



Effects of bush encroachment control in a communal managed area in the Taung region, North West Province, South Africa

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DECLARATION

I, Reamogetswe Olebogeng Mokgosi (21003149), hereby declare that the dissertation titled: **Effects of bush encroachment control in a communal managed area in the Taung region, North West Province, South Africa**, is my own work and that it has not previously been submitted for a degree qualification to another university.

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This thesis has been submitted with my approval as a university supervisor and I certify that the requirements for the applicable M.Sc degree rules and regulations have been fulfilled.

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Abstract

The communally managed Taung rangelands are degraded because of bush encroachment. Bush encroachment is defined as a natural continuous retrogressive ecological succession, resulting in the increase of both alien and indigenous encroacher woody species and a reduction in grass species composition. This in turn result to changes in soil chemical and physical properties. The knowledge of the interaction between bush encroachment, land-use and soil conditions is essential to sustainably manage these areas.

More than 80 % of the respondents in the Taung area owns cattle. To mitigate poverty stress; many pastoralists in the Taung area resorted to high stocking rates, leading to high grazing pressures locally and thereby, led to bush encroachment.

The Working for Water (WfW) programme identified the need to implement both mechanical and chemical bush control strategies within the Taung area. Eight study sites were selected for this study. Each of the selected sites had a control and an uncontrolled (benchmark) site. The prominent woody encroacher species within these rangelands were *Senegalia mellifera*, *Vachellia tortilis*, *V. karroo* and *Tarchonanthus camphoratus*. This posed a threat towards the water resources in Taung communal communities and their economic status. Soil samples were collected and analysed for soil chemical properties such as soil pH, carbon (C) and nitrogen (N) concentrations, C: N ratio, soil magnesium (Mg) and exchangeable magnesium content (Mg^{2+}), soil phosphorus (P) and sodium (Na) concentrations, soil calcium (Ca) content, soil exchangeable (Ca^{2+}) concentration, soil CEC and EC values and the percentage base saturation. The results revealed that, soil pH and carbon concentrations were slightly higher in the uncontrolled sites as compared to the controlled sites. Soil Ca^{2+} , Mg^{2+} and K concentrations and CEC values were higher in controlled sites as compared to the uncontrolled sites. P concentration, N availability and C: N ratios were limited in both the controlled and uncontrolled sites. EC values varied between the controlled and uncontrolled sites.

Keywords: Bush encroachment, carbon and nitrogen concentrations, electrical conductivity (EC), Magnesium (Mg), overgrazing, soil pH, soil chemical properties, Taung communal area.

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Glossary of Abbreviations

Ca – Calcium

Ca²⁺ - Exchangeable calcium

CEC – Cation-Exchange-Capacity

EC – Electrical Conductivity

GDP – Gross Domestic Product

LSU - The equivalent of one head of cattle with a body weight of 450 kg and gaining 500 g per day

Mg – Magnesium

Mg²⁺ - Exchangeable magnesium

Na – Sodium

NWP – North West Province

NWU – North-West University

P – Phosphorus

PES – Payment for Ecosystem Services

WfW – Working for Water

CHAPTER 1

INTRODUCTION

1.1 Background of the study

Rangelands are globally important and cover about 51% of the Earth's land surface (Bond & Midgley, 2000; Sankaran *et al.*, 2005) and approximately 33% of South Africa (Abebe, 2000; Oba *et al.*, 2000; Wiegand *et al.*, 2006; Mucina & Rutherford, 2006; Wang *et al.*, 2010; Mussa *et al.*, 2016). Savanna rangelands make up the largest managed land-use area and are estimated to cover about 25% of the Earth's land surface (Asner *et al.*, 2004; Liebig *et al.*, 2006; Angassa & Oba, 2008; Archer, 2010; Kgosisikoma & Mogotsi, 2013). These rangelands are often used as grazing lands for livestock and/or game due to the grass cover.

The savanna biome is characterised by scattered trees and a herbaceous layer which include grass species (Knoop & Walker, 1985; Sankaran *et al.*, 2005; Wiegand *et al.*, 2006; Mucina & Rutherford, 2006). Savanna ecosystems are important for both maintaining environmental services such as biodiversity conservation and as a source of sustainable livelihood systems, especially in communal areas (O'Connor, 2005; Eriksen & Watson, 2009; Muhumuza & Byarugaba, 2009; Kgosisikoma & Mogotsi, 2013). Plant species richness of the savanna biome in southern Africa is relatively high compared to the other southern African biomes, with only the fynbos biomes' plant species richness being higher. However, per unit area, the plant diversity of savannas is lower compared to the fynbos, forest, grassland or succulent karoo biomes (Scholes, 1997; Harmse, 2013). The savanna biome contains 3-14 species/m² and 40-100 species per 0.1 ha, not significantly different to the rest of southern African biomes (Scholes, 1997; Harmse, 2013).

Savannas are considered to be variable in terms of vegetation structure, composition and geographic distribution. Ecosystems in savannas are mainly determined by primary (climate and soil properties) and secondary parameters (herbivory and fire) (Scholes & Archer, 1997; Scholes *et al.*, 2002; Sankaran *et al.*, 2005; Wiegand *et al.*, 2006; Sankaran & Anderson, 2009; Higgins *et al.*, 2010). The main functional distinction between the savannas of southern Africa is the broad-and fine-leaved woody components. Fine-leaved savannas occur in nutrient-rich, arid environments and the broad-leaved savannas in nutrient-poor,

semi-arid environments. An exception is the broad-leaved *Colophospermum mopane* savannas which forms part of fertile arid savannas (Scholes, 1997; Harmse, 2013). The fine-leaved savannas contain relatively palatable grass species, attracting high ungulate densities, thus preventing fires, as the high grass biomass production, needed for fires, is constantly being removed. Broad-leaved savannas can be distinguished from fine-leaved savannas by having less palatable grass species, fewer ungulates and more fires (Scholes *et al.*, 2002; Harmse, 2013).

Although the high variability in the savanna ecosystems, most studies undertaken for comparative assessment of the effects of management on savanna ecosystem dynamics, are site specific (Dahlberg, 2000; Asner *et al.*, 2004; Smet & Ward, 2005; Tefera *et al.*, 2010; Kgosikoma *et al.*, 2012). By considering the interactions between natural resources, rainfall and soil types and including human-induced factors (e.g. cultural and land management practices), the knowledge of how a specific factor influences vegetation conditions is improved (Scholes & Archer, 1997; Hoffman & Ashwell, 2001; Groffman *et al.*, 2007).

The vegetation structure and composition of savanna rangelands have sustained influential changes overtime, and ecosystem deterioration resulting from bush encroachment, has been recorded to be one of the modern expressions of those dynamic changes (Bester & Reed, 1997; Briske *et al.*, 2003; Reynolds *et al.*, 2007; Eldridge *et al.*, 2011a; Belayneh & Tessema, 2017). Changes in vegetation structure and composition in the savannas influence the sustainability of livestock production and ecosystem function (Tainton, 1999; Sankaran *et al.*, 2005; Kgosikoma *et al.*, 2012; O'Connor *et al.*, 2014; Mussa *et al.*, 2016). Therefore, rangeland degradation, as a result of bush encroachment, appears to threaten the ecosystem integrity of these delicate ecosystems and may diminish the grazing capacity of a rangeland by as much as 90% (Archer *et al.*, 1995; Du Preez & Snyman, 2003; De Klerk, 2004; Sankaran *et al.*, 2005; UNEP, 2009; Yanoff & Muldavin, 2008; Kahumba, 2010).

The drivers of changes in savanna ecosystems are highly complex and debated (Smit, 2005; Vetter, 2005; Van Auken, 2009; Daryanto, 2013; O'Connor *et al.*, 2014; Belayneh & Tessema, 2017). This is due to the impacts of land-use, frequency of wild fires and climatic conditions (such as rainfall variability, global warming and the increase of CO₂ levels), and especially the increase in stocking numbers of both domestic and native animals

over large scales of rangelands. Several scientists have, however, reported that all the aspects have contributed to a reduction in grass biomass production (Archer *et al.*, 1995; Smit, 2005; Kgosikoma *et al.*, 2012; Daryanto, 2013; O'Connor *et al.*, 2014). It is, however, challenging to qualify an individual factor or a set of factors as the cause for bush encroachment clearly because most factors are spatially related and scale dependent (both over space and time) (Walker, 1971; Archer *et al.*, 1995; Scholes & Archer, 1997; Van Auken, 2000; Briske *et al.*, 2003; Ward, 2005; Van Auken, 2009; Archer, 2010; Tessema *et al.*, 2012; Belayneh & Tessema, 2017).

Savannas are degraded as a result of inappropriate rangeland management practices (for example, inappropriate use of fires, mismanagement/over-utilisation of natural resources, climate change and the elimination of mega-herbivores) (Smit, 2004; Kahumba, 2010; Franci, 2011; Kgosikoma *et al.*, 2012; Daryanto, 2013; Mohammed, 2013). Poor management of rangelands may lead to the weakening of the grass sward, through over-utilisation and subsequent replacement of palatable grasses by woody plants, also called “bush encroachment” (Kahumba, 2010). Degraded savannas are dominated by spiny woody plants that out-compete grasses, causing a decrease in grass species frequency and cover and density increase of woody species (Van Vegten, 1984; Abule *et al.*, 2007; Angassa & Oba, 2008; Angassa *et al.*, 2012).

Bush encroachment, also known as the increase in alien or indigenous trees or shrub densities, is a global phenomenon. The increase in the density and cover of woody species may be indigenous or alien species, particularly in grasslands and savanna regions (Van Auken, 2009; Kgosikoma *et al.*, 2012; Daryanto, 2013; Eldridge & Soliveres, 2015). Kyalangalilwa *et al.* (2013) and O'Connor *et al.* (2014) defined bush encroachment as a directional increase in the cover of indigenous woody species (generic use of *Acacia* sp., for African species – now classified as *Vachellia* sp. and *Senegalia* sp.) at the expense of the herbaceous component. According to Belayneh and Tessema (2017), bush encroachment is a term used in association with other frequently-used terms such as bush thickening (Van Auken, 2009), woody plant re-growth (Eldridge *et al.*, 2013), invasion of woody weeds (Ayres *et al.*, 2001), xerification (Archer, 2010) and invasion of shrubs (Noble & Rodolfo, 1997).

Bush encroachment is common in savannas and has surfaced as one of the top three recognised rangeland problems across 25% of the South African magisterial districts. It is expected that its impact may escalate over time, depending on the Natural Resources Management (NRM) strategies applied in certain areas (Hoffman *et al.*, 1999; O' Connor *et al.*, 2012; Higgins & Scheiter, 2013; Moncrieff *et al.*, 2014; O'Connor *et al.*, 2014). For instance, bush encroachment has affected the agricultural productivity of 10 to 20 million ha in South Africa (Ward, 2005) and 37 000 km² in Botswana in 1994 (Moleele *et al.*, 2002), thereby threatening the feasible production of livestock systems and human well-being (Kgosikoma & Mogotsi, 2013). After evaluating the recent problem of bush encroachment in the North West Province, Northern Cape, Eastern Cape and Limpopo Provinces of South Africa, it was reported that 42% of the rangelands were already affected by bush encroachment (Hoffman & Ashwell, 2001; Harmse, 2013). According to Ward (2005), 10 to 20 million ha of rangelands in South Africa have been reported to experience a decline in grazing capacity and biodiversity due to bush encroachment. The invasive woody plants involve both indigenous and alien species (Brown & Archer, 1989; Hoffman & O'Connor, 1999; Oba *et al.*, 2000; Smit, 2001; Ward, 2005; Gemedo *et al.*, 2006a; O'Connor *et al.*, 2014; Belayneh & Tessema, 2017).

Bush encroachment has a negative effect on the land user's economic viability, especially beef ranches and has cost Namibia an annual loss of approximately N\$700 million in agricultural productivity (De Klerk, 2004). This escalates rural privation and this reduces food stability in rural communities, which depend extensively on livestock farming for their livelihoods (Moyo *et al.*, 1993; Kahumba, 2010).

1.2 Importance of ecological restoration in savanna rangelands

The combined effect of management, soil and climatic factors on rangeland degradation has led to reduced floral biodiversity, contributing to a reduction in environmental quality (Jama & Zeila, 2005). The restoration of degraded rangeland remains a challenge. Scientific studies have demonstrated that damaged vegetation can recover in a relatively short time when protected from grazing impacts (Yayneshet *et al.*, 2009). Rangeland rehabilitation and restoration measures take various forms, which include re-seeding or allowing the progression of natural regeneration of seed or propagules in the soil, as well as general soil and water conservation measures (Mussa *et al.*, 2016). For rehabilitation to be effective and

successful, it should target the underlying causes of degradation and reverse the degradation process (Li *et al.*, 2011). The introduction of appropriate rangeland management legislation together with effective restoration and rehabilitation practices contribute significantly to halting and reversing bush encroachment and improving the carrying capacity of rangelands (AU-IBAR, 2012). Bush encroachment control and rehabilitation practices are often very expensive, especially in widespread application. The prevention of rangeland degradation is preferred over rehabilitation, not only in terms of cost, but also due to the progressive and reinforcing nature of degradation once it has crossed a threshold and reached irreversible effects (Hobbs & Norton, 1996; Puigdefabregas, 1998; Aronson *et al.*, 2007; Richardson *et al.*, 2007; Kellner, 2008; Bullock *et al.*, 2011; Mussa *et al.*, 2016).

1.3 Rangeland restoration types

Restoration requires an in-depth understanding of how the ecosystem works and what the causes for degradation are (Blench & Florian, 1999; Aronson *et al.*, 2007). In general, there are two types of restoration, i.e. (1) passive restoration (restoration of degraded habitats by ceasing anthropogenic perturbations that are causing degradation, such as the adaptation of grazing practices to decrease the grazing pressure) and (2) active restoration (biotic manipulation that is practiced by reintroduction of animal or plant species that have been extirpated from an area). These restoration practices occur mainly through re-vegetation or re-seeding practices and the application of some soil cultivation practice (Harmse, 2013; Mussa *et al.*, 2016).

1.3.1 Passive restoration

Passive restoration interventions in semi-arid savannas are applied in systems with a high resilience and limited functional damage (Visser *et al.*, 2007; Kellner, 2008). These interventions include removing stresses such as heavy grazing by implementing rotational grazing management (withdrawing livestock) to allow vegetation with longer periods of time to recover (Whisenant, 1995; Milton & Dean, 1995; Snyman, 1999; Tainton *et al.*, 1999; Curtin, 2002; Müller *et al.*, 2007; Scholes, 2009). In instances of a limited occurrence of rangeland degradation, these systems are capable of self-recovery due to the available seed remaining in the soil-seed bank (Kellner, 2008). If the land user has the knowledge to implement sustainable rangeland management practices, low input costs would be required and the practices can be affected with relative ease. The impacts from hooves of livestock

on the soil surface and uniform utilisation of vegetation are regarded as positive tools for rangeland restoration (Briske *et al.*, 2011). If the plants are underutilised and fewer disturbances occur, a decline of soil conditions could result, which may lead to competition between plant species (Briske *et al.*, 2011).

1.3.2 Active restoration

Active restoration interventions are applied in savanna systems that lost its function and structure with low-resilience, high woody plant densities and a decrease in grass cover and density (Aronson *et al.*, 2007; Kellner, 2008). To facilitate improved grass establishment and reduce woody densities, active restoration interventions will include shrub clearing (mechanical or chemical), veld burning, brush packing, fertilisation and/or cultivation (Milton & Dean, 1995; Visser *et al.*, 2004; 2007; Scholes, 2009; Teague *et al.*, 2010). The soil-seed bank in these degraded rangelands is usually depleted and will require re-vegetation through re-seeding practices to ensure re-establishment of perennial grass species (Milton, 1994; Van den Berg & Kellner, 2005; Kellner, 2008; Scholes, 2009; Harmse, 2013).

1.4 Restoration techniques

Rangeland restoration techniques include re-vegetation (re-seeding) of rangelands, prescribed fires, bush encroachment control, rangeland enclosures and grazing management (Mussa *et al.*, 2016).

1.4.1 Re-vegetation of degraded rangelands

In arid and semi-arid areas, prolonged heavy grazing pressures combined with the recurrent drought have changed large areas of rangelands to bare soil (Mussa *et al.*, 2016). Rangelands in such situations are prone to wind and soil erosion, which in turn lead to a decline in the fertile soil-seed bank (Tessema *et al.*, 2011). In such extremely degraded rangelands, where soil-seed banks have been depleted or in a situation where the relative proportion desirable species have fallen below critical levels (less than 10-15%), the degradation of rangelands can be reversed through re-seeding/re-vegetation (Abule & Alemayehu, 2015). Re-seeding technology has been used successfully as a means of rehabilitating and restoring degraded rangelands (Musimba *et al.*, 2004; Mussa *et al.*, 2016). This practice is not common in

communal managed rangelands because of high livestock numbers and areas not protected from grazing after re-vegetation (Van den Berg & Kellner, 2005; Opiyo *et al.*, 2011). A study conducted in southeast Ethiopia showed the possibility of restoring degraded rangeland with re-seeding of Rhodes grass (*Chloris gayana* Kunth) with simple tillage and manure application (Mussa *et al.*, 2016).

Re-seeding involves collecting seeds from local ecosystems or buying it from a seed merchant and then sowing it on bare ground which has been cultivated to improve the soil moisture regime. Re-seeding techniques also include soil preparations, making use of fertilizers and continued maintenance, and also encouraging pastoralists to gather seeds of plants in the growing season to plant when needed (Blench & Florian, 1999; Mussa *et al.*, 2016). Native grasses and local ecotypes are well suited to the harsh environment of semi-arid areas (Van den Berg & Kellner, 2005). Naturally occurring grasses not only provide necessary habitat for many indigenous animals, but are also able to provide a suitable pasture base for animal production (Oba & Kotile, 2001; Mussa *et al.*, 2016).

1.4.2 Prescribed fire

In African savannas, fire is a natural management tool that could have considerable impacts on ecosystem structure and functioning (Higgins *et al.*, 2000). The most noticeable result of fire is the eradication of mature, dead vegetation, which is replaced by young re-growth, i.e. green-flush re-growth (Mussa *et al.*, 2016). Herbivores are attracted to this re-growth and feed on the post-fire re-growth of woody and herbaceous plants (Higgins *et al.*, 2000; Frank *et al.*, 2003; Mussa *et al.*, 2016). Various scientific studies have demonstrated that post-burn savanna plants have a higher above-ground nutrient concentration compared to unburned plants during the post-fire growth season (Tainton, 1999; Higgins *et al.*, 2000; Frank *et al.*, 2003).

Fire is generally a useful mechanism for controlling woody plant densities, eradicating dead biomass, promoting grass re-growth and addressing pest control (Herlocker, 1999). It was illustrated by Gebru *et al.* (2007) that fire application in the Borana rangelands in southern Ethiopia increased the cover of *Themeda triandra* between 18% and 40% and that the basal cover and the amount of bare ground was accordingly reduced after burning. Bond and Keeley (2005), Gebru *et al.* (2007), Sankaran *et al.* (2008), Archibald *et al.* (2010), O'Connor

et al. (2014), Joubert *et al.* (2012), Rohde and Hoffman (2012), O'Connor *et al.* (2014) and Chirwa *et al.* (2015) have all suggested that, if prescribed fire is implemented properly, and used in conjunction with other appropriate range management practices (reduction of stocking rates and adequate rainfall regime), prescribed fire can be used to reduce bush encroachment and increase the forage production and quality for grazing animals. Fire has an important ecological role in shaping the formation and arrangement of rangeland vegetation (Angassa & Oba, 2008). Incorrect management practices and absence of fire would most likely lead to an increase of woody species density that could lead to the area becoming a dense woodland (Tainton, 1999; Hoffman & Ashwell, 2001; Angassa & Oba 2008; Stephen *et al.*, 2009; Ward *et al.*, 2014; Mussa *et al.*, 2016).

1.4.3 Bush encroachment control

Control methods intent to reduce bush encroachment alter rangeland vegetation from a state being dominated by woody plants to that of herbaceous vegetation. The control of the bush is aimed at creating suitable habitat for increased grass production which satisfies grazers (Angassa & Oba, 2008; Mussa *et al.*, 2016). Thus, a decrease in woody species leads to an increase in forage production (Tainton, 1999; Hoffmann & Ashwell, 2001; Angassa & Oba, 2008; Daryanto, 2013). Different types of bush encroachment control methods are available (Barac, 2003). Control methods can be divided into mechanical, biological, chemical or combined methods (Barac, 2003; Lesilo *et al.*, 2013; Belachew & Tessema, 2015). However, to implement these methods, public awareness has to be developed and a participatory approach to control the invasive woody species should be adopted where it becomes a problem for sustainable rangeland management (Patel, 2011; Mussa *et al.*, 2016).

Angassa (2007) argued that the implementation of bush clearing methods are valuable in the management and recovery of the rangelands, following prescribed burning. Scientific evidence indicates that there are positive and negative feedback loops between grass and soil following bush removal (Ward, 2005; Abule *et al.*, 2007; Bikila *et al.*, 2014; Buyer *et al.*, 2016). Strong evidence regarding the role of plant–soil feedback in driving plant community composition exists (Pendergast *et al.*, 2013). The changes in soil chemistry and microbial communities following bush removal could promote either grass establishment (positive feedback) or bush re-growth and encroachment (negative feedback) (Buyer *et al.*, 2016; Mussa *et al.*, 2016). According to Perkins and Nowak (2013) and Buyer

et al. (2016), soil nutrients as well as soil microbial communities have shown to be involved in plant–soil feedback systems. Grass growth, following woody plant clearing, may be improved by nutrient availability and facilitation from decomposing tree residues (Buyer *et al.*, 2016). According to Tainton (1999), Smit (2005) and Samuel (2009), rangelands that are encroached by *Vachellia* species improve the under-storey vegetation production and soil fertility if they are thinned beyond certain densities.

Buyer *et al.* (2016) concluded that bush removal initially perturbs the soil ecosystem, but over a period of 3-9 years, the system recovers to a state resembling that of undisturbed grass in a bush encroached savanna only if the seed bank of the climax grass species is still intact. Therefore, removal of bush may provide a way to restore both the above ground and below ground components of bush encroached savanna ecosystems to a more grass dominated state (Tainton, 1999; Smit, 2004; Kahumba, 2010; Lohmann *et al.*, 2012; Ward *et al.*, 2014; Chirwa *et al.*, 2015). According to Buyer *et al.* (2016) additional scientific research is necessary to fully evaluate the role of soil microbes in the restoration of savanna rangelands modified by bush encroachment.

1.4.4 Rangeland enclosures

One familiar technique that has successfully been proven in restoring deteriorated rangelands is the use of enclosures whereby grazing is excluded for a specified period of time (Tainton, 1999; Oba *et al.*, 2008; Mussa *et al.*, 2016). According to Mussa *et al.* (2016), rangeland enclosures can be applicable systems for the restoration of deteriorated land if they have definite users, resource perimeter and realistic local stable rules. However, the outcome of long-term studies of managing land in this manner also implies that the conception of bush encroachment is a considerable treat in these enclosures over time, compared to more frequently grazed rangelands (Angassa, 2007). Therefore, exceptional care should be taken when incorporating scientific and indigenous knowledge in the management of rangeland enclosure (Mussa *et al.*, 2016).

1.4.5 Grazing management

The basic principles of range management require that livestock numbers comply with the available forage supply, maintenance of livestock numbers corresponding to available forage supply, consistent allocation of animals within the range, resting of vegetation over changing periods of grazing and the use of most suitable variety of cattle (Tainton, 1999; Hoffmann & Ashwell, 2001; Kgosikoma *et al.*, 2012; Mussa *et al.*, 2016). The reductions of livestock numbers, together with best practice of rangeland management are important for sustaining the productivity and health of rangelands (Illius *et al.*, 1998; Ash *et al.*, 2011). In degraded rangelands, reduced stock numbers and controlled grazing have been suggested to facilitate rehabilitation (Wessels *et al.*, 2007; Li *et al.*, 2011). According to Woodfine (2009), the aim of sustainable land management practices is to maximise the retention, penetration and storage of rain water into soil layers. This encourages a constructive terrain for vegetation cover, soil organic carbon and results in sustainable utilisation of above and below ground biodiversity (Mussa *et al.*, 2016).

The implementation of proper grazing practices as a management tool for enhancing range productivity and restoration also needs to consider the grazing history of the rangeland (Woodfine, 2009; Mussa *et al.*, 2016). This is particularly important if the degraded lands have a historical trajectory of large herbivores, including livestock, utilising the area extensively over time (Papanastasis, 2009). In case of rotational and deferred grazing, it is recommended that the partitioning of land should be based on ecological variation (for example plant species richness, inter and intra-plant species competition, seasonal rainfall patterns) and the timing and duration of grazing be worked out separately for each rangeland in order to account for biophysical variations, mainly due to soils and vegetation types (Abel & Blaikie, 1989; Mussa *et al.*, 2016). Besides its significance in range restoration, improved grazing management will improve the functioning of the hydrological systems in rangelands and contribute to the protection and restoration of biodiversity (Woodfine, 2009). According to the International Union for Conservation of Nature (IUCN), unsustainable livestock management has been identified as a major threat to biodiversity of several grass species (Neely *et al.*, 2010).

1.5 The cost of woody plant encroachment in South Africa

In South Africa water is managed by the government through the rules and regulations stipulated in the Water Services Act of 1997 together with the National Water Act established in 1998 (WWF-SA, 2016). The South African National Water Act (NWA) is rooted on the concept that water forms part of a meaningful complementary water cycle and must, therefore, be administered accordingly. The NWA incorporates an in depth rationale for the use, protection, management and conservation of South Africans' water resources. The important objectives governing water resources are specified in the National Water Resource Strategy (NWRS) (DWA, 2010; WWF-SA, 2016). Woody alien encroacher species usually use more water than neighbouring indigenous species and, therefore, reduces water availability by up to 4%. Should the density of these encroaching woody species advance, reductions in water availability could expand to an approximated 16%. Therefore, bush encroachment can adversely impact available water supplies, especially stream flows, thereby, favour related increases in silt formation which negatively impact water quality (WWF, SA, 2016). According to Van Wilgen *et al.* (2012) an approximated R6.5 billion of the R152 billion of possible ecosystem services (water, grazing and biodiversity) is lost annually as a result of increased bush densities (alien and indigenous woody species) and the loss would have been an estimated additional R41.7 billion, had no control measures be implemented. This suggests a saving of R35.2 billion per annum (approximately 4.8% of South Africa's annual GDP) which represents approximately 30% as a consequence of biological control (control browsing by using goats) (Van Wilgen *et al.*, 1998; De Lange & Van Wilgen., 2010; Van Wilgen *et al.*, 2012). The demand for water supply is further displayed by the fact that, in developing countries, water shortages is contributing to hunger, poverty and diseases (Van Wilgen *et al.*, 1998; Adato *et al.*, 2005; Marais *et al.*, 2008; DWA, 2009).

1.6 Working for Water (WfW) programme in South Africa

1.6.1 The establishment of the WfW programme

In 1995, the WfW programme was the first programme to be acknowledged as part of the Natural Resource Management (NRM) programme of the Department of Water Affairs (DWA) formerly known as the Department of Water Affairs and Forestry (DWAF) (DWA, 2010; Coetzer & Louw, 2012; WWF-SA, 2016). This programme was subsidized by funds

contributed by the National Reconstruction and Development Programme (RDP) budget which its main goal was to control the spread of invasive woody plants (Van Wilgen *et al.*, 1998; McQueen *et al.*, 2001; Van Wilgen *et al.*, 2001; Richardson *et al.*, 2007; Koenig, 2009; Van Wilgen *et al.*, 2011; Klein, 2011; Morena & Hoffmann, 2011). On the 1st of April 2011 the National Resources Management (NRM) programme, was reassigned from the Department of Water Affairs (DWA) to join the Department of Environmental Affairs (DEA) (Coetzer & Louw, 2012). The NRM programme presently includes various programmes such as the Working on Fire, Working for Wetlands, Working for Ecosystems and Working for Forests. These programmes have in turn employed more than 51 300 people (over the last 3 years), of which the majority are woman and youth (WWF-SA, 2016).

The reassignment of the WfW programme to DEA, lead to its unification with other initiatives and has considerably expanded funding opportunities to the scientific society to promote improvements in ecosystem management (Van Wilgen *et al.*, 2012). As a result, the WfW programme has received international acknowledgement and is repeatedly referred to as an innovative, comprehensive and outstanding approach to the management of problematic encroacher species (Magadlela & Mdzeke, 2004; Mark & Dickinson, 2008; Pejchar & Mooney, 2010; Van Wilgen *et al.*, 2011) and has successfully cleared 2.7 million hectares of rangelands over the last 20 years (Van Wilgen *et al.*, 2012; WWF-SA, 2016).

1.6.2 The significance of the Working for Water (WfW) programme

The WfW programme focuses on four main areas to support strategies for dealing with the bush encroachment problem (McQueen *et al.*, 2001; Van Wilgen *et al.*, 2011; Coetzer & Louw, 2012; Van Wilgen *et al.*, 2012):

- National jobs development programme
- Biological control
- Education and community programme
- Legislative framework

The fundamental principle of the WfW programme is to preserve water through the eradication of invasive woody plants as part of the RDP initiative. Operating as a labour-intensive Extended Public Works Programme (EPWP), it echoes the South African

government's obligation to the conservation of natural resources, training, job creation, capacity building and poverty elimination (Magadlela, 2000; McQueen *et al.*, 2001; DEA, 2010; Van Wilgen *et al.*, 2012).

The objectives of the WfW programme are to enhance water security, improve the ecological principle of natural systems, the restoration of the valuable potential lands, to invest in the most marginalised sectors in South Africa and hence the nations quality of life through job creation, and to develop industrial benefits from wood, land, water and trained people (WFF-SA, 2016). Through the promotion of small business and entrepreneurship development, the WfW programme has momentous social benefits for the country's poor (Magadlela & Mdzeke, 2004; Rogerson, 2008; Coetzer & Louw, 2012).

The WfW programme's underlying goal is to mitigate poverty stresses by creating short- to medium-term employment for unskilled workers through the eradication of invasive species (Coetzer & Louw, 2012; Van Wilgen *et al.*, 2012; WFW-SA, 2016). In relation to other environmental management programmes, WfW has to engage in complex socio-ecological environments, in which it is regularly necessary to monitor outcomes, learn from experience and accommodate new approaches (Van Wilgen *et al.* 2012). Van Wilgen *et al.* (2012) recommended that the WfW programme continues a joint interest among scientists and practitioners, in which cooperate work will be implemented and by so doing, address significant future challenges.

1.6.3 Funding for the WfW programme

Funding for WfW expanded from R25 million during 1995/1996 to R250 million in 1997/1998, at which phase it was approximated that R600 million would be required annually over the next 20 years (presuming that invasive vegetation spreads at a rate of 5% yearly), to scale down the problem to a level where these encroaching species could be managed at a relatively low cost (Van Wilgen *et al.*, 2012; WWF-SA, 2016). McConnachie *et al.* (2011) stated that, in the fynbos biome alone, R855 million has been spent on clearing encroaching woody species. Regardless of these vast investments, WfW could reach only a minimum fraction of the invaded areas, which continue to spread, although less rapidly (Van Wilgen *et al.*, 2012). According to Van Wilgen *et al.* (2012) it materializes that the original estimates of the degree of spread of about 5% per year extremely low. The advanced spread

of invasive pine trees in unreachable mountain areas, threatens to disturb fynbos flora over extensive areas, with deliberate consequences for water resources, catchment stability and the risk of wildfires (Kraaij *et al.*, 2011; McConnachie *et al.*, 2012). The increasing threat of alien vegetation on suitable lands, poses an obligation to increase financial contributions which are acceptable for alien clearing programmes, hence, co-financing is demanded from both private and governmental associations to effectively manage the areas affected by bush encroachment (WWF-SA, 2016).

In the WfW programme, representatives are active small-scale entrepreneurs who provide services by restoring lands under various types of ownership. The WfW representative selection criteria demands that employees must have been formerly unemployed. Proposals for restoration contracts are done by the WfW representatives as opposed to landowners (Magadlela & Mdzeke, 2004; Turpie *et al.*, 2008; Rodricks, 2010). Financing from this source is then cast on the control of bush encroachment with widely recognized negative impacts on water resources (DWAF, 2007; Van Wilgen *et al.*, 2011).

1.7 Problem statement of this study

According to Oba *et al.* (2000) the progressive establishment of unwanted woody plant species is an indication of land degradation. Bush encroachment is considered as the extensive form of land degradation in both arid and semi-arid regions of South Africa (De Klerk, 2004; Joubert *et al.*, 2009; Schröter *et al.*, 2010). Bush encroachment has been considered a rangeland problem in the savannas of southern Africa for practically over a hundred years (Bews, 1917; Kgosikoma & Mogotsi, 2013; O' Connor *et al.*, 2014). Nine of the 28 magisterial districts in the North Western Province include areas that are so severely encroached with invasive woody species that the land cannot be reclaimed by farmers without desperate bush control measures (Van Vuuren, 2003; Franci, 2011). Consequently, bush encroachment accompanies a reduction in the livestock carrying capacity of savanna ecosystems (Ward, 2005). The latter has severe negative implications for food security as well as the agricultural productivity of savanna rangelands (Kgosikoma & Mogotsi, 2013). Large areas in South Africa and Botswana have been affected by bush encroachment (Moleele *et al.*, 2002; Ward, 2005; Archer, 2010; Ward & Esler, 2011). Bush encroachment has not only impacted sustainable livestock production systems but also human well-being (Lamprey, 1983; Moyo *et al.*, 1993; Scholes & Archer, 1997; Kgosikoma & Mogotsi, 2013).

In South Africa, a response to the bush encroachment problem was brought about through the implementation of the WfW programme. To control bush encroachment in the Taung area, the WfW working teams started to get involved in 2011, whereby both chemical and mechanical control methods in selected sites were carried out. The aim was to improve the grazing capacity for improved livestock keeping. Pastoralists' perception and ecological knowledge were often not considered during the application of control activities, despite their knowledge of the local environment (Barkes *et al.*, 2000; Angassa & Oba, 2008; Roba & Oba, 2009; Daryanto, 2013). The participation of local communities is viewed to be of fundamental value for understanding environmental changes that take place (Fernandez-Gimenez, 2000; Quinn *et al.*, 2008). To ensure positive rangeland restoration/rehabilitation, community participation and awareness are necessary and should be encouraged within the Taung area, especially in regard to grazing lands. Both beneficiaries and stakeholders (the local residents and tribal authorities) and the WfW programme representatives should work together in implementing restoration techniques such as re-seeding in bare areas, reduction in stocking rates (to prevent overgrazing) and introduction of eco-rangers (to facilitate the use of the natural resource base).

In the Taung area, the most problematic species in the study sites selected included *Senegalia mellifera*, *Vachellia tortilis*, *V. karroo* and *Tarchonanthus camphoratus*. The non-woody exotic species such as *Opuntia spirostachya* and *O. ficus-indica* were also present in the sites. Overgrazing, misuse of natural resources (wood harvesting) and inappropriate use of fires have accelerated the effects of bush encroachment in the Taung area. The lack of follow-up practices by the WfW working teams in the Taung area has contributed to increased bush densities through woody re-growth (coppicing), thereby, enhancing the bush encroachment problem. However, the re-establishment of palatable perennial grasses was also evident in the controlled sites, thereby, indicating positive effects of bush removal towards grass species establishment.

1.8 Aim of the study

The aim of this study was to:

1. Assess the impacts of bush control activities by the WfW programme on the woody re-growth and changes in grass species composition in selected sites in the Taung area,
2. Assess the changes in soil chemical and physical properties after bush control, and

3. Determine the impact of bush encroachment and control on the social life of the communities in the Taung area.

1.9 Framework

The thesis is presented in 8 chapters.

Chapter 1: This chapter provides a brief background of the study. It also addresses the implementation, role and significance of the WfW programme towards controlling the bush encroachment problem in South Africa. This chapter also addresses the problem statement and the aim of the study.

Chapter 2: This chapter provides a literature review on the problem of bush encroachment in the savanna biome of South Africa especially, in communal areas of South Africa. The effects and causes of bush encroachment in rangelands of the savanna biome are also discussed. Walter's (1939) two-layer hypotheses, also including the positive and negative effects that bush encroachment has on the herbaceous vegetation are discussed. Bush control methods are also discussed in this chapter. A literature review on social studies is also included.

Chapter 3: This chapter provides the location of the North West Province in South Africa and the location and history of the Taung area. The chapter also includes general features of the Taung area (such as household structure, education and economic status). Vegetation of the Taung area, environmental challenges and management of the study area are discussed. Environmental features such as rainfall, temperature, geology and soils of the Taung area are also explored. Demarcation and the description of the study sites are also included.

Chapter 4: This chapter includes all the materials and methods used to determine the woody, herbaceous and soil components of the study sites, as well as methodologies to carry out the social surveys at specific areas.

Chapter 5: This chapter includes the results and discussion of the changes in woody and grass abundance in bush controlled and uncontrolled sites in the Taung study area.

Chapter 6: This chapter includes the results of the soil chemical properties of the study sites.

Chapter 7: This chapter includes the results and discussion of the social surveys carried out at the study sites in the Taung area where bush control was carried out.

Chapter 8: This chapter includes some conclusion and recommendations to be implemented in the affected sites to rehabilitate the area for a better functioning ecosystem.

CHAPTER 2 LITERATURE REVIEW

2.1 The savanna rangelands and the problem of bush encroachment

The savanna biome forms part of the tropical or sub-tropical ecosystems which are complex and in a continuous state of change, due to of natural and anthropogenic (human-induced) factors such as land use practices (Scholes & Archer, 1997; Walker & Abel, 2002; Simioni *et al.*, 2003). The biome is found in a transitional zone between forest regions and desert and its ecosystems are characterised by a continuous layer of herbaceous plants, such as, grasses and casually populated plots of trees and shrubs (Frost *et al.*, 1986; Scholes & Archer, 1997; Kgosikoma & Mogotsi, 2013; Tessema *et al.*, 2017). Savannas accommodate an extensive portion of the world's human population, variety in floral and faunal species compositions with the majority of its rangelands being useful for livestock production (Scholes & Archer, 1997; Sankaran *et al.*, 2004; O' Connor *et al.*, 2014). The savanna biome is the largest biome in the southern African sub-continent and contributes to up to 32.8% of the surface area of South Africa, which relates to 399 600 km² (Mucina & Rutherford, 2006). Savannas are considered to have balanced ecosystems around one or more stable states or points of equilibrium (Illius & Swift, 1988; Illius & O'Connor, 1999) although, they are exceedingly dynamic over various geographical scales and differ in rainfall regimes, soil nutrient availability, fire frequencies and herbivory (Rietkerk & Van De Koppel, 1997; Briske *et al.*, 2003; Kgosikoma & Mogotsi, 2013; Ward, 2015). In South Africa, this biome is mostly found at altitudes below 1500m but do extend to 2000m above sea level (Mucina & Rutherford, 2006). Rainfall patterns, nutrient availability, fire and herbivory are all key determinants of the vegetation structure and composition of savannas (Scholes *et al.*, 2002). Many genera and species of the savannas of southern Africa are shared with the savannas of central-and east Africa and can also be found in the Nama-Karoo Biome of southern Africa (Scholes, 1997; Harmse, 2013). Rangelands are described as geographical regions on which indigenous vegetation is mostly dominated by grasses, grass-like plants, forbs and shrubs and support various grazing and browsing animals (Allen *et al.*, 2011; Sive, 2016).

The central area of Southern Africa is considered semi-arid to arid (Hoffman & Ashwell, 2001). The study area for this study, in the Taung region of the North West Province, may be considered as a semi-arid savanna, although significantly drier (mean annual precipitation [MAP] = c. 300-400 mm) than many other study areas in southern

Africa, previously studied (O'Connor, 1995; Bond *et al.*, 2003; Higgins *et al.*, 2007; Buitenwerf *et al.*, 2012). Due to the occurrence of woody vegetation (trees and shrubs) in savannas, these ecosystems are prone to bush encroachment. The Namibian agricultural sector has experienced annual losses of about N\$ 700 million as a result of bush encroachment (De Klerk, 2004). This economic loss has escalated to N\$ 1.6 billion, thereby, reducing beef production by 50% (Christian, 2010; Muroa, 2013).

Bush encroachment is a form of land degradation which mainly results in the loss of total vegetation cover or an increase of alien/invasive woody vegetation at the cost of palatable perennial grasses and herbs (Skarpe, 1990; Jeltsch *et al.*, 2000a; Buitenwerf *et al.*, 2012; Lohmann *et al.*, 2012). The encroaching woody plants are generally unpalatable to domestic livestock and, therefore, reduce the grazing capacity of viable lands (Donaldson, 1980; Lamprey, 1983; Scholes & Archer, 1997).

Bush encroachment is regarded as a natural phenomenon, affecting rangelands worldwide. This process is usually irrevocable for several years and reduces the supply of forage biomass, thus, affecting livestock production as well as other ecosystem services such as water retention capacity and protection from soil erosion (Gillson & Hoffman, 2007; Graz, 2008; Rohde & Hoffman, 2012; O'Connor *et al.*, 2014). Bush encroachment has negative implications on ecosystem services such as a decline in the overall grass and livestock productivity, ground-water restoration, carbon sequestration or the prevention of soil erosion (UNCCD, 1994; Jeltsch *et al.*, 2000a; Graz, 2008; Lehmann & Rousset, 2010; Buyer *et al.*, 2016) as well as momentous losses in biodiversity across taxonomic groups (Blaum *et al.*, 2009). Savanna rangelands have been subjected to drastic vegetation changes which resulted in a shift from a state previously dominated by perennial grasses to one which is influenced and dominated by woody plant invaders (Fensham *et al.*, 2005; Wigley *et al.*, 2010; Buitenwerf *et al.*, 2012; Lohmann *et al.*, 2012).

The reasons for bush encroachment are seen in several interacting (mostly human induced) factors on the local, regional and global scale (Lohmann *et al.*, 2012). At the local and regional scale, the most influential drivers for bush encroachment, in semi-arid rangelands in particular, are unsuitable rangeland management practices (e.g. high livestock densities and fire suppression) which are provoked by increasing pressure on natural resources (Walker *et al.*, 1981; Skarpe, 1991; Van Langevelde *et al.*, 2003; Reynolds *et al.*, 2007;

Graz, 2008; Joubert *et al.*, 2008; Marchant, 2010; Ward & Esler, 2011; Joubert *et al.*, 2012; Lohmann *et al.*, 2012; Rohde & Hoffman, 2012). At global scale, climate change and rising atmospheric CO₂ levels, but also changes in global markets and the rising demand in food products are increasingly considered as drivers of savanna rangeland degradation (UNCCD, 1994; Reynolds *et al.*, 2007; Tietjen *et al.*, 2010; Wigley *et al.*, 2010; Kgope *et al.*, 2010; Lohmann *et al.*, 2012).

