Impact of livestock grazing intensity on the plant diversity of species-rich montane grassland in the northern Drakensberg, South Africa

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Declaration

I declare that this Dissertation is my own work. It is being submitted for the Degree of Master of Science at the University of the Witwatersrand, Johannesburg. It has not been submitted by me before for any other degree, diploma or examination at any other University or tertiary institution.

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Date

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ABSTRACT

Livestock grazing intensity is expected to impact on the plant species composition and plant diversity of Drakensberg grasslands. These montane grasslands are important for providing goods and services for the local and the national population, in part through supporting livestock on communal rangelands. Montane communal rangelands are generally heavily stocked, although grazing pressure is expected to be concentrated around kraals and to show a decreasing gradient with distance. The aim of this study was to determine the impact of livestock grazing on plant species composition and diversity of montane grassland in the northern Drakensberg (Thabo Mofutsanyane district). A grazing gradient from a kraal site to beyond the average foraging distance of cattle was examined. A second approach examined fence-line contrasts between communal rangeland and the protected areas of Golden Gate Highlands National Park (GGHNP) and Royal Natal National Park (RNNP). Nine transects were sampled (using 90 5 x 5 m plots) in order to define a gradient, ensuring relative uniformity of environment by sampling only spurs and crests. The fence-line contrast between GGHNP and QwaQwa was sampled using 40 adjacent pairs of plots (5 x 5 m), that of RNNP and communal rangeland using 20 adjacent pairs. The percentage cover of each species on each plot was estimated using the Domin scale. Soil from each plot was analysed for levels of P, K, Ca, N, Mg, Zn, Mn, Cu, exchangeable acidity, acidity saturation, total cations, pH (KCl), organic carbon and percent clay.

The presence of a grazing gradient defined by distance was supported by a decrease in the amount of cattle dung with increasing distance from a kraal, described by a non-linear relationship. Distance was not confounded with altitude, solar radiation, or slope, nor with variation in soil physico-chemical properties as described by the first three axes of a Principal Component Analysis (PCA). An effect of grazing intensity was supported by a relationship between increasing distance and increasing species richness, an increasing abundance of graminoids, indigenous plants, and perennial plants, a decreasing abundance of shrubs, dwarf shrubs, annual herbaceous plants, and alien plants (especially *Richardia brasiliensis* and *Hypochaeris radicata*), and a change in herbaceous composition. However, most compositional variation was unexplained.

The studies of fence-line contrasts provided some further support for an effect of grazing intensity. For the contrast between GGHNP and the QwaQwa communal area, species composition was different and GGHNP supported more endemic species. RNNP compared with the adjoining communal rangeland showed greater species richness and plant diversity, a greater abundance of grasses, dicotyledons and indigenous plants, and a lesser abundance of dwarf shrubs. In conclusion, a gradient study and fence-line contrasts both provided support for an effect of livestock grazing on the composition and diversity of montane grassland in the northern Drakensberg. However, despite a long history of communal livestock grazing a total of more than 320 indigenous plant species indicates these grasslands make some contribution to biodiversity conservation, but this contribution is compromised in the vicinity of kraals. Rangeland under reduced grazing intensity may therefore offer a means of maintaining the plant diversity of communal grassland grazing systems. The state of the grassland on the RNNP side of that fence-line contrast perhaps presents a benchmark grassland state that could be achieved in this communal grazing region if conservative livestock numbers could be maintained.

KEYWORDS: Fence-line contrasts, grazing intensity, kraals, piospheres, plant diversity, and plant groups

LIST OF ABBREVIATIONS AND ACRONYMS

ANOVA	Analysis of Variance
CCA	Canonical Correspondence Analysis
DAFF	Department of Agriculture, Forestry and Fisheries
DBSA	Development Bank of Southern Africa
DEM	Digital Elevation Model
DEAT	Department of Environmental Affairs and Tourism
DRDLR	Department of Rural Development and Land Reform
ECP	Eastern Cape Province
Esri	Environmental Systems Research Institute
FS	Free State Province
FAO	Food and Agricultural Organization
EKZN	Ezemvelo KwaZulu-Natal wildlife
FSDP	Free State Development Plan
GGHNP	Golden Gate Highlands National Park
GIS	Geographic Information System
GPS	Global Positioning Systems
HDC	Highlands Development Corporation
Hdf	Hierarchical Data Format
KZN	KwaZulu-Natal
IUCN	International Union for Conservation of Nature
LM	Linear Models
MDTP	Maloti-Drakensberg Transfrontier Park
MODIS	Moderate Resolution Imaging Spectroradiometer
OFS	Orange Free State
PCA	Principal Component Analysis
QGIS	Quantum Geographic Information Systems
QNP	QwaQwa National Park

RNNP	Royal Natal National Park
SAGNDP	South African Government's National Development Plan
SANBI	South African National Botanical Institute
SAIRR	South African Institute of Race Relations
SANParks	South African National Parks
SAWS	South African Weather Service
SA	South Africa
TBVC	Transkei, Bophuthatswana, Venda, and Ciskei
WCCA	Witsieshoek Community Conservation Area

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Chapter 1: GENERAL INTRODUCTION

In general, grazing lands globally make-up the largest land uses (Liebig et al. 2006). The grazing lands are estimated to cover close to 25% of the global terrestrial land (Asner et al. 2004). This makes rangelands an important source both for the livelihoods of rural communal communities and in maintaining the world's biodiversity (O' Connor 2005; Eriksen and Watson 2009; Muhumuza and Byarugaba 2009). Montane grasslands rangelands have been utilized for centuries for extensive livestock grazing, with both commercial and communal grazing rangelands being the dominant ones (Rohde et al. 2006; Masike and Urich 2008; Terefa 2011). Most southern African communal rangelands are found within tribal land and the land belongs to all pastoral farmers (Rohde et al. 2006; Masike and Urich 2008). They generally keep mixed livestock with cattle and goats making-up the largest numbers (Wigley et al. 2010). These communal rangelands are being degraded by the impact of livestock grazing, which potentially threatens their sustainability and the livelihoods of the communal communities who rely on livestock rearing (Vetter 2005; Darkoh 2009).

1.1 Rationale for the study

Livestock grazing is the main land use of natural grassland in South Africa (SA) in both commercial and communal rangelands. Communal rangelands are areas that are not privately owned but belong to the whole community, and half the livestock population of the country is found in these areas (Scogings et al. 1999). Communal rangeland in SA is usually subjected to continuous grazing at stocking rates higher than commercial rangeland (Tapson 1993; National Livestock Statistics 2006, 2017), whereas commercial rangelands mainly use rotational grazing systems involving paddocks (O'Regain and Turner 1992).

The impact of grazing on grassland depends on grazing pressure that is determined by stocking rate (Tainton 1999). An increase in stocking rate usually causes changes in grassland species composition including an increased abundance of species that are resistant to or tolerant of grazing, and a decreased abundance of species that are vulnerable which are usually the grass species most desirable for animal production (Latting et al. 1972; Crawley 1983).

A current problem of some montane grasslands in South Africa is that they support a large population of livestock under communal grazing management (Tainton 1999), which renders these grasslands susceptible to overgrazing and transformation. A general view in South Africa is that many communal grasslands are poorly managed and overgrazed (Shackleton

1993). Overgrazing is believed to be a direct contributor to reduced plant species diversity (O'Connor et al. 2010). Perhaps of even more importance is the perception that the productivity and resilience of these grasslands are being irreversibly damaged and that eventually these grasslands will be but a shadow of their former state (Shackleton 1993).

Montane grasslands in SA are important centres of plant diversity and support many endemic species (Cowling and Hilton-Taylor 1997; Carbutt and Edwards 2006; Hanke et al. 2014; O' Connor et al. 2011; Scott-Shaw and Morris 2015). The South African Government's National Development Plan (SAGNDP) 2030 has recognized the need to protect the natural environment in all its aspects (Government of South Africa 2012). However, empirical evidence on the effects of livestock intensity grazing on plant species diversity in South African montane grasslands is scarce (O'Connor et al. 2010). The question of whether heavily grazed communal rangelands have reduced plant diversity or whether protected areas under lenient grazing have maintained plant species diversity is of interest in this study. A related question is whether a gradient in grazing intensity exists from where livestock are corralled (or kraaled), a region of high intensity, to outlying areas where livestock grazing intensity is lower. This can be likened to the study of a piosphere, which "is an ecological system of interactions between a watering point, its surrounding vegetation and the grazing animal" (Lange 1969). Grazing intensity gradients of this nature provide an advantageous means of assessing the impacts of livestock grazing both on plant communities and soil properties (Thrash and Derry 1999; Morris 2017; Wang et al. 2017).

Fence-line contrasts are another suitable approach for studying the effects of grazing provided the management histories on both sides of the fence are well known, and that management has been consistent on both sides of the fence-line for at least ten years (Todd and Hoffman 1999; Noy-Meir and Oron 2001; Teague et al. 2008; Zimmermann 2009).

This survey sought to examine the impacts of livestock intensity grazing on plant species composition and plant species diversity using a grazing gradient, akin to a piosphere, and fence-line contrasts between lightly grazed protected areas and heavily grazed communal rangeland.

1.2 Research aims, objectives and questions

1.2.1 Aims

- **a.** Determine the impact of communal livestock grazing on vegetation composition, diversity and plant groups in the northern Drakensberg montane grassland.

1.2.2 Objectives

a. Assess the impact of communal livestock grazing using a gradient of grazing intensity on vegetation composition and diversity.

b. Assess the impact of communal livestock grazing using fence-line contrasts between protected areas supporting wildlife versus communal grassland grazed by livestock on vegetation composition and diversity.

c. Account for the effect of fire and other confounding environmental factors when assessing vegetation responses to livestock grazing.

1.2.3 Questions

a. Does a gradient of grazing pressure exist with increasing distance from where livestock are kraaled?

b. If this gradient exists, is there any corresponding pattern in botanical composition, plant diversity and plant groups?

c. Are there any differences in botanical composition, plant diversity and plant groups across a fence-line contrast between communal and protected areas?

d. Are grazing gradients confounded with fire regime or soil properties?

Chapter 2: Literature review

2.1 Introduction

The loss of biodiversity is a worldwide phenomenon. FAO (2013) estimates that between 26% and 70% of the world's agricultural area is covered by the grassland biome; the system is home to an estimated one billion inhabitants around the globe. The Grassland Biome in South Africa covers 16.5% of the land surface (Neke and du Plessis 2004). Despite an extensive global distribution of grassland, the Grassland biome is a threatened biome (Gibson 2009). In SA, less than 5% of the Grassland Biome is protected (DEAT 1996), a total which falls well below the recommended percentage. The International Union for Conservation of Nature (IUCN) recommends that a minimum of 10% of any landscape should be conserved in order to save 50% of the biodiversity it contains (Shafer 1999). It is estimated that at least 29.2% of the Grassland Biome in SA has been transformed by human land use and is second after the fynbos in terms of the most threatened biodiversity rich biomes (Fairbanks and Benn 2000; Neke and du Plessis 2004).

Grasslands are heterogeneous and gradually evolving systems with high diversity and take long periods to return to their natural state after intense heavy disturbances (Jacobs et al. 1999; Little et al. 2005). These natural ecosystems are important as they provide ecosystem services, notably forage for livestock and wildlife (Fisher 2013). It is estimated that at least 26.7% of South Africa's Grassland Biome remains untransformed or in a semi-pristine state and 56% is still under a natural grassland state (Matsika 2007). Some of the common land use transformations include afforestation, pasture production, and irrigated or dryland cropping (Allan et al. 1997; Lipsey and Hockey 2010). Most of the existing native grassland is largely disintegrated and inadequately conserved (Mucina and Rutherford 2006; SANBI 2014a). A large portion of South Africa's grassland biodiversity resides in rangelands. A common view is that these areas are substantially overstocked and subsequently overgrazed (Shackleton 1993). Communal areas are said to have double or treble the stocking rates compared to commercial areas (Bond 1999; Cousins 1995).

Communal rangelands in South Africa have been subjected to a multiplicity of land lease forms (Scoones 1994; Walker 2002). Unlike commercial rangeland, communal rangeland is open access with diminished individual responsibility to natural grassland management (Ainslie 1998; Kotze et al. 1999). Despite that, the juxtaposition of communal, protected and commercial areas reflects a concession that not all communal rangelands are in a state of degradation. Certain communal rangelands may hold increased basal cover and more palatable grasses than adjacent protected areas (de Bruyn et al. 1998; Goqwana 1998; Shackleton 1998). Venter et al. (1989), Shackleton (1998) and other studies have reported slight differences in soil erosion or herbage accumulation between communal rangeland and protected or commercial rangeland. This evidence therefore suggests that communal rangelands should be approached on a case-by-case basis and not based on generalizations (Peden 2004).

Rural households have a range of reasons why they normally keep a large number of livestock at their disposal. Livestock are utilized for milk consumption or selling, power, lobola, fertilizer, ritual slaughter and transportation (Thobela et al. 1998; Allsopp et al. 2003; Letty et al. 2003; Dovie et al. 2006). In the past, livestock were used as a symbol of status and wealth (Cousins 1995; Thobela et al. 1998), but current studies argue that it is not customarily the rationale (Letty et al. 2003). The intention of communal households in keeping high stocking rates makes economic sense at the individual household level because of these livestock, hence rangelands, support a range of livelihood strategies.

Livestock in communal rangelands are normally kept in kraals during night-time and are released during the day-time. The existence of a cardinal point for animal activity in a landscape creates spatial impacts as a function of animal density (Andrew 1988; Palmer et al. 2003). The focus of herbivore densities around a water station brings about a piosphere (Lange 1969). This is a gradient of grazing intensity which is at peak closer to the watering point and decreases per unit distance away from it. A similar gradient of utilization should establish around a kraal site.

The movement of livestock is generally done by herders or pastoralists in many communal rangelands. Herding is considered to be a critical method in the effective supervision of diverse natural grassland forage availability and highly inconstant precipitation (Ellis and Swift 1988; Behnke et al. 1993; Scoones 1994; McCabe 1994). Nomadic herding is assumed to amplify the quantity of livestock that can continuously forage an area (Sandford 1983; McCabe 1994).

Topography is generally a major influence on the hydrology and climate of mountainous environments. It further influences the distribution of soil properties, plant growth and development. Topographical variation in grasslands can thus create differences in plant species and associations in physical environmental variables namely aspect, elevation and slope (Bennie et al. 2006; Gong et al. 2008). Elevation gradients in mountains can be associated with an increase in precipitation and solar radiation and a decrease in temperature as elevation increases (Barry 2008). Plant species richness generally decreases with increasing elevation (Whittaker 1952, 1960; Rahbek 2005; Watkins et al. 2006).

Fire has a key ecological role in the terrestrial ecosystem and especially on the Grassland Biome. Both anthropogenic and natural fires are a common phenomenon on the montane grassland of Southern Africa (Everson and Hatch 1999). Many montane grassland plant species are adapted and dependent on fire to complete their life cycles and to maintain their populations (Tainton and Mentis 1984; Bond et al. 2003; Uys et al. 2004; Joubert et al. 2017). Without fire many areas of the Grassland Biome and montane grassland could potentially transform into closed woodlands (Hilliard and Burtt 1987; Trollope 1999; O'Connor 2005). Fire also increases tiller production of the grass sward (Everson et al. 1985; Trollope 1999) and promotes resprouting of natural mesic grassland forb species from their underground storage tissues (Fynn et al. 2004). It is thus important to clearly recognize the role of fire when studying these grassland systems.

2.2 Communal rangelands grazing in the context of South Africa

Rangelands are believed to constitute more than 50% of the terrestrial landscape of planet earth and >70% of South Africa's surface area (Hoffman et al. 2018). Most of the land in South Africa is under private tenure, namely 79%, 14% is under communal tenure (former Bantustans) and 8% is unaccounted for (Scogings et al. 1999; Land audit report 2018). Livestock grazing is one of the main uses of communal land, together with cropping and residential areas (Cousins 1999). It is estimated that half of South Africa's livestock is in the communal areas, and that they support a quarter of South Africa's population (de Bruyn 1998; Scogings et al. 1999). Communal rangelands in South Africa are mainly found in the former homelands (QwaQwa, Kwandebele, Kangwane, KwaZulu, and Lebowa) and the former TBVC (Transkei, Bophuthatswana, Venda, and Ciskei) states.

Animal husbandry is an important economic activity in South African communal rangelands, although livestock ownership in these areas is generally skewed towards the few wealthy and politically connected members of the community (Cousins 1995; Scogings et al. 1999). Livestock are of good value in communal areas because they provide multiple scarce resources (meat, milk, cattle loaning and manure) (Shackleton et al. 2000), and other uses such as animal traction, investment, and use in traditional ceremonies. These scarce resource benefits are also shared with the non-livestock holders. It is estimated that 7% of the take home profit generated from livestock, accumulate to family units not in a position to own livestock (Shackleton et al. 1999). In addition, livestock are important to communal farmers because they are less vulnerable to failure than crops, and they provide financial security. As a consequence, livestock owners generally wish to own more cattle, which generally result in overgrazing as the number of cattle exceeds the carrying capacity of the land (Claassen et al. 1991).

Historical records show that the numbers of cattle, sheep and goats per capita of human population declined by almost two thirds between 1924 and 1974 (Knight and Lenta 1980; Lahiff 1997). A proportion of communal livestock household owners decreased from 71% in 1948 to only 30% in 2000 (Ainslie 2002). However, the national livestock survey from 1996-2018, shows the amount of livestock in communal rangelands has remained significantly high (DAFF 2018).

The problem with overgrazing in these rangeland systems has been a contentious one over the past years precisely over the numbers of livestock. Once upon a time it was a norm to associate the commonage grazing rangelands set up in most African villages with high grazing intensity and long-term soil erosion, however research studies done in other villages of southern and eastern Africa subsequently the late 1980's has disputed this one-dimensional representation (Ellis and Swift 1988; Behnke et al. 1993; Scoones 1993). It has been argued further that both semi-arid regions and savanna grazing rangelands in Africa are not equilibrium environments, but can be regarded as having high and spatial temporal variability of grazing natural grassland. This allows cattlemen to carry livestock across the land in order to capitalize and forage the freely accessible natural grassland. This is in contrast to grassland systems that are in a state of equilibrium which adhere to an expected continuity model and can also be determined by controlling ungulates numbers, whereas in non-equilibrium systems changes in grazing grassland can be closely aligned to changes in rainfall rather than changes to ungulates numbers. The non-equilibrium approach further argues that it would be unproductive to apply low stocking numbers in systems whereby the rainfall is not predictable, because farmers that are moderately stocked up would not have an enough opportunity to increase their herds in time to forage all the grazing land on hand during rainfall years, without turning to buying in new livestock (DeAngelis and Waterhouse 1987; Ellis and Swift 1988; Westoby et al. 1989; Behnke et al.

1993). Many scholars have now agreed that both equilibrium and non-equilibrium are present in rangelands (Fernandez-Gimenez and Allen-Diaz 1999; Desta and Coppock 2002; Briske et al. 2003). Instead of taking the approach of either or the rangeland systems can rather be described as continuum between equilibrium and non-equilibrium (Wiens 1984; Derry and Boone 2010). This is where everything in between is in dis-equilibrium and the position along the gradient is influenced by the strength of coupling between plants and animals (Vetter 2005; Derry and Boone 2010). There is immense critique though about the extent to which the dis-equilibrium concept applies to South African rangeland systems and the gravity of degradation in communal rangelands (Fernandez-Gimenez and Allen-Diaz 1999; Cowling 2000; Desta and Coppock 2002; Sullivan and Rohde 2002).

In conclusion, traditional concepts of heavy intensity grazing are subjective and are found upon generalizations obtained from commercial farming, or what commercial farming in a certain farmland system should represent (Shackleton 1993). Shackleton (1993) further argues that both arid and semi-arid grasslands are naturally unstable rangelands, controlled by precipitation and fire. These unpredictable and severe environmental conditions need an adaptive and flexible management system. Therefore, management systems must take into account the interests of livestock, the environment and pastoralists.

2.2.1 Carrying capacity of communal rangelands in South Africa

The term carrying capacity with reference to livestock in communal areas has been a source of confusion for planning of communal grazing systems. The debate that has arisen in the literature concerning whether communal rangeland management centres on ecological versus economic carrying capacity (Abel 1993). Smith et al. (1999) indicated that the advancement of a flourishing grazing management structure is primarily reliant on the number of ungulates that a particular area of veld (or rangeland) can potentially handle. Economic carrying capacity is the stocking rate that offers maximum economic return in a market economy and is determined by the output of the marketable meat and milk which depends on the condition of individual animals (Abel 1993; Grossman et al. 1999). This is in contrast to communal farmers, where the value of the cattle is dictated by a number of factors such as draught power, transport, manure and milk production, food security, and as stable investments in times of death of family members. These values are achieved through holdings of large numbers of livestock.

The view that high stocking rates are as a consequence of irrational behaviour and poor management are therefore baseless to communal farmers. High stocking rates are rather an economic necessity for rural people. Cattle are therefore usually maintained at an ecological carrying capacity determined by external factors including climate, soils, and vegetation type (Abel 1993; Grossman et al. 1999). Rural people tend to concentrate their efforts on the maintenance and survival of animals and ignore the ecological degradation of the rangeland.

Many different plant species are used every day by communal rangeland communities. Communities in northern South Africa, for example, often make use of approximately 300 plant species (Shackleton et al. 1999; Magasela et al. 2001). The southern parts of South Africa use fewer plant species, despite higher biodiversity in those regions (Shackleton et al. 2002). Traditional healers may utilize a plethora of several plant species and with most homesteads use approximately 24 species for self-treatment for a number of minor illnesses (Shackleton et al. 2001). The use of plant species as a resource in these communal rangelands is not uniform. Work conducted in India, Zimbabwe and South Africa indicate that uses differ between well-off and struggling homesteads, with the poor households being more reliant upon goods and services furnished by the communal rangeland (Qureshi and Kumar 1998; Cavendish 2000; Ntshona 2001; Shackleton et al. 2002).

2.2.2 South Africa's betterment scheme

There have been a number of efforts by the previous government and donors to bring about agricultural development in communal areas but without much success. In the 1950s, a betterment planning scheme was introduced in an attempt to improve land management in rural communal rangelands in South Africa through introducing a natural resource management scheme that divided the communes into three main resource areas, *viz*. crops fields, rangeland, and residential areas, each of which had a different resource tenure organization and procedures (von Maltitz 1998; Allsopp et al. 2003).

The betterment scheme substituted the traditional methods of natural resource management and enforced new regulations on managing cattle movements without involving the community. This top-down approach excluded local people from decision making. Although the scheme had ecologically sound principles and was well structured, it fell short in solving the problems of environmental degradation because of lack of consultation and participation of local communities (Everson and Hatch 1999). Unsuitable institutional frameworks, poorly implemented rural development assistance, and the lack of impartial local governance systems are some of the reasons that led to the general failure of betterment programmes (von Maltitz and Evans 1998; Hesse and Trench 2000). QwaQwa was an example of a betterment scheme. The limited success of the betterment scheme is that it was a top-down planning approach although aimed at maintaining or improving the state of the natural grassland. The scheme would have succeeded if it was supported by and involved the community, and that it demonstrated benefits for the community and not simply for a few farmers. The aim should have been a value-added natural resource scheme set up and managed by the community (Squires et al. 1992; Hesse and Trench 2000).

2.2.3 QwaQwa Communal area

QwaQwa is located in the foothills of the Maloti Drakensberg. The mountainous former homeland covers 620 km^2 (Figure 2.1). It was originally inhabited by the San people from about 8000 years ago (Irwin et al. 1980). Bantu speaking people, who owned livestock and practiced agriculture, arrived in QwaQwa and elsewhere in the Drakensberg toward the end of the 17^{th} century (Bank 1995). In 1867, QwaQwa was occupied by the Sotho people under the chieftaincy of Wetsie (Bank 1995). The then Orange Free State (OFS) Government offered Witsieshoek to a group of refugees under the leadership of Mokhachane who had supported white settlers in the Sotho wars (Quinlan 1986).

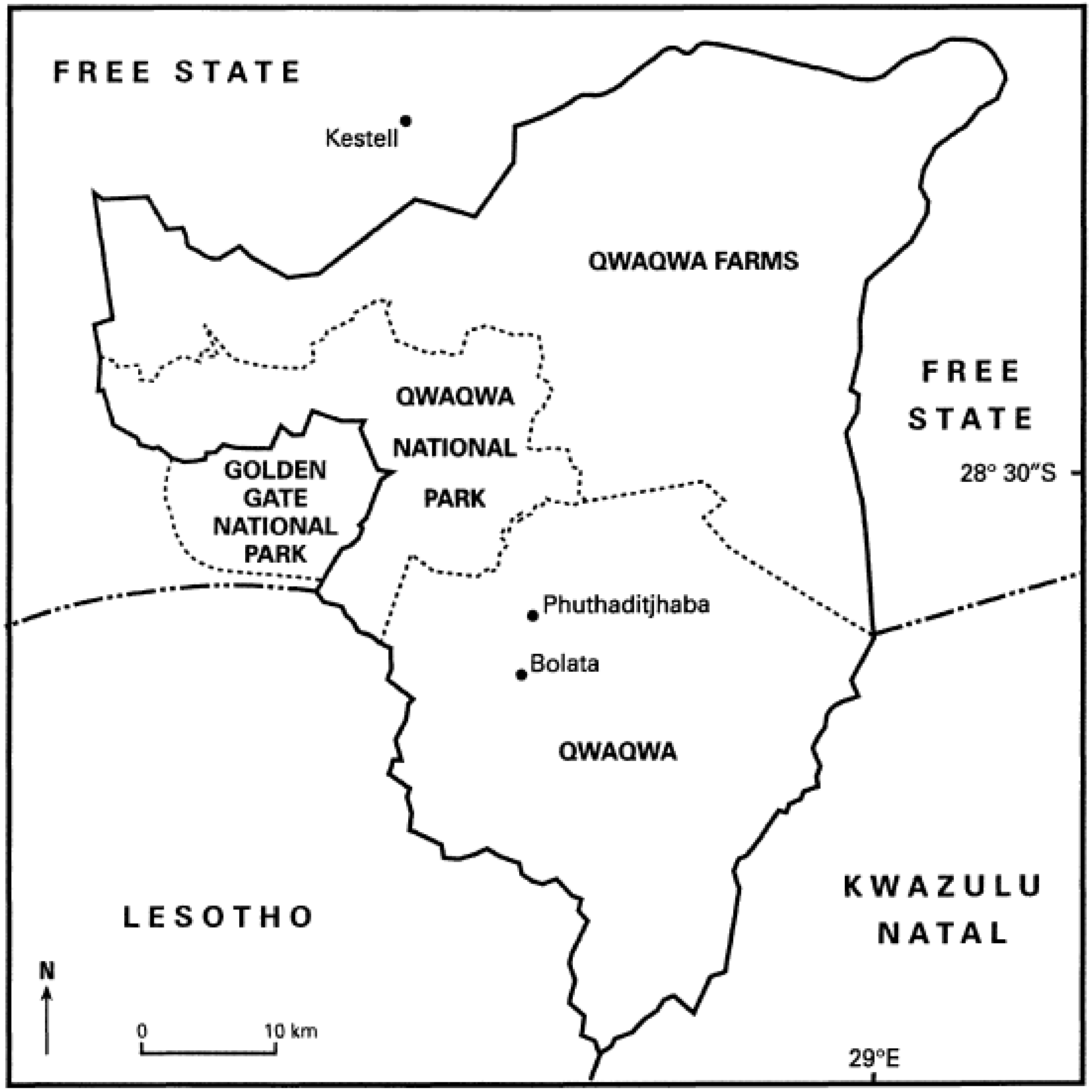

Figure 2.1. Map showing Greater QwaQwa. (Source: Slater 2002).

