successional process.

The patch dynamics theory suggests that the savanna ecosystem consists of a mosaic of patches at different cyclical succession phases dominated by either woody or herbaceous plant species (Wiegand *et al.*, 2005; Meyer *et al.*, 2007). The shifting mosaic is mainly caused by variations in abiotic factors such as rainfall and inter-tree competition for resources (Wiegand *et al.*, 2006). Successive years of increased precipitation triggers mass tree/shrub seedling recruitment, which is the most critical stage in the bush encroachment process (Kraaij and Ward, 2006). The spatio-temporal variability of rainfall in arid environments accelerates development of a patchy vegetation pattern (Wiegand *et al.*, 2006; Meyer *et al.*, 2009). The inter-tree competition regulates the spatial distribution of trees and shrubs through self-thinning (Wiegand *et al.*, 2005) and grazing has minimal influence on bush encroachment (Brown and Archer, 1999; Kraaij and Ward, 2006).

The Tribal Grazing Land Policy (TGLP) was introduced in Botswana in 1975 to address land degradation in communal grazing areas through the privatisation of ranches (Botswana Goverment, 1975; Tsimako, 1991). This was necessary because rangeland degradation was occurring at an alarming rate, and threatening the livelihoods of pastoral farmers (van Vegten, 1984; Tsimako, 1991). For example, it is reported that rangeland degradation affected 25% of Botswana in 1980

(Vanderpost *et al.*, 2011) and an estimated 37 000 km² (6.4 %) of land especially around watering points mainly boreholes and kraals was encroached by woody plants in 1994 (Moleele *et al.*, 2002). However, there is limited evidence to suggest that the land use change through TGLP has achieved its goals. Understanding the dynamics of bush encroachment in response to rangeland management systems over a broad range of environmental conditions is essential for sustainable management of rangelands in arid environments (Milchunas and Lauenroth, 1993).

This study was carried out to investigate bush encroachment dynamics in relation to rangeland management systems in areas with different environmental conditions in Botswana. The specific objectives of the present study were to determine:

i. Woody plant cover and density in relation to communal grazing land and ranches.
ii. Diversity of woody vegetation in relation to communal grazing land and ranches
iii. Rooting systems of encroacher and non encroacher woody vegetation
iv. Relationships between woody vegetation dynamics and environmental variables

4.2 Material and methods

4.2.1 Study sites

The study was conducted at Goodhope, Matlolakgang, Xanagas ranches and their surrounding communal grazing lands during 2009 and 2010. The three sites vary in rainfall, soil and vegetation type. A detailed description of the study sites is given in Chapter 1.

4.2.2 Data collection

A vegetation composition survey was conducted towards the end of the growing season of 2009 (April–May). This was not repeated in 2010 because the woody vegetation was expected to remain fairly constant over this short period (Bonham, 1989). This stability makes trees/shrubs better indicators of persistent shift in vegetation composition than annual plants (Vincke *et al.*, 2010). The woody vegetation data were collected along 300 m line transects located at predetermined distances in randomly selected paddocks within both ranches and communal grazing land. At each site, four transects were established in communal grazing land, with the exception of Goodhope which had three transects as explained in Chapter 1. In additional, four transects were randomly established within each ranch. Collecting data from many plots ensured coverage of a good representative area (Bonham, 1989).

The woody vegetation was measured in 10 × 10 m quadrats located at 100, 200 and 300 m along transects, giving a total sampling area of 300 m^2 per transect. Individual woody plants in each quadrat were identified and classified according to height class and counted. They were classified as seedlings (<0.5 m), small shrubs (0.5 <1.0 m), shrubs (1<2.0 m), and tall shrubs/trees (>2.0 m) (Angassa and Baars, 2000; Dahlberg, 2000b; Angassa and Oba, 2008a). Plant height was measured to the highest living part using a 2 m long measuring rod. For each height class, at least three woody plants were selected at random and their crown diameter measured to calculate the crown area as follows:

$$\text{л x } (D/2)^2$$

where л = 3.14 and D is mean diameter of woody plants (Bonham, 1989). The tree/shrub density per transect was then multiplied by the crown area to calculate woody vegetation cover (%) (van de Vijver *et al.*, 1999). The density of woody plants, which is the total number of individual plants per given area (Bonham, 1989), was calculated per transect (300 m^2) and a conversion factor (10000/300) was used to convert to number of plants per hectare (ha^{-1}). The total density included individual plants of different height classes, except seedlings because they had not emerged above grass layer (Roques *et al.*, 2001) and are less likely to develop into shrubs due to competition from herbaceous vegetation plants.

The diversity of woody plants was estimated using species richness (R) and the Shannon-Weiner diversity index (*H')*. Species richness measures the total number of species present in a particular habitat and is also referred to as α-diversity (Waite, 2000). The Shannon-Wiener index was also used to estimate woody vegetation heterogeneity per transect and was calculated as:

$$H' = -\sum p_i \ln p_i$$

where p_i (n_i/N) is the proportion of each species in the sample (Reyes *et al.*, 2010). High value of *H'* reflects high heterogeneity or uncertainty that a randomly selected individual from a community of N individuals will be the same species as the previous one (Smith and Smith, 2001).

Three dominant encroacher woody plant species and one non-encroacher were selected based on their abundance across study sites during the survey in 2009 and their rooting system measured in 2010. The dominant encroacher woody species selected were *Terminalia sericea, Dichrostachys cineria,* and *Grewia flava. Bauhinia petersiana* was also selected as a dominant non-encroacher plant for comparison with encroacher plants. However, *G. flava* was the only species investigated at Goodhope because other selected species were absent. Individual woody plants of height greater than 0.5 m but less than 2 m (n = 44) were selected at random along randomly selected transects and their roots excavated to the greatest feasible depth. Excavation was done manually, and therefore it was not always possible to dig out the whole main root, particularly tap roots, and large trees were avoided (Belsky, 1994). The root system was classified as a taproot if it mainly penetrated the soil vertically, or was recorded as lateral if roots grew horizontal to the soil surface (Hipondoka and Versfeld, 2006). Characterising woody species is important in developing measures to control bush encroachment (Engelen and Santos, 2009).

In addition to the survey data, woody plant characteristics were related to environmental factors such as rainfall and soil characteristics at each site. Mean annual rainfall data were calculated for the period from 1988 to 2008 (see Chapter 1) and was obtained from the Department of Agricultural Research. The soil properties data collected in 2009 and 2010 have been described in chapter 2.

4.2.3 Statistical analysis

The general linear model (GLM) was used to compare the effects of multiple categorical factors such as management and mean annual rainfall on woody vegetation cover, density and diversity across the study sites. Comparisons of rangeland management systems effects on woody vegetation cover and diversity at each site were made using one-way analysis of variance (ANOVA). The non-parametric Mann-Whitney U test was used to compare woody plant densities at different height classes between the communal grazing land and ranches because these data were not normally distributed and could not be improved by transformation. Chi-square test was used to determine if dominant plants had uniform rooting types. Simple regression was used to identify the relationships between woody plant cover, total density, species richness and environmental variables, while stepwise regression analysis was used to determine the best explanatory variables. The visual checks of the histogram and normal probability plot were used to test for normality of data (Hair *et al.*, 1998). All statistical analyses were run in Minitab 15.

4.3 Results

4.3.1 Woody plants cover and density in relation to rangeland management systems

The general linear model (GLM) showed that the woody plant cover was significantly influenced by site effects (P<0.001), but did not differ between the ranching and communal grazing lands (P>0.05) (Table 4.1). The highest woody plant cover was observed at Matlolakgang rangelands and the lowest at Goodhope (Figure 4.1). There was no significant difference in woody plant cover between the ranching system and communal grazing land at all sites (P> 0.05, Figure 4.1)

Table 4. 1: Effect of site and rangeland management systems on woody plant cover across semi-arid savannas of Botswana

Source of variation	Df	F	P-value
Site	2	35.14	< 0.001
Rangeland Management system	1	0.00	0.985

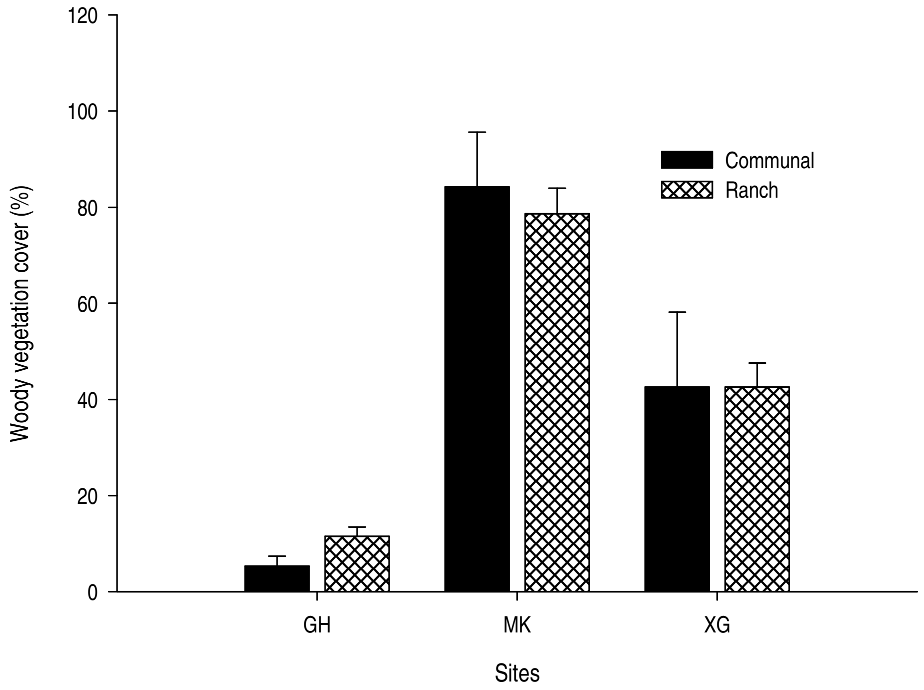

Figure 4.1: Woody vegetation in rangelands at Goodhope (GH), Matlolakgang (MK), and Xanagas (XG)

The woody vegetation cover at Goodhope communal grazing land was dominated by *Grewia flava* (26.5 %), *Acacia mellifera* (23.4 %), *Acacia hebeclada* (19.7 %), *Diospyros lyciodes* (18.1 %) and the least recorded woody plants were *Ziziphus mucronata* (7.9 %) and *Acacia erioloba* (4.51 %). The patchy woody plant cover in the ranch at Goodhope was dominated by *G. flava* (22.3 %), *Z. mucronata* (22.3 %), *Acacia karoo* (19.8 %), *A. erioloba* (18 %) and *A. hebeclada* (12.3 %). Other woody species present included *A. mellifera, A. tortilis, Tarchonanthuis camphoratus,* and *D. lyciodes*.

Woody plant cover at Matlolakgang communal grazing land was dominated by *Terminalia sericea* (34.6 %), *Dichrostachys cineria* (27.7 %), and *Bauhinia petersiana* (14.2 %). Other woody plant species present included *A. erioloba. Acacia fleckii, Acacia gerrardii, D. lycioides, Ehretia rigida, Grewia avellana, Grewia flavescens, Maytenus senegalensis, Maytenus tenuispina, Ochna pulchra, Peltophorum africanum, Rhus tenuivervis,* and *Ziziphus mucronata.* Matlolakgang ranch was dominated by *T. sericea* (30.9 %), *B. petersiana* (13.3 %)*, A. erioloba* (12.4 %), and *D. cineria* (11.7 %). Other woody plants found at Matlolakgang ranches included *A. fleckii, A. gerrardii, A. mellifera, Boscia albitrunca, Combretum apiculatum, D. lycioides, G. avellana, G. flava, G. flavescens, G. retinervis, M. senegalensis, M. tenuispina, O. pulchra, P. africanum,* and *R. tenuinervis.*

The dominant woody plant species at Xanagas communal grazing land included *D. cineria* (22.8 %), *G. flava* (22.3 %), *T. sericea* (21.9 %), *A. mellifera* (14.8 %), and *Acacia burkie* (8.27 %). Other scattered woody plants such as *A. fleckii, A. hebeclada, B. petersiana, B. albitrunca, E. rigida, G. avellana, Ozoroa paniculosa, Rhigozum brevispinosum, R. tenuinervis*, and *T. camphoratu* made a small contribution to total woody plant cover. *Terminalia sericea* (44.3 %), G. *flava* (23.8 %) and *B. petersiana* (19.2 %) dominated the woody plant cover at Xanagas ranch. Few scattered trees of *A. fleckii, A. mellifera, B. petersiana, Commiphora angolensis, D. cineria, G. avellana, Lonchorarphus nelsii, O. paniculosa,* and *T. camphoratus* were found at Xanagas ranch.

There was a strong correlation between the woody vegetation cover and total woody plant density (seedlings excluded) ($r = 0.87$, $P<0.001$). There was a high variation in woody plant density across study sites as shown by high coefficient of variation of 22−103 %. The GLM analysis showed that the total woody plant density differed significantly between sites ($F=26.45$, $P< 0.001$), but no significant difference was observed between the communal grazing land and the ranching system ($P>0.05$). One-way ANOVA showed that site effects explained 73.4 % of the variability in woody plant density. Matlolakgang rangelands had the highest woody plant density of 8017 plants ha^{-1}, followed by Xanagas with 3375 plants ha^{-1} and Goodhope rangelands had the lowest woody plant density of 467 plants ha^{-1}. The total plant density showed no significant differences between communal and ranching land at each site ($P>0.05$). The woody plant height classes are as shown in Figure 4.2 and indicate that all sites were dominated by seedlings and shrubs. However, a pair-wise comparison (Mann-Whitney test) of woody plant density of different height classes at each site showed no significant difference between the communal grazing land and ranching system ($P>0.05$).

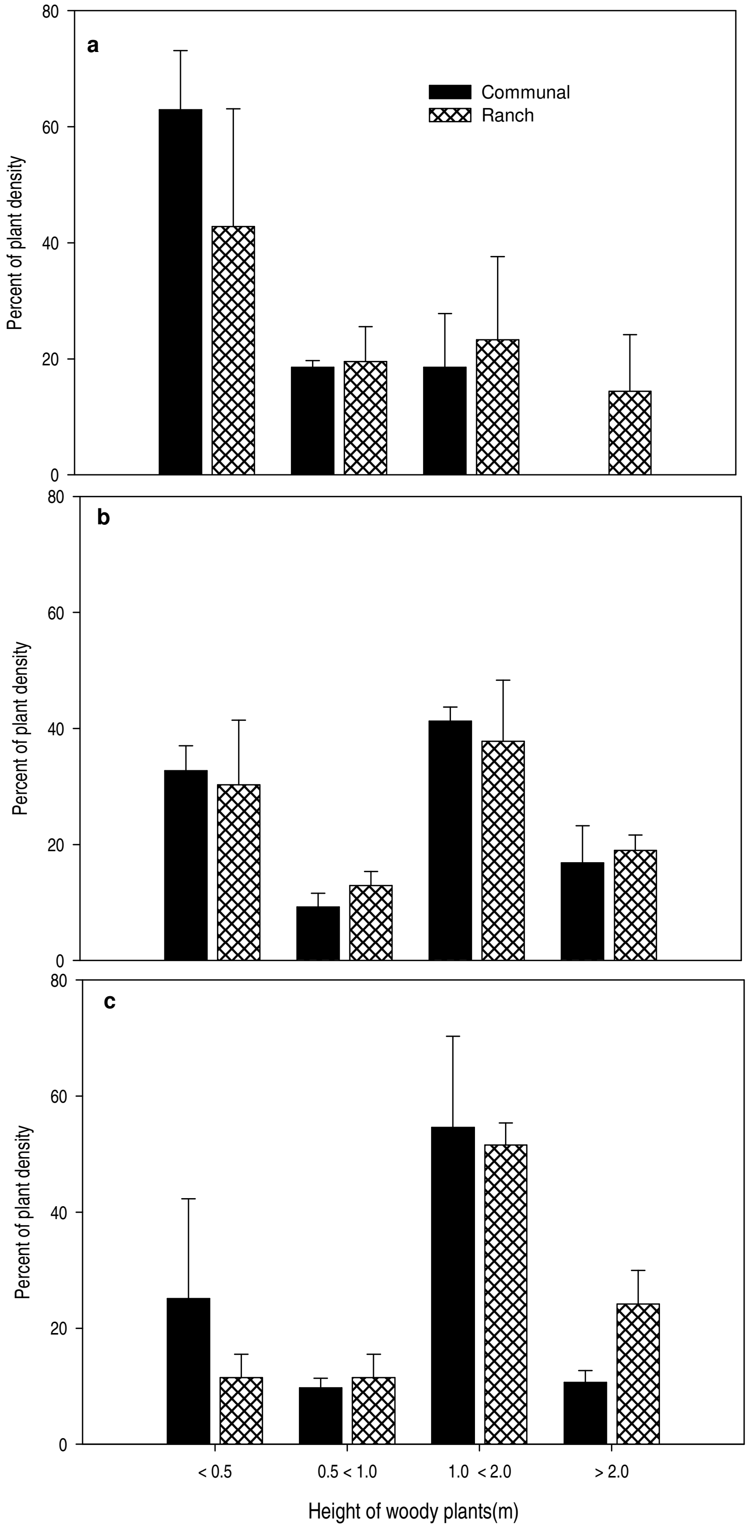

Figure 4.2: Mean percent of woody plant density (± SE) in rangelands at Goodhope (a), Matlolakgang (b) and Xanagas (c).