Careful management is required to avoid degradation of savannas. However, most evident causes of degradation are erratic climatic changes, varied dynamics of rangeland degradation and raising population pressures on the natural resource base. This poses a strenuous task to land managers, hence, a universal explanation for the causes of the degradation of savannas has not yet been identified (UNCCD, 1994; Gillson & Hoffman, 2007; Lohmann *et al.*, 2012). Opportunistic management strategies have been promoted in the context of South African land reforms especially regarding communal rangeland management (Cowling & Lombard, 2002). According to Lohmann *et al.* (2012), it is, however, unclear as to what extent such management strategies are suitable and viable for rangeland management in semi-arid environments from an ecological as well as from an economic perspective. It is expected that, the eradication of some or all the woody plants would, therefore, normally result in increased grass biomass and improved grazing capacity under normal rainfall patterns (Teague & Smit, 1992; Tainton, 1999; Hoffman & Ashwell, 2001; Smit, 2004; Smit, 2005; Kgosikoma, 2012; Angassa *et al.*, 2014).

2.2 The economic significance of African savannas

Animals influence the vegetation on which they feed and are assumed to shape the structure of savanna rangelands (Cumming, 1982; Sinclair, 1983; Tesemma *et al.*, 2011). The African savannas accommodate more hoofed animal species than other continents (Du Toit, 2003). Furthermore, African savannas are the most relevant ecosystems for raising herbivores (Prins, 1988; Tesemma *et al.*, 2011). African savannas have been used for rangeland resources as grazing lands for livestock and millions of people depend on them for variable pastoral production systems (Pratt & Gwynne, 1977; Skarpe, 1991; Tesemma *et al.*, 2011). These rangelands have amazing landscapes that support important ecosystem services, such as supporting suitable habitat for wildlife populations and domestic herbivores and yielding admirable livestock products (Desta & Coppock, 2004; Coppock *et al.*, 2011).

Livestock products provided by savanna ecosystems contribute about 14 to 25% of South Africa's entire agricultural production (Scoones & Graham, 1994; Hagmann & Speranza, 2010). Various livestock species are a persuasive way of satisfying human needs in semi-arid African savanna rangelands (Richardson *et al.*, 2010). Pastoral production systems are a meaningful, however, declining economic sectors in African countries (Prins, 1992; Coppock *et al.*, 2011). Therefore, rangeland degradation poses a threat to pastoral based livelihood strategies (Gemedo *et al.*, 2006b; Harris, 2010; Ho & Azadi, 2010).

In South Africa, land degradation and bush encroachment have been reported to be the most common factors threatening livestock production in communal rangelands (Sive, 2016). According to Sive (2016), livestock production in communal areas is regarded as a major source of food and livelihood for those who trade/sell because goats, sheep and cattle. Solomon *et al.* (2014) stated that communal farmers generate income by selling their livestock to send their children to school. Stroebel *et al.* (2011) elaborated that communal farmers in South Africa generate their income from livestock production. Bush encroachment and land degradation are threatening both resilience of communal rangelands and the livelihoods of those who depend on them (Seymour & Desmet, 2009; Archer *et al.*, 2011). Bush encroachment is regarded as an environmental and economic problem that affects livestock production by its suppressive effect on herbaceous vegetation in communal rangelands (Oba *et al.*, 2000). Moreover, Gwaze *et al.* (2009) argued that livestock in communal rangelands can also be influenced by seasonal fluctuation in forage quality and quantity in arid and semi-arid regions. Sive (2016) stated that, the decrease of grazing capacity in savanna communal rangelands is linked to rapid increase of woody cover which ultimately poses negative implications towards the economic productivity in communal rangelands.

Grazing is one of the monetary ways of using rangelands, particularly in communal and/or pastoral areas (Lesilo *et al.*, 2013). The legislation of biodiversity conservation and ecosystem establishment within rangelands preserve the ecological value of savanna ecosystems (Daryanto, 2013; Lesilo *et al.*, 2013; Angassa *et al.*, 2014). Therefore, restoring or rehabilitating rangeland ecosystem health and resilience is an important practice to ensure the future supply of the ecosystem services, which are essential for the future sustenance of human societies. These services include the maintenance of stable soils, reliable and clean water supply and the natural development of floral and faunal communities to meet the

expected cultural values aimed at improving human livelihoods (Grice & Hodgkinson, 2002; Teague *et al.*, 2008). Therefore, arid and semi-arid rangelands experience various forms of land degradation, where overgrazing may have played a major role (Pungru & Lazaaro, 2000; Van Auken, 2000; O'Connor *et al.*, 2014).

2.3 Effects of bush encroachment

Bush encroachment can either be caused by an increase in the biomass of already established plants, or the expansion of tree density as a result of seedling development (Strang, 1973; Tainton, 1999; Smit, 2004; Isaacs *et al.*, 2013). Various factors contribute to bush encroachment, encroachment, either singularly or in combination. These include excessive grazing (Tainton, 1999; Hoffman & Ashwell, 2001; O'Connor *et al.*, 2014), inappropriate use of fires (Kahumba, 2010; Eldridge & Soliveres, 2015), absence of mega-herbivores, for example, elephants that can damage trees (Kerley & Whitford, 2009; O'Connor *et al.*, 2014), increased nitrogen (N) deposits and atmospheric carbon dioxide (CO₂) (Archer, 2010; Kambatuku *et al.*, 2011; Bond & Migley, 2012; Buitenwerf *et al.*, 2012; Leakey & Lau, 2012; O'Connor *et al.*, 2014; Ripple *et al.*, 2015), due to extended climate changes (Wenig *et al.*, 2003; Knapp *et al.*, 2008; O'Connor *et al.*, 2014; Ward *et al.*, 2014).

2.3.1 Grazing

Scientific research has proved that overgrazing (predominantly by livestock) results in bush encroachment (Walker & Noy-Meir, 1982; Hobbs & Mooney, 1986; Scholes & Archer, 1997; Roques *et al.*, 2001; O'Connor *et al.*, 2014; Ward *et al.*, 2014). Savanna ecosystems are under severe degradation pressure due to overgrazing of palatable grass species, expansion of crop cultivation, soil erosion and the thickening of undesired woody species (MOA, 2000; Angassa & Oba, 2008; Kgosikoma *et al.*, 2012). Within the savanna biome, excessive grazing further leads to modified competitive alterations between the woody and the grassy vegetation layers as a result of defoliation (Tainton, 1999; Hoffman & Ashwell, 2001; Daryanto, 2013). Excessive grazing has been recorded alter the positive effects of shrubs on a variety of ecosystem functions (Eldridge *et al.*, 2013). Former studies of water availability beneath shrubs have tended to focus on the presence of shrubs *per se* with limited recognition on the potential effects that certain levels of grazing may have on a variety of shrubs (Allington & Valone, 2013). Changes in grazing severity can probably explain the contradictor effects of shrubs on ecosystem dynamics, simply because woody

plants the mostly preferred by herbivores during drought periods (Van den Berg & Kellner, 2005).

In areas that have not experienced a long transformative history of grazing by mega-herbivores, the effects of overgrazing on ecosystem processes have normally been understood to be negative (Milchunas *et al.*, 1988; Eldridge *et al.*, 2015). Low to moderate levels of grazing have been discovered to lead to biotical compelled changes in plant community structure and establishment (Eldridge *et al.*, 2011b; Eldridge *et al.*, 2015). Continuous overgrazing reduces vegetation cover, resulting in loss of connectivity among vegetation patches, thereby, negatively affecting soil water and nutrient availability to plants during favourable conditions (Reitkerk *et al.*, 2000; Tongway *et al.*, 2003). Overgrazing also increases soil compaction, disturbs the soil surface layer, impairs basic structural complexity (Tongway *et al.*, 2003) and modifies the allocation of resources by moving eroded interspace soil into the shrub patches (Okin *et al.*, 2009). Overgrazing also reflects negatively on grass development and composition (Smit, 2005; Angassa & Oba, 2008; Angassa *et al.*, 2012; O'Connor *et al.*, 2014) and results in a reduction of grass biomass and ultimately livestock productivity.

According to Wiegand *et al.* (2006), heavy grazing will benefit woody species development due to the reduction in the competition regime. *Vachellia tortilis* and *V. karroo* have deep lateral root systems, while *Senegalia mellifera* has a shallow root system, thus implying that these species have an added advantage to access more soil water and nutrients (Ward *et al.*, 2014).

2.3.2 Soil conditions (or properties) as a result of bush encroachment

The removal of above-ground plant biomass by livestock grazing reduces the soil quality properties such as soil physical and chemical characteristics (Walker & Desanker, 2004; Hagos & Smit, 2005; Kgosi, 2012; Daryanto, 2013). Overgrazing reduce vegetation cover, which limits organic matter added to the soil and subsequently contribute to reduced soil structure stability, protection against rainfall impact, infiltration rate and ecosystem activities (Roose & Barthes, 2001; Snyman & Du Preez, 2005). Many shrubs are unpalatable to livestock and are often more competitive than grasses under grazing and more resistant than herbaceous plants to fires, soil salinity and low temperatures (e.g. frost) (Richmond &

Chinnock, 1994; Booth *et al.*, 1996; Smit, 2005; Kgosikoma, 2014). Domestic grazing animals also have an influence on plant structure and composition (Smit, 2005; Klumpp *et al.*, 2009; Kgosikoma, 2012), which in turn affects soil productivity (Scholes, 1990) as a result of changes in root biomass (Klumpp *et al.*, 2009; Kgosikoma, 2012) and the quality of organic matter (Kgosikoma, 2012). Soil nutrient depletion eventually causes a decline in the primary production of the herbaceous plants and carrying capacity for livestock (Girmay *et al.*, 2008; Kgosikoma, 2012). Grazing also increases bare soil surface which increases erosion potential and leads to unfavourable changes in soil chemical and physical properties (increasing bulk density, decreasing water infiltration and nutrient concentrations) (Castellano & Valone, 2007; Stavi *et al.*, 2008). The impact of irrational grazing practices seems to be specifically severe during drought seasons when water stress negatively affects grass biomass production (Brintton & Sneva, 1981).

Soil compaction due to livestock grazing (Geissen *et al.*, 2009), depends on animal size, stocking density, soil texture, soil moisture and vegetation cover (Bilotta *et al.*, 2007). Hoof action compacts soil through reduction in pore spaces (Drewry, 2006), which increases bulk density, especially on the soil surface (Liebig *et al.*, 2006). Soil compaction, leads to reduced water infiltration, thus increasing water runoff (Asner *et al.*, 2004) during rainfall. Eldridge and Soliveres (2015) stated that, regardless of the evident rates of encroachment by both shrubs and trees, irrespective of whether it is natural or human-induced process is a major issue. This is since the majority of pastoralists acknowledge that bush encroachment threatens the development of their pastoral enterprises. Scientific and pastoral communities agree that, woody plant invasion is related to declining ecosystem functioning and stability as well as desertification (Noble & Rodolfo, 1997; MEA, 2005; Archer, 2010; Eldridge & Soliveres, 2015).

Bush encroachment has contributed to alterations on soil physical properties such as soil pH, soil organic carbon content, carbon sequestration and nitrogen availability. Literature has provided evidence which suggests that, changes in land-use practices coupled with overgrazing has largely contributed to increases in woody plant densities which in turn has influenced the soil carbon sink and soil nitrogen availability (Hoffman & Ashwell, 2001; Wiegand *et al.*, 2006; Angassa *et al.*, 2012; Lesilo *et al.*, 2013; O'Connor *et al.*, 2014).

According to Buyer *et al.* (2016) it is clear that bush encroachment has the potential to alter soil microbial community biomass, structure and diversity. Plant species composition is known to affect microbial species composition and diversity (Wardle, 2006; Maul & Drinkwater, 2010). Invasive plants have been shown to alter soil microbial communities (Batten *et al.*, 2006), biogeochemical cycling, nutrient availability and ecosystem functioning (Weidenhammer & Callaway, 2010). Soil microbial biomass has been reported to increase with increasing woody plant density and age (Liao & Boutton, 2008; Buyer *et al.*, 2016).

2.3.3 Rainfall variability

Rainfall availability could also be one of the driving factors leading to bush encroachment (Angassa & Oba, 2008; Kgosikoma, 2012; O'Connor *et al.*, 2014). Additional scientific studies in the semi-arid areas of South Africa have demonstrated that irregular high rainfall events has contributed to woody plant encroachment (Archer *et al.*, 1998; O'Connor, 1995; O'Connor & Crow, 1999; Wiegand *et al.*, 2005; Kraaij & Ward, 2006; Wiegand *et al.*, 2006; Meyer *et al.*, 2007a; Meyer *et al.*, 2007b; Joubert *et al.*, 2012). Kraaij and Ward (2006) found compelling evidence on woody plant recruitment following extremely high rainfall events. It is reported that the grass competitive ability is reduced in years of below-average rainfall preceding above-average rainfall events, and this aids for the development and establishment of woody species (Booth, 1986; O'Connor, 1995; Wiegand *et al.*, 2005; Wiegand *et al.*, 2006; Ward & Elser, 2011). Meyer *et al.* (2007a) and Joubert *et al.* (2012) reported that, although, *Senegalia mellifera* does not have a persistent seed bank, its recruitment is effectively linked to above-average rainfall events. However, other encroaching species such as *V. tortilis* and *Rhigozum trichotomum* have a persistent seed bank and *Tarchonanthus camphoratus* mostly has clonal reproduction, thus, the relationship between rainfall and recruitment might not be as vigorous for all woody species as it is for *S. mellifera* (Ward *et al.*, 2014). Irregular high rainfall events and possible increases in atmospheric CO₂ function collectively and may be thought-out to be the cause of extensive woody plant encroachment in certain parts of South Africa (Ward *et al.*, 2014). Woody plant encroachment in areas receiving <400 mm/a usually leads to an increase in available carbon stored in these ecosystems (Jackson *et al.*, 2002; Hibbard *et al.*, 2003; Knapp *et al.*, 2008) although a decrease is noted above 400 mm/a rainfall (Ward, 2009).

Increased CO₂ has the ability to cause bush encroachment by favouring woody vegetation during periods of drought (Eamus & Palmer, 2007; Leakey & Lau, 2012). Disturbances such as extended drought periods, followed by unusually high rainfall events, may lead to bush encroachment (O'Connor, 1995; Meyer *et al.*, 2007b; Moustakas *et al.*, 2009; Ward & Elser, 2011). O'Connor and Crow (1999) also reported that in the Eastern Cape Province of South Africa, an increase in bush densities was closely related to high rainfall events immediately after drought seasons. Gordijn *et al.* (2012) noted that in the KwaZulu-Natal Province of South Africa, increased bush densities were as a result of constant rainfall changes. In seasons of more frequent droughts, when there is a great variance in rainfall, the increase in abundant precipitation events also favours bush encroachment (Karl & Knight, 1998; Kraaij & Ward, 2006; Bates *et al.*, 2008; Volder *et al.*, 2010).

Savanna ecosystems are mainly water-limited and as a result bush encroachment is correlated with inter-annual rainfall availability (Angassa & Oba, 2007). At regional scale, encroacher plants such as *Senegalia mellifera* require a minimum of three consecutive favourable yearly rainfall regimes, to ensure successful recruitment (Joubert *et al.*, 2008). Elevated soil moisture availability, specifically when there is restricted competition from grasses, grant woody plant seedlings the opportunity to grow and advance into impenetrable bush thickets (Kgosikoma & Mogotsi, 2013). Contrarily, drought, through limited plant growth, seed germination and elevated competition for preferential ground water supply at high shrub densities, leads to the death of some plants (Roques *et al.*, 2001) and thus reduced the devastating effects of bush encroachment. Bush encroachment is, therefore, a cyclic natural phenomenon determined by recruitment and death of encroacher plants in response to rainfall patterns (Wiegand *et al.*, 2006; Kgosikoma & Mogotsi, 2013). There are alarming declines of water availability as a result of bush encroachment, therefore, water is a primary determinant for both grassy and woody plants (Tainton, 1999; De Wit *et al.*, 2001; Hoffman & Ashwell, 2001; Van Auken, 2009). Based on this concept it is hypothesised that grasses use only surface water while woody plants mostly use subsoil water (Noy-Meir, 1982; Tainton, 1999; Lesilo *et al.*, 2013; O'Connor *et al.*, 2014). Thus, potential water reductions in South African are anticipated to be 8 times higher should current bush densities increase and utilize their maximum potential extent (De Wit *et al.*, 2001). The invasion of woody plants is a considerable cost to South Africa's economy, estimated at R 6.5 billion per annum, which is approximately 0.3% of South Africa's Gross Domestic Product (GDP) of an approximated R 2 000 billion and with the potential to increase to up-to 5% of GDP if

these woody plants remain uncontrolled (Van Wilgen *et al.*, 2001; De Lange & Van Wilgen, 2010).

2.3.4 Fire management

The continual exclusion of rangeland fires, farming of marginal lands and constant grazing on the remaining fraction of the communal rangelands have been reported to encourage the spread of bush encroachment to a level of more than 60%. This has led to reductions in grass cover, unfertile rangeland condition and consequently poor livestock productivity (Oba, 1998; Oba *et al.*, 2000; Lesilo *et al.*, 2013; Angassa *et al.*, 2014; O'Connor *et al.*, 2014). It is generally acknowledged that fire is extraordinary determinant in arid and semi-arid savannas, owing to the decline of fuel loads supported by grasses (Teague & Smit, 1992; Meyer *et al.*, 2005; Higgins *et al.*, 2010; Kraaij & Ward, 2006; Ward *et al.*, 2014). Bond (2008) and Higgins *et al.* (2010) acknowledge that fire in African savannas is an important tool especially in a region inheriting above 750 and 820 mm MAP (mean annual precipitation) respectively.

Regular burning restrain woody plant growth by destroying the shrubs and young trees and thus prevents them from developing into mature woody plants, normally resistant to fire and out of reach for browsers (Mphiyane *et al.*, 2011). However, policy makers in Africa fail to acknowledge the importance of fire as a management tool in savanna ecosystems and thus forbid the use of fires in rangelands (Dalle *et al.*, 2006; Kgosikoma & Mogotsi, 2013). Respectively, pastoralists and ecologists contend that the lack of regular burning has allowed invasion of woody plant vegetation (Kgosikoma *et al.*, 2012). Fire should be a fundamental part of savanna ecosystems management (Kgosikoma & Mogotsi, 2013). Given the significant role of fire, it is imperatively necessary to institutionalize sustainable burning intervals (Fatumbi *et al.*, 2008) and incorporate controlled regular burning of savanna ecosystems (Kgosikoma & Mogotsi, 2013). Uncontrolled burning increases pastoralist's vulnerability to undesirable impacts of drought and favours elevated levels of carbon into the atmosphere. The continued use of fire as a rangeland management tool, therefore, depends upon prior knowledge granted by future climatic conditions and the ability to minimise its negative impacts minimise its negative impact, such as, greenhouse effects, air pollution and carbon losses (Kgosikoma & Mogotsi, 2013).

2.3.5 Climate change

Both inter- and intra-annual precipitation patterns are anticipated to change in arid and semi-arid savannas in the system of climate change, leading to an increased number of intense climatic events (IPCC, 2007). For semi-arid regions, especially in South Africa, but also elsewhere, climate change predictions include many changes (Lohmann *et al.*, 2012). While an increase of mean temperature seems inevitable, model predictions regarding the amount of precipitation and its intra- and inter-annual distribution vary (IPCC, 2007; Higgins & Scheiter, 2013). Temperature increases and changes in precipitation patterns can lead to significant changes in water availability, which is the key driver of semi-arid savanna dynamics (Graz, 2008; Tietjen *et al.*, 2010; Lohmann *et al.*, 2012).

Additionally, predictions include increases in average atmospheric CO₂ levels, temperature variations and changes in the average annual precipitation (MAP) for arid and semi-arid regions (IPCC, 2007). Unfortunately, climate model forecasts are the main cause of confusion because direction of change can differ between different climate models (IPCC, 2007). According to Lehmann (2010), increased evaporation and transpiration rates will result in water availability shortages. The water use productivity in turn will be positively affected by increased levels of atmospheric CO₂ and possibly favour woody vegetation over grasses due to their distinct carbon pathway (Bond, 2008; Tietjen *et al.* 2010; Kgope *et al.*, 2010; Buitenwerf *et al.* 2012).

2.3.6 Increased CO₂ levels

There is reasonable controversy concerning the role of global climate changes on bush encroachment, specifically with regard to increased CO₂ levels (Archer *et al.*, 1995; Bond & Midgley, 2000; Körner, 2006; Ward, 2010; Bond & Midgley, 2012; Leakey & Lau, 2012). However, O'Connor *et al.* (2014) indicated that increased atmospheric CO₂ is the primary driver of bush encroachment. It has been reported by Körner (2006) and Leakey and Lau (2012) that, the effects of increased CO₂ are generally greater during drought periods rather than in rainy years. Ward (2010) has noted that the ecological result of the C₃/C₄ ratio differences has favoured the growth of C₃ plants which are characterised by woody plants and decrease carbon losses. Reductions in carbon losses may be caused by increased carbon levels in polyphenols, which is an insignificant metabolite in encroacher

species, such as *Senegalia mellifera* and *Vachellia tortilis* (Lawler *et al.*, 1997; Kanowski, 2001; Mattson *et al.*, 2004).

Some small-scale experiments in South Africa which utilized elevated CO₂ in bins (Kgope *et al.*, 2010) or from natural CO₂ springs (Stock *et al.*, 2005) had different results, with the prior study bringing forth outcomes that were dependable with increased effectiveness of C₃ plants and the most recent not supporting these outcomes (Ward *et al.*, 2014). There is some dispute as to whether the increased photosynthetic effectiveness of C₃ plants are presently better than those of C₄ plants (herbaceous plants) resulting from elevated CO₂ levels of about 350 to 440 ppm (parts per millilitre) (Wolfe & Erickson, 1993; Bond & Midgley, 2000; Ward, 2010). Assuming that, C₃ plants are more effective may render CO₂ as the primary determinant responsible for bush encroachment (Bond & Midgley, 2000).

According to Bond and Midgley (2000), increases in global CO₂ levels have led to a proliferation of woody plants. However, several mechanisms have been proposed, which may explain a positive influence of atmospheric CO₂ on woody plant abundance. These are as follows:

- (i) raising CO₂ levels favour C₃ synthesis relative to C₄ synthesis, thus increasing the quantum yield and thus growth of C₃, but not C₄ plants,
- (ii) elevated CO₂ may reduce transpiration rates of grasses, causing deeper percolation of water and thereby favouring woody species (Bond & Midgley, 2000; Polley *et al.*, 2003). Furthermore, an increased amount of carbon may become available for:
 - (iii) prevention of seedlings from damage by grass fires (Bond & Midgley, 2000) or for,
 - (iv) investments in carbon-based compounds, such as tannins, which are the main defence compounds for many encroaching trees but not of grasses (Ward & Young, 2002).

The benefits of increased CO₂ to C₃ plants have probably not been discovered yet (Bond & Midgley, 2012). Higgins and Scheiter (2013) modelled woody plant encroachment in African savannas by the aid of global CO₂ increases and suggested that there should be localised effects of bush encroachment, assuming rainfall remains constant. Several studies have demonstrated that elevated atmospheric CO₂ also increases water use efficiency of plants, thereby, soil water availability (Centritto *et al.*, 2002; Nelson *et al.*, 2004; Widodo & Rehmarestia, 2008; Leakey *et al.*, 2012). However, Körner (2006) and Leakey and Lau

(2012) noted that, immediate increases in the efficiency of plants due to increased CO₂ have not always been sustained in the long term. Additionally, Zak *et al.* (2003) pointed out that, plants may ultimately be limited by soil nutrients, especially nitrogen, which will constrain the benefits accrued by growing in a CO₂ richer environment.

Bond and Midgley (2000) suggested that savanna trees will benefit from increased levels of atmospheric CO₂ because of their C₃ photosynthetic pathway. Accordingly, the resulting increased growth rates of woody plants will allow for faster regeneration and growth after fires and consequently an escape from the so-called 'flame-zone' (Higgins *et al.*, 2000; Kgope *et al.*, 2010; Higgins & Scheiter, 2013). Other studies argue with a shift in the competitive balance between trees (C₃) and perennial grasses (C₄) to explain the predicted increase in tree cover (Tietjen *et al.* 2010; Ward *et al.*, 2014). Wiegand *et al.* (2005) and Kraaj and Ward (2006) attributed the increases in woody densities to factors such as increases in atmospheric CO₂ levels, rainfall patterns, increased nitrogen oxide levels, drought and soil surface disturbances (Hagos & Smit, 2005).

2.4 Walter's (1939) two-layer hypothesis

Various theories pursue to explain tree-grass co-existence, however, the spatial resource partitioning theory is the most acknowledged but also broadly challenged (Walker, 1939; Ward & Noy-Meir, 1982; Jeltsch *et al.*, 2000b; Ward, 2005; Lehmann, 2010). According to this theory, tree and grass roots use of different vertical niches, which support each vegetation layer with specific access to vital plant growth requirements such as water and minerals (Breshears & Barnes, 1999; Kambatuku *et al.*, 2011). This means that, when woody and herbaceous niches do not overlap, there is no competition for plant nutrient requirements between the different plant species in this vicinity and, therefore, the tree-grass co-existence is achieved (Ward *et al.*, 2014).

An example of such spatial theory is the Walter's (1939) two-layer hypothesis (Walter *et al.*, 1981; Walter & Noy-Meir, 1982) which has repeatedly been contradicted or even rejected (Higgins *et al.*, 2000; Jeltsch *et al.*, 2000b; Higgins & Scheiter, 2013; Kulmatiski *et al.*, 2008; Ward *et al.*, 2014). Ward *et al.* (2014) elaborated that, in this two-layer hypothesis, it is suggested that grasses have an active shallow root system and would, therefore, access water only from the topsoil surface layer. In contrast, woody plant species

would have limited access to the topsoil water but absolute access to, and mainly depends on, subsoil water beneath grass roots. Spatial niche separation in the soil would determine the success of the well-balanced co-dominance of trees and grasses in rangelands (Ward *et al.*, 2014). In contrast, the non-equilibrium concept (Walter, 1939) suggests that climate variability, particularly rainfall is the key determinant of vegetation productivity in arid and semi-arid rangeland ecosystems (Ellis & Swift, 1988; Kgosikoma, 2012). According to Walter (1939) trees have a deeper root system compared to grasses and therefore there is a uniform state between trees and grasses in their access to water. When grasses are removed as a result of overgrazing, trees may access topsoil water and easily to germinate considerably thus leading to bush encroachment (Walter, 1939; Ward *et al.*, 2014).

In arid and semi-arid savannas trees may have positive and negative effects on their nearby environment. This ensures the establishment of sub-habitats under tree canopies which differ from that found in grasslands and exerts distinctive influences on the herbaceous layer (Belsky *et al.*, 1989; Anderson *et al.*, 2001; Smit, 2004; Abdallah *et al.*, 2008). The attributes of these profound effects depend on whether the encroaching woody species are trees or shrubs and their natural geographical characteristics. Comparatively, Angassa and Baars (2000) indicated that, in the Borana rangelands in the southern region of Ethiopia, the overall result of rangeland conditions was minimum under bush encroached rangeland than non-encroached rangelands. This, therefore, implies the fact that a definite definition of bush encroachment could not be supported with critical clarification of available information. To ensure reasonable evaluation of the effects (whether negative or positive) of bush encroachment on the structure and functioning of various ecosystems a broader concept of land degradation in the form of bush encroachment is necessary (Belayneh & Tessema, 2017).

2.4.1 Positive effects of trees on grasses

Woody plant encroachment has been observed to have positive impacts on the savanna ecosystems, however, this has not been generally accepted by livestock farmers (Kgosikoma & Mogotsi, 2013). In the African context, pastoralists have indicated that woody plants contribute considerably towards livestock feed, particularly during extensive drought periods thereby reducing the expense of supplementary feed (Moleele, 1998; Smit, 2004; Kgosikoma *et al.*, 2012; Kgosikoma & Mogotsi, 2013). In addition, leguminous woody

vegetation, such as *Senegalia mellifera* has the ability to enhance soil quality through nitrogen fixing and could also contribute considerably towards carbon sequestration, which leads to a improved ecosystem dynamics under the woody thickets (Kgosikoma & Mogotsi, 2013).

The positive relationships among plants have been shown to be crucial in various ecosystems (Greenlee & Callaway, 1996; Abdallah *et al.*, 2008). Trees have shown to improve the robustness of other plants by providing shade, increased soil water storages, soil nutrient availability and the protection smaller plants from severe weather conditions (Belsky *et al.*, 1989; Vetaas, 1992; Rhoades, 1996; Mishra *et al.*, 2003; Smit, 2004). Trees can also provide protection from herbivory, especially grasses that grow below or between woody vegetation (Callaway & Tyler, 1996). In arid and semi-arid environments that are affected by wind erosion, the grass growing under a suitable woody cover are shielded against the negative influences of the wind erosion (Montana *et al.*, 1991; Abdallah *et al.*, 2008). Depending on their density, trees can also reduce soil erosion and reverse desertification by protecting the soil from increased soil temperatures leading to soil crusting and increased water runoff rates (Young, 1989). Furthermore, the existence of woody plant species promotes unique habitats that support a larger variety of species including herbivores compared to other ecosystems without woody plants.

2.4.2 Negative effects of trees on grasses

According to De Klerk (2004), the negative effects of woody plant species on grasses include the fact that high bush densities results in unbalanced tree-grass ratios. This causes a reduction in rangeland carrying capacity which affects stocking rates and in turn compromise economic implications. Kahumba (2010) pointed out that, some indigenous woody encroacher species, such as *S. mellifera* and *Dichrostrachys cinerea* have increased considerably, therefore, negatively affecting the growth and development of perennial grasses throughout semi-arid savanna rangelands to the extent that small fragments of historic open savanna habitats prevail only where livestock grazing has been limited. Lubbe (2001) also reported that, the carrying capacity of more viable rangelands experiencing an average rainfall of 450 mm/a have been altered by woody plants to that which is drought sensitive and infertile dwarf shrub savanna having a rainfall of 200 mm/a, causing a so-called “pseudo drought”. Semi-arid African rangelands experience severe vegetation and

soil degradation due to heavy grazing causing negative impacts on ecosystems, livestock production and livelihood diversification (Kelkay, 2011).

The proliferation of encroacher woody species has been explained to be an extensive problem, generally affecting semi-arid and arid rangelands (Bews, 1917; Archer, 1990; Hoffman & O'Connor, 1999; Oba *et al.*, 2000; Smit, 2001; Gibbens *et al.*, 2005; Gemedo *et al.*, 2006b; O'Connor *et al.*, 2014), and the problem is most severe in the African continent. Bush encroachment has been linked to rangeland degradation and usually has negative impacts (Gemedo *et al.*, 2006a; Belayneh & Tessema, 2017). The causal linkages between encroachment of bush and rangeland degradation have been the focal point of scientific researches around the world, especially in the African continent, plainly because bush encroachment poses serious difficulties to the socio-economic development of rangelands (Vetter, 2005). Bush encroachment is acknowledged as the main aspect that aggravates rangeland degradation in arid and semi-arid areas globally (Belay *et al.*, 2013). More than 70% of rangelands have already been reported to be affected by severe rangeland degradation (Dregne & Choun, 1992; Belayneh & Tessema, 2017). Furthermore, the invasion of woody plant densities into openly grazed areas contribute mainly to rangeland degradation (Smit, 2001; Ward, 2005; Archer, 2010). Depending on the severity of bush encroachment and the nature of encroaching plants, bush encroachment could negatively alter the soil biological, physical and chemical characteristics of the local biodiversity of rangeland ecosystems (Biederman & Boutton, 2009; Belayneh & Tessema, 2017).

2.5 Bush encroachment in communal rangelands of South Africa

According to Hardin (1968) cited in Vetter (2005), communal land tenure is characterized whereby the natural resource base is communally owned by the residing individuals and where individual benefit is maximized at the expense of the available communal resource. Communal land rights have been and are still are exercised by native indigenous communities for several many years (Pienaar, 2008). Pienaar (2008) further stated that, communal land ownership tenure is defined in terms of its comprehensive inclusive nature and displays the following features:

- Land rights are confined in a range of social relationships (such as family and kinship networks) and diversified community memberships, often multiple and related in character.

- Land rights are comprehensive rather than restricted in nature, being shared and relative, but in most cases secure. In a specific community, rights may be individualised (residence), shared (hunting, trapping, fishing and grazing) or in combination of both (seasonal cropping combined with grazing and other social activities).
- Access to land is approved by norms and values incorporated in the communities' land ethics. This suggests that access through specific social rights differs from those being ruled by governing authorities and administrators.
- Community rights are developed from accepted associations of a public unit and can be acquired by birth, relationship/partnership, adherence or transactions. Political, social and resource use perimeters are usually fair, but often adaptable and variable, and can sometimes create tension and conflict.
- The balance of power between gender, contending communities, people who have the right to the land, land legislation authorities and traditional authorities is flexible.
- The constitutional flexibility and negotiability of land ownership rights means that they are well-suited to changing conditions, but vulnerable to confiscation by powerful external authorities (government officials) or superior investments.

Communal rangelands of South Africa and their surrounding homesteads occupy about 13% of the agricultural land surface in South Africa whereas 4.8% was identified as degraded land with 12.7 million people residing in these areas (Vetter *et al.*, 2006; Bennett & Barrett, 2007; Bennett *et al.*, 2012). The major challenge for the communal livestock system is the complexity of rangeland resource management due to multi-ownership of the resource (Sive, 2016). In this system, all communal members have an equal access to rangeland or have equal right to keep their livestock on the communal rangeland (Mapekula, 2009). Multi-ownership makes it difficult for communal farmers to make decisions on which rangeland management practice to use (Hardin, 1968). Increasing population pressure, invasion of rangelands by land use mismanagement, control of livestock diseases leading to high numbers of livestock and the failure of common resource administration structures are the main contributors of land degradation (Vetter, 2005). Livestock production is a foundation stone for rural livelihoods and livestock rely on these rangelands for their feed (Sive, 2016). The trend of livestock population in communal rangelands is currently expected to be significantly higher due to exponential growth of human population (Palmer & Ainslie, 2006). However, the transformation of grasslands and savanna to woodlands has a negative impact on livestock production (Sive, 2016). The suppressive effect of

encroaching woody species tend to reduce forage production which is valuable to livestock and also lead to the decline of profitability of rangeland in many communal rangelands of South Africa (Jacobs, 2000; Smit, 2004; Britz & Ward, 2007a).

Management of communal grazing systems has come under considerable assessment criteria concerning its fundamental ecological and economical negative assumptions. (Behnke & Abel, 1996; Sullivan, 1996; Sullivan & Rohde, 2002; Vetter, 2005). Poor norms concerning rangeland administration and absence of proper fences needed for rotational grazing have resulted into uncontrolled grazing which promotes under or over-grazing of certain areas (Sive, 2016).

2.6 The involvement of indigenous knowledge towards bush encroachment

Pastoralists have managed their production system for many centuries, accumulating detailed knowledge of the environment of their grazing landscape (Oba & Kotile, 2001; Mapinduzi *et al.*, 2003). Turner *et al.* (2000) suggested that, traditional knowledge of the successful pastoralists is fundamentally important in the management of local resources. According to Fernandez-Gimenez (2000), Angassa and Oba (2008) local ecological knowledge of rangeland resources have provided useful information for the development, sustainable utilization and conservation of natural resources (Abate, 2016). However, land degradation is largely as a consequence of poor land management by unsuccessful pastoralists (Tainton, 1999).

Communal farmers are known to have extensive indigenous knowledge which could complement scientific research (Barkes *et al.*, 2000; Oba & Kotile, 2001; Ladio & Lozada, 2009) and contribute to improved understanding and sustainable management of savanna ecosystems (Ladio & Lozada, 2009; Reed *et al.*, 2011). According to Angassa and Oba (2008), communal farmers can identify invasive and non-invasive species that threaten their rangelands. Bart (2006), Roba and Oba (2009) and Oba and Kaitira (2006) stated that the knowledge of farmers could be used to provide long term ecological view-points of bush encroachment or vegetation changes due to impacts, such as rainfall, fire and grazing management. Understanding the general causes of woody plant encroachment, therefore, depend upon an integrated approach that will ensure the application of combined ecological knowledge acquired through both indigenous and scientific communities (Van Auken, 2009; Sop & Oldeland, 2011). This approach also ensures that strategies adopted to

address the problem are economically, culturally and environmentally suitable for the local conditions (Kgosikoma & Mogotsi, 2013).

In the African context, there is limited long-term ecological data and the indigenous ecological knowledge on vegetation and other environmental changes accumulated through long-term observation and land use (Allsopp *et al.*, 2007) could complement the scientific knowledge by providing a long-term perspective on vegetation change and underlying causes (Bart, 2006). Despite the availability of this valuable resource, researchers and development experts have previously overlooked indigenous knowledge in the evaluation of rangeland (Abate, 2016). Roba and Oba (2008), Dabasso *et al.* (2012) and Abate (2016) have suggested that, integrating the knowledge of local communities would improve the current understanding of the mechanisms involved in range degradation. Other studies (Angassa & Oba, 2008; Roba & Oba, 2009) have indicated that a combination of pastoral indigenous knowledge and modern scientific information would help provide a better understanding of the environment from the perspective of resource utilization (Abate, 2016).

Most rangeland development projects have failed because they focused on addressing the technological aspect, without addressing the socio-economic factors (Kgosikoma & Mogotsi, 2013). Therefore, the use of both scientific and indigenous ecological knowledge ensures that a common goal is set and strategies (policy) adopted to curb bush encroachment also take into consideration the livelihood of that community. New grazing policies need to promote transparent decision making that is flexible to changing circumstances and embraces a diversity of knowledge and values. Given that factors such as rainfall and soil properties are not manipulative, management of bush encroachment needs to focus on regulating grazing pressure and optimum burning intervals (Kgosikoma & Mogotsi, 2013).

2.7 Bush control methods

Generally, encroacher woody species can be controlled mechanically (physical/manual clearing), chemically (use of arboricides) or biologically (use of natural agents) (Smit *et al.*, 1999; De Klerk, 2004). Chemical control of problematic woody species is the most active and rapid, potentially selective and virulent method of bush control. Mechanical clearing by means of felling and eradication of trees is most selective but time consuming

and may be costly if expensive machinery opposed to manual labour is used (De Klerk, 2004).

Smit *et al.* (1996) stated that, the main purpose of bush control is not the complete eradication of trees, but the reduction of tree numbers to an optimum level, thereby restoring the veld to a state of optimum grazing and browsing production which promotes ecological stability. The eradication of some or all the woody plants usually results in the re-establishment of grass species and thus increases grazing capacity (Pienaar, 2006). However, the outcome of woody plant eradication may vary between veld types, with the outcome determined by both negative and positive responses to tree removal. This is as a result of physical determinants, biological interactions and individual species characteristics which are unique to geographical and temporal conditions, especially in savanna rangelands (Pienaar, 2006). In addition, historic management practices are complex and result in various kinds and degrees of ecosystem modification (Teague & Smit, 1992).

The expansion of woody plant seedlings following the eradication of some or all the mature woody plants may reduce the time spent on bush control measures (Pienaar, 2006). In various cases the re-growth and development of new seedlings may eventually result into a state which is worse than before control (Smit *et al.*, 1999). To prevent the spread of problematic invasive species, scientific and conservative range management strategies are essential for all rangeland farmers. Bush control methods vary depending on the severity of the encroachment and resources available. Aftercare (post-treatments) after bush control is, however, important but due to the excessive costs, less implemented (Scifres, 1977; Trollope, 1978; Tainton, 1999; Barac, 2003; Kahumba, 2010; Mpati, 2015).

Veld management responses are adaptive, reactive or preventative in nature and depend on the state in which the rangeland is in (either transitional towards bush dominance where bush densities have already developed to a critical state or whether a desirable grassy state still prevails) (Joubert *et al.*, 2014). In cases where bush encroachment has already prevailed, reactive and expensive bush control measures must be implemented to restore damaged savanna ecosystems to a more desired grassy state. Natural self-thinning dynamics due to inter-specific competition or increased mortality rates due to environmental factors such as drought stress, maturity or fungal pathogens usually exceed an economically

reasonable time aspect competition (Scholes & Archer, 1997; Smit *et al.*, 1999; Joubert *et al.*, 2008; Joubert *et al.*, 2014; Lukomska *et al.*, 2014; Harmse *et al.*, 2016).

Bush control measures should adhere to the two key requirements before they can be considered successful (Pienaar, 2006). They must be ecologically reasonable and economically justifiable. In the South African context, based on these two principles, it is considered that very few attempts have been successful in controlling bush encroachment (Barac, 2003; Reynolds *et al.*, 2007; Angassa & Oba, 2008; Pienaar, 2006; Harmse, 2013). Campbell *et al.* (2000) suggested that, a control programme for the management of invasive plants must include tree phases, namely:

- Initial control: notable eradications of the problematic species (for example: cut tree, removal of plant biomass, control stumps, plant grass) with the aid of manual (hand) or aerial (arboricides applied by means of aeroplanes) are necessary.
- Follow-up control: after-care strategies must be implemented to prevent the establishment of woody seedlings, root suckers and coppice growth (production of re- growing individual stems on the remaining stratum).
- Maintenance control: sustaining unpalatable plant densities at relatively low frequencies with minimum annual control costs (example: burn high fuel loads of grass). In this stage, problematic invader species no longer persist.

However, is crucial to manage the areas previously affected by bush encroachment two or three times a year usually during spring, mid-summer and autumn periods to avoid re-encroachment, spread and advancements of undesired plants, thereby, reducing control costs (Campbell *et al.*, 2000).

According to Pienaar (2006) selection of appropriate control measures must consider the following factors:

- Plant species phenology (climatic and seasonal changes) and morphology (form and structural biological characteristics),
- densities of the undesired woody plant species,
- landscape,
- restoration specification,
- underground water and mineral resources availability and,

- the method and timing of control (herbicides can promote rapid thinning, biological control is slow but can be a permanent solution for some species).

Bush clearing methodologies such as mechanical control (tree cutting) as well as chemical control (herbicide application), generally affects wildlife habitat by altering vegetation structure (Smit, 2005; Kgosikoma, 2012). Fire as a bush management control tool is likely to achieve more favourable results, once applied to rangelands, as it destroys the majority of woody saplings beneath grasses, thereby, controlling the spread of problem species (Rothauge & Joubert, 2002). This will favour a more open state as opposed to a bush encroached one, which only happens occasionally (Kahumba, 2010).

The three methods used to control bush encroachment are discussed below:

2.7.1 Mechanical control

The mechanical clearing of trees is the most expensive methodology and is normally used together with other forms of control. These methods range from simply cutting down trees and shrubs by hand or stumping, ‘holting’ and using the ball and chain drag method with the employment of specialised equipment including heavy machinery such as tractors and bulldozers. The chopping down of woody plants using chain saws, machetes and axes is usually labour intensive and is not effective because of the selective thinning of woody species (Angassa & Oba, 2008; Kahumba, 2010; Mpati, 2015). Mechanical or manual control by chopping down woody plants temporarily creates open canopies without necessarily reducing woody plant densities (trees/ha) (Kahumba, 2010). Unless chemical treatments or ‘hot fires’ are applied on the remaining tree stumps, coppicing will occur. According to Lubbe (2001) most *Vachellia* species have the ability to coppice vigorously and if allowed to re-grow, various woody species will create an even denser thicket which that is worse than before. The use of fire, during rainy seasons, have similar effects as cutting but reduces stump coppicing ability (Kahumba, 2010).