The agreement signed by Mokhachane and the OFS government recognized Mokhachane as a chief of the Kwena tribe (Bank 1995) and marked the birth of the BaKwena tribe. Similar to other rural areas in South Africa, QwaQwa was administered by cultural traditional leaders up until 1974, and thereafter it became a homeland (Slater 2002). With only 12% of the available land being arable, the area offered limited opportunity for growing cash crops (Vrey and Smith 1980).

The Batlhokwa tribe was also settled in the area by the OFS government in 1875. The Batlhokwa inhabited the higher areas that were appropriate for grazing only (Kotze 1982). QwaQwa has many villages, with Phuthaditjhaba being the main one (Slater 2002). After the area became a homeland in 1974, the population increased nearly tenfold (Table **2.1**). Most of the people that arrived were farm workers from central and southern parts of the Free State. It was one of the highly densely populated homelands, with most of the arable land (2000 ha) occupied by residential housing (DBSA 1986). This did not help much because it offered the smallest amount of arable land (0.02 ha) per family of all the homelands.

Table 2.1. Human population size of QwaQwa since 1916

Year	**Population**	**Population density per (km^2)**	**Source**
1916	5000	8	(Beaumont commission report 1916)
1950	12000	25	(Murray 1983)
1970	24000	50	(SAIRR 1977)
1980	200 000	415	(Surplus Peoples Project 1983)
1989	450 000	933	(Sharp 1994)
1996	353 238	81.43	(Census 1996)
2001	360 549	83.11	(Census 2001)
2011	335 784	77.41	(Census 2011)
2016	353 452	81.48	(Census 2016)

After the 1994 democratic elections, a Land Reform Programme was initiated in order to redress the racial imbalances in land ownership through redistribution of land. A section of 22000 ha was proclaimed the QwaQwa National Park (QNP) in 1992. The park boundary was fenced whereas internal fences were removed in order to allow for free activity of 3000 wild animals that were brought in in 1993 (Murray 1998). QNP was administered through the Highlands Development Corporation (HDC). After the dawn of democracy QwaQwa was no longer independent and became part of the new Free State province (FS), when it forfeited its statutory role as an independent homeland. In 1995, QNP was put under the jurisdiction of Agri-Eco, a government parastatal that emerged from the re-organization of former QwaQwa government agencies (Slater 2002).

The fence that divided QNP from the adjoining Golden Gate Highlands National Park (GGHNP) was removed in 1992, which meant that QNP acted as a security area for curbing poaching and land invasion in GGHNP (Slater 2002). Prior to the land being declared a national park, residents had used the park for livestock foraging and subsistence farming without any restrictions. After 1992 when QNP was proclaimed a park, use of this land by residents, particularly for livestock grazing and extraction of resources, became regulated. The legislation decreed that residents had no land tenure or grazing rights in the park. This presented a conflict and a dilemma between the park residents and park management, because residents were dependent on the park for acquiring wage labour. The park management was trying to meet responsible conservation requirements and at the same time contribute to the human development of park residents.

A limit of 15 large stock units per household was introduced in 1992. This quota was based on concerns over soil erosion. Other restrictions that were introduced to the residents of the park included cutting of forests, water use and harvesting of medicinal plants. Responses from residents were mixed. Some residents within the park opted to move their livestock and paid a fee for grazing on farms that belonged to black commercial farmers. The majority of the communal farmers refused to cut down their livestock and continued to graze their cattle in QNP.

The use and conservation of rangeland by chiefs in the QwaQwa area goes back before the colonial times. This was done through seasonal preservation of grass, whereby grazing was minimally allowed along the river banks and on the veld during summer so that there would be grass for winter grazing (Mwalukomo and Patel 2012). Grass was also used as weaving material for household goods including hats, mats, baskets, ropes and brooms. However, the practice of conserving grass has diminished over the years. For instance, the betterment policies of livestock culling that were introduced in the 1950’s were in contradiction with traditional and cultural practices. The chiefs preferred indigenous methods such as slaughtering an animal during annual and circumcision ceremonies (Bank 1995). This traditional decline of conservation of grass continued further during the apartheid period. The chief’s independence was compromised and they had no power because they were paid by the government, hence under such conditions it would have been difficult to stick to traditional methods of grass use (Goodenough 2002). The dawn of democracy in 1994 may also have added to the decline of these practices, as councilors and chiefs have alleged that people were starting grass fires as a means of demanding their democratic rights. Today, these traditional practices and conservation of grass are no longer practiced. This is due to the current commercialization of grass weaving which is in high demand in cities, which inevitably has increased the demand for grass. The grass weavers have extended their supply by procuring grass from outside QwaQwa. This meant that chiefs have lost their grass supply chain and are no longer involved.

2.3 Grazing effects on plant species diversity in the context of South Africa

There have been general concerns expressed about the impact of heavy livestock grazing on plant diversity in South Africa. South African grassland systems are being degraded through continuous increased grazing pressure (Neke and du Plessis 2004; Anderson and Hoffman 2007; O' Connor et al. 2011). Although grazing by livestock is perceived to be impacting negatively on plant species diversity, grazing cannot simply be ignored in managing the grassland system. Grazing and accompanying trampling action by wild ungulates and nutrient enrichment have an important role on montane grassland plant diversity (Collins 1992; Blair et al. 2008; O'Connor et al. 2011). For example, selective grazing may change the make-up and species composition of montane grassland (Milchunas et al. 1998; Owen-Smith 1999). Rangeland biodiversity studies in South Africa have tended to focus on commercial ranching, with generalized findings applied to communal rangeland (O'Connor et al. 2010).

Grasslands are among the most diverse vegetation communities on the earth, and one of the most diverse biomes in South Africa (Wilson et al. 2012). A number of authors have suggested (Shackleton et al. 1994; Hadden and Westbrooke 1999; Niamir-Fuller and Turner 1999; Pykala 2000; O' Connor et al. 2010) that grazing can lead to either an increase or to a decrease in plant diversity and species richness (Todd and Hoffman 1999). These consequences may be due to various external environmental factors such as rainfall patterns, habitat productivity, stocking rates and soil substrates. Plant species diversity is ordinarily much higher on nutrient poor substrates than on nutrient rich substrates (Tilman 1988). Observations by O'Connor et al. (2011) from natural groups of large ungulates and from livestock at low stocking frequencies indicate that plant species richness is normally favoured under these conditions. O'Reagain and Turner (1992) have shown that stocking rate has the greatest effect on range condition and livestock production.

In the southern Drakensberg region, the impact of grazing on grassland plant diversity was slight in conservation areas, intermediate on commercial rangeland, and strong on communal rangeland (O'Connor 2005). However, it has been suggested that grasslands which are continuously grazed, tend to be subjected to major deterioration in plant species diversity (O'Connor et al. 2010).

There is limited empirical evidence supporting the notion that intense livestock communal grazing has adverse effect on plant species diversity. In southern KwaZulu-Natal, grazing influenced species composition but its influence on the average number of species per square metres was evident for protected areas (78 species) being greater than communal (52 species) or commercial (49 species) (O'Connor 2005). The findings presented on work done at Cathedral Peak protected area and a nearby communal area produced inconclusive results of the impact of livestock on plant species diversity, but the difference in stocking density between the two rangelands was not known (Peden 2004). The work presented by Hoare (2003) in the grassland of the Eastern Cape Province found that there was less mean species richness per square metres in communal areas (26.3) than in commercial areas (33.8). In support, declines in plant species diversity were observed on communal rangelands which were thought to be in a poor condition for livestock production than grassland in commercial farming systems (Kiguli et al. 1999).

A number of authors have hypothesized that under heavy grazing the competitiveness of palatable and dominant grass species is reduced thereby promoting shorter and less palatable species. Continuous grazing by ungulates may diminish the forage value of rangelands and plant species diversity and richness. A fence-line contrast study in the Succulent Karoo shows that high grazing intensity promotes unpalatable shrub species (Desmet 2007). Heavily grazed areas are normally dominated by higher numbers of annuals, species with a basal growth form, and an increasing abundance of alien species (Borchardt et al. 2013). After these areas have been degraded by livestock grazing, seed availability and dispersal limitation become strong inhibitors for the establishment of productive natural grassland (Tadey 2006; Simons and Allsopp 2007).

Equally, lack of grazing can also result in unpalatable species dominating and excluding other species, thereby reducing diversity (Milchunas et al. 1988; Osem et al. 2002; Cingolani et al. 2005; Altesor et al. 2006; Cingolani et al. 2008). In addition, lack of grazing may potentially result in decreased diversity of palatable species in grasslands (West 1993; Tainton 1999; Allsopp 1999; Watkinson et a1. 2001). Cingolani et al. (2014) also found that when grazing is excluded, woody plants and tall tussock grasses are promoted. These tall-tussock grasses compete for light with shorter plants eventually driving them out (Westoby 1998; Weiher et al. 1999). Grazing by wild ungulates and domestic herbivores supports the increase of more nutritious and palatable species in some rangelands. Tremont (1994) has shown that native plant species decline in the absence of grazing. Grazing can also keep grasses from becoming moribund and dying out (Tainton 1999).

The Drakensberg grasslands of South Africa are a significant reservoir of plant diversity, in which many endemic species are found (Cowling and Hilton-Taylor 1997). The Drakensberg harbours a rich flora of 2520 vascular plants, including 334 endemic and 595 near endemics plants (Carbutt and Edwards 2006). The foothills of the Drakensberg grassland provide the large human population living there with the goods and services of livestock farming, regulation of catchment water supply, medicines, and building material (Turpie et al. 2008). Hence, these grasslands are susceptible to transformation and there is a need to understand these dynamics.

The mesic rangelands have most likely never maintained the grazing intensities presently experienced in communal rangelands (Uys 2006). The grass swards in this region are highly productive over the summer growing season, but their palatability decreases over winter (Ellery et al. 1995; Tainton 1999). Continued intense grazing of South Africa's mesic rangelands has resulted in smaller tufts, decreased basal cover and a change to less palatable grass species compared to adjacent commercial systems (O'Connor et al. 2003). This raises concerns to conservation managers over the degradation rate of these rich grasslands. These grazing effects however have not been explored adequately along a gradient of grazing intensity and through using fence line contrasts in these grasslands.

2.4 Piospheres and 'fence-line' contrasts

The movement of livestock in many arid environments is ordinarily centered on water points. The principal idea behind the piosphere concept is that a gradient of grazing impact exists and is maximum adjacent to the water stations and declines as per unit of distance from it (Thrash and Derry 1999). The site neighbouring a water station undergoes greater pressure and is called the sacrifice zone. By contrast, sites at distances less travelled by ungulates from water points encounter the least utilization intensity (Van Wyk and Fairall 1969; Graetz and Ludwig 1978).

In winter watering points are not easily available, because it is the dry season. Hence, grazing pressure is concentrated mostly on one or few remaining perennial watering points. Grazing pressure near the water points only decreases at the start of the rainy season (Funston et al. 1994).

The following ecological variables can serve as indicators of piosphere pattern: build-up of livestock faeces, a gradient of soil nutrient availability with a maximum near water and declining away from it, and corresponding gradients for soil compaction, amount of bare soil, reduced grass basal area, and biomass of herbage defoliation. The amount of faeces accumulating at a site is a useful example indicating the occupancy of a site by animals. A grazing gradient affords an opportunity to investigate ecological patterns and processes in response to grazing pressure at a relatively fine scale. Each animal species present at a watering point should cause a spatial pattern in accordance with its foraging behaviour; the foraging distances of ungulates would define the piosphere range of each that would collectively create a piosphere pattern (Derry 2004).

Spence and Angus (1971); Edroma (1981); Friedel (1988) have found that the assemblage of grazing ungulates utilization in the area of newly built watering point cause a reduction of a number of grazing-intolerant groups of herbaceous plant species. Perkins and Thomas (1993) found that in particular, the palatable species preferred by grazing ungulates declined in number. On the other hand, Thrash et al. (1993) showed no uniform trends associated with distance from water points and herbaceous plant species richness. Tolsma et al. (1987) in Botswana also found that species diversity decreases adjacent to water stations. These studies are not sufficient and are confusing to inform land management of these areas.

An equivalent of the concept of a piosphere is the foraging pattern in relation to stocking posts. A gradient of grazing pressure declines with distance from kraals generally prevails around stocking posts is created when ungulates are sent out every day from a stock post to forage (Hoag 2016).

The use of kraals to examine the effect of grazing along a grazing gradient remains largely unknown in South African mesic grasslands. However, in the Succulent Karoo grazing intensity is much higher in the area closer to the stock post (Riginos and Hoffman 2003). This approach would be applicable in the mesic grasslands or sub-humid environments of South Africa where such studies have been lacking. Therefore, equivalent to the piosphere concept, this study seeks to investigate grazing gradients emanating from kraal sites in mesic/sub-humid montane grasslands of the Drakensberg.

Fence-line contrasts can be regarded as natural “experiments” that provide a unique opportunity for examining long term impacts of heavy grazing when such differences have existed for a long time either side of the fence (Todd and Hoffman 1999; Noy-Meir and Oron 2001; Todd and Hoffman 2009; Chamane et al. 2017; Nenzhelele et al. 2018). Grazing management has been markedly different between communal and protected areas in South Africa. The vegetation of communal areas is mostly seen to be impacted heavily in contrast to adjacent protected areas and commercial areas (Tapson 1993). In Karoo vegetation, a prominent change linked with communal lands has been a decrease in palatable woody and succulent shrubs and a change towards annuals and geophytes (Todd and Hoffman 1999). However, research conducted from akin semiarid Mediterranean regions suggests contrasting findings (Naveh and Whittaker 1979; Noy-Meir et al. 1989; Olsvig-Whittaker et al. 1993). A long term (20 years) study of a fence-line contrast in the Namaqualand South Africa observed a decrease of plant cover in both communal and commercial rangelands (Nenzhelele et al. 2018). The key question is whether there is a difference in plant species richness and plant species composition in the mesic grassland system when contrasting the communal and protected rangelands.

2.5 Impact of livestock grazing on plant groups

Grazing is considered as a key influence affecting the morphology and structure of plants and regulating the assembly and performance of grassland systems (Milchunas and Lauenroth 1993; Pakeman 2004; Cingolani et al. 2005; Semmartin et al. 2008). Plant functional types have been studied to understand the shifts to environmental changes and disturbance as well as trade-offs between various roles contained by a plant (Diaz et al. 2001; Garnier et al. 2001; Wright et al. 2004).

Livestock grazing has since been recognized as an important influence on plant functional types (Diaz and Cabido 2001; Cingolani et al. 2005; Klimesova et al. 2008; Pillar et al. 2013; Hu et al. 2015). A functional adaptation that is widespread in a plant community can provide insight about ecosystem processes. Plant functional types are becoming an emerging key tool in comprehending plant growth and survival in relation to environment. These tools have recently become popular in predicting the effects of grazing on plant species fauna and flora (Zheng et al. 2010).

Dorji et al. (2013) have shown for specific spiny species that leaves of grazed seedlings are smaller in surface area and have more surface spines than ungrazed leaves. Lamas et al. (2013) have reported that plant cover is the trait most affected by grazing pressure at plant community and at species levels. This decrease in plant cover has been reported widely across many ecosystems, including arid and semiarid ecosystems (Schlesinger et al. 1990; Bisigato and Bertiller 1997; Bertiller and Bisigato 1998; Whitford 2002).

Other findings suggest that plants with plant functional types such as a small growth form, lateral spread, or an annual life cycle can withstand high grazing pressure (Castro et al. 2010; Wesuls et al. 2012; Borchardt et al. 2013). Diaz and Cabido (2001) have also shown that species with shorter height and smaller, more tender leaves respond positively to grazing pressure. In this study plant groups (soft traits) were used to infer to plant functional types. In this study I identify a set of plant groups that indicate the impact of grazing pressure.

2.6 Spatial distribution of livestock grazing

There are various factors that influence the spatial pattern of livestock grazing in rangelands. Biotic and abiotic elements may influence the accumulative pressure of spatial distribution of livestock grazing along a grazing gradient (Wu and Loucks 1995; White and Jentsch 2001; Smith et al. 2009). In many types of rangeland grasslands, the distribution of livestock grazing can be influenced by topography, fire, soil, water, nutrient availability, kraaling/shelter and track density (Risser et al. 1981; McNaughton 1983; Gibson 1988; Collins and Gibson 1990; Vinton and Hartnett 1992; Vinton et al. 1993; Noy-Meir 1995; Cingolani et al. 2003).

The most recognizable limiting factors on grazing distribution are the sources of food and water (Adler et al. 2001). The hierarchical foraging model of Senft et al. (1987) states that ungulates often opt for the grass sward that is loaded in resources, followed by the most productive communities, down to the palatable species within a feeding post. In most cases in grazing systems, grazing pressure decreases exponentially away from water points (Valentine 1947, Andrew 1988; Pickup et al. 1998; Turner 1999; Nash et al. 1999). The physical reliance of livestock on water results in movement being intensive in the vicinity of permanent water (Lange 1969; Foran 1980; Andrew and Lange 1986; Ludwig et al. 2000). Hence, water points play a key role in structuring where livestock pressure accumulates. In addition, physical barriers such as steep slopes may restrict access to high resource areas (Coughenour 1991; Bailey et al. 1996).

Grazers prefer grazing locations that are near water points, areas with gentle slopes and areas with high quantity and quality vegetation (Stephenson 2015). Herders around the world have been able to sustain livestock in changing environments for ages (Fernandez-Gimenez and Swift 2003). They have been able also to coordinate their activities with spatial and temporal availability of resources. Herd mobility in communal rangelands has been seen as an ecologically sound means of maximizing the use of the limited resources (Adriansen 2003; Niamir-Fuller and

Turner 1999), it is also a risk avoidance method (Swallow 1994), and an approach that maximizes digestible nutrient intake during the growing seasons (Western 1982).

Track density can be used to provide a spatial description of shifts in grazing movements. The areas of greater track density normally indicate zones that have been utilized more intensely and for longer periods (Pringle and Landsberg 2004). Livestock tracks generally diverge out from water stations and become less clear as ungulates spread out to graze (Arnold and Dudzinski 1978). Therefore, areas of high track density should reveal areas where grazing activity was heavy in the past, but more current grazing activity could be taking place in areas where distinct tracks have not yet developed (Pringle and Landsberg 2004).

Stocking history also controls livestock movements, because, if the grasslands around water points become dormant, livestock are forced to travel long distances away from water stations to graze (Hunt 2001). Therefore, the amount of disturbance is most likely influenced by kraal age and past stocking rates. In addition, areas that regenerate after fire tend to entice livestock as well (Dyer et al. 2001). Therefore, in this study we seek to quantify these confounding gradients and to determine their impact on plant species diversity along a grazing gradient.

2.7 Spatial patterns of plant species diversity

The geographical patterns of biodiversity along environmental gradients have been debated in ecological and environmental sciences. It is pertinent that we comprehend how biodiversity is distributed across South Africa's rich rangeland grasslands, as this will assist conservation planning in these grasslands (Cowling et al. 1997). According to Rosenzweig (1995), both plant species richness and species diversity are not distributed randomly in space but they follow discrete spatial patterns along environmental gradients (Orme et al. 2005; Stohlgren et al. 2005; Hortal et al. 2013; Slaton 2015). In this study I investigate species richness at regional scales and along a gradient of grazing pressure.

Several hypotheses have been put forward to predict the distribution of plant species richness at a regional scale. These comprise the relations between species richness, sample area, environmental heterogeneity, environmental productivity, environmental favourableness, resource abundance (seasonality and unevenness), the spreading range of species, speciation history and the fashion in which local and regional richness come together (Cowling et al. 1997). However, the most applicable hypotheses seem to be the ones that correlate species richness to patterns of primary resource quantity and quality, i.e. the resource richness hypotheses (Cowling et al. 1997). Mean precipitation determines patterns of floristic diversity across southern Africa (Linder 1991; O'Brien 1993; O'Connor and Bredenkamp 1997; O'Brien et al. 1998; Caylor et al. 2004; Uys 2006).

Topographic diversity usually increases with altitude indicating that an increase in habitat heterogeneity should support greater species richness in the mountains than in lowlands (Uys 2006). In mountains, temperature usually decreases with increasing altitude, which is expected to result in lower species richness and diversity. Mean annual precipitation usually increases with altitude (Schulze 1997), meaning an increase in environmental favourableness and therefore an expected increase in species richness (Cowling et al. 1997). Primary production is directly related to mean annual precipitation over the southern African grasslands, such that increasing species richness might be expected with increasing altitude in mountains if other factors were not involved (Rutherford 1980; O'Connor and Bredenkamp 1997; Cowling et al. 1997). Increasing mean annual precipitation is also associated with a lower coefficient of variation in precipitation (Schulze 1997), indicating fewer extreme years of precipitation should occur that is also expected to result in an increase in species richness (Cowling et al. 1997). Therefore, if one wishes to isolate an effect of grazing on plant diversity in mountains, then one should not unduly confound differences in grazing pressure with differences in altitude.

2.8 Topography in mountains

Topography generally influences local and regional microclimates. Mountainous topography forms highly diverse small scale environmental conditions that may potentially change precipitation, temperatures and insolation as a consequence of aspect and slope angle (Kane et al. 2015). Topography may also influence the energy and water balance through differences in incident solar radiation and the distribution of soil moisture (Armstrong et al. 1988). The north and south facing slopes have largely different environments by virtue of differences in insolation, snow quantity and soil development (Pfister et al. 1977).

Aspect alone plays a key role in determining maximum temperatures, with the south-facing aspects in the southern hemisphere producing lower temperatures than north-facing slopes, with the opposite trend evident in the northern hemisphere (Smithers 1961).

South-facing slopes in the southern hemisphere ordinarily receive less annual solar radiation per unit area and are wetter and colder than other slopes, although north-facing slopes are relatively humid and warm in contrast to the northern hemisphere (Armstrong et al. 1988). East and west-facing slopes depict the same but with less change, with east-facing slopes being open directly to sunlight in the morning when temperatures are lower, making them relatively cooler and moister than west-facing slopes (Barnes et al. 1980). Insolation also controls soil water, soil temperatures and air temperatures which are determinants of the formation and growth of plants (Barnes et al. 1980). Fine-scale variation in aspect and slope implies that slope heterogeneity in mountainous landscapes produces differences in vegetation community composition, structure and ecosystem functioning (Barnes et al. 1980).

Gentle slopes, concave and converging landscapes are envisaged to accrue water in the soil, while steep slopes, convex and diverging areas tend to discard water (Logsdon 2015). Equally steeper slopes have more rapid drainage of soil moisture than flatter slopes (Kane et al. 2015). A study done by Yu et al. (2015) found that the correlation between soil moisture change and precipitation was more evident on sunny slopes than on shady slopes, except only when the initial soil moisture was greater than 40%. On lower slopes, the availability of soil water (Daws et al. 2002; Markesteijn et al. 2010) and soil nutrients increases (Wilcke et al. 2011; Wolf et al. 2011), while wind exposure is reduced. As a consequence, productivity and canopy height are higher on lower slopes than on upper slopes (Webb et al. 1999; Oesker et al. 2008).

2.9 Fauna and Vegetation

2.9.1 Fauna (Ungulates)

The Grassland Biome of southern Africa has various key ungulates that form part of the system, including within the study area. This diversity of ungulate grazers includes *Connochaetes gnou* (black wildebeest), *Damaliscus dorcas phillipsi* (blesbok), *Alcelaphus buselaphus* (red hartebeest), *Redunca arundinum* (reedbuck), *Redunca fulvorufula* (mountain reedbuck), and *Equus quagga burchelli* (Burchell's zebra). These ungulates are thought to have been hunted and poached to extirpation in the latter part of the 19th century AD.

Other indirect factors that have caused the large decrease in these ungulates were human settlement (housing development), horticulture, and intense grazing by domesticated sheep, goat and cattle (du Plessis 1969; Ansell 1972; Lynch 1983).

Historical sources suggest that these grazers migrated from the Sourveld of the eastern Free State, Mpumalanga and Gauteng to the sweet and mixed velds of the western and central

Free State and KwaZulu-Natal adjacent to the Drakensberg-Mountains (du Plessis 1969; von Richter 1971; Lynch 1983). The mixed veld and warmer temperatures of KwaZulu-Natal are thought to have functioned as winter protection for these grazers. They further migrated westward at the beginning of the summer rains to the Sourveld of the eastern Free State (FS) and Gauteng Province (GP). Towards the end of summer, populations migrated to the sweet and mixed veld of the central and western Free State (von Richter 1971).

The movement patterns of these grazing wild animals are generally aligned to the change in nutrient composition cycle of the grassland. According to the climate gradient and soil distribution of the biome, grasses can be divided up into three velds (sweet, sour and mixed veld), subject to nutrient content and palatability at different periods of the year (van Oudtshoorn 1992; Ellery et al. 1995). Grasses with a high nutrient content relative to carbon that maintained their palatability during the entire year are referred to as sweet, in contrast to sour grasses which are only edible during the growing season (Ellery et al. 1995). The relationship to carbon is influenced by light intensity, water availability and temperature. An increase in light intensity causes an increase in the photosynthetic uptake of carbon. In the growing season low mean daily temperatures and high moisture content support more carbon assimilation over nutrient mineralization (Ellery et al. 1995). In certain micro-environments a mixed veld occurs where conditions favour the growth of both sweet veld and sour grasses (Tainton and Mentis 1984; van Oudtshoorn 1992).

2.9.2 Vegetation

The current study area forms part of Grassland Biome of Southern Africa (Mucina and Rutherford 2006). Vegetation type's growth and distribution is mainly influenced by temperature, precipitation and soils in the Grassland Biome. The moist Bioregion of this montane region is influenced by the underlying geology, soils, climate and microclimate. Geology influences vegetation through its effect on soil type and on nutrient availability.

The combination of precipitation, temperature and soil found in the study area favour plant communities that are more than 75% C_4 grasses (Ellis et al. 1980), with herbaceous dicotyledons and woody shrubs constituting most of the remainder. However, above 2700 m the number of C_4 grasses start to diminish with increased altitude (Morris 2013). The number of temperate C_3 grasses begins to increase at this point because of a decrease in minimum and maximum growth season temperatures and evaporation (Killick 1978; Ellis et al. 1980; Tainton and Mentis 1984; Acocks 1988; Morris and Derry 1999). Vegetation growth activity is generally most productive towards the beginning and at the end of the summer. There is limited precipitation in winter, with frequent frosts and low temperatures. These factors limit plant growth.

2.10 Fire and land use

In general, fire and grazing are the two most common and important complementary disturbances of a grassland ecosystem. The history of fire dates back from charcoal records, which envisage that wildfires began soon after the arrival of plants in the Silurian times (420 million years ago). All through the history of terrestrial life, a fire event invites the inference that fire must have prominent evolutionary effects on the fauna and flora (Davies et al. 2009). Grassland ecosystems are old, intricate and slowly developing structures of diverse vegetation communities that normally take long periods to recover after severe environmental stress (Jacobs et al. 1999; Little et al. 2005). However, the disturbance of fire is responsible for maintaining vegetation diversity, species richness and heterogeneity (Collins 1992; Freeman 1998; Uys et al. 2004; Bond and Keeley 2005; Everson and Everson 2016). In the Grassland Biome of South Africa, fire plays a key role in the composition, structure and functioning of the system as a whole (Bond 1997; O'Connor and Bredenkamp 1997). Winter and spring fires generally maintain these grassland systems (Mucina and Rutherford 2006).