4.3.2 Woody vegetation diversity in relation to rangeland management systems

The woody plant species richness differed significantly between sites (F= 13.97, P<0.001), but not between the communal grazing land and the ranching system (P>0.05). Matlolakgang had the highest species richness (10.00 ± 0.95), followed by Xanagas (8.00 ± 0.68), while Goodhope rangeland had the least woody species diversity (5.00 ± 0.43). Analysis of variance at each site showed significant differences in species richness between the communal grazing land and the ranching system at Matlolakgang ($F=7.42$, $R^2 = 0.55$, $P<0.05$), but not at other sites (P>0.05) (Table 4.2). However, the Shannon-Wiener index showed no significant difference either between sites or rangeland management systems (all P>0.05). Further comparative analysis of Shannon–Wiener index at site level failed to show significant differences between the communal grazing land and ranching system (P>0.05). However, the Shannon-index reflected a similar pattern to that followed by species richness with the lowest woody vegetation diversity at Goodhope and the highest diversity at Matlolakgang (Figure 4.3).

Table 4.2: Mean (± SE) woody species richness across the semi-arid savannas of Botswana

Site	Management systems	
	Communal	Ranch
Goodhope	4.00 ± 0.58	5.00 ± 0.58
Matlolakgang	8.00 ± 1.08 a	12.00 ± 0.85 b
Xanagas	8.00 ± 1.22	7.00 ± 0.75

Different letters show significant differences

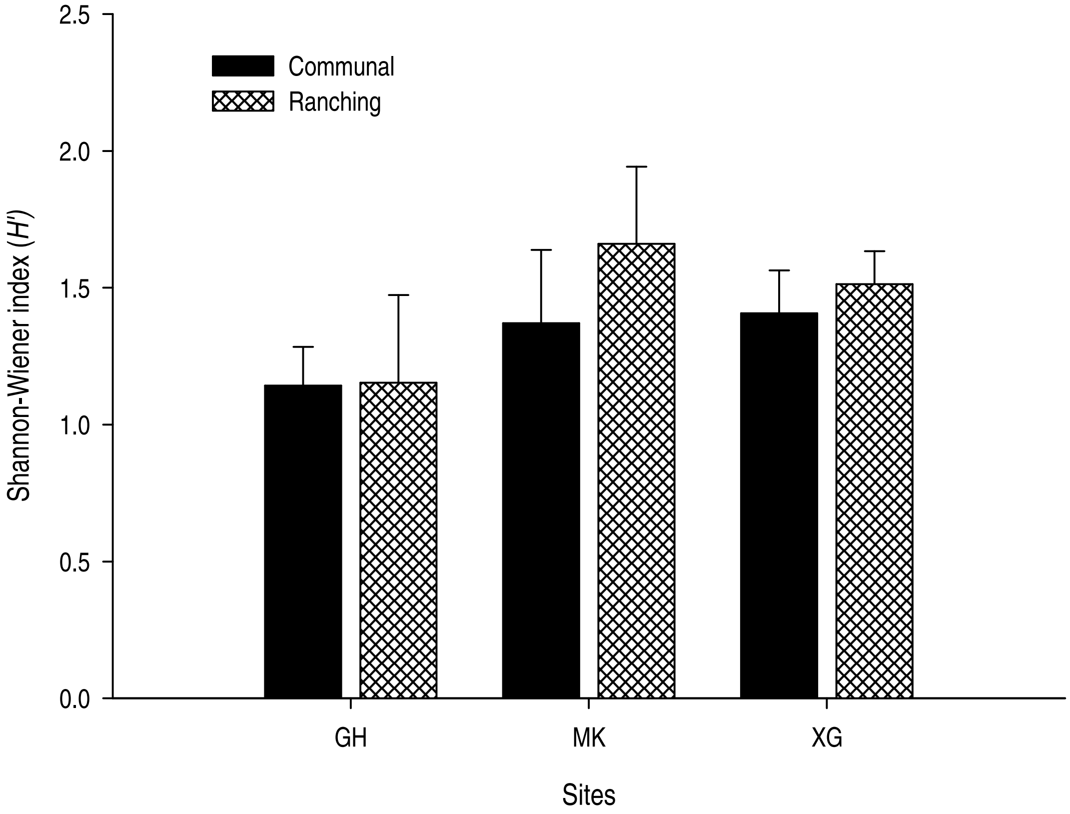

Figure 4.3: The Shannon-Wiener diversity index (± SE) for woody vegetation in relation to management systems at Goodhope (GH), Matlolakgang (MK), and Xanagas (XG).

4.3.3 Rooting systems of encroacher and non-encroacher woody vegetation

The dominant woody plants species in rangelands at the three study sites showed significant differences in rooting systems (χ^2 = 20.78, $P<0.001$), though three quarters of the woody plant species had numerous lateral roots. *Bauhinia petersiana* shrubs exhibited both taproot (n= 5) and lateral roots (n=4, Table 4.3). *Dichrostachys cinerea* had many lateral roots (n=7), and few trees (n = 4) had a taproot. The *Grewia flava* rooting system (n=14) was dominated by shallow lateral roots found at an average soil depth of 50 cm (Table 4.3). All *Terminalia sericea* trees (n=10) had a tap rooting system with a root diameter of approximately 2.69 cm at the top, narrowing to about 0.53 cm at the end of the root.

Table 4.3: Shrub height and root characteristics (±SE) of woody plants selected in rangelands at Goodhope, Matlolakgang and Xanagas.

Woody plant Species	Height (m)	Root diameter at beginning (cm)	Root diameter at end (cm)	Max root length excavated (m)
Bauhinia petersiana	1.08 ± 0.11	3.14 ± 0.43	1.33 ± 0.27	1.58 ± 0.19
Dichrostachys cineria	1.19 ± 0.10	2.84 ± 0.45	0.46 ± 0.09	1.74 ± 0.24
Grewia flava	1.13 ± 0.08	2.72 ± 0.25	0.63 ± 0.11	2.63 ± 0.39
Terminalia sericea	1.14 ± 0.11	2.69 ± 0.36	0.53 ± 0.34	1.19 ± 0.06

4.3.4 Relationships between woody vegetation dynamics and environmental variables

The general linear model showed that both woody plant cover ($P<0.001$) and species richness ($P<0.01$) varied significantly in relation to rainfall variability. The rangeland management systems had no significant effect on either woody plant cover or species richness ($P>0.05$). A stepwise regression analysis with multiple predictors (including rainfall, soil organic carbon, herbaceous biomass, and species richness) however showed that soil clay content was the best predictor of woody plant cover ($R^2=0.41$, $P<0.01$), density ($R^2 =0.34$, $P<0.01$) and species richness ($R^2=0.24$, $P<0.05$). Applying a polynomial regression analysis improved the explanatory power of soil clay content on woody vegetation cover ($P<0.001$), woody plant density ($P<0.01$) and species richness ($P<0.01$) respectively (Figures 4.4, 4.5 and 4.6). The woody plant cover, density and species richness decreased with increase in soil clay content, but not in a linear way. The woody plant cover, density and species richness varied largely between sites (Figure 4.4, 4.5 and 4.6) in response to soil clay content. The soil clay content did not vary much at each site except at Goodhope and this was subsequently reflected in the variation of woody plant cover. However, woody plant cover in one transect at Xanagas communal grazing land was very high (89%) (Figure 4.4 and 4.7) and differed to the woody plant cover observed in other transect at the same site. The woody plant cover also showed a negative linear relationship with herbaceous plant biomass, whereby increase in woody plant cover resulted in decreased herbaceous plant biomass (P=0.056) (Figure 4.7). Approximately 16% of the variation in herbaceous plant biomass was explained by changes in woody vegetation cover leaving a large proportion of the variation unexplained. There was a large variation in herbaceous plant biomass at Goodhope rangelands, compared to Matlolakgang and Xanagas (Figure 4.7).

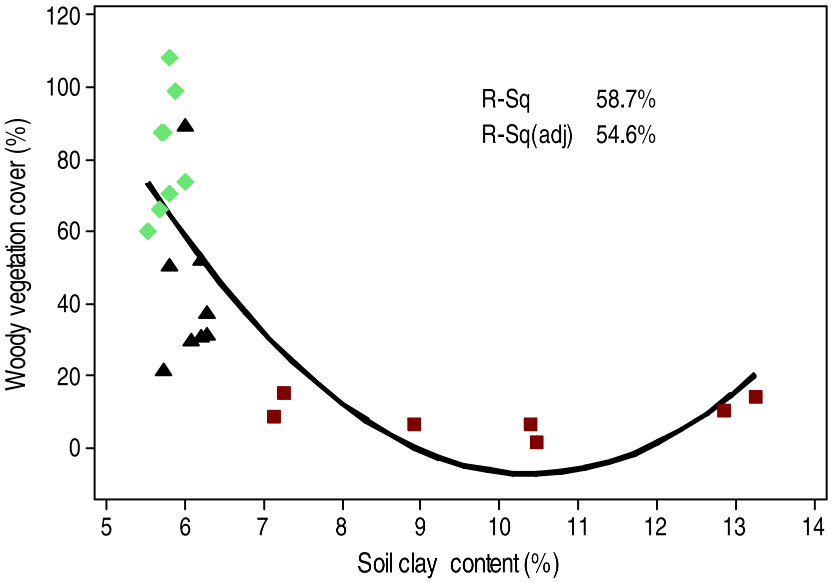

Figure 4.4: Relationship between woody cover and soil clay content across semi-arid savannas of Botswana (■ Goodhope, ♦ Matlolakgang, ▲ Xanagas).

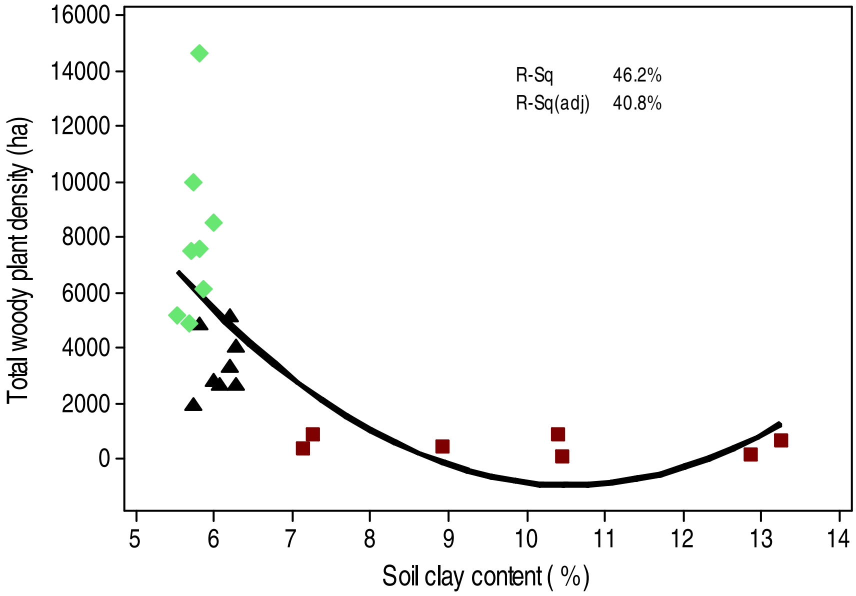

Figure 4.5: Relationship between woody plant density and soil clay content across savannas of Botswana (■ Goodhope, ♦ Matlolakgang, ▲ Xanagas)

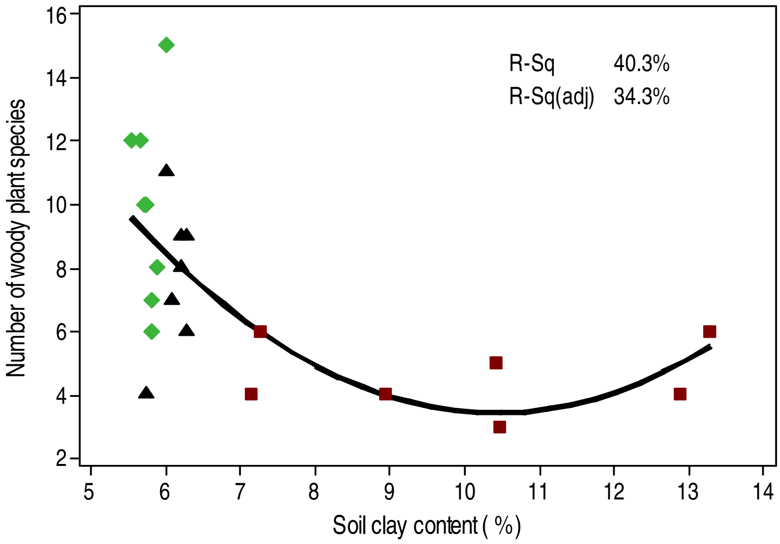

Figure 4.6: Relationship between woody plant species richness and soil clay content in semi-arid savannas of Botswana (■ Goodhope, ♦ Matlolakgang, ▲ Xanagas)

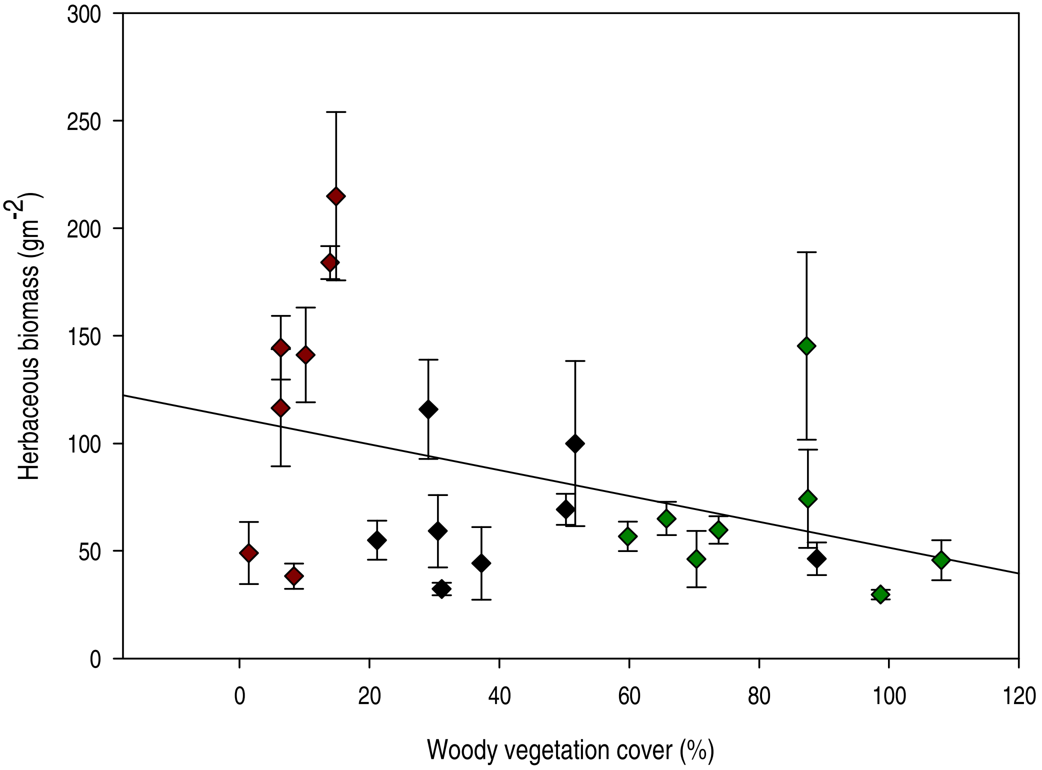

Figure 4.7: Relationship between herbaceous biomass (±SE) (g / m²) and woody vegetation cover across semi-arid savannas of Botswana (♦ Goodhope, ♦ Matlolakgang, ♦ Xanagas)

4.4 Discussion

4.4.1 Woody plants cover and density in relation to rangeland management systems

The studied rangelands exhibited heterogeneity in woody vegetation cover and density both at site level and between sites. At site level, some transects had patches of land with high woody plant density, which is consistent with the explanation of savanna patch dynamics (Meyer *et al.*, 2007; Meyer *et al.*, 2009). Goodhope rangeland had the lowest woody plant cover and density, suggesting that bush encroachment is minimal, despite the dominance of encroacher species such as *Grewia flava* and *Acacia mellifera* (Tews *et al.*, 2004; Joubert *et al.*, 2008). The low woody plant cover at Goodhope rangelands could probably be attributed to local environmental conditions, especially rainfall and soil fertility that favours high growth by dominant herbaceous plant species and thus limit tree seed germination, seedling establishment and survival through competition for water and soil nutrients (Hagenah *et al.*, 2009). The alternative explanation could be that woody plants are cut for fuel-wood and construction material by local communities (Nkambwe and Sekhwela, 2006) because the site is closer to a highly populated major village than other sites and some cuttings were observed during fieldwork.