Manual clearing methods such as the use of bulldozers are destructive and expensive (Tainton, 1999; Kahumba, 2010). Making use of bulldozers in rangelands dominated by large colonies of *Senegalia mellifera* generally aggravates, rather than solve the existing problem (Stoddart *et al.*, 1975; Kahumba, 2010). This is because using bulldozer causes a

disturbance in topsoil stability (De Klerk, 2004). Such excessive cost intervention is only recommended when thickets are to be replaced with cash crops or planted pastures. Due to the high cost involved, mechanical methods of controlling bush encroachment play only a limited role in most 'problem' areas (Todd & Hoffman, 1999; Mpati, 2015). According to Tainton (1999) and Kahumba (2010), the manual control of woody encroacher species is very specific, labour-intensive, time consuming and does not have lasting effects unless follow-up treatments are applied.

2.7.2 Biological control

The biological control of woody plants includes the incorporation of browsers and the spread of biological control agents effective for controlling woody plant species. In savanna rangelands, the use of mega-herbivores as biological control agents has been reported to produce effective and desired results (Kahumba, 2010). However, the use browsers together with controlled fire regimes have a significant and considerable impact on the control of bush densities (Kahumba, 2010). Biological methods, such as, the use of veld-fires and Boer goats or game may be used as a follow-up treatment in areas where bush densities have been reduced, rather than as an initial method of control (Trollope, 1978; Zimmermann *et al.*, 2003).

When using browsers to control bush densities, stock numbers must comply with the grazing capacity of that particular rangeland (Kahumba, 2010). Using browsers at the initial stage of bush control does not result in an effective impact on the biomass and densities of encroaching woody species. However, herbivores in savanna rangelands are important when incorporated in combination with other natural processes to maintain tree-grass balances (Van der Waal *et al.*, 2011). Angassa and Oba (2008) stated that, making use of herbivory alone does not reduce bush densities. Angassa and Oba (2008) further indicated that, the incorporation of herbivores is only successful in controlling woody sapling recruitment (Muroua, 2013; Mpati, 2015).

Stem burning using dung or brush is a very effective, non-expensive and a selective method of controlling bush densities. Stem burning, in which a low intensity fire smoulders (burn) for a long period of time around the base stem of the woody plant is effective in the selective control of individual trees (Pienaar, 2008). This method is not expensive and any available

fuel may be used, however, it is time consuming and labour intensive. Nonetheless, stem burning is an unsuitable method for controlling small plants or multi-stemmed trees (Smit *et al.*, 1999).

2.7.3 Chemical control

Chemical control is a primary choice when the woody layer is very dense and covers extensive areas (Smit *et al.*, 1999; Harmse, 2013). Herbicides are effective, however, the pros and cons should be considered before using them (Angassa & Oba, 2008; Kahumba, 2010; Muroua, 2013; Mpati, 2013). In the southern African region, the most popular chemical control practice among farmers is the use of systemic, photosynthesis-inhibiting arboricides, whose active ingredient enters the soil after rain, where it is absorbed by the roots of woody species and leads to death once carbohydrate reserves are exhausted (Du Toit & Sekwadi, 2012; Bezuidenhout *et al.*, 2015). Substituted urea arboricides, based on the non-selective Tebuthiuron as the active ingredient, are very common in use and highly effective in killing woody species at relatively low dosage rates (Moore *et al.*, 1985; Du Toit & Sekwadi, 2012; Harmse *et al.*, 2016). A major concern using Tebuthiuron pertains to its accumulation and persistence in the soil for several years, i.e. with potential long-lasting threats to also non-target species including herbaceous life forms. For example, Du Toit and Sekwadi (2012) observed Tebuthiuron-treated soil to exclude any plant recovery for at least eight years post-application in semi-arid South African grassland. In contrast, Haussmann *et al.* (2016) reported an immediate re-infestation of chemically cleared sites by an undesirable perennial shrub in the Highland savanna of Namibia. Moore *et al.* (1985) found Tebuthiuron-based arboricides to express some degree of selectivity for woody species from the semi-arid Kalahari, with obviously no detrimental effects on local grass species (Moore *et al.*, 1985; Richter *et al.*, 2001). The fate of Tebuthiuron in soils varies with the decomposition rate (half-life) and mobility in the soil as influenced by the content of clay particles and organic matter (adsorption), rainfall amounts (leaching), as well as soil temperatures (Du Toit & Sekwadi, 2012; Bezuidenhout *et al.*, 2015). In dryer areas with summer rain and well drained sandy soils of low organic matter, persistence of Tebuthiuron in at least the upper soil layers can thus be expected to be of only short duration, as would be the case in the Kalahari environments (Bezuidenhout *et al.*, 2015; Harmse *et al.*, 2016).

Bush control *via* chemical methods is unselective if a general-purpose herbicide is applied broadly, but can also be very selective if a specific herbicide is applied in spots (Tainton, 1999; Mpati, 2015). Chemical control is primarily suited for the initially thinning stage of bush control, although, it can be used in follow-up operation (Pienaar, 2008). Until the late 1950's, most herbicide use was as individual-plant treatments and available herbicides were not highly selective (Kahumba, 2010; Mpati, 2015). Chemical arboricides can be applied either selectively by means of spot control (application by hand/manually) and this is known as selective chemical bush control, or broadly by means of an aeroplane (aerial application) and this is known as the non-selective chemical bush control (Harmse, 2013; Harmse *et al.*, 2016).

2.8 Social studies

The common understanding of range dynamics by both researchers and communal farmers is very crucial because it improves livestock production through good quality forage production of the rangelands (Kessler & Stroonsnijder, 2010). Therefore, it is of paramount importance to understand how communal farmers perceive rangeland conditions and the extent of bush encroachment in communal areas (Sive, 2016). Mixed research can be conceptualized as combining qualitative or quantitative research in a concurrent, sequential, conversation (Tashakkori & Teddlie, 2003; Tashakkori & Teddlie, 2010), parallel (Onwuegbuzie & Leech, 2004) or fully mixed (Leech & Onwuegbuzie, 2005; Tashakkori & Teddlie, 2010) manner. Quantitative and qualitative approaches can be combined in these ways whether the study represents primary research (Johnson & Onwuegbuzie, 2004; Tashakkori & Teddlie, 2010) or mixed synthesis of the extent literature (i.e. integrating the findings from both quantitative studies in a shared area of empirical research) (Sandelowski *et al.*, 2006). Further, quantitative and qualitative approaches can be combined in these ways regardless of which approach has priority in the study (Creswell *et al.*, 2006; Onwuegbuzie & Johnson, 2006). Linked quantitative and qualitative methods also assist in the transition of social science information to predominantly natural science contexts, which can subsequently be applied to policy and management (Spoon, 2014). Quantitative research in environmental anthropology is on the decline, whereas, calls for the questioning of natural and social science data in the development of environmental policy are on the rise (Charnley & Durham, 2010). Quantitative social science data have the potential to be integrated more easily with natural science information, especially useful in research on coupled social-

ecological systems (Spoon, 2014). Furthermore, adding a qualitative component creates a thicker description assisting in the interpretation of the quantitative information so that it does not stand alone in a vacuum (Spoon, 2014).

CHAPTER 3

STUDY AREA

3.1. Location of the North West province in South Africa

The North West Province (NWP) is one of the nine provinces of South Africa and is located in the north-western part of the country (Figure 3.1). The NWP is located between 22 and 28 degrees longitude east of the Greenwich Meridian and between 25 and 28 degrees latitude south of the equator (Cowley, 1985). The province is situated at the centre of the northern border of South Africa and shares borders with four other Provinces in South Africa, namely Northern Cape, Free State, Gauteng and Limpopo, and with Botswana to the north (NWREAD, 2015). It is the fourth smallest province in South Africa and it consists of 57% arid and 43% semi-arid areas (Hoffman *et al.*, 1999; NWREAD, 2014) and it is also fringed by the Kalahari Desert to the west (NWREAD, 2014). The NWP lies at the heart of the Bushveld region, characterised by a generally flat savanna landscape (Figure 3.2). Its rich natural resource value includes mineral resources such as platinum and chromium, which has earned the province the trademark “The Platinum Province” of South Africa (NWREAD, 2015). The NWP occupies an area of 106 512 km² of South Africa (Acha, 2014). Most industrial activities, such as mining, is mainly in the southern region between Potchefstroom and Klerksdorp, as well as Rustenburg and the eastern region, where more than 80 percent of the province's economic activity takes place (NWPG, 2009).

3.2 Location and history of the Taung area

The NWP consists of four districts namely Dr. Ruth Segomotsi Mompati in the west, Ngaka Modiri Molema in the central parts, Bojanala Platinum in the eastern parts and Dr. Kenneth Kaunda in the south (NWREAD, 2015), see Figure 3.2. The Taung area, where the study was carried out, is part of the Dr. Ruth Segomotsi Mompati District Municipality. This municipality is 43 700 km² in size and is characterised by rural areas and remotely located settlements with many communities living in poverty (NWREAD, 2015). As per national census in 2011, the South African population was 51 770 561 and the population of the NWP was estimated at 3 509 953, the Dr. Ruth Segomotsi Mompati District municipality has a population of 463,815, which is 13.2% of the population of the NWP (NWP Integrated Development Plan, 2015).

Greater Taung is located near the following town and cities, i.e. 75km south of Vryburg, 25km from Dry-Harts location and 175km from Kimberley (Acha, 2014). About 95% of the Greater Taung municipal area is predominantly rural and there are approximately 106 scattered villages with an estimated population of 177 642 which represents 38.3% of the total population (Makgalemane & Mofokeng, 2011; NWP Integrated Development Plan, 2015). About 98% of the population in the Taung area is Setswana speaking and it is the head-quarters of Greater Taung Local Municipality (NWP Integrated Development Plan, 2015). Greater Taung was the area where the “Taung Child” was discovered and identified as the *Australopithecus africanus*, a predecessor of humans (GTLM, 2012).

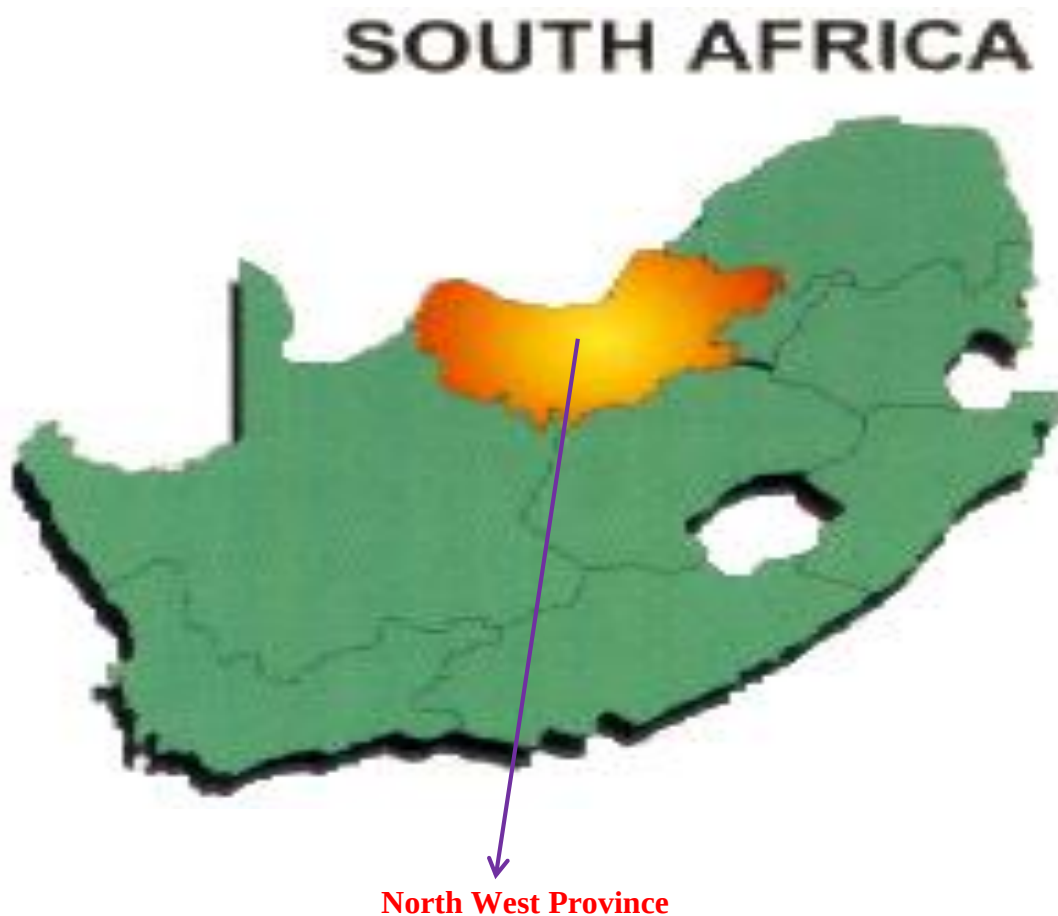


Figure 3.1: Map of South Africa with special reference to the North West Province.



Figure 3.2: The four main municipal districts in the North West Province, South Africa (source: www.mapsofworld.com).

3.3 Household structure, education and economic status of residents in the Taung area

The average size of households in the Taung area amounts to 4 persons per household. Most of the Taung population live in small, low-intensity settlements consisting of mostly informal housing that are scattered throughout the eastern parts of the municipal area. The Taung area has the highest population density within the Dr. Ruth S Mompati District Municipality (Acha, 2014). Most of the population (93%) reside within the Greater Taung Local Municipality and only 7% live in urban areas (Gender Statistics for South Africa, 2011; Acha, 2014). In 2010, 52.8% of households were headed by men as compared to 48.2% by females.

Regarding education, a substantial proportion of the population (75%) have completed at least some secondary school education. Currently only 31% have completed school and only 13 percent have tertiary education. The low education percentages have a negative impact on human development capacity in the area (Gender Statistics for South Africa, 2011; Acha, 2014).

According to Gender Statistics for South Africa (2011), agriculture is the main economic activity in the study area consisting mainly of cattle farming and intensive agriculture (irrigation) for crop production in the Great Harts River Valley.

3.4 Vegetation of the Taung area

According to Mucina and Rutherford (2006) the Greater Taung area forms part of the SVk4 Kimberley thornveld which is located under the savanna biome (Van Rooyen & Bredenkamp, 1996; Mucina & Rutherford, 2006). This savanna consists of plains vegetation with a continuous grassy layer and scattered layer of woody vegetation (Ward *et al.*, 2014). On the flatter and gently undulating plains, most of the woody plants are *Vachellia tortilis* trees with scattered *Senegalia mellifera* shrubs and occasional scattered *Vachellia erioloba* trees. On the slopes of the small sandstone hills, *Tarchonanthus camphoratus* shrubs are common (Mucina & Rutherford, 2006; Ward *et al.*, 2014). Scattered *Vachellia tortilis* trees also occur on the hillsides (Ward *et al.*, 2014). Woody species contributing to bush encroachment in this study are *Senegalia mellifera*, *Vachellia karroo*, *V. tortilis* and *Tarchonanthus camphoratus*. According to Acocks (1988), climax grass species include *Eragrostis superba* (Saw-tooth Love grass), *Cymbopogon plurinodis* (Narrow-leaved Turpentine grass), *Themeda triandra* (Red grass), *Heteropogon contortus* (Spear grass) and *Panicum coloratum* (Small Buffalo grass).

3.5 Rainfall and temperature of the Taung area

The area receives variable rain with scattered thunder storms and occasional flooding. During hot summers, there is high evaporation and elevated temperature (Tekana & Oladele, 2011; GTLM, 2011; Acha, 2014). Taung receives variable rainfall with an average of 322 mm/a, most rainfall occurring mainly during summer (October–April) (Acha, 2014). During hot summers, there is high evaporation and elevated temperature (Tekana & Oladele, 2011; GTLM, 2011). The region is the coldest during July when the mercury drops to 0.7°C on average during the night. Taung is characterised as a semi-arid area which occasionally experiences hail and frost (Acha, 2014).

Figure 3.3 indicates that the annual rainfall is highest during January and March. The average rainfall declines from March until September and increases again from September to December. The lowest rainfall occurs during June and July and the highest in January

(120 mm) and December (140mm). According to Figure 3.3, the average temperatures during midday range from 20°C in January to 30°C in March. The temperature then decreases from April (15°C) and gradually increases from September until December (41°C). The Taung area generally has high temperatures of about 40°C in February and December.

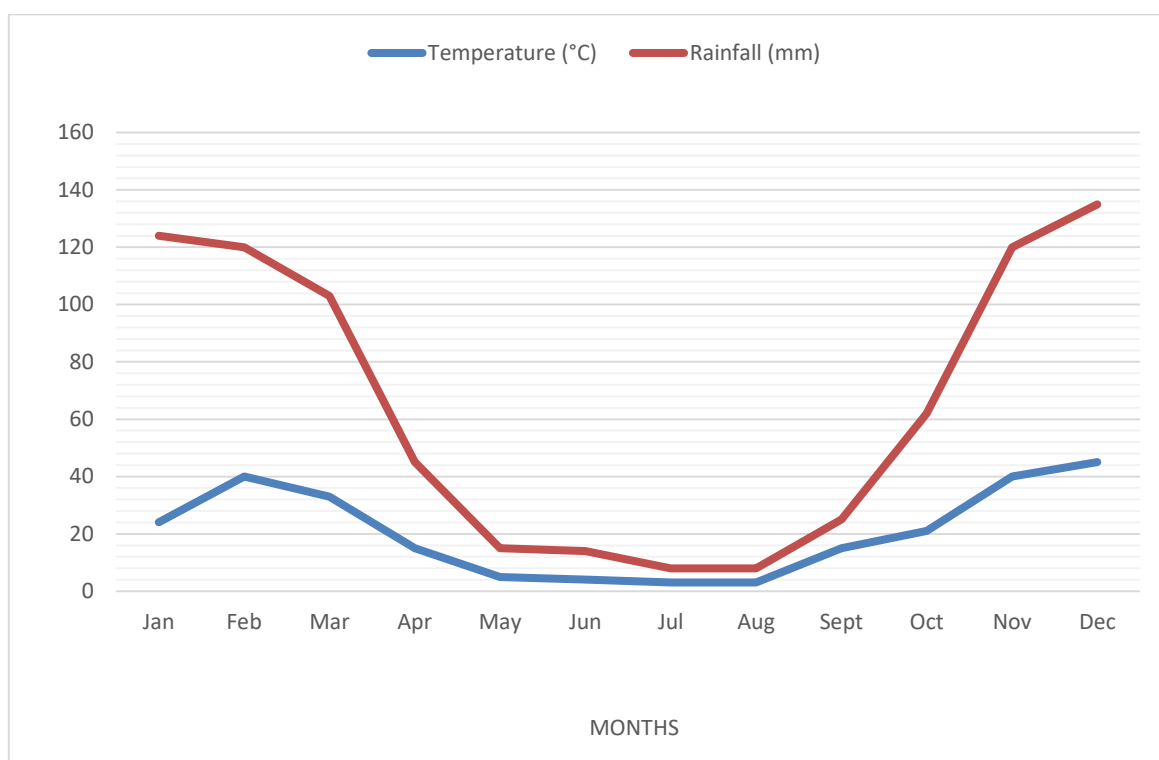


Figure 3.3: Annual rainfall and temperature ranges for the Taung area. The red trend line indicates annual rainfall and the blue trend line indicates the average annual temperatures.

3.6 Geology and soils of the Taung area

Soils of the Kimberly thornveld are made up of the andesitic lavas of the Allanridge Formation in the north and west and fine-grained sediments of the Karoo Supergroup in the south and east. They are deep (0.6-1.2 m) sandy to loamy soils of the Hutton soil form (Ae and Ah land types) on slightly undulating sandy plains (Mucina & Rutherford, 2006). According to Mucina and Rutherford (2006) influences on soil properties, such as, variation on a scale of as small as 0.25 ha may have a profound influence on the pattern and type of tree-grass coexistence in an area.

According to the Department of Agriculture, Forestry and Fisheries (DAFF) (2010), the soils are characterised mainly by calcareous types in the central to western parts and

eutrophic soils in the northern and north-western parts of Taung. Approximately 60% of the Taung area is Glenrosa and Miapah soils types (Acha, 2014). According to Greater Taung Local Municipality (GTLM) (2011), soils in Greater Taung Local Municipality are predominantly shallow and with outcrops, however deep red soils also occur mainly in the cultivated areas. The rocky and shallow outcrop areas are mostly utilised as natural veld for grazing. The deep red soil areas are suitable for dry land farming (Acha, 2014). The north-western part of Taung is characterised by red-yellow apedal soils which are prone to wind erosion and as a result of the low erratic rainfall, these soils are not cultivated and are, therefore, mostly utilised as natural veld (DrRSDM, 2013; Acha, 2014).

3.7 Demarcation and description of study sites

A total of eight study sites each representing controlled and uncontrolled areas (site) were demarcated by the WfW working teams for the purpose of this study (Numbered 1-8 in Figure 3.4). In each site, a benchmark site (uncontrolled sites), where no bush control was carried out and a site where bush control by WfW was conducted (controlled sites) were selected (Figure 3.4). The environmental conditions (soil, topography, climate and general habitat) of the uncontrolled and controlled sites were similar. The WfW working teams made use of chemical application of arboricides which was placed on the tree stumps that were cut manually (so called 'cut-stump' application) (see Chapter 4). In some places, the branches of the trees were placed on the bare soil patches (so called 'brush packing') to prevent further grazing and promote the rehabilitation process.

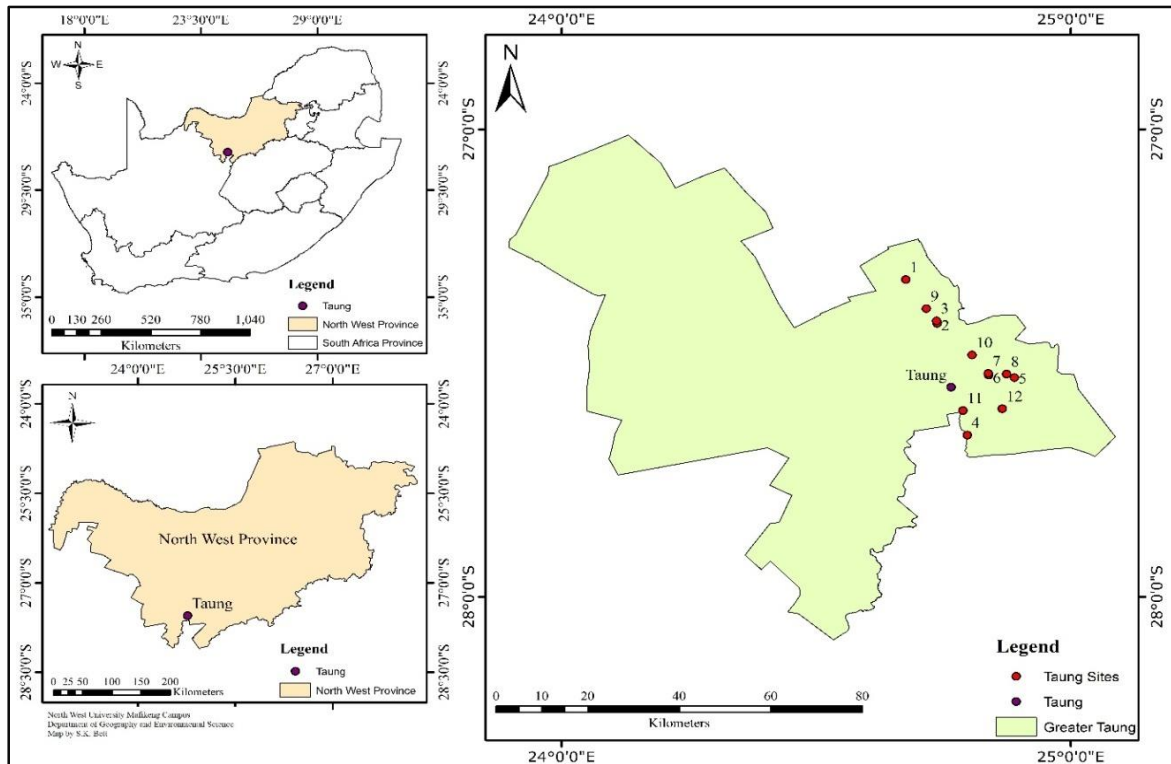


Figure 3.4: Location of the study sites in the Greater Taung Local Municipality District within the North West province of South Africa. (Moretele, Myra (87), Myra (76), Magogong, Manthe, Taung Dam (102), Taung Dam (98), Taung Dam (100), Pudumoe, Cokonyane, Maphoitsile, Pitsong).

Table 3.1: Location (Grid Reference) of study sites (controlled and uncontrolled) where the vegetation and soil surveys were conducted.

Number of study site according to Figure 3.4	Name of study site	Grid references of study sites by GPS coordinates
1	Moretele controlled site	(S 27° 19.267')(E 024° 40.576')
	Moretele uncontrolled site	(S 27° 19.435')(E 024° 40.173')
2	Myra (87) controlled site	(S 27° 24.894')(E 024° 44.266')
	Myra (87) uncontrolled site	(S 27° 25.046')(E 024° 44.393')
3	Myra (76) controlled site	(S 27° 24.566')(E 024° 44.204')
	Myra (76) uncontrolled site	(S 27° 24.522')(E 024° 44.184')
4	Magogong controlled site	(S 27° 39.238')(E 024° 47.822')
	Magogong uncontrolled site	(S 27° 39.095')(E 024° 47.870')
5	Manthe controlled site	(S 27° 31.858')(E 024° 53.375')
	Manthe uncontrolled	(S 27° 31.767')(E 024° 53.491')
6	Taung Dam (102) controlled site	(S 27° 31.536')(E 024° 50.346')
	Taung Dam (102) uncontrolled site	(S 27° 31.502')(E 024° 50.374')

7	Taung Dam (98) controlled site	(S 27° 31.314') (E 024° 50.294')
	Taung Dam (98) uncontrolled site	(S 27° 31.320') (E 024° 50.143')
8	Taung Dam (100) controlled site	(S 27° 31.400') (E 024° 52.463')
	Taung Dam (100) uncontrolled site	(S 27° 31.438') (E 024° 52.498')

3.7.1 Moretele

This site is characterised by dense stands of *S. mellifera* shrubs and spars shrubs. There is also a mixture of other woody species such as *T. camphoratus*, *V. tortilis*, *V. karroo* and *Z. mucronata* occurring at low densities. The terrain of the area is shallow sandy soils with dolerite intrusions and a stony surface.

3.7.2 Myra

There were two sites in Myra namely Myra (87) and Myra (76):

3.7.2.1 Myra (87)

The Myra (87) site is located near a mountain slope with rocky outcrops. The site is characterised by high densities of *Tarchonanthus camphoratus* shrubs and spars shrubs followed by *Vachellia tortilis*, *V. karroo* as well as low densities of *Senegalia mellifera*, *Ziziphus mucronata* and *Diospyros lycioides*. Both annual grasses and perennial grasses occur in controlled and uncontrolled sites. These sites have sandy soils with a rocky terrain.

3.7.2.1 Myra (76)

This site is situated just opposite the Myra (87) study site. The division between the Myra (76) and (87) study sites is a vast mountain slope. Most prominent woody species occurring in the Myra (76) survey sites are high densities of *T. camphoratus* shrubs and spars shrubs followed by *V. tortilis*, *V. karroo* as well as low densities of *S. mellifera*, *Ziziphus mucronata* and *Diospyros lycioides*. However non-encroacher woody species such as *Searsia lancea* and *S. burchellii* were also present. A mixture of both annual and perennial grass species was evident in the site.

3.7.3 Magogong

Magogong is characterised by a mixture woody encroacher species such as *V. karroo* and *V. tortilis* followed by lower densities of *T. camphoratus* and *Grewia flava*. The non-woody *Opuntia ficus-indica* was also present.

3.7.4 Manthe

The controlled and uncontrolled survey sites are located near the Manthe village. This area is characterised by bare rocky soils with scattered woody plants such as *V. tortilis*, *T. camphoratus* and low densities of *S. mellifera*. The non-woody exotic species such as *Opuntia spinosior* and *O. ficus-indica* were also present in the area. Soil erosion by water runoff was evident. Wind erosion as a result of bare soil patches was also evident in the area.

3.7.5 Taung Dam

There were three sites in the Taung Dam area namely Taung Dam (102), Taung Dam (98) and Taung Dam (100). The soils in the Taung Dam area are shallow sandy soils with rock intrusions. Vegetation in the Taung Dam (102) uncontrolled was sampled along a rocky mountain slope. Annual and perennial grasses are present in the area. Pseudo soil erosion was evident in the Taung Dam (98) area.

3.7.5.1 Taung Dam (102)

The Taung Dam (102) survey site was characterised by dense stands of *S. mellifera* shrubs and spars shrubs *V. tortilis*, *G. flava* and *Ziziphus mucronata* were also present. The non-woody exotic species *O. spiriosibacca* and *O. ficus-indica* were also present. Re-growth of *S. mellifera* and *V. tortilis* was evident in the controlled sites (See Chapter 5, Figure 5.11).

3.7.5.2 Taung Dam (98)

The Taung Dam (98) survey site is characterised by *S. mellifera* shrubs and spars shrubs. Other woody encroacher species such as *V. tortilis*, *T. camphoratus* and *V. karroo* were also present.

3.7.5.3 Taung Dam (100)

The prominent woody species in the Taung Dam (100) survey sites were *S. mellifera*, *V. tortilis* and *T. camphoratus*. *O. ficus-indica* was also present in the area. This is an indication that both annual and perennial grass species are present in the area.

CHAPTER 4

MATERIALS AND METHODS

4.1 Introduction

Bush controlled and uncontrolled sites (Sites 1-8, Figure 3.4) were selected within the Taung area in the North West Province. The woody and herbaceous (mainly grasses) components were assessed at each of the sites and a social survey was carried out at the local community living near the sites. The bush controlled sites refer to the sites in which bush clearing methods were implemented by WfW programme (Chapter 1, Section 1.7). The abundance of the main woody species was recorded over a period of two years in controlled and uncontrolled sites, i.e. during April 2014 and April 2015 as well as in September 2015. Both mechanical (manual cutting) and chemical application methods were applied during the bush clearing (control) projects implemented by the WfW programme in 2011.

The control methodology included the cutting of tree stems using chain saws and axes after which chemical arboricides were applied on the cut stumps. No follow-up strategy was implemented by WfW in any of the controlled sites. No clearing (control) of the woody species was applied in the uncontrolled sites. The latter sites were used as reference sites (benchmark sites) and compared with the bush controlled sites. The woody and herbaceous components and certain chemical and physical parameters of the soil components were monitored at each site.

4.2 Woody Component

The belt transect method (Yamamoto *et al.*, 2011; Teshome *et al.*, 2012) was carried out in a 400m² (4m × 100m) area. Two, one hundred meter measuring tapes were used for the two transects. A rod of 2m was placed on both sides of the measuring tape used for the 100m transects to get the 400m² belt. The two transects were placed 40m parallel from each other in the same habitat characterised by similar soil and other environmental conditions. The 2m rods were marked at every centimeter and used to measure the height of the individual woody species. The different growth forms within the 400m² belt transect were also monitored. The growth forms of the individual woody species were categorized as follows:

- Shrubs – individual woody species with more than 5 stems at the ground surface,
- Sparse shrubs – individual woody species with less than 5 stems at the ground surface,

- Trees – individual woody species with one stem irrespective of height class.

The density of woody species per height class was counted within the 400m² belt transect. The type of control, rate of control on a scale of 0% - 100% (according to the amount of stems cut, e.g. 0% only a few stems were cut and 100% - all stems were cut) and the rate of re-growth (coppicing) on a scale of 1-5 (1-little re-growth (<10%) to 5 – much re-growth (>75%) of the woody individuals were recorded. All the information was recorded on the data sheet (Appendix A). Woody plant densities were expressed in total of individuals of each species per 1 hectare area.

4.3 Grass component

The frequency (%) of the grass species was determined using the descending point method on the 100m transect (Kioko *et al.*, 2012; Joubert & Myburgh, 2014). Grass species composition was recorded at every one meter distance on the two parallel 100m transect lines. Only the nearest plant in a radius of 30cm was recorded. If no grass tuft was within the 30cm radius, bare ground was recorded. All the data was recorded on the data sheet designed for the herbaceous component of the study (Appendix B).

4.4 Soil component

Three soil samples per site were collected at a depth of 30cm (Joubert & Myburgh, 2014) using a soil auger. The three samples were mixed to form one composite sample per site. Soil samples were analysed for physical and chemical properties by the Eco-Analytica soil analysis laboratories of the NWU¹.

All soil samples were air dried before analytical analysis was carried out. According to Soil Survey Staff (2014), soil samples must be dried out before analysis as the moisture in soil can influence the chemical properties of the soil. A short description of soil parameters and methodology used to test the chemical and physical properties is described below:

¹ Eco Analytica P.O.Box 19140 Noordburg 2422

4.4.1 Soil pH

Soil pH (water) and pH (potassium chloride (KCl)) both in 1:2:5 solution (10 g soil and 250 ml water/ KCl) (McLean, 1982; Kalra, 1995; Huluka, 2005) were carried out. In the case of pH (water) being lower than 4.5, the lime requirement testing, according to the agricultural method, was carried out (Bailey *et al.*, 1991).

4.4.2 Soil carbon and nitrogen

Total carbon (C) present in the soil was determined by means of the LECO method (Leco Corporation, 2002; Leco Cooperation, 2003). Soil nitrogen (N) was calculated by means of total N, using Phosphate Bray-1 and NH₃ methods (Woudenberg *et al.*, 2010).

4.4.3 Electrical conductivity (EC) and cation-exchange capacity (CEC)

Electrical conductivity was performed on a saturated extract of the soil (Khorsandi & Yazdi, 2011). The soil cation-exchange capacity (CEC) was done using Ammonium acetate at pH 7 (not 2:1 solution) was used to perform the test for exchangeable cation capacity (Rowell, 2014).

4.4.4 Base saturation

Base saturation was determined using the methods of BaCl₂ – TEA extractable acidity and NH₄OAc- extractable bases (Holmgren & Nelson, 1977; Soil Survey Staff, 2014).

4.5 Social surveys

A total of 79 respondents were interviewed individually in six villages within the Taung area near the study sites. The rural villages visited were Pudumoe, Cokonyane, Maphoitsile, Pitsong, Taung and Manthe (see Chapter 3, Figure 3.4). Small-hold cattle farmers were targeted because the main aim of the study was to investigate how bush encroachment and control methods affected the pastoral activities in the area and to what extent it influenced the social behaviour of the pastoralists.

A questionnaire (Appendix C) was used to conduct social surveys. Both open-and closed ended questions were used to gather social information. Open ended questions refer to the

questions in which the respondents were required to elaborate or explain their answers. Close ended questions refer to questions wherein only numerical values are required e.g. number of cattle, the age or status of education.

The questionnaire was divided into two sections namely, (A) personal information of the respondents and (B) environmental information. In section (A) of the questionnaire, the respondents were asked general questions such as age, gender, marital status, education status and household livelihood (access to water, electricity and medical benefits). In section (B), the respondents were asked to provide information based on their environmental constraints such land use practices, cattle and livestock farming practices, introduction of eco-rangers, the role of the WfW project in their area. Respondents were also required to provide an input on how their area could be improved, as well as what impacts the bush encroachment problem had in their residential area. The respondents were also asked about the acceptance and roles of eco-rangers if introduced in their area.

Although, the interviews were conducted individually, the results were grouped in various categories and expressed in percentages. The data were then compared with the results reported in other research studies (Musara *et al.*, 2010; Gender Statistics for South Africa Report, 2011).

CHAPTER 5

RESULTS AND DISCUSSION OF CHANGES IN WOODY ABUNDANCE AND GRASS ABUNDANCE FREQUENCIES IN BUSH CONTROLLED AND UNCONTROLLED SITES

5.1 Abundance of woody species in bush controlled and uncontrolled sites

The abundance of the main woody species was recorded over a period of two years in controlled and uncontrolled sites, i.e. 2014 and 2015. The most prominent woody species in the study sites were *Senegalia mellifera*, *Vachellia tortilis*, *V. karroo* and *Tarchonanthus camphoratus*. In the controlled sites, *V. tortilis* showed a decrease in plant density in 88% of the sites and *V. karroo* in 50% of the sites, see Table 5.1.

Table 5.1: Species increasing and decreasing in uncontrolled and controlled study sites, coppicing in controlled sites, and the control effectiveness. (Key: *D.* = *Diospyros*, *S.* = *Senegalia*, *Se.* = *Searsia*, *V.* = *Vachellia*, *T.* = *Tarchonanthus*)

Site	Uncontrolled Site		Controlled Site			Control Result
					Species Coppicing	
	Species decreasing	Species increasing	Species decreasing	Species increasing		
Site 1 Moretele	<i>S. mellifera</i> <i>V. tortilis</i>	<i>V. karroo</i> <i>T. camphoratus</i> <i>Z. mucronata</i>	<i>V. tortilis</i> <i>Z. mucronata</i>	<i>S. mellifera</i> <i>T. camphoratus</i> <i>V. karroo</i>	<i>V. karroo</i> <i>V. tortilis</i> <i>S. mellifera</i>	-
Site 2 Myra (87)	<i>V. tortilis</i> <i>S. mellifera</i> <i>T. camphoratus</i> <i>D. lycioides</i>	<i>V. karroo</i>	<i>V. tortilis</i> <i>Z. mucronata</i> <i>S. mellifera</i> <i>D. lycioides</i>	<i>T. camphoratus</i> <i>V. karroo</i>	<i>V. karroo</i> <i>V. tortilis</i> <i>T. camphoratus</i> <i>S. mellifera</i>	+
Site 3 Myra (76)	<i>T. camphoratus</i> <i>V. tortilis</i> <i>V. hebeclada</i> <i>Se. burchellii</i> <i>Se. lancea</i>	<i>S. mellifera</i> <i>V. karroo</i>	<i>T. camphoratus</i> <i>V. tortilis</i>	<i>V. karroo</i> <i>S. mellifera</i> <i>Z. mucronata</i>	<i>V. karroo</i> <i>T. camphoratus</i> <i>S. mellifera</i>	-
Site 4 Magogong	<i>V. tortilis</i>	<i>V. karroo</i> <i>Z. mucronata</i>		<i>V. karroo</i> Dead tree <i>V. tortilis</i>	<i>V. tortilis</i> <i>V. karroo</i>	+

Site 5 Manthe	<i>V. tortilis</i> <i>Z. mucronata</i>	<i>T. camphoratus</i> <i>B. albitrunca</i>	<i>T. camphoratus</i> <i>D. lycioides</i> <i>S. mellifera</i>	Dead tree <i>V. tortilis</i> <i>V. hebeclada</i>	<i>V. tortilis</i>	+
Site 6 Taung Dam (102)	<i>S. mellifera</i> <i>V. tortilis</i> <i>G. flava</i> <i>Se. burchellii</i> <i>T. camphoratus</i> <i>Z. mucronata</i>	<i>B. albitrunca</i>	<i>S. mellifera</i> <i>V. tortilis</i> <i>G. flava</i>	<i>Z. mucronata</i> Dead tree	<i>S. mellifera</i> <i>V. tortilis</i>	+
Site 7 Taung Dam (98)	<i>V. tortilis</i>	<i>S. mellifera</i> <i>T. camphoratus</i> <i>V. karroo</i> <i>Z. mucronata</i> <i>D. lycioides</i>	<i>T. camphoratus</i> <i>V. tortilis</i>	<i>S. mellifera</i> <i>V. karroo</i> <i>Z. mucronata</i> <i>D. lycioides</i>	<i>S. mellifera</i> <i>T. camphoratus</i> <i>V. karroo</i> <i>V. tortilis</i>	-
Site 8 Taung Dam (100)	<i>V. tortilis</i>	<i>S. mellifera</i> <i>T. camphoratus</i> <i>V. karroo</i>	<i>V. tortilis</i> <i>Z. mucronata</i>	<i>S. mellifera</i> <i>T. camphoratus</i> <i>V. karroo</i>	<i>V. tortilis</i> <i>T. camphoratus</i> <i>Z. mucronata</i>	-

5.2 Changes in woody plant abundance for each study site

The abundance (density) of woody species per hectare, as well as the coppicing rate of the woody species (re-growth of individual species per hectare) will be discussed for each study site below.

5.2.1 Moretele

5.2.1.1 Controlled sites

Senegalia mellifera was present at a density of 1716 individuals/ha in 2014 and increased to 2600 individuals/ha surveyed in 2015 (Figure 5.1 and Table 5.1). The same results were obtained for *T. camphoratus* which increased in density from 26 individuals/ha to 208 individuals/ha. *V. karroo* increased from an absence of specimens to a density of 247 individuals/ha, see Table 5.1. 468 *V. tortilis* individuals/ha were present in 2014 and this density decreased to 65 individuals/ha in 2015. Similar results were obtained for *Z. mucronata* which decreased from a density of 156 individuals/ha to 52 individuals/ha (Figure 5.1).

5.2.1.2 Uncontrolled sites

The density of *S. mellifera* in Moretele uncontrolled site decreased from 3341 individuals/ha in 2014 to 2983 individuals/ha in 2015 (Figure 5.1). There was an increase in density of *T. camphoratus* from 13 individuals/ha in 2014 to 390 individuals/ha in 2015 and *Ziziphus mucronata* from an absence to 26 individuals/ha in 2015 (Figure 5.1). *V. karroo*, however, increased from a total absence in 2014 to 260 individuals/ha in 2015 (Figure 5.1).