Hilliard and Burtt (1987) suggest that fires in the Drakensberg area have also been started by "natural fire regimes" through lightning and rock falls, but humans have been instrumental as well. The occurrences of lightning events have been observed by the Department of Forestry since mid-1950. Killick (1963) reported that most lightning-induced fires occur in the spring months when the grass swards are dry and flammable. Consequently, naturally caused fires have had a large effect on the succession of vegetation in the Drakensberg system bringing about a grassland-dominant vegetation community (Killick 1963).

The use of fire to manage grassland vegetation communities is to offer a green flush of nutrient rich sward, eliminate old plant litter and moribund material, prevent ingress of woody plants and curb ticks (Bond 1997; van Wilgen 1997). Herbivores are attracted to freshly burnt grassland because it provides an increased protein concentration, resulting in a concentrated grazing pressure as soon as enough growth has taken place (Oliver et al. 1978; Rowe-Rowe 1982; Everson et al. 1985a). However, there is concern about the potential unfavourable impacts of unnatural regular fires on grassland communities (Swengel 2001; Collins and Calabrese 2012). Using ecologically sound approaches to fire regimes is important in maintaining productive grasslands.

Fire regimes can have strong effects on grassland species composition. Nanni (1969) proposed that an irregular fire regime was best for maintaining grassland conditions in the Drakensberg. In addition, Edwards (1984) proposed that biennial spring or annual winter burns be applied to maintain good vegetative cover and palatable species for grazing. Uys et al. (2004) found that there were large compositional changes towards woody species if fires were left out for more than ten years. Annual burning regardless of litter accumulation has been shown to have a negative impact on plant diversity in South Africa's moist highland grassland communities (van Wilgen 1997; Everson and Hatch 1999; Joubert et al. 2014). Burning in late spring and early summer in the Highland Sourveld of KwaZulu-Natal can lead to severe loss of grass cover (Dillon 1979; Everson et al. 1985b), and to declines in populations of important palatable grasses such as *Themeda triandra* (Tainton and Mentis 1984). Autumn burning leads to some plants, in particular the grasses, not flowering at all but others flower periodically (Killick 1963). Winter burning allows the flowering of most grasses in spring. Uys et al. (2004) found that fire frequency and season of burn influence dominant grasses and with no consistent trends in forb composition. Grasslands that are excluded from fire are normally composed of perennials, shrubs, C_3 species, unpalatable broad leaved evergreens and deciduous species (Collins et al. 1998; Heisler et al. 2003; Grund et al. 2005; Kahmen and Poschlod 2008).

The loss of cover and habitat has been widely perceived as major drivers of biodiversity loss in the world and in South Africa (Vitousek 1994; Millennium Ecosystem Assessment 2005; Jetz et al. 2007; Jewitt et al. 2015). The province of KwaZulu-Natal's landscape has been experiencing major losses in natural habitat (Jewitt et al. 2015). Most of the grassland of the Drakensberg mountains flora and endemics is protected within the UKhahlamba Drakensberg Park (Metroplan 2001). But the species rich flora of the foothills is threatened by land use change (O'Connor 2005). South Africa has a mean annual precipitation of 450 mm (Schulze 1997), which makes most of the country unsuitable for production enterprises such as intensive agriculture, dairy production, and plantation forestry. Consequently, high rainfall areas such as the Drakensberg foothills are in demand for such activities. Thus the species rich natural vegetation of the mesic grassland is being threatened by loss of natural habitat because of increasing agriculture comprised of orchards, commercial and subsistence crops and timber plantations (Jewitt et al. 2015).

In the past information pertaining to fire in grasslands has been derived from methods that were unaffordable, tedious, time consuming. The recent technological advances in remote sensing have presented unique applications needed for fire management (Zhan et al. 2002; Pereira 2003; Chu and Guo 2015). Remote sensing offers cheap and fairly consistent observations of fire activity through providing information on the magnitude, period of occurrence, locality and spatial extent of fire (Laris 2005; Archibald et al. 2009). Several studies have been shown to be successful in using remote sensing to map out and quantify fire regimes.

Moderate resolution imaging Spectroradiometer (MODIS) and Landsat TM imagery has been used in South Africa to map out burnt areas in the Savanna Biome (Hudak et al. 2004). Archibald et al. (2010) and Goodwin et al. (2013) have utilized the MODIS sensor to produce a visual assessment of fire treatments in southern Africa and a time series of Landsat TM/ETM+ imagery to look at fire frequency in the state of Queensland, Australia respectively. In a more recent study, Buthelezi et al. (2016) undertook a spatial and temporal assessment of fire treatments on different plant types using MODIS, in KwaZulu-Natal province, South Africa.

This study sought to investigate fire return interval and season of occurrence in the study area in order to determine the extent to which fire return interval or season of burning may be confounded with grazing patterns.

Chapter 3: Study area and Methods

3.1 Description and location of the study area

The study was undertaken in the Phuthaditjhaba communal rangelands, Golden Gate Highlands National Park (GGHNP), the Witsieshoek Community Conservation Area (WCCA) (not yet officially declared), and the Royal Natal National Park (RNNP) within the Maloti-Drakensberg Transfrontier Park World Heritage Site (Figure 3.1). The GGHNP is in the foothills of the Maluti Mountains situated in the eastern Free State Province of South Africa. The park was established in 1963 with the intention of protecting Clarens sandstone formations and montane Afro-Alpine grassland (SANParks 2004). The Witsieshoek Community Conservation Park is a newly established area in the foothills of Maluti Drakensberg Transfrontier Park (MDTP), prior to which it was communal rangeland.

The study was situated between the Free State and KwaZulu-Natal provinces of South Africa, where they are bounded by the Maloti-Drakensberg Mountains of Lesotho. QwaQwa (S 28^{o} 29’7.5’’, E 28^{o} 49’ 29.68’’) is under the Maluti-a-Phofung local municipality and falls within the authority of Thabo Mofutsanyana District Municipality (FSDP 2001). Phuthaditjhaba is the urban core of QwaQwa and is about 60 km away from the town of Harrismith and 30 km from Kestell. GGHNP (S 28^{o} 28’’ 47.7’’; E 28^{o} 45’ 33.4”) forms part of the greater Drakensberg-Maluti mountain system. GGHNP is under the authority of South African National Parks (SANParks) and is 19 km away from the town of Clarens.

Royal Natal National Park (RNNP) (S 28^{o} 42’ 18.56’’, E 28^{o} 55’12.88’’) is part of the OKhahlamba district municipality, approximately 40 km away from the rural town of Bergville. RNNP is under the authority of Ezemvelo KwaZulu-Natal Wildlife (EKZN), which is an organization responsible for conservation in the province. The geology, soils, climate, vegetation and fauna of these areas generally fall within the sub-humid, climatic climax grasslands of the Grassland Biome of southern Africa (Mentis and Huntley 1982; Rutherford and Westfall 1994).

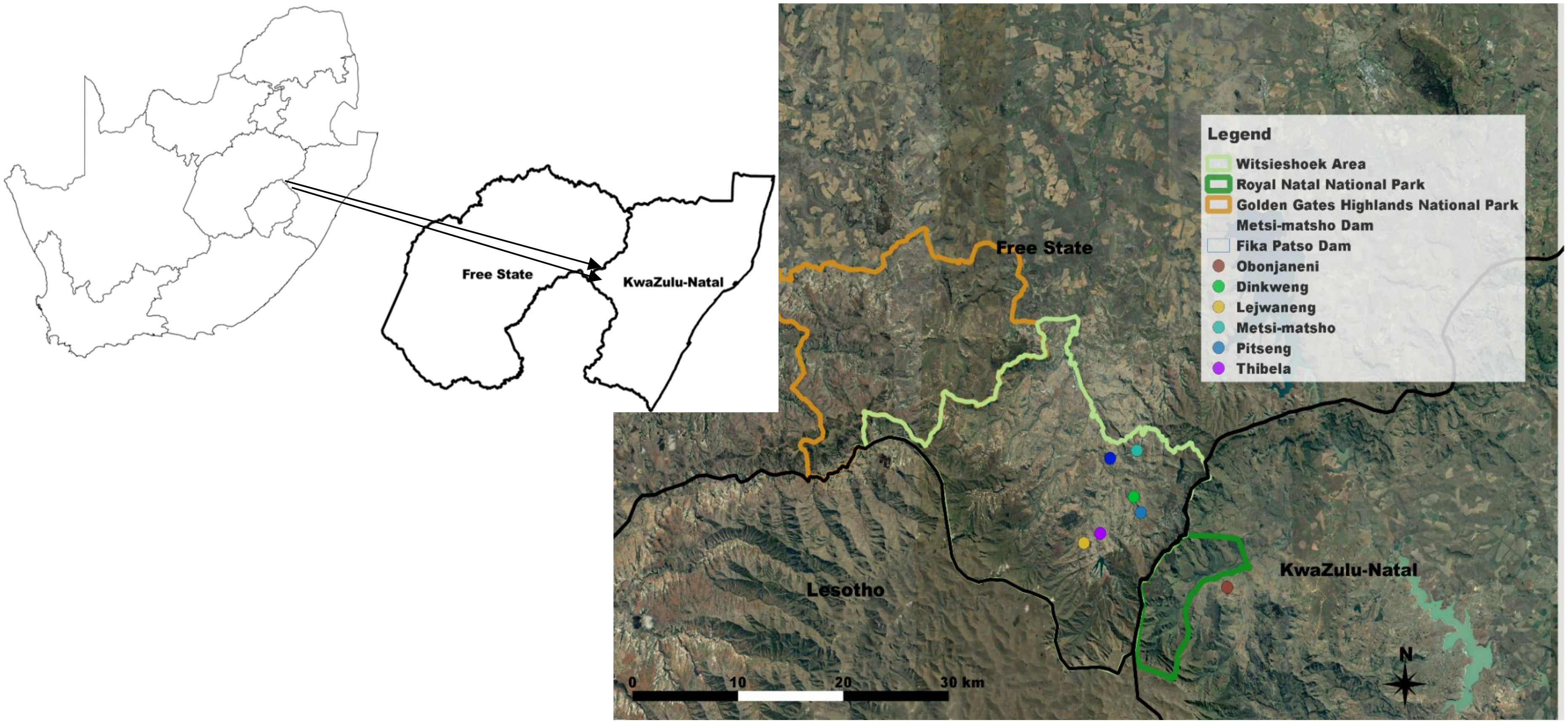

Figure 3.1. Location of the study area, with the different dot circle colours representing villages that were sampled and colour boundaries of protected areas (source: Google Earth 2018).

3.1.1 Climate

The climate of the eastern Free State is humid subtropical (Schulze 1947; Moeletsi et al. 2011). Humid subtropical climates are characterized by high relative humidity (Mousavi et al. 2015). The mountainous eastern Free State highlands are considered to have low air temperatures (Moeletsi and Walker 2012). Temperatures below zero are common in winter and spring mornings. Much of the rain falls in summer as convectional storms. Summers are wet and hot, winters are dry with regular frost. Snow falls occur annually between July and September; but low lying areas receive occasional events of snow. Wind also influences the climate of eastern Free State, with wind blowing from either west or east. The westerly winds are dry and warm, which sometimes reach gale force during the months of August to September and often lead to dust storms. Easterly winds can cause an extensive decrease in temperature especially along the escarpment and in high-lying areas (Roux 1986). The area experiences frost conditions which support the dominance of grassland vegetation and prevent the growth of trees (Low and Rebelo 1996). In lower lying areas, the mean annual precipitation is between 601-800 mm and rises to over 1000 mm in higher lying areas in a southerly direction (www.agis.agric.za/).

Weather data were obtained from the South African Weather Service (SAWS) for the closest weather stations. Temperature and rainfall data are from RNNP (1512 m), Bethlehem (1642 m) and Sterkfontein (1728 m) weather stations. In winter, the mean annual temperature is 4^{o}C and the maximum is 20^{o}C. In summer, the mean annual temperature is 14^{o}C and the maximum is 32^{o}C (https://malutiaphofung.fs.gov.za/). The warmest month was recorded in December, with an average maximum temperature of 26.4^{o}C. The coldest month was recorded in July, with an average minimum of -1.7^{o}C (Figure 3.2). The highest rainfall recorded was in January and the lowest in July (Figure 3.3).

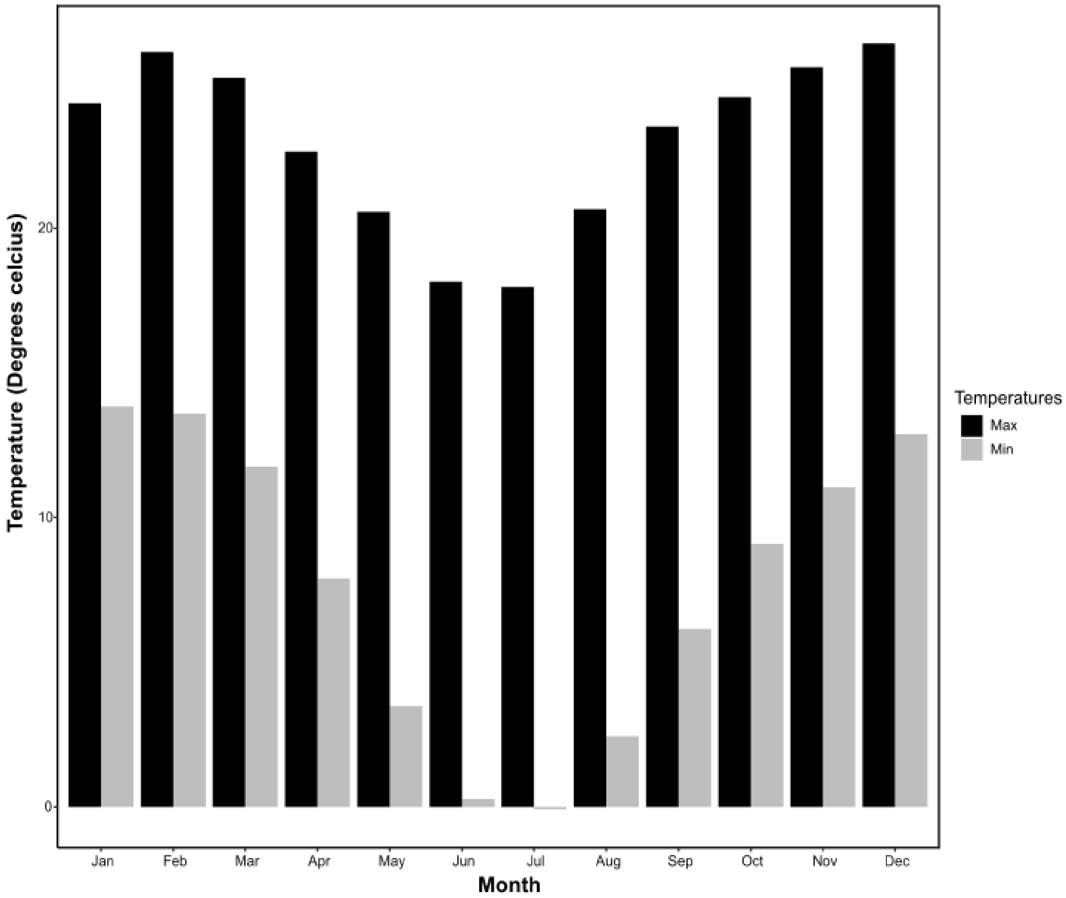

Figure 3.2. Average monthly minimum and maximum temperatures in the study area for the period January 2000 to December 2017 (Source: SAWS 2018).

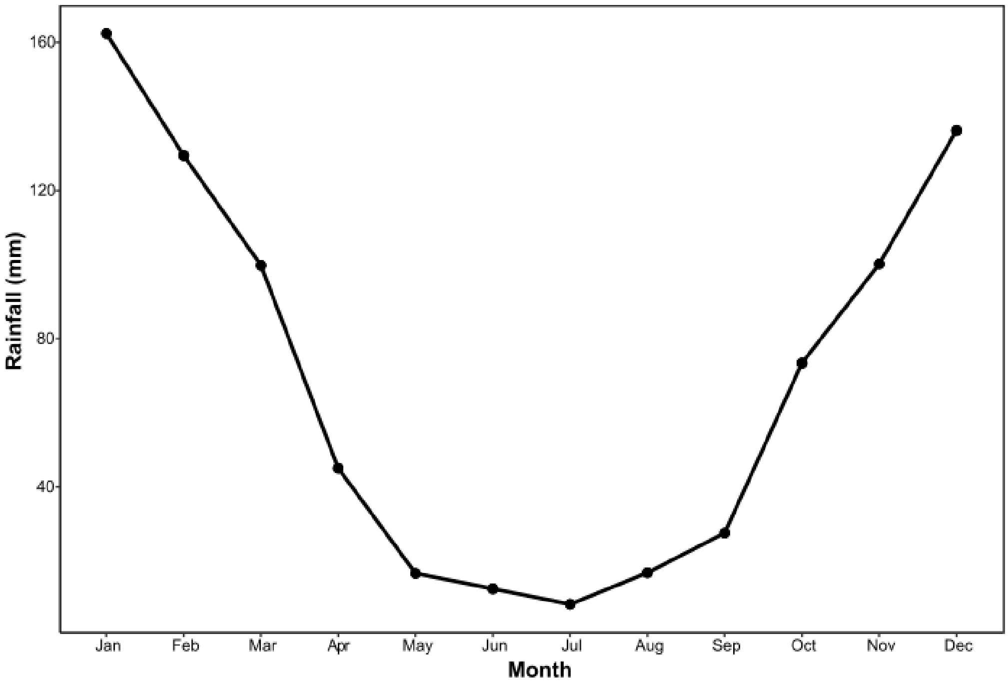

Figure 3.3. Mean monthly rainfall in the study area for the period January 2000 to December 2017 (Source: SAWS 2018).

3.1.2 Topography

The northern Drakensberg mountains form the watershed of the Orange and Vaal River systems to the south and west, and the Vaal and Tugela River systems to the east (Roux 1986), which are key systems for fresh water provision in South Africa. The study area has a well-defined drainage pattern and drains mainly in a northerly direction. The area falls within the catchments of Namahadi River, Wilge River, Klip River and Caledon River, with Fika Patso and Metsi-matsho being the only two water scheme dams. The topography is littered with drainage systems. Large portions of the area are not habitable and fall within the flood-line areas. The high Lesotho Highland in the southern boundary of the area has a definitive impact on both climate and drainage systems (Free State Department of Agriculture 1998).

3.1.3 Geology

The study area is underlain by sedimentary rocks of the Beaufort and Stormberg Groups (Karoo Super group) deposited between 250 and 180 million years ago during the Mesozoic era, onto which basalt lavas of the Drakensberg Group were extruded 180 million years ago (McCarthy and Rubidge 2013).

The Beaufort Group is the oldest surface geology in the study area, having been deposited in the greater Karoo basin around 250 million years ago at the start of the Mesozoic period. The Beaufort Group forms the foothills of the Drakensberg Escarpment and some portions of the eastern Free State plateau. This group consists of sedimentary rocks, mainly mudstone, sandstone, shale and quartzite.

Deposition of the Stormberg Group on top of the Beaufort group commenced about 220 million years ago. The Stormberg Group is divided into three sub-groups, namely the Molteno (Sandstone), Elliot (Red beds or Mudstones) and Clarens (Sandstone) Formations. These sedimentary rocks are more susceptible to weathering than the overlying basalt caps, which protect them (Whitmore et al. 2006).

The Drakensberg Group consists of basalt and dolerite that formed from an outpouring of magma through faults in the sedimentary deposits of the Karoo Supergroup that commenced about 180 million years ago. This outpouring of magma occurred as a sequence of flows to eventually deposit approximately 1.5 kms of basalt that is evident today as the summits of the Drakensberg Mountains and the cliffs of the escarpment. Basalts erode and weather more gradually than the underlying sedimentary rock types, which is evident in the escarpment cliffs overlying highly eroded hill slopes (Whitmore et al. 2006). Magma remaining in the faults cooled slowly to form the dolerite dykes and sills that can be seen to intersect the sedimentary Karoo Supergroup deposits throughout the study area. Therefore, a diverse geology of both igneous and sedimentary character was available for soil formation.

3.1.4 Soils

Soil formation is strongly influenced by the climatic factors of temperature and precipitation. These factors further influence soil moisture, thereby causing weathering of soil minerals and translocation of soil components within the soil profile. In addition, this causes leaching and decomposition of organic matter (Schimtz and Rooyani 1987). In addition, soil formation over time can also be influenced by topography, rock parent materials and time (Jenny 1941).

Soils in high rainfall areas become extremely weathered that result in poorly developed horizons. Because of extreme leaching these soils become acidic and nutrient poor soils. This strongly influences the vegetation dynamics that they support. By contrast, areas that experience low to intermediate rainfall have less weathered soils that are more eutrophic because of less leaching. These areas have fersiallitic, solonetzic and planosolic soils which tend to have well defined A and B horizons.

Rock weathering is a common feature in the area, which gives rise to formation of Orthic topsoil horizons. The predominant land types of the area are Ac land type which have red and yellow, apedal, well drained, dystrophic and mesotrophic soils. The predominant soil forms of the Ac land type are Hutton, Shortlands, Clovelly, Avalon, Westleigh and Katspruit forms (MacVicar 1991). The Bb land type has dystrophic and mesotrophic soils with a plinthic catena, margalithic and duplex soils, while red soils are uncommon (Smit 1992).

The soils are mostly thin on the summit and deeper on the foothills. In montane areas such as the current study area, shallow soils are evident because of low temperatures that limit weathering and erosion especially on steep slopes.

The other most common soil forms found in the study area are Glenrosa, with Hutton and Clovelly forms occurring mainly on sandstone plateaus (Groenewald 1989). The south-facing slopes have Bonheim form, and Mayo and Shortlands forms on the north-facing slopes (Groenewald 1989). The basalt slopes are considered to be very fertile for southern African (Staples and Hudson 1938). The soils in the study area are predominantly 10-90 cm deep.

3.2 Methods

3.2.1 Field vegetation sampling

3.2.1.1 Grazing gradient

The approach was to sample several transects representing grazing gradients from an origin (a kraal) to a distance beyond the limit of the average foraging distance of cattle. A total of 9 transects, comprising 90 plots were surveyed in the 2016/17 growing season. All transects were orientated with respect to topographic features, sampling mainly spurs and crests in order to sample a uniform physical environment. Sampling was restricted to grasslands avoiding all forests and bushy areas. All 90 plots (5 x 5 m) were placed along spurs and at as regular distances as possible to each other to capture the gradient. Every species within a plot was listed and its cover scored using the Domin scale (Jager and Looman 1987). Plant species that were unidentifiable in the field (outside protected areas) were collected, pressed, and later identified at the John Bews herbarium, Pietermaritzburg, and at the KwaZulu-Natal herbarium, Durban. Plant nomenclature follows that of SANBI's local database (http://redlist.sanbi.org/).

Indicators of grazing pressure measured were the number of dung pats (cattle dung), and the amounts of moribund material and grazing offtake from the sward, and the extent of soil erosion. The numbers of cattle dung pats were counted and grazing offtake (percentage loss of volume) was estimated in a 50 x 50 m plot (90) containing the vegetation plot. The amounts of moribund material (percentage of volume) and of erosion (cattle paths) were estimated in the 5 x 5 m plot. Visual estimations used a scale of very little (0-10%), moderate (10-30%), and abundant/heavy (30-50%).

Environmental data determined for each plot included location, altitude, aspect, slope, geology, soil type, land type, distance to settlement, distance to the closest kraal, and distance to the nearest perennial water. Location and altitude were measured using a Garmin Global Positioning System (GPS). Distances, aspect and slope were determined using the Geographic Information System (GIS) (ArcMap ESRI 2010) and a 50 m Digital Elevation Model (DEM) (U.S. Geological Survey 2017). Only the sandstone geology was considered for sampling.

3.2.1.2 Fence-line contrasts

The fence-line contrast between GGHNP (S 28^o 30' 5", E 28^o 34' 43") and QwaQwa (S 28^o 24' 21.8", E 28^o 57' 10.3") was sampled in March 2017. A second fence-line between Witsieshoek (S 28^o 41' 8.5", E 28^o 53' 56.8") and Royal Natal National Park (S 28^o 42' 18.56'', E 28^o 55'12.88'') was sampled in March 2018. Vegetation composition was recorded in 40 adjacent pairs of plots (5 x 5 m) along a 4 km fence-line for the GGHNP-QwaQwa contrast, and in 20 adjacent pairs for the RNNP-Witsieshoek contrast. The plots were placed 100 m apart as paired plots 10 m away from the fence to prevent trampling impacts by livestock. A visual estimation of percentage cover of each species on each plot was done using the Domin scale (Jager and Looman 1987). A Garmin GPS was used to record altitude in metres and to mark plots.

3.2.2 Field soil sampling along the distance gradient or across the fence-line contrasts

Soil environmental variables determined for each plot included levels of P, K, Ca, N, Mg, Zn, Mn, Cu, exchangeable acidity, acidity saturation, total cations, pH (KCl), organic carbon and percent clay. Soil form at each sampling location was identified by extracting a soil column (in the centre of the plot, on rocky plots it was moved slightly) using an auger. A column was carefully extracted depth by depth and laid out on plastic for identification of horizons from master through to subsoil horizons (Soil Classification Working Group and MacVicar 1991). Soil colour was determined using the Munsell Soil Colour book (Munsell colour charts 2009). Each soil horizon sample was sieved through a King Test sieve (2 mm aperture) in order to remove gravel and roots, packed, then sent to the Cedara Soils Analytical Services laboratory (Laboratories of Analytical Services, KwaZulu-Natal Department of Agriculture and Rural Development) for analysis of chemistry and texture (details in Appendix B).

3.2.3 Fire

Burnt area data sets covering a period of 17 years (2001-2017) were sourced from MODIS (MCD64A1) and used to map fire scars in the study area. The sourced data sets include a temporal imagery of 30 days and a spatial resolution of 500 m. The sensor records the date and location of each fire. Roy and Boschetti (2009) have shown that MODIS can identify atleast 75% of the burnt scars on an accuracy assessment done in South Africa.

A MCD64A1 burned area algorithm adopted from MODIS burned area product for collection 6 was used to detect fires that occurred monthly and gives them unique identities (Giglio et al. 2009). The MCD64A1 burned area product is a monthly Level 3 gridded 500 m product comprising per pixel burning and quality data. The hierarchical data format (hdf) images for a period of 17 years were extracted using the R MODISTSP package. The hdf images were projected and georectified and then converted to polygon shapefiles on QGIS 3.2.2 (QGIS Development Team 2010). Polygons that had matching fire scars were dissolved to produce a single fire record over a month and a year (Buthelezi et al. 2016). Lastly, fire frequency, seasonality and burnt area were analysed over a period of 17 years.

An annual burnt area data from 2001-2017 was produced by merging a monthly burned area dataset (Buthelezi et al. 2016). The seven months (May-November) selected reflected large fire frequencies as identified by MODIS images (Archibald et al. 2010; Buthelezi et al. 2016). An ANOVA was carried out to determine the differences among months and fire frequency per plot. The annual burnt area was overlaid with study area to calculate total burnt area. The annual burnt area layers were combined using a raster calculator. A histogram output was produced showing the distribution of burned area per month. The total area studied was 138557.51ha.

3.2.4 Vegetation response variables

3.2.4.1 Diversity

Diversity indices were used to assess whether there was a gradient in diversity with distance from kraals, and whether diversity differed across fence-line contrasts. A single measure for assessing community diversity is not feasible because no single index can adequately represent complex patterns of species richness and evenness (Hurlbert 1971; Peet 1974; DeJong 1975). Consequently, this study used a number of diversity measures, each emphasizing a different facet of diversity. Species richness (**S**) refers to the total number of species per plot. This measure accords equal weight to each species, is the least complex index, is independent of species abundance distribution, and is commonly considered the most reasonable measure of diversity (Chaneton and Facelli 1991). It is widely used in conservation assessments and monitoring (Olff and Ritchie 1998; Proulx and Mazumder 1998; Uys et al. 2004; Scott-Shaw and Morris 2015).