The woody plant cover observed at Matlolakgang communal grazing land and ranching system was significantly higher than the 40% considered equilibrium between the encroached and non-encroached rangeland condition (Roques *et al.*, 2001; Dalle *et al.*, 2006). The results of the present study indicate that both the communal grazing land and ranching system at Matlolakgang are encroached by woody plants, which is supported by the dominance of encroacher species such as *Terminalia sericea* and *Dichrostachys cineria* (van Vegten, 1984). A high density of woody plant seedlings was observed in both the communal grazing land and ranching system suggesting that the problem is likely to increase if no control measures are taken to address the problem as suggested by pastoralists as would be discussed in Chapter 5. Previous studies also found high woody plant cover in overgrazed areas (Skarpe, 1990b), suggesting that overgrazing could be causing bush encroachment at Matlolakgang rangelands especially that stocking intensity in communal lands were high as 5 hectares per livestock unit (Chapter 1).

The data collected on woody plant cover at Xanagas showed that rangelands are crossing the threshold (40%) between encroached and non-encroached rangeland condition. The woody plant density was above the critical threshold of 2400 plants ha $^{-1}$ (Roques *et al.*, 2001) in both the communal grazing land and ranches suggesting that bush encroachment is already a problem at this site. The woody plant cover observed at Xanagas rangelands was slightly lower than the 50% and 70% woody plant cover reported for overgrazed areas nearby at Ncojane and Ghanzi, respectively (Skarpe, 1990a). However, extremely high woody plant cover of 89%, dominated by encroacher species such as *Acacia mellifera, D. cineria* and *T. Sericea,* was observed along one transect in communal grazing land at Xanagas, which is consistent with Skarpe's (1990a) findings for neighbouring areas. This also demonstrates that encroached patches of land in a particular landscape vary in size, which could reflect different stages of a cyclic succession (Smith and Smith, 2001; Wiegand *et al.*, 2005).

There was no significant difference in woody plant cover and density between the communal and ranching system, which is consistent with other studies (Smet and Ward, 2005; Tefera *et al.*, 2008b). However, our results contradict observations in South Africa (Wigley *et al.*, 2010), which showed that communal grazing land had lower woody cover than ranches, probably because woody plants in the communal grazing land are felled and utilised by local communities for firewood and construction material (Wigley *et al.*, 2009; Wessels *et al.*, 2011) and also from browsing pressure. The results of the present study suggest that bush encroachment is a common problem in both the communal grazing land and ranching system. This is contrary to assumptions of the TGLP that bush encroachment occurred mostly in communal grazing land due to overstocking and poor rangeland management (Botswana Goverment, 1975; Moleele and Perkins, 1998) and that ranches would promote proper utilisation of rangeland resources and limit land degradation (Botswana Goverment, 1975). Contrary to the assumptions of the TGLP, other studies have observed higher degradation in ranches than communal grazing land (Vanderpost *et al.*, 2011) which could be attributed to overstocking and poor management.

The woody plant density at all sites indicated that few trees/shrubs grew above 2 m high probably due to browsing pressure since all of the dominant woody plants are excellent browse species (Ringrose and Matheson, 1991; Moleele, 1998). Subsequently, browsing could also be responsible for proliferation of woody plants through seed dispersal by livestock (Tews *et al.*, 2004). The significant differences in woody cover between the three sites observed in this study could suggest that bush encroachment in rangelands is best explained by site-specific variables such as soil and rainfall variability.

4.4.2 Woody vegetation diversity in relation to rangeland management systems

The results of this study showed that overgrazed rangelands had high woody vegetation diversity. The bush encroached Matlolakgang rangelands had the highest species numbers, though the number is lower compared to the species richness observed at Letlhakeng rangelands (Ringrose and Matheson, 1991), which are close to Matlolakgang. The high species richness at Matlolakgang rangelands could probably be attributed to disturbances caused by overgrazing, which reduces competition from herbaceous plants (Jacobs and Naiman, 2008), thus providing a favourable environment for seeds of different woody plants deposited with animal faeces to germinate and establish, especially after rainfall (Kraaij and Ward, 2006). Browsing pressure in overstocked rangelands could also increase mortality in dominant, but palatable woody species and reduce interspecific competition, thus allowing a range of less dominant woody species to establish. This could eventually lead to increase in woody plant species diversity (Reyes *et al.*, 2010). The lowest woody plant species richness was observed at Goodhope, despite this site having the highest mean annual rainfall, and therefore contradicts other studies which reported a positive influence of precipitation on woody plant species richness (Speziale *et al.*, 2010).

The lack of a significant difference in woody plant species richness between the communal grazing land and ranching system at Goodhope and Xanagas could suggest that both systems have limited influence on woody plant diversity. It is possible that seed germination and establishment of woody plants is strongly influenced by soil and rainfall (Wiegand *et al.*, 2005). But, the results of present study are in agreement with (Dahlberg, 2000b) who found no significant difference in woody plant species richness between the communal grazing land and ranches in the north-east district of Botswana. The Shannon-Weiner index did not show significant differences between management systems and was generally low since the rangelands were mostly dominated by a few woody plant species (Waite, 2000). This increased the probability that the next individual plant randomly selected would be the same as previous one in each rangeland (Smith and Smith, 2001).

4.4.3 Rooting systems of encroacher and non-encroacher woody vegetation

The difference in root characteristics of woody vegetation facilitates coexistence with other plant species because they allow the use of various resources at different soil depths (Smith and Smith, 2001). The results of this study showed that selected dominant woody plant species at all study sites differed significantly in root type and most of them, particularly *G. flava* had several lateral roots spreading within 50 cm depth from the soil surface. According to (Hipondoka *et al.*, 2003) both the grass and woody vegetation roots occupy the surface soil layers in the Kalahari sandveld of Botswana. The dominance of shallow rooted woody plants has been attributed to their ability to absorb nutrients (Tolsma *et al.*, 1987) and soil moisture (Cole and Brown, 1976; Hipondoka *et al.*, 2003) in the surface soil layers. This contradicts the two layer hypothesis, which suggests that woody plants monopolise the use of soil moisture at deeper soil layers (Walker *et al.*, 1981).

Although most woody plant species in this study had lateral roots, few had taproots, which make it difficult to compare encroacher species and their non-encroacher counterparts such *Bauhinia petersiana*. The variation in the rooting systems did not follow any clear pattern (e.g. rainfall gradient) and therefore could be attributed to individual plant response to micro habitat condition such as soil moisture, fertility and competition (Hipondoka and Versfeld, 2006). For example, in this study all samples of *Terminalia sericea* had taproots, which contradicts other studies findings (Hipondoka and Versfeld, 2006; Holdo and Timberlake, 2008). However, the growth of *T. sericea* roots is known to be adaptive to local conditions and had exhibited taproots in other savanna ecosystems across Southern Africa (Hipondoka *et al.*, 2003) as observed in this study. The rooting depth and distribution of tree species could be strongly influenced by edaphic factors (Holdo and Timberlake, 2008) and probably in the case of our results sandy soils allowed *T. sericea* to develop a taproot capable of accessing soil moisture that escape lateral roots in the surface soil.

4.4.4 Relationships between woody vegetation dynamics and environmental variables

Woody plant cover and species richness in this study were significantly associated with mean annual rainfall and decreased with increasing rainfall which contradicts results of other studies (Scholes *et al.*, 2002). Continental scale analysis of African savanna ecosystems have revealed that woody plant cover increases with mean annual rainfall in arid and semi-arid ecosystems with annual rainfall below 650 mm (Sankaran et al., 2005). Even though our results showed a negative relationship between woody cover and species richness whilst other study showed positive relationship (Sankaran *et al.*, 2008) it still confirm that rainfall is a crucial factor in regulating woody vegetation growth in savanna ecosystems. The woody plant cover at Goodhope was low despite higher rainfall than other sites and this could be an indication that factors other than rainfall are more important in regulating woody community structure. Alternative explanation of low woody plant cover at Goodhope could be previous land use such as arable farming that is common at Goodhope such that one transect had to be abandoned during the survey after it was discovered that the area was recently ploughed. Previous studies have demonstrated that past land use practices increased woody plant cover at different rates in a South African savanna (Wigley *et al.*, 2010).

A multiple factor analysis showed that edaphic factors, particularly clay content were the most important predictors of woody vegetation cover, density and diversity in a non-linear model. Matlolakgang and Xanagas rangelands, located in the Kalahari sandveld, showed higher woody vegetation cover and diversity than Goodhope, located in the hardveld. These results are consistent with observation in other African savanna systems (Sankaran *et al.*, 2008), though these authors identified rainfall as the key determinant of woody plant cover. Sankaran *et al.* (2008) also observed a negative relationship between woody plant cover and soil clay content, which supports our findings. Water infiltration in sandy soils is higher than in clay soils, even in the absence of vegetation, because the soil absorbs rainwater well and uniformly (Rietkerk *et al.*, 1997). In semi-arid environments, where soil water limits plant growth, good infiltration therefore enhances the development of a continuous vegetation cover during good rainy years (Rietkerk *et al.*, 1997). As a result, other researchers (Sankaran *et al.*, 2005) suggested that the Kalahari sandy soils sustain higher woody vegetation cover, which again supports findings of the present study.

The results of our study also showed that the increases in woody plant cover lead to decline in herbaceous biomass. Goodhope rangelands, which had the lowest woody plant cover, showed the highest herbaceous biomass (Chapter 3). The reduction in herbaceous biomass in relation to increase in woody vegetation cover has also been reported in studies conducted elsewhere (Dalle *et al.*, 2006) and is consistent with pastoralists perception to be discussed in Chapter 5. Herbaceous biomass reduction through grazing create space and reduces competition for resources, which allows tree seedlings to develop (Kraaij and Ward, 2006) and eventually accelerates bush encroachment. The reduction in herbaceous biomass threatens the sustainability of the livestock industry, especially cattle that are dependent on grass productivity in the rangelands. However, introducing browsers such as goats in bush encroached rangelands could help reduce the woody plant cover (Augustine and McNaughton, 2004) and would also diversify livelihood of pastoral farmers.

4.5 Conclusion

Rangeland management systems had no significant effects on woody vegetation cover, density and species richness, which is contrary to the assumptions of the Tribal Grazing Land Policy of Botswana. Bush encroachment is a problem at Matlolakgang and Xanagas rangelands, which are dominated by encroacher plant species, but not Goodhope rangelands. The woody plant diversity was high in encroached rangelands, but not different between the communal and ranching grazing land. Dominant woody plant species had different root types and mostly had shallow lateral roots. There was no clear distinction between root types of encroacher and non-encroacher plant species. The heterogeneity in woody cover, density and species richness were influenced by site factors such as soil type and rainfall. The soil texture, particularly clay content, was strongly related to woody cover and subsequently, rangelands located in Kalahari sandveld are more likely to experience increase in woody cover than those located in hardveld.

Chapter 5

Pastoralists' perception and ecological knowledge on grazing savanna ecosystem in semi-arid Botswana

5.1 Introduction

The savannas are largely used for communal grazing, and thus are considered to be degraded due to poor management by pastoralists (Hardin, 1968; Fernandez-Gimenez, 2000). Overgrazing by livestock promotes increase of undesirable herbaceous plant species and bush encroachment, which are all indicators of rangeland degradation (Katjiua and Ward, 2007). However, the perspective and interpretation of rangeland degradation are highly controversial (Abel and Blaikie, 1989; Dougill *et al.*, 1999; Dahlberg, 2000b; Skarpe, 2000) and dominated by researchers and extension workers (Dougill *et al.*, 2002; Reed and Dougill, 2010). Pastoralists' perception and ecological knowledge of vegetation changes are often ignored (Roba and Oba, 2009) despite the debate in their role in rangeland degradation (Butt, 2010). Yet, pastoralists are known to have extensive ecological knowledge (Berkes *et al.*, 2000; Ladio and Lozada, 2009) which could complement scientific knowledge (Oba and Kotile, 2001) and contribute to improved understanding and sustainable management of savanna ecosystems (Reed *et al.*, 2011).

In Botswana, livestock production is a dominant economic activity practiced in communal and ranching lands (Masike and Urich, 2008). The general perception of policy makers is that the ranching management system is environmentally sustainable and therefore a solution to overgrazing in communal land (Botswana Goverment, 1975). However, that perspective is challenged by non-equilibrium theory, which attributes vegetation changes in arid environments mainly to stochastic abiotic factors, particularly rainfall variability (Ellis and Swift, 1988; Westoby *et al.*, 1989). This creates doubts on the relevance of rangeland carrying capacity, stocking rates and grazing policy, which are considered to be key to sustainable management of ranches (Ellis and Swift, 1988). Although both the communal and ranching lands are well researched (Thomas *et al.*, 2000), there are limited long term studies that compare vegetation conditions between grazing management systems. Given that pastoralists interact with their environment regularly and share ecological knowledge between generations (Roba and Oba, 2009), they could provide a long-term ecological perspective of vegetation changes in rangelands and also of the underlying causes (Bart, 2006) which is normally lacking in most ecological studies (Brook and McLachlan, 2008; Knapp and Fernandez-Gimenez, 2008).

Pastoral communities are known to have elaborate knowledge of plant species composition, their palatability to grazing animals and also of trends in vegetation change in response to factors such as rainfall variability and grazing pressure (Davis, 2005; Oba and Kaitira, 2006). This knowledge and perception has significant influence on the management strategies adopted to exploit a particular ecosystem (Ellis and Swift, 1988). Pastoral communities' participation in addressing rangeland degradation also depends on their perceived impact of degradation on livestock production (Mbaiwa and Stronza, 2011). Hence, it is essential to understand how pastoral farmers perceive rangeland conditions and the extent of degradation in both communal grazing land and ranches. Common understanding of rangeland degradation by both researchers and pastoral communities is crucial for combating rural poverty and environmental degradation (Kessler and Stroosnijder, 2010) and is relevant to all stakeholders. More complete understanding would allow development of rangeland management strategies that are effective and justified because both land users and policy makers are then aware of degradation problems (Kessler and Stroosnijder, 2006).