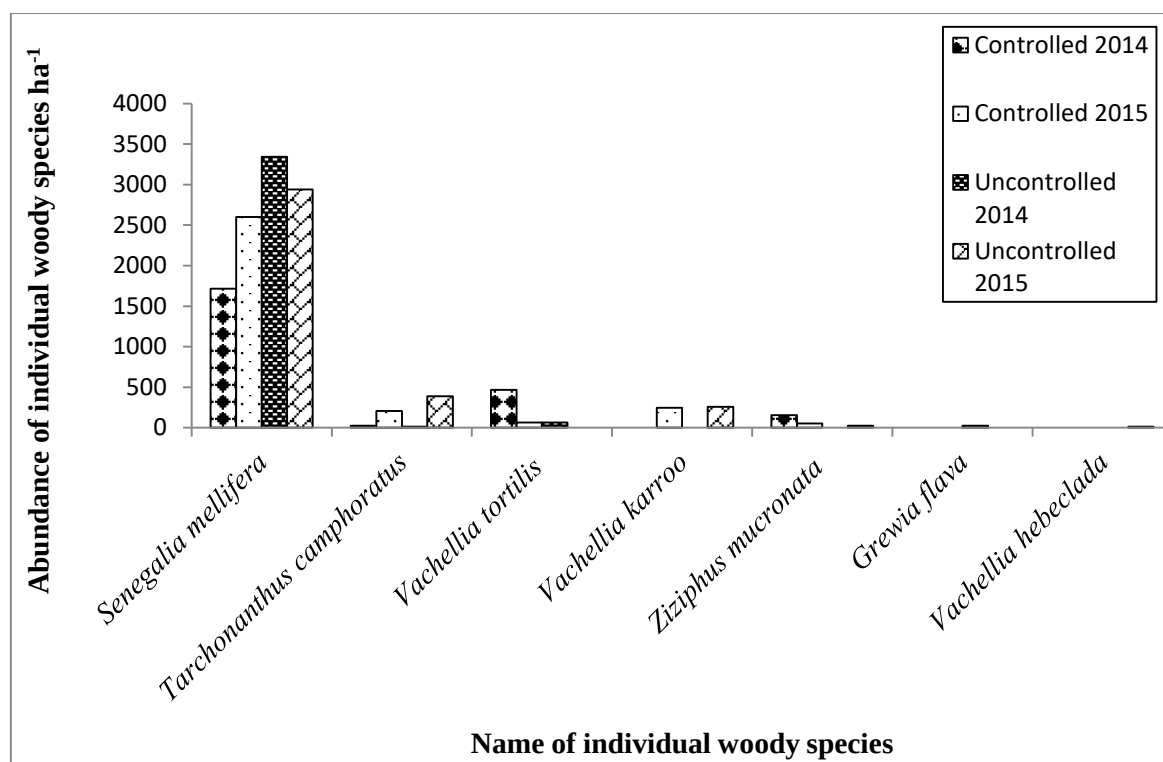


Figure 5.1: Abundance of individual woody species per hectare in the controlled and uncontrolled study sites at Moretele from 2014 and 2015.

5.2.1.3 Coppicing of woody species after control

Coppicing (re-growth) of *V. karroo* was measured at 11%, 20% by *V. tortilis* and 1% by *S. mellifera* trees (Figure 5.2).

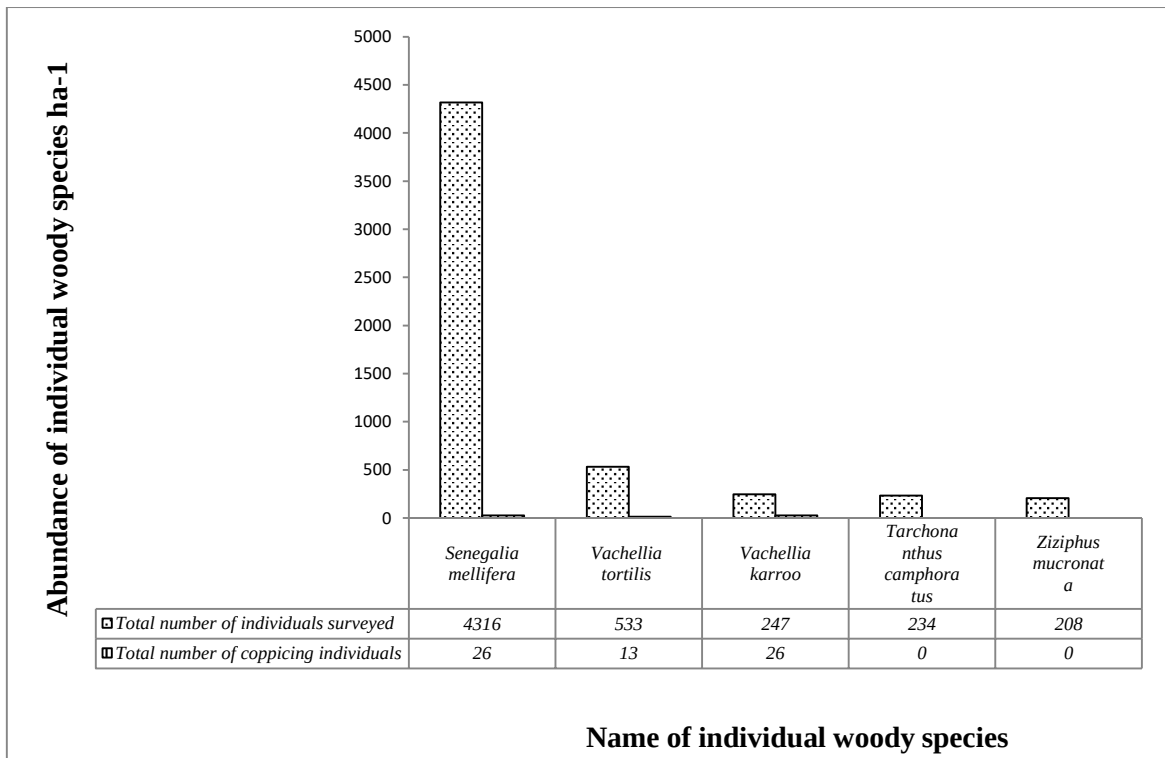


Figure 5.2: Abundance of coppicing individuals per hectare compared to the total number of species surveyed in the Moretele study site that was controlled.

5.2.1.4 Success of woody plant control in Moretele controlled site

Table 5.1 indicates that the control implemented at the Moretele study site was not successful due to the increase of *V. tortilis*, *S. mellifera* and *T. camphoratus*. The control of woody species in Moretele can be considered as negative (not successful), because of the high re-growth of, especially *V. tortilis* (Table 5.1).

5.2.2 Myra (87)

5.2.2.1 Controlled sites

The density of *V. tortilis* decreased from 1196 individuals/ha in 2014 to only 351 individuals/ha in 2015 and *Z. mucronata* from 26 individuals/ha to 13 individuals/ha in 2015. *S. mellifera*, however, decreased from 130 individuals/ha to 78 individuals/ha during the same time (Figure 5.3). Figure 5.3 also indicates that *T. camphoratus* increased from 507 individuals/ha in 2014 to 936 individuals/ha in 2015. In 2014 *Diospyros lycioides* was present at a density of 78 individuals/ha and decreased to a density of 13 individuals/ha surveyed in 2015. *V. karroo*, however, increased from an absence in 2014 to a total of 728 individuals/ha in 2015 (Figure 5.3).

5.2.2.2 Uncontrolled sites

T. camphoratus had a density of 1417 individuals/ha in 2014 and decreased to 819 individuals/ha surveyed in 2015 (Figure 5.3). *S. mellifera* decreased from 104 to 39 individuals/ha in 2015, while *V. karroo* increased from an absence in 2014 to a density of 702 individuals/ha in 2015 (Figure 5.3). *V. tortilis*, surveyed in 2014 was present at a density of 1248 individuals/ha and decreased to 559 individuals/ha in 2015 (Figure 5.3).

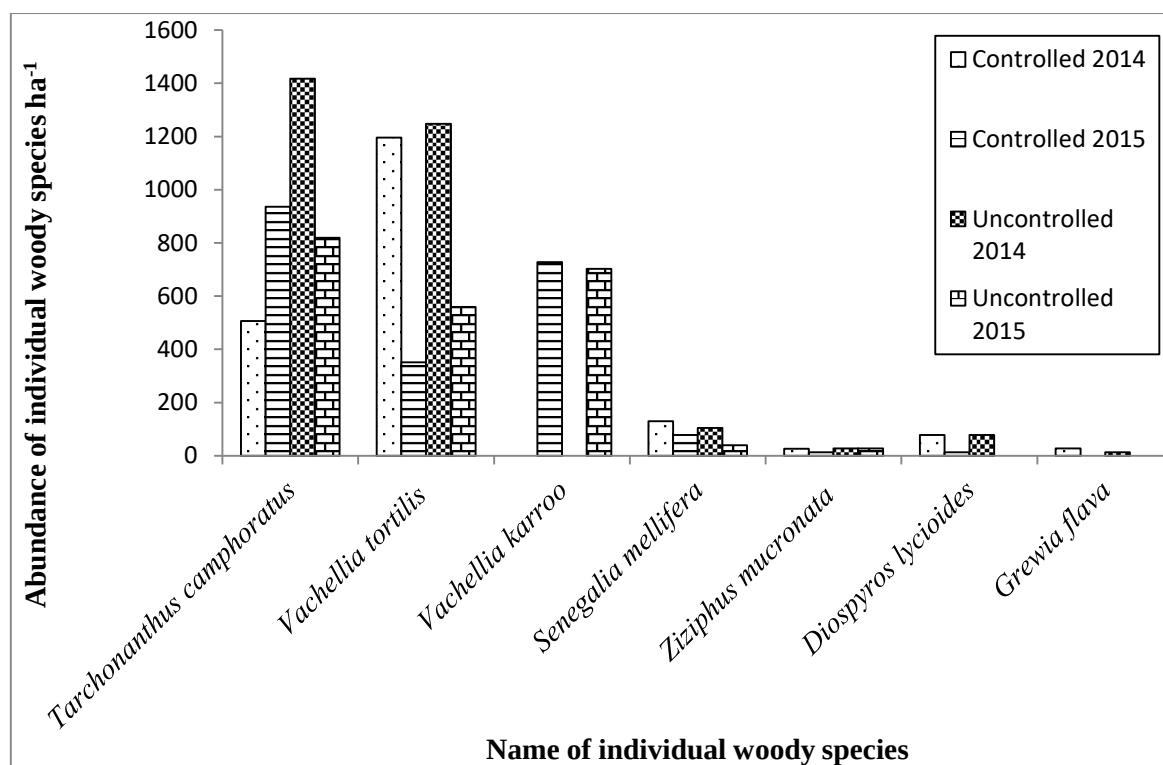


Figure 5.3: Abundance of individual woody species per hectare in the controlled and uncontrolled study sites at Myra (87) from 2014 and 2015.

5.2.2.3 Coppicing of woody species after control

The highest coppicing percentage was that of *V. karroo* recorded at 32%, followed by *V. tortilis* and *S. mellifera* at 27% and 13% respectively. *T. camphoratus* had the lowest coppice percentage of 9% (Figure 5.4).

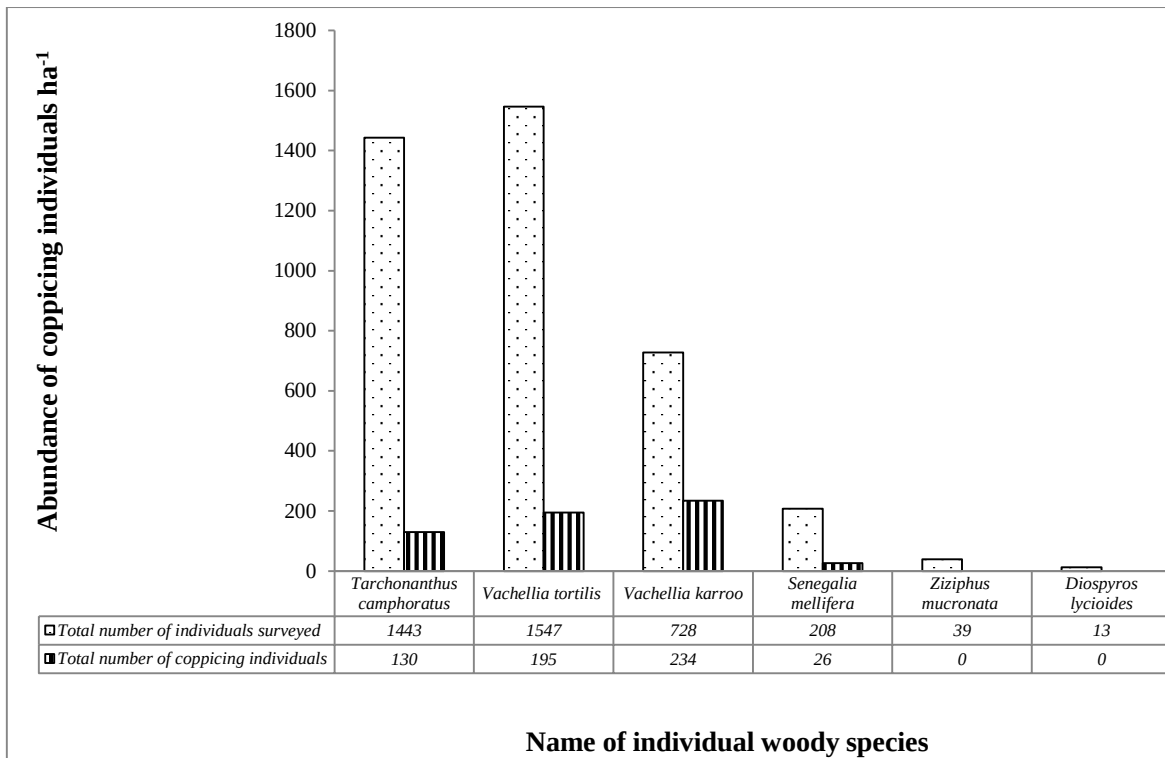


Figure 5.4: Abundance of coppicing individuals per hectare compared to the total number of species surveyed in the Myra (87) study site that was controlled.

5.2.2.4 Success of woody plant control in Myra (87) controlled site

Table 5.1 indicates that the woody control in Myra (87) was successful because of the decrease in density of *V. tortilis*, *Z. mucronata*, *S. mellifera* and *D. lycioides*. High coppicing of *V. karroo* and *V. tortilis* trees was, however, noticed (Table 5.1).

5.2.3 Myra (76)

5.2.3.1 Controlled sites

T. camphoratus trees were present at a density of 1339 individuals/ha in 2014 and decreased to 1079 individuals/ha in 2015 (refer to Figure 5.5). Likewise, *V. tortilis* decreased from 559 individuals/ha in 2014 to 65 individuals/ha in 2015. *V. karroo* was absent in the controlled sites in 2014 and increased to 455 individuals/ha in 2015, while *S. mellifera* trees increased from 13 individuals/ha in 2014 to 65 individuals/ha in 2015. *Z. mucronata* also increased from 13 individuals/ha in 2014 to 52 individuals/ha in 2015 (Figure 5.5).

5.2.3.2 Uncontrolled sites

S. mellifera trees increased from a density of 39 individuals/ha in 2014 to 130 individuals/ha in 2015, while *T. camphoratus*, *V. tortilis*, *Diospyros lycioides* and *V. hebeclada* showed a decrease in density from 2014 to 2015 (Figure 5.5).

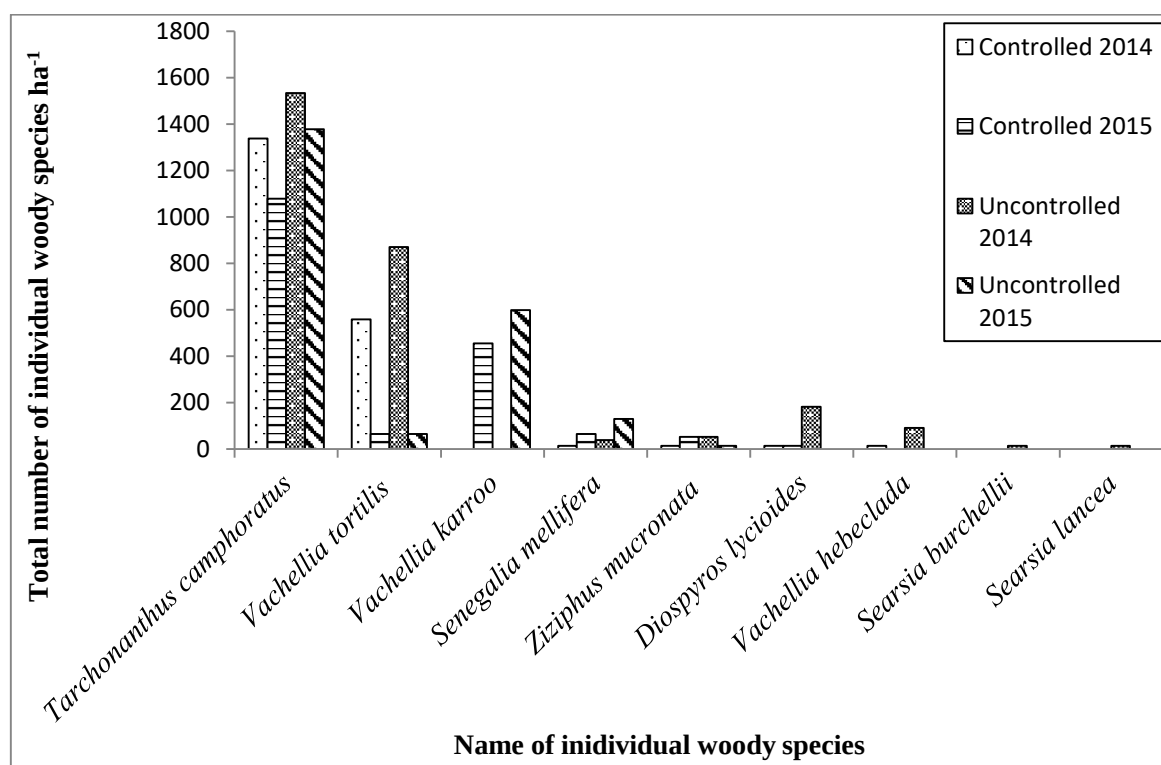


Figure 5.5: Abundance of individual woody species per hectare in the controlled and uncontrolled study site at Myra (76) from 2014 and 2015.

5.2.3.3 Coppicing of woody species after control

The results in Figure 5.6 indicates that *V. karroo* had the highest re-growth percentage (40%), followed by *S. mellifera* at 17% and *T. camphoratus* having the lowest coppice percentage of 2%. The high re-growth of *V. karroo* was as a result the previously present woody individuals that were not properly controlled.

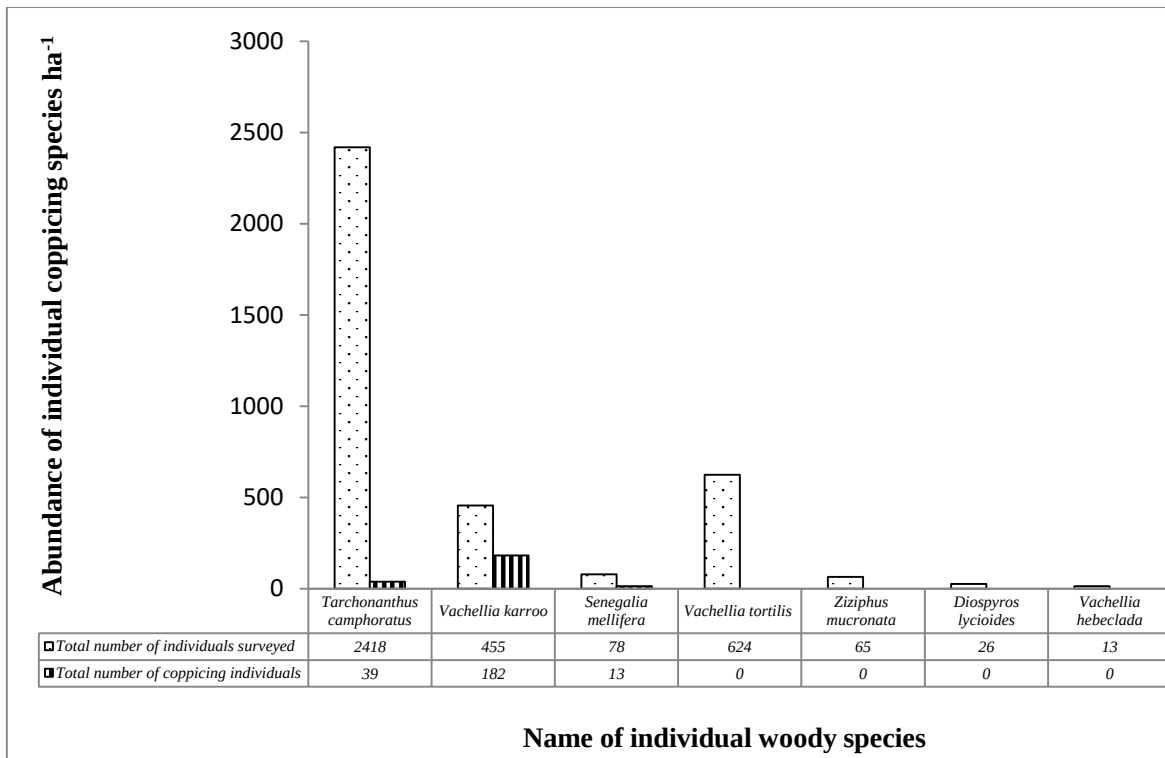


Figure 5.6: Abundance of coppicing individual per hectare compared to the total number of species surveyed in the Myra (76) study site that was controlled.

5.2.3.4 Success of woody plant control in Myra (76) controlled site

According to Table 5.1 the control implemented by the WfW working teams in Myra (76) controlled site could be considered unsuccessful due to the increase of *V. karroo*, *S. mellifera* and *Z. mucronata*. Although, Table 5.1 indicates that the density of *T. camphoratus* decreased, the re-growth (coppicing) of this species was evident.

5.2.4 Magogong

5.2.4.1 Controlled sites

The results obtained in the Magogong controlled survey site revealed that there was a total of 1872 *V. tortilis* individuals/ha in 2014 and 1989 individuals/ha in 2015 (Figure 5.7). The density of *V. karroo* increased from 949 individuals/ha in 2014 to 1599 individuals/ha in 2015. It was also observed that in 2014, there was no evidence of any dead trees in the Magogong controlled survey site, however in 2015, 234 individuals/ha of dead trees were recorded (Figure 5.7).

5.2.4.2 Uncontrolled sites

The density of the woody encroacher, *V. tortilis* was recorded at a density of 1274 individuals/ha in 2014 and 1170 individuals/ha in 2015 (Figure 5.7). *V. karroo* was present at a density of 1248 individuals/ha in this site in 2014 and increased to a density of 1885 individuals/ha in 2015 (Figure 5.7). *Z. mucronata* also increased in density from 13 individuals/ha to 130 individuals/ha from 2014 to 2015 (refer to Figure 5.7).

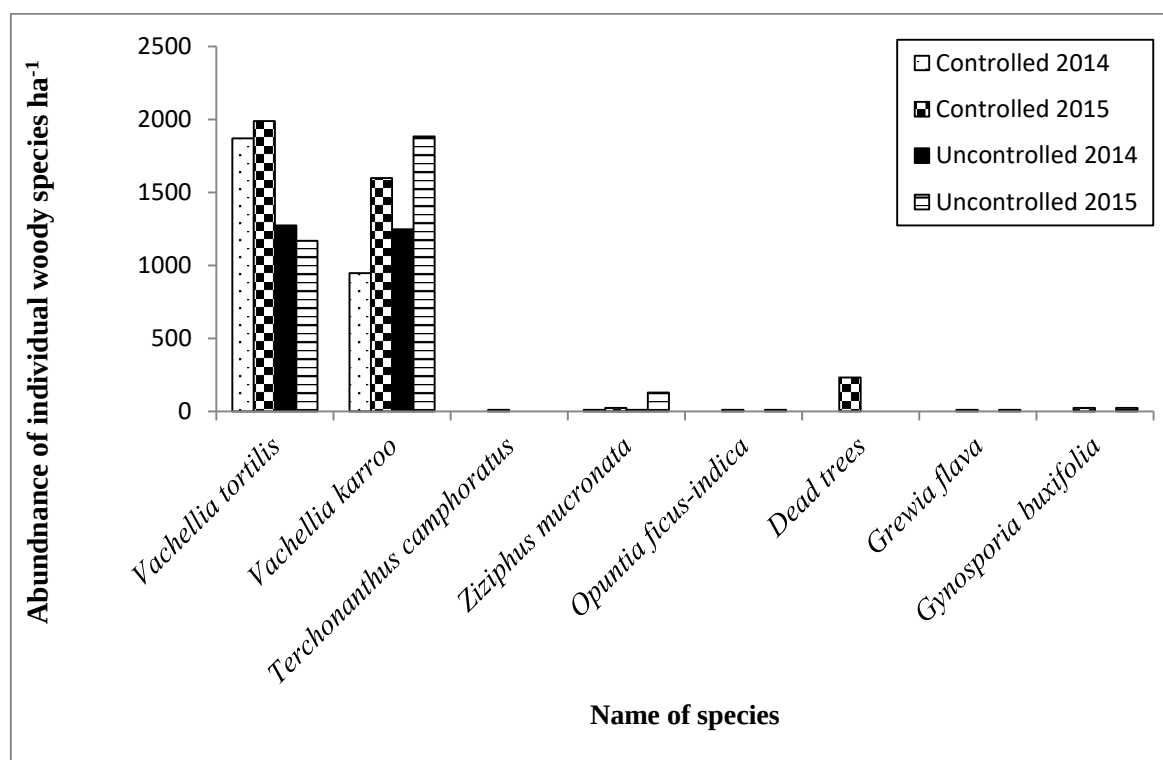


Figure 5.7: Abundance of individual woody species per hectare in the controlled and uncontrolled study site at Magogong for 2014 and 2015.

5.2.4.3 Coppicing of woody species after control

V. tortilis and *V. karroo* showed the highest coppicing (re-growth) at 35% and 11% respectively. The high coppicing of these species in the study site could be attributed to the high densities of these species in 2015 (Figure 5.8).

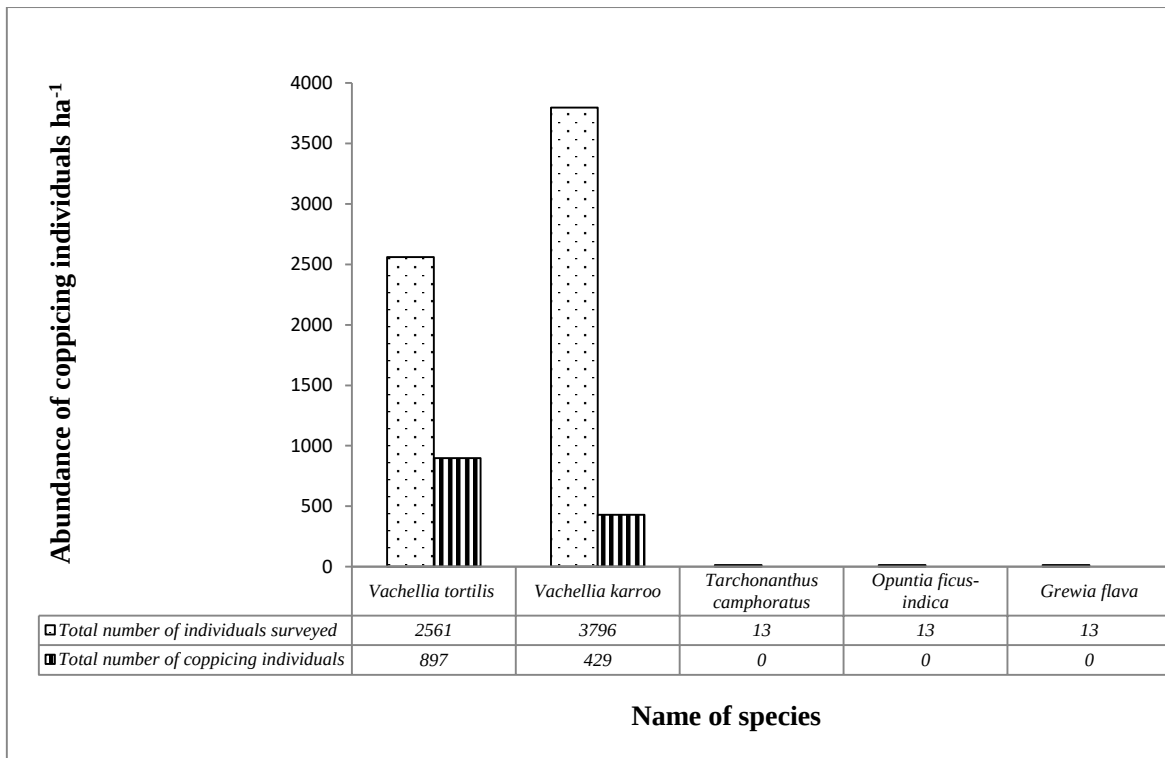


Figure 5.8: Abundance of coppicing individual per hectare compared to the total number of species surveyed in the Magogong study site that was controlled.

5.2.4.4 Success of woody plant control in Magogong

According to Table 5.1, the control in Magogong was successful. Evidence of this is the 245 dead trees found on this site in 2015. However the high re-growth of *V. tortilis* and *V. karroo* is indicative of the need of follow-up treatments by the WfW programme.

5.2.5 Manthe

5.2.5.1 Controlled sites

The results in Figure 5.9 indicate that the most prominent woody species in the Manthe controlled survey site were *V. tortilis*, *Diospyros lycioides* and *T. camphoratus*. The density of *V. tortilis* surveyed during 2014 were 1118 individuals/ha which increased slightly to 1131 individuals/ha in 2015 (Figure 5.9). *T. camphoratus* decreased from a density of 195 individuals/ha to only 13 individuals/ha during the same time interval. *D. lycioides* individuals were present at 390 individuals/ha in 2014 and decreased to a total absence in 2015 (Figure 5.9). Although not a woody species, the exotic cactus, *Opuntia ficus-indica* and *O. spinosior* were also present in the Manthe controlled site but absent in the

uncontrolled site (Figure 5.9). No dead trees were observed in 2014, however, a total of 169 individuals/ha were recorded in 2015 (Figure 5.9).

5.2.5.2 Uncontrolled sites

The density of *T. camphoratus* increased from 182 individuals/ha to 429 individuals/ha from 2014 to 2015, while the density of *V. tortilis* decreased from a density of 1963 individuals/ha surveyed in 2014 to 1573 individuals/ha in 2015 (Figure 5.9). The density of *S. mellifera* individuals was unchanged from 2014 to 2015 (Figure 5.9).

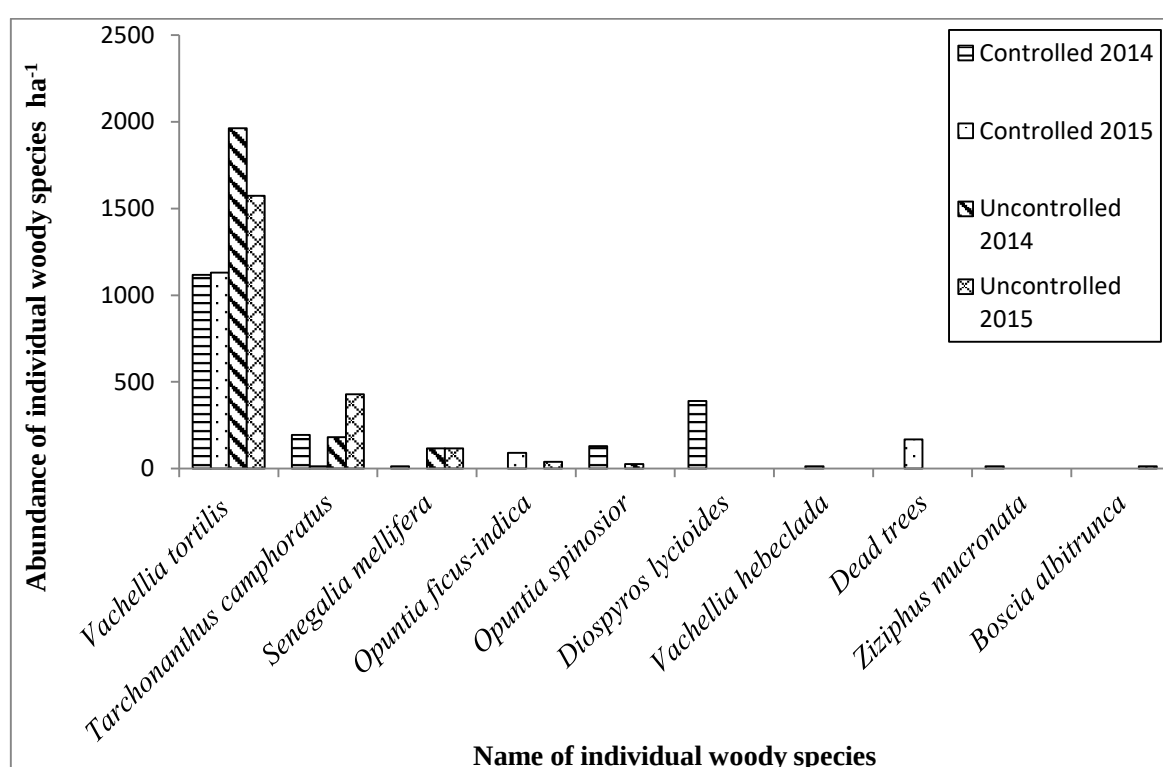


Figure 5.9: Abundance of individual woody species per hectare in the controlled and uncontrolled study site at Manthe from 2014 and 2015.

5.2.5.3 Coppicing of woody species after control

V. tortilis was the only coppicing species, representing a coppicing (re-growth) rate of 13% of the individuals after being treated both manually (cutting) and chemically (application of arboricides) in 2014 (Figure 5.10).

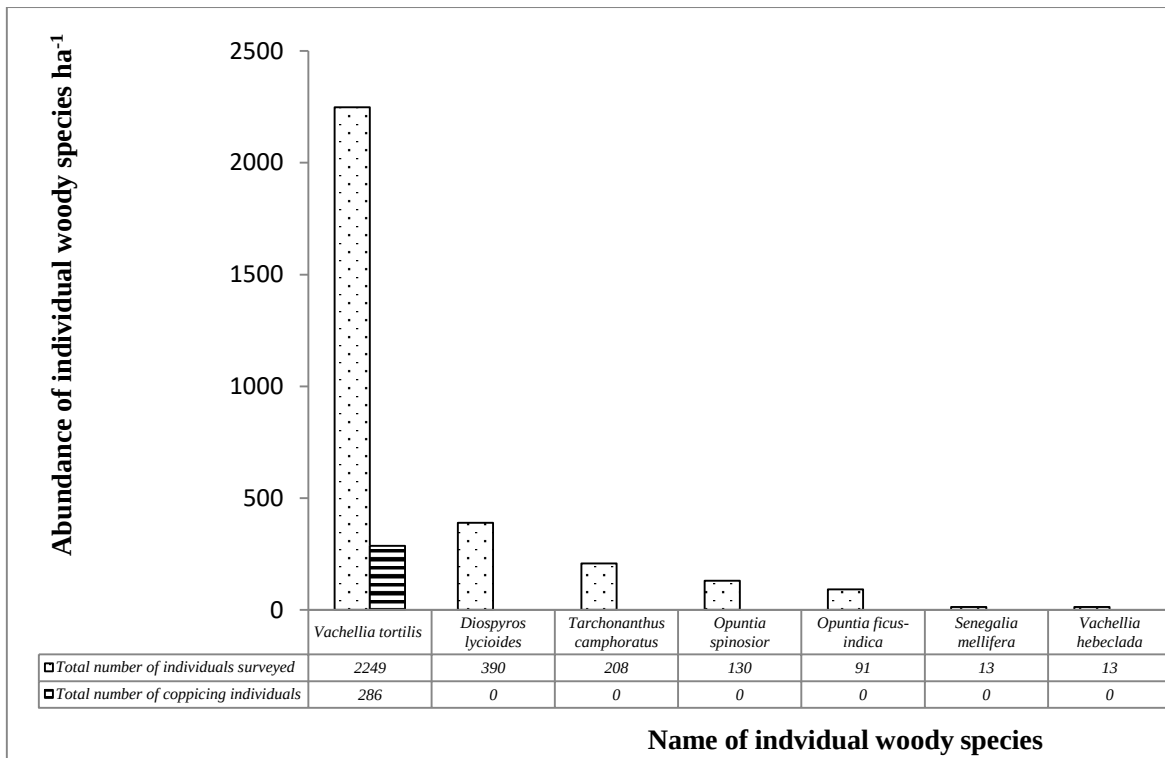


Figure 5.10: Abundance of coppicing individual per hectare compared to the total number of species surveyed in the Manthe study site that was controlled.

5.2.5.4 Success of woody plant control in Manthe

Results in Table 5.1 indicate that, the only increasing woody species in the controlled site were *V. tortilis* and *V. hebeclada*. The control in this site could, therefore, be considered successful due to the evidence of dead trees present in the site and also, the fact that *V. tortilis* was found to be the only woody encroacher species indicating signs of re-growth.

5.2.6 Taung Dam (102)

5.2.6.1 Controlled sites

The results in Figure 5.11 indicates, that in the Taung Dam (102) controlled site, the density of *S. mellifera* surveyed in 2014 was measured at a density of 1196 individuals/ha which decreased to 1170 individuals/ha in 2015. The density of *V. tortilis* decreased from 1352 individuals/ha to 923 individuals/ha in 2015 (Figure 5.11). *Z. mucronata* individuals, however, increased from an absence in 2014 to 39 individuals/ha in 2015 (Figure 5.11).

The non-encroacher, exotic cactus species, *O. ficus-indica* was absent in 2014 but, however, increased to 39 individuals/ha in 2015. Similarly, *O. spinosibacca* was recorded at 91 individuals/ha in 2014 but decreased to an absence in 2015 (Figure 5.11).

5.2.6.2 Uncontrolled sites

In the Taung Dam (102) benchmark (uncontrolled) site, *S. mellifera* trees were recorded at a density of 1612 individuals/ha in 2014 which decreased to 702 individuals/ha in 2015. Similar results were recorded for *V. tortilis* which decreased in density from 1352 individuals/ha surveyed in 2014 and to a density of 377 individuals/ha in 2015 and *T. camphoratus* decreased from 364 individuals/ha to 156 individuals/ha during the same time interval (Figure 5.11).

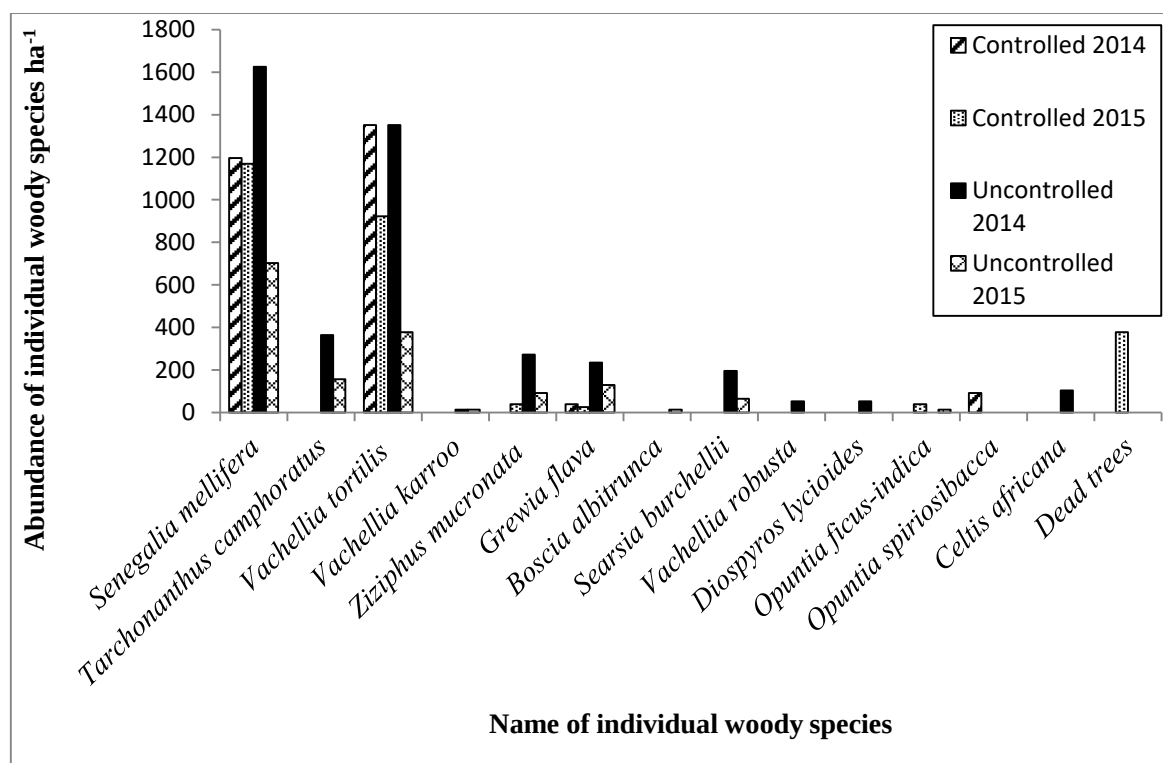


Figure 5.11: Abundance of individual woody species per hectare in the controlled and uncontrolled study site at Taung Dam (102) from 2014 and 2015.

5.2.6.3 Coppicing of woody species

Opuntia spinosibacca and *G. flava* showed no signs of coppicing (Figure 5.12), whereas *V. tortilis* had a high coppicing (re-growth) percentage of 51%, compared to *S. mellifera* which only had a coppicing percentage of 42% (Figure 5.12).

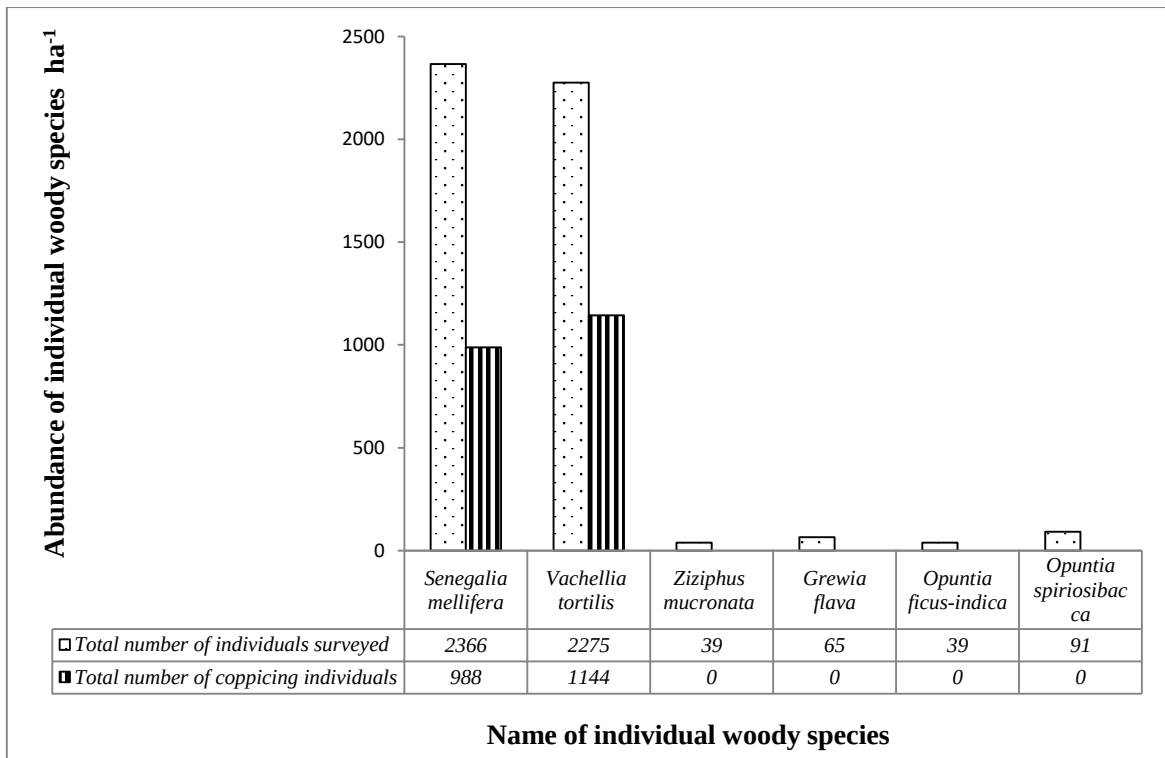


Figure 5.12: Abundance of coppicing individuals per hectare compared to the total number of species surveyed in the Taung Dam (102) study site that was controlled.