Species diversity is a measure of the structure and complexity of a community. It is influenced by the total number of species, their relative abundance (evenness), their spatial distribution (size of sampling unit), and sample size ('capture' of uncommon species in a sample). A derived index is therefore a composite measure of both species richness and evenness (Smith and Wilson 1996). The two indices of species diversity used were the Shannon-Wiener index (**H'**) and Simpson's index (**D′**), the index of evenness used was Pielou's index (**J′**). $\mathbf{H'} = -\sum_{i=1}^{s} \mathrm{p}i\ln_{\mathrm{p}i}$ and $\mathbf{D'} = \frac{1}{\sum_{i=1}^{s} \mathrm{p}i^{2}}$, whereby **s** is the number of species in a plot, and pi is the proportion of S contributed by the ith species. **H′** places greater emphasis on richness and rare species, whereas **D′** underscores evenness and the dominant species (McGarigal and Marks 1994; Haines-Young and Chopping 1996; Riitters et al. 2000). Pielou's index was calculated as $\mathbf{J'} = \frac{\mathrm{H'}}{\ln \mathrm{S}}$, with **H'** as defined above (Pielou 1969).

3.2.4.2 Plant groups

Plant community responses were conducted at the level of individual species and with species aggregated into groups using life and growth forms. Regarding plant groups, annual plants complete their life cycle within one year whereas perennial plants require more than one year.

A growth form is defined as a group of species that share the same habit of growth but are not necessarily related. Sedges and grasses were grouped as graminoids. Herbs refer to all herbaceous dicotyledonous species, non-graminoids monocotyledons, and geophytes. Herbs are therefore seed-bearing plants lacking a woody stem. Geophytes are species which possess belowground storage organs (bulb, tuber, or rhizome) and spend part of the annual cycle without living aboveground parts. Shrubs are species with persistent woody stems, grow >0.5 m in height, and lack succulent leaves or stems. Dwarf shrubs are woody shrub species usually less than 0.5 m in height. In summary, the following groups were distinguished: herbs (annuals, biennials and perennials); monocotyledons (grasses, indigenous and sedges); dicotyledons (aliens, geophytes, shrubs, dwarf shrubs and endemics).

3.2.5 Data analysis

3.2.5.1 Grazing gradient

Linear or non-linear relationships were sought between a numbers of variables with distance from a kraal site. If the best fit was linear, it was fitted using linear regression. If the relationship of any variable with distance was non-linear, the form of relationship was identified using the multiple non-linear regression routine of R (R, Version 3.5.1; R Core Team 2018, R package easy Nonlinear Model 5.0). The first objective was to determine whether distance could serve as a surrogate for long-term grazing pressure, which was expected to decline with distance. Relevant variables included cattle dung, moribund grass, and the amount of biomass recently removed by grazing. A second objective was to determine if any such relation might be confounded with a spatial pattern in soil properties or other environmental variables, specifically altitude, solar radiation or slope. A third objective was to determine whether vegetation variables showed a relation with distance, these variables being diversity indices, and the abundance of plant groups or of individual species.

3.2.6 Pattern of soil variation with distance and along fence line contrasts

Principal Component Analysis (PCA) was undertaken to assess relations among the soil environmental variables and to display variation in the soil environment across sites (Ludwig et al. 1988). The method is advantageous because it is able to minimize dimensionality (Manly 1994). The soil environmental variables were centered and standardized because they were measured in different scales (R Version 3.5.1; R Core Team 2018, R package Vegan 2.5-2). In order to determine if the pattern of variation in the soils was related to distance from a kraal site, site scores for each of the first three PCA axes were related to distance. A significant relation would indicate that soil gradients are confounded with distance from kraal sites.

In order to determine if the soils of communal and protected area were different across a fence-line contrast, separate paired *t*-tests using the PCA site scores of the first three axes were conducted. After careful inspection, five plots from a grazing gradient were left out from the analysis because there were either on dolerite dykes or old fire breaks trenches which were impossible to see in the field because of dense vegetation cover.

3.2.7 Species community response to distance

Species composition was analyzed using an ordination method, Canonical Correspondence Analysis (CCA) using R (Version 3.5.1; R Core Team 2018, R package Vegan 2.5-2). It is a direct gradient analysis approach that is able to directly relate a fixed set of environmental variables to a set of species (Ter Braak 1986). It is also able to identify the forms of change in community structure that can clearly be described adequately with environmental variables (Ter Braak 1986). CCA analysis was undertaken to determine if distance from a kraal site influenced composition, as expected. Species recorded in ten or more plots were included in the analysis.

An ordination biplot shows species and plots as points and environmental variables as arrows. The diagram also indicates the primary forms of change in community structure that can best be described by the supplied environmental variables (Ter Braak 1986). It also shows the patterns of species on each environmental variable. This method assumes that species have a Gaussian distribution response to multiple environmental variables (Ter Braak 1986).

3.2.8 Fence-line contrasts

Paired two sample *t*-tests (two tailed) were done at the 0.05 significance level to compare Shannon’s diversity index, Simpson’s diversity index, Pielou’s evenness index and species richness between protected and communal areas for each of the contrasts. Paired *t*-tests are normally conducted when there are two sets of data from matched sites or samples (Howell 1997). Paired *t*-tests were executed to assess whether there was any true difference that is due to the effect of the two land uses. Paired *t*-tests were also used to determine whether the aerial cover of each plant group was different between protected and communal areas for each of the two contrasts. Paired *t*-tests were also applied on PCA site scores and on individual soil variables to determine whether grazing had any effect on soil environmental variables.

Chapter 4: Results

4.1 Grazing intensity gradient

4.1.1 Distance as a proxy for grazing impact

The amount of cattle dung decreased with distance from a kraal site according to a negative power relationship (Figure 4.1). Cattle dung was the only strong indicator of a gradient in grazing with distance. Other indicators of grazing such as moribund grass and amount of biomass recently removed by grazing did not show any relation (*P*>0.05) with distance from kraal sites. It was further confirmed that distance was not confounded with certain physical environmental variables (*P*>0.05), specifically altitude, solar radiation, or slope.

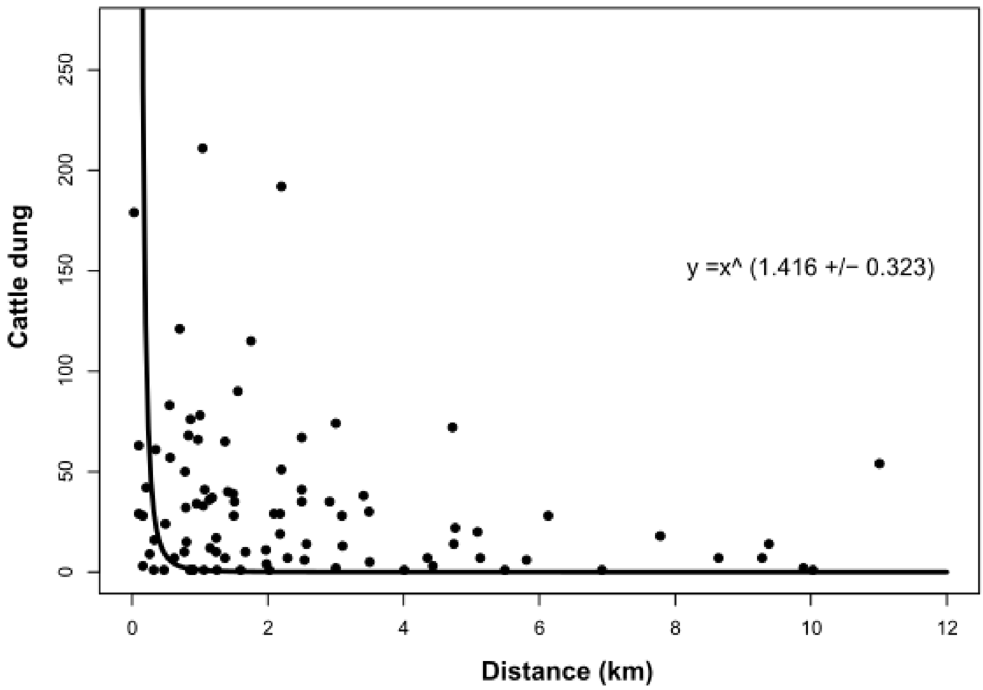

Figure 4.1. Cattle dung (number of pats in a 50 by 50 m quadrat) in relation to distance from kraal sites (Mean Square Error = 1.416±0.323).

4.1.2 Pattern of soil variation with distance

The first two axes explained 38.7% and 19.1% of the total variance of 14 soil variables (Table 4.1). The first axis described a gradient from soils with relatively high total cations dominated by calcium and magnesium, with associated high nitrogen concentration and clay amount, through to soils with high acidity saturation (Figure 4.2;

Table ***4.2***). The second axis describes a gradient from soils with relatively high exchangeable acidity, with associated high phosphorus concentration, through to soils with a high pH (Figure 4.2;

Table ***4.2***). The third axis, which explained a further 12.3% of the variance (Table 4.1), describes a gradient from zinc rich sites which were positively correlated to potassium and manganese through to sites with the converse (Figure 4.2; Table **2.1**). There was no relation between distance and the PCA site scores of either axis 1(r= 0.1440, P= 0.1885), axis 2 (r=-0.1566; P= 0.1523), or axis 3 (r= -0.0116, P= 0.9158), indicating that the grazing gradient was not confounded with a gradient in soil physico-chemical properties.

Table 4.1. Summary of PCA output of soil environmental variables along a grazing gradient.

	PCA 1	PCA 2	PCA 3
Eigen values	5.4189	2.6709	1.7217
Proportion Explained	0.3871	0.1908	0.1230
Cumulative Proportion	0.3871	0.5778	0.7008

Table 4.2. Summary of the loading of each soil environmental variable on each of the first three axes of a PCA describing a grazing gradient (n=85).

Variables	PCA 1	PCA 2	PCA 3
P	-0.2072	-0.8851	0.6347
K	0.7636	0.2392	0.7317
Ca	1.3340	0.3455	-0.3988
Mg	1.3224	0.0601	-0.4835
Cu	0.9617	-0.3626	0.4348
N	1.0769	-0.8610	-0.2272
C	1.0185	-0.9977	-0.1794
pH (KCl)	0.6295	0.7857	0.0524
Zn	0.5022	0.1292	1.1592
Mn	0.8922	-0.1180	1.0105
Clay	1.1116	-0.5537	-0.1614
Total cation	1.4169	0.0061	-0.4508
Acidity saturation	-1.0663	-0.8986	-0.0026
Exchangeable acidity	-0.4156	-1.3957	-0.1258

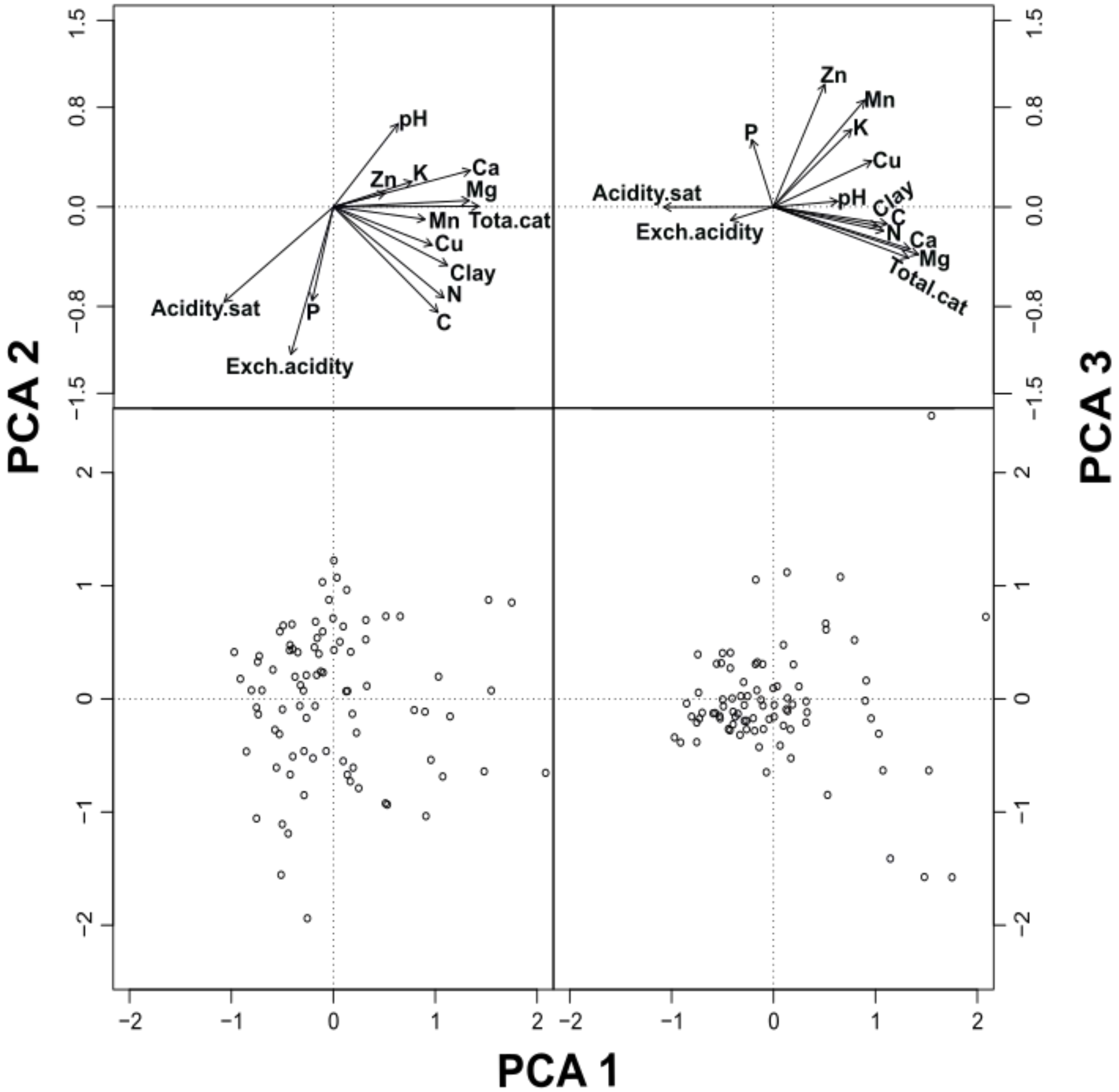

Figure 4.2. Principal component analysis of soil environmental variables along a grazing gradient (n= 85). Abbrev: Acidity.sat. = Acidity saturation, Exch.acidity = Exchangeable acidity, Total.cat. = Total cation, C= Carbon, N = Nitrogen, P = Phosphorus, Zn = Zinc; Mg = Magnesium, Mn = Manganese, Ca = Calcium, K =Potassium, Cu = Copper, pH = pH (KCl), Clay = Clay. The open circles represent plots.

4.2 Fire

Fire frequency per plot differed over the 17 years ($F_{(4,145)}$ =7.349; P < 0.00204). The fire return interval ranged from short (1-2 years), through moderate (3-5 years), long (6-9 years), very infrequent (10-11 years), to none (0 in 17 years) (Figure 4.3). In general, there had been no fires < 300 m from a kraal over the period of record, an area which contained 22% of the plots, ostensibly because this area was heavily grazed. A short return fire interval (1-2 years) accounted for 24.7% of the plots. Fire-return intervals of 3-5, 6-9, and 10-11 years accounted for 33.3%, 16.7% and 3.3% of the plots, respectively (Figure 4.3). These plots were generally situated far away from the kraals. In conclusion, fire frequency was confounded with distance along the grazing gradient which was considered to be an unavoidable outcome owing to the influence of livestock grazing. However, the potential influence of this confounding factor was lessened by the relatively small proportion (5.9%) of the study area which burned each year.

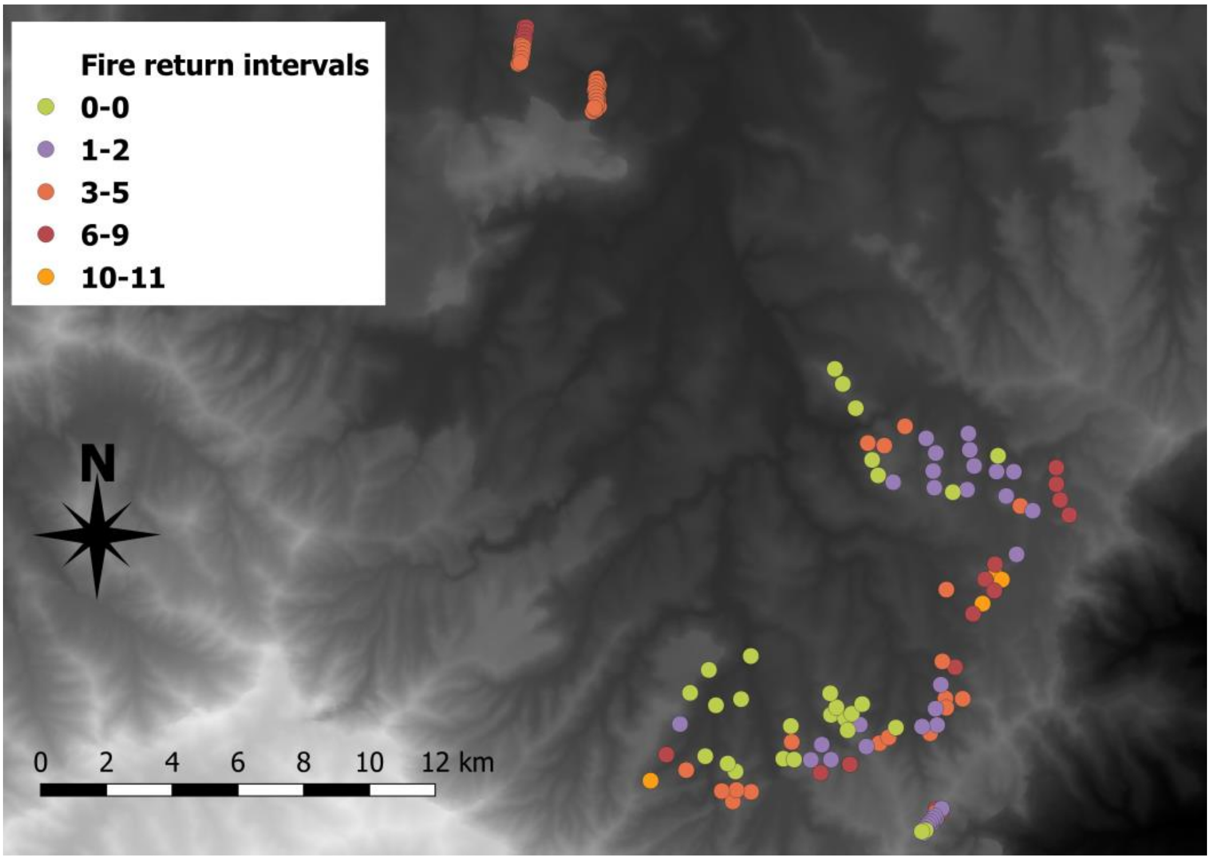

Figure 4.3. Fire frequency per plot in years from 2001-2017 in the study area, based on MODIS imagery.

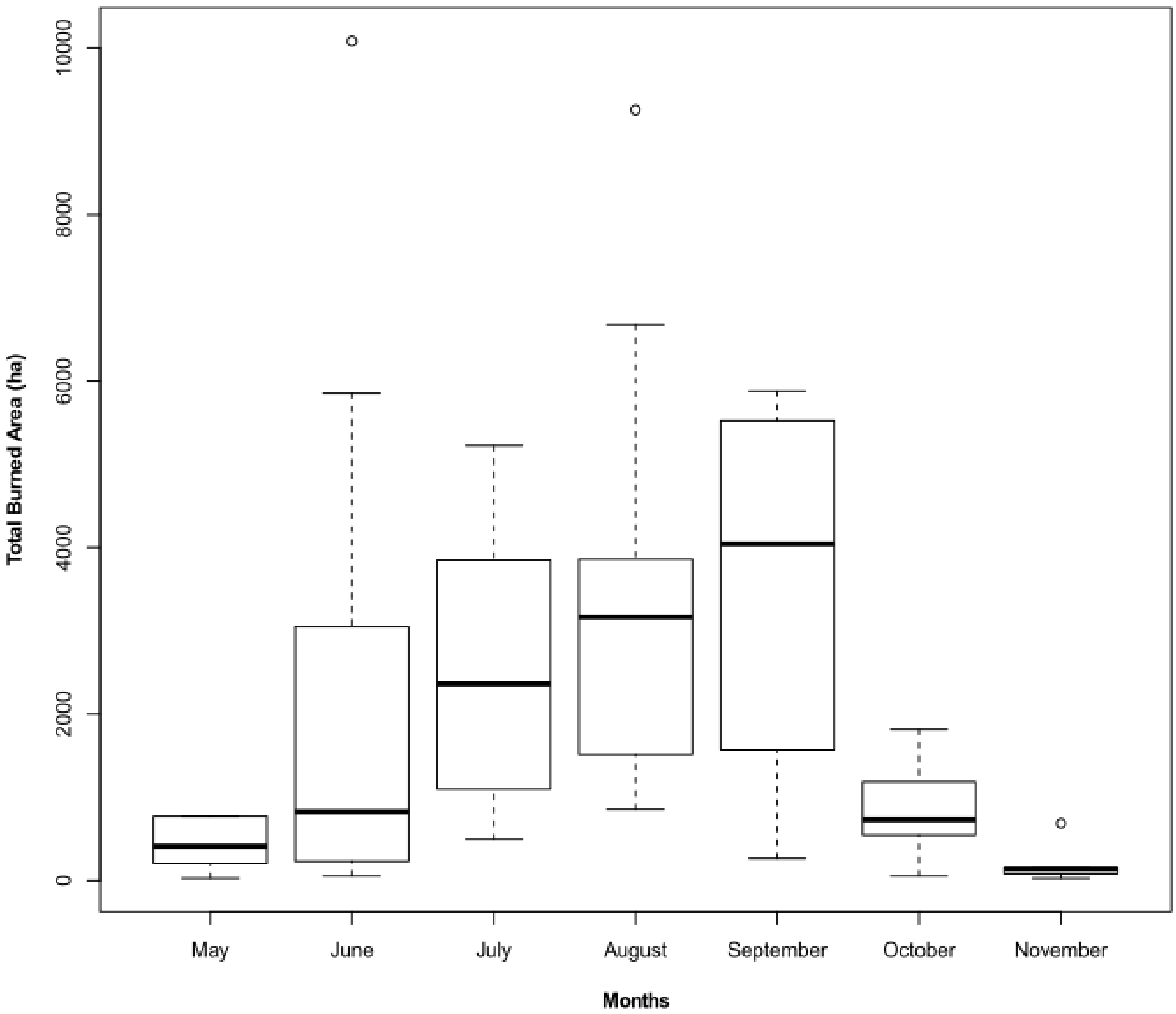

Figure 4.4. Average of burnt area per month from May to November from 2001-2017. The horizontal lines within the bigger box represent the mean, the boxes represent (mean ± SE) and the whisker plots represent (mean ±SD). Outliers are shown as small circles.

The area burned per month differed among months for the seven months from May to November ($F_{(6, 85)}$ =6.73; $P < 0.00072$). Only a small area burnt at the beginning of the dry season in May, increasing consistently to a peak in September, before decreasing markedly by the end of the dry season in November (Figure 4.4 and Figure 4.5). However, variability among years in the total area burned was high for July to September inclusive (Figure 4.4). The proportion of the study area burnt each year ranged from a minimum of 1.6% in 2001 to a maximum of 9.2% in 2007, with an average of 5.9% burnt per annum for the 17 years of record (Figure 4.6). In correspondence, fire season varied from three to seven months in any particular year (Table 4.3).

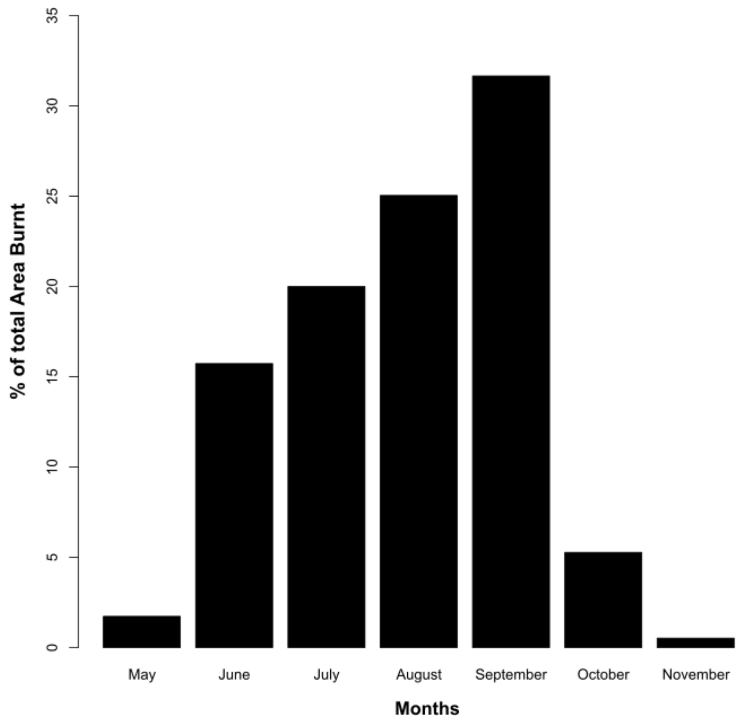

Figure 4.5. Percentage of the total area burnt on average per month from May to November (2011-2017).

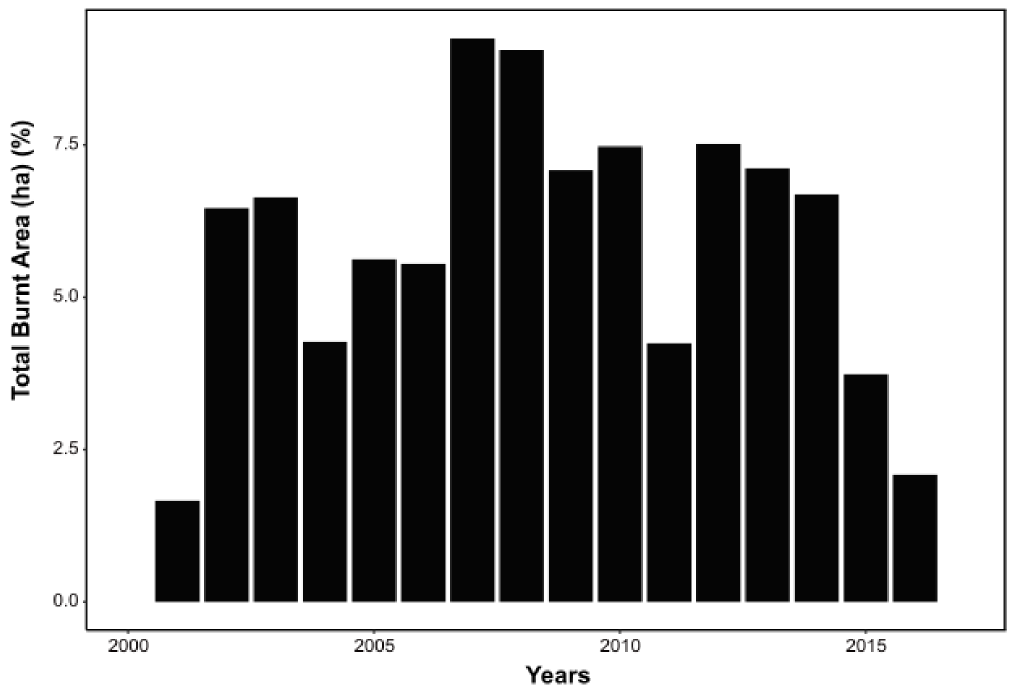

Figure 4.6. Percentage of the study area burnt each year from 2001-2017.