Previous studies conducted in Botswana had already provided broad knowledge of local perceptions on environmental degradation associated with dominant livelihoods such as arable farming and veldt products (Chanda, 1996; Ringrose *et al.*, 1996). However, community-based ecological knowledge differs based on interests like veldt products and grazing (Oba *et al.*, 2008) and this study specifically focuses on knowledge and perception of vegetation dynamics and livestock grazing in both communal grazing lands and ranches. Traditional ecological knowledge (TEK) is unique to different places and communities (Fernandez-Gimenez, 2000) and even to different individuals within a community. Therefore, examining the pastoralists' ecological knowledge under different cultural and environmental conditions is expected to provide a broader understanding of ecosystem dynamics. The objectives of the present study were to use pastoral ecological knowledge to (i) characterise the composition of grass species (ii) characterise the dynamics of woody vegetation and

(iii) compare effects of rangeland management systems on vegetation and livestock production in different ecological systems.

5.2 Material and methods

5.2.1 Study area

The study was carried out in three principal villages or ranch (Xanagas, Matlolakgang and Goodhope) located in different regions of Botswana as described in Chapter 1. Communal grazing lands do not have well defined boundaries and are shared by communities from several different villages. Subsequently, our sampling area at Xanagas included pastoral farmers from Karakubis and Charleshill, while Matlolakgang sample area consists of Ngware, Malwelwe and Molepolole. In Goodhope, the sampled farmers included those from Sheepfarm and Metlojane. The study sites also represent different cultural settings (e.g. Bakgalagadi, Basarwa, Baherero, Bakwena, Barolong and other Tswana tribes). The three areas are located in savanna biome, but are characterised by different biophysical features (e.g. rainfall and soil type) as described in Chapter 1.

5.2.2 Sampling and data collection

Data on vegetation dynamics in grazing lands were collected through a structured questionnaire (Appendix 4) between May and June 2009. The structured questionnaire was selected because it allowed the researcher to address specific objectives (Huntington, 2000) and also minimised differences between interviews by having standardised questions (Bryman, 2004). The interviews were conducted face-to-face with individual pastoral farmers using the national language of Botswana, Setswana, which is understood throughout the country (Mbaiwa and Stronza, 2011). Some local grass species' names are not specific enough and therefore sometimes the pastoralists were asked to show us the grass species in the field where possible or describe it in more details to allow easy identification. A total of 125 respondents from the pastoral communities in the three study sites were selected; Goodhope (n=38), Matlolakgang (n = 42), and Xanagas (n = 45). Initially the target sample size per site was 50, but this was not achieved because some pastoralists were away from both home and cattle-posts during the survey period. There was no formal list of pastoral farmers at each site and therefore respondents were selected from alternating households from a random starting point (household) to ensure that households had fair chance of being selected irrespective of their social background.

The pastoralists had to characterise the grass composition within their grazing area by describing; (i) dominant species; (ii) grazing desirability or palatability of dominant grasses as very palatable, palatable or unpalatable (see Appendix 3, section

3). They also had to describe (iii) trends of individual grass species by indicating whether a particular species had increased (increasers), decreased (decreaser) or remained stable (stable) within the past 5 years or historical knowledge. Other authors (Roba and Oba, 2009) had indicated that pastoral farmers commonly use terms such as increasing, not changing and decreasing during assessment of species trends. Pastoralists were also asked whether grass composition has changed historically and the causal factors linked to the observed changes.

In order to characterize woody vegetation within their grazing lands, pastoralists were asked to describe; (i) dominant woody plant species (ii) their trend as stable, increasing or decreasing; (iii) their suitability (palatability) to grazing animals as very palatable and not palatable (Appendix 3, section 4). Pastoral farmers were also asked whether bush encroachment was a problem within their grazing lands, its impact and current management strategies meant to address the problem. Woody plant species associated with bush encroachment were also identified by pastoralists. The association between rangeland degradation and grazing management systems was established by asking whether observed degradation was common in either communal or ranching grazing lands. The pastoralists were further asked to indicate whether the perceived changes in grass composition and bush encroachment were more pronounced in any particular grazing management system.

5.2.3 Data analysis

The collected data on rangeland condition were analysed quantitatively in Statistical Package for the Social Sciences (SPSS 15). Cross-tabulation and chi-square test (χ^2) were used to determine differences between pastoralists' responses between sites. The response rates to some questions such as plant palatability to livestock were low to the extent that it was not possible to compare between sites. In such cases, the responses were grouped together irrespective of study sites and general pattern reported. A comparison between young ($\leq$30 years) and old pastoralists' perception of changes in grass composition and bush encroachment was determined using chi-square test.

5.3 Results

5.3.1 Grass compositions as perceived by pastoralists

Pastoralists identified a range of different grasses as dominant at the three sites (Table 5.1). However, *Aristida congesta* and *Stipagrostis uniplumis* were considered dominant by more than half of respondents in all three grazing lands, with *A. congesta* being identified as dominant by 97.4 % and 88.9 % of pastoralists at Goodhope and Xanagas respectively. A significant proportion of pastoral farmers (81 %) at Matlolakgang identified *Megaloprotachne albescens* as a dominant grass in their grazing area. All pastoralists at Goodhope mentioned *Cynodon dactylon* as abundant grass species in their grazing lands. Pastoralists' perception and knowledge of the palatability of dominant grasses was not widely shared among pastoral communities as shown by differences in response rate (Table 5.2). However, the majority of pastoralists classified only *Cynodon dactylon* and *Schmidtia pappophoroides* as very palatable, while most of the dominant grasses were largely perceived as only moderately desirable for grazing. Among dominant grasses, *A. congesta* and *M. albescens* were mainly perceived as undesirable for grazing.

Table 5.1: Proportions of pastoralists that considered different grasses dominant in their grazing lands

List of grasses		% of respondents who named grass species as dominant				
Scientific name	Local name	Goodhope (n = 38)	Matlolakgang (n = 42)	Xanagas (n = 45)	χ^2	P value
Aristida congesta	Seloka	97.4	52.4	88.9	28.5	***
Cenchrus ciliaris	Mosekangwetsi	10.5	0	2.2	NA	NA
Cynodon dactylon	Motlho	100	19	2.2	93.2	***
Eragrostis pallens	Motsikiri	68.4	26.2	0	46.6	***
Eragrostis rigidior	Rathethe	26.3	31	0	16.1	***
Megaloprotachne albescens	Mohaha	0	81	0	92.3	***
Schmidtia pappophoroides	Tshwang	0	11.9	55.6	39.9	***
Stipagrostis uniplumis	Tshikitshane	55.3	59.5	53.3	0.35	NS
Urochloa trichopus	Phoka	0	2.4	55.6	51.6	***

NA => 20 % cells has expected values less than 5;***P< 0.001, NS P>0.05

Table 5.2: The grazing palatability of dominant grasses as reported by pastoralists in Botswana

Herbaceous species	No of Respondents	% Response		
		Very palatable	palatable	Unpalatable
Aristida congesta	88	4.5	44.3	51.2
Cenchrus cilairis	6	0	66.7	33.3
Cynodon dactylon	45	86.7	11.1	2.2
Eragrostis pallens	24	0	83.3	16.7
Eragrostis rigidior	19	21	73.7	5.3
Megaloprotachne albescens	32	15	31.3	53.1
Schmidtia pappophoroides	19	57.9	36.8	5.3
Stipagrostis uniplumis	60	30	58.3	11.7
Urochloa trichopus	21	47.6	47.6	4.8

The rangelands at Goodhope and Matlolakgang were mainly (63 and 73 % respectively) perceived to have undergone changes in grass compositions, which significantly differed with the perception of Xanagas pastoralists who largely (53.3 %) indicated that grass composition has not changed (χ^2=22.3, df=4, P<0.001). This perception was consistent between young (≤30 years) and older pastoralists (χ^2 = 4.4, df = 2, P > 0.05) but a significant number of pastoralists at Goodhope (21.1 %) did not know whether grass composition has changed (Figure 5.1). The change in grass composition was characterised by an increase in unpalatable grasses such as *A. congesta* and *M. albescens* and a decline in moderately palatable grasses such as *E. pallens*, *E. rigidior* and *S. uniplumis* (Table 5.3). The changes in grass composition were mainly attributed to rainfall variability (Table 5.4) especially at Goodhope (63.2 %) and Xanagas (82.2 %). Overgrazing was another perceived causal factor for changes in grass composition particularly at Goodhope (60.5 %), though less so at Xanagas where fire was suggested by 57.8 % of respondents.

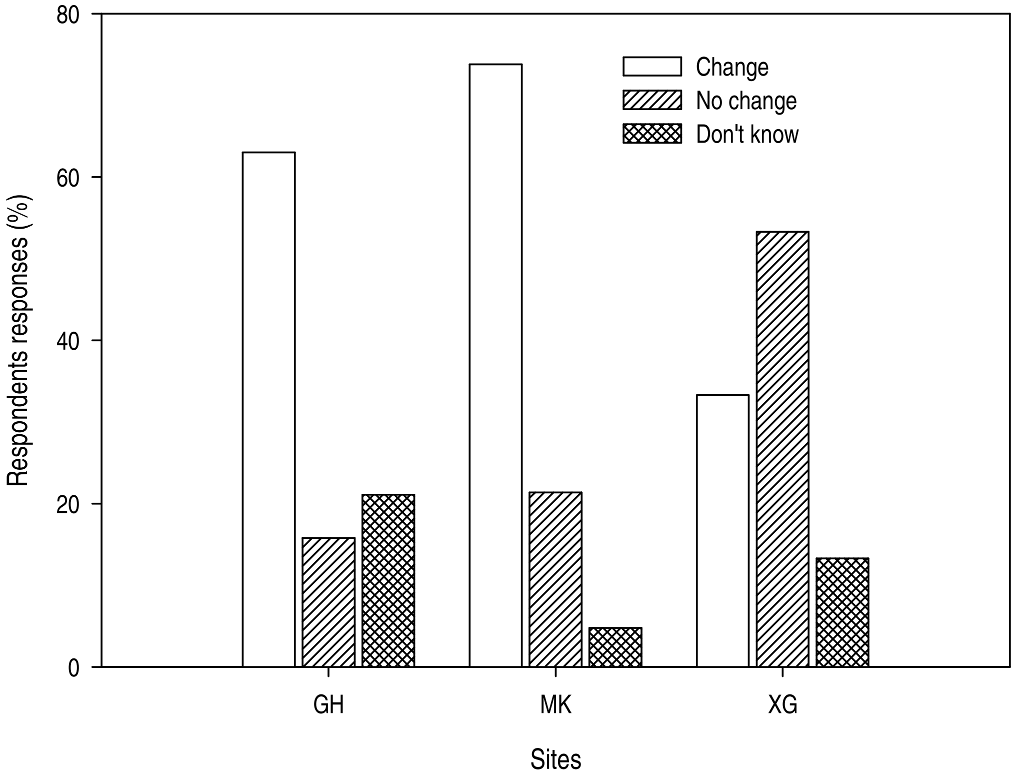

Figure 5.1: Proportions of pastoralists' perceptions on changes in herbaceous composition in the last 5 years

Table 5.3: Trends of herbaceous plant species in all rangelands as suggested by pastoralists

Species name	No of respondents	% response on trends of herbaceous species		
		Increasing	Stable	Decreasing
Aristida congesta	56	50	41.1	8.9
Cenchrus dactylon	32	31.3	62.5	6.3
Eragrostis pallens	26	19.2	26.9	53.8
Eragrostis rigidior	17	0	35.3	64.7
Megaloprotachne albescens	20	70	10	20
Stipagrostis uniplumis	33	3	12.1	84.8

Table 5.4: Factors that are considered to cause changes in composition of grasses in the last 5 years

Factors	% of respondents at different sites				
	Goodhope (n= 38)	Matlolakgang (n= 42)	Xanagas (n = 45)	χ^2	P value
Fire	7.9	7.1	57.8	38.2	***
Overgrazing	60.5	23.8	15.6	21.1	***
Rainfall	63.2	31	82.2	24	***

***P< 0.001

5.3.2 Woody vegetation composition as perceived by pastoralists

Pastoralists identified different woody plant species as dominant within their grazing lands though *Acacia* species were abundant in all regions (Table 5.5). A few woody plants species such as *Acacia burkei* (31 %) and *Dichrostachys cinerea* (78.6 %) were perceived to be dominant by pastoralists at Matlolakgang only, while *Boscia albitrunca* was considered dominant by 65.8 % of pastoralists at Goodhope. *Terminalia sericea* was largely perceived to be dominant by pastoralists grazing in Matlolakgang (83.3 %) and Xanagas rangelands (95.6 %) both of which are located in sandveld (Table 5.5). Pastoralists' generally acknowledged the importance of woody vegetation as a browsing resource, though not all woody plants were indicated as suitable for browsing (Table 5.6). The *Boscia albitrunca,* dominant at Goodhope, was largely considered very suitable for browsing by 80.8 % of pastoralists. Most dominant woody plant species were considered moderately desirable for browsing, with the exception of *Acacia erioloba*, which was largely (64.3 %) perceived as unsuitable for browsing.

Table 5.5: The dominant woody plants species as perceived by pastoralists

Scientific name	Local name	% of presence response at different sites				
		Goodhope (n= 38)	Matlolakgang(n = 42)	Xanagas (n = 45)	χ^2	P value
Acacia burkei	Mokgwa	0	31	0	NA	NA
Acacia erioloba	Mogotlho	86.8	23.8	28.9	39.3	***
Acacia hebeclada	Sekhi	73.7	4.8	68.9	49.3	***
Acacia mellifera	Mongana	63.2	14.3	75.6	35.8	***
Acacia tortilis	Mosu	97.4	14.3	0	97.9	***
Boscia albitrunca	Motlope	65.8	16.7	0	49.5	***
Dichrostachys cinerea	Moselesele	0	78.6	0	88.6	***
Grewia flava	Moretlwa	92.1	21.4	71.1	44.9	***
Lonchocarpus nelsii	Mhata	0	47.6	0	47.1	***
Ochna pulchra	Monyelenyele	0	9.5	0	NA	NA
Terminalia sericea	Mogonono	21.1	83.3	95.6	54.5	***
Ziziphus mucronata	Mokgalo	52.6	7.1	77.8	44.4	***

***P< 0.001, NA = > 20 % cells has expected values less than 5

Table 5.6: Browsing suitability of different woody plants as suggested by pastoralists

Woody plant species	No of Respondents	% Response		
		Very suitable	Suitable	Unsuitable
Acacia burkei	9	---	55.6	44.4
Acacia erioloba	42	7.1	28.6	64.3
Acacia hebeclada	36	27.8	72.2	---
Acacia mellifera	36	25	61.1	13.9
Acacia tortilis	38	7.9	78.9	13.2
Boscia albitrunca	26	80.8	15.4	3.8
Dichrostachys cinerea	29	24.1	44.8	31
Grewia flava	54	31.5	50	18.5
Lonchocarpus nelsii	17	58.8	41.2	---
Ochna pulchra	5	40	40	20
Terminalia sericea	69	18.8	66.7	14.5
Ziziphus mucronata	45	60	37.8	2.2

Bush encroachment was perceived as an environmental problem by the majority of both young (≤30 years) and old pastoralists (χ^2=2.33, df = 1, P>0. 05) at all sites, though the perception differed significantly between sites (χ^2=9.94, df = 2, P<0.01). Most pastoralists (88.1 %) at Matlolakgang considered their rangelands to be highly encroached by woody vegetation. Goodhope and Xanagas rangelands were perceived to be encroached by bushes by 68.4 % and 57.8 % of respondents respectively. Most of the dominant woody plant species identified as encroachers were increasing in respective rangelands, with the exception of *B. albitrunca and O. pulchra,* which were largely considered to be declining in the rangelands (Table 5.7). *Dichrostachys cinerea* was largely (94.4 %) considered by pastoralists to be increasing in Matlolakgang rangelands.