5.2.6.4 Success of woody plant control in Taung Dam (102)

Table 5.1 shows that the control in the Taung Dam (102) site was positive due to the decrease of *S. mellifera* and *V. tortilis*. The presence of dead trees was also evident and this contributed significantly to the decrease of woody densities in this site (Table 5.1). However, *S. mellifera* and *V. tortilis* showed signs of re-growth (Table 5.1), therefore, indicating that follow-up strategies are necessary in this site.

5.2.7 Taung Dam (98)

5.2.7.1 Controlled sites

The density of *S. mellifera* increased from 234 individuals/ha surveyed in 2014 to 702 individuals/ha in 2015 and *V. karroo* from an absence in 2014 to 312 individuals/ha surveyed in 2015. *V. tortilis* showed a decrease from 299 individuals/ha in 2014 to a density of 91 individuals/ha in 2015 (Figure 5.13). The results in Figure 5.13 also indicated that there was an increase in both *Z. mucronata* and *D. lycioides* woody species recorded from 2014 to 2015.

5.2.7.2 Uncontrolled sites

In 2014, the density of the woody encroacher species *S. mellifera* was reported at 65 individuals/ha which increased in 2015 to 1469 individuals/ha (Figure 5.13). Figure 5.13 also indicates that there were 962 individuals/ha for *V. tortilis* in 2014 which decreased to 117 individuals/ha surveyed in 2015. *T. camphoratus* was recorded at a density of 156 individuals/ha and increased to a density of 422 individuals/ha from 2014 to 2015 (refer to Figure 5.13) respectively. The density of *V. karroo* remained unchanged from 2014 to 2015, however, *D. lycioides* increased from an absence in 2014 to 65 individuals/ha in 2015 (Figure 5.13).

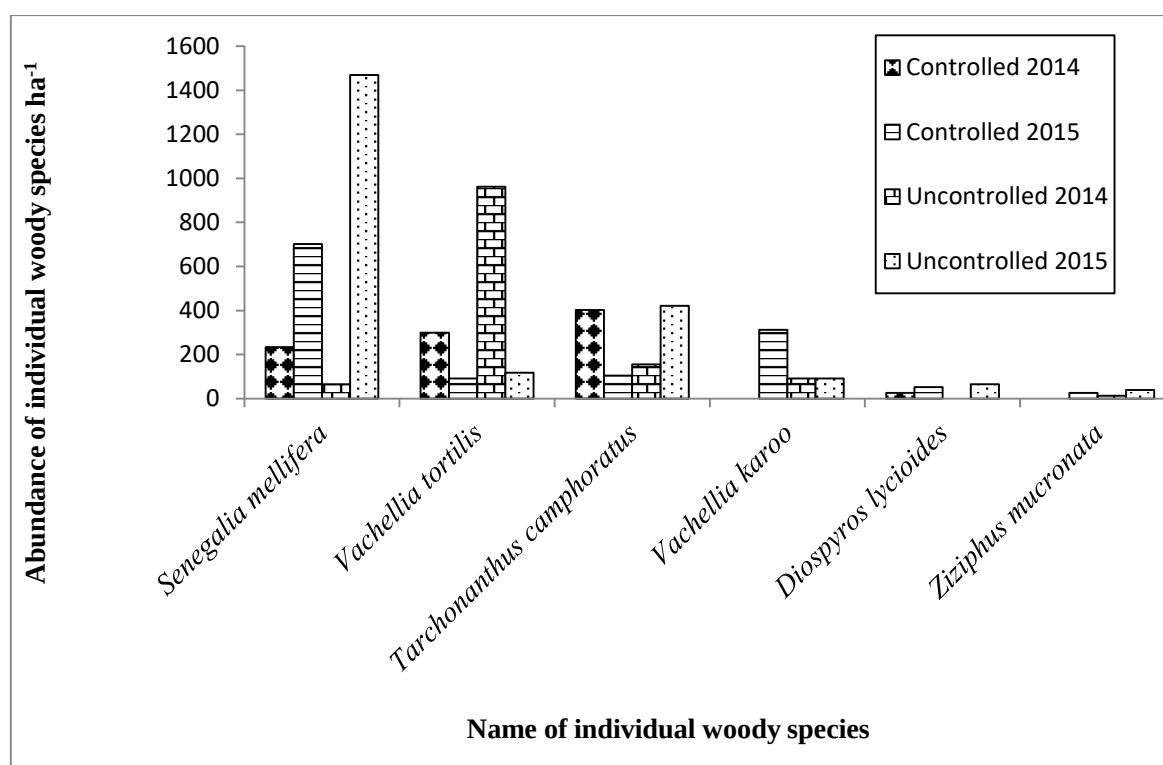


Figure 5.13: Abundance of individual woody species per hectare in the controlled and uncontrolled study site at Taung Dam (98) from 2014 and 2015.

5.2.7.3 Coppicing of woody species after control

In the Taung Dam (98) controlled site, the coppicing percentage of *V. karroo* was the highest at 58%, followed by *T. camphoratus* at 51%, *V. tortilis* at 20% and *S. mellifera* at 10% (Figure 5.14).

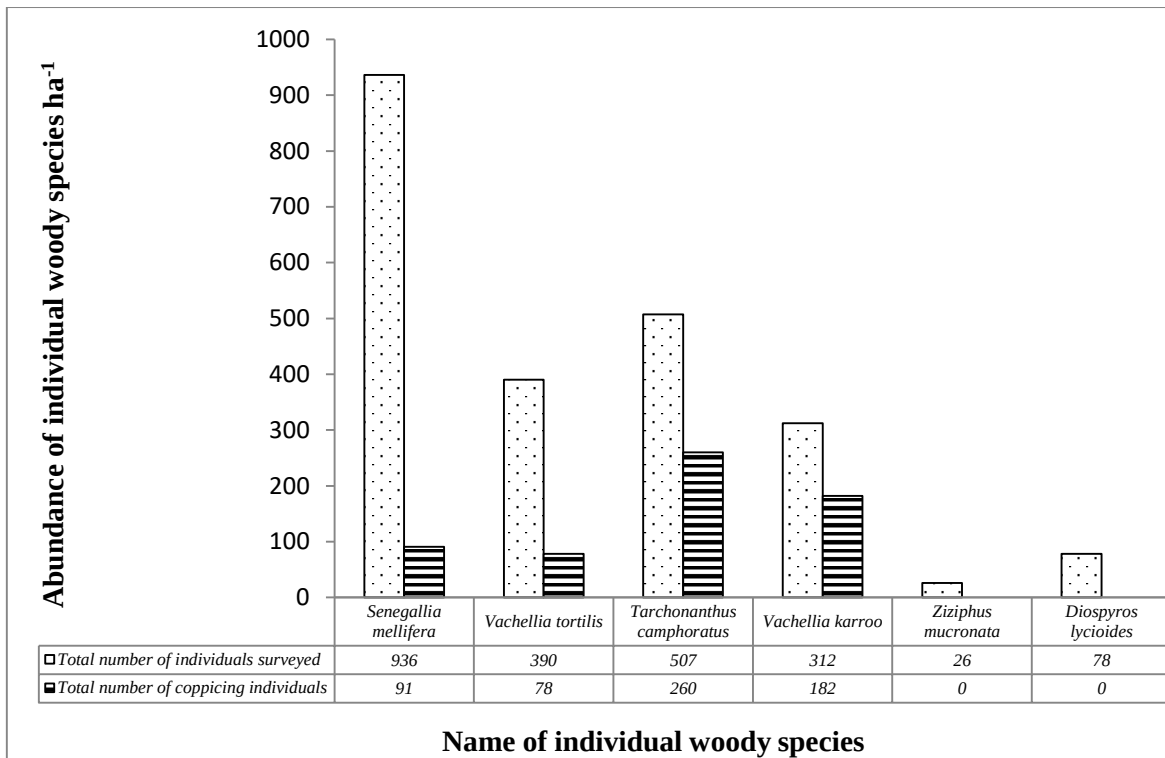


Figure 5.14: Abundance of coppicing individuals per hectare compared to the total number of species surveyed in the Taung Dam (98) study site that was controlled.

5.2.7.4 Success of woody plant control in Taung Dam (98)

According to Table 5.1 there was an increase in woody densities of *S. mellifera*, *V. karroo*, *Z. mucronata* and *D. lycioides*, thus indicating that the control in this site was unsuccessful. There was also evidence of re-growth with regard to *V. karroo*, *S. mellifera*, *V. tortilis* and *T. camphoratus* (Table 5.1). This is, therefore, a clear indication that follow-up strategies need to be implemented in this area.

5.2.8 Taung Dam (100)

5.2.8.1 Controlled sites

From 2014 to 2015, the density of *S. mellifera* increased from 182 individuals/ha to 351 individuals/ha. The densities of *T. camphoratus* and *V. karroo* also increased during this time from 65 individuals/ha to 377 individuals/ha and 13 individuals/ha to 117 individuals/ha respectively. The density of *V. tortilis* individuals, however, decreased from 442 to 13 individuals/ha.

5.2.8.2 Uncontrolled sites

S. mellifera had a density of 169 individuals/ha in 2014 and in 2015 this species increased to 2431 individuals/ha (Figure 5.15). *T. camphoratus* individuals increased from 13 individuals/ha to 377 individuals/ha and *V. karroo* from a total absence to a density of 845 individuals/ha (Figure 5.15). A decrease in density occurred for *V. tortilis* from 221 individuals/ha in 2014 to 39 individuals/ha in 2015.

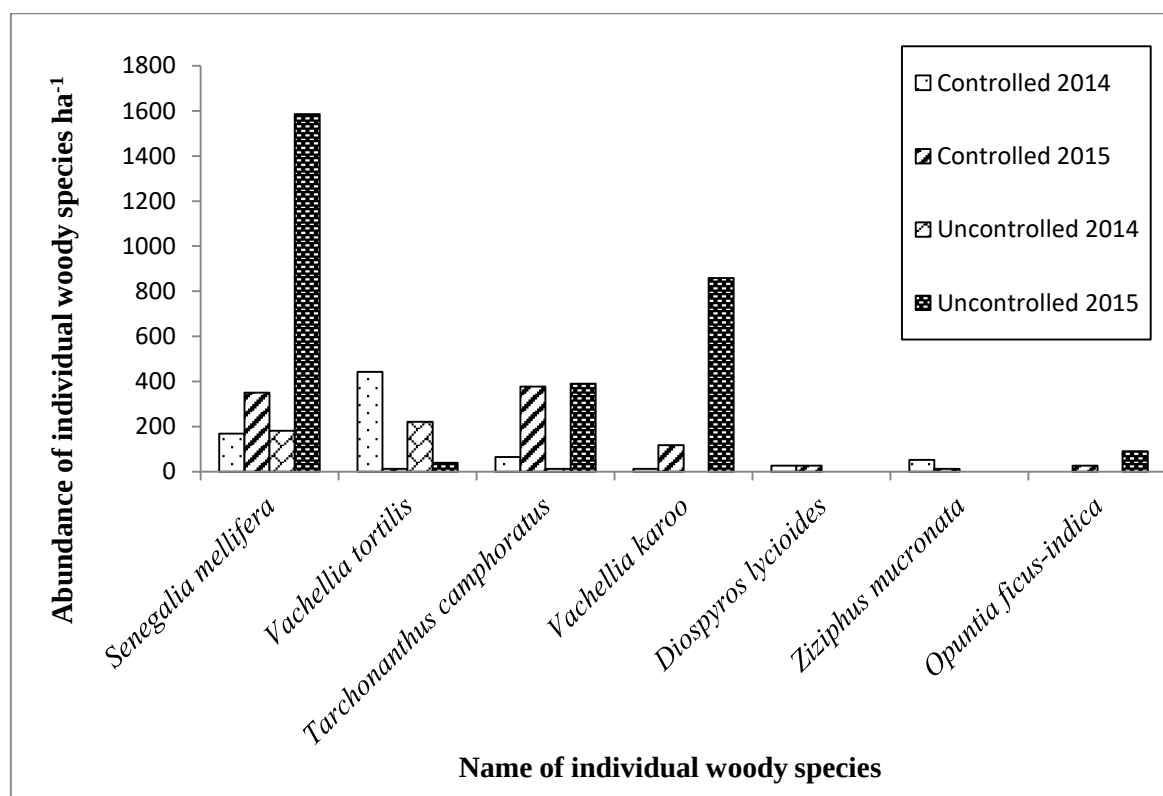


Figure 5.15: Abundance of individual woody species per hectare in the controlled and uncontrolled study site at Taung Dam (100) from 2014 and 2015.

5.2.8.3 Coppicing of woody species after control

The results indicate that *T. camphoratus* (24%), *V. tortilis* (23%) and *Z. mucronata* (20%) have all three coppiced (Figure 5.16). Woody species such as *S. mellifera* did not show any indication of coppicing (Figure 5.16).

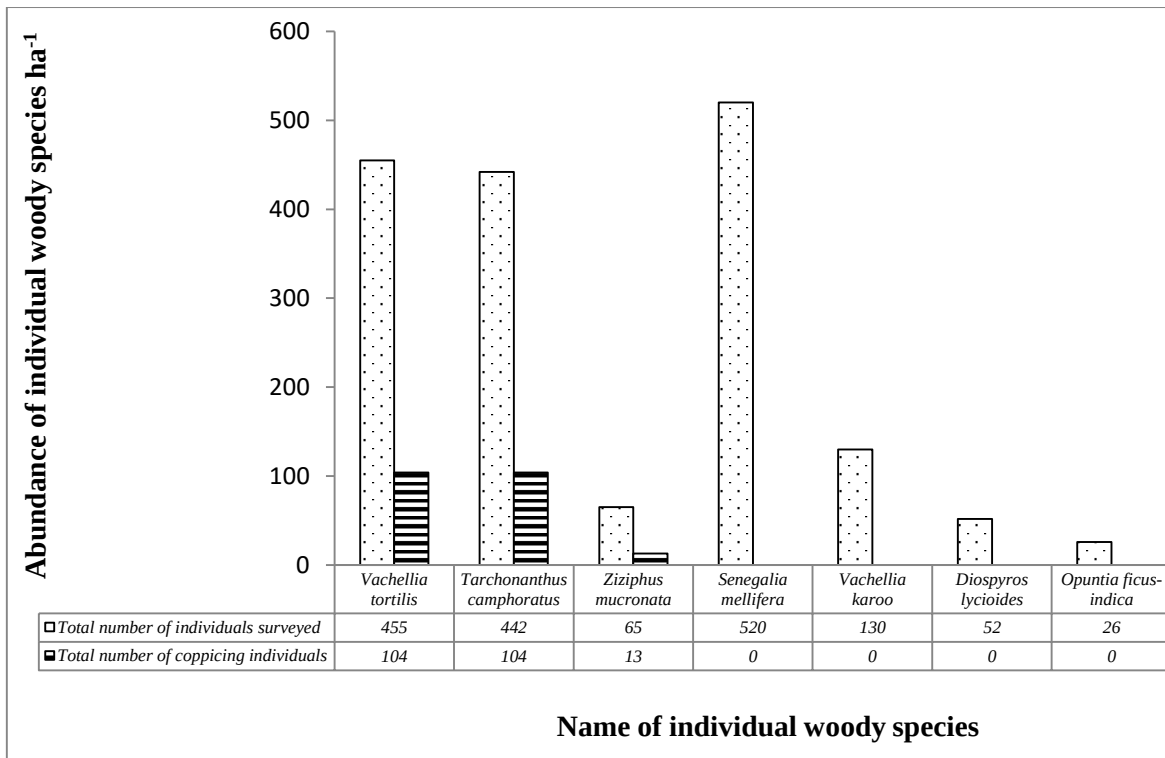


Figure 5.16: Abundance of coppicing individual per hectare compared to the total number of species surveyed in the Taung Dam (100) study site that was controlled.

5.2.8.4 Success of woody plant control in Taung Dam (100)

The control success in the Taung Dam (100) could be considered negative due to the increase in woody density of *V. karroo*, *T. camphoratus* and *S. mellifera* present in the area (Table 5.1). However, according to Figure 5.16 and Table 5.1 there was no evidence of re-growth (coppicing) of *S. mellifera* in 2014 or 2015.

5.2.9 General discussion of the results obtained for the woody component in the study sites

The WfW programme was not successful in controlling woody species, neither through mechanical nor chemical or a combination of the two methods. This is indicated by the increase in densities of *V. karroo*, *T. camphoratus* and especially *S. mellifera* in the controlled sites and coppicing of *V. tortilis* and *S. mellifera*. *S. mellifera* increased in density by 63% over the two survey years, even after the control programmes were implemented. Re-growth occurred in all the sites, especially for *V. karroo*, *V. tortilis* and *S. mellifera*. Control in site 4 (Magogong), site 5 (Manthe) and site 6 (Taung Dam (102)) were the only examples where the control project of WfW programme can be considered successful. Dead

trees in these sites reveal that the control was successful, however, the high coppicing of *V. tortilis* and *S. mellifera* is an indication that a follow-up control programme should be implemented. The establishment of new seedlings, coppicing and re-thickening of *S. mellifera* was also noticed by Smit (2014) in the savanna biome. Smit (2014) indicated that the establishment of new seedlings is due to soil disturbance and that re-growth of controlled *S. mellifera* trees often lead to a worse encroaching situation than before the control.

Schleicher *et al.* (2011) examined how woody plant interactions and soil type affect growth and spatial distribution of *T. camphoratus* and *S. mellifera* in Kimberley thorn bushveld, South Africa. Schleicher *et al.* (2011) reported that, vast numbers of *S. mellifera* woody plants was recorded in the rocky plot (total number of woody plants: rocky plot experiment 269 individuals, sandy plot experiment 141 individuals, rocky and sandy plot experiment 113 individuals). This was possibly as a result of improved soil conditions caused by high soil rock particles in this area (Britz & Ward, 2007b). According to Schleicher *et al.* (2011) the number of plants per species differed significantly within various plots, with *T. camphoratus* being most abundant in the sandy area, whereas *S. mellifera* was more abundant in the rocky area. The differences in shrub densities can be ascribed to (1) niche separation, with *T. camphoratus* preferring sandy soil and *S. mellifera* preferring soils containing adequate rock particles and/or (2) a competitive impact, in which *S. mellifera* may be a vigorous competitor on rocky soils compared to *T. Camphorates* (Schleicher *et al.*, 2011).

Ward and Esler (2011) embarked on a study on Pniel Estates near Barkly West, Northern Cape Province of South Africa and demonstrated that, reduced grass densities in sandy soil caused by overgrazing had a large positive influence of sapling recruitment of *S. mellifera* compared a rocky substrate. *S. mellifera* is one of the exceptional influential woody encroacher plants in South African savannas. Ward and Esler (2011) reported that, *S. mellifera* encroached effortlessly on rocky areas compared to sandy environments, although, it exhibits a larger growth form in sandy environments. According to Ward and Esler (2011) rocky substrates absorb more water than sandy substrates, which is crucial during plant recruitment in arid and semi-arid environments. Kraaij and Ward (2006), Meyer *et al.* (2007a) and Joubert *et al.* (2008) have demonstrated that, rainfall availability is a major role in the colonization of *S. mellifera* individuals. Ward and Esler (2011) observed

additional recruitment of saplings on rocky environments as a result of sufficient water availability. Mbatha and Ward (2010) found that there was about 65-70% lower biomass production of grasses in rocky environments at the Pniel Estates than that of the sandy soils in open savannas. Ward and Esler (2011) suggested that increased colonies of *S. mellifera* individuals may be ascribed to low densities of grasses competing against *S. mellifera* saplings in the vicinity of a rocky terrain.

The results reported by Schleicher *et al.* (2011), Ward and Esler (2011) and Smit (2014) are in agreement with the results in Figure 5.11, Figure 5.13 and Figure 5.15 which indicated high densities of *S. mellifera*. The soil surface terrain in the Taung Dam area is very rocky and thus made it difficult for the WfW programme working teams to control *S. mellifera* in the Taung Dam survey sites. Ward and Esler (2011) indicated that, the increase in *S. mellifera* densities within a rocky terrain may be attributed to the fact that there is sufficient water under the rocks which could easily be accessed by *S. mellifera* individuals. The high coppicing percentages of *S. mellifera* also attributed to the high densities in these sites. Another factor that could have attributed to the increase of virtually all the woody species in the Taung Dam (100) controlled site could be the fact that no follow-up practices were implemented by the WfW programme working teams on this site.

Grazing pressure is a key determinant of bush encroachment because it creates opportunities for colonization, thus, permitting overall woody recruitment in grass dominated areas. Plant herbivory promotes the spread of unpalatable species and reduces inter-plant competition (Coetzee *et al.*, 2008). The successful plant survival of unpalatable woody species such as *Vachellia* species, could lead to further encroachment. Herbivory or mechanical treatment (manual cutting) stimulates shoot production of mature *Vachellia* trees (Dangerfield & Modukanele, 1996). *D. lycioides* is not considered as an invasive species and contributes to veld stability through soil protection from harsh environmental factors. It increases water infiltration and reduces high temperatures affecting bare grounds. The Taung residents also depend on *D. lycioides* and other species, such as *V. tortilis*, for fuelwood purposes. The *V. tortilis* pods are browsed by domestic cattle, thus making it a good substitute for forage during the dry seasons. Mechanical removal of shrubs without proper follow-up remedies usually results in temporary changes in plant community structure (Robson, 1995; Allegretti *et al.*, 1997), and stimulates plant re-growth, leading to subsequent increases of woody

plants (Ruthven III *et al.*, 1993). In the majority of cases, shrubs persist to dominate even after complete grazing removal (Guo, 2004; Mengitsu *et al.*, 2005; Angassa & Oba, 2007). In the Taung Dam (98) controlled site, both manual cutting and chemical control practices were implemented by the WfW programme to control the encroaching woody species *S. mellifera*, *T. camphoratus* and *V. tortilis*. While *V. tortilis* could have been used for fuel wood, it is extremely unlikely that *S. mellifera* and *T. camphoratus* would have been used by the local people because the species were much smaller and also burn rapidly (Ward *et al.*, 2014). Because Taung is a communally managed area, the inhabitants make use of *V. tortilis* for fuel purposes and *T. camphoratus* for traditional purposes such as medicinal use. This could have contributed to the removal of these two species.

Luoga *et al.* (2004) observed that 83% of the 30 harvested (manual cutting) woody species in the forest reserve in the forest savanna woodland of Tanzania had re-sprouted after harvesting, compared to 90% of the 39 species in the communal lands. Trees of unique Miombo woodlands of eastern and south-central Africa coppice from roots and stumps once above-ground parts have been damaged, eradicated, harvested or killed by fire (Frost *et al.*, 1986; Grundy, 1995). The coppicing shoots grow more rapidly compared to newly established saplings, because they already have a well-developed root system which stores reserves (Chidumayo, 1993; Grundy, 1995). Plant re-growth and development can best be improved by implementing follow-up strategies and therefore managing bush encroachment (Grundy, 1995; Luoga *et al.*, 2004). Luoga *et al.* (2004) indicated that stump height had limited influence on the resulting coppicing shoots than stump size, although stump heights had an effect on the height of re-growing individual stumps. Brudvig and Asbjornsen (2007) added that, for restoring ecosystems degraded by bush encroachment, active eradication of the problematic woody invader species is necessary. This technique may be profitable in situations where invasion has resulted in severe structural modifications, such as canopy closure or changes in vegetation structure and development, which might inhibit the re-establishment of historic fire regimes.

Changing land patterns in the Taung region can be viewed as another factor influencing woody plant establishment. The mechanical (tree cutting) and chemical (herbicide application) control methods implemented by the WfW programme in Taung region could have also played a role in the recruitment of new woody individuals, thus resulting in high densities of *S. mellifera* and *V. tortilis*. These species generally have extensive root systems,

both vertical and horizontal, which promotes accelerated recovery after cutting (Malimbwi *et al.*, 1994; Mistry, 2000) and most dominating woody species found in the Miombo woodland have tap roots which extend to a depth of about 5 metres (Mistry, 2000). *V. tortilis* and *T. camphoratus*, however, have deep taproots and can extract moisture from the deep soil layer.

The high coppicing percentages of these species are as a result of no follow-up strategies being implemented by the WfW programme in the study areas. Pienaar (2006) conducted a study in the Marekele National Park in the Limpopo Province and reported that, the coppiced plants had increased number of stems compared to normal uncut plants, which suggested that tree thinning treatments produces structural changes of woody plants. Barac (2003) reported that, single-stemmed trees, which were previously controlled through the aid of mechanical control measures, tend, to advance into multi-stemmed plants when coppicing occurs. Pienaar (2006) stated that, if the coppicing of the encroacher species is not restricted and controlled in time, the re-encroachment that normally occurs is usually worse than the previous woody encroached state. This, therefore, implies that follow-up treatments are an integral component of all bush control management programmes (Ritchter, 1991; Smit *et al.*, 1999; Barac, 2003). According to Pienaar (2006), the continual damage to the coppice shoots of most *Vachellia* species contribute to the reduced re-growth of plants during the following season. Pienaar (2006) conducted a study in the Marakele National Park and reported that, the lack of follow-up treatments following initial control is the key factor supporting the swift re-growth of woody plants in all controlled (treated) plots. Lack of follow-up measures also led to coppice shoots in all controlled survey sites. Pienaar (2006) recorded that, the coppice of basically all woody plants that were mechanically treated highlights the importance of follow-up treatments, either chemically, mechanically or a combination of both.

According to Smit (2014), the purpose of bush control must be cleared out before a control programme is initiated. Smit (2014) stated that, bush control often leads to situations that are worse than before the control of woody plants, due to re-thickening and coppicing of encroaching species. Smit (2014) suggested that, bush management must be the main focus, rather than total control and that the financial benefits of the controlled woody plants (charcoal etc.) must be considered before such a control programme is initiated.

V. tortilis is a keystone species, growing across arid ecosystems in Africa and the Middle East and has the ability to coppice vigorously, following defoliation (Teague & Walker, 1988). In a study conducted by Luoga *et al.* (2004) the increased numbers of coppicing individuals on communal lands compared to private lands, is an outcome of relief from self-thinning mechanisms, brought about by reduced levels of competition between woody plants due to reduced canopy cover. Coppicing ability of plants varies in regard to phenology and morphology of the species being controlled, the size/age of plant, stump height and the percentage of stand being removed (Shackleton, 2000; Luoga *et al.*, 2004). Other factors shown to affect woody coppicing effectiveness are site characteristics (soil structure and composition), angle of cutting and the effectiveness of the cutting tools (sharp tools stimulates intensive re-growth from the stumps) (Grundy, 1995; Luoga *et al.*, 2004). Tall stumps usually produce a larger number of coppice shoots compared to shorter stumps (Mushove & Makoni, 1993). This tendency, is also reported South African savanna trees and is associated with an increase in land surface area, resulting in the growth of extra buds on the cut stump (Luoga *et al.*, 2004). The seasonal timing during which mechanical control measures are implemented also contributes to the number of coppice shoots produced.

The best time for plants re-growth is just before the onset of the rains when abundant moisture facilitates recovery from harvesting (Pawlick, 1989; Luoga *et al.*, 2004). Cutting plant stem too close to the ground; may result in fungal infection as a result of available soil moisture or stump decay, while cutting too high may cause compromised plant growth ability and poor shoot development. Therefore, stump height has a definite positive influence on sprouting height, but varying effects on the total number of coppice shoots (Luoga *et al.*, 2004).

Water availability is the main determinant of woody plant densities in semi-arid and arid environment (650mm mean annual precipitation [MAP]) while disturbance-based factors contribute significantly to tree cover dynamics in areas receiving 650mm MAP (Sankaran *et al.*, 2005). Kraaij and Ward (2006) reported that rainfall availability is the basic factor determining the establishment of *S. mellifera* colonies. In savanna rangelands, sapling establishment is favoured by the initiation of rainy seasons after being stimulated dry seasonal fires (Gashaw & Michelsen, 2002; Ward *et al.*, 2014). However, a rainy season drought following germination may cause considerable mortality of tree seedlings

(Hoffmann & Franco, 2003; Gignoux *et al.*, 2009; Vadigi & Ward, 2013). Therefore, recurrent rainy seasons are the key determinant of tree seedling establishment compared to the overall mean annual precipitation (Kraaij & Ward 2006; Meyer *et al.*, 2007a; Angassa & Oba, 2008; Kahumba, 2010; Vadigi & Ward, 2013).

The decrease in woody densities in the uncontrolled sites (See Chapter 5, Section 5.2) can be attributed to the fact that during winter season the Taung local residents make use of fire wood as a source of energy (Figure 7.5b & Figure 7.5c). This is especially evident in the Moretele, Myra (87), Myra (76), Manthe and Taung Dam (102) uncontrolled sites. The Moretele and Manthe sites are located near villages. Myra (87) and Myra (76) are located near Pudumoe village. Taung Dam is located close to Cokonyane village (See Chapter 3, Figure 3.4). This, therefore, means that residents do not have to travel long distances for them to collect fire wood.

5.3 Changes in grass species frequency in each study site

The changes of grass species abundances will be simultaneously discussed for the uncontrolled and controlled sites.

5.3.1 Moretele

The most dominant grass species in the Moretele controlled site was *Eragrostis lehmanniana* (45%), a perennial grass characterising moderate overgrazing and disturbance in the Taung area (Van Oudsthoorn, 1992) (Figure 5.17). The frequency of *Digitaria eriantha* (14%), a perennial, palatable large tufted grass in the controlled site is much lower, followed by annual grasses depicting overgrazing and disturbance, such as *Tragus racemosus* and *T. berteronianus* (Van Oudsthoorn, 1992) (Figure 5.17). The type of grazing regime in this area was observed to be continuous and selective grazing of palatable, climax species was observed. Although the dominant grass species in the uncontrolled site were *Stipagrostis uniplumis* (21%) and *E. lehmanniana* (15%), their frequencies were very low with little contribution to the phytomass production and grazing potential (Figure 5.17). Other grasses that occurred in the uncontrolled site included *Aristida stipitata* and *A. adscensionis*, indicating disturbance and poor veld condition (Van Oudsthoorn, 1992). The results from the Moretele study site indicate high bush densities in the uncontrolled (benchmark) site as compared to the controlled site (Figure 5.1) and high grass frequencies

(%) in the uncontrolled site as compared to the controlled site. The frequency of grass species in the Moretele uncontrolled site was higher when compared to the controlled site (Figure 5.17).

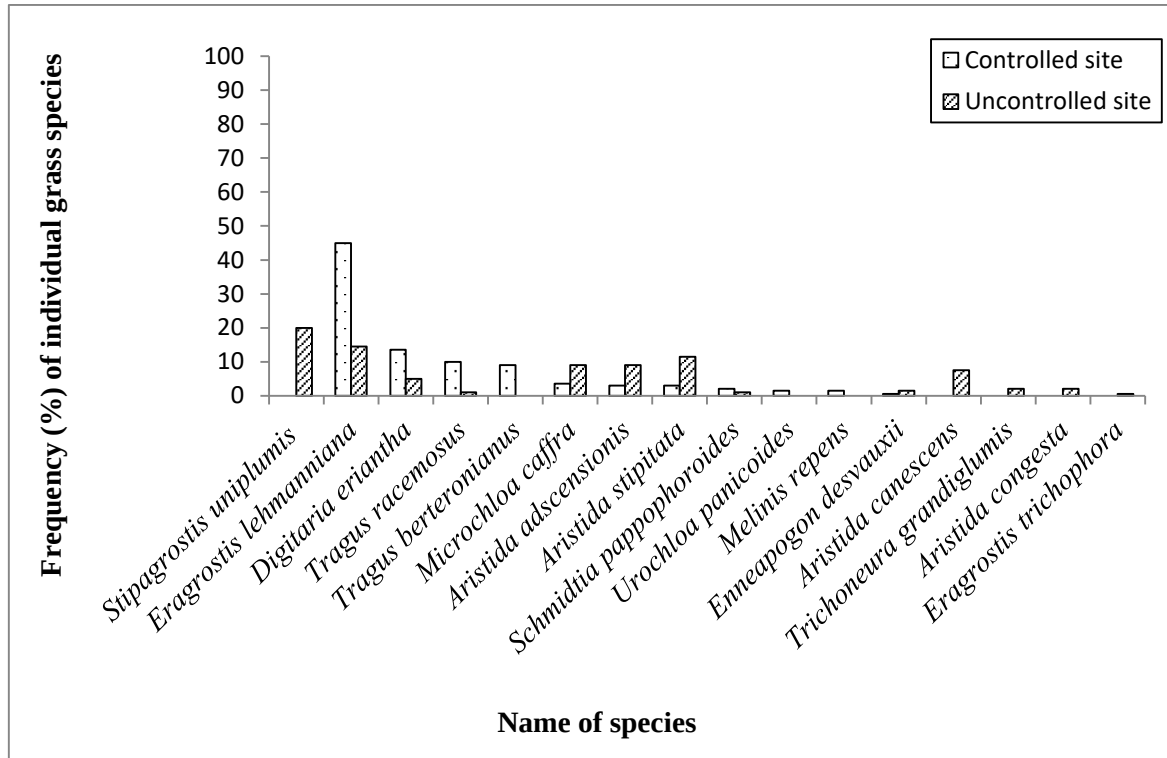


Figure 5.17: Frequency (%) of individual grass species in the woody controlled and uncontrolled sites at Moretele.

5.3.2 Myra (87)

The most prominent grass species surveyed in the Myra (87) controlled site were the tufted perennial grasses *D. eriantha* (16%) and the short-lived perennial (sometimes annual) grass *A. congesta* (12%) (Figure 5.18). *D. eriantha* (15%) was noted in the uncontrolled site and is a palatable grass that is regarded as one of the best natural and cultivated pasture species in southern Africa. It is normally an indicator of good veld condition, whereas *A. congesta* is an indicator of veld disturbance or degraded land (Van Oudsthoorn, 1992). Both grass species provide stability to disturbed soils and bare patches under degradation conditions (Van Oudsthoorn, 1992). In the Myra (87) uncontrolled site, the most prominent grass species were the tufted perennial grasses, *Aristida canescens* (18%), followed by *D. eriantha* (15%) (Figure 5.18).

According to Smit and Rethman (1999), colonisation of bare ground by grasses increased with increased levels of tree thinning. Annual grasses were the main colonisers of bare ground, while perennial grasses only constituted a small proportion of the grass species composition. This corresponds with the results obtained for *Aristida congesta*, as this species is well established in this area where the control of woodies was only recently applied (Figure 5.18). The establishment of perennial grasses was unsuccessful and due to the semi-arid nature of the area, it is anticipated that successful succession will fail progress to a point where perennial grasses will once more prevail, especially if these species are not protected from grazing (Smit & Rethman, 1999).

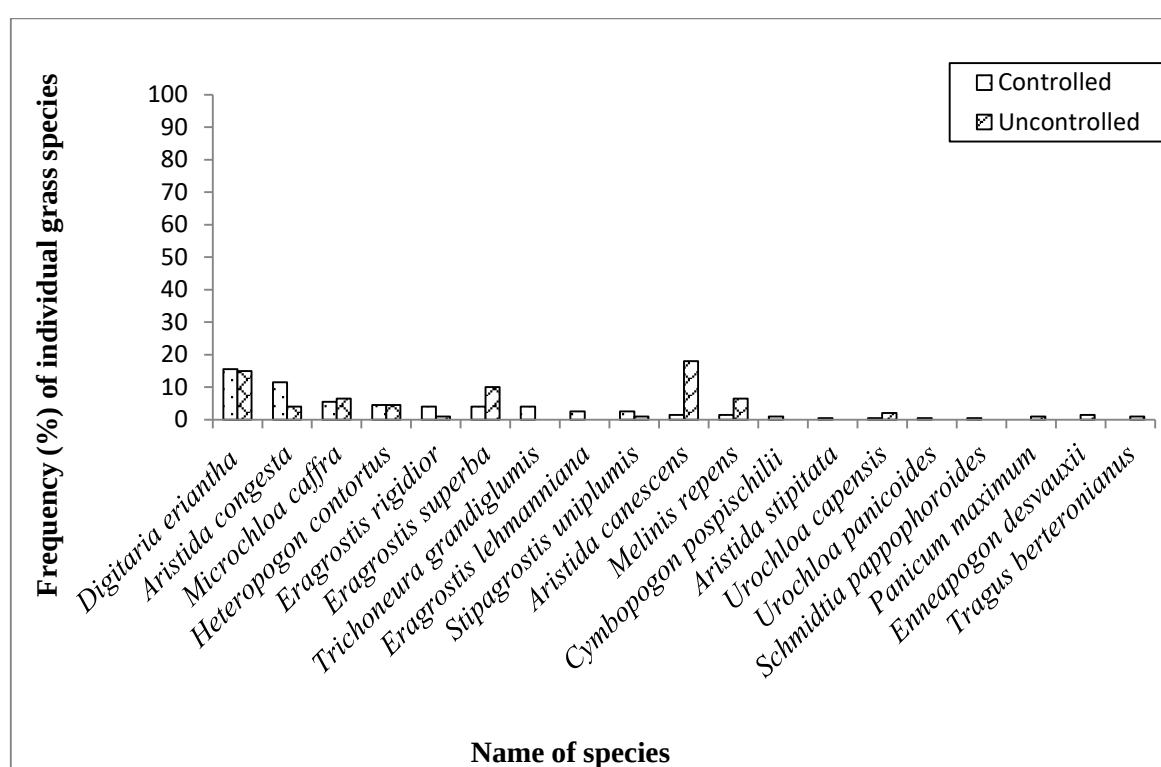


Figure 5.18: Frequency (%) of individual grass species in the woody controlled and uncontrolled sites at Myra (87).

5.3.3 Myra (76)

The prominent grass species in the Myra (76) controlled site were the short-lived perennial (sometimes annual) and non-palatable grass *A. congesta* (38%), followed by the sparse tufted annual grass, *T. berteronianus* (9%), *A. congesta* and *T. berteronianus* are indicators of severe veld disturbance and degradation (Figure 5.19) (Van Oudsthoorn, 1992). *A. congesta* was also present in the uncontrolled site, but occurred at lower frequencies (Figure 5.19). Both *A. congesta* and *T. berteronianus* are generally unpalatable (Van

Oudsthoorn, 1992) and have the ability to protect the soil and provide improved “conditions” for growth by other tufted and perennial grasses. In the uncontrolled site (Figure 5.19), the most prominent grass species were *A. congesta* (22%) and a hard (non-palatable) perennial tufted grass, *Eragrostis rigidior* (13%). Both these grass species are considered to be good indicators of veld disturbance and soil erosion. Under natural conditions, both *Cynodon dactylon* and *E. rigidior* are average to good pasture grasses and excellent soil stabilisers (Van Oudsthoorn, 1992). According to Smit and Rethman (1999), annual grasses are the dominant pioneers of bare ground observed after tree thinning, while perennial grasses only contribute a small proportion of the grass species composition. Colonisation of the herbaceous layer increased with an increase in tree thinning. Annual grasses are observed to be most effective colonisers and establish easier than pioneer grasses in bare ground, post thinning (Smit & Rethman, 1999). It was also observed that perennial grasses showed a general lack of successional trends. These findings support the results obtained in the Myra (76) controlled site which indicate that *A. congesta* and *T. berteronianus* were the most prominent grass species (Figure 5.19). The other supporting factor of this finding is that the increased production of herbaceous vegetation following bush clearing many result in the replacement of desirable, low-fibre grass species by more undesirable, high-fibre, sour grass species (Smit & Rethman, 1999). The results in Figure 5.19 also indicated that, in the controlled site, the frequency of the perennial tufted grass, *E. rigidior*, were lower, compared to the uncontrolled site.

Although there was a high variety of grass species in the uncontrolled site, the highest grass frequencies were observed in the controlled site as compared to its respective uncontrolled site. This includes the short-lived perennial (sometimes annual) grass, *A. congesta*, followed by the sparse tufted annual grass, *T. berteronianus* (Figure 5.19).

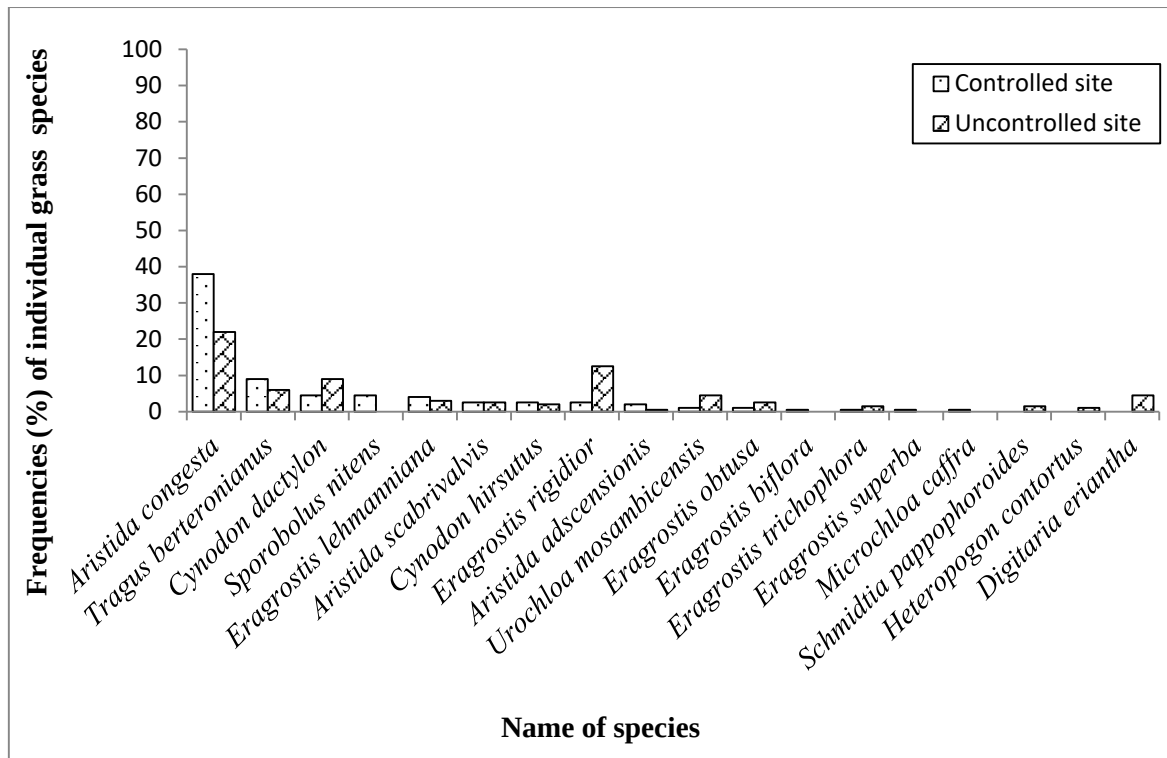


Figure 5.19: Frequency (%) of individual grass species in the woody controlled and uncontrolled sites at Myra (76).