Table 4.3. Fire seasonal changes, emphasizing the month of maximum fire frequency, extent of the fire period (seasonality) and months with minimum burnt area.

Year	Length of fire season	Months of small to no burnt area	Month of maximum fire frequency
2001	3 months	-	September
2002	4 months	-	July
2003	6 months	November	September
2004	5 months	-	September
2005	4 months	-	August
2006	6 months	October	September
2007	5 months	-	June
2008	5 months	-	August
2009	6 months	May	September
2010	6 months	May	July and September
2011	3 months	May, June and October	August
2012	7 months	November	September
2013	7 months	November	September
2014	6 months	May	June
2015	7 months	November	August
2016	5 months	June	August
2017	7 months	May	September

4.3 Species richness and diversity

A total of 303 vascular plant species were recorded. In terms of South African or Drakensberg endemics, a total of 12 species were recorded. Species richness increased in a linear fashion with distance from kraal sites at a rate of 25 species per kilometer (Figure 4.7c). Shannon diversity also increased linearly with distance (Figure 4.7a; Table 4.4). Pielou’s index also increased linearly with distance (Figure 4.7d; Table 4.4), whereas Simpson’s diversity index tended to do so (Figure 4.7b, Table 4.4). Patterns of diversity with distance were therefore primarily related to species richness. A total of 50 plant families were recorded. The dominant families in terms of richness were Asteraceae, Poaceae, Cyperaceae and Fabaceae (Figure 4.8). These findings indicate that an increase in grazing intensity had resulted in a minimal decrease (0-2.5 km) in plant species richness and in plant diversity.

Table 4.4. Indices of diversity in relation to distance from kraal sites, n=90.

Variables		Shannon H' (LM)			Pielou's D' (LM)			Simpson's D′ (LM)	
	Estimate	**Std. Error**	**P**	**Estimate**	**Std. Error**	**P**	**Estimate**	**Std. Error**	**P**
intercept	2.5542	0.0646	<2E-16 ***	0.7683	0.0156	<2E-16***	0.8667	0.0097	<2E-16 ***
Distance (km)	0.0435	0.0172	0.0135 *	0.0087	0.0041	0.039 *	0.0050	0.0026	0.0545
R^2		0.0673			0.0475			0.0413	
Df		89			89			89	
Note							***p<0.001; **p<0.01; *p<0.05; .p<0.1		

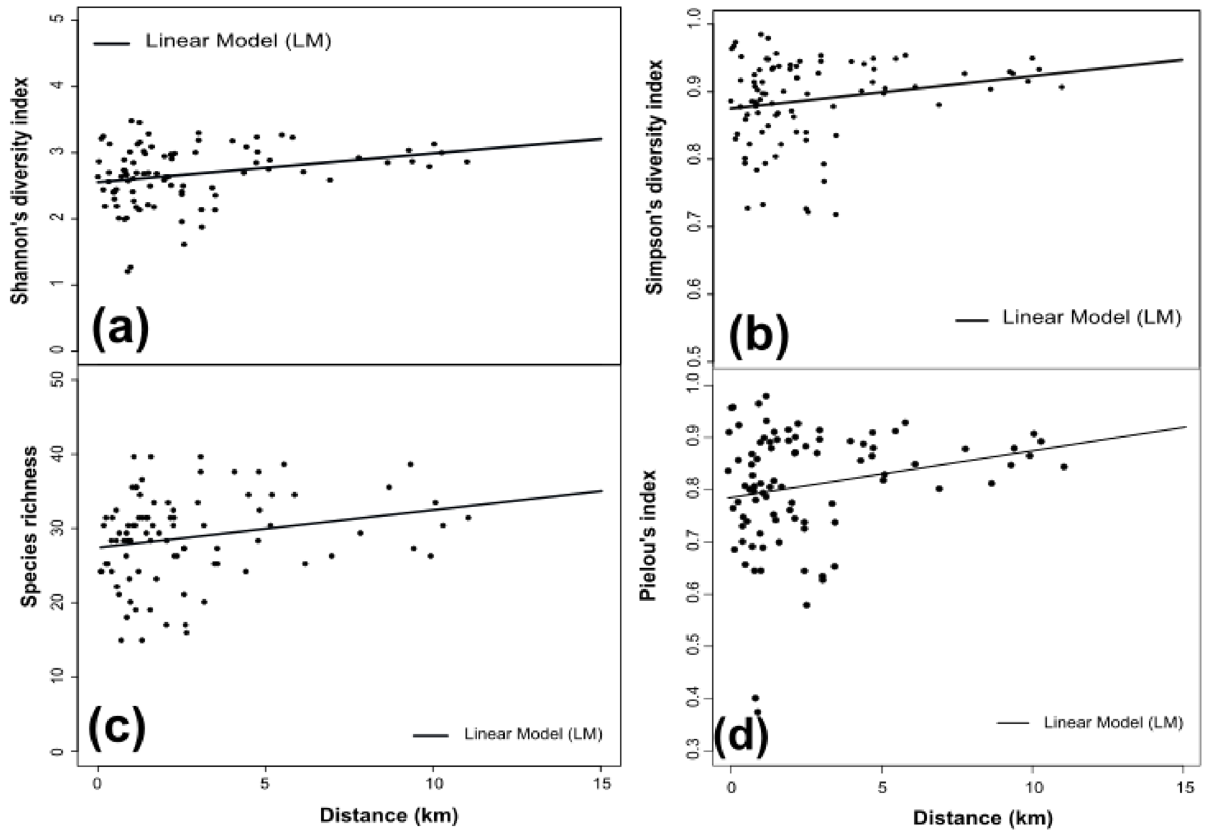

Figure 4.7. Indices of diversity in relation to distance from kraals along a grazing gradient (n=89). (a) Shannon's diversity index; (b) Simpson's diversity index; (c) Species richness and (d) Pielou's index.

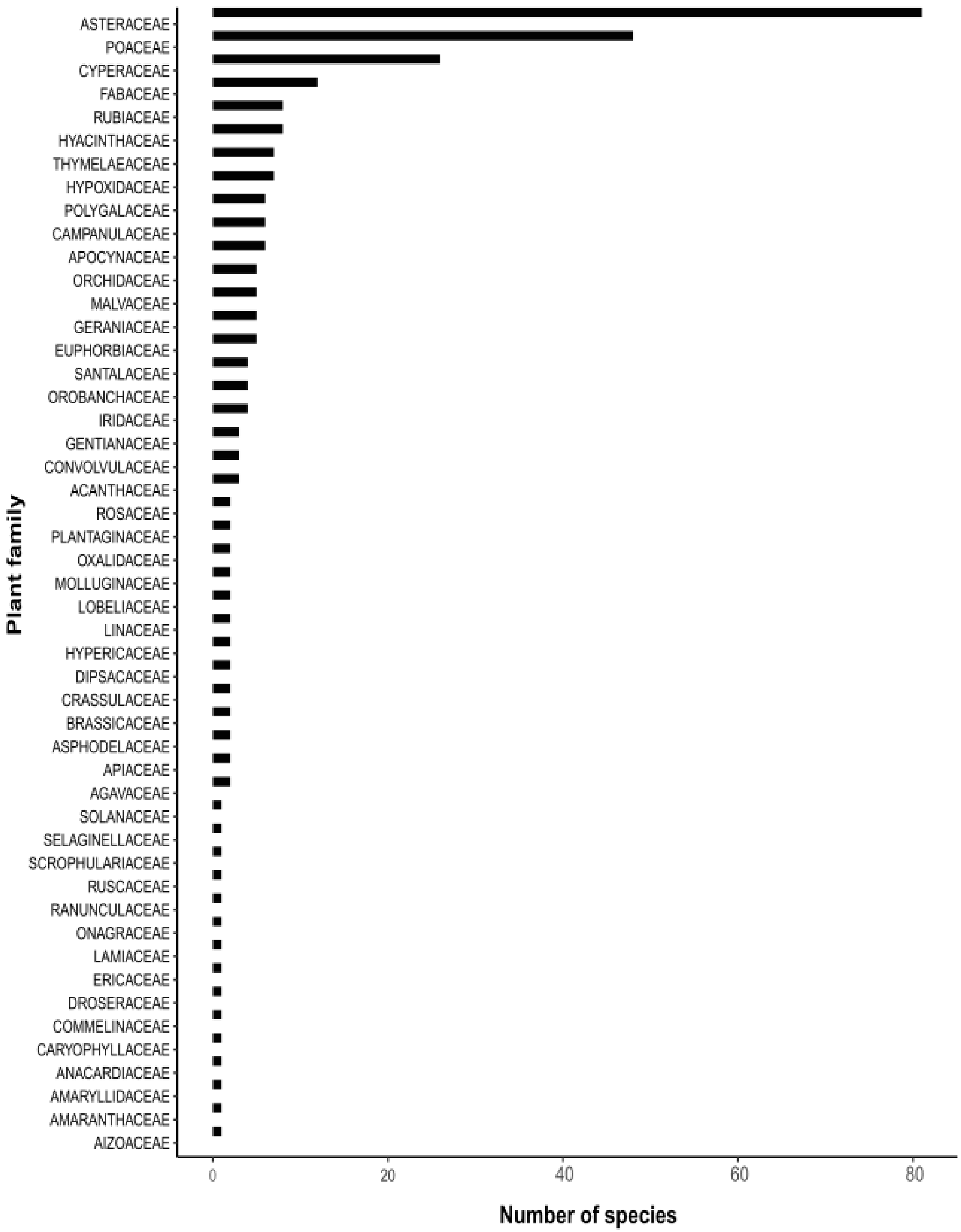

Figure 4.8. Contribution of each plant family to the total number of flowering plant species along a grazing gradient (n= 303 species).

4.4 Species community response to distance

A canonical correspondence analysis (CCA) model of species cover abundance revealed a significant (F= 2.0936, P< 0.01) relationship with distance from kraal sites (

Figure **4.9**). The first axis of CCA explained 9.3 % of compositional variation (to other environmental factors.

Table 4.5), indicating that most compositional variation was related to other environmental factors.

Table 4.5. Summary output of species community responses under a Canonical Correspondence Analysis (CCA) model from kraal sites, (n=90).

	CCA 1	**CCA 2**	**CCA 3**
Eigenvalues	0.2280	0.1773	0.1632
Species environmental correlations	0.5983		
Proportion explained	0.0702	0.0545	0.0502
Cumulative explained	0.0934	0.1480	0.1982
Canonical coefficient : Distance	0.3515		

The first axis described a gradient from grazing sites with *Bulbostylis boeckeleriana*, *Craterocapsa tarsodes*, *Sebaea natalensis*, *Helichrysum appendiculatus* and *Alloteropsis semialata* (Group 2;

Figure **4.9**) found further away from kraal sites, whereas *Acalypha punctata*, *Digitaria monodactyla*, *Gnidia kraussiana*, *Themeda triandra*, *Scabiosa columbaria*, *Monsonia attenuata* and *Microchloa caffra* were closer to kraal sites (Group 5;

Figure **4.9**). The species that were found at intermediate distance from the kraal sites along axis 1 were *Ajuga ophrydis*, *Aristea woodii* and *Heteropogon contortus*. After distance had been taken into account, the second axis was defined by (Group 1) species in response to an unidentified environmental gradient. In conclusion, herbaceous composition therefore changed significantly with increasing distance indicating an effect of grazing intensity. However, most compositional variation remains unexplained.

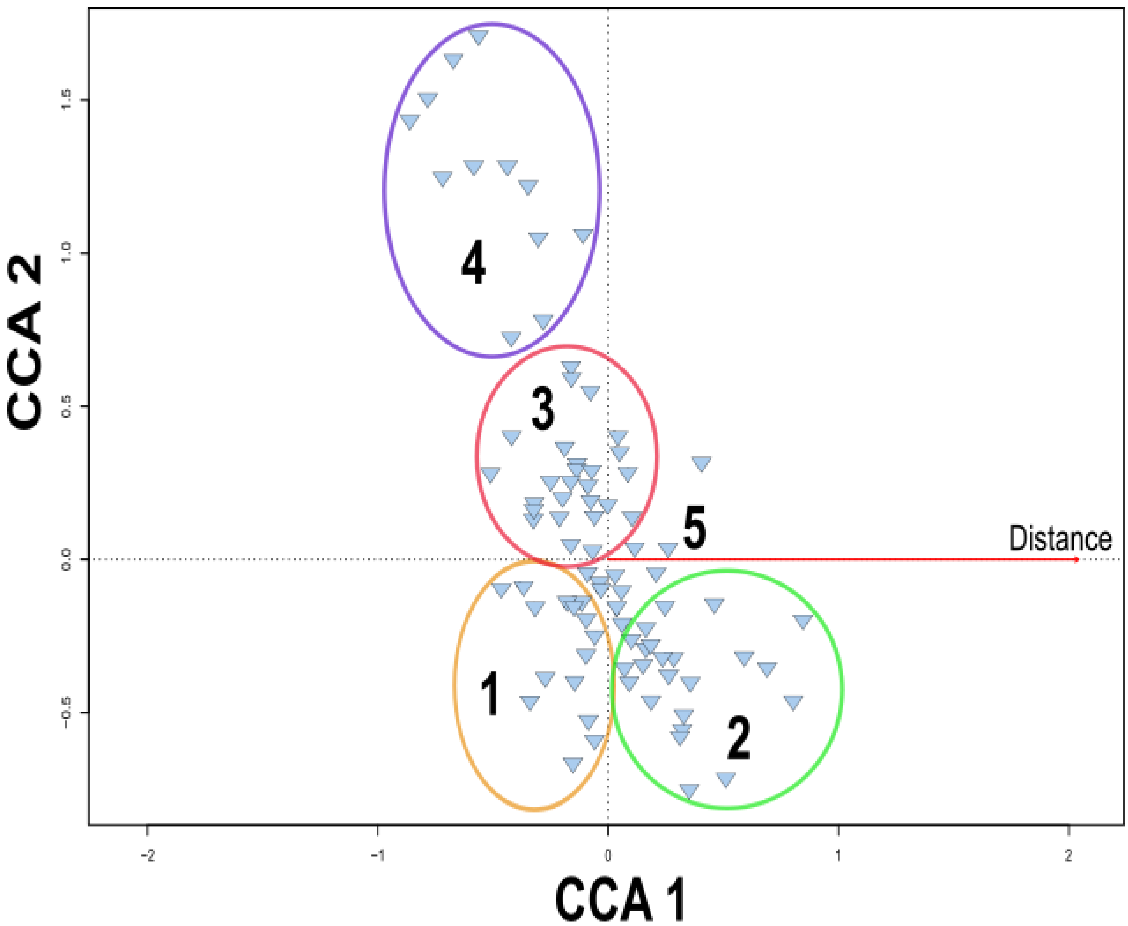

Figure 4.9. Canonical correspondence analysis output model of species cover abundance along a grazing gradient from kraal sites. Key to species: (Group 1) *Paspalum natalensis*, *Helichrysum spiralepis*, *Helichrysum chionosphaerum*, *Hypoxis accuminata*, *Senecio rhomboideus*, *Bulbostylis humilis*, *Berkheya rhapontica*, *Senecio glaberrimus*, *Tristachya leucothrix*, *Aristea cognata*, *Brunsvigia radulosa*, *Bulbostylis scleropus*, *Vernonia natalensis*, *Dicoma anomala*, *Cyperus rupestris*; (Group 2) *Gnidia kraussiana* , *Diheteropogon filifolius*, *Sebaea sedoides*, *Heteropogon contortus*, *Polygala ohlendorfiana*, *Ficinia stolonifera*, *Sebaea natalensis*, *Cyperus semitrifidus*, *Thesium pallidum*, *Harpochloa falx*, *Acalypha punctata*, *Haplocarpha scaposa*, *Ajuga ophrydis*, *Trachypogon spicatus*, *Monocymbium ceresiforme*, *Craterocapsa tarsodes*, *Andropogon distachyos*, *Bulbostylis boeckeleriana*, *Bulbostylis oritrephes*, *Helichrysum appendiculatus*, *Watsonia lepida*, *Alloteropsis semialata*, *Hirpicium armeriodes*; (Group 3) *Anthospermum rigidum*, *Helichrysum oreophilum*, *Senecio coronatus*, *Oxalis obliquifolia*, *Stiburus alocoperoides*, *Elionurus muticus*, *Helichrysum pallidum*, *Helichrysum herbaceum*, *Helichrysum miconifolium*, *Gnidia baurii*, *Anthospermum herbaceum*, *Tolpis capensis*, *Hypoxis anguistifolia*, *Felicia muricata*; (Group 4) *Richardia brasiliensis*, *Hypochaeris radicata*, *Lepidium africanum*, *Eragrostis chloromelas*, *Helichrysum callicomum*, *Rendlia altera*, *Monopsis decipiens*, *Crabbea acaulis*, *Eragrostis capensis*, *Sonchus nannus*, *Helichrysum dasycephalum*; (Group 5) *Euphorbia striata*, *Ledebouria sp*., *Helichrysum adenocarpum*, *Themeda triandra*, *Aristea woodii*, *Digitaria monodactyla*, *Crassula lanceolata*, *Monsonia attenuata*, *Microchloa caffra*, *Scabiosa columbaria*, *Linum sp*., *Eragrostis racemosa*, *Helichrysum pilosellum*.

4.5 Response of plant groups and species to distance from kraals

An impact of grazing intensity, as reflected by distance from a kraal, was evident for some plant groups; with increasing distance from kraals there was an increase in the cover of indigenous plants (Figure 4.10a; Table 4.6), graminoid plants (Figure 4.10b; Table 4.6) and of perennials (Figure 4.10e; Table 4.6) and a decrease in the cover of alien plants (Figure 4.10c). It was further confirmed that the plant groups of annuals, perennials, or geophytes, did not show a relation with distance from kraal sites ($P>0.05$). However, the cover of geophytes increased with increasing altitude (Figure 4.10d).

Table 4.6. Summary of linear models in plant groups in relation to distance away from the kraals.

Variables	Indigenous (LM)			Graminoids (LM)			Perennials (LM)		
	Estimate	**Std. Error**	**P**	**Estimate**	**Std. Error**	**P**	**Estimate**	**Std. Error**	**P**
intercept	115.071	5.767	<2e-16 ***	74.611	5.544	< 2E-16 ***	4.7381	0.0361	< 2E-16 ***
Distance (km)	3.707	1.533	0.0177 *	3.916	1.482	0.0097 **	0.0787	0.0297	0.00957 **
R^2		0.0629			0.0735			0.0738	
Df		88			88			88	
Note							*****p<0.001; **p<0.01; *p<0.05; .p<0.1**		

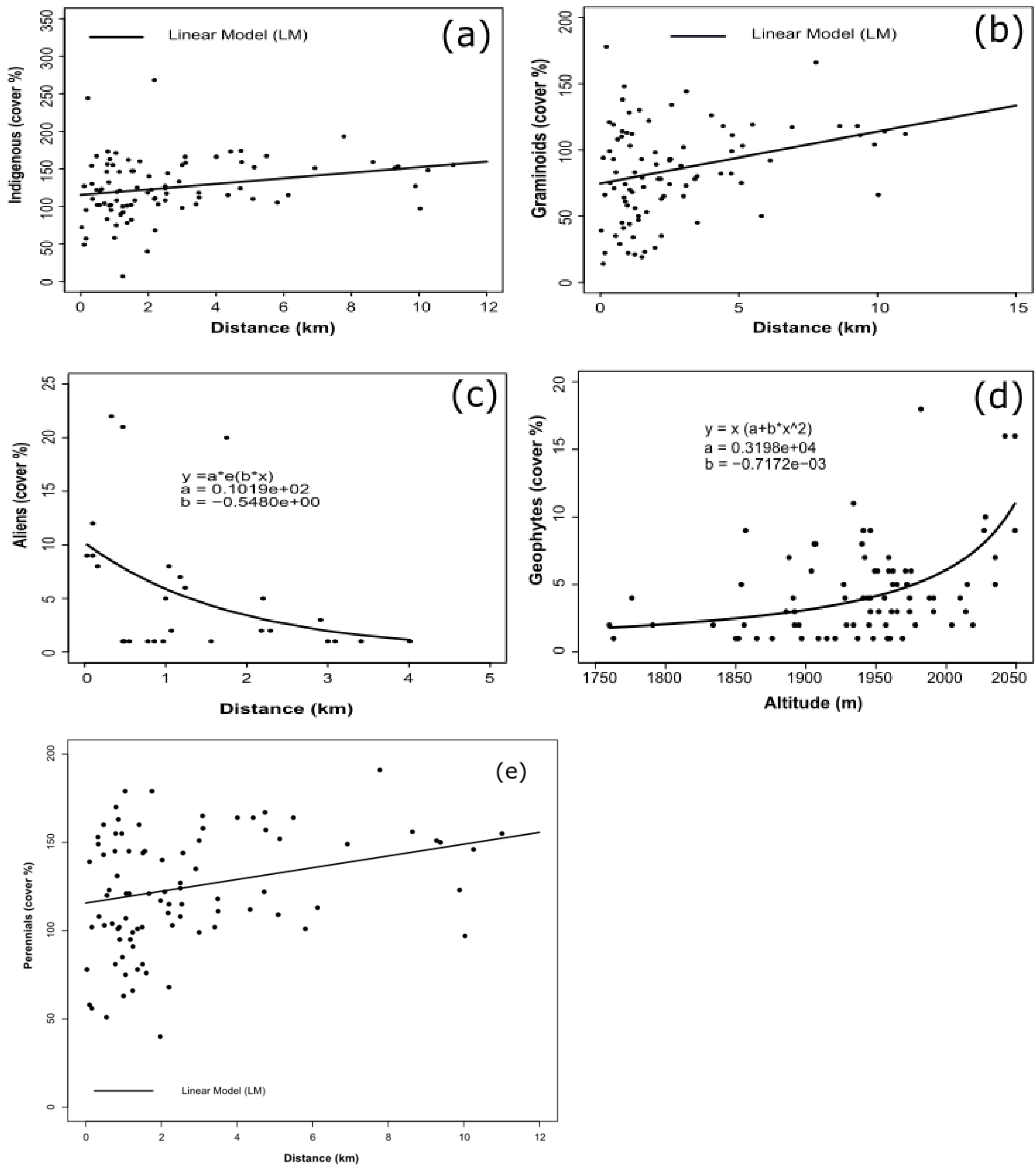

Figure 4.10. Aerial cover of plant groups in relation to distance away from kraal sites. (a) Indigenous; (b) Graminoids; (c) Aliens; (d) Geophytes and (e) Perennials.

Only one species, *Harpochloa falx*, showed a relation to distance from a kraal site according to non-linear exponential function (Figure 4.11). Other common plant species including *Eragrostis racemosa*, *Oxalis obliquifolia*, *Senecio glaberrimus*, *Tristachya* *leucothrix*, *Trachypogon spicatus*, *Aristida junciformis* and *Themeda triandra* did not show any relation with distance (*P*>0.05) from kraal sites.

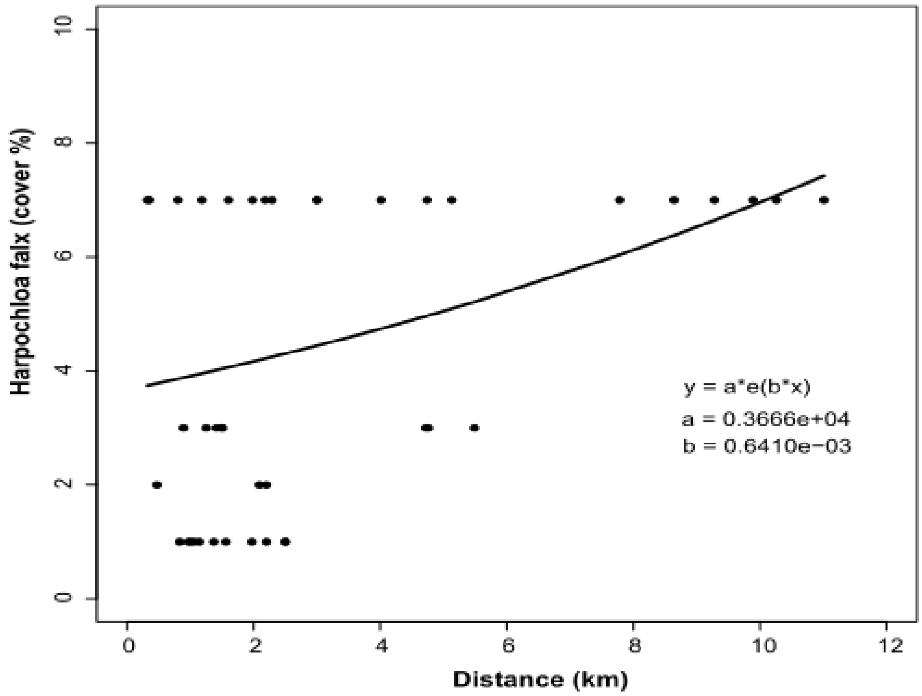

Figure 4.11. *Harpochloa falx* in relation to distance away from the kraals.

4.6 Fence-line contrast

4.6.1 Golden Gate Highlands National Park and QwaQwa communal area: Species richness and diversity

A total number of 173 vascular plant species was recorded, with the protected area having 132 species and the communal area 127 species. Only 86 species occurred on both sides of the fence, 26 of which were Poaceae, and 87 species occurred on only one side. In terms of South African or Drakensberg endemics, a total of eight species (*Anthospermum streyi*, *Aster pleiocephalus*, *Felicia fascicularis*, *Hypericum aethiopicum Thunb.subsp. Aethiopicum*, *Nolletia rarifolia*, *Selago monticola*, *Solanum capense* and *Thunbergia vernosa*) were recorded, two of which occurred on both sides of the fence. A total of 38 families were recorded, with 31 in the protected area (Figure 4.12) and 34 in the communal area (Figure 4.13), of which 27 families were shared. The dominant families in terms of number of species were Asteraceae, Poaceae, Cyperaceae and Fabaceae. There was a significant difference (t= 4.7299; *P*=0.00014 df= 19; GGHNP = 60.05±5.177; QwaQwa =32.25 ± 1.9123) in aerial cover of Poaceae family between the two rangelands.