Table 5.7: Trends of dominant woody plants density in rangelands of Botswana

Name of woody plant	No of respondents	Encroachers	% respondents		
			Increasing	Stable	Decreasing
Acacia erioloba	37	**	10.8	64.9	24.3
Acacia hebeclada	41	**	63.4	31.7	4.9
Acacia mellifera	46	**	47.8	34.8	17.4
Acacia tortilis	28	**	57.1	39.3	3.6
Boscia albitrunca	22		--	9.1	90.9
Dichrostachys cinerea	18	**	94.4	5.6	---
Grewia flava	53	**	45.3	34	20.8
Lonchocarpus nelsii	5		40	20	40
Ochna pulchra	4		----	25	75
Terminalia sericea	51	**	60.8	23.5	15.7
Ziziphus mucronata	44	**	50	25	25

** identified as major encroachers

The pastoralists' perceptions of the effect of bush encroachment on herbaceous vegetation at Goodhope, Matlolakgang and Xanagas differed significantly ($\chi^2 = 15.3$, df = 4, P=0.004). The majority of respondents at Goodhope (68.4 %), Xanagas (66.7 %) and Matlolakgang (47.6 %) indicated that bush encroachment led to reduced primary grass production (Figure 5.2). However, some pastoralists indicated that bush encroachment had no effect on grass productivity, almost a third of the respondents at Xanagas, while some 28.6 % pastoralists at Matlolakgang indicated that they do not know the effects of bush encroachment on herbaceous vegetation. There was a strong agreement among pastoral farmers at all sites on the current control measures of bush encroachment ($\chi^2 = 8.6$, df = 6, $P<0.05$). Most pastoral farmers (80.8 %) indicated that nothing was being done to mitigate bush encroachment, while only 14.4 % of them indicated that they were cutting down trees to control bush encroachment. Only 3.2 % of the farmers indicated that they used rotational grazing whilst and 1.6 % mentioned use of fire to control bush encroachment.

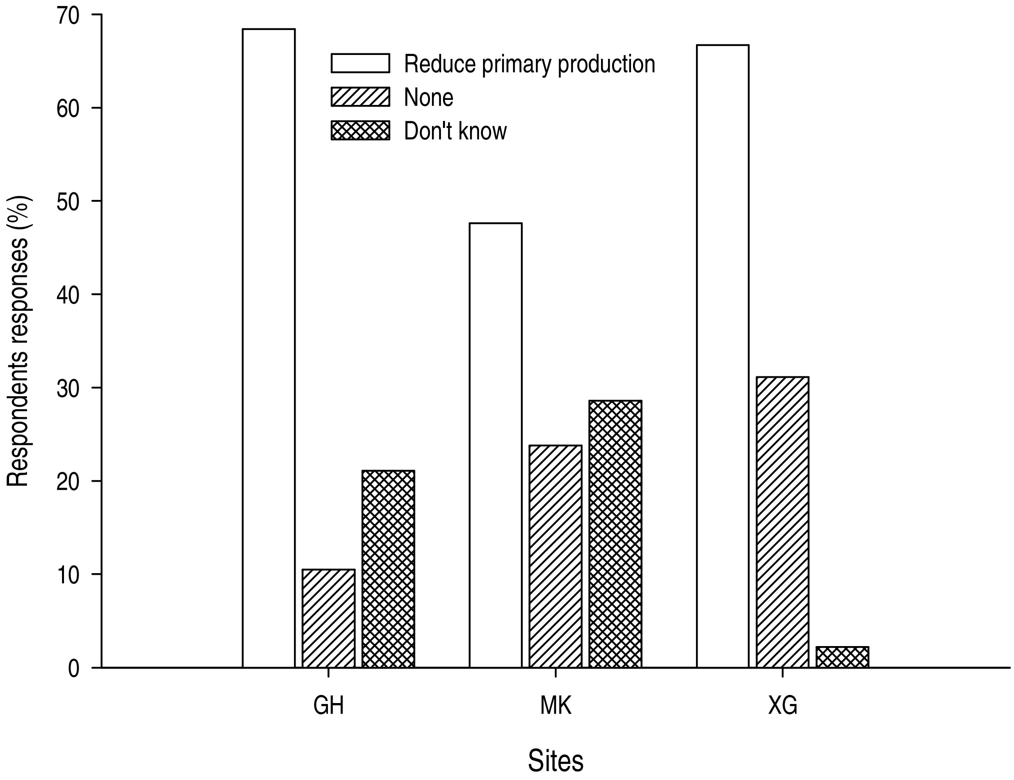

Figure 5.2: Impact of bush encroachment on herbaceous vegetation at Goodhope (GH), Matlolakgang (MK), and Xanagas (XG) as suggested by pastoral farmers

5.3.3 Effects of rangeland management systems on vegetation and livestock production

The pastoral farmers' perception of the effect of rangeland management systems on vegetation changes differed both locally and between sites. The changes in grass species composition were considered more pronounced in communal grazing land than ranches especially at Matlolakgang (Figure 5.3a). But, many pastoralists particularly at Xanagas (31.1 %) indicated that changes in grass composition were also common under the ranching system, while others indicated that both management systems are affected similarly (Figure 5.3a). The pastoral farmers' perception also differed strongly on the occurrence of bush encroachment in both communal and ranching rangelands between the study sites ($\chi^2 = 30.8$, df = 6, $P<0.001$). Bush encroachment was considered to be a problem in communal grazing areas at all sites (Figure 5.3b), though some pastoralists particularly in Xanagas (37.8 %) also reported increase in woody vegetation under the ranching system. However, 36.8 % of pastoralists at Goodhope did not know whether bush encroachment was a problem in either of the two systems. Pastoralists largely (70.4%) considered communal grazing lands to be more prone to degradation than ranches (5.6 %), while 20.8 % of them reported that both systems have a similar degrading effect on the environment. Ranches were considered more to be appropriate for livestock production than a communal grazing system by 74.4% of respondents. About 18.4 % of the pastoral farmers perceived both rangeland management systems as suitable for livestock production, while only 7.2 % of pastoralists favoured the communal grazing system.

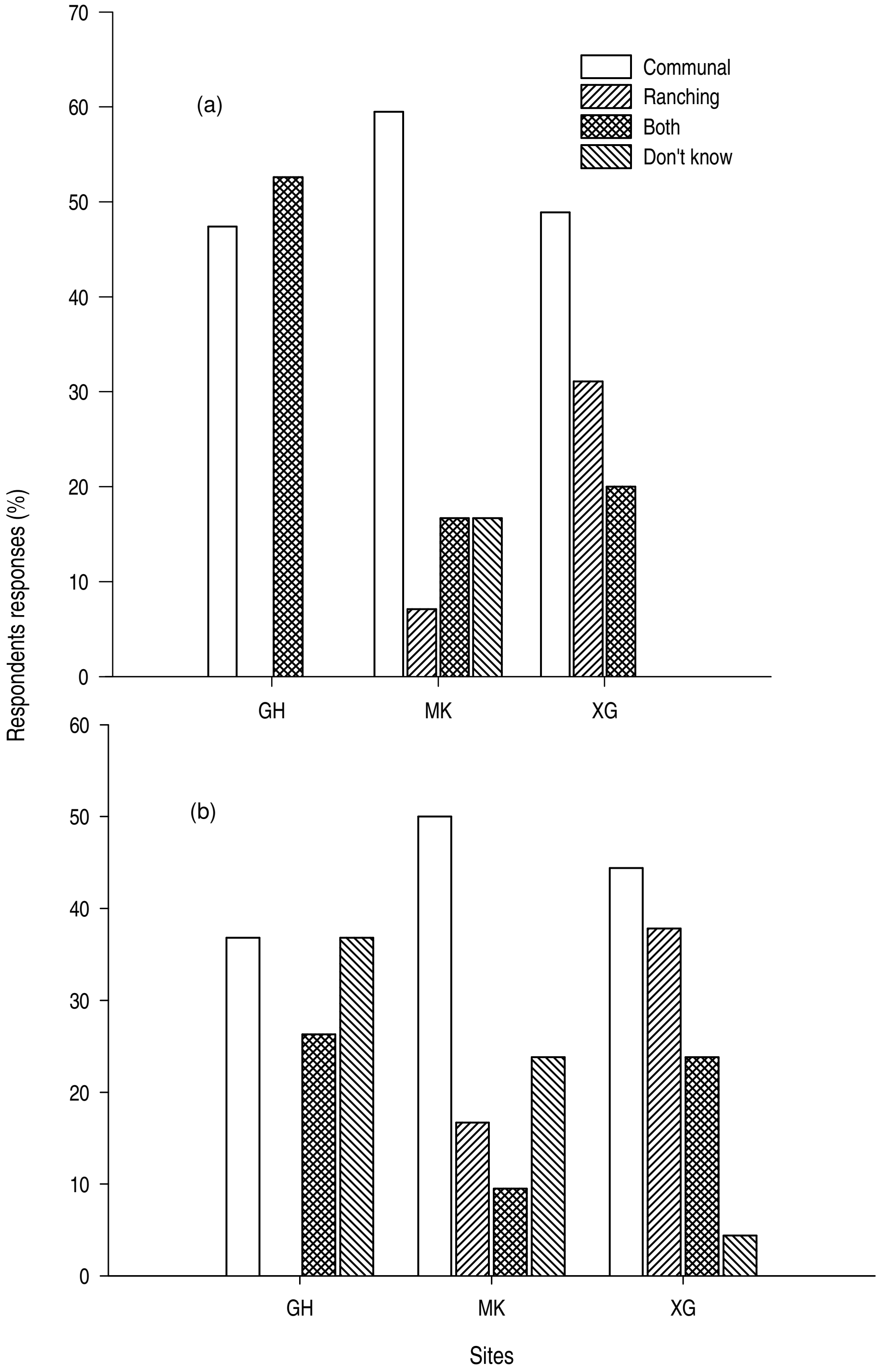

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Figure 5.3: Grass composition changes (a) and bush encroachment (b) in relation to management systems as perceived by pastoral farmers at Goodhope (GH), Matlolakgang (MK) and Xanagas (XG)(zero columns= no respondents).

5.4 Discussion

The results of this study show that most pastoral farmers are knowledgeable of grass composition, palatable plant species, trends of vegetation and their causes within their local environment as suggested in the literature (Fernandez-Gimenez 2000). However, pastoralists' knowledge is not uniformly shared within or between pastoral communities as indicated by lack of agreement on some aspects of vegetation dynamics, which is consistent with other studies (Knapp and Fernandez-Gimenez, 2008; Kassam, 2009). This could be partly attributed to loss of local ecological knowledge that is not regularly used because herding, which refer to controlled movement of animal by a herder, is no longer practiced in Botswana. The retention of local ecological knowledge depends on its regular use and therefore lack of its regular use may lead to loss of specific knowledge (Kassam, 2009) which could be the case in Botswana.

5.4.1 Grass compositions as perceived by pastoralists

Pastoralists' responses showed significant differences in dominant grass species in their respective grazing lands even at local level, which probably suggest that rangelands are spatially heterogeneous and supported by ecological data (Chapter 3). The heterogeneity of the grass composition across the landscape was most likely to be influenced by soil characteristics, variation in grazing pressure (Cole and Brown, 1976; Roba and Oba, 2009) and rainfall variability (Thomas and Twyman, 2004).

Thus, different patches of savannas observed in present study are either in different successional states or in transitions as demonstrated by other authors (Westoby *et al.*, 1989). The dominance of *Aristida congesta*, which is an increaser II species (Trollope *et al.*, 1989), indicated that overgrazing is widespread across all grazing rangelands. The results also show that grazing rangelands had few very palatable grass species, which suggests that these species are overgrazed as reported in some regions of Botswana (Reed *et al.*, 2008).

The pastoralists particularly at Goodhope and Matlolakgang indicated that the composition of grass species had changed in their grazing land. The general trend was that palatable grass species such as *E. rigidior,* and *S. uniplumis* were declining and replaced by unpalatable species such as *A. congesta* and *M. albescens*, which indicate heavy grazing (du Plessis *et al.*, 1998). Similarly, Roba and Oba (2009) had reported that rangelands undergoing degradation tend to lose palatable grass species, which are then replaced by unpalatable species. However, the distribution of grasses is also dependent on species' ability to adapt to local conditions such as soil fertility and moisture (Muhumuza and Byarugaba, 2009) and therefore differences in dominant grass compositions and trends could also be explained by rainfall variability (Chapter 1) and soil fertility (Chapter 2) between sites.

In fact, the pastoralists mainly attributed changes in grass species composition to rainfall variability. They indicated that during years of good rainfall, there is sufficient forage for livestock, while poor rainfall leads to poor growth of herbaceous species, which in turn has adverse effects on livestock production. This observation is consistent with the non-equilibrium explanation (Ellis and Swift, 1988; Westoby *et al.*, 1989) and is supported by studies conducted in the northeast part of Botswana, where local communities indicated that changes in their ecosystem were caused by variable rainfall (Chanda, 1996; Dahlberg, 2000a). Pastoral farmers in other arid environments also attributed changes in rangeland condition to rainfall variability (Fernandez-Gimenez, 2000; Oba and Kaitira, 2006; Klintenberg *et al.*, 2007) which supports our finding. Given that climate change is likely to increase aridity in southern Africa (Eriksen and Watson, 2009b), then it could be argued that vegetation and pastoral production are likely to become more vulnerable to the effects of global climate change.

Overgrazing was another factor perceived by pastoralists to contribute towards change in grass composition, though more than 50 % of them did not associate vegetation changes with livestock populations. Chanda (1996) suggested that pastoral farmers tend to evade blame for overgrazing and this could explain why most of the respondents failed to mention overgrazing as a cause of vegetation changes in this study. Although fire is also known to regulate the composition of savanna vegetation (Furley *et al.*, 2008; Devineau *et al.*, 2010), it was scarcely mentioned by respondents in this study except at Xanagas. The possible explanation could be that the laws of Botswana do not allow the use of fire as a management tool in rangelands. In addition, the government has invested in the construction and maintenance of firebreaks to reduce fire incidences in rangelands. The other possible explanation is that livestock grazing removes fuel load which in turn reduces the possibility of fire (van Langevelde et al., 2003).

5.4.2 Woody vegetation composition as perceived by pastoralists

The perceived dominant woody plant species differed between the three study sites although *A. mellifera* and *G. flava* were widely distributed in all three sites. The discrepancy in pastoral farmers' perception of dominant woody plants could be explained by landscape patches and heterogeneity both within each site and between sites, which is a function of rainfall variability (Sankaran *et al.*, 2005) and soil fertility (Boone, 2005; Tefera *et al.*, 2008a). The abundance of *Boscia albitrunca* in Goodhope rangelands indicate that these rangelands are in good condition because the decrease in abundance of this species in grazing areas is considered to be a good indicator of rangeland degradation (Reed *et al.*, 2008). Most of the woody plant species, such as *A. mellifera*, *G. flava* and *Dichrostachys cinerea,* that were identified as dominant in grazing lands by pastoralists are associated with bush encroachment throughout the Kalahari ecosystem (Tews *et al.*, 2004; Thomas and Twyman, 2004) and are considered to be an indicator of rangeland degradation (Reed *et al.*, 2008).

Pastoralists further confirmed that bush encroachment was occurring within their rangelands in all sites. However, there was variability on the perceived level of bush encroachment, which suggests that the establishment and growth of woody plants is progressing at different rates across each study site. Pastoralists' descriptions suggest that the landscape consists of patches of land highly encroached or not encroached, as suggested by patch dynamics (Meyer *et al.*, 2009), and this was also confirmed during the ecological assessments in this study (Chapter 4). The pastoral farmers' perception of the occurrence of bush encroachment is supported by the results of studies conducted in other parts of Botswana (Moleele *et al.*, 2002) and elsewhere (Angassa and Beyene, 2003; Dalle *et al.*, 2006), where an increase in the density of woody plants was also reported.