5.3.4 Magogong

The dominant grass species in Magogong controlled site was *Urochloa panicoides* (59%), a tufted annual palatable grass, depicting veld disturbances and overgrazing (Daemane., 2012). This was followed by perennial palatable grasses such as *C. dactylon* (36%), *Panicum coloratum* (2%) and *T. racemosus* (2%) respectively (Figure 5.20). *C. dactylon* is an average palatable grass, depicting veld recovery after disturbances and is able to withstand overgrazing (Augustine & McNaughton, 2004). Noticeable is the higher frequency of *C. dactylon* in the controlled site as compared to the uncontrolled site (Figure 5.20). The most dominant species in the Magogong uncontrolled site was the annual tufted and palatable grass, *U. panicoides* (65%). This was followed by perennial grasses such as *C. dactylon* (21%) and annual grass such as *T. racemosus* (12%) (Figure 5.20). However, the frequency of the palatable perennial grass species, *P. coloratum*, had decreased in the Magogong uncontrolled site as compared to Magogong controlled site (Figure 5.20).

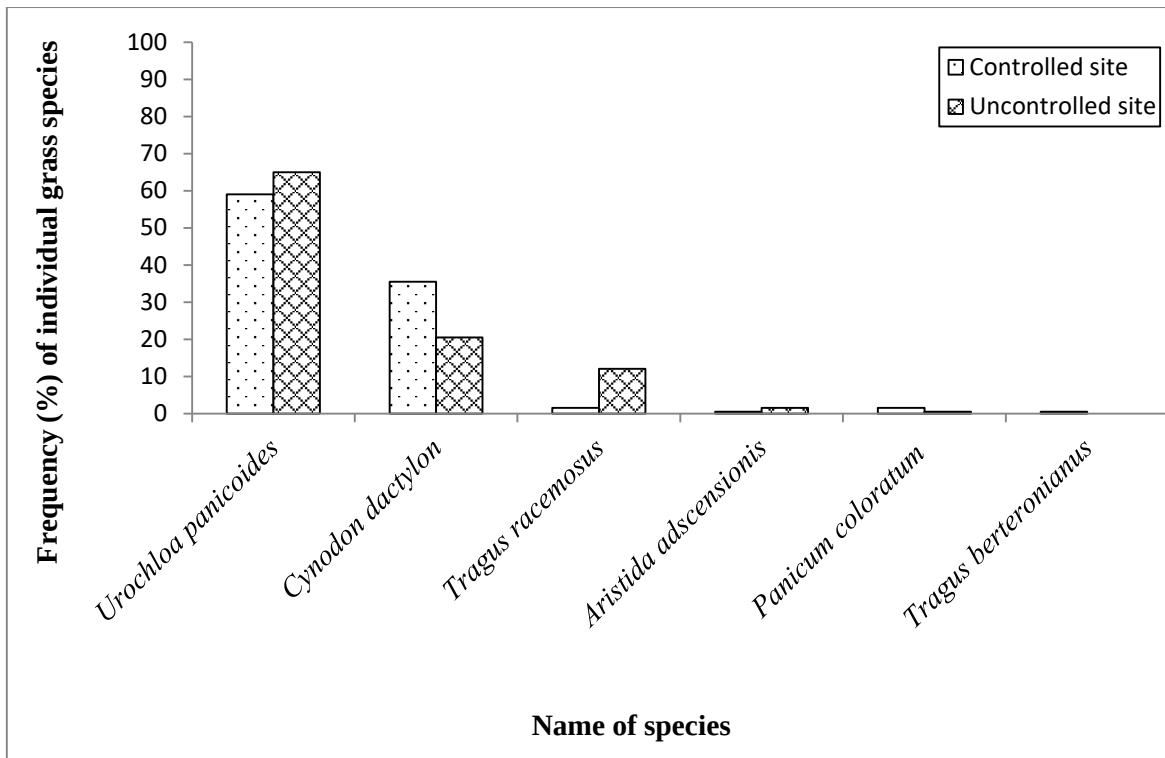


Figure 5.20: Frequency (%) of individual grass species in the woody controlled and uncontrolled sites at Magogong.

5.3.5 Manthe

The most prominent grass species in the controlled site were the sparse tufted annual grass, *T. berteronianus* (18%), followed by *C. hirsutus* (10%) (Figure 5.21). *Tragus berteronianus* is a good indicator of veld disturbance and is useful in stabilising soil conditions, thus creating opportunity for other grass species to establish (Van Oudsthoorn, 1992). In the uncontrolled site, the most prominent grass species were the perennial (sometimes annual) *T. berteronianus* (17%) and *A. congesta* (16%) (Figure 5.21). Both are generally unpalatable (Van Oudsthoorn, 1992) and have the ability to protect the soil and provide improved conditions for growth by other perennial grasses.

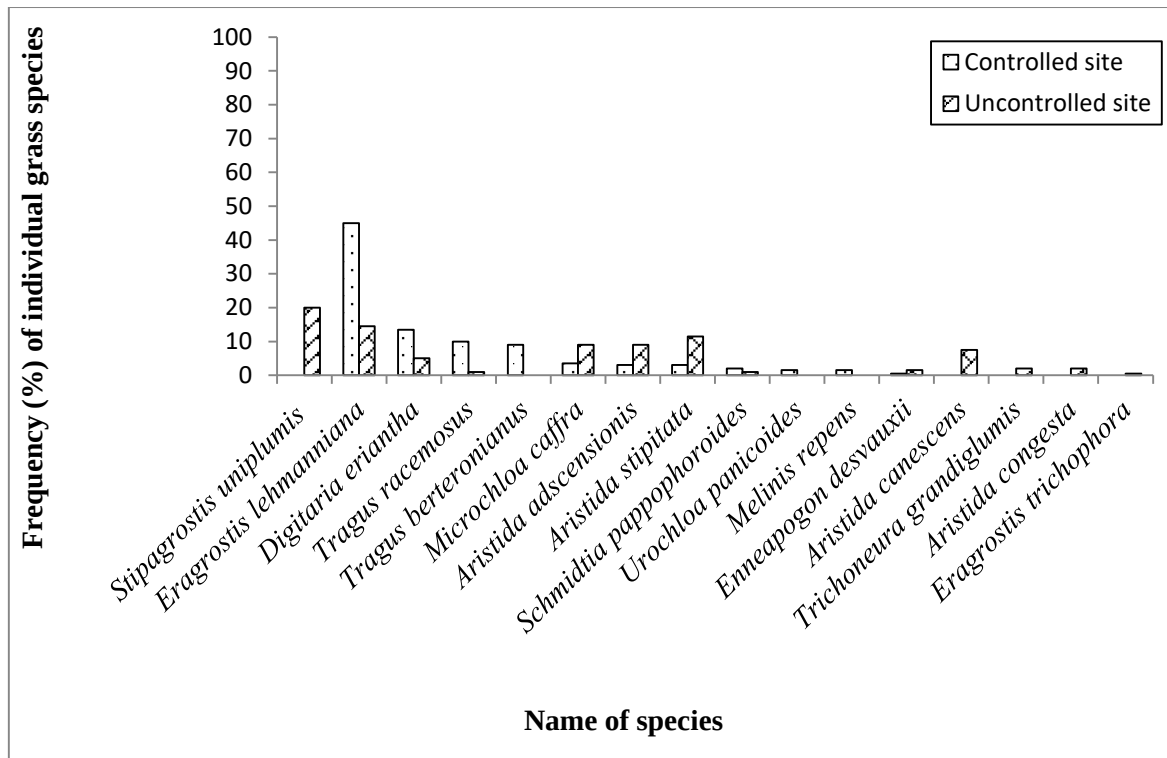


Figure 5.21: Frequency (%) of individual grass species in the woody controlled and uncontrolled sites at Manthe.

There was a total of 3 survey sites selected at the Taung Dam study site, namely, Taung Dam (102), Taung Dam (98) and Taung Dam (100). These will be discussed below:

5.3.6 Taung Dam (102)

In the Taung Dam (102) controlled site, the most prominent grass species were *C. hirsutus* (58%) and *T. berteronianus* (2%), while *D. eriantha* (24%) was the most prominent in the uncontrolled site (Figure 5.22). The results presented in Figure 5.22 were rather unexpected because a “good” palatable grass, such as *D. eriantha*, occurred in the uncontrolled site, but was absent in the controlled site.

Comparable to this finding, Karuaera (2011) noted that sites with fewer bush densities had a higher grass cover as to bush encroached sites. These findings do not correspond with the observations in Taung Dam (102) controlled and uncontrolled sites (Figure 5.22). Although *T. berteronianus* and *C. hirsutus* had the highest frequencies (%), it is indicated that a wider grass species variation occurred in the controlled site as compared to the uncontrolled site, although with low frequencies (%) (Figure 5.22). This could be attributed to the high bush densities surveyed in the uncontrolled site as compared to the controlled site (refer to

Figure 5.11). According to Richter *et al.* (2001), a reduction in woody plant densities did not have any effect on the grass species composition in the Molopo and mixed vaalbos thornveld types. This contradicts the findings in Figure 5.11, where it could be seen that bush clearing does result in grass species composition change (Figure 5.22).

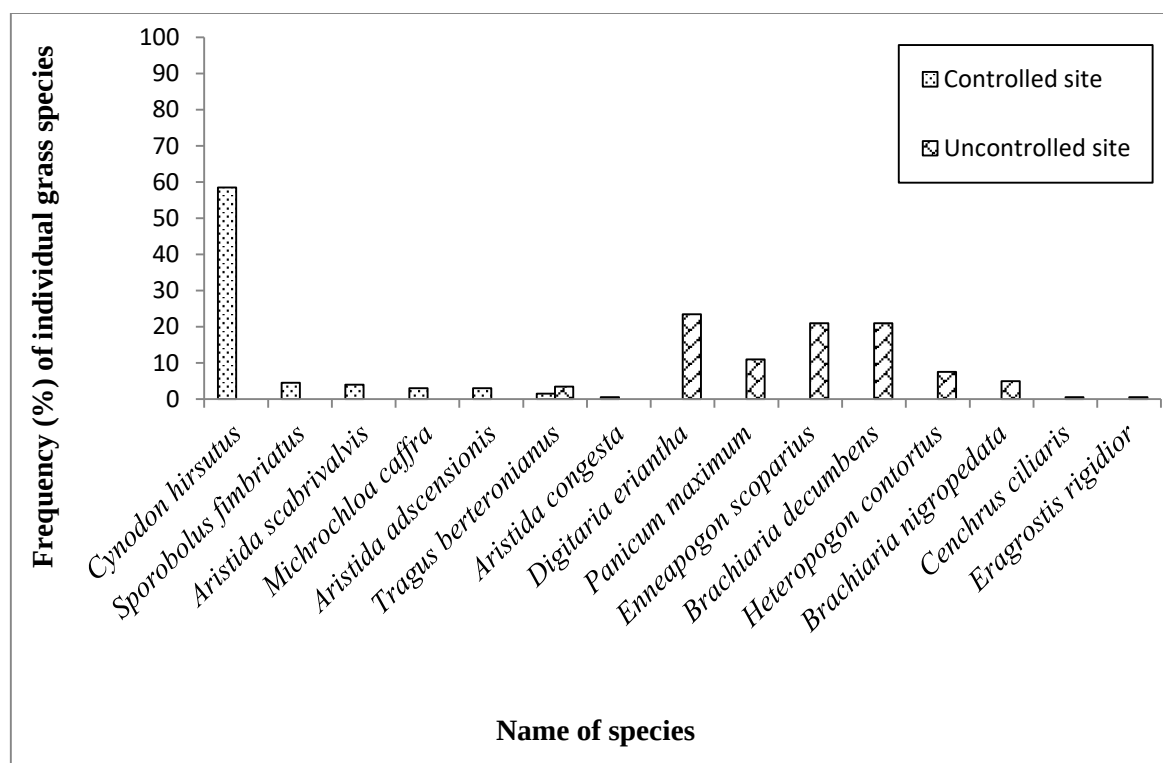


Figure 5.22: Frequency (%) of individual grass species in the woody controlled and uncontrolled sites at Taung Dam (102).

5.3.7 Taung Dam (98)

The most prominent grass species in the Taung Dam (98) controlled site were the well adaptable and palatable perennial grass, *C. dactylon* (50%), followed by the tufted, non-palatable annual grasses, *A. adscensionis* (14%) and *U. panicoides* (10%) (Figure 5.23). These grass species are found in “disturbed” areas where the vegetation is disturbed by overgrazing, trampling or in uncultivated lands. The most prominent grass species in the uncontrolled site was the perennial grass, *C. dactylon*, followed by the tufted, annual grass, *A. adscensionis* and the sparse tufted annual grass, *T. berteronianus* (Van Oudsthoorn, 1992) (Figure 5.23). *C. dactylon* and *A. adscensionis* were both observed to be the most prominent grass species in both the controlled and uncontrolled sites (Figure 5.23). The uncontrolled site was observed to experience continuous non-selective grazing due to high stocking rates.

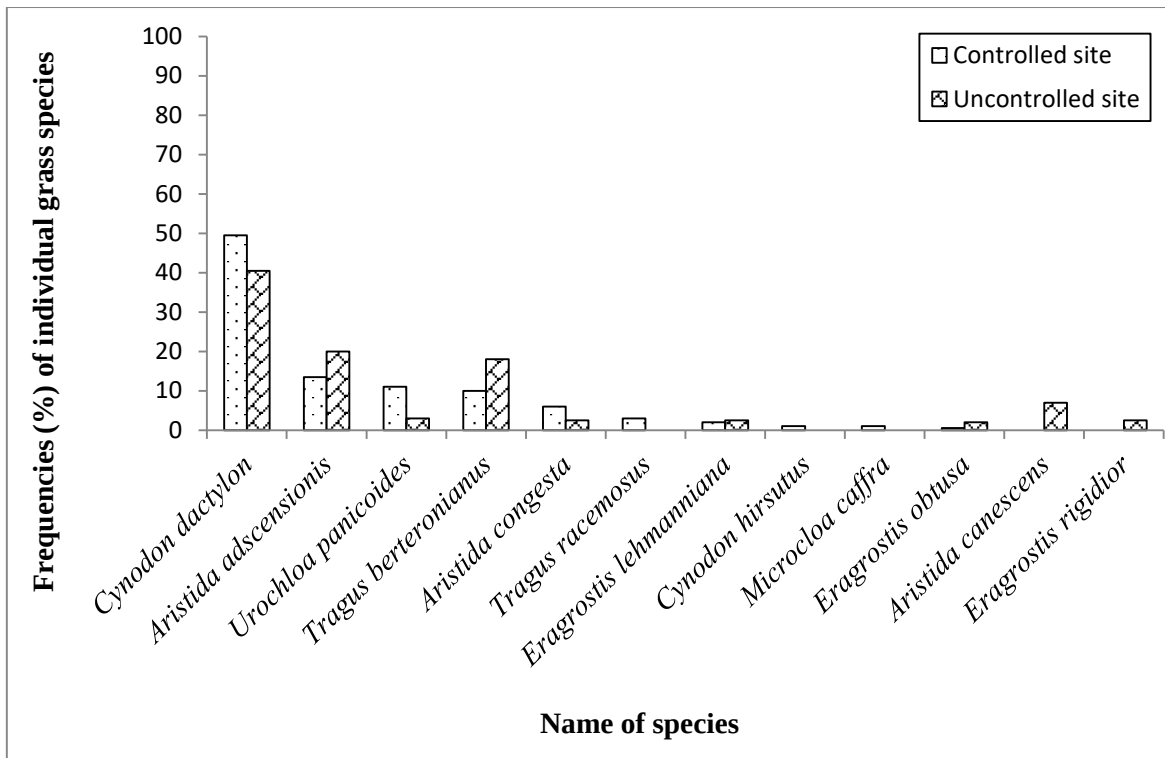


Figure 5.23: Frequency (%) of individual grass species in the woody controlled and uncontrolled sites at Taung Dam (98).

5.3.8 Taung Dam (100)

The most prominent grass species in the Taung Dam (100) controlled site were the perennial (sometimes annual) *A. congesta* (46%), followed by the tufted perennial grass, *A. canescens* (12%) and the annual grass, *E. lehmanniana* (6%) (Figure 5.24). *A. congesta* is considered as a palatable grass in semi-arid regions. *A. congesta*, *A. adscensionis* and *E. lehmanniana*, provide stability to disturbed soils and bare patches under severe conditions (Van Oudsthoorn, 1992). However, the high occurrence of the perennial grass, *A. congesta* (46%), was as a result of the bush clearing project implemented by the WfW programme (Figure 5.24). The bush clearing project at the Taung Dam (100) controlled site also contributed to the establishment of *A. adscensionis* and *E. lehmanniana*, although still at a low frequency (Figure 5.24). The results in Figure 5.24 also indicate that, in the Taung Dam (100) uncontrolled site, the most prominent grass species were the perennial (sometimes annual) grass *A. congesta* (32%), followed by *A. scabrivalvis* (8%) and *C. dactylon* (7%) (Figure 5.24).

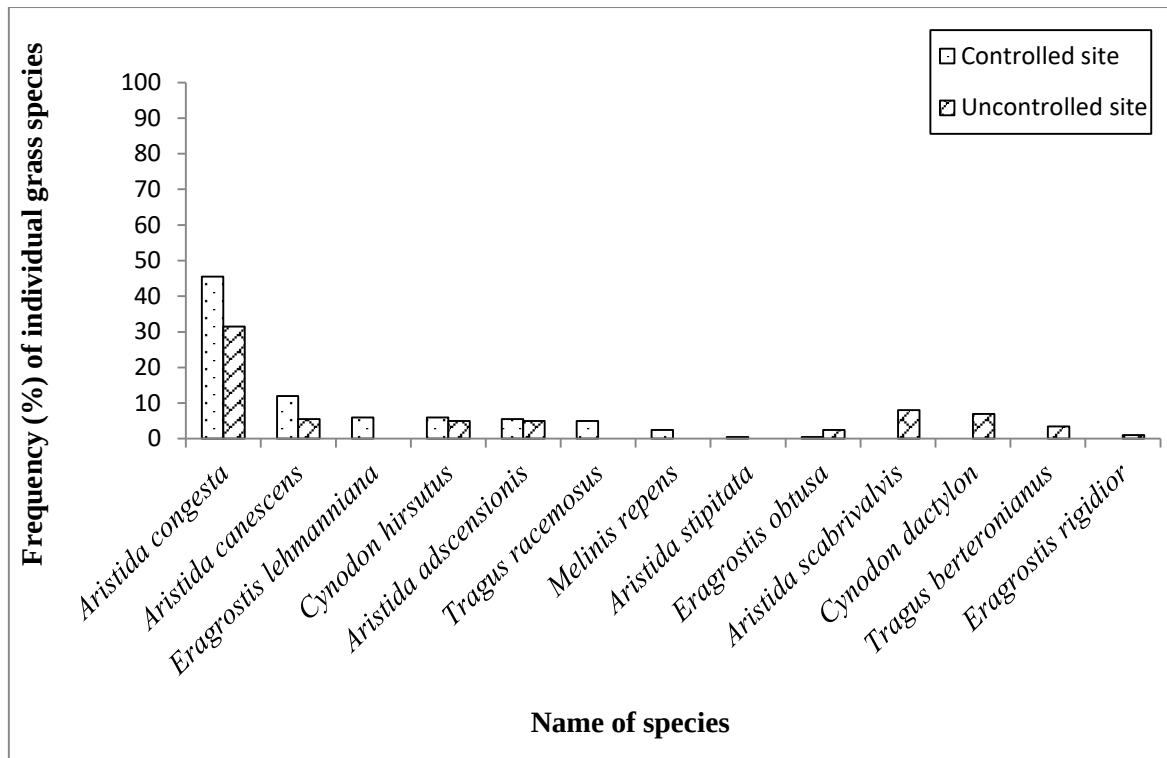


Figure 5.24: Frequency (%) of individual grass species identified on controlled and uncontrolled sites at Taung Dam (100).

5.3.9 General discussion of the grass component in the study sites

Although *D. eriantha* is a preferred and palatable grazing grass and *A. canescens* is considered unpalatable and an un-preferred grazing grass, the latter species has the ability to grow and survive in disturbed and degraded rangelands. The adaptability ensures that the species can act as a soil stabiliser and then serves as “nursing” plants for the climax, perennial, palatable grasses that will establish later (Van Oudsthoorn, 1992). According to Abdallah *et al.* (2008), grasses growing beneath a woody cover are mostly protected from grazing effects. However, grass species’ variability is higher in the uncontrolled sites than compared to the controlled sites. This, however, can be accounted to the fact that there were high woody species densities in certain controlled sites than compared to the uncontrolled sites. Grass species differ in morphology (rhizomatous/stoloniferous or tufted), phenology and palatability, thus, grass species composition is a key determinant in acceptability for grazing herbivores (Snyman, 1999). Various scientific studies recognise that soil nutrients availability plays a major role in vegetation structure, composition and productivity (Snyman, 2002). Tefera *et al.* (2010) observed that perennial grasses such as *P. maximum* and *C. ciliaris* were most frequent in areas with low stocking rates (commercial areas), whilst *T. berteronianus*, *B. eruciformis* and *A. bipartita* had highest occurrence in areas with

high and medium stocking rates, including areas that are communally managed. Tefera *et al.* (2010) also indicated that, under heavy grazing, normally occurring in communal lands, highly desirable perennial grasses are prone to extinction, while, on the contrary, short-lived and less desirable species, which produce large amounts of smaller or better dispersed seeds, such as *T. berteronianus* and *Aristida* species, may not be as vulnerable. These findings, corresponds, with the results obtained in the Manthe controlled and uncontrolled sites. The most prominent grasses in these sites were *T. berteronianus* and *C. hirsutus* (Figure 5.21). Nevertheless, the limitations of the classification of the grasses based on their desirability indicate the nature of the grass layer in terms of ecology (response to grazing) and palatability values (Abule *et al.*, 2007). *T. berteronianus* is a good indicator of veld disturbance and it is useful in stabilising soil conditions, thus acting as a nurse plant, creating the “opportunity” for other grasses to establish and grow (Van Oudsthoorn, 1992).

The impact of grazing on species diversity does not only depend on the degree of which grazing stimulates disturbances, but also on moisture availability before and after disturbance (Engle *et al.*, 2000). Increased bush densities transform susceptible grazing lands into dense thicket-forming noxious trees/shrubs and suppress palatable grasses and forbs through competition, thus resulting in unsuitable rangelands for browsing and grazing animals (Negasa *et al.*, 2014). Oba *et al.* (2000) reported that, in the Borana rangelands bush encroachment has negatively affected livestock productivity and survival especially during drought periods, when forage scarcity is greatest. In recent studies of Negasa *et al.* (2014), a total of 3 annual and 13 perennial grass species were recorded after the control actions that were applied in the study areas. The high distribution of perennial grass species may be associated with the eradication of invasive woody plants which have a negative impact on perennial grass establishment through vigorous competition for available water, nutrients and light. Increased competition between tree and grass species before control also contributed to increased bush densities observed in the selected survey sites.

Smit (2005) stated that, *S. mellifera* can directly compete with the grass sward because of its shallow tap root system. *S. mellifera* was the most prominent woody species in the controlled sites as compared to the uncontrolled sites. The results in Figure 5.12, Figure 5.13, Figure 5.14 and Figure 5.16 indicate that *S. mellifera* was the most dominant woody encroacher; however, Figure 5.24 indicates a well-established grass sward with *A. congesta*, occurring at a frequency of 46%, *A. adscensionis*, *C. hirsutus* and

E. lehmanniana at 6% and *Tragus racemosus* at 5% in the controlled sites, as was expected that grass establishment will be limited beneath dense stands of *S. mellifera*. Grass competition has the ability to suppress woody plant growth at all life stages and has profound effects on seedling survival, growth and colonization. Grass competition can suppress growth at all life stages of woody plants with a pronounced effect on sapling growth, survival and establishment (Riginos & Young, 2007; Riginos, 2009; Cramer *et al.*, 2010; Kambatuku *et al.*, 2011; Cramer *et al.*, 2012). Grass competition restrains seedling establishment even in the vicinity of abundant nutrient availability and, therefore, has a negative effect on seedling survival and development (Kraaij & Ward, 2006; Grellier *et al.*, 2012; Vadigi & Ward, 2013). Hence, grass competition will negatively affect sapling survival and growth (Vadigi & Ward, 2013).

CHAPTER 6

CHANGES IN SOIL CHEMICAL ANALYSIS IN BUSH CONTROLLED AND UNCONTROLLED SITES

The changes in chemical soil parameters for both the uncontrolled and controlled sites in the Taung area will be discussed together.

6.1 Soil pH

Figure 6.1 indicates that the recorded soil pH is higher in the uncontrolled sites than the controlled sites, except for soils at the Moretele, Magogong and Taung Dam (102) study sites. The soil in the Magogong controlled site had a neutral soil pH of 7.17 as compared to the soil in the uncontrolled site which was slightly acidic at a pH of 6.80 (Figure 6.1). The soil pH in the Magogong controlled and uncontrolled sites are worth mentioning, because these pH values are considered to be suitable for optimum plant growth (Austin, 2002; Pausas *et al.*, 2003; Medinski, 2007). The highest soil pH of 8.17 was recorded in the Moretele uncontrolled site and soils at the Taung Dam (100) controlled site was lowest at a pH value of 5.46 (Figure 6.1). Overall, the pH of the soils at all the study sites was slightly acidic to neutral and ranged between 5.4 and 8.2 (Table 6.1 and Figure 6.1).

The soil pH in the controlled and uncontrolled sites concurred with the findings of Doughill and Cox (2007) who did studies in the Kalahari and compared soil properties in a controlled area and a bush encroached zone. It was concluded by Doughill and Cox (2007) that, there was no significant difference in surface and subsurface moisture and pH levels between encroached and non-encroached sites. Belsky *et al.* (1989), Hagos and Smit (2005) and Abdallah *et al.* (2008) reported a high soil pH under *V. tortilis* (subsp.) *raddiana* canopies. Comparable results were also reported by Abule *et al.* (2005) who recorded a higher soil pH in habitats with a denser tree canopy. According to Abdallah *et al.* (2008) the exact reasons for these differences are not known. A higher pH under canopies of trees is often correlated with a higher content of exchangeable cations in the sub-habitat (Hagos & Smit, 2005). The higher or lower cover and production of herbaceous plants under the canopies of *V. tortilis* (subsp.) *raddiana* trees and in open sub-habitats may also have an influence on the soil nutrient status (Abdallah *et al.*, 2008).

According to Belsky *et al.* (1989), Callaway *et al.* (1991), Davids *et al.* (2005) and Abdallah *et al.* (2008), soils beneath tree canopies are more fertile than soils from the surrounding grasslands. Hagos and Smit (2005) reported that *S. mellifera* possessed some beneficial properties such as leaf litter which contribute positively to the increase of soil pH under tree canopy cover. High bush densities at the Moretele, Magogong and Taung Dam (102) (See Chapter 5 Figure 5.1; Figure 5.7 and Figure 5.11) could account for the high soil pH in these sites (Figure 6.1).

Table 6.1: Soil pH in the different study sites.

Name of Study site	pH	Acidity
1.1 Moretele controlled	8.17	Moderately alkaline
1.2 Moretele uncontrolled	6.97	Neutral
2.1 Myra (87) controlled	5.57	Moderately acidic
2.2 Myra (87) uncontrolled	6.06	Slightly acidic
3.1 Myra (76) controlled	6.19	Slightly acidic
3.2 Myra (76) uncontrolled	6.30	Slightly acidic
4.1Magogong controlled	7.17	Neutral
4.2Magogong uncontrolled	6.80	Neutral
5.1 Manthe controlled	5.59	Moderately acidic
5.2 Manthe uncontrolled	5.61	Moderately acidic
6.1 Taung Dam (102) controlled	5.94	Moderately acidic
6.2 Taung Dam (102) uncontrolled	5.87	Moderately acidic
7.1 Taung Dam (98) controlled	5.79	Moderately acidic
7.2 Taung Dam (98) uncontrolled	6.07	Slightly acidic
8.1 Taung Dam (100) controlled	5.46	Strongly acidic
8.2Taung Dam (100) uncontrolled	6.07	Slightly acidic

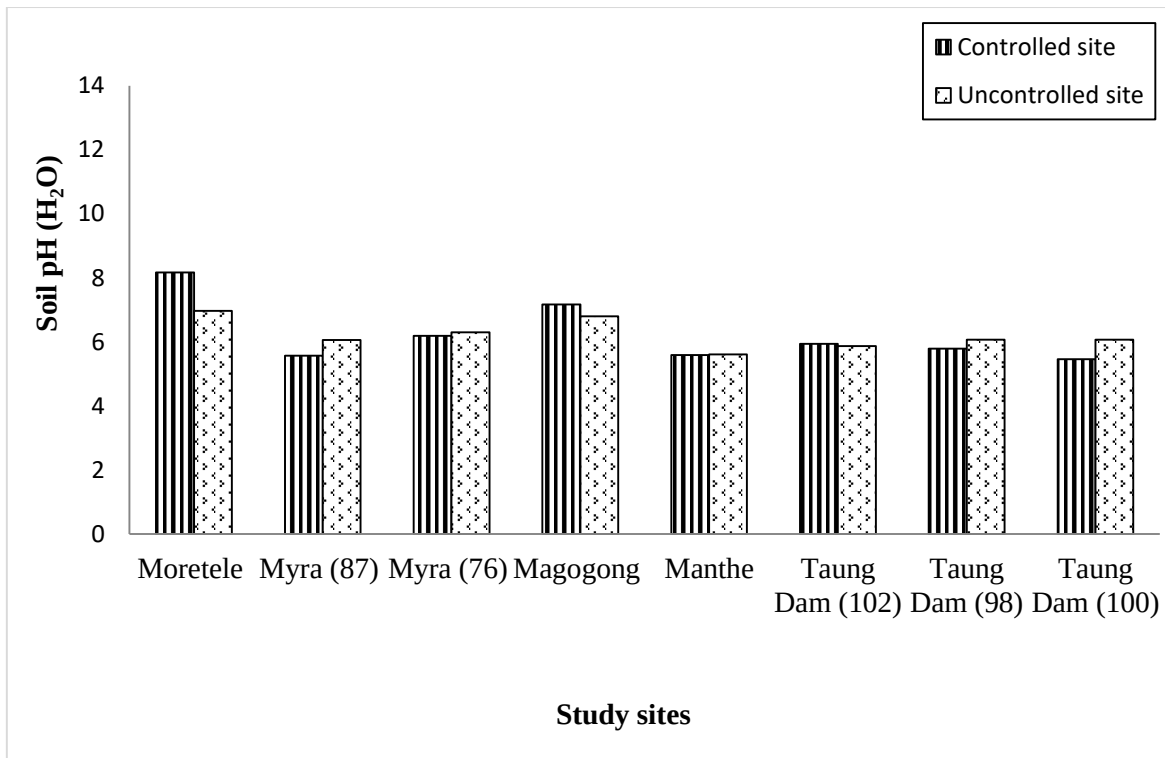


Figure 6.1: Soil pH in the different study sites within the Taung area.

6.2 Soil organic carbon (SOC)

The SOC was higher in the uncontrolled sites, compared to the controlled sites, with the exception of sites at Manthe, Taung Dam (98) and Taung Dam (100) controlled sites (Figure 6.25). The Taung Dam (102) uncontrolled site recorded the highest SOC of 1.81%, followed by Taung Dam (100) uncontrolled site at 0.71% and the Magogong uncontrolled site at 1.68% (Figure 6.25). The highest SOC was recorded in the Taung Dam (102) controlled site at 1.55%, followed by Magogong 1.42% and Myra (76) controlled site at 1.13% (Figure 6.25).

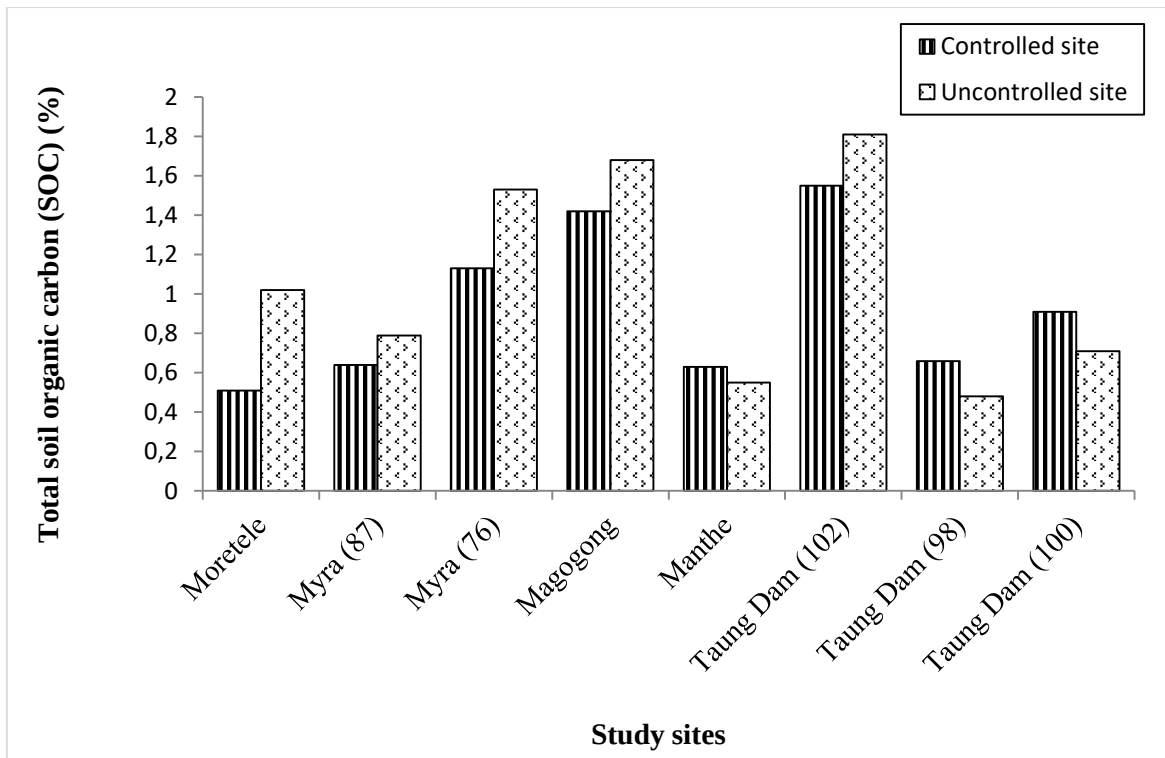


Figure 6.25: Total soil organic carbon (SOC) in the different study sites in Taung.

Woody species structure and composition, litter quality and quantity and soil characteristics can account for the differences in the concentrations of SOC (Boutton & Liao, 2010; Archer *et al.*, 2011). According to Hudak *et al.* (2003), Blaser *et al.* (2014) and Eldridge and Soliveres (2015), soil carbon sequestration may initially increase with bush encroachment, but then decline when bush densities become too high which will also inhibit understory grass growth. This corresponds with the results observed in Figure 6.25, where SOC was generally higher in the uncontrolled sites as compared to the controlled sites, except for the Manthe, Taung Dam (98) and Taung Dam (100) sites. Mzezewa and Gotosa (2009) illustrated that, SOC was higher under *Dichrostrachys cinerea* trees than in open areas. Shifting dominance among herbaceous and woody vegetation alters primary production, plant nutrient allocation, rooting depth and soil faunal communities in the underground soil layers (Nazeeruddin *et al.*, 1993; Wurth & Menzel, 2000). The latter affects nutrient cycling and carbon storage (Patthey *et al.*, 1999). When an increase in SOC does occur, it is often observed in the upper soil profile (0-20cm) (Boutton *et al.*, 2009) with accumulation rates ranging from 80 to 300 kg C ha⁻¹ yr⁻¹ (Wheeler *et al.*, 2007), and associated increases in soil N often occur (Wheeler *et al.*, 2007).

6.3 Available soil organic nitrogen (SON)

Figure 6.3 indicates that there was no SON deposits recorded, except for the Myra (76) uncontrolled site, Magogong controlled and uncontrolled sites and also in the soil of the Taung Dam (102) sites. However, there was no difference in these sites at Myra (76), Magogong and Taung Dam (102). The Magogong controlled site showed lower N availability of 0.01%, compared to the uncontrolled site which was measured at 0.03% (Figure 6.6.3). The soil in Taung Dam (102), however, indicated high N availability in the controlled site (0.05%), as compared to the uncontrolled site (0.03%) (Figure 6.3).

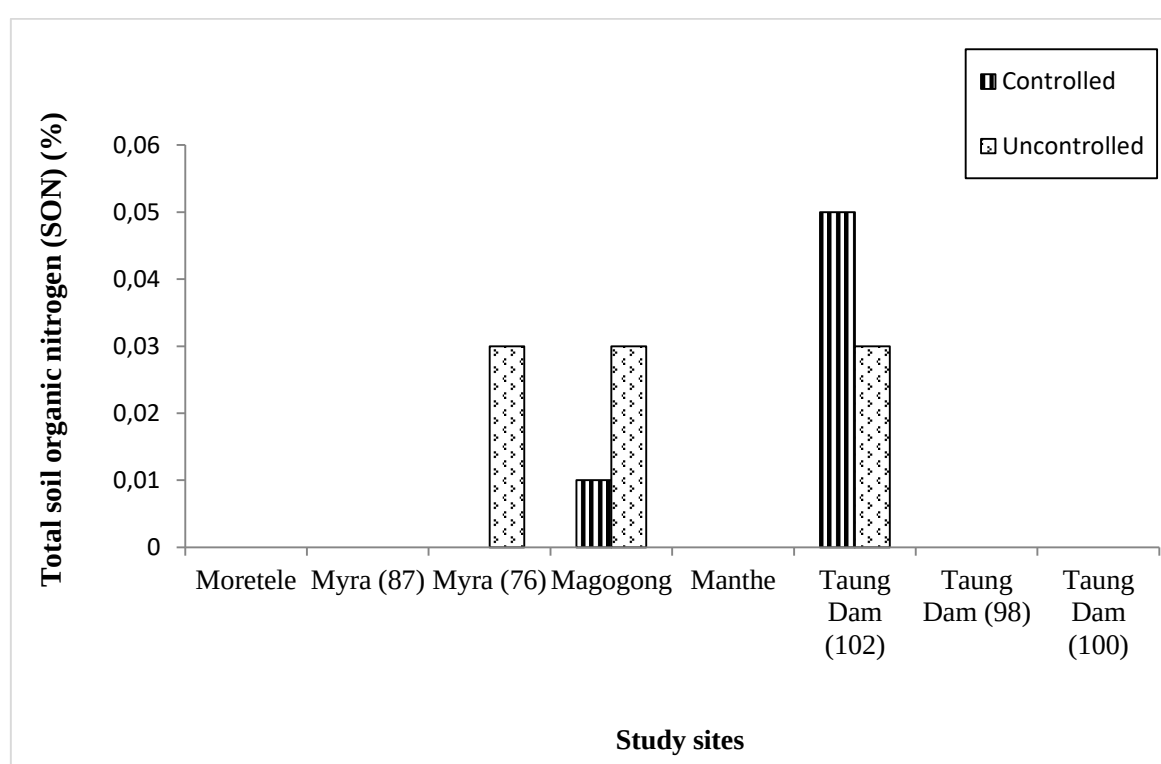


Figure 6.3: Total soil organic nitrogen (SON) availability of the soil sampled in the different study sites in Taung area.

The results in Figure 6.3 contradicts the findings reported by Wiegand *et al.* (2005) who revealed that, in semi-arid savannas, soil N content beneath large trees was about double than under small trees. Soil N availability, or the lack thereof in the study sites (Figure 6.3), may also be attributed to factors such as soil texture, nutrient leached sandy soils to a deeper soil profile and soils that have higher compaction and erosion rates. The latter will also affect the nutrient and water availability. Nitrogen availability may not necessarily depend on bush densities (Figure 6.3 and Table 6.1) but rather on available underground water (Sankaran *et al.*, 2008; Kgosisikoma, 2012). Grazing impacts may

influence soil C and N stocks. Studies have shown mixed results of grazing impacts on soil C and N. Studies by Reeder and Schuman (2002) showed increasing, neutral (Shrestha & Stahl, 2008) or even decreasing effects of soil C and N due to grazing (Pei *et al.*, 2008). Grazers can also affect legume species abundance, and hence N fixation rates. The latter may reduce N inputs to the soil (Allard *et al.*, 2007). Loss of soil C and N, associated with grazers, arises mainly from changes in SON decomposition and mineralisation rates or increased erosion under heavy grazing (Savadojo *et al.*, 2004; Wang *et al.*, 2010).

Soil N availability reaches a maximum at a neutral soil pH, because it is the most “favourable” range for soil microbes to mineralise N into organic matter, including organisms that fix N symbiotically (Foth, 1990; Boutton & Liao, 2010). According to Figure 6.1, soils at Myra (76), Magogong and Taung Dam (102) are all within the pH range of 6-8. All 16 survey sites (See Chapter 3, Figure 3.4) in this study are subjected to continuous non-selective grazing practices, which may have affected the C and N levels in the soil and changed the plant C and N below-ground (Reeder *et al.*, 2004; Semmartin *et al.*, 2010; Mohammed, 2013).

Sankaran *et al.* (2008) stated that, in 161 savanna sites across Africa, woody cover showed a strong negative dependence on soil N availability, which decreases sharply as soil N mineralisation potential increased to approximately $20 \text{ mg.g}^{-1}.\text{soil}^{-1}.\text{7 days}^{-1}$. The limited soil N in the study sites (Figure 6.3) is in agreement with findings by Kraaij and Ward (2006) and Sankaran *et al.* (2008), but differs from Hudak *et al.* (2003), indicating a higher soil N content in encroached sites. One explanation for the negative dependence of woody cover on N availability (Figure 6.3) could be an increase in the competitive vigour of the herbaceous layer under conditions of high N availability (Sankaran *et al.*, 2008). Evidence to support this comes from experiments which show that woody seedling survival and growth can decrease with soil N enrichment, either from direct pre-emption of nutrients by the herbaceous vegetation, or indirectly as a result of lowered light or water availability following the stimulation of herbaceous growth (Kraaij & Ward, 2006; Daryanto, 2013). Sankaran *et al.* (2008) reported that, long-term N enrichment (e.g. by N deposition) can potentially cause directional shifts in savanna structure towards lower tree densities or a more open system. Hagos and Smit (2005) reported that, for total N, concentrations of organic matter and Ca were recorded the highest near the area surrounding the stem base of *S. mellifera* as compared to further away from the stem.

6.4 Soil C:N ratios

Due to limited soil N, the only measurable C: N ratio was in the Myra (76) uncontrolled site and Magogong and Taung Dam (102) sites (Figure 6.4). The highest soil C: N ratio of 142:1 was as found in soil of the Magogong controlled site, as compared to 56:1 in the uncontrolled site (Figure 6.). Soil C:N in Myra (76) uncontrolled site was recorded at 51:1 and in the Taung Dam (102) controlled and uncontrolled sites at 36:1 and 52:1 respectively (Figure 6.4).