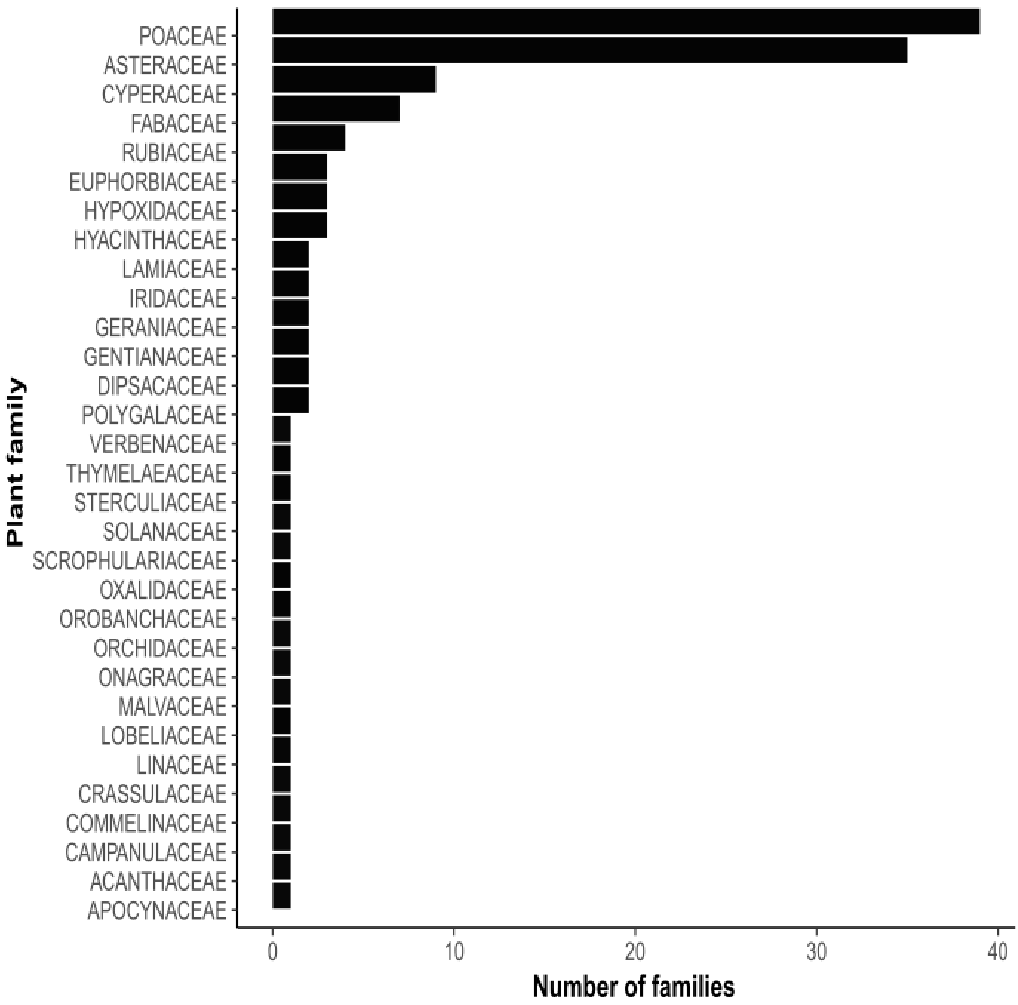

Figure 4.12. Contribution of each plant family to the total number of flowering plant species recorded in the Golden Gate Highlands National Park area (n= 132 species).

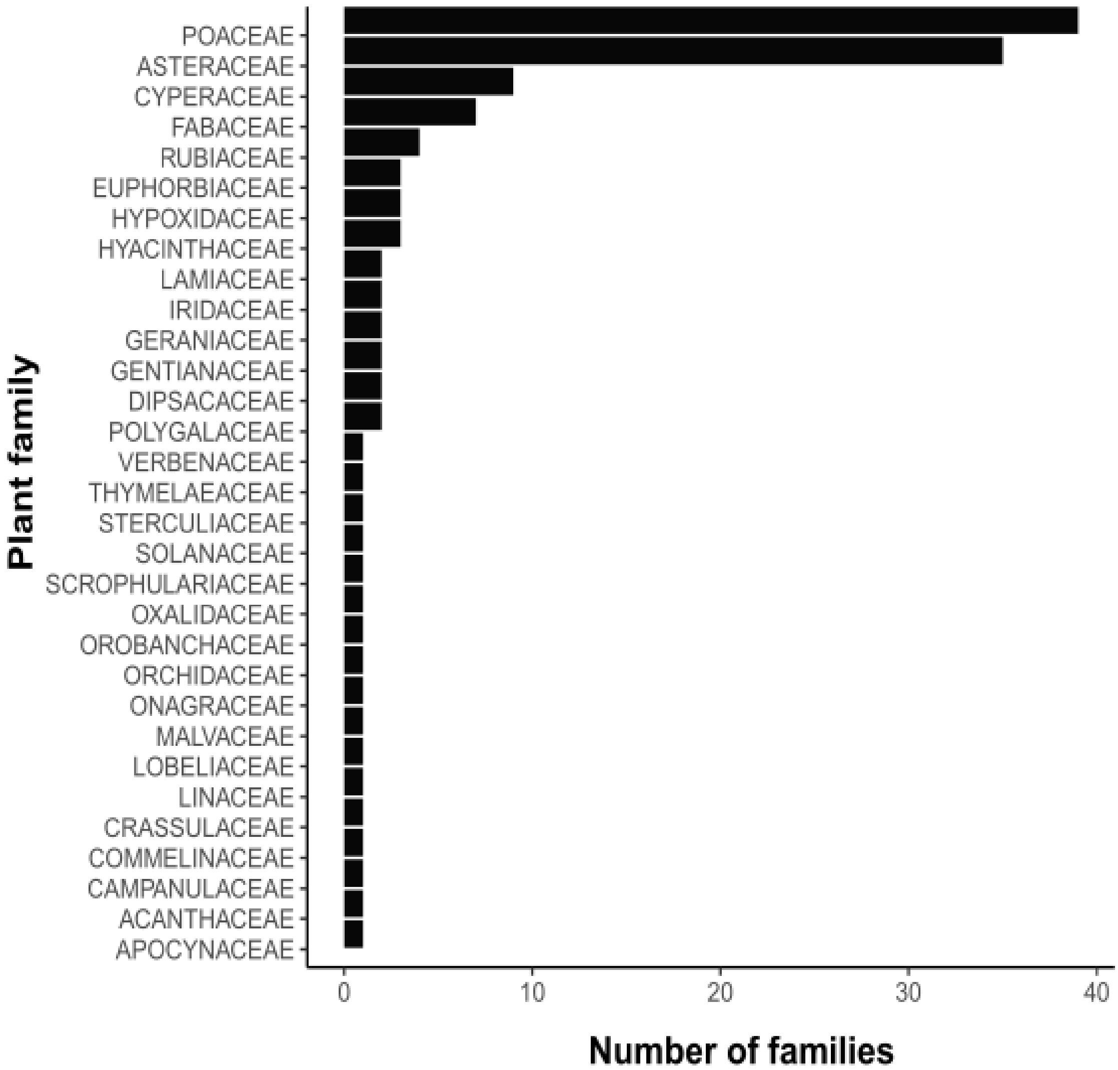

Figure 4.13. Contribution of each plant family to the total number of flowering plant species recorded in the QwaQwa communal area (n= 127 species).

The protected area supported grassland with greater plant diversity than the communally grazed area, as shown by a greater ($P<0.05$) Simpson diversity index, Shannon diversity index, and Pielou's index, although the two did not differ ($P>0.05$) for species richness (Table 4.7). The spatial pattern of variation in plant diversity on one side of the fence was reflected on the other side to a degree, as shown by a correlation (r=0.7319; df =19; $P=0.0002$) between the two for Simpson diversity and for Pielou's index (Table 4.8). The protected grassland would therefore appear to be slightly more complex than the communal grassland.

Table 4.7. Fence-line differences between grassland of a protected area and an adjacent communally grazed area for measures of plant diversity (n= 20), r is the similarity in spatial pattern either side of the fence.

Index of diversity	Protected area	Communal area	t-value	P	r	P
Simpson's D'	0.9211±0.0090	0.8849±0.0144	3.6279	0.0017	0.7319	0.0002
Shannon-Wiener H'	2.8281±0.0694	2.6371±0.0856	2.0877	0.0505	0.3175	0.1724
Pielou's J'	0.8829±0.0180	0.8385±0.0206	-2.1998	0.0403	0.4651	0.0387
Species richness	25.3±1.2692	24.45±1.6693	0.3736	0.4388	-0.1834	0.7128

There was a higher cover (P<0.05) of annual herbs, shrubs and a slight difference in alien species cover between the communal area and protected area (Table 4.8). Grazing had a slight impact on the plant groups between the protected area and the communal area. Grazing had no significant differences (r= 0.8670, df = 19, t= -0.3834, P>0.3973) in terms of family richness especially on grasses between the protected area and the communal area.

Table 4.8. Aerial cover (%) and number of species per plot (mean ±SE) for plant groups on each land use (n=20).

Plant groups		**Protected area (aerial cover)**	**Communal area (aerial cover)**	***t*-test (Aerial cover)**	**P**	**Number of species (Protected area)**	**Number of species (Communal area)**
1. Herbs	Annuals	1±0.3485	0.45±0.2105	1.9272	0.0690	0.6±0.1887	0.3±0.1304
	Biennial	0	0.15±0.1860	-1	0.3299	0	0.05±0.0620
	Perennial	17.9±2.1929	20.05±3.1177	-0.7047	0.4895	9.7±1.0536	10.05±1.2961
2. Monocotyledons	Grasses	62.2±4.4563	63.25±7.3621	-0.145	0.8862	9.4±0.6324	9.6±0.9454
	Sedges	3.35±1.1498	3.4±0.8364	-0.0462	0.9636	0.95±0.246	1.25±0.2827
	Other	1.2±0.3438	0.65±0.2172	1.4734	0.157	0.7±0.1822	0.45±0.1364
	Indigenous	92.35±4.822	94.75±7.9119	-0.3010	0.7666	24.6±1.498	22.95±2.1791
	Shrubs	3.35±1.2181	1.4±0.4258	2.1023	0.0490	1.45±0.543	0.65±0.2254
3. Dicotyledons		25.35±2.648	27.3±3.7341	-0.6032	0.5535	13.65±1.28	12.25±1.2625
	Dwarf shrubs	2.2±0.4552	4.45±2.7032	-1.069	0.2985	1.6±0.3168	1.6±0.3753
	Geophytes	3.9±1.2432	2.05±0.5508	1.5483	0.1381	1.85±0.351	1.25±0.2680
	Endemics	2.3±0.6917	1.5±0.5934	1.3314	0.1988	1.05±0.246	0.55±0.1903
	Aliens		0.65±0.3278	4.55±2.5042	-1.8972	0.0731	0.25±0.123
	Indigenous	92.35±4.822	94.75±7.9119	-0.3010	0.7666	24.6±1.498	22.95±2.1791

4.6.2 Royal Natal National Park and Witsieshoek communal area: Species richness and diversity

A total of 125 vascular plant species was recorded, with the protected area having 107 species and the communal area 81 species. Only 60 species (44.6 %) occurred on both sides of the fence, 14 of which were Poaceae, and 65 species occurred only on one side. Three species of South African or Drakensberg endemics were recorded (*Aster pleiocephalus*, *Helichrysum krebsianum* and *Vernonia thodei*), of which only one occurred on both sides. A total of 31 families was recorded, with 28 in the protected area (Figure 4.14) and 25 in the communal area (Figure 4.15), of which 20 families were shared. The dominant families in terms of number of species were Asteraceae, Poaceae, Cyperaceae and Orchidaceae. There was a significant difference (t=2.4853; *P*= 0.0346; RNNP = 69.8±5.3806; Witsieshoek = 55.8±2.6449) in aerial cover of Poaceae family between the two rangelands. Grazing therefore appears to have affected the representation of grasses on either side of the fence.

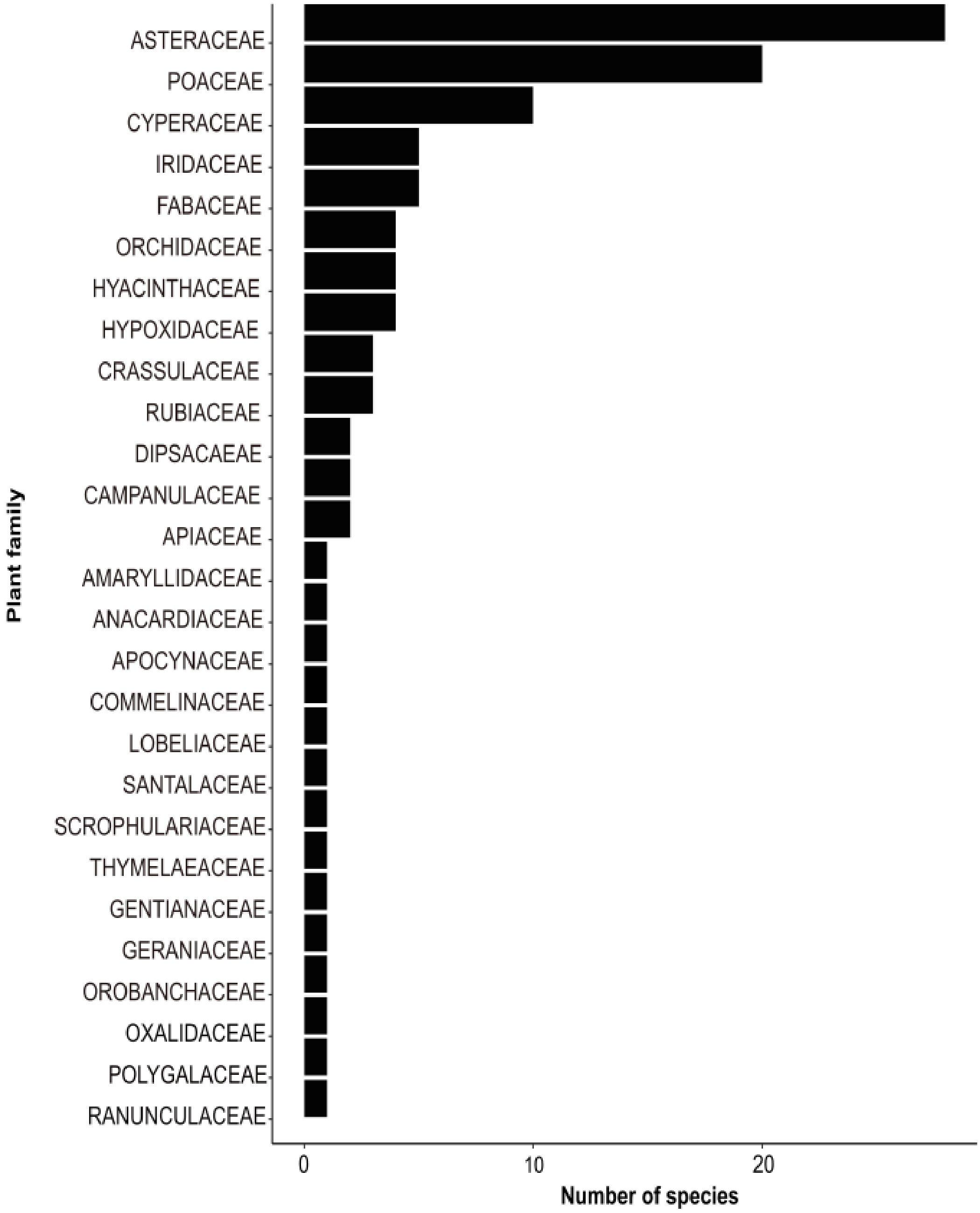

Figure 4.14. Contribution of each plant family to the total number of flowering plant species recorded in the Royal Natal National Park (n=107 species).

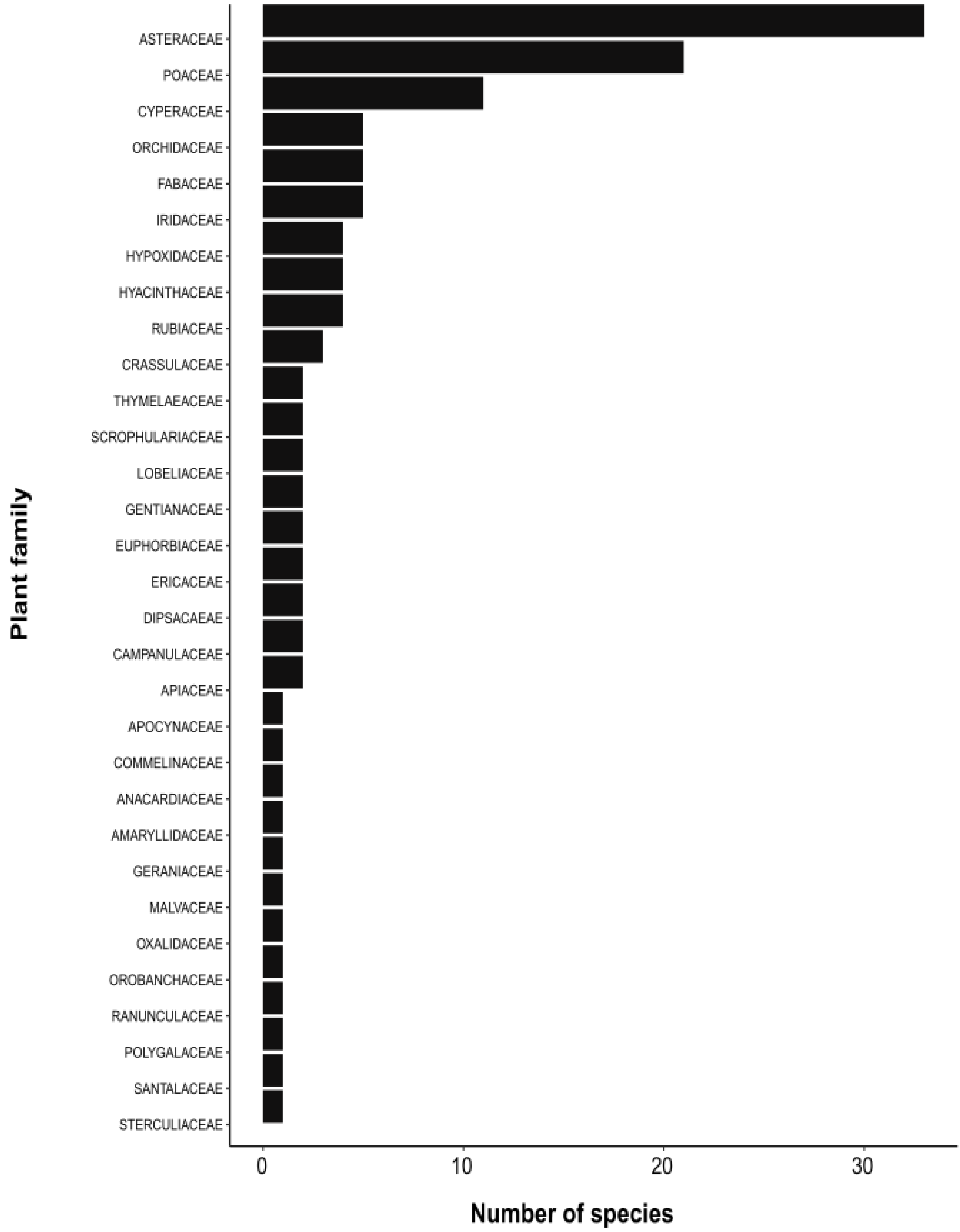

Figure 4.15. Contribution of each plant family to the total number of flowering plant species recorded in the Witsieshoek communal area (n=81 species).

Table 4.9. Fence-line differences between grassland of a protected area and an adjacent communally grazed area for measures of plant diversity (n= 10) r is the similarity in spatial pattern either side of the fence.

Index of diversity	Protected area	Communal area	t-value	P	r	P
Simpson's D'	0.9182±0.0798	0.9872±0.0070	-0.8493	0.6460	-0.1663	0.4176
Shannon-Wiener H'	3.2899±0.071	3.0878±0.0337	3.4366	0.0074	0.5695	0.0856
Pielou's J'	0.8939± 0.0114	0.8900± 0.0057	0.3236	0.7535	0.1254	0.7298
Species richness	39.9±1.7792	32.2±0.8537	4.7743	0.0010	0.4257	0.2199

The protected area supported grassland with a greater (P<0.05) species richness and Shannon diversity than the communal rangeland, but the two did not differ (P>0.05) for Simpson diversity or Pielou's evenness index (Table 4.9). There was a higher cover (P<0.05) of indigenous plants or grasses on the protected area than the communal rangeland, but a higher cover of dwarf shrub species on the communal than on the protected area. It would therefore appear that grazing influenced the representation of these plant groups on protected area versus communal rangeland (Table 4.10).

Table 4.10. Aerial cover (%) and number of species per plot (mean ±SE) for plant groups on each land use (n=10).

Plant groups		Protected Area (aerial cover)	Communal Area (aerial cover)	*t*-test (Aerial cover)	P	Number of species (Protected area)	Number of species (Communal area)
1. Herbs	Annuals	0.5±0.2041	0.3±0.2035	1	0.3434	0.5±0.2041	0.3±0.2035
	Perennial	28.5±1.3960	26.0±6.2861	0.3529	0.7322	18.5±0.5485	15.5±2.193
2. Monocotyledons	Grasses	69.8±4.9118	55.8±2.5218	2.4853	0.0346	11.2±0.5409	9.4±0.4311
	Sedges	16.1±3.3373	8.5±2.5236	1.5946	0.1453	4.2±0.5409	2±0.3760
	Other	2.6±0.4944	2.5±0.2562	0.1982	0.8473	1.7±0.3058	2.2±0.2378
	Indigenous	119.6±5.7028	96±2.9128	3.6588	0.0052	39.8±1.5916	32.1±0.834
3. Dicotyledons		32.6±1.1320	28.1±1.6108	1.9121	0.0881	20.8±0.3277	17.5±0.879
	Shrubs	1.7±0.2738	2.3±0.4933	-0.8017	0.4433	1.5±0.2041	1.8±0.3706
	Dwarf shrubs	2.2±0.2652	3.3±0.3194	-2.1807	0.0571	1.5±0.2041	2±0.2461
	Geophytes	4.2±0.5900	4.8±0.6162	-0.5858	0.5724	2.5±0.3118	2.2±0.2770
	Endemics	0.4±0.2434	0	1.5	0.1679	0.2±0.1217	0
	Indigenous	119.6±5.7028	96±2.9128	3.6588	0.0052	39.8±1.5916	32.1±0.834
	Aliens	0	0		-	0	0

4.6.3 Pattern of soil variation

The first two axes of a PCA explained 61% of the total variance in 14 soil variables of the fence-line contrast between GGHNP and the QwaQwa area (Table 4.11). The first axis described a gradient from soils with a high total cation concentration, dominated by calcium and magnesium, through to soils with high acidity saturation and high exchangeable acidity (Figure 4.16; Table 4.12). The second axis described a gradient from carbon rich soils also with high concentrations of nitrogen and zinc, through to soils with high concentrations of manganese and copper (Figure 4.16; Table 4.12). The third axis described a gradient from soils with high exchangeable acidity and acidity saturation, and phosphorus concentration, through to soils with a high pH (Figure 4.16; Table 4.12). There was no signifcant difference in PCA site scores of axis 1 (t= 0.8582, P=0.402), whereas in axis 2 there was a signifcant difference (t= -16.035, P = 4.199E-12) according to a paired *t*-test. There was no significant difference (P>0.3282; n = 14) in each soil variable according to a paired *t*-test. The first two axes could only explain 67% of the total variance in 14 soil variables of the fence-line contrast between Royal Natal National Park and the Witsieshoek communal area (Table 4.11). The first axis described a gradient from soils with a high pH through to soils with high concentrations of phosphorus, zinc, carbon, nitrogen, copper and manganese, and high exchangeable acidity (Figure 4.17;Table 4.13). The second axis described a gradient from soils with a high total cation content dominated by calcium and and magnesium, through to clay-rich soils with high levels of acidity saturation (Figure 4.17; Table 4.13). The third axis was relatively weak and is not considered further (Figure 4.17; Table 4.13). There was a signifcant difference between the PCA site scores of axis 1 (t= -3.1338, *P*= 0.0139), whereas axis 2 was not signifcant (t= -0.6629, *P*= 0.526) according to a paired *t*-test. There were significant differences on each soil variable namely, phosphorus (t=3.5058; *P*=0.0080; n=14), zinc (t=4.406; *P*=0.0022; n=14) and clay content (t=4.0847; *P*=0.0035; n=14) according to a paired *t*-test.

Table 4.11. Summary output of PCA analysis of soil environmental variables along the fence- line contrast at Royal Natal National Park (n= 9 for each site) and Golden Gate Highlands National Park (n = 19 for each site).

	PCA 1	PCA 2	PCA 3
Golden Gate Highlands National Park/QwaQwa			
Eigenvalues	4.8230	3.7276	1.7655
Proportion Explained	0.3445	0.2663	0.1261
Cumulative Proportion	0.3445	0.6108	0.7369
Royal Natal National Park/Witsieshoek			
Eigenvalues	5.1588	4.2210	2.0712
Proportion Explained	0.3685	0.3015	0.1479
Cumulative Proportion	0.3685	0.6700	0.8179

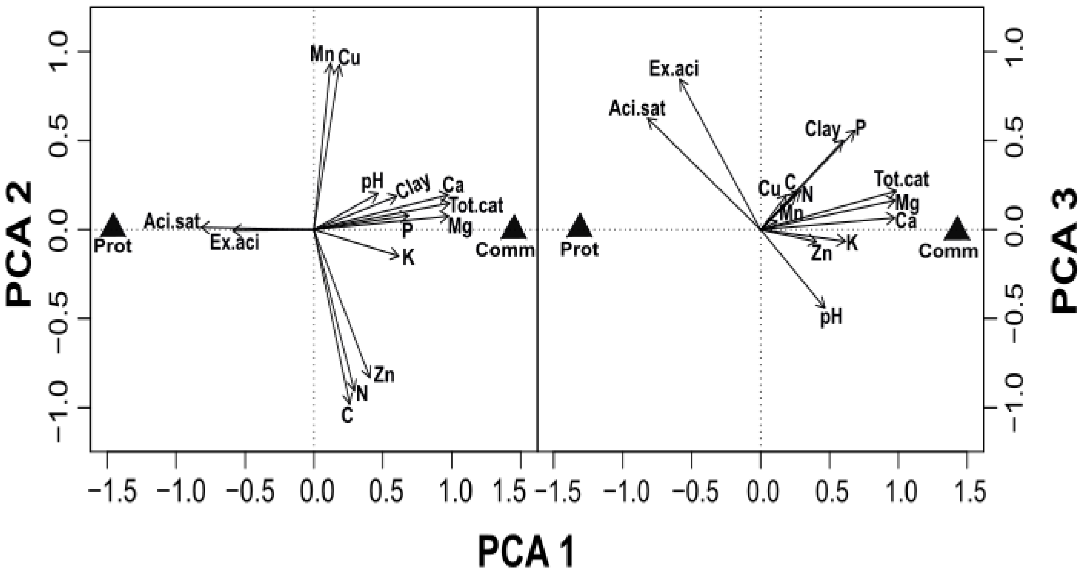

Figure 4.16. Plot of soil environmental variables on the first three axes of a Principal Component Analysis for a fence-line contrast at Golden Gate Highlands National Park/QwaQwa communal area (n= 19 for each side). Key to abbrev: Prot = Protected area, Comm = Communal area, Aci.sat = Acidity saturation, Ex.aci = Exchangeable acidity, C= Carbon, Mn = Manganese, Tot.cat = Total cation, K = Potassium, P = Phosphorus, Zn = Zinc, Cu = Copper, N = Nitrogen, Mg = Magnesium, Ca = Calcium.