Several dominant woody plants such as *G. flava*, and *D. cineria* were identified as responsible for bush encroachment, which is consistent with other studies in Botswana (Moleele and Perkins, 1998; Tews *et al.*, 2004) and other savanna ecosystems (Oba *et al.*, 2008). Furthermore, the pastoral communities indicated that the increase in encroacher woody plants in rangelands suppresses grass productivity, which reduce grazing area for livestock. This view is supported by the results of studies conducted in other parts of Botswana (Dougill *et al.*, 1999) and elsewhere (Angassa and Oba, 2008b), which also show that bush encroachment is a widespread environmental problem. Thus, bush encroachment threatens the livelihoods of pastoral communities (Angassa and Oba, 2008a) especially in developing countries like Botswana.

Despite being concerned by bush encroachment, pastoral farmers also appreciated the importance of woody plants as feed for livestock. They indicated that the increase in woody plants in rangelands was beneficial as an essential fodder resource that supports livestock as suggested by other authors (Moleele, 1998; Dougill *et al.*, 1999). Woody plants such as *B. albitrunca, G. flava* and *D. cineria* are recognized as an important component of cattle diet especially during dry periods because of their nutritional pods and their ability to flush early after the dry season, which is in agreement with observation by Moleele (1998). In addition, most of the woody plants that are considered to be suitable for browsing by pastoral farmers in the present study were leguminous species and are likely to have a high crude protein content because of their ability to fix nitrogen from the atmosphere (Moleele, 1998). The importance of woody plants as a fodder source could explain why most pastoralists are currently not doing much to control bush encroachment. However, the consumption of the pods of woody plants by livestock is likely to enhance bush encroachment through seed dispersal (Tews *et al.*, 2004). Thus, it is essential for pastoralists and researchers to work together to control bush thickening in savanna ecosystems.

5.4.3 Effects of rangeland management systems on vegetation and livestock production

The effects of communal and ranching management on vegetation change were as debatable among pastoralists as they are among ecologists. However, a significant proportion of pastoralists in the present study indicated that changes in grass composition and bush encroachment were more pronounced in communal grazing land, although the ranching system was also affected. Pastoralists acknowledged that the communal grazing lands are more degraded than ranches, but a satellite-based long-term assessment of Botswana rangeland conditions showed that ranches were more degraded than other land uses including communal land (Vanderpost *et al.*, 2011), which contradicts pastoralists' perception in this study. The Ethiopian Borana pastoralists also claimed that change from pastoral land use to ranches and private enclosures was a major cause of encroachment of woody vegetation in the rangelands (Dalle *et al.*, 2006; Angassa and Oba, 2008b) which contradicts perception of pastoralists in current study. Similarly, another study conducted in the Republic of South Africa showed that there was less bush encroachment in communal grazing areas because woody plants are cut and utilised by local communities (Wigley *et al.*, 2009). The Botswana pastoralists' perception that ranches promote rangeland conservation could have been influenced by the Tribal Grazing Land Policy (Botswana Goverment, 1975) promoted by the government of Botswana through its extension services, which promote ranching as sustainable grazing land use.

Most pastoralists indicated that the ranching management system was more suitable for livestock production. This could be partly attributed to dual grazing rights, which allows those allocated ranches to continue having access to communal grazing land (Tsimako, 1991). Hence, most pastoralists prefer to have ranches because it allows them to expand their herd sizes, but their management does not necessary change to promote ecosystem conservation (Motlopi, 2006). In addition, the pastoralists in Botswana no longer practice livestock herding (Reed *et al.*, 2008) like pastoralists elsewhere such as Masaai of Kenya and Borana of Ethiopia and therefore are likely to loose livestock in communal grazing lands. Therefore, fenced ranches provide a more secure alternative for rearing livestock especially since livestock owners spend more time away from the farm. However, privatization of common-pool resource such as communal rangelands by absentee landowners threatens both the sustainability of ecosystems and the livelihood of local communities (Altrichter and Basurto, 2008). Possibly, pastoral farmers of Botswana are unaware of the implications of privatization and the shrinking of communal rangelands.

5. Conclusion

Pastoralists were knowledgeable of vegetation changes in their grazing land though ecological knowledge was not uniformly shared. Their rangeland assessment indicated that savannas are highly heterogeneous spatially, dominated by different grasses and woody plant species, which is supported by the ecological assessment reported in this study in Chapter 3 and 4. Yet, loss of palatable grass and bush encroachment was common in most grazing lands, which indicate that rangeland degradation is occurring in some patches of grazing savanna. The pastoral communities largely attributed changes in grass composition to rainfall variability, though overgrazing and fire were also mentioned. The pastoralists' perception is that bush encroachment negatively affects livestock production by suppressing grass productivity, but also contributes positively as a valuable feed resource particularly during dry period. Pastoralists largely associated rangeland degradation with communal grazing lands, though some indicated that ranching lands were similarly affected. The results showed that pastoralists' knowledge could complement scientific knowledge and improve our understanding of traditional management practices. However, it is essential that pastoral communities and policymakers' work together to address changes in vegetation composition, particularly the control of bush encroachment.

Chapter 6

General discussion on effects of grazing management systems and environmental factors on savanna ecosystem

6.1 Introduction

Proper management of savanna ecosystems is essential to sustain environmental services, and pastoral livelihoods (Eriksen and Watson, 2009a). Sustainable agro-ecosystem management is dependent on improved understanding of the effects of different land use and environmental factors on ecosystems dynamics (Wallgren *et al.*, 2009). The current grazing policy of Botswana and other developing countries (e.g. Lesotho and Kenya) are based on few paradigms, which are highly controversial (Rohde *et al.*, 2006; Nyangito *et al.*, 2008). In Botswana, Tribal Grazing Land Policy (TGLP) of 1975 was introduced to halt rangeland degradation (Dougill *et al.*, 1999) which was perceived unavoidable in communal grazing lands as illustrated by "Tragedy of Commons" (Hardin, 1968). Rangeland degradation, which refers to a reduction of the biological or economic productivity of rangelands (Thomas, 2008) and vegetation diversity decline are indicators of ecological change (Detsis, 2010) as a result of livestock overgrazing (Cheng *et al.*, 2011) particularly under traditional production system (communal land) (Fernandez-Gimenez, 2000). The (TGLP) policy therefore promotes change of grazing lands from common pool resource to privately owned ranches.

This study was therefore aimed at understanding savanna ecosystem in relation to rangeland management systems and different environmental conditions in semi-arid Botswana. The thesis focussed on:

a. Soil dynamics in relation to rangeland management systems and environmental variability (Chapter 2).
b. Compare herbaceous vegetation dynamics between rangeland management systems (Chapter 3).
c. Compare bush encroachment under communal grazing and ranching systems (Chapter 4).
d. Pastoralists' ecological knowledge on the savanna ecosystem dynamics in relation to rangeland management systems (Chapter 5).

The current chapter thus provides a synopsis of all questions addressed in Chapter 2-5 to achieve the overall aim (above) of this thesis, and then go on to discuss the implications of this evidence to rangeland management and policy in Botswana and other developing countries in southern Africa.

6.2 Soil dynamics in relation to management systems and environmental conditions

Soil samples were collected along transects located in both communal and ranching systems. These were analyzed for soil texture, bulk density, pH and soil organic carbon (chapter 2.2).

6.2.1 Soil characteristics in relation to management systems

The results showed that soil properties in savanna ecosystems do not respond consistently to any particular management systems (Chapter 2.3). The soil texture, bulk density and pH did not differ between management systems, though occasionally significant differences were observed. Based on our study results (Chapter 2) it is evident that management systems have no or similar effect on soil physical properties despite differences in grazing intensities between and within the study sites. These results are partially supported by the findings of another comparative study (Tefera *et al.*, 2007), which also found no significant management effect on soil texture, bulk density and pH. Moussa *et al.,* (2008) also found no significant differences in soil chemical properties (e.g. pH) between grazed communal land and ungrazed land in South Africa and that could explain lack of significant differences between grazing management systems in our study. Hence, it could be argued that differences in grazing intensities under communal grazing and ranching systems have no significant effects on soil properties especially in sandy regions. Other studies have indicated that grazing management systems affect soil bulk density (Liebig *et al.*, 2006), and acidicity (Geissen *et al.*, 2009) which apparently contradicts our observation. The differences between current results and others could be attributed to differences in environmental conditions such as rainfall or management practices.

6.2.2 Soil organic carbon and its determinants

The amount of soil organic carbon (SOC) was found to be generally low in our study sites, particularly in sandveld regions consisting of Xanagas and Matlolakgang (Chapter 2). SOC was not strongly affected by management systems except at Goodhope. Soil organic carbon is dependent on the addition of carbon in plant and manure residues (Rees *et al.*, 2005), and therefore our results largely indicated that both communal grazing and ranching management have similar effects on the amount of carbon added to soil. This observation is consistent with another study (Tefera *et al.*, 2007), but contradicts another (Smet and Ward, 2006), which observed significant differences in SOC close to watering points. The significant differences observed by Smet and Ward (2006) could be due to manure deposits around watering points, which would be dependent on the amount of time the animals spent around water point and their numbers.

The communal rangelands at Goodhope had lower soil organic carbon than ranching lands (Chapter 2). This could indicate that the ranch at Goodhope was not overgrazed since it is mainly used to keep bulls, and these are moved to different ranches during the breeding period. Because of this, there is high standing herbaceous biomass, as observed in Chapter 3, and consequently, there is a lot organic matter being deposited back into the soil as plant litter. Goodhope communal land may also be at an early stage of degradation because the degraded savanna ecosystems exposed to high grazing intensity generally have lower soil organic carbon (Rutherford and Powrie, 2010). This could then explain our observation of soil organic carbon differences between management types at Goodhope, especially since communal grazing lands had higher grazing pressure than ranch at this site (see Chapter 1).

SOC concentration in a specific ecosystem is dependent on site-specific conditions (e.g. soil texture, & rainfall) and other controlling factors such as land use and period of use (Girmay *et al.*, 2008) and these could explain variations between the effects of grazing management systems on soil carbon observed in this study and others elsewhere. In fact, there were significant site effects on the variability of soil properties (Chapter 2) which further confirmed that local environmental conditions like rainfall variability had a strong influence on soil heterogeneity observed across the savanna ecosystems of Botswana.

The results of our study showed that rainfall had a significant effect on soil organic carbon, which is consistent with results from other studies (Ringrose *et al.*, 1998). Rainfall promotes plant growth, which leads to increased organic input to the soil from litter and contributes towards increased organic carbon. This was confirmed by the results of multifactor analysis, which showed a positive relationship between herbaceous biomass and soil organic carbon across all sites. However, the main predictor of soil carbon was soil clay content, which was shown to have a strong linear relationship with soil organic carbon (Chapter 2, Figure 2.2). It is acknowledged that clay content and rainfall are also moderately correlated, and therefore, could lead to overriding of the effects of other variables in analyses even when these are important (Quinn and Keough, 2002). However, clay particles protect and stabilize soil carbons (Walker and Desanker, 2004) and therefore lead to an increase in soil organic carbon which explains our observations. In addition, clay particles improve soil water holding capacity, which supports high vegetation growth as observed at Goodhope. The litter from improved vegetation growth would also accumulate if not overgrazed and add organic matter to the soil. The positive relationship between herbaceous biomass and soil organic carbon is also clearly demonstrated by our data, as 46 % variation in herbaceous biomass was attributed to soil carbon (Chapter 3.3).

6.3 Herbaceous vegetation dynamics in relation to management and environmental factors

The herbaceous vegetation composition was assessed using the wheel point method along 300 m transects located in both communal and ranching lands at each site (Chapter 3).

6.3.1 Herbaceous compositions in relation to management systems

The herbaceous vegetation compositions were highly heterogeneous spatially, both at local scale and between sites. The dominant herbaceous species differed between sites and even between management systems at each site (Chapter 3). The variation in herbaceous composition between sites could be explained by differences in soil texture, soil fertility (see Chapter 2) and rainfall (Chapter 1) as suggested by other authors (Carr *et al.*, 2009). Our rangeland assessment showed that Matlolakgang and Xanagas rangelands had relatively high bare ground and presence of unpalatable forbs associated with heavy grazing (Makhabu and Marotsi, 2011; Tessema *et al.*, 2011) compared to Goodhope rangelands, which suggests that these rangelands are experiencing ecological stress (Whitford *et al.*, 1998). In addition, the Matlolakgang and Xanagas rangelands were also dominated by unpalatable herbaceous plants, and subsequently the condition of both rangelands was classified as poor based on a range condition guide (Mosley *et al.*, 1986). The Goodhope communal and ranching rangelands were in fair condition, characterized by high grass cover, limited bare ground and abundance of palatable herbaceous plants. Goodhope rangelands also had a balanced composition of decreaser, increaser I and II species, which indicated that they were not overgrazed (Trollope *et al.*, 1989) like some other sites, particularly Matlolakgang.

The herbaceous composition indicated that rangeland conditions were mostly not statistically different between management system at each site (Chapter 3) which could imply that both communal and ranching lands were similarly affected despite the differences in grazing intensities (Chapter 1). Tefera *et al.,* (2007) observed higher abundance of palatable herbaceous plant in ranches compared with communal grazing lands in Ethiopia, which is consistent with our observation at Xanagas. However, our observations at the other sites showed no significant differences in palatable herbaceous plant cover between communal and ranching lands, which are supported by other studies in the African savannas (Parsons *et al.*, 1997; Tefera *et al.*, 2008b). Goodhope communal land had high grazing intensity, but the rangeland was still in good condition. This could suggest that rangeland vegetation does not always respond in a linear way to grazing intensity as suggested by other authors (Sasaki *et al.*, 2010), partly because local environmental conditions regulate plants' ability to cope with defoliation. High rainfall (Chapter 1) and soil fertility (Chapter 2) observed at Goodhope could explain plants' ability to cope with high defoliation.

The herbaceous vegetation composition in savanna ecosystems was generally undergoing changes. This is suggested by the high cover of Increaser II herbaceous plants recorded, particularly at Xanagas and Matlolakgang rangelands. These rangelands were declining in rangeland condition as indicated by decrease in grass cover, biomass and increase of forbs cover as was clearly observed in Matlolakgang rangelands and further illustrated by PCA in Chapter 3 (Figure 3.5). This observation is consistent with the literature (Oba *et al.*, 2000).

6.3.2 Herbaceous biomass in relation to management systems

The results of the current study revealed that herbaceous biomass was highly variable at local spatial scale and between Goodhope and the other two sites (chapter 3). This observed variability in herbaceous biomass is recognised as an attribute of arid and semi-arid ecosystems (Mashiri *et al.*, 2008). Herbaceous biomass variability between the study sites reflected both soil fertility (Chapter 2), and rainfall differences as shown in Chapter 1. The Goodhope rangelands, for example, were characterized by high soil fertility (indicated by high soil organic carbon) and high rainfall, which was consequently reflected in high recorded levels of herbaceous biomass. In fact, our results showed that 46 % of the variability of herbaceous plant biomass was explained by soil organic carbon (Chapter 3). Bush encroachment was shown to be another local factor that could have contributed to herbaceous biomass variation (Chapter 4). The results of the current study (Chapter 4 & 5) and others (Hagos and Smit, 2005) have shown that bush encroachment suppresses herbaceous plant productivity. As a result, the highly encroached Matlolakgang rangelands had low herbaceous biomass, while Goodhope rangelands, characterized by low woody cover, had high herbaceous biomass.

The herbaceous biomass was also significantly affected by grazing management systems. The ranching lands at Goodhope and Matlolakgang exhibited higher herbaceous biomass than communal grazing lands at these sites, whereas no significant differences were observed between management systems at Xanagas (Chapter 3). Herbaceous biomass differences between grazing management systems at a local scale are due to the higher grazing intensities of the communal grazing lands at Goodhope and Matlolakgang compared with their respective ranches (Chapter 1) reflecting greater consumption of herbaceous biomass in the communal lands. The opposite results were observed at Xanagas, which further confirms that herbage yield is highly responsive to grazing pressure as also suggested in other studies (Sternberg *et al.*, 2000; Mphinyane *et al.*, 2008).