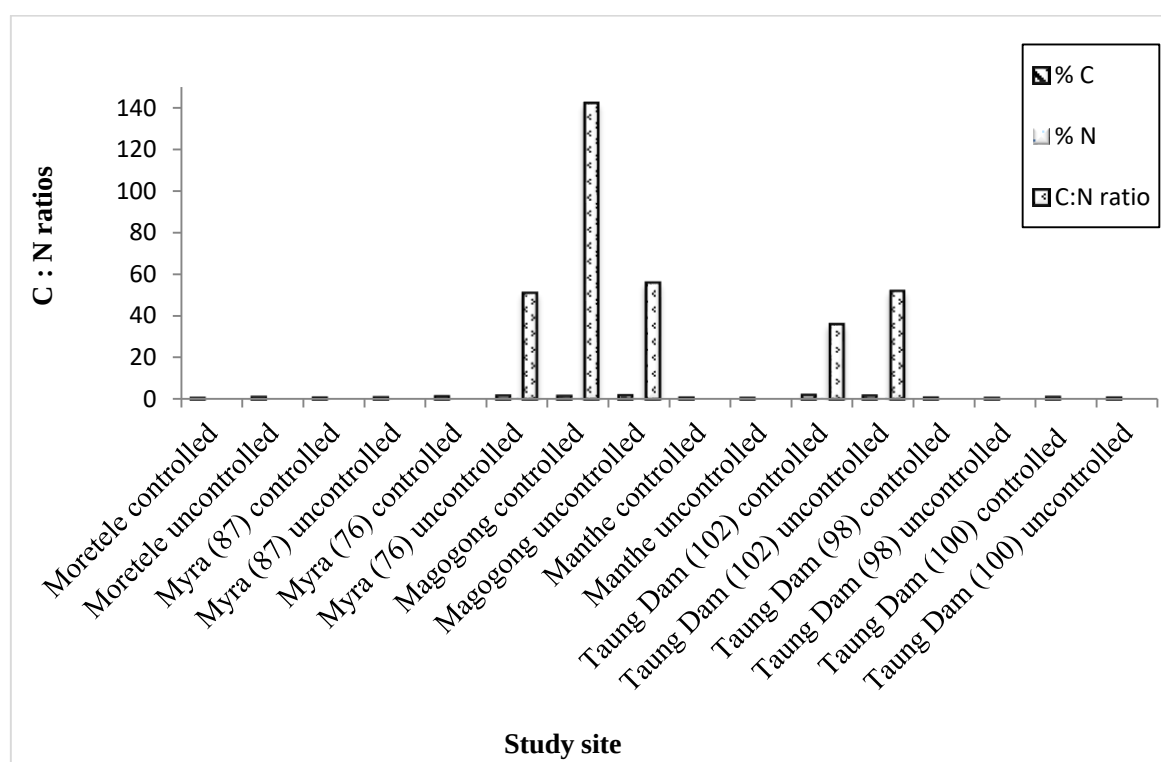


Figure 6.4: C:N ratios of the soil sampled in the different study sites in Taung area.

According to Hibbard *et al.* (2001), soil C and N concentrations under woody canopies were higher or equal to surface soil C and N concentrations between woody canopies. Similarly, Hudak *et al.* (2003) noted that, soils beneath a woody over-story exhibited higher C contents than soils beneath a solely herbaceous canopy, except on red clay loam soils. Woody plants such as *S. mellifera* enrich nutrient poor sandy soils in dry savannas through nitrogen fixing (Hagos & Smit, 2005). According to Kraaij and Ward (2006), changes in soil N could alter the competitive outcome between *S. mellifera* and *Eragrostis* species due to the different soil N requirements of these species. Soils beneath *S. mellifera* canopies were found to have higher levels of total N, SON and Ca than soils some distance away from the trees (Hagos & Smit, 2005). Findings of Hagos and Smit (2005) corresponds with the results reported in

the Taung Dam (102) controlled site (See Chapter 5, Figure 5.11) which indicated high bush densities of *S. mellifera*, and high levels of total N (Figure 6.3), SON and exchangeable base cations with regard to Ca^{2+} (See Chapter 6, Figure 6.15). Lal (2004) and Rutherford and Powrie (2010) demonstrated that, high grazing pressures also reduce soil C and N content because of a decline in plant cover, leading to reduced organic inputs. The results in Figure 6.4 concur with those reported by Lal (2004) and Rutherford and Powrie (2010), as all sites in this study were subjected to continuous non-selective grazing.

Hudak *et al.* (2003) reported that, litter at sampling points with a woody component had a higher N% and a lower C:N ratio than litter in sites with an herbaceous composition. This is, however, in contrast with the results in Figure 6.3, showing that soil N was limited. Severely encroached areas had lower C:N ratios throughout the soil profile than less encroached areas (Hudak *et al.*, 2003; Blaser *et al.*, 2014). With regard to the results in Myra (76) uncontrolled site and Taung Dam (102) survey sites; the results presented in Figure 6.3 and Figure 6.4 are in agreement with Hudak *et al.* (2003) and Blaser *et al.* (2014), indicating that encroached areas had lower soil C:N ratios compared to less encroached areas. This, however, contradicts results reported for soil in Magogong (Figure 6.4).

6.5 Calcium (Ca) content

Soil calcium (Ca) content was higher in controlled sites as compared to the uncontrolled sites, except for sites at Taung Dam (100) and Taung Dam (102) (Figure 6.5). The highest soil Ca concentrations were reported in the Moretele controlled site and the Magogong controlled site (Figure 6.5). Soil in the Magogong controlled site recorded the highest Ca content of 3379 mg/kg while the Ca content of the soils in the Taung Dam (98) controlled site was recorded the lowest at 720 mg/kg (Figure 6.5). The soil Ca content in the Taung Dam (100) uncontrolled site was higher at 946 mg/kg compared to 832 mg/kg of Ca in the controlled site (Figure 6.5).

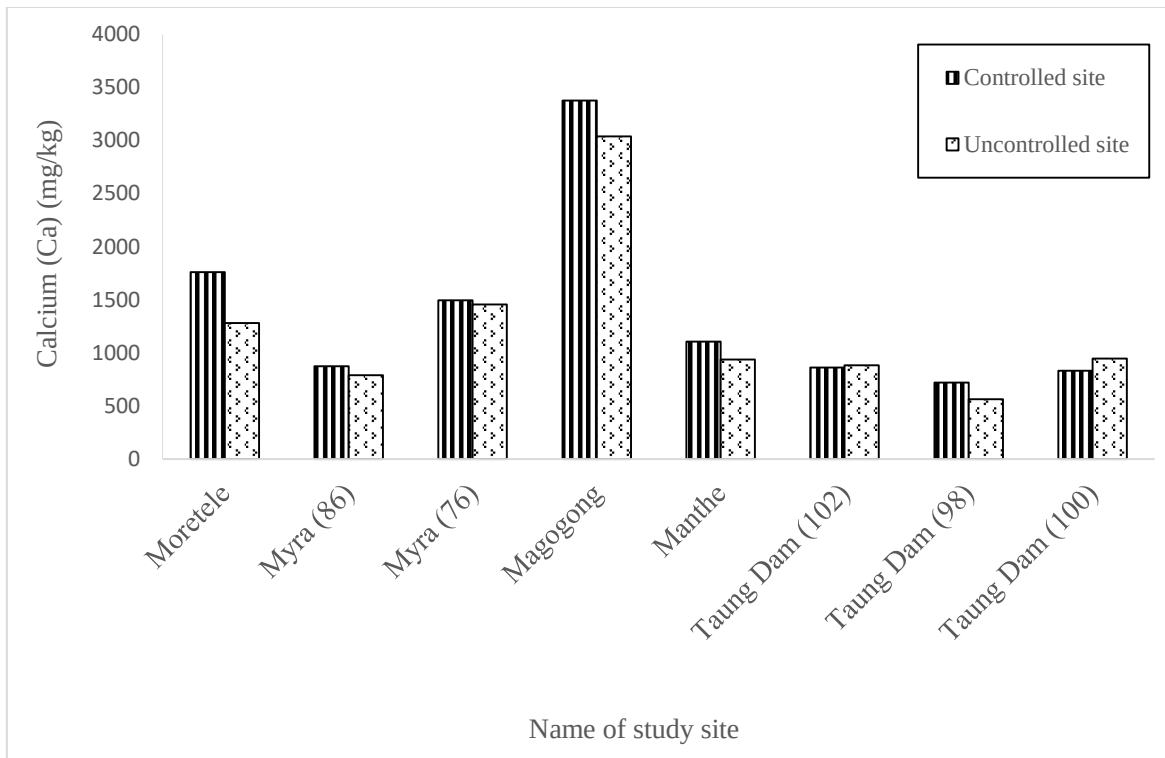


Figure 6.5: Mean Calcium (Ca) content of soil in the study area.

Soil nutrient accumulation is positively related to soil pH (Angassa *et al.*, 2012; Spectrum Agronomic Library, 2015). According to Spectrum Agronomic Library (2015), soils with a low pH have less available Ca and, therefore, lower soil Ca^{2+} content. The Magogong controlled site had the highest soil Ca content of 3379 mg/kg, compared to the uncontrolled sites having a Ca content of 3041 mg/kg (Figure 6.5). Both these sites had a high soil pH of 7.17 and 6.80 respectively (Figure 6.1). According to Belsky *et al.* (1989), reports of the effects of trees on savanna grasses and soils are not consistent. Soil organic matter (SOM, 0-10cm soil depth) and extractable P, K and Ca (0-15cm soil depth) were highest adjacent to trunks of both tree species but declined quickly within the canopy zone (Belsky *et al.*, 1989). Extractable K and Ca levels were approximately 15 times greater next to tree trunks, compared to the grassland zone (450 against 300 and 960 against 600 mg/kg dry soil, respectively) while extractable P levels were 27 times higher near the trunks (Belsky *et al.*, 1989). Essah *et al.* (2003) and Zhang *et al.* (2010) stated that, increased Ca was observed to increase sodium influx in a study on *Arabidopsis*. The results on soil Ca in Taung Dam (100) and Taung Dam (102) corresponds to the findings by Angassa *et al.* (2012), who reported higher soil Ca content in encroached sites than in non-encroached sites (Figure 6.5). These results also correspond with Molatlhegi (2008) and Mogodi (2009)

who found high soil Ca contents in the encroached sites as compared to the non-encroached sites.

6.6 Soil exchangeable calcium (Ca^{2+})

Figure 6.6 indicates high soil Ca^{2+} content in controlled sites as compared to the uncontrolled sites. The highest value of soil Ca^{2+} was recorded in the Magogong controlled site, recording 16.86 mg/kg and 2.81mg/kg in the soil of the Taung Dam (98) uncontrolled site was the lowest (Figure 6.6).

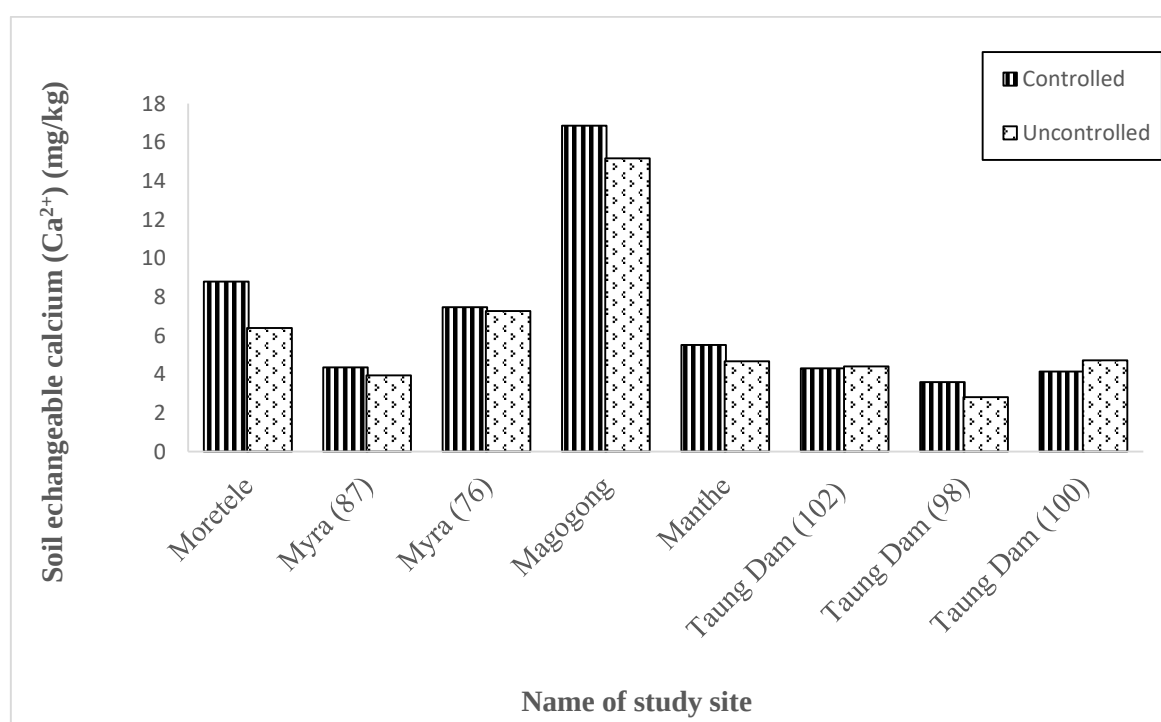


Figure 6.6: Mean exchangeable Calcium (Ca^{2+}) content of soil in the study area.

Exchangeable Calcium (Ca^{2+}) is the most prominent cation in sandy soils in semi-arid areas (Zhang *et al.*, 2013). Dryland soils generally have high levels of total Ca, reaching more than 5% by weight and occupy 75% - 85% of the cation-exchange sites (CEC) (Troeh & Thompson, 1993). Plant uptake and plant litter can accumulate Ca on the soil surface (Jobbágy & Jackson, 2001). An increased soil Ca^{2+} was reported by Zhang *et al.* (2013) in *Caragana microphylla* plantations occurring in semi-arid sandy soils in China. Zhang *et al.* (2013) observed an increase of Ca^{2+} in soil 22 years after *C. microphylla* plantations were established. The results shown in Figure 6.6 agree with the results observed by Zhang *et al.* (2013) that although there was high soil Ca^{2+} , the Ca accumulation in soil declined. It

was also observed that soil pH has a positive influence to soil Ca^{2+} and soil Ca accumulation. Zhang *et al.* (2013) stated that the mean available soil Ca^{2+} content may also be attributed to the co-effect of plant accumulation and different leaching rates of base cations, when compared to the soil Mg^{2+} content. The soil in the Magogong controlled site was observed to have the highest accumulated soil Ca content (3379 mg/kg) and soil in the Taung Dam (98) study site indicated the lowest soil Ca content of 563.5 mg/kg (Figure 6.). In this study, high soil Ca (Figure 6.5) accumulation occurred where there was also high soil Ca^{2+} concentrations (Figure 6.6).

6.7 Magnesium (Mg) content

Magnesium (Mg) content is higher in soils of the controlled sites, compared to the uncontrolled sites (Figure 6.7). Soil in the Magogong and Myra (76) uncontrolled sites, however, showed a higher Mg content compared to their respective controlled sites (Figure 6.7). The soils at the Magogong uncontrolled site recorded a Mg content of 824.5 mg/kg compared to 808 mg/kg in the controlled site. The Myra (76) uncontrolled site had a soil Mg content of 320 mg/kg which was higher than in the controlled site. The Moretele site had the lowest recorded soil Mg content of all the sites, with 65 mg/kg measured in the uncontrolled measured and 66.5 mg/kg in the controlled site (Figure 6.7).

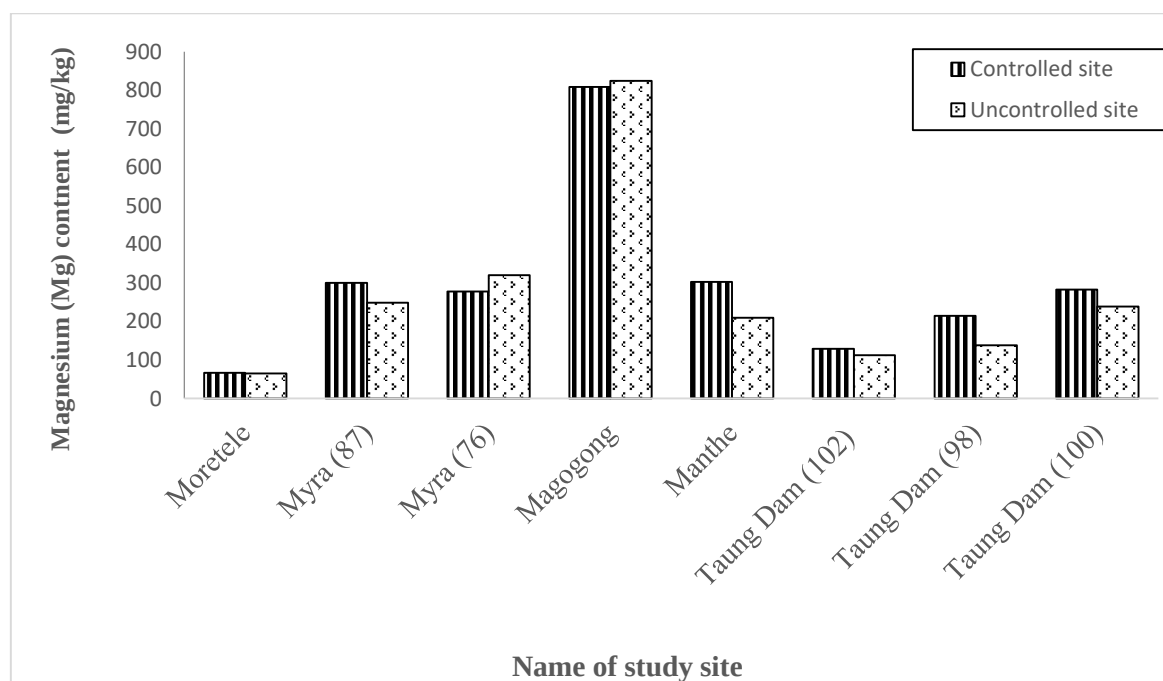


Figure 6.7: Mean magnesium (Mg) content of soil in the study area.

The results in Figure 6.7 are in agreement with Hagos and Smit (2005) who found high soil Mg under tree canopies compared to non-encroached areas. It has been reported that high soil Mg^{2+} contribute to soil structural properties and lowered infiltration rates compared to similar soils with a high Ca content (Dontsova & Norton, 1998).

The results in this study indicated that soils having a higher soil pH (Figure 6.1) also indicate higher soil Mg content (Figure 6.7). This, therefore, is an indication that a positive relationship exists between the soil pH and Mg content. Staugaitis and Rutkauskiene (2010) observed that, in the top soil layers (0-30cm) soil pH contributed to a high available soil Mg content. This corresponds with the results in the Magogong survey sites, which indicate high soil pH (Figure 6.1) and also a higher Mg content (Figure 6.). Evidence suggests that Mg content in the soil depends on soil texture, soil type, pH and humus content (Staugaitis & Rutkauskiene, 2010).

6.8 Soil exchangeable Magnesium (Mg^{2+})

The lowest recorded Mg^{2+} content was observed in the soils at the Moretele controlled (0.55 mg/kg) and uncontrolled (0.53 mg/kg) sites respectively (Figure 6.8). In the controlled Myra (76) study site, soil Mg^{2+} was recorded at 2.29 mg/kg and 2.63 mg/kg in the uncontrolled site (Figure 6.8). The results in Figure 6.8 indicate that the highest mean Mg^{2+} content of the soil was recorded in the Magogong uncontrolled site at 6.75 mg/kg.

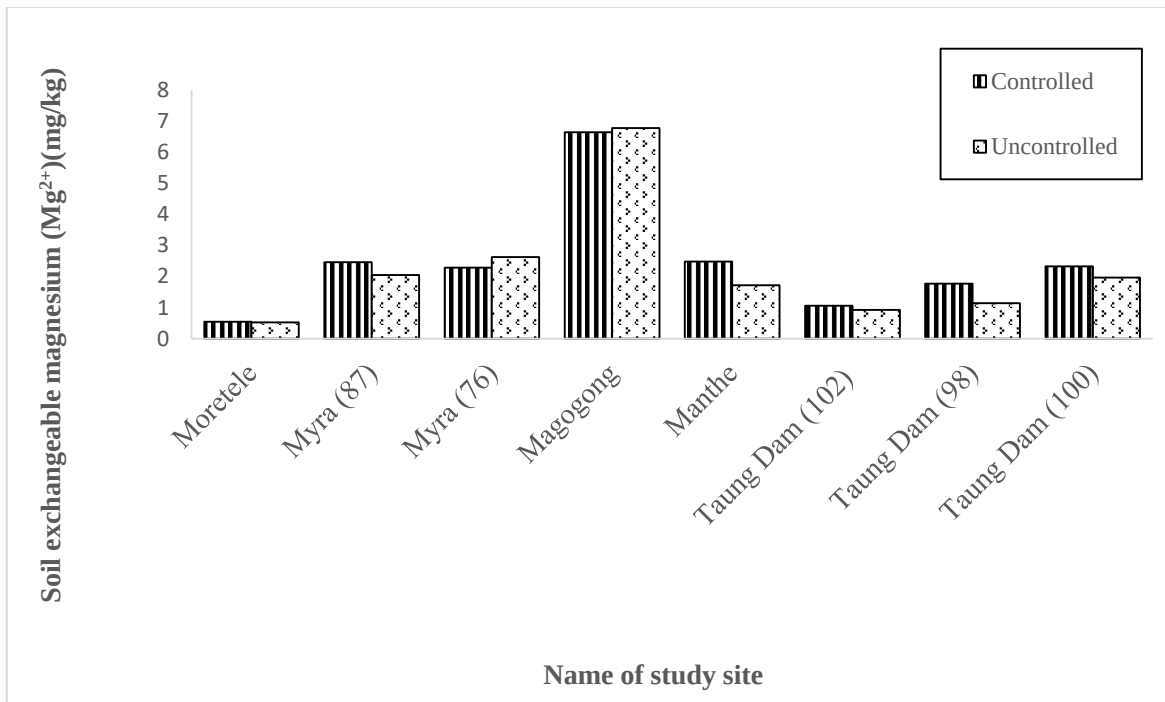


Figure 6.8: Mean exchangeable magnesium (Mg^{2+}) content of soil in the study area.

According to Okpamen *et al.* (2011) and Okpamen *et al.* (2013), the Mg content in soil is dependent on the status of the primary Mg bearing material found in such soils, which is subjected to vast environmental forces such as weathering and physical factors. Okpamen *et al.* (2013) conducted a study on four soil parent materials (Rhodicpaleudults, Rhodictropudalfts, Oxictropudalfts and Aquatic tropossament) and reported that soil depths and pH were correlated with water soluble Mg^{2+} , exchangeable Mg^{2+} and non-available Mg^{2+} .

According to Okpamen *et al.* (2013), soil pH in various profiles and parent materials were low and fell within the acid pH range. Kamaljit *et al.* (2005) studied the status of major elements in the soils of Sri Lanka grasslands in central Queenstown in Australia and reported high amounts of K, Mg and Ca in deep soils. Okpamen *et al.* (2013) observed that there was a positive correlation between soil pH and exchangeable Mg^{2+} ($r = 0.693$). Soil pH in the Myra (76) controlled site was 6.19 which was similar to that of the soil in the uncontrolled site, which was recorded at 6.30 (refer to Table 6.1). The results observed in Moretele and Myra (76) study sites, therefore, indicate that the soil exchangeable Mg^{2+} content increased with an increase in soil pH (refer to Table 6.1). This corresponds with Okpamen *et al.* (2013) that there exists a positive relationship between soil pH and exchangeable Mg^{2+} content. These findings were also in agreement with Fayemi and Lombin (1975) that, there is a

significant positive relationship between soil pH, exchangeable Mg^{2+} and soil depth. Angassa *et al.* (2012) observed a decline in soil pH and attributed this decline to the depletion of basic cations such as Ca^{2+} and Mg^{2+} in soil as well as high leaching rates.

According to Okpamen *et al.* (2013), it would be expected that the Magogong uncontrolled site would have a high soil pH when compared to the controlled site (Table 6.1) since it contains a higher soil exchangeable Mg^{2+} . The results in Table 6.1, however, indicated that the soil in the Magogong controlled site had a high pH (7.17) compared to the uncontrolled site (6.30). The high exchangeable Mg^{2+} content of the soil in the Magogong uncontrolled site can, therefore, be attributed to high bush densities (See Chapter 5, Figure 5.7). The results observed in the Magogong survey sites, therefore, corresponds with Zhang *et al.* (2013) who observed an increase in soil exchangeable Mg^{2+} under plantation of *Caragana microphylla* in semi-arid sandy areas in China. This was also the case for the soils in the Manthe controlled and uncontrolled study sites, as well as Myra (87) controlled and uncontrolled sites (Table 6.1). The results in Figure 6.8 can be attributed to soil physical and chemical properties, as indicated by Okpamen *et al.* (2011) and soil depth as indicated by Fayemi and Lombin (1975).

6.9 Potassium (K) content

Potassium (K) content was generally high in the soils of the controlled sites compared to the uncontrolled sites, excluding the Magogong uncontrolled site which was slightly higher than that in the controlled site (Figure 6.9). The highest concentration of soil K was recorded in the Magogong uncontrolled site at 566 mg/kg compared to its controlled site which recorded a K content of 557.5 mg/kg. The Myra (76) uncontrolled site indicated the lowest soil K content at 140 mg/kg and its controlled site recorded a K content of 170 mg/kg (Figure 6.). The Myra (76) controlled site, however, measured a higher soil K concentration than in the Taung Dam (98) site (Figure 6.9).

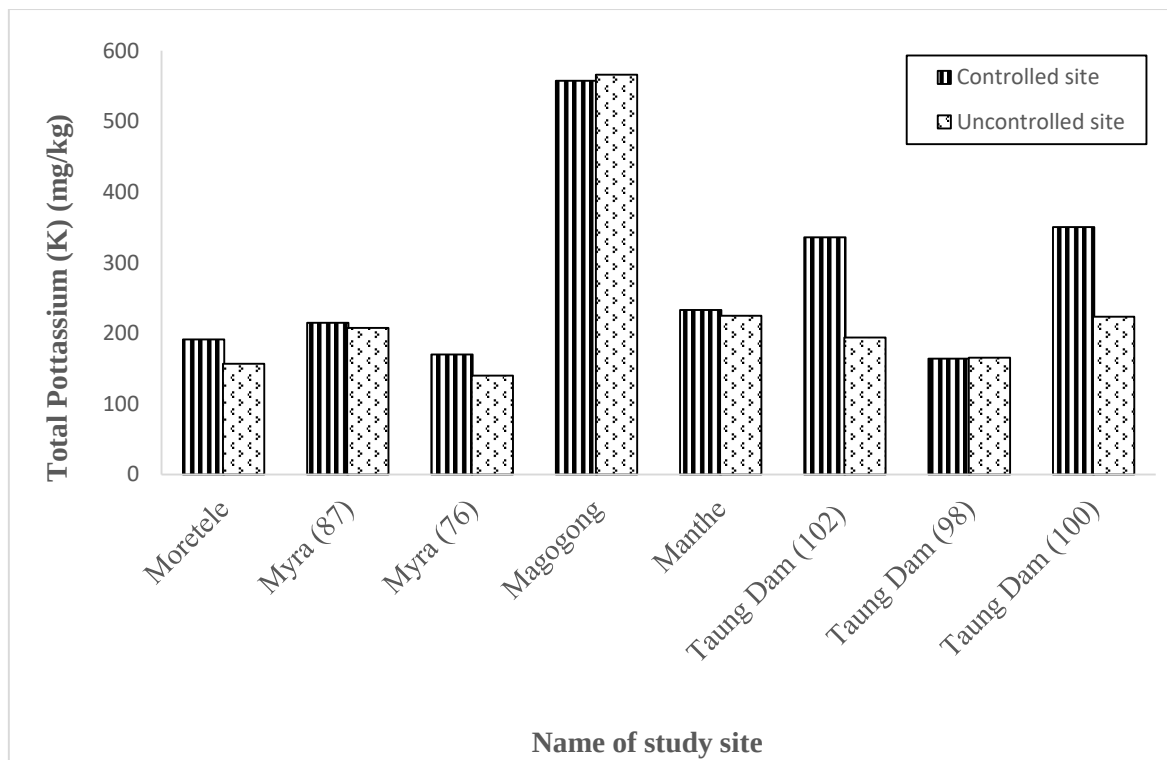


Figure 6.9: Mean potassium (K) content of soil in the study area.

Molatlhegi (2008) observed that the K content of the soil in the uncontrolled reference site in the Molopo was lower than in the sites where bush encroachment occurred. Jackson *et al.* (2002) and Asner *et al.* (2003) reported that, shifts in soil nutrient contents are influenced by the process of bush encroachment. The results in Figure 6.9 corresponds to Jackson *et al.* (2002) and Asner *et al.* (2003) who also reported higher concentrations of soil K content in the controlled sites as compared to the uncontrolled sites.

Bosch and Van Wyk (1970) found higher N, P, K, Mg and Ca in soil under *Combretum apiculatum*, *Boscia albitrunca*, *V. tortilis* and *V. senegal* canopies in comparison to soils in open areas. These results are in agreement with Figure 6.9. The results of the study confirmed the differences in the soil nutrient status between the various sub-habitats, which occurred in a specific spatial gradient from the stem base of the plants under the canopy towards the open areas. Soil P and K contents and to a lesser extent Mg, were also higher under the tree canopies, while pH was lower. Nutrient enrichment of soils occur in areas with high *S. mellifera* densities in the Kalahari thornveld, compared to a lower nutrient status of sandy soil, commonly found in this area (Hagos & Smit, 2005). The higher woody plant densities (See Chapter 5, Table 5.1) could account for the observed soil K content in the Magogong and Myra (76) uncontrolled sites (Figure 6.9). This corresponds to the results

reported by Bosch and Van Wyk (1970) and Hagos and Smit (2005) who found an increase in soil nutrient status as a result of the presence of woody species. Soil nutrient status can be directly or indirectly (*via* trampling and recycling of nutrients respectively) affected by livestock grazing (Augustine, 2003; Solomon *et al.*, 2007), mostly due to the deposition of animal dung.

6.10 Soil exchangeable Potassium (K^+)

Soil in the Magogong uncontrolled site had slightly higher soil exchangeable potassium (K^+) content compared to the soil in the controlled site. Both recorded a mean value of 1.45 mg/kg for the uncontrolled site and 1.43 mg/kg respectively (refer to Figure 6.10). All other sites indicated a higher mean soil K^+ in the controlled sites as compared to their respective uncontrolled sites (Figure 6.10).

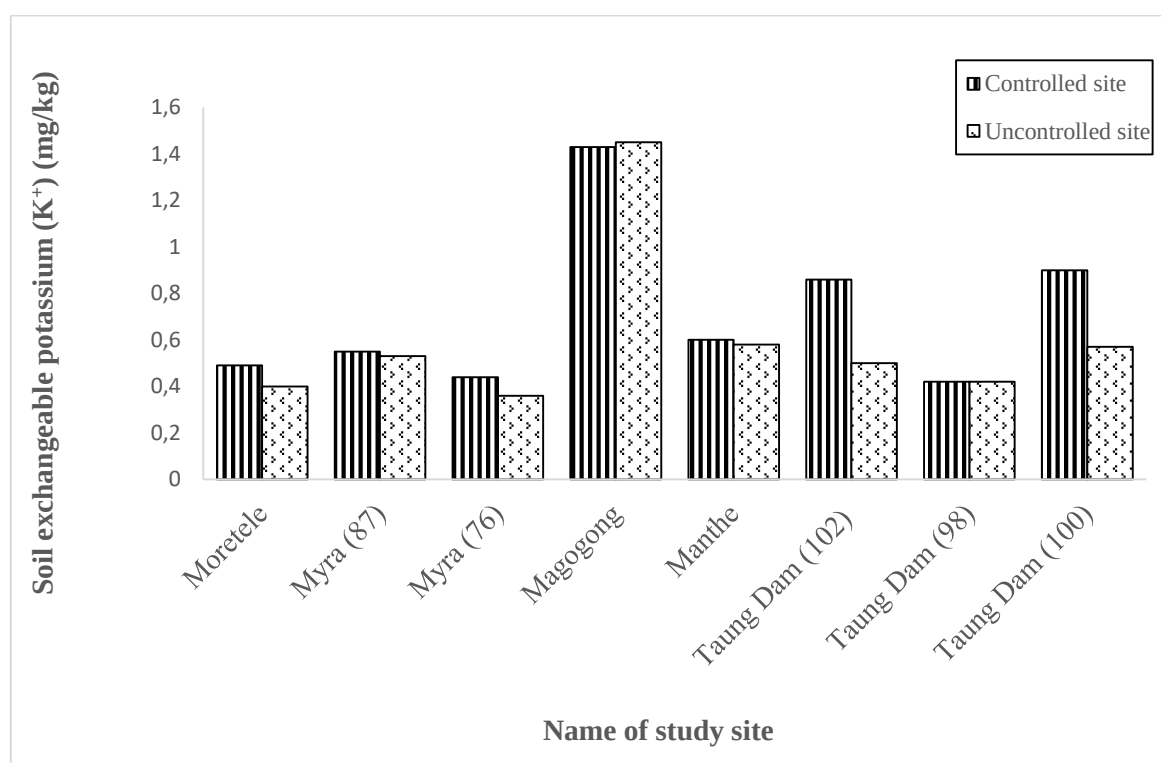


Figure 6.10: Mean exchangeable potassium (K^+) content of soil in the study area.

In dryland ecosystems, K^+ accumulates at the surface due to upward transport by plant roots and accumulation of K in plant litter (Jobbágy & Jackson, 2001). According to Zhang *et al.* (2013), hydrated K ions are the same size as ammonium ions and adsorbed with about the same strength, but more weakly adsorbed when compared to Ca and Mg ions. This may contribute to the higher exchangeability of the K ions (Marschner & Rengel, 2007).

The differences in the observed values of soil K^+ in the Magogong controlled and uncontrolled sites (Figure 6.10) can be attributed to the different woody densities (See Chapter 5, Figure 5.7). This corresponds to results obtained by Zhang *et al.* (2013), who reported a positive correlation between soil K^+ and higher woody densities. The observed soil K^+ content (Figure 6.10) may have been influenced by the soil texture and depth at which the soils were sampled in the different study sites (Zhang *et al.*, 2013).

6.11 Sodium (Na) content

Figure 6.11 indicates that the soils Na concentration in the Taung study area are generally higher in the controlled sites than in the uncontrolled sites. However, the soil in Moretele, Myra (76) and Taung Dam (102) sites recorded higher Na in the uncontrolled sites compared to the controlled sites (Figure 6.11). Soil samples from uncontrolled sites in Moretele, Myra (76) uncontrolled site and Taung Dam (102) were observed to be higher than their respective controlled sites (Figure 6.11). The highest recorded soil Na content was 9.5 mg/kg observed in soil sampled in the Taung Dam (97) controlled site. The lowest soil Na content was observed in soil sampled in the Taung Dam (98) and the Taung Dam (100) uncontrolled study sites which were both measured at 1.5 mg/kg respectively (refer to Figure 6.11).

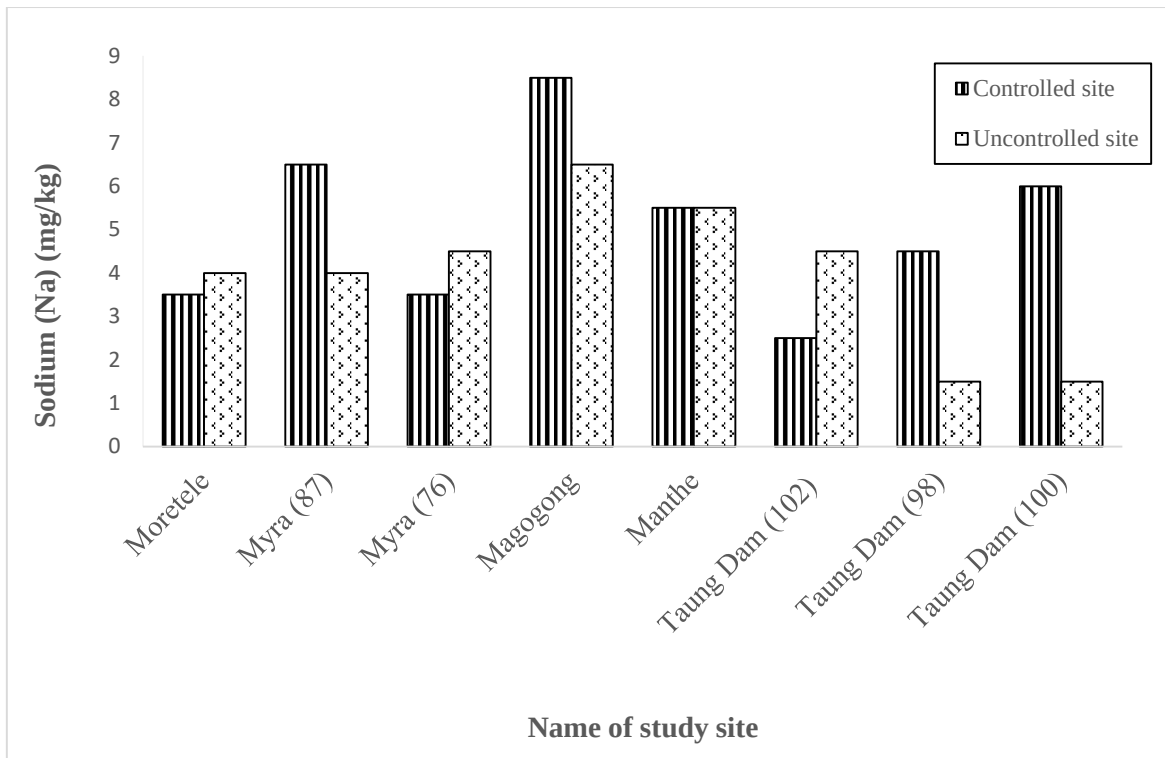


Figure 6.11: Mean sodium (Na) content of soil in the different study sites.

Molatlhegi (2008) conducted a similar study in ten study sites in the former Molopo Magisterial District of the North West Province and indicated a positive relationship between soil pH and soil Na availability. The results observed in the Taung area are in agreements with those results obtained, with regard to the high soil Na content (Figure 6.11) and soil pH (Table 6.1). Mogodi (2009) stated that, soil Na content did not indicate any significant differences between encroached and non-encroached sites.

Although Na is not essential for plant growth, it is normally higher in the vicinity of woody plants (Archer, 1995; Hagos & Smit, 2005). This corresponds with the findings in the controlled sites in Magogong, Moretele, Myra (87), Myra (76) and Taung Dam (100) and in the Taung Dam (102) and Taung Dam (98) uncontrolled sites (Figure 6.11).

Soils with a high concentration of Na have a significant decrease in K concentration and a decrease in the K:Na ratio (Maathuis & Amthman, 1999; Shabala *et al.*, 2006). The soils in this study also recorded high K concentrations (Figure 6.9) and lower Na concentrations (Figure 6.11). The results in this study are thus in agreement with Shabala *et al.* (2006). Therefore, K:Na ratio was negatively affected with respect to Na (lower concentration) as opposed to K (higher concentration) (Figure 6.9 and Figure 6.11).

The highest recorded soil pH (8.17) was observed in the Moretele controlled site (Figure 6.1). The high soil pH in this site did have an effect on the available K in the study site (Figure 6.9).

6.12 Available organic phosphorus (P) content

There were limited soil phosphorus (P) in both the controlled and uncontrolled sites with the only exception being the Moretele uncontrolled site and the Magogong controlled and uncontrolled sites (Figure 6.12). The highest available soil P content was observed in soil sampled at the Taung Dam (102) controlled site and was measured at 36.4 mg/kg, while no soil P was recorded in the other sites (Figure 6.12). The mean soil P in the Magogong controlled and uncontrolled sites was recorded at 3.8 mg/kg and 1.4 mg/kg respectively. The mean available soil P in the Moretele uncontrolled site, was recorded at 3.8 mg/kg (Figure 6.12).

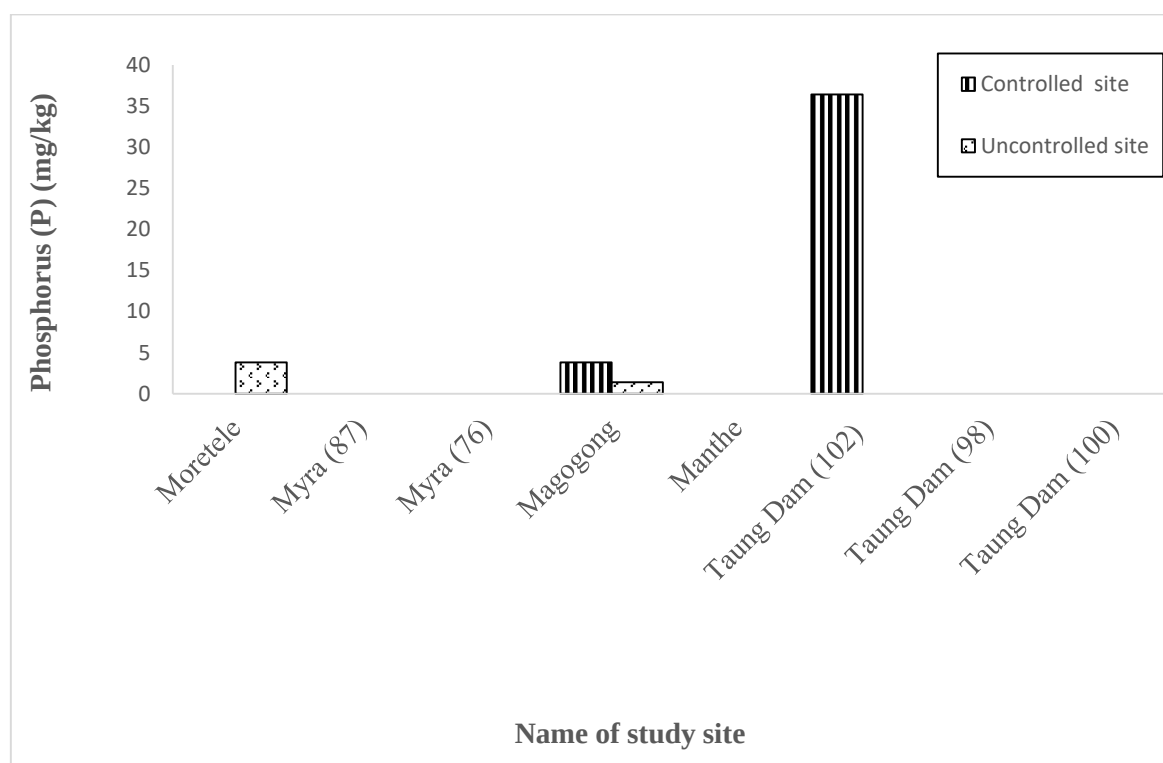


Figure 6.12: Mean phosphorus (P) content of soil in the study area.