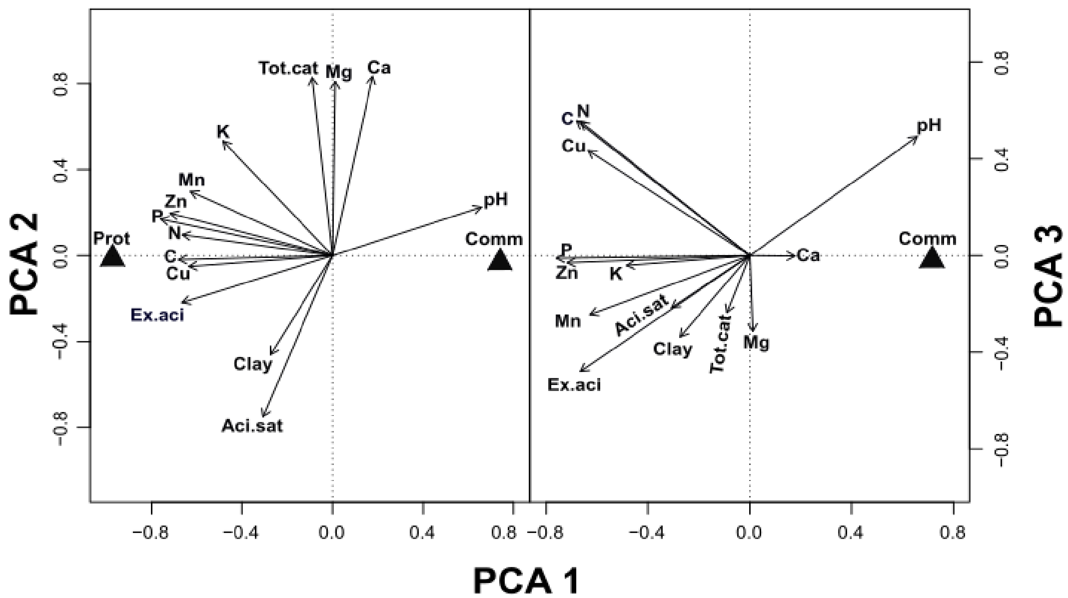

Figure 4.17. Plot of soil environmental variables on the first three axes of Principal Component Analysis from a fence line contrast at Royal Natal National Park/Witsieshoek communal area (n= 9 for each site). Key to abbrev: Prot = Protected area, Comm = Communal area, Aci.sat = Acidity saturation, Ex.aci = Exchangeable acidity, C= Carbon, Mn = Manganese, Tot.cat = Total cation, K = Potassium, P = Phosphorus, Zn = Zinc, Cu = Copper, N = Nitrogen, Mg = Magnesium, Ca = Calcium.

Table 4.12. Summary of the loading of each soil environmental variable on each of the first three axes of a PCA describing a fence-line contrasts at Golden Gate Highlands National Park/QwaQwa communal area (n= 19 for each site).

Variables	PCA 1	PCA 2	PCA 3
P	0.8059	0.0931	0.6546
K	0.7223	-0.1749	-0.0767
Ca	1.1431	0.2298	0.0780
Mg	1.1466	0.0893	0.1934
pH (KCl)	0.5471	0.2405	-0.5174
Zn	0.4774	-0.9790	-0.0794
Mn	0.1407	1.1003	0.0558
Cu	0.2185	1.0881	0.2279
N	0.3446	-1.0659	0.2586
C	0.3050	-1.1543	0.2395
Clay	0.7063	0.2212	0.5897
Total cation	1.1563	0.1749	0.2554
Acidity saturation	-0.9634	0.0145	0.7347
Exchangeable acidity	-0.6910	-0.0060	0.9929

Table 4.13. Summary of the loading of each soil environmental variable on each of the first three axes of a PCA describing a fence-line contrasts at Royal Natal National Park/Witsieshoek communal area (n= 9 for each site).

Variables	PCA 1	PCA 2	PCA 3
P	-0.8942	0.1997	-0.0130
K	-0.5702	0.6262	-0.0484
Ca	0.2087	0.9802	-0.0022
Mg	0.0136	0.9532	-0.3683
pH (KCl)	0.7735	0.2647	0.5785
Zn	-0.8429	0.2307	-0.0351
Mn	-0.7382	0.3502	-0.2867
Cu	-0.7465	-0.0599	0.5113
N	-0.7803	0.1138	0.6492
C	-0.7993	-0.0226	0.6542
Clay	-0.3223	-0.5422	-0.3957
Total cation	-0.1053	0.9735	-0.2770
Acidity saturation	-0.3614	-0.8824	-0.2530
Exchangeable acidity	-0.7819	-0.2573	-0.5633

Chapter 5: Discussion and Conclusion

5.1 Grazing intensity gradient

5.1.1 Distance as a proxy for grazing impact

A first question to address was whether distance from a kraal site could serve as a proxy measure for grazing intensity. Although it has been effectively employed in many piosphere studies (Nangula and Oba 2004; Derry 2004; Sandhage-Hofmann et al. 2015), a mountainous terrain such as the northern Maloti-Drakensberg study area imposes additional complications of greater environmental variation in relation to topography. The value of distance as a proxy will therefore depend on whether distance is confounded with other environmental factors that could be expected to influence vegetation composition. The relationship between distance and three measures of short-term (<1 year) grazing pressure were examined. The amount of cattle dung decreased non-linearly with an increase in distance, in support of distance serving as a proxy for grazing intensity. However, no relation was found for the amount of moribund grass or of biomass removed by grazing.

The employment of distance as a proxy for a gradient in grazing intensity from kraals therefore had limited support. However, the abundance of dung has been shown elsewhere to reflect a grazing gradient of wild herbivore species (Dorji et al. 2013). The value of cattle dung as a proxy for grazing intensity over the long-term can be questioned because the sites of dung deposition may not be the same as the sites of grazing (Omaliko 1981). However, as Adler and Hall (2005) suggest, distance from a kraal site or similar represents a cumulative impact over a long period of time rather than only a year or season as reflected by the short-term indicators available for this study. On the basis of dung deposition distance being related to distance, and further that dung counts were considered to be the most accurate and direct measure of the three variables reflecting short-term grazing intensity, distance was accepted as a proxy for a gradient in grazing intensity. Future use of this proxy in similar environments should, however, attempt to determine if the presence or abundance of human settlement influences the distance livestock prefer to graze from kraals, which was a possible pattern for this study area. There may have been dung removal by residents for purposes of manure and fuel for burning. This is the case at high altitude in Lesotho as fire wood is scarce.

Dorji et al. (2013) has also argued that the spatial patterns did not reflect grazing intensity in their study. Grazing intensity further depends on the heterogeneity in the vegetation communities, forage availability, and disturbances across the landscape, with livestock expected to flock on areas with greater forage availability and to avoid disturbed places (Dorji et al. 2013), which contrasts with the findings of Manthey and Peper (2010).

Using distance as a proxy does not mean that grazing intensity would change in a uniform manner with distance. This was shown by the negative power relationship between dung abundance and distance (Figure 4.1). In the mesic grassland the area used by livestock from any individual kraal would be expected to overlap with the area used by other herds in a spatial pattern that could not be easily shown (Moeslund et al. 2013). So any gradient may in effect be disguised.

Distance has served as a reliable proxy for grazing intensity in many piosphere studies around waterholes in semi-arid systems (Thrash 2000; Riginos and Hoffman 2003; Sandhage-Hofmann et al. 2015), which was a basis for adopting distance as a proxy measure in this study. However, a major distinction between piosphere studies in semi-arid environments and the distance gradient of this study is the degree of environmental heterogeneity coupled with differences in vegetation composition and diversity. Most piosphere studies in semi-arid steppe grasslands have been conducted in regions of flat terrain with little variation in soil properties (Andrew 1988; Thrash and Derry 1999; Sternberg 2012; Kotze et al. 2013). By contrast, the montane environment of this study was topographically highly variable. Effort was therefore made to sample as uniform a topo-edaphic environment as possible. This was successful because distance was shown not to be confounded with aspect, altitude, solar radiation (4.1.1), or soil physico-chemical properties (4.1.2). However, the plots sampled nonetheless showed a degree of variation in topo-edaphic properties that can be expected to have increased the variability of supported vegetation. It becomes more difficult to show the effect of any agent when the response variable is intrinsically highly variable. The floral diversity of the grasslands studied is also more species-rich and hence more variable than their species-poor counterparts in semi-arid environments (Saayman et al. 2016; Eldridge et al. 2017), which adds to the disguising effect of variability. However, despite this disguised gradient a reasonable gradient intensity was achieved.

The sample size (n=90), sampling plot size (25 m^2) and the range of distances from kraal sites to sampling plots (± 400 m-12 km) of the study may be perceived as shortcomings to the study. However, our sample size concurs with other surveys elsewhere that have used surrogates to assess grazing pressure (Fernandez-Gimenez and Allen-Diaz 2001; Manthey and Peper 2010; Morris 2017), but differed in terms of sampling plot size (Hendricks et al. 2005; Hoshino et al. 2009; Manthey and Peper 2010; Dorji et al. 2013) but compares well with distance from kraal sites (Fernandez-Gimenez and Allen-Diaz 2001; Sasaki et al. 2007; Manthey and Peper 2010; Tarhouni et al. 2010; Dorji et al. 2013). Another reason for the shortcomings of the study might be that livestock does not stray at greater distances from kraal sites.

5.1.2 Pattern of soil variation with distance and along fence-line contrasts

Different grazing regimes should differ in terms of impacts on soil properties that, in turn, might affect vegetation. High levels of livestock grazing have been shown to impact soil physical and chemical properties (Greenwood and McKenzie 2001). N is an important source for growth reproduction in plants (Jones et al. 2005; Näsholm et al. 2009; Warren 2014). Increases in livestock intensity have clear impacts on soil cations and anions and soil fertility (Eldridge et al. 2017). Soil environmental properties were found not to be confounded with distance from the kraal sites.

Soil physico-chemical properties did not differ, which contrasts with a view that large cattle dung deposits and urine increases soil nutrients (Savory and Parsons 1980; Savory and Butterfield 1998). Although dung pats and urine may possibly increase soil nutrients, the high stocking rates of ungulates from the communal areas may result in increased offtake of nutrients through grazing (Tiedemann 1986; Frank et al. 1995). The impact of livestock trampling, although not measured in this study, can cause a decline in soil permeability to water flow (Heathwaite et al. 1990; Pietola et al. 2005). It is also worth mentioning that livestock grazing does have positive impacts on soil properties but when there is only intermediate grazing intensity (Savory and Butterfield 1998).

Although soil physico-chemical properties were not confounded with distance in this study, observational studies done in the Mongolian rangeland along a grazing gradient show that C, N and P increased with grazing pressure, and livestock densities decreased with increasing distance from water stations (Fernandez-Gimenez and Allen-Diaz 2001; Stumpp et al. 2005). Experimental and enclosure studies also suggest that soil quality does decline under increased grazing intensity. Heavily grazed grasslands generally have a negative (Bauer et al. 1987; Sandhage-Hofmann et al. 2015) or a positive (Abril and Bucher 1999) relationship between C and N, whereas ungrazed grasslands remain unchanged. The general consensus is that high grazing intensity generally reduces the nutrient status of the soil by indirectly exposing the topsoil to erosion (Belsky et al. 1993, Campbell et al. 1994, Abule et al. 2005), whereas the high nutrient values may be ascribed to the positive effect of animal manure, particularly on the sacrifice zone (Sandhage-Hofmann et al. 2015).

5.1.3 Influence of grazing on plant groups and species

The main objective of the study was to determine the impact of communal heavy grazing on botanical composition and plant diversity. There was a modest increase in species richness and change in botanical composition with an increase in distance (4.3). The effect of grazing only influenced species composition. This is in line with other surveys in the mesic and temperate grasslands (Scott-Shaw and Morris 2015; Lyseng et al. 2018) and across biomes (Rutherford and Powrie 2013). The observed increased species richness under the grazing intensity gradient in this study may be attributed to moderate grazing impact which was found elsewhere to favour or promotes richness (Milchunas et al. 1998; Bai et al. 2001). Species richness and particularly the number of palatable species is reported to tend to increase with distance and the effect is more pronounced on mountainous areas than on the foothills and plains (Hendricks et al. 2005). The limited findings in this study may be attributed to a history of more than a century of continuous heavy grazing before this study was undertaken. Therefore, a long term history of heavy grazing may have disguised any gradient of grazing intensity.

The objective was also to determine if grazing had an influence on plant species composition and plant diversity between the intensely grazed communal area and lightly grazed protected area. The impact of intense grazing pressure was more pronounced on the fence-line contrasts, especially the one involving the Royal Natal National Park/Witsieshoek communal area compared to Golden Gate Highlands National Park/QwaQwa communal area (4.6.1 and 4.6.2). RNNP had been under protection from heavy grazing for decades before this study, hence a lesser impact of heavy grazing. These observations are in line with other studies in the mesic grassland in South Africa and elsewhere (Waser and Price 1981; Noy-Meir et al. 1989; Noy-Meir and Oron 2001; Scott-Shaw and Morris 2015). The GGHNP side has been subjected to continuous livestock grazing for decades and has only been protected for (±25 years) before the study. Plant diversity might have been compromised during the period of commercial livestock ranching. These findings on the GGHNP are in accordance with other studies in South Africa and elsewhere, where heavy grazing on a communal rangeland and light grazing from a commercial rangeland was shown to have had a slight or no major influence on species richness and botanical composition (Todd and Hoffman 1999; O’ Connor 2005; Uys 2006; O’ Connor et al. 2011; Nenzhelele et al. 2018). Therefore, the effect of grazing appears to have only influenced species composition and not species richness.

Species composition changed significantly along the gradient of grazing intensity from the kraals. Species that were closer to the kraals and decline with distance included the unpalatable shrubs *Felicia muricata* and *Euphorbia striata*, which are good indicators of heavy grazing (Todd 2006). There was grazing tolerant species such as *Ledebouria sp.*, this species can morphologically tolerate grazing through its flat rosette and hard leaves (Todd 2006). Palatable intolerant grazing species such as *Themeda triandra* are reported to be scarce in the vicinity of cattle posts in Lesotho but this was measured on basalt geology (Morris 2017). Some of these species that were closer to kraal sites might have come as a result of exploitation of the high availability of nutrients from ungulates urine and dung pats (Badía et al. 2008; Hoag 2016; Morris 2017).

Species that decreased with increasing distance were the aliens *Richardia brasiliensis* and *Hypochaeris radicata*. *Richardia brasiliensis* is able to mainly avoid grazing because of its flat, short rosette leaves (O' Connor 2005). Species found at intermediate distance from kraal sites included the palatable and grazing tolerant grass species such as *Heteropogon contortus*, *Monocymbium ceresiforme,* and *Trachypogon spicatus,* and the short herbaceous palatable species *Sebaea natalensis* and *Craterocapsa tarsodes*. This pattern suggests that these species benefitted from intermediate levels of grazing intensity.

5.1.4 Influence of grazing on plant groups and species

There was an increase on graminoids, indigenous plants and perennials with a decrease in grazing intensity along the distance gradient, whereas shrubs and dwarf shrubs, and annual herbs showed the converse response. Different factors influence whether or not, or the extent to which, different groups or species are affected by heavy grazing. Geophytes have leaves and bulbs with high concentrations of alkaloids, and glycosides of calcium oxalate (Frohne and Jensen 1979). These substances discourage most herbivores from grazing the leaves of most geophytes (El-Bahri et al. 2000), hence their persistence under intense grazing pressure. Geophytes were not a dominant plant group in the Maloti-Drakensberg but they are in heavily grazed dry Mediterranean grasslands (Noy-Meir and Oron 2001). *Richardia brasiliensis* is an example of a species that generally persists under communal heavy grazing because it is a low growing prostrate species (O' Connor 2005), whereas *Aristida junciformis* is a rarely grazed grass except when young on account of its tough leaves (van Oudtshoorn 1992). Species such as *Ledebouria* and *Brunsvigia* had their flowers heavily utilized by livestock, but the plants survived because they grow with their flat, hard leaves appressed to the ground. Species in these QwaQwa grasslands therefore exhibited a wide range of means of tolerating or avoiding livestock impact.

The Maloti-Drakensberg is renowned for its perennial plant diversity. These grasslands are comparable with heavily grazed rangelands in Israel where high diversity is directly and indirectly influenced by a long evolutionary history under constant stress by grazing, drought and fire (Naveh and Whittaker 1979). There were marginal differences between RNNP and Witsieshoek communal area in the cover of plant families of Asteraceae, Campanulaceae, Crassulaceae, Apiaceae, Fabaceae, Gentianaceae, Hyacinthaceae, Hypoxidaceae, Orchidaceae, Poaceae and Oxalidaceae. This is supported by the cover of plant groups with the RNNP having more grasses, dicotyledons and indigenous plants, whereas the communal area grazing supported more dwarf shrubs. The number and cover of geophytes were very low, despite their effective avoidance mechanisms (narrow and very hard leaves) against grazing (Noy-Meir and Oron 2001).

The cover and number of the plant groups such as perennials remained the same in this study, whereas in the Namaqualand fence line contrast (Todd and Hoffman 1999) and Tibetan pastures (Wang et al. 2017), heavy grazing changed the vegetation cover from perennials to annuals. The low cover and number of annuals in this study is consistent with the mesic grassland grazing system, whereas this is in sharp contrast with the drylands of southern Africa (Namaqualand and Karoo) and other parts of the world. Annual plants in semi-arid and dry environments may benefit from sustained communal heavy grazing pressure (Todd and Hoffmann 1999; Nenzhelele et al. 2018). By contrast, other plant groups such as perennial herbs and Maloti-Drakensberg endemics did not increase because they were more vulnerable to trampling and grazing.

Plant cover plays an important role in preventing soil loss which might have adverse implications for vegetation composition. Plant cover increases as grazing pressure decreases with increasing distance from water (Hendricks et al. 2005; Todd 2006). In alpine meadows grazing has been found to both decrease and increase the cover of graminoids and forbs respectively (Wang et al. 2017). In general, continuous grazing on communal areas leads to an increase in annual forbs and unpalatable perennial growth forms or plant groups, and a decrease in the cover of palatable perennial plants (Anderson and Hoffman 2007).

5.1.5 Fire pattern in relation to grazing gradients

Fire is generally considered to be the most important disturbance agent affecting montane grasslands through suppressing or reversing woody ingression and by promoting fresh growth of perennial grasses and herbaceous plants (Uys et al. 2004). In this study, fire frequency was confounded with distance along the grazing gradient which was considered to be an unavoidable outcome owing to the influence of livestock grazing. However, the potential influence of this confounding factor was lessened by the relatively small proportion of the study area (4.2 %) that burned on average each year. Between 2007 and 2008, the area burned was much higher this was ascribed to years of high rainfall which brought about high grass production.

The lowest percentages of area burnt were recorded in May and November (Figure 4.5) which marked the beginning and end of the dry season. This can be explained by low standing biomass and less fuel accumulation. The high fire frequency was observed between July and September, which is a peak season for fire events. This is also the case in many other places as fire frequency is maximum in the dry season after the senescence stage (Bucini and Lambin 2002; Hudak et al. 2004; Govender et al. 2006). These high frequency events for example are a norm in many communal rangelands, as communities believe that fire gives way to a new fresh sward for livestock in the next season (Kepe 2005). The grassland of the study area is commonly described as Sourveld, where fires are used regularly to suppress woody ingression and to remove alien plants (Trollope et al. 2002; SANBI 2014b). The grassland of the study area would appear to have responded similarly to that of other grasslands in the KwaZulu-Natal Drakensberg, by remaining resistant to compositional change in the face of variable fire-return intervals (Gordijn et al. 2018).

5.2 Conclusion

Communal livestock grazing was expected to have a pronounced negative impact on plant composition and diversity along a gradient of grazing pressure from kraal sites. The gradient of grazing pressure with increasing distance from kraal sites was weak. This gradient of grazing pressure showed that species richness increased modestly with distance from kraal sites. A long history of communal livestock grazing pressure showed a minor effect on species composition along a grazing intensity gradient from kraal sites. Continuous livestock grazing pressure on plant groups showed that graminoids, indigenous and perennials plants increased with a decrease in grazing pressure from kraal sites. In contrast to shrubs, dwarf shrubs, annual herbs and aliens showed a decrease with an increased distance along a gradient. Spatial heterogeneity in relation to topographic variability, in conjunction with a long history of continuous livestock grazing, may have contributed to the strength of grazing impacts recorded. Grazing gradients were confounded with fire frequency, however soil physical properties showed the converse.

Light livestock grazing is considered to be essential for maintaining or improving the diversity of these communal grassland grazing systems. Communal livestock grazing had an effect on species richness as it was lower on the Witsieshoek communal area when compared with RNNP area. The impact of grazing on GGHNP and QwaQwa communal area showed no differences on species richness and species composition. The state of the grassland on the RNNP side of the fence-line contrast perhaps presents a typical grassland state that could be achieved in this communal grazing region if conservative livestock numbers could be maintained. Plant species composition in the study area has remained resilient despite a long history of communal livestock grazing. This was evident in terms of grazing tolerant species. However, this may not be sustainable going into the future if grazing pressure remains consistently high, as this may eventually result in transformation from grazing-tolerant perennial species, including palatable grasses, to communities dominated by unpalatable woody plants, annual plants, and alien plants as witnessed in other areas. Such a shift would present uncertainties in terms of the forage quantity and quality potential, hence livestock production, of the study area.

The implications might be negative firstly for the communities occupying the area and, secondly, in terms of water production for the country because the Maloti-Drakensberg is known as the water tower of the country. The degradation of the grazing resource in the QwaQwa communal area is likely to continue because of open-access grazing by herders. Measures for controlling livestock numbers, cattle movement through herding and resting camps as applied to commercial rangelands would probably be unsuccessful because communal farmers do not have the financial means of doing so.

Future studies for improving our understanding of the impact of livestock on plant diversity in the QwaQwa region may expand by study of both the sandstone and basalt geology and altitudinal gradients thereby increasing the grazing gradient. A clearer understanding is also required of the landscape pattern of grazing intensity. This study would have benefitted from having knowledge about the relationship between site of grazing and of defecation, which could be gained from tracking ungulate movements in conjunction with recording of defecation. The reciprocal influence of fire on grazing intensity needs to be explored further by looking at fire regimes, fire intensity and fire return intervals, in order that their joint effect on grassland plant diversity can be better understood.

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APPENDICES
APPENDIX A: Plant species list, with synonyms and Family

Species name	Synonym	Family
Abilgaardia ovata (Burm.f.) Kral		CYPERACEAE
Acalypha peduncularis E.Mey. ex Meisn.		EUPHORBIACEAE
Acalypha punctata Meisn. var. punctata		EUPHORBIACEAE
Acalypha schinzii Pax	*Acalypha depressinervia* (Kuntze) K.Schum.	EUPHORBIACEAE
Agrostis eriantha Hack.		POACEAE
Ajuga ophrydis Burch. ex Benth.		LAMIACEAE
Albuca setosa Jacq.		HYACINTHACEAE
Alectra sessiliflora (Vahl.) O. Kuntze		OROBANCHACEAE
Alepedia natalensis J.M.Wood & M.S.Evans		APIACEAE
Allotorepsis semialata (R.Br.) Hitchc. subsp. semialata		POACEAE
Aloe boylei Baker		ASPHODELACEAE
Amaranthus sp.		AMARANTHACEAE
Andropogon appendiculatus Nees		POACEAE
Andropogon distachyos L.		POACEAE
Andropogon schirensis Hochst. ex A.Rich.		POACEAE
Anthospermum herbaceum L.f.		RUBIACEAE
Anthospermum basuticum Puff		RUBIACEAE
Anthospermum pumilum subsp.pumilum		RUBIACEAE
Anthospermum rigidum Eckl. & Zeyh.		RUBIACEAE
Anthospermum streyi Puff.		RUBIACEAE
Anthoxanthum ecklonii (Nees ex Trin.) Stapf		POACEAE
Argyrolobium hedyantha		FABACEAE
Argyrolobium marginatum Bolus		FABACEAE
Argyrolobium sankeyi Dummer		FABACEAE
Argyrolobium variopile N.E Br		FABACEAE
Aristea cognata N.E.Br. ex Weim	*Aristea abyssinica* Pax.	IRIDACEAE
Aristea woodii N.E.Br.	*Aristea torulosa* Klatt.	IRIDACEAE
Aristida congesta Roem. & Schult..		POACEAE
Aristida junciformis Trin. & Rupr..		POACEAE
Asclepias affinis (Schltr.) Schltr.		APOCYANACEAE
Asclepias albens (E.Mey.) Schltr.		APOCYANACEAE
Asclepias cucullata (Schltr.) Schltr.		APOCYANACEAE
Asclepias gibba (E.Mey.) Schltr.		APOCYANACEAE
Asclepias multicaulis Vell.		APOCYANACEAE
Aspidoglossum woodii (Schltr.) Kupicha		APOCYANACEAE
Aster pleiocephalus (Harv.) Hutch.		ASTERACEAE
Athrixia arachnoidea J.M.Wood & M.S.Evans ex J.M.Wood		ASTERACEAE

Athrixia fontana MacOwan		ASTERACEAE
Berkheya rhapontica (DC.) Hutch. & Burtt Davy		ASTERACEAE
Berkheya setifera DC.		ASTERACEAE
Berkheya speciosa (DC.) O.Hoffm.		ASTERACEAE
Brachiara serrata (Thunb.) Stapf		POACEAE
Bromus sp.		POACEAE
Brunsvigia radulosa Herb. i		AMARYLLIDACEAE
Bulbostyis humilis (Kunth) C.B.Clarke		CYPERACEAE
Bulbostylis boeckeleriana (Schweinf.) Beetle		CYPERACEAE
Bulbostylis contexta Nees,Bodard		CYPERACEAE
Bulbostylis humilis (Kunth) C.B.Clarke		CYPERACEAE
Bulbostylis oritrephes (Ridl.) C.B.Clarke		CYPERACEAE
Bulbostylis schoenoides (Kunth) C.B.Clarke		CYPERACEAE
Bulbostylis scleropus C.B.Clarke		CYPERACEAE
Cephalaria oblongifolia (Kuntze) Szabó		DIPSACACEAE
Cerastium arabidis E.Mey. ex Fenzl		CARYOPHYLLACEAE
Chelianthus quadripinnata (Fodk) Kuhn.		PTERIDACEAE
Chlorophytum acutum (C.H.Wright) Nordal		AGAVACEAE
Chlorophytum fasciculatum Bak.	*Anthericum fasciculatum* Baker	AGAVACEAE
Chrysocoma tenuifolia P.J.Bergius		ASTERACEAE
Cliffortia repens Schltr.		ROSACEAE
Commelina africana L.		COMMELINACEAE
Convolvulus natalensis Bernh. ex Krauss		CONVOLVULACEAE
Conyza floribunda Kunth		ASTERACEAE
Conyza podocephala DC.		ASTERACEAE
Conyza chilensis Spreng.		ASTERACEAE
Cotula hispida (DC.) Harv.		ASTERACEAE
Crabbea acaulis N.E.Br.		ACANTHACEAE
Crassula dependens Bolus		CRASSULACEAE
Crassula lanceolata (Eckl. & Zeyh.) Endl.		CRASSULACEAE
Crassula vaginata Eckl. & Zeyh.		CRASSULACEAE
Craterocapsa tarsodes Hilliard & B.L.Burtt		CAMPANULACEAE
Cymbopogon validus (Stapf) Stapf ex Burtt Davy	*Cymbopogon nardus* (L.) Rendle	POACEAE
Cynodon dactylodon (L.) Pers		POACEAE
Cynodon incomplectus Nees		POACEAE
Cyperus C.F sanguin		CYPERACEAE
Cyperus obstusiflorus var. flavissimus (Schrad.) Boeck.		CYPERACEAE
Cyperus marginatus Thunb.		CYPERACEAE
Cyperus rupestris Kunth var. rupestris		CYPERACEAE
Cyperus schlechteri C.B.Clarke		CYPERACEAE
Cyperus schoenoides (R.Br.) Poir.	*Fimbristylis squarrulosa* F.Muell.	CYPERACEAE
Cyperus semitrifidus Schrad		CYPERACEAE