The surface plant litter was also significantly lower in communal lands than ranches at all sites, which further confirmed higher consumption of herbaceous plants such that there is less biomass accumulating and degenerating into litter. Given that herbaceous biomass was collected only during 2010 for this study, it is possible that the differences observed could be temporal and dependant on the timing of grazing. It would be interesting to observe and measure repeatedly over longer periods especially since biomass was shown to be significantly influenced by rainfall variability and soil fertility (Chapter 3).

6.3.3 Herbaceous diversity in relation to management systems

The herbaceous species diversity was highly variable, both temporally and spatially, but did not differ between communal and ranching lands at each site. The Matlolakgang rangelands had the highest herbaceous diversity of the three sites, partly due to establishment of forbs and annual grasses. Annual herbaceous vegetation is highly abundant in heavily grazed areas (Skarpe, 2000), and subsequently increases species richness, as confirmed by our observation at Matlolakgang and Xanagas rangelands (Chapter 3). Other authors (Cheng *et al.*, 2011) have also reported a pronounced increase in species richness due to heavy grazing, while others (Tessema *et al.*, 2011) have reported higher species diversity in lightly grazed sites compared to heavily grazed sites, which contradicts our observation. The abundance of annuals at Xanagas and Matlolakgang in this study also explains some of the temporal variation in herbaceous diversity because annual cover is likely to vary depending on annual rainfall variability (Sternberg *et al.*, 2000; Hassler *et al.*, 2010). Our results actually confirmed that herbaceous diversity was significantly related to spatial rainfall variability as suggested by other authors (Adler and Levine, 2007; Cheng *et al.*, 2011). Continuous grazing, which is practiced in communal lands, is also considered to promote high species diversity (Sternberg *et al.*, 2000), but this was not reflected in our study as no statistically significant disparities were observed between management systems.

6.4 Bush encroachment in relation to management systems and environmental factors

The woody plant cover was assessed in 10 x 10 m quadrats located at 100, 200 and 300 m along transects situated in both communal and ranching lands (Chapter 4).

6.4.1 Bush encroachment in relation to rangeland management

The ecological data showed that patches of woody vegetation differed across the savannas particularly between sites (chapter 4). This observation is consistent with patch dynamics theory (Smith and Smith, 2001; Meyer *et al.*, 2007). The results (Chapter 4.3) indicated that Godhope rangelands were least encroached by woody vegetation, while other rangelands, particularly Matlolakgang, were highly encroached by woody vegetation. The rangelands were encroached by different plants such as *Acacia mellifera, Dichrostachys cineria, Grewia flava* and *Terminalia* *sericea*, which are generally classified as encroacher plants by other ecologists (van Vegten, 1984).

Bush encroachment was not strongly associated with any particular management system in the present study (Chapter 4), despite differences in grazing intensities between communal and ranch lands. This lack of association between bush encroachment and grazing management systems is also reported by other authors (Oba *et al.*, 2000) and contradicts the perceptions of most pastoralists (chapter 5) and also the assumptions of Tribal Grazing Land Policy (TGLP) that communal gazing lands are more encroached on by bushes than ranching grazing lands. Other authors (Wigley *et al.*, 2009) reported that bush thickets expanded more slowly in communal grazing lands than ranches, which also disputes the assumptions of TGLP.

6.4.2 Factors related to bush encroachment

Our analysis showed that abiotic factors were the primary regulators of the structure of the woody plant community, which is consistent with the findings of other authors (Alados *et al.in press*). The woody cover and diversity increased with decreasing soil clay content (Chapter 4), but not in a linear way. Thus, bush encroachment is most likely to occur in sandy soils, as was observed in this study at Matlolakgang and Xanagas that are both located in Kalahari sandveld. Analysis of the determinants of woody cover in African savannas also reported that sandy soils promote increased woody cover (Sankaran *et al.*, 2005), which further supports our results.

6.5 Pastoralists’ assessment of rangeland conditions

The dominant vegetation compositions at different sites were also characterized based on pastoralists’ ecological knowledge (Chapter 5). A structured questionnaire was used to collect information on vegetation dynamics in grazing lands. The ecological data (chapter 3 & 4) provided current rangeland conditions and the pastoralists’ ecological knowledge complemented it by providing information on observed changes in vegetation over time.

6.5.1 Grass composition as characterized by pastoralists

The pastoral communities in the three sites identified different grasses as dominant in their respective grazing lands (Chapter 5). The differences in dominant grasses between grazing lands indicate the heterogeneity of savanna ecosystems due to soil (Chapter 2) and rainfall variability (Chapter 1) and grazing pressure (Chapter 1). Pastoralists’ descriptions of dominant grasses were largely supported by our ecological data (Chapter 3) but we realized that our definitions of dominance could be different because some of the grasses classified as dominant by pastoralists were recorded as either common or present in our ecological data (Chapter 3). The pastoralists indicated that grass compositions have changed over time, and most of the palatable grasses were being replaced by unpalatable species, which is consistent with the high cover of Increaser II species (Chapter 3). Grass composition changes were mainly associated with communal grazing lands, which could have been influenced by perception of policymakers. However, the majority of pastoralists identified rainfall as the main driver of change in grass composition, which was also identified as one of key drivers of vegetation dynamics between sites in our ecological data analysis in this study. Livestock overgrazing and fire were other causes of change in grass compositions identified by pastoralists.

6.5.2 Bush encroachment as perceived by pastoralists

The pastoralists confirmed our ecological data (Chapter 4) that bush encroachment was an environmental problem in some grazing lands. However, some pastoralists at Goodhope stated that their grazing rangelands were encroached, which contradicts our ecological data. Pastoralists' perceptions are based on different patches of landscape and therefore some patches may be encroached and those pastoralists settled closer to those encroached patches would have a different perception than others settled in the open savanna observed at Goodhope. This could also highlight the significance of using pastoral ecological knowledge because it allows a researcher to assess a large area of grazed land more easily than is possible by conventional methods.

According to most pastoralists, the increase in woody cover resulted in decline in herbaceous productivity, and their perception was supported by our ecological data (Chapter 3) and other studies (Hagos and Smit, 2005). Matlolakgang rangelands were highly encroached by bushes, and herbaceous biomass was in turn low. Low standing biomass implies that there is shortage of forage for grazing animals (Dalle *et al.*, 2006). Animal productivity is adversely affected and this in turn threaten pastoralists' potential income (Angassa and Oba, 2008a). However, pastoralists also indicated that woody vegetation is a valuable browsing resource, especially during dry period (Chapter 5) which is supported by other ecological studies (Moleele, 1998). Woody plant species such as *Boscia albitrunca* were considered suitable for browsing by most pastoralists, but the literature suggests that browsing of some woody plants, like *Grewia flava,* could also enhance bush encroachment through seed dispersal (Tews *et al.*, 2004) and regeneration (Noumi *et al.*, 2010).

6.5 Results implication to management and policy

The current grazing policy (TGLP) was introduced to address rangeland degradation and is based on the assumptions that ranching would promote sustainable land use and conservation of rangeland resources in Botswana. Our results showed that this assumption is wrong. Rangeland degradation is occurring on both communal and ranching lands, particularly in sandveld regions (Chapter 3 & 4). Actually, Vanderpost *et al.,* (2011) had reported that rangeland degradation is highest in ranches compared with other land use (including communal grazing) in Botswana. This had led to others (Rohde *et al.*, 2006) arguing that the current grazing policy had failed to address the land degradation problem and had instead exacerbated it. In addition, the communal grazing land which supports the large population of livestock continues to shrink in size (Boone, 2005) as more land is demarcated into ranches.

Subsequently, grazing pressure is intensified in communal grazing land (Bennett *et al.*, 2010) especially since owners of private ranches continue to use communal grazing land in addition to their own land (Tsimako, 1991). Considering the limited land now available for grazing by pastoral communities (Bennett *et al.*, 2010) the existing policy should be reviewed and dual grazing by ranchers should be eliminated. This would address the current inequity of land distribution, which could ultimately threaten the sustainability of the entire savanna ecosystem (Eriksen and Watson, 2009a).

The results from this study showed that soil carbon, and bush encroachment (Chapter 2 & 4) which reflects long-term ecological change (Walker and Desanker, 2004; Vincke *et al.*, 2010), did not differ strongly between communal and ranching grazing lands in the three sites in Botswana despite differences in grazing intensities. This suggests that other abiotic factors (rainfall and soil properties) regulate ecosystem response to grazing and therefore grazing policy should consider these. Our results revealed that soil properties (especially clay content) and rainfall have strong influence on soil fertility and woody cover, which are key indicators of degradation in savanna ecosystem in Botswana. Meanwhile, the current policy emphasises stocking rates only and therefore needs to be reviewed to account for environmental variability and the strong influence that this has on savanna ecosystem dynamics (Chapter 3 & 4). Grazing management was however shown to have a strong influence on herbaceous productivity. Ranching land had higher standing biomass than communal grazing land due to lower stocking rates (Chapter 3) and this could lead to increased animal gain. It is recognized however that standing biomass is correlated with rainfall (Sternberg *et al.*, 2000) and therefore likely to vary annually. Conversely, it has been reported that productivity and grazing management had not improved in most individually owned ranches in Botswana, partly due to absentee ranch management (Rohde *et al.*, 2006). The government ranches used in this study had all of the attributes (e.g. correct stocking rates & rotational grazing) associated with proper ranching management practices, but there was no significant and consistent differences observed in comparison to communal grazing lands.

With this in mind, future grazing policy should take into account pastoralists' ecological knowledge and management practices (Chapter 5). Most pastoralists are knowledgeable of savanna ecosystem dynamics as illustrated in Chapter 5. Participation of local communities and land users is essential for win-win outcomes in environmental-development issues (Reynolds *et al.*, 2007). Pastoral communities' understanding of rangeland degradation improves their willingness to participate in addressing the problem (Kessler and Stroosnijder, 2010). Our study had found that pastoralists are aware of the range degradation that was indicated by an increase of unpalatable grasses and bush encroachment in their grazing lands. This shared understanding of rangeland degradation by land users and policy makers could contribute towards protecting the environment in dryland agro-ecosystems and alleviating poverty (Thomas, 2008) as it promotes positive attitudes towards natural resource conservation (Mbaiwa and Stronza, 2011). The grazing policy should also reflect the interests of all of the different land users (Slocombe, 1998) and allow some flexibility in management to adapt to environment variability including the licensed use of fire as a management tool to bush encroachment.

Grazing land policy should consider management concepts such as Community Based Natural Resource Management (CBNRM) (Mbaiwa and Stronza, 2011) to facilitate active participation of land users in the management of communal grazing lands. The pastoral communities would then be involved in the decision-making processes of grazing land use, including monitoring and addressing rangeland degradation. The seven pillars of ecosystem management including social values, boundaries, health, stability, diversity, sustainability and scientific knowledge (Lackey, 1998) could also be relevant to sustainable management of rangelands, though establishment of boundaries is likely to be challenging. Researchers meanwhile could contribute towards sustainable rangeland management by carrying out directed ecological research to guide management especially that ecological knowledge is not widely shared among pastoral communities. Policy should be based on all relevant ecological knowledge, consisting of both scientific and pastoral ecological knowledge. Thus, the results of this study and others could contribute significantly towards guiding grazing policy of Botswana.

6.6 Conclusion and way forward

The results of this current study clearly showed that communal grazing and ranching do not affect the savanna ecosystem differently, which contradicts the assumptions of TGLP policy. The indicators of rangeland degradation such as low soil organic carbon, high cover of increaser II herbaceous species and bush encroachment were more evident in sandveld than hardveld regions, but did not differ strongly between grazing management systems. The shift in vegetation composition in savanna ecosystem (e.g. bush encroachment) of Botswana was largely attributable to environmental variability, particularly soil characteristics. The soil clay content strongly associated with soil organic carbon, which in turn influenced herbaceous biomass. In addition, bush encroachment increased with a decrease in soil clay fraction, but not linearly. The pastoral communities also acknowledged that changes in grass compositions and bush encroachment were occurring across some savannas in semi-arid Botswana. Pastoralists also recognised that inter-annual rainfall variability had a strong influence on herbaceous productivity.

This study substantiates the need to address rangeland degradation because it threatens the livelihood of pastoralists. The pastoralists and researchers need to find ways to share and apply their existing knowledge in order to rehabilitate degraded rangelands and control bush encroachment. Long-term social and scientific studies are necessary to guide the future management of savanna ecosystems, and maintain a threshold of bush encroachment that is beneficial to growth of herbaceous vegetation and livestock productivity in savanna ecosystems under different environment conditions. Quantitative and standard methods of rangeland assessment methods also need to be established to allow easy comparison between studies especially that current methods are not sufficient to provide reliable assessment that reflects all aspects (e.g. social and ecological) of land degradation (Foster, 2006). While there is an agreement that loss of rangeland biological or economical productivity indicates degradation, there is no quantitative measure of how much loss is associated with degradation. This is particularly a challenge in semi-arid environments where the ecosystems' productivity tends to fluctuate in response to rainfall variability, such that it becomes difficult to distinguish between temporary and permanent decline in ecosystem productivity.

Thus, targeted experimental research should be carried out to clearly separate the effects of environmental factors (e.g. soil clay, rainfall) on herbaceous biomass and woody cover observed in this study. Experimental studies would be more precise than the observational approach adopted in this study as the researcher would be able to manipulate some variables while maintaining others constant. Future research studies should also investigate the effects of soil mineral content such as phosphorus, and calcium on savannas' vegetation variability. This combined knowledge must be able influence policy making in relation to rangeland management in Botswana.

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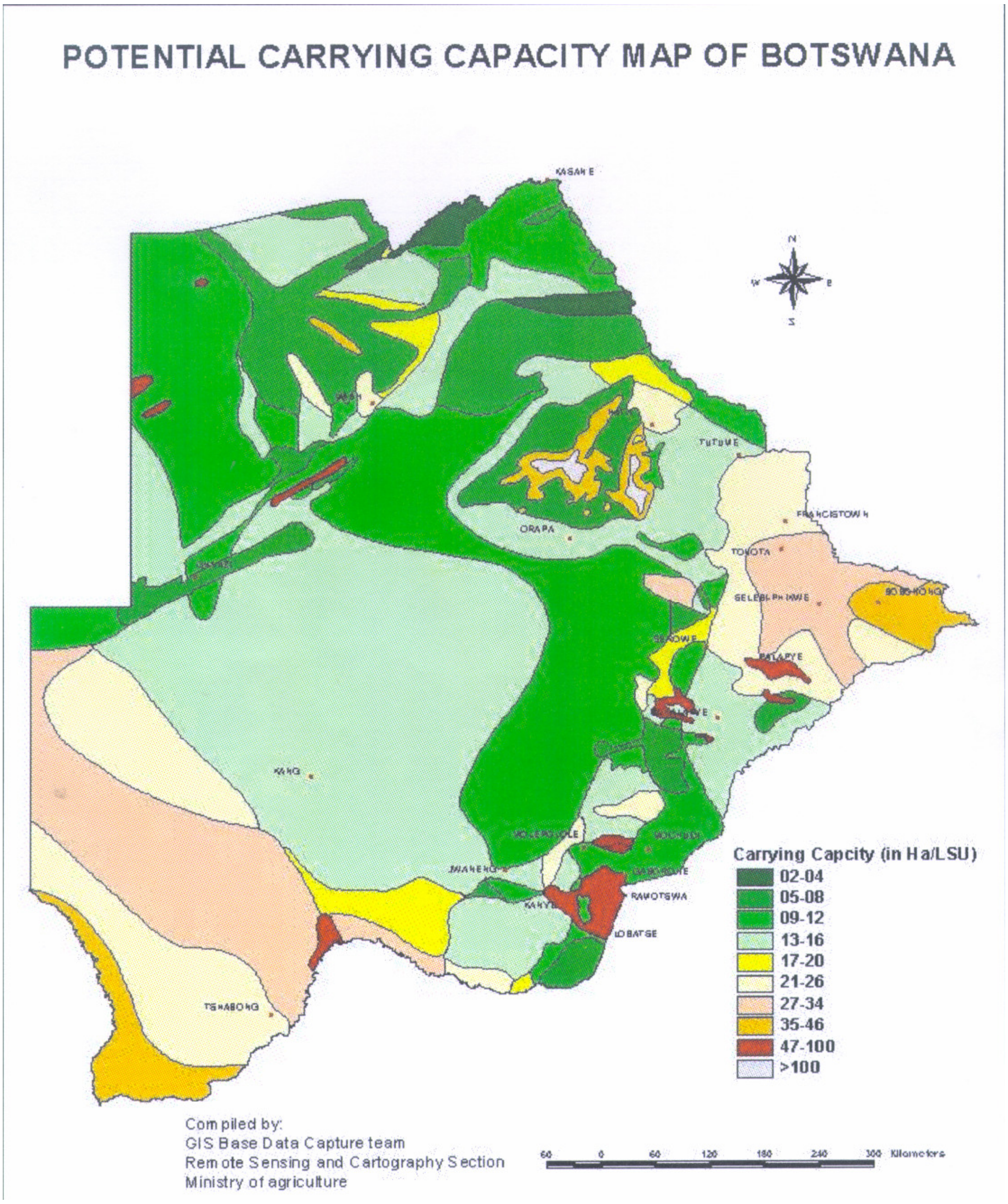
Appendix 1

Map of potential carrying capacity of livestock in Botswana



Appendix 2

Correlation coefficients (r) matrix showing interrelationships between soil properties and environmental indicators

Variables	1	2	3	4	5	6	7	8	9	10
1.Rainfall										
2.Herbaceous SR	-0.31*									
3.HBS	0.46 *	-0.30								
4.Litter	0.31	-0.16	0.60**							
5.SOC	0.66***	-0.29*	0.67***	0.43*						
6.pH	-0.12	-0.19	0.27	-0.00	0.54*					
7.Clay	0.61**	-0.40	0.56*	0.25	0.90***	0.59**				
8. Silt	0.77***	-0.50*	0.65**	0.42*	0.85***	0.27	0.75***			
9. Sand	-0.74***	0.48*	-0.65**	-0.36	-0.93***	-0.48*	0.93 ***	-0.94 ***		
10. Woody SR	-0.27	0.48*	-0.24	0.12	-0.41	-0.45*	-0.53*	-0.40	0.47*	
11. Woody cover	-0.27	0.49*	-0.40	0.04	-0.47	-0.40	-0.64**	-0.42*	0.56*	0.66**

* < 0.05;** <0.005; *** < 0.001, SR= species richness, HBS= herbaceous biomass

Appendix 3

Mean herbaceous composition and cover (±SE) under communal and ranch management in the three study sites during 2009 and 2010

Species	Life-form	Goodhope		Matlolakgang		Xanagas	
		Communal	Ranch	Communal	Ranch	Communal	Ranch
Aconthosicyos nauctinian	G					2.38 ± 0.71	3.04 ± 0.54
Acrotom iflota	F					0.08 ± 0.00	0.08 ± 0.00
Anthephora pubescens	G		7.92 ± 1.00		0.13 ± 0.13	0.25 ± 0.00	
Aptosium elongatum	F			0.96 ± 0.04	0.50 ± 0.25		
Aristida congesta	G	9.28 ± 1.83	1.00 ± 0.67	9.83 ± 1.83	10.63 ± 2.46	9.33 ± 1.17	0.54 ± 0.13
Aristida granitica	G	2.67 ± 0.56	0.25 ± 0.17	4.29 ± 1.13	4.96 ± 1.38	4.08 ± 0.50	0.67 ± 0.08
Aristida meridionalis	G		0.17 ± 0.08		6.38 ± 0.88		
Aristida stipitata	G	1.06 ± 1.06	0.83 ± 0.75				
Bare ground		0.94 ± 0.06	0.33 ± 0.00	10.54 ± 2.71	2.58 ± 1.00	3.42 ± 2.83	6.08 ± 0.25
Blepharis intergrifolis	F			0.25 ± 0.00	0.08 ± 0.00	0.25 ± 0.25	0.50 ± 0.00
Brachiaria nigropedata	G	1.83 ± 0.50	5.63 ± 0.13	0.46 ± 0.04	1.88 ± 0.38	0.17 ± 0.17	0.17 ± 0.00
Cassia biensis	G					0.21 ± 0.21	
Cassia italica	F	0.56 ± 0.00			0.25 ± 0.00	0.75 ± 0.25	0.21 ± 0.04
Crinum polyphyllum	F				0.04 ± 0.04		
Cymbogon plurinodis	G	0.72 ± 0.17	4.54 ± 0.96		0.04 ± 0.04		
Cynodon dactylon	G	14.28 ± 0.61		0.08 ± 0.00			
Dicoma schinzii	F			0.08 ± 0.00	0.08 ± 0.00	1.25 ± 0.00	0.33 ± 0.00
Digitaria eriantha	G	23.94 ± 3.72	18.17 ± 0.25	8.33 ± 2.58	18.33 ± 3.17	0.13 ± 0.04	1.88 ± 1.13

Herbaceous composition and cover continues

Species	Life-form	Goodhope	Matlolakgang	Xanagas			
		Communal	Ranch	Communal	Ranch	Communal	Ranch
Elephantorrhiza elephantina	F	0.22 ± 0.00	0.25 ± 0.00	1.08 ± 0.33	0.58 ± 0.00	3.04 ± 2.79	
Elyonurus argenteus	G		1.25 ± 1.17				
Enneapogon scoparius	G	0.44 ± 0.00	11.25 ± 1.92				
Eragrostis denudata	G			0.92 ± 0.42	0.04 ± 0.04	0.21 ± 0.04	0.33 ± 0.08
Eragrostis lehmanniana	G	0.89 ± 0.67	0.13 ± 0.13	14.92 ± 0.83	6.042 ± 0.38	13.71 ± 0.21	1.38 ± 0.71
Eragrostis pallens	G		0.92 ± 0.25		0.08 ± 0.08		0.71 ± 0.21
Eragrostis rigidior	G	2.89 ± 0.56	1.17 ± 0.67	0.92 ± 0.17		3.38 ± 0.29	6.08 ± 0.75
Eragrostis superba	G		0.17 ± 0.17				
Eragrostis trichophora	G	5.00 ± 0.00	4.75 ± 0.00				
Eustachys paspaloides	G		0.53 ± 0.49				
Fimbristylis hispidula	G		0.17 ± 0.17	1.29 ± 1.29	0.13 ± 0.13	1.17 ± 0.00	2.08 ± 1.17
Fingerhuthia Africana	G	7.00 ± 1.00	2.29 ± 0.63				
Gesikia pharaceodes	F					0.13 ± 0.13	0.04 ± 0.04
Harpagophytum procumbens	F			0.33 ± 0.00			
Heliotropium steudneri	F					0.08 ± 0.08	0.17 ± 0.17
Hermania species	F			0.21 ± 0.13			
Heteropogon contortus	G	8.94 ± 1.28	15.50 ± 3.25	2.67 ± 2.67			
Hyparrhenia filipendula	G		1.25 ± 1.25				

Herbaceous composition and cover continues

Species	Life-form	Goodhope	Matlolakgang	Xanagas			
		Communal	Ranch	Communal	Ranch	Communal	Ranch
Indigofera bainesiis	G			0.04 ± 0.04	0.33 ± 0.17	0.83 ± 0.00	
Indigofera daleoides	F	0.61 ± 0.17		5.96 ± 0.80	6.21 ± 0.29	0.17 ± 0.08	0.08 ± 0.00
Indigofera milelidenea	F	0.06 ± 0.06		3.58 ± 1.08	3.83 ± 2.08	3.08 ± 0.75	0.58 ± 0.33
Ipomoea bolusiana	G					0.17 ± 0.00	0.96 ± 0.13
Litter		0.33 ± 0.00	2.00 ± 0.00	4.17 ± 3.42	5.00 ± 3.17	0.58 ± 0.00	5.04 ± 1.29
Mariscus species	F			2.92 ± 0.00	0.33 ± 0.00	0.75 ± 0.00	0.67 ± 0.00
Megaloprotachne albescens	G			4.88 ± 1.21	1.38 ± 1.29	0.17 ± 0.00	0.67 ± 0.00
Melhania burchelli	F			0.08 ± 0.00			
Melhania forbesii	F			0.13 ± 0.13	0.04 ± 0.04	0.08 ± 0.08	
Melinis repens	G	1.44 ± 0.56	0.50 ± 0.42	0.33 ± 0.33	1.00 ± 0.00	7.67 ± 5.00	2.54 ± 1.21
Otoptera burchellii	F					0.63 ± 0.46	0.58 ± 0.08
Oxygonum alatum	F			3.58 ± 0.83	2.63 ± 1.38	1.54 ± 1.21	0.50 ± 0.42
Panicum coloratum	G	0.50 ± 0.17	0.75 ± 0.00				
Perotis patens	G	0.11 ± 0.00		7.04 ± 0.04	2.50 ± 0.67		
Pogonarthria squarrosa	G	0.56 ± 0.33	0.17 ± 0.17	0.75 ± 0.50	8.50 ± 0.42	0.08 ± 0.00	
Raphionacme burkei	F						0.17 ± 0.00

Herbaceous composition and cover continues

Species	Life-form	Goodhope	Matlolakgang	Xanagas			
		Communal	Ranch	Communal	Ranch	Communal	Ranch
Raguiena sphaerospherma	F			0.42 ± 0.00	1.79 ± 0.21	5.50 ± 0.00	
Rhynchosia totta	F			0.42 ± 0.25	0.08 ± 0.08		
Schimidtia pappophoroides	G	8.56 ± 0.89	10.83 ± 0.33	2.08 ± 0.25	2.54 ± 0.04	0.88 ± 0.21	9.88 ± 0.04
Setaria verticiliata	G		0.17 ± 0.00				
Stipagrostis uniplumis	G	0.67 ± 0.22	2.88 ± 0.04	4.83 ± 1.42	10.42 ± 2.50	25.21 ± 0.38	35.17 ± 7.33
Talium crispatulum	F			0.08 ± 0.00		0.42 ± 0.33	0.38 ± 0.04
Themeda triandra	G		3.25 ± 0.00				
Tragus berteronianus	G						0.17 ± 0.17
Tragus racemosus	G	0.11 ± 0.00				0.83 ± 0.33	
Tricholena monachne	G					0.17 ± 0.00	0.08 ± 0.00
Trivaphis andropogonoide	G						0.08 ± 0.00
Tylosema esculenta	F			0.50 ± 0.17	0.54 ± 0.29	3.50 ± 0.67	4.25 ± 1.17
T. orkanyva	F					0.21 ± 0.04	0.13 ± 0.13
Urochlora trichopus	F	7.44 ± 6.89	6.33 ± 5.75	0.67 ± 0.08	0.17 ± 0.08	8.58 ± 7.42	14.50 ± 3.75
Unknown	F	0.39 ± 0.28	0.33 ± 0.00	1.92 ± 1.75	0.13 ± 0.04	0.25 ± 0.08	0.25 ± 0.00
Urginea sanguinea	F						0.25 ± 0.00
Vernonia poskena	F					0.08 ± 0.08	
Waltheria indica	F			0.54 ± 0.13	0.17 ± 0.17		

G = Grass, F = Forbs, Dominant (>15 %), common (<15 -5 %), less common (<5-1%) and rare (<1%)

Appendix 4

Pastoral farmer's perception on vegetation change

This questionnaire is designed to collect information on pastoral farmers' perception of vegetation change in their grazing area. This information will improve our understanding of traditional ecological knowledge and its application to range management. The surveys were carried out in the local dialect.

1. Farmer's demographic

1.1. Name:	1.2. Age:
1.3. Years of education.	1.4. Years of farming:
1.5. Location:	1.6. Gender:

1.7 Type of farmer: a. full-time farmer _____
b. part-time farmer _____
c. Herder _____

2. Production systems

2.1. Which type of production system do you rear your livestock? (tick right answer)
a. Communal system (cattlepost) ____________
b. Ranching ____________
c. Both ____________

2.2. What is your estimated number of the following animals do you rear in your production systems?

Animal type	Numbers
Cattle	
Goats	
Sheep	

2.3 Which of the production systems do you consider suitable for livestock production?
a. Communal system ______________
b. Ranching ______________
c. Both ______________
d. None ______________

2.4 Which of the production systems do you consider to have negative impact on vegetation?
a. Communal system _____________
b. Ranching system _____________
c. Both _____________
d. None of above _____________

3. Herbaceous vegetation change

3.1 What are dominant grasses in your grazing area?

____
____
____
____

3.2 Please rank all dominant grasses (3.1) in terms of their suitability (1 very suitable, 2 Suitable, 3 Not Suitable)

Name of grass	Very suitable	Suitable	Not suitable

3.3 Has there been any change in composition of grasses in the last 5 years?

a. Yes _____
b. No _____
c. Do not Know _____
(if answer is "No" , go to Q3.6)

3.4 . If yes, please indicate changes in composition of grasses in the last 5 years in table below.

Name of grass	Increasing	Stable (constant)	Decreasing

3.5 What factors do you think cause change in grass composition?

____
____
____

3.6 Rank the following factors in their relevance to change in grass composition

Factor	Rank (1 most important------5 least important)
a. Rainfall	
b. Drought	
c. Fire	
d. Overgrazing	
e. Management	

3.7 Under which production system are changes in grass composition more common?

a. Communal system _______________

b. Ranching system _______________

c. Both ________________

3.8 What is currently being done to minimize change in grass composition?

____

____

____

4. Woody plant vegetation change

4.1 Which woody plant species are dominant in your grazing area?

____

____

____

4.2 Please rank all dominant woody plants in terms of their suitability for browsing (1 very suitable, 2 Suitable, 3 Not Suitable)

Name of woody plants	Very suitable	Suitable	Not suitable

4.3 Is bush encroachment a problem in your grazing area?

a. Yes ____________

b. No ____________

(if answer is "No", go to Q4.5)

4.4 If yes, how can you classify the degree of bush encroachment in your area?

a. High __________

b. Medium __________

c. Low __________

d. None __________

e. Do not Know________

4.5 Under which production system do you think the problem of bush encroachment is more common?

a. Communal system ____________

b. Ranching system ____________

c. Both ____________

d. Do not know ____________

4.6 What are major encroacher woody plant species in your grazing area?

____

____

____

4.7 Please rank encroacher woody plants in terms of their dominance in your grazing area.

Name of woody plants	Rank (1 Most dominant- 5 least dominant)

4.8 What do consider as causes of bush encroachment?

____

____

____

4.9 Rank the following factors in terms of importance towards facilitating bush encroachment (1 most important-----4 least important)

a. Overgrazing __________
b. Rainfall/drought __________
c. Change in use of woody plant species __________
d. Lack of browsing/grazing __________

4.10 Use the table below to indicate browsing suitability and trends of bush cover in the last 5 years (1 very suitable, 2 Suitable, 3 Not Suitable)

Name of woody plant	Suitability (1-3)	Trend (1-increase, 2-constant, 3-decrease)

4.11 What is the impact of bush encroachment on herbaceous vegetation?

____

____

____

____

____

4.12 What is currently being done to address problem of bush encroachment?

____

____

____

4.13Is there any benefit associated with increase in bushy plants?

a. Yes______
b. No ______
(if answer is NO, go to Q5.0)

4.14List all benefits associated with individual plant species

Name of woody plant species	Benefits

5.0 Fire history

5.10 Has there been some fire in your grazing area?

a. Yes________
b. No_________

5.11 If yes, how frequent have fires burned in and around the unit?

a. Every year ______
b. 2 years ______
c. 3 years ______
d. 4 years and more ______

5.12 Please list the years that you experienced fires in your grazing area?

____

____

____

THANK YOU FOR YOUR SUPPORT