According to Fitzpatrick (1990) and Foth (1990), deficiency of P is caused by increased Ca in the soil. Molatlhegi (2008) also reported, high K content in soil samples having high Ca content, thus indicating a positive relationship between soil Ca and soil K content. The low soil Ca content in the Moretele uncontrolled site (Figure 6.5) could account for the high

accumulation of soil P in this site (Figure 6.12). This is in contrast with the results reported by Molatlhegi (2008) who found higher soil P in non-encroached sites compared to the encroached sites in the Molopo area. A positive relationship between soil Ca and P content were, however, found in this study (Figure 6.5 and Figure 6.12). The soil in the Magogong controlled and uncontrolled sites both expressed high Ca content and were the highest of all the sites (Figure 6.5 and Figure 6.12). These results are in contrast with the results reported by Fitzpatrick (1990) and Foth (1990) but are in agreement with the results reported by Molatlhegi (2008).

According to Moore *et al.* (1998), soils with a neutral pH are P deficient. The soil pH in Moretele uncontrolled site and Magogong (controlled and uncontrolled) were recorded at 6.97, 7.17 and 6.80 respectively (Table 6.1) and thus fall within the neutral soil pH range, which can account for the low soil P content observed (Figure 6.12). Yerokun (2008) stated that, soil P is strongly associated with soil clay, Al_{ox} and Fe_{ox} , which are contributing to immobilisation of P (Yerokun, 2008).

Angassa *et al.* (2012) reported that, the available P content of the soil in the Borana rangelands in southern Ethiopia was very low and are thus in agreement with the findings of this study (Figure 6.12). Agbenin and Tiessen (1995) and Buresh *et al.* (1997) have indicated that organic matter production in semi-arid rangelands is low due to climate change and lower soil organic P rates may be expected. The low SOC in the controlled and uncontrolled sites (Figure 6.25) could also account for the low or unavailability of soil organic P contents in the different sites (Figure 6.12). Ibia and Udo (1993) reported soil P values of between 5 mg/kg and 434 mg/kg for savanna soils in Eastern Nigeria. Other studies, however, revealed soil P contents lower than 72 mg/kg (Shodeke *et al.*, 2006; Raji & Ongunwale, 2006).

Plants growing in a P-deficient soil allocate a greater proportion of assimilates to root growth and tend to have fine roots and especially root hairs (Gahoonia *et al.*, 2001; Nigussie *et al.*, 2003). Plants, having such a root system, are effective in scavenging P from the soil environment because of a large surface area of contact with the soil (Rengel & Marschner, 2005). This is especially true for *Vachellia* species present in both the controlled and the uncontrolled sites in this study. *Vachellia* species have a deep lateral root system and have an advantage over grasses to access available soil P.

Hagos and Smit (2005) reported that, the P content of the soil in a site encroached by *S. mellifera* was limited and that the highest concentration of soil P was recorded around the stem base and declined linearly towards the open area. This corresponds with the high bush densities of *S. mellifera* reported in the Taung Dam (102) controlled site (Chapter 5, Figure 5.11) which revealed the highest soil P content (Figure 6.). With regard to Taung Dam (102) controlled site, the deficiency of soil available P can be attributed to low SON, which was observed to be lower in the uncontrolled site as compared to the controlled site.

6.13 Cation-Exchangeable Capacity (CEC)

The highest soil CEC was recorded in the Taung Dam (98) controlled site at 51.03 cmol(+)/kg (Figure 6.13). The soil sampled in the Magogong controlled site had a CEC value of 24.62 mg/kg and the soil in the uncontrolled sites had CEC values of 28.13 cmol(+)/kg (Figure 6.13). The lowest soil CEC was measured in the Moretele and the Taung Dam (98) uncontrolled sites at 9.02 cmol(+)/kg and 8.66 cmol(+)/kg respectively (Figure 6.13).

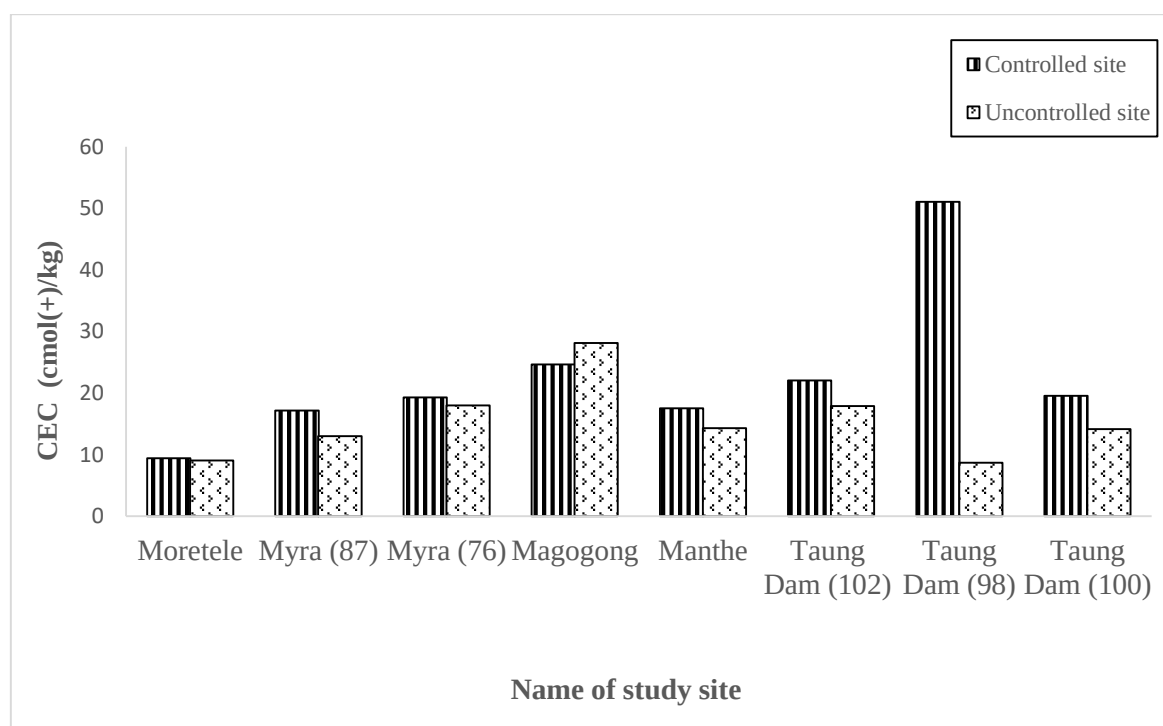


Figure 6.13: Cation-Exchange-Capacity (CEC) of the soils in the Taung study area.

EC is defined as the ability of a material to transmit an electrical current between the different elements in the soil (Doerge *et al.*, 2004). Soil EC is a measurement that correlates

to soil properties affecting crop productivity, including soil texture, CEC, drainage conditions, organic matter level, subsoil characteristics, depth of clay pans and Ca^{2+} and Mg^{2+} values (Doerge *et al.*, 2004). Soil EC with field variation had been related to specific properties that affect crop yield, such as topsoil depth, pH, salinity, water holding capacity and CEC. There is thus a positive relationship between soil CEC and EC, which implies that the higher the CEC, the higher the EC will be (Moore & Walcott, 2001; Doerge *et al.*, 2004). Soil EC has also been closely correlated with soil texture and, therefore, soil pH (Hartsock *et al.*, 2000; Mogodi, 2009). Angassa *et al.* (2012) indicated that, low soil pH, high soil EC and CEC and also high CaCO_3 and certain nutrients such as K^+ , Ca^{2+} , Mg^{2+} , N, OC and P are generally associated with bush encroachment.

The CEC of the soil is a measurement of the total negative charges on its cation-retaining properties (Archer, 1985; Foth & Ellis, 1988; Molatlhegi, 2008; Mogodi, 2009). Based on the strength of positive and negative charges in the soil, one can produce a sound soil nutrient treatment plan for growing vegetation (Harris, 2010). According to Fitzpatrick (1990), the negative charges in the soil are directly related to soil pH. Higher soil CEC is a likely indication that the fertility will be higher. The CEC is, therefore, a good indicator of soil quality (Cooper & Regents, 2006; Mogodi, 2009). According to Harris (2010), a soil pH increase lead to the CEC increase in the soil. Sangha *et al.* (2005) reported that soil pH increased in controlled sites as compared to uncontrolled sites and also increased soil exchangeable Ca, Mg and Na and CEC.

Both the Magogong survey sites revealed neutral soil pH values (Figure 6.1) and are closely related to soil CEC (Figure 6.13). A positive relationship between soil pH and CEC was observed (Table 6.1 and Figure 6.13).

6.14 Electrical Conductivity (EC)

The highest soil EC was observed in the Magogong controlled and uncontrolled sites which were recorded at 44 mS/m and 31 mS/m respectively, followed by soil in Moretele and Taung Dam (102) uncontrolled sites, which measured readings of 28 mS/m and 22 mS/m respectively (Figure 6.14). Soil in Myra (87), Manthe and Taung Dam (100) uncontrolled sites, all recorded the similar EC values of 10 mS/m (Figure 6.14). Soil in

Taung Dam (98) controlled and uncontrolled sites had the lowest EC readings of 8 mS/m (Figure 6.14).

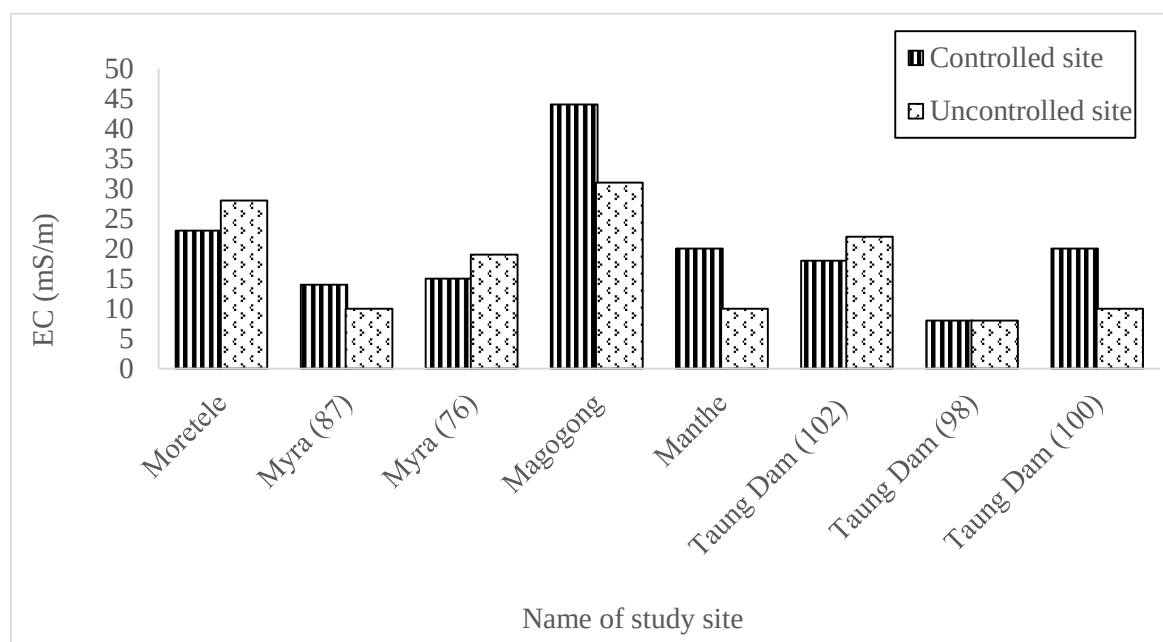


Figure 6.14: Mean electrical conductivity (EC) of soils in the different study sites in the Taung area.

Although, the soil sampled in the Magogong controlled site, had a higher EC value as compared to the corresponding uncontrolled site, the corresponding CEC value was, however, lower. The EC value was expected to be higher in the Magogong uncontrolled site as compared to the controlled site as a result of a high soil CEC. This, however, corresponds with the results reported by Molatlhegi (2008) and Mogodi (2009) who observed that, although some bush encroached sites had a higher soil EC these soils revealed a lower CEC when compared to a non-encroached benchmark site. The results observed in soil in the Magogong uncontrolled site indicate a higher CEC value and a lower EC value (Figure 6.13 and Figure 6.14). The soil CEC recorded in the Taung Dam (98) controlled site was 51.03 cmol(+)/kg compared to the soil in the uncontrolled site which measured 8.66 cmol(+)/kg (Figure 6.13). There was, however, no change in the soil EC value in both sites (8 mS/m) (Figure 6.14).

CEC is important for maintaining adequate quantities of plant available Ca, Mg and K in soils. Other cations include Al^{3+} (when $\text{pH} < 5.5$), Na^+ , and H^+ (Spectrum Agronomic Library, 2015). Soil particles and organic matter have negative charges on their surfaces

which can be adsorbed by mineral cations and hence, soil exchangeable cations are highly 'dependent' upon soil texture and organic matter content (Hepper *et al.*, 2006; Gogo & Pearce, 2009).

Obi *et al.* (2010) reported that descriptive statistics indicated that the distribution of soil particles sizes, exchangeable Ca, Mg and K and their ratios were not "similar". It had been reported that variability of soil properties could be as a result of intrinsic (factors of soil formation), extrinsic (management and land use) variation or experimental error (Shukla *et al.*, 2004; Botros *et al.*, 2009; Souza *et al.*, 2009). Enloe *et al.* (2006) stated that, Ca dominate the exchange site with Mg, K, NH_3 , while Na has lower concentrations. Angassa *et al.* (2012) reported a significantly positive correlation between the effects of land use changes and certain nutrients in soil such as K^+ , Ca^{2+} , Mg^{2+} and SON.

6.15 Percentage base saturation of soils in the Taung area

Base saturation is defined as the proportion of the CEC occupied by exchangeable bases (Fageria & Baliger, 2008) and is commonly used as a criterion for liming recommendations (Fageria & Baligar, 2008). Variable results on soil base saturation in different study sites were recorded (Figure 6.15). The highest base saturation was recorded in soil sampled in the Moretele controlled (104.84%) site, followed by the Magogong controlled (101.46%) site (Figure 6.15). Soils in the uncontrolled study sites of Myra (87), Myra (76), Taung Dam (98) and Taung Dam (100) also showed higher base saturation as compared to the respective controlled sites (Figure 6.15). The latter revealed a soil base saturation that was in the range of 50.33% and 57.26% respectively (Figure 6.15). Soils in Myra (87) uncontrolled site and all three Taung Dam controlled sites recorded a base saturation of less than 50% (Figure 6.15). The soil pH values recorded in the Taung Dam controlled sites (102, 98 and 100) all fall within the range of 5-6 (Figure 6.1), indicating that these soils are moderately to highly acidic. Obi *et al.* (2010) observed that K had a positive correlation with clay soil, while sandy soil correlates with Ca/Mg and Mg/K increases.

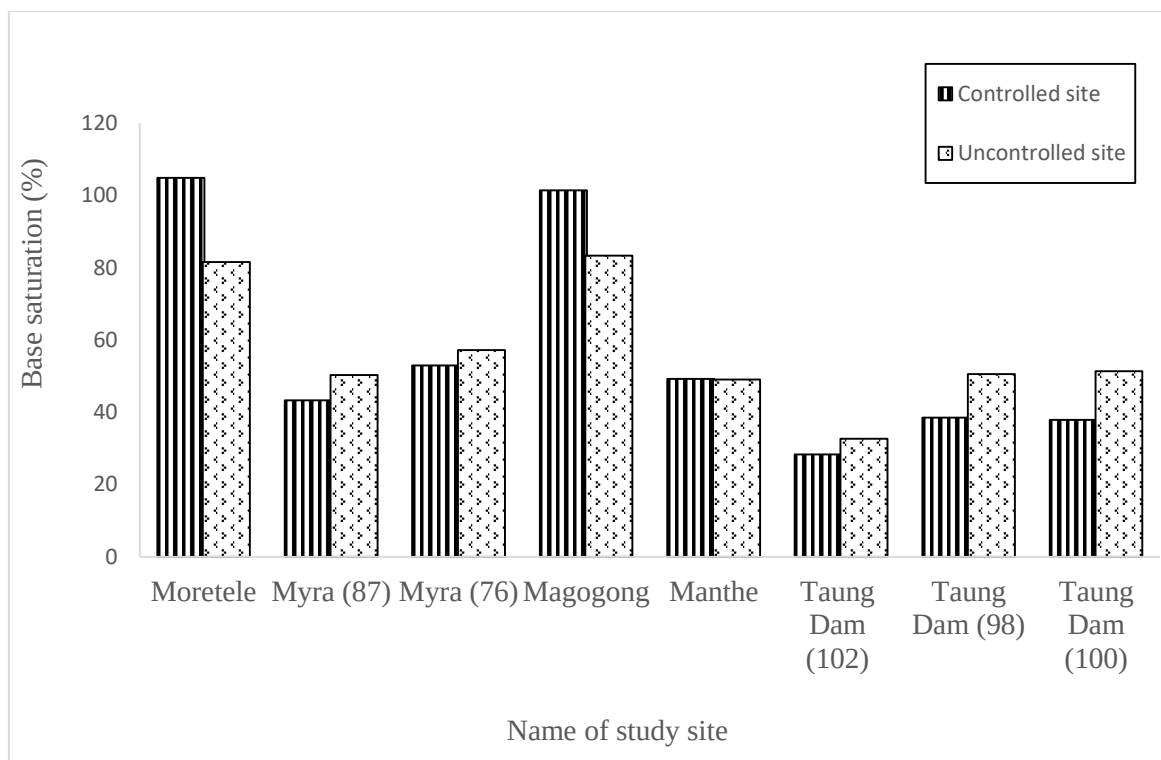


Figure 6.15: Percentage base saturation of soil in the Taung area.

For crop production, base saturation levels in soil may be grouped into extremely low (lower than 25%), low (25-50%), medium (50-75%) and high (>75%) (Fageria & Gheyi, 1999). Extremely low and low base saturation refers to a predominance of adsorbed Hydrogen (H) and Aluminium (Al) on the exchange (Fageria & Baligar, 2008). According to FAO (1999), soils with a base saturation of greater than 50% are regarded as fertile soils while soils with less than 50% were regarded as non-fertile soils. This implies that, according to the FAO (1999), soils in the Taung sites can be regarded as fertile.

There is a positive relationship that between soil pH (Table 6.1) and base saturation (Figure 6.15), but a negative relationship exists between soil CEC (Figure 6.9) and base saturation of soil (Figure 6.15). The lower CEC values could be attributed to the accumulation of Al^{+} and H^{+} ions saturated at the exchange sites (Fageria & Baligar, 2008). This was true for the Moretele controlled and uncontrolled sites and the Magogong controlled and uncontrolled sites with regard to soil pH and CEC (Table 6.1 and Figure 6.13). The results obtained in Figure 6.15 indicate that the soil in Moretele and Magogong are the most fertile compared soil in other sites.

6.16 Conclusion

Soil pH was higher in the uncontrolled sites as compared to the controlled sites, except for soils in Moretele, Magogong and Taung Dam (102) (Figure 6.1). The SOC was higher in the uncontrolled sites, as compared to the controlled sites, with the exception of the soils in Manthe and Taung Dam (98) and Taung Dam (100) controlled sites (Figure 6.25). There was no SON deposits recorded, except in the Myra (76) uncontrolled site, Magogong controlled and uncontrolled sites and also in soil of the Taung Dam (102) sites (Figure 6.3). Due to the lack of N, the only measurable C:N ratio was in the Myra (76) uncontrolled site and Magogong and Taung Dam (102) sites (Figure 6.4). The percentage base saturation of soils was generally higher in the uncontrolled, as compared to the controlled sites, except for sites in Moretele and Magogong (Figure 6.15). There was virtually no available soil phosphorus (P) in both the controlled and uncontrolled sites with the only exception being the Moretele uncontrolled site and the Magogong survey sites (Figure 6.12).

Soil Ca content was higher in controlled sites as compared to the uncontrolled sites, except for soil in Taung Dam (100) and Taung Dam (102) (Figure 6.5). Mg content in soil was higher in the controlled sites as compared to the uncontrolled sites, except for Myra (76) and Magogong sites (Figure 6.7), while K content in soil was higher in the controlled sites as compared to the uncontrolled sites, excluding the Magogong uncontrolled site soil which recorded a slightly higher K than in the controlled site (Figure 6.9). The results indicated that the mean K^+ content was higher in the controlled compared to the uncontrolled sites, excluding in the Magogong sites (Figure 6.10). The soil Na concentrations in the Taung study area are generally high in the controlled sites as compared to the uncontrolled sites, excluding for Moretele, Myra (76) and Taung Dam (102) study sites (Figure 6.11).

Soil CEC and EC was generally higher in the controlled sites as opposed to the uncontrolled sites indicating a higher soil fertility.

CHAPTER 7

RESULTS AND DISCUSSION ON THE SOCIAL SURVEYS DONE IN THE TAUNG AREA

A total of 79 respondents were interviewed in five villages within the Taung region near the survey areas. The rural villages visited included Pudumoe, Cokonyane, Maphoitsile, Pitsong, Taung and Manthe (Chapter 3, Figure 3.3). The interviewed individuals (referring to respondents in the text) were farmers living in the villages within the Taung region. The aim of the study was to investigate how bush encroachment and bush clearing methods affected the pastoral activities of the farmers in the area and to what extent it influenced their social behaviour and grazing activities of the livestock.

7.1 Personal information of respondents

7.1.1 Gender

The gender distribution in this study constitutes of 51% males and 49% female respondents (Figure 7.1a).

Sive (2016) conducted a study in the Sheshegu communal lands of the Eastern Cape Province of South Africa and reported that there were more female respondents (62%) as males (38%). Gender distribution is, therefore, not restricted to area and changes from one geographical to the next may appear. Admasu *et al.* (2010) found that, in southern Ethiopia, males were more interested in farming compared to females. Similar results are reported in the Dr. Ruth S. Mompati District of the North-West province of South Africa (Chapter 3, Section 3.2) where the female population (239,097) constitutes 51.55% of the total population (463,815), while males constitute 48.45% (NWP Integrated Development Plan, 2016). This is however, in contrast with the results reported in Figure 7.1(a).

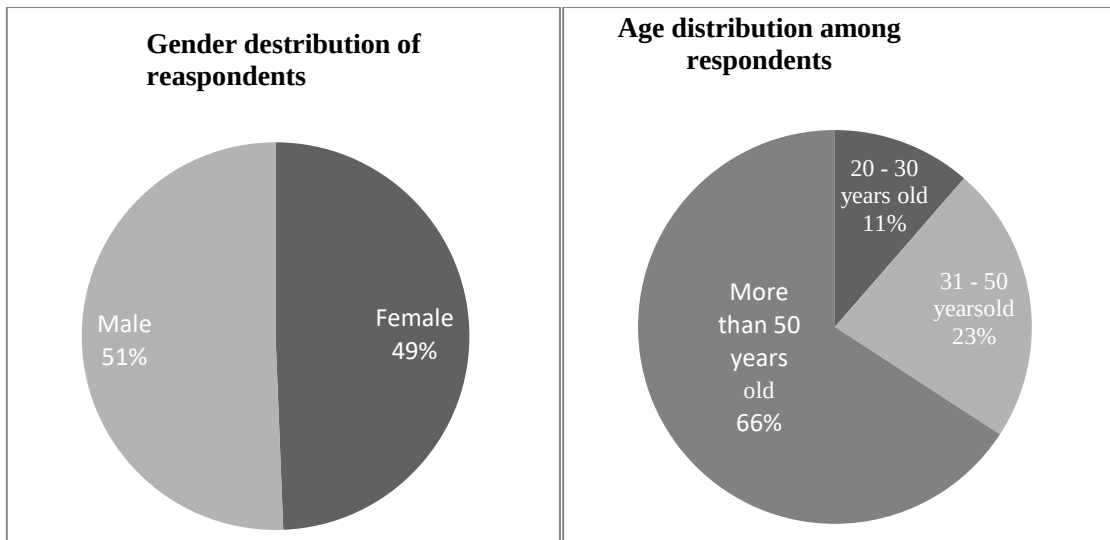
7.1.2 Marriage status

Only 28% of the respondents were married (Figure 7.1 (c)) of which were between 30-69 years old. The number of individual respondents who had children (45%) is higher than the recorded 28% of married couples (Figure 7.1(c)). Respondents not married but having children indicated that, most are dependent on close family members and/or funding from the South African Social Services Agency (SASSA) and self-employment (small businesses) for their income to support the children.

In Taung study site, generally only 3.8% of female teenagers are married or living together with a male partner, compared to 54.0% and 59.8% of woman in the age groups 30-39 years and 40-49 years respectively who are married (Gender Statistics for South Africa Report, 2011). This corresponds to the results observed in Figure 7.1(a), Figure 7.1(b) and Figure 7.1 (c) found in this study. Access to childcare in communal areas is substantially lower than in commercial areas (more developed areas) at 21.0%, which means that it is less likely for children to be attending childcare facilities in the rural areas (Gender Statistics for South Africa Report, 2011).

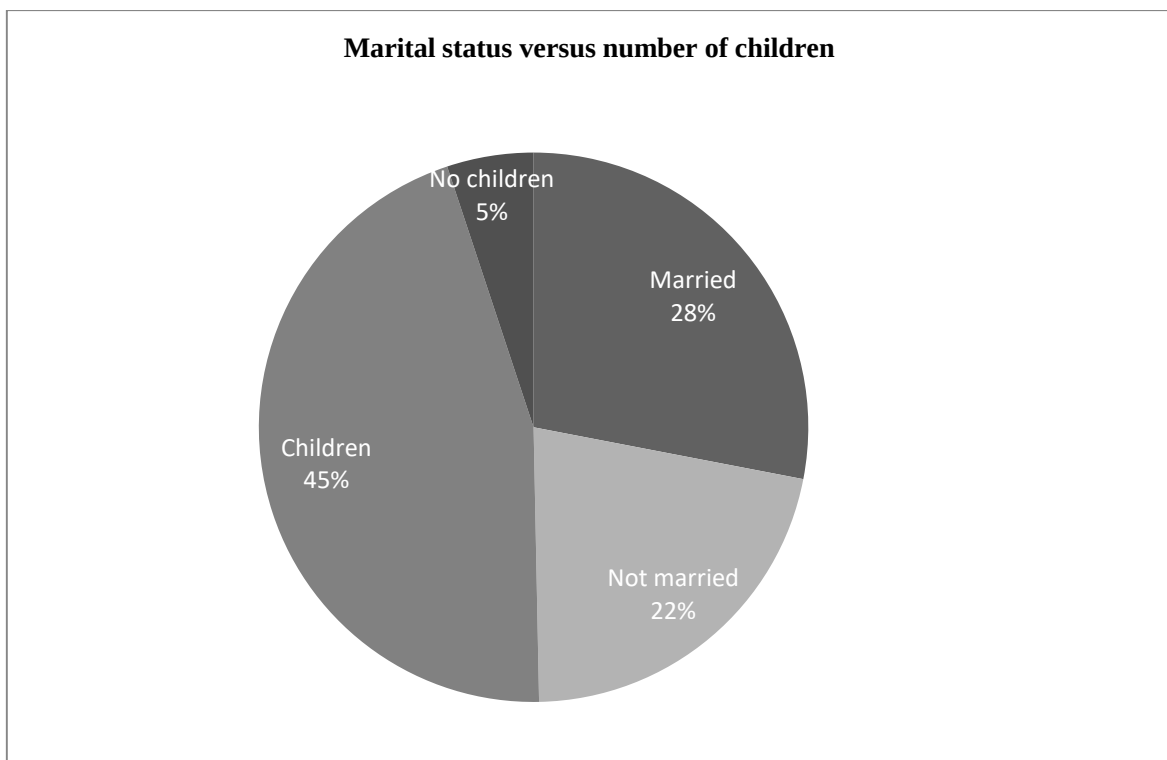
7.1.3 Age distribution

Figure 7.1(b) indicates that, 66% of the respondents were 50 years and older, followed by 23% of the respondents falling within the 31-50 years age group and only 11% being in the range of 20-30 years of age (Figure 7.1(b)). In the Taung area, it is mostly the older respondents who are interested in livestock farming and, therefore, own more livestock especially cattle. The younger respondents interviewed indicates that they only participate in livestock farming during school holidays when they look after the cattle.



(a)

(b)



(c)

Figure 7.1: (a) Gender distribution of respondents; (b) age distribution of respondents; (c) marital status versus number of children of respondents.

Sive (2016) reported that, in the Sheshegu communal rangelands in the Eastern Cape Province of South Africa, 40% of the respondents were between the ages of 34 and 45 years, while 38% ranged from 56 to 75 years of age. According to Lesilo (2011), youth in communal rangelands of the Eastern Cape Province of South Africa areas show low

interest in livestock production but are more engaged in other small businesses, such as education and private enterprises and others.

7.1.4 Education status through official schooling

Figure 7.2(a) indicates that 29% of the respondents had attained a primary level of education, followed by 27% secondary level and 23% of respondents who had never attended any education through schooling. In a study conducted by Sive (2016), most of the communal farmers had some form of formal education (official schooling), with only 8% of them being illiterate. The results of this study found that respondents with a tertiary qualification owned fewer cattle as compared to those who attained only a primary or secondary qualification Figure 7.2(d).

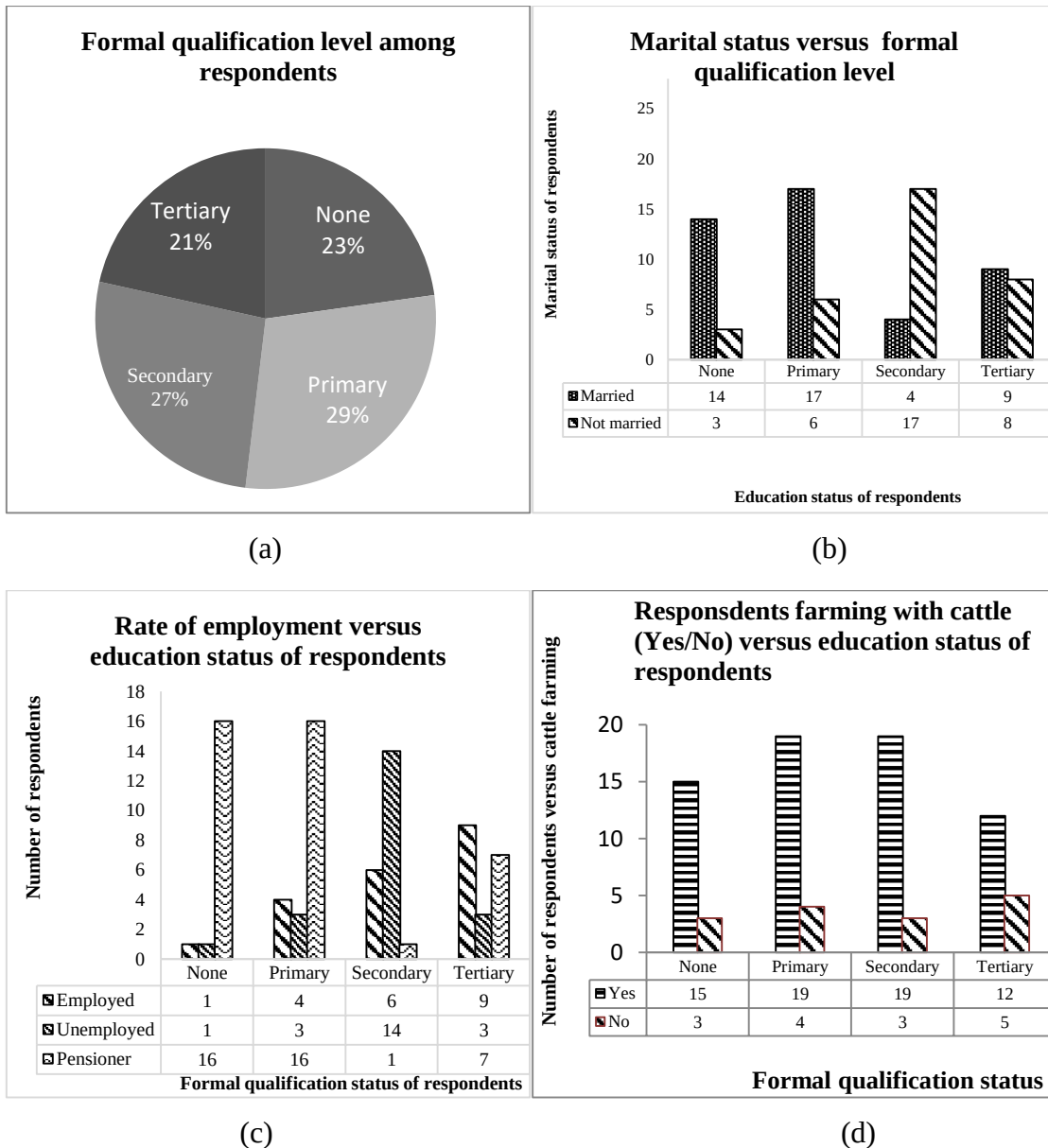


Figure 7.2: (a) Formal qualification level among respondents; (b) marital status versus level of education; (c) rate of employment versus education status of respondents; (d) respondents keeping cattle versus education status.

Education status differs from one communal area to the next (Sive, 2016). According to studies conducted by Gwelo (2012) in the Kwezana and Dikidikana communal rangelands located in the Eastern Cape Province, South Africa and Sive (2016) formal qualification level in communal areas in the Eastern Cape Province of South Africa are gradually improving which may lead to less poverty through farming in communal societies. This statement is in agreement with the results of this study which indicate that the number of respondents in possession of a secondary qualification was higher than the respondents with

no form of education (Figure 7.2(a)). Ellis and Freeman (2005) stated that there is an inverse relationship between poverty and formal qualification levels. This corresponds with the results observed in Figure 7.2(c), where the majority of respondents with some sort of education are employed and are less poor (Figure 7.2(a) and 7.2(c)). According to the Gender Statistics for South Africa Report (2011), the high unemployment number in South Africa is generally as a result of poor level of education, especially among the rural residents. According to Torr (2007) there is a clear relationship between level of education and marital status in that a lower education status may resort to having more children as compared to those having a higher qualification (schooling) status. The latter children would also be expected to provide an 'income safety net' to the family as they grow up (Torr, 2007; Gender Statistics for South Africa Report, 2011). According to Beck and Nesmith (2001), the Eastern Cape Province mainly consists of communally managed areas, with livestock production playing a vital role in the reduction of poverty. This is similar to the Taung area where this study is carried out. Tekana and Oladele (2011) indicated that, 33% of pension fund is the greatest contributor of the economy in the Taung area. According to Gender Statistics for South Africa (2011), only 40% of the population in the area, are formally employed. This is in comparison with the results reported in Figure 7.2 (c). According to the Gender Statistics for South Africa (2011) report and Acha (2014), the majority of the households in the Taung area rely on external economic activities such as state grants (for example, child grants and pension funds).

According to Figure 7.2(d) most of the respondents in possession of either a primary or secondary school level of education keep more livestock, such as sheep, goats and cattle. Livestock types in the Taung area constitutes mainly of cows, sheep, goats, donkeys and horses (Figure 7.3(a)). The results in this study are similar to Sive (2016) who indicated that in the Sheshegu communal rangelands, Eastern Cape Province of South Africa, farmers with secondary school qualifications owned large numbers of goats, cattle and sheep. Moyo *et al.* (2008) also reported that literacy level is an important determinant of livestock production, particularly on accepting new interventions of rangeland management. With regard to school education, a large proportion of the population (75%) have completed at least some secondary school level of education. Currently only 31% have completed school and only 13% have tertiary education. The low school education percentages have a negative impact on human development capacity in the area (Gender Statistics for South Africa,

2011; Acha, 2014). According to the NWP Integrated Development Plan (2015) there has been a general increase in the number of pupils (aged 5-24 years) from 1996 to 2011 that are enrolled at schools in all municipalities in the Taung district, with the highest being in the Greater Tang local municipality (77%). Although there was an increase in the school enrolment, the NWP Integrated Development Plan (2015) indicated that 38.8% unemployment rate occurs in the Dr. Ruth S. Mompati municipality. The highest unemployment rate is recorded at 50% in the Taung area. Majority of the residents in the Taung area have a historical disadvantaged background and poverty level in the area stands 49.8% (Acha, 2014). This is in comparison with the results reported in Figure 7.2 (a &c) of this study.

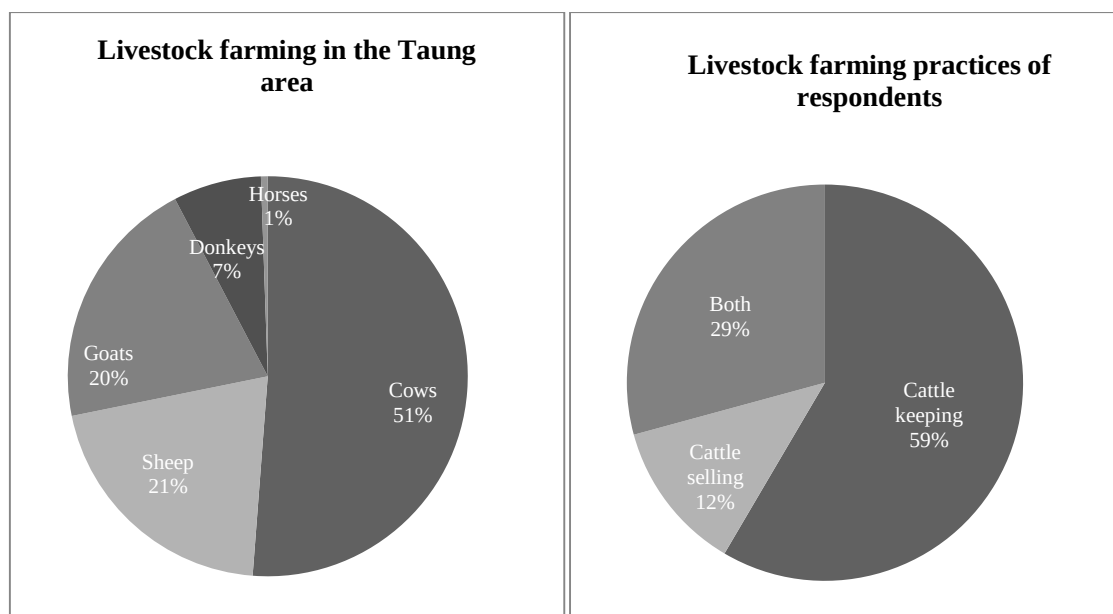
7.2 Farming practices in the Taung area

7.2.1 General livestock farming in the area

Most of the communal farmers (59%) in the Taung area are cattle farmers, of which only 29% occasionally sell their cattle (Figure 7.3(b)), as 12% of the farmers prefer selling their cattle rather than keeping it. For the purpose of this study, high stock numbers refer to the number of cattle beyond 50 LSU²/ha and a low stock number refers to less than 50 LSU/ha. Respondents with high numbers of livestock were more interested in selling their cattle to generate household income (e.g. paying for school fees, food and clothing) as compared to farmers who only have a few livestock (cattle, sheep and/or goats) (Figure 7.3 (b)).

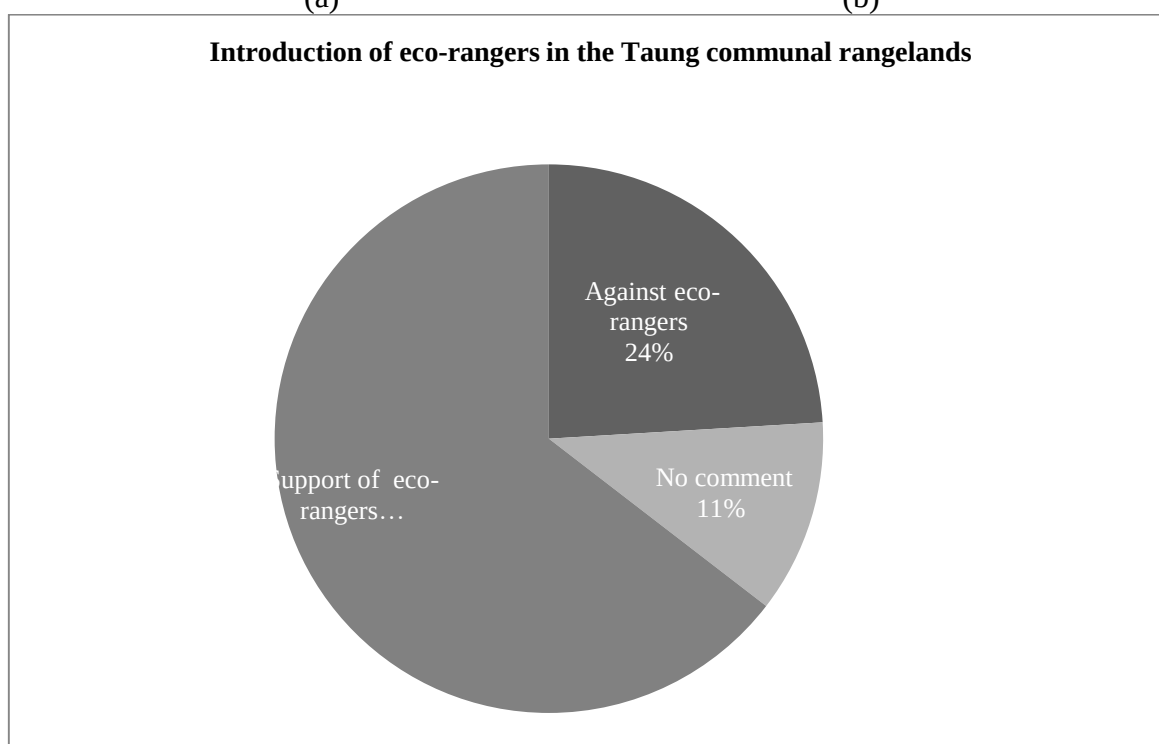
Figure 7.3 (c) reveals that, 65% of the respondents prefer the introduction of eco-rangers within their grazing areas, while 24% indicated that they are not in support of this strategy. Only 11% of the smallholder farmers had no comment on the introduction of eco-rangers in the grazing lands (Figure 7.3 (c)).

² LSU: The equivalent of one head of cattle with a body weight of 450 kg and gaining 500 g per day (Meissner *et al.*, 1983).



(a)

(b)



(c)

Figure 7.3: (a) Livestock farming practices in the Taung area; (b) livestock farming practices of the respondents; (c) introduction of eco-rangers in the Taung communal rangelands.

Shackleton *et al.* (2001) and Twine (2013) stated that poor households tend to rely on a greater range of benefits from their livestock, such as cash to buy food and other products (Ainslie, 2002; Ainslie, 2005; Mapiye *et al.*, 2009), as well as direct products from livestock,

including milk, meat and hides (Schakleton *et al.*, 2005). This is in agreement with findings from this study where poor communal farmers prefer selling cattle products such as milk, as means of food security, provision of household income, school fees, to buy agricultural goods and apparatus, as well as sheep and goats to the community Musemwa *et al.* (2010) (Figure 7.3(b)). They would also use donkeys and horses for the collection of firewood and as draught animals. Livestock farming practices contributes 38.1% of the Taung economy, poultry 41.9%, vegetables stand at 6.0% and other crops 4.3%. In the Taung area, animals only contribute 81.3%, mixed farming at 12.4%, crops production contributed 6.3% of the local economy (Gender Statistics for South Africa