Cyperus sphaerocephalus Vahl		CYPERACEAE
Diclis reptans Benth.		SCROPHULARIACEAE
Dicoma anomala Sond.		ASTERACEAE
Dierama sp.		IRIDACEAE
Digitaria diagnolis (Nees) Stapf var. diagonalis		POACEAE
Digitaria longiflora (Retz.) Pers.		POACEAE
Digitaria monodactyla (Nees) Stapf		POACEAE
Digitaria tricholaenoides Stapf		POACEAE
Diheteropogon filifolius (Nees) Clayton		POACEAE
Disa cooperi Rchb.f.		ORCHIDACEAE
Disa sp.		ORCHIDACEAE
Drimia sp.		HYACINTHACEAE
Drosera natalensis Diels		DROSERACEAE
Elephantorrhiza elephantina (Burch.) Skeels		FABACEAE
Elionurus muticus (Spreng.) Kuntze		POACEAE
Eragrostis capensis (Thunb.) Trin.		POACEAE
Eragrostis centrifugus Nees		POACEAE
Eragrostis chloromelas Steud.		POACEAE
Eragrostis gummiflua Nees		POACEAE
Eragrostis patentissima Hack.		POACEAE
Eragrostis plana Nees		POACEAE
Eragrostis porosa Nees		POACEAE
Eragrostis racemosa (Thunb.) Steud.		POACEAE
Eragrostis sclerantha Nees		POACEAE
Erica algida Bolus		ERICACEAE
Erica woodii Bolus		ERICACEAE
Eriosema cordatum E.Mey		FABACEAE
Eriosema salignum E.Mey		FABACEAE
Eriospermum ornithogaloides Baker		RUSCACEAE
Eucomis automnalis (Mill.) Chitt.		HYACINTHACEAE
Euphorbia striata Thunb.		EUPHORBIACEAE
Euphorbia claviroides Boiss.		EUPHORBIACEAE
Euphorbia gueinzii Boiss.		EUPHORBIACEAE
Euryops transvaalensis Klatt subsp. setilobus (N.E.Br.) B.Nord.		ASTERACEAE
Eustachys paspaloides (Vahl) Lanza &Matter		POACEAE
Felicia fascicularis D.C		ASTERACEAE
Felicia filifolia (Vent.) Burtt Davy		ASTERACEAE
Felicia muricata (Thunb.) Nees		ASTERACEAE
Festuca scabra Vahl		POACEAE
Ficinia filiculmea B.L.Burtt		CYPERACEAE
Ficinia filiformis (Lam.) Schrad.		CYPERACEAE
Ficinia gracilis Schrad		CYPERACEAE
Ficinia stolonifera Boeckeler		CYPERACEAE

Fingerhuthia sesleriaeformis Nees		POACEAE
Gazania krebsiana Less.		ASTERACEAE
Gerbera ambigua (Cass) Sch.Bip		ASTERACEAE
Gerbera aurantiaca Sch.Bip.		ASTERACEAE
Gnaphalium drakensbergense Harv.		ASTERACEAE
Gnidia baurii C.H.Wright		THYMELAEACEAE
Gnidia fastigiata Rendle		THYMELAEACEAE
Gnidia gymnostachya (C.A Mey) Gilg.		THYMELAEACEAE
Gnidia kraussiana Meisn.		THYMELAEACEAE
Gnidia microcephala Meisn.		THYMELAEACEAE
Gnidia nodiflora Meisn.		THYMELAEACEAE
Gnidia renniana Hilliard & Burtt		THYMELAEACEAE
Gnidia woodii C.H Wright		THYMELAEACEAE
Graderia scabra Benth.		OROBANCHACEAE
Gymnopentzia bifurcata Benth.		ASTERACEAE
Haplocarpha scaposa Harv.		ASTERACEAE
Harpochloa falx (L.f.) Kuntze		POACEAE
Helichrysum acutatum DC.		ASTERACEAE
Helichrysum adenocarpum DC.		ASTERACEAE
Helichrysum aureonitens Sch.Bip.		ASTERACEAE
Helichrysum auriceps Hilliard		ASTERACEAE
Helichrysum bellum Hilliard		ASTERACEAE
Helichrysum caespititium (DC.) Sond. ex Harv.		ASTERACEAE
Helichrysum callicomum Harv.		ASTERACEAE
Helichrysum cephaloideum DC.		ASTERACEAE
Helichrysum chionospermum DC.		ASTERACEAE
Helichrysum coriaceum (DC.) Harv.		ASTERACEAE
Helichrysum cymosum Wood & Evans		ASTERACEAE
Helichrysum dasycephalum O.Hoffman		ASTERACEAE
Helichrysum eckloonis Bolus		ASTERACEAE
Helichrysum epapposum Bolus		ASTERACEAE
Helichrysum glomeratum Klatt		ASTERACEAE
Helichrysum griseum Sond.		ASTERACEAE
Helichrysum herbaceum (Andrews) Sweet		ASTERACEAE
Helichrysum infaustum Wood & Evans		ASTERACEAE
Helichrysum krebsianum Less.		ASTERACEAE
Helichrysum krookii Moeser		ASTERACEAE
Helichrysum microcephalum A.Cunn. ex DC		ASTERACEAE
Helichrysum miconifolium DC.		ASTERACEAE
Helichrysum montanum DC.		ASTERACEAE
Helichrysum nannum Klatt		ASTERACEAE
Helichrysum nudifolium (L.) Less.		ASTERACEAE
Helichrysum odoratissimum (L.) Sweet		ASTERACEAE

Helichrysum oreophilum Klatt		ASTERACEAE
Helichrysum pilosellum (L.f.) Less.		ASTERACEAE
Helichrysum rosum (Berg) Less		ASTERACEAE
Helichrysum rugulosum Less		ASTERACEAE
Helichrysum spiralepis Hilliard & B.L.Burtt		ASTERACEAE
Helichrysum subglomeratum Lees.		ASTERACEAE
Helichrysum trilineatum DC.		ASTERACEAE
Helichrysum pallidum DC.		ASTERACEAE
Helichrysum vernum Hilliard		ASTERACEAE
Helictotrichon turgidulum (Stapf) Schweick.		POACEAE
Heliophila rigidiuscula Sond.		BRASSICACEAE
Hermannia erodioides Kuntze		MALVACEAE
Hermannia gerrardii Harv.		MALVACEAE
Hermannia woodii Schinz		MALVACEAE
Hemarthria altissima (Poir.) Stapf & C.E.Hubb.		POACEAE
Heteropogon contortus (L.) Roem. & Schult.		POACEAE
Hibiscus aethiopicus L.		MALVACEAE
Hibiscus malacospermus (Turcz.) E. Mey. ex Harv.		MALVACEAE
Hibiscus trionum L.		MALVACEAE
Hirpicium armerioides (D.C) Roessler		ASTERACEAE
Hyparrhenia hirta (L.) Stapf		POACEAE
Hypericum aethiopicum Thunb.subsp. Aethiopicum		HYPERICACEAE
Hypericum lalandii Choisy		HYPERICACEAE
Hypochaeris radicata L.		ASTERACEAE
Hypoxis accuminata Baker		HYPOXIDACEAE
Hypoxis angustifolia Lam.		HYPOXIDACEAE
Hypoxis argentea Harv. ex Baker		HYPOXIDACEAE
Hypoxis costata Baker		HYPOXIDACEAE
Hypoxis filiformis Baker		HYPOXIDACEAE
Hypoxis rigidula Baker		HYPOXIDACEAE
Indigofera hedyantha Eckyl &zeyl		FABACEAE
Indigofera sp.		FABACEAE
Ipomoea crassipes Hook.		CONVOLVULACEAE
Ipomoea pellita Hallier		CONVOLVULACEAE
Isolepis angelica B.L.Burtt		CYPERACEAE
Isoetes transvaalensis Jermy & Schelpe		CYPERACEAE
Junceus effusus L.		CYPERACEAE
Koeleria capensis Nees		POACEAE
Kohautia amatymbica Eckl. & Zeyh.		RUBIACEAE
Lactuca indica L.		ASTERACEAE
Lampranthus stipulaceus N.E. Br.		AIZOACEAE
Ledebouria sp.		HYACINTHACEAE

Ledebouria cooperi (Hookef)Sessop		HYACINTHACEAE
Ledebouria ovatifolia (Baker) Jessop		HYACINTHACEAE
Ledebouria zebrina (Baker) S.Venter		HYACINTHACEAE
Leonotis intermedia Lindl.	*Leonotis ocymifolia var. raineriana* (Vis.) Iwarsson	LAMIACEAE
Lepidium africanum (Burm.f.) DC.		BRASSICACEAE
Leucosidea sericea Eckl. & Zeyh.		ROSACEAE
Limeum viscosum subsp. Viscosum varoglomeratum		MOLLUGINACEAE
Linum sp.		LINACEAE
Linum thunbergii Eckl & Zehyl		LINACEAE
Lobelia erinus L.		LOBELIACEAE
Lobelia flaccida(Presl.A.) D.C.		LOBELIACEAE
Lophacme digitata Stapf		POACEAE
Lotononis calycina (E.Mey.) Benth.		FABACEAE
Lotononis eriantha Benth.		FABACEAE
Lotononis foliosa Bolus		FABACEAE
Lotononis lotononoides (Scott-Elliot) B.-E.van Wyk		FABACEAE
Lotononis procumbens Bolus		FABACEAE
Mariscus congestus (Boeckeler) Kuntze		CYPERACEAE
Melolobium alpinum Eckl. & Zeyh.		FABACEAE
Melinis repens (Willd.) Zizka		POACEAE
Merxmuellera disticha (Nees) Conert	*Tenaxia disticha* (Nees) N.P.Barker & H.P.Linder	POACEAE
Melinis nerviglumis (Franch.) Zizka		POACEAE
Microchloa caffra Nees		POACEAE
Monocymbium ceresiforme (Nees) Stapf		POACEAE
Monopsis decipiens (Sond.) Thulin		LOBELIACEAE
Monsonia angustifolia E. Mey. ex A. Rich.		GERANIACEAE
Monsonia attenuata Harv.		GERANIACEAE
Muraltia saxicola Chodat		POLYGALACEAE
Nicandra physalodes (L.) Gaertn.		SOLANACEAE
Nolletia rarifolia (Turcz.) Steetz		ASTERACEAE
Oenothera indecora Cambess.		ONAGRACEAE
Oenothera rosea L'Hér. ex Aiton		ONAGRACEAE
Ornithogalum graminifolium Thunb.		HYACINTHACEAE
Ornithogalum juncifolium Jacq.		HYACINTHACEAE
Orchid sp.		ORCHIDACEAE
Osteospermum caulescens Harv.	*Dimorphotheca caulescens*	ASTERACEAE
Othonna natalensis Sch.Bip.		ASTERACEAE
Oxalis corniculata L.		OXALIDACEAE
Oxalis obliquifolia Steud. ex A.Rich.		OXALIDACEAE
Pachycarpus scaber (Harv.) N.E. Br.		APOCYNACEAE
Panicum ecklonii Nees		POACEAE
Panicum natalense Hochst.		POACEAE

Paspalum notatum Flüggé		POACEAE
Paspalum scrobiculatum L.		POACEAE
Paspalum urvillei Steud.		POACEAE
Pelargonium alchemoloides (L.) Aiton		GERANIACEAE
Pelargonium luridum (Andrews) Sweet		GERANIACEAE
Pennisetum clandestinum Hochst. ex Chiov.		POACEAE
Pentanisia angustifolia (Hochst.) Hochst.		RUBIACEAE
Pentaschistis oreodoxa Schweick.		POACEAE
Plantago major L.		PLANTAGINACEAE
Plantago virginicus L.		PLANTAGINACEAE
Polygala amatymbica Eckl. & Zeyh.		POLYGALACEAE
Polygala gricilenta Burtt Davy		POLYGALACEAE
Polygala hottentotta C.Presl		POLYGALACEAE
Polygala ohlendorfiana Eckl. & Zeyh.		POLYGALACEAE
Polygala rehmanii Chodat		POLYGALACEAE
Psammotropha mucronata (Thunb.) Fenzl		MOLLUGINACEAE
Psammotropha myriantha Sond.		MOLLUGINACEAE
Pygmaeothamnus chamaedendrum (Kuntze) Robyns		RUBIACEAE
Ranunculus meyeri Harv.		RANUNCULACEAE
Rendlia altera (Rendle) Chiov.	*Microchloa altera* (Rendle) Stapf	POACEAE
Rhodohypoxis baurii (Baker) Nel		HYPOXIDACEAE
Rhus dentata Thunb.		ANACARDIACEAE
Rhynchosia totta (Thunb.) DC.		FABACEAE
Richardia brasiliensis Gomes		ASTERACEAE
Ruellia glomerata A.D.C		ACANTHACEAE
Satyrium longicauda Lindl.		ORCHIDACEAE
Satyrium sp.		ORCHIDACEAE
Scabiosa columbaria L.		DIPSACACEAE
Schistostephium crataegifolium (DC.) Fenzl ex Harv.		ASTERACEAE
Schoenoxiphium filiforme (Nees) Steud		CYPERACEAE
Schoenoxiphium sparteum C.B CL		CYPERACEAE
Scilla nervosa (Burch.) J.P.Jessop	*Schizocarphus nervosus* (Burch.) van der Merwe	HYACINTHACEAE
Scleria bulbifera Hochst. ex A.Rich.		CYPERACEAE
Scleria distans Poir.		CYPERACEAE
Scleria dregeana Kunth	*Scleria holcoides* Kunth	CYPERACEAE
Sebaea grandis (E.Mey.) Steud.		GENTIANACEAE
Sebaea natalensis Schinz		GENTIANACEAE
Sebaea sedoides Gilg		GENTIANACEAE
Selaginella sp.		SELAGINELLACEAE
Selago monticola J.M. Wood & M.S. Evans		ASTERACEAE
Senecio sp.		ASTERACEAE
Senecio affinis D.C		ASTERACEAE
Senecio adnatus DC.		ASTERACEAE

Senecio arabidifolius O.Hoffm.		ASTERACEAE
Senecio barbatus DC.		ASTERACEAE
Senecio bupleuroides DC.		ASTERACEAE
Senecio cathcartensis Hilliard & Burtt		ASTERACEAE
Senecio coronatus Harv.		ASTERACEAE
Senecio erubescens Ait var.dichotomous D.C		ASTERACEAE
Senecio glaberrimus DC.		ASTERACEAE
Senecio harveianus MacOwan.	*Senecio inaequidens DC.*	ASTERACEAE
Senecio hypochionaeus Boiss.		ASTERACEAE
Senecio hypochoerideus D.C		ASTERACEAE
Senecio inornatus DC.		ASTERACEAE
Senecio macrocephalus D.C		ASTERACEAE
Senecio rhomboideus Harv.		ASTERACEAE
Senecio scitus Hutch. & Burtt Davy		ASTERACEAE
Senecio sandersonii Harv.		ASTERACEAE
Senecio speciosus Willd.		ASTERACEAE
Sida rhombifolia Linn.		MALVACEAE
Solanum capense L.		SOLANACEAE
Sonchus dregeanus Harv.		ASTERACEAE
Sonchus nannus Sond. ex Harv.		ASTERACEAE
Sopubia cana Harv.		OROBANCHACEAE
Sporobolus africanus (Poir.) Robyns & Tournay		POACEAE
Sporobolus centrifugus Nees		POACEAE
Stiburus alopecuroides (Hack.) Stapf	*Eragrostis tincta* S.M.Phillips	POACEAE
Stoebe vulgaris Levyns	*Seriphium plumosum* L.	ASTERACEAE
Striga bilibiata (Thunb.) Kuntze		OROBANCHACEAE
Sutera neglecta (J.M.Wood & M.S.Evans) Hiern	*Chaenostoma neglectum* J.M.Wood & M.S.Evans	SCROPHULARIACEAE
Taraxacum officinale (L.) Weber ex F.H.Wigg.	*Taraxacum campylodes* G.E.Haglund	ASTERACEAE
Tephrosia polystachya E.Mey.		FABACEAE
Tephrosia semiglabra Sond.		FABACEAE
Tetraria cuspidata (Rottb.) C.B.Clarke		CYPERACEAE
Themeda triandra Forssk.		POACEAE
Thesium costatum A.W.Hill		SANTALACEAE
Thesium goetzianum Engl.		SANTALACEAE
Thesium pallidum A. DC.		SANTALACEAE
Thesium zehyeri A. DC		SANTALACEAE
Thunbergia venosa C.B. Clarke		ACANTHACEAE
Tolpis capensis (L.) Sch.Bip.		ASTERACEAE
Trachyandra asperata Kunth		ASPHODELACEAE
Trachyandra saltii (Baker) Oberm. var. saltii		ASPHODELACEAE
Trachypogon spicatus (L.f.) Kuntze		POACEAE

Trichoneura grandiglumis (Nees) Ekman		POACEAE
Tristachya leucothrix Trin. ex Nees		POACEAE
Verbena bonariensis L.		VERBENACEAE
Vernonia capensis (Houtt.) Druce	*Hilliardiella pinifolia* (Less.) H.Rob.	ASTERACEAE
Vernonia natalensis Oliv. & Hiern		ASTERACEAE
Wahlenbergia fasciculata Brehmer		CAMPANULACEAE
Wahlenbergia huttonii (Sond.) Thulin.		CAMPANULACEAE
Wahlenbergia sp.		CAMPANULACEAE
Wahlenbergia squamifolia V.Brehm		CAMPANULACEAE
Wahlenbergia virgata Engler.		CAMPANULACEAE
Watsonia lepida N.E.Br.		IRIDACEAE
Xysmalobium parviflorum Harvey ex S. Elliot		APOCYNACEAE
Zaluzianskya microsiphon K. Schum.		SCROPHULARIACEAE

APPENDIX B: Laboratory Soil Analysis

ANALYTICAL METHODS USED BY THE SOIL FERTILITY AND ANALYTICAL SERVICES SECTION

By

A.D. Manson & V.G. Roberts

KwaZulu-Natal Department of Agriculture and Environmental Affairs

October 2000

KZN AGRI-REPORT No. N/A/2001/04

ANALYTICAL METHODS USED BY THE

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KZN Agri-Report No. N/A/2001/4, October 2000

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The Soil Fertility and Analytical Services Section of the KwaZulu-Natal Department of Agriculture and Environmental Affairs performs a variety of analyses on soil, plant tissue, and compost, manure and water samples. This document gives brief descriptions of the analyses most commonly performed.

Soil analysis

The Soil Fertility Laboratory routinely performs the following analyses as part of the Department's Fertilizer Advisory Service, using the rapid procedures described by Hunter (1975) and Farina (1981): Ambic-2-extractable P, K, Cu, Mn and Zn, KCl-extractable Ca, Mg and acidity, and pH (KCl); NIRS-estimates of organic carbon and clay content are also done routinely. Other soil analyses done by the Section are: EC, Ca, Mg, K, and Na in the saturation extract, ammonium-acetate extractable Na, pH (water), total carbon, nitrogen, and sulphur, Walkley-Black organic carbon, and particle size distribution. These methods are briefly described below.

Sample preparation: Soil Fertility Laboratory

Soil samples are air dried at room temperature; they are spread out in drying trays and air is forced over them. When dry, the samples are crushed between rubber belts on a soil crusher and passed through a 1-mm sieve. Material coarser than 1 mm that cannot be crushed (such as stones, gravel and concretions), is discarded.

Batch handling: Soil Fertility Laboratory

Samples are scooped into trays which each contain 11 PVC cups (capacity 70 mL); a tray is used for nine unknown samples, one standard soil sample (for quality control) and one blank. For operations such as dispensing and stirring, and for quality control, batches of three trays (27 samples, three unknowns, and three blanks) are used. Multiple dispensers and diluter/dispensers are used to dispense aliquots of extractant or reagent to three samples at a time.

Sample density

Soil samples are analysed on a volume rather than a mass basis. To enable the conversion of the results to a mass basis, the mass of a 10-mL scoop of a dried and milled sample is measured and the calculated sample density is reported.

pH (KCl)

10 mL of soil is scooped into sample cups. 25 mL of 1 *M* KCl solution is added and the suspension is stirred at 400 r.p.m. for 5 min using a multiple stirrer. The suspension is allowed to stand for about 30 minutes, and the pH is measured using a gel-filled combination glass electrode while stirring. De-ionised water is substituted for the 1 *M* KCl solution if pH (water) is required.

Extractable (1 M KCl) calcium, magnesium and acidity

2.5 mL of soil is scooped into sample cups. 25 mL of 1 *M* KCl solution is added and the suspension is stirred at 400 r.p.m. for 10 min using a multiple stirrer. The extracts are filtered using Whatman No.1 paper. 5 mL of the filtrate is diluted with 20 mL of 0.0356 *M* $SrCl_2$, and Ca and Mg determined by atomic absorption. To determine extractable acidity, 10 mL of the filtrate is diluted with 10 mL of de-ionised water containing 2-4 drops of phenolphthalein, and titrated with 0.005 *M* NaOH.

Extractable (Ambic-2) phosphorus, potassium, zinc, copper and manganese

The Ambic-2 extracting solution consists of 0.25 *M* NH_4CO_3 + 0.01 *M* Na_2EDTA + 0.01 *M* NH_4F + 0.05 g L^{-1} Superfloc (N100), adjusted to pH 8 with a concentrated ammonia solution. 25 mL of this solution is added to 2.5 mL soil, and the suspension is stirred at 400 r.p.m. for 10 min using a multiple stirrer. The extracts are filtered using Whatman No.1 paper. Phosphorus is determined on a 2 mL aliquot of filtrate using a modification of the Murphy and Riley (1962) molybdenum blue procedure (Hunter, 1974). Potassium is determined by atomic absorption on a 5 mL aliquot of the filtrate after dilution with 20 mL de-ionised water. Zinc, Cu and Mn are determined by atomic absorption on the remaining undiluted filtrate.

Effective CEC (ECEC) and Acid saturation

Effective CEC is calculated as the sum of KCl-extractable Ca, Mg, and acidity and Ambic-2 extractable K. Percent acid saturation of the ECEC is calculated as "extractable acidity" x 100 / (Ca + Mg + K + "extractable acidity").

Estimate of organic carbon by near-infrared spectroscopy

Organic carbon is estimated for all soil samples routinely analyzed in the Soil Fertility Laboratory, by near-infrared reflectance, using the air-dry, milled soil samples.

Estimate of clay content by near-infrared spectroscopy

Clay content is estimated for all soil samples routinely analyzed in the Soil Fertility Laboratory, using a combination of near-infrared reflectance using the air-dry, milled soil samples, and the measured sample density.

Total carbon and nitrogen in soil

Total C, N and S are analyzed by the Automated Dumas dry combustion method using a LECO CNS 2000 (Leco Corporation, Michigan, USA; Matejovic, 1996). Briefly, this method involves weighing samples into a ceramic crucible to which 0.5g of vanadium pentoxide is added as a combustion catalyst. The vanadium pentoxide also improves the recovery of sulphur. The crucible is introduced into a horizontal furnace, where the sample is burned in a stream of oxygen at 1350^{o}C. The gases produced are passed through two infra-red cells where the sulphur (as SO_2) and carbon (as CO_2) are determined. Nitrogen is determined (as N_2) in a thermal conductivity cell.

Organic carbon by the Walkley-Black method

This method is based on the Walkley-Black procedure (Allison, 1965) and measures the readily oxidizable organic carbon. The organic matter is oxidized by potassium dichromate in a sulphuric acid medium. The excess dichromate is determined by titration with standard ferrous sulphate solution.

Particle size distribution of soils and soil texture

Suspended clay and fine silt are determined after dispersion and sedimentation, and sand fractions are determined by sieving (Day, 1965). A 20 g soil sample (<2 mm) is treated with hydrogen peroxide to oxidise the organic matter. The sample is made up to 400 ml with de-ionized water and left overnight. The clear supernatant is siphoned off and the sample puddled. A further addition of de-ionized water is added, the sample stirred and left overnight. The clear supernatant is again siphoned off. Dispersing agents (NaOH and sodium hexametaphosphate) are added and the sample stirred on Hamilton Beach stirrers. The suspension is made up to 1 litre in a measuring cylinder and the clay (<0.002 mm) and fine silt (0.002-0.02 mm) fractions measured with a pipette after sedimentation. Fine silt plus clay is measured after 4-5 min (exact time depends on temperature) at 100 mm, and clay after 5-6 h at a depth of 75 mm. Sand fractions include very fine sand (0.05 - 0.10 mm), fine sand (0.10 - 0.25 mm), medium sand (0.25 - 0.50 mm) and coarse sand (0.50 - 2.0 mm) which are determined by sieving. Coarse silt (0.02-0.05 mm) is estimated by difference.

Determination of textural class by means of a textural triangle

Once the particle size distributions of the two soils are known, their textural class is determined from a diagram (textural triangle) defining particle size limits of the various textural classes (Soil Classification Working Group, 1991).

Salinity and sodicity

Saturation extracts of soils are analyzed for electrical conductivity (EC) using an EC meter, and Ca, Mg, Na and K are determined by atomic absorption. Sodium adsorption ratio (SAR) is determined by calculation. SAR = $[Na]/\{([Ca]+[Mg])/2\}^{0.5}$, where [Na], [Ca] and [Mg] are concentrations of the elements expressed in $mmol_c/L$.

Exchangeable Na

Exchangeable Na is determined by atomic absorption after extraction of a 2.5 mL soil sample with 25 mL of 1 *M* ammonium acetate (Soil Classification Working Group, 1991).

Ammonium - N (NH_4^+) and nitrate - N (NO_3^-)

Ammonium - N (NH_4^+) and nitrate - N (NO_3^-) in filtered extracts are measured by segmented flow analysis with a Perstorp Flow Solution III analyzer using the sodiumsalicylate-sodium nitropusside-hypochlorite method for NH_4^+ - N (Perstorp Analytical, 1993) and the sulphanilamide-naphthyl-ethylenediamine method for NO_3^- - N plus NO_2 - N after having reduced nitrate to nitrite with copperized cadmium wire (Willis & Gentry, 1987).

Plant tissue, manure, and compost analysis

Plant material, manures, and compost samples are analyzed using the following procedures. The batch-handling procedures are similar to those used for the soil analyses described above. Samples are dried at 75EC, and milled to pass through a 0.84 mm sieve. Subsamples are then dry ashed at 450EC overnight and taken up in 1 *M* HCl. The P concentration is determined colorimetrically by the same method used for soil extracts, and K, Na, Ca, Mg, Cu, Mn, and Zn are determined by atomic absorption. New development:- P,K,Al,Ca,Mg,Na,Cu,Mn and Zn is determine on ICP. Samples for B analysis are ashed separately, and B is determined photometrically by the azomethine H method (Gaines & Mitchell, 1979).

Nitrogen is determined using one of the following methods:

- before ashing, using near-infrared reflectance (Eckard *et al*., 1988);
- after macro-Kjeldahl digestion, by bubble-segmented flow analysis using the colorimetric phenol-hypochlorite method (de Figueredo & Thurtell, 1998); or

- by the Automated Dumas dry combustion method using a LECO CNS 2000 (Leco Corporation, Michigan, USA; Matejovic, 1996), described above (***Total C and N in soil***).

Water analysis

Water samples are analyzed to determine their suitability for irrigation, hydroponics, and for use with herbicide and pesticide sprays. Electrical conductivity (EC) is measured using an EC meter, and Ca, Mg, Na, K, Fe and Mn are determined by atomic absorption. Sodium adsorption ratio (SAR) is determined by calculation. SAR = $[Na]/\{([Ca]+[Mg])/2]^{0.5}$, where [Na], [Ca] and [Mg] are concentrations of the elements expressed in $mmol_c/L$.

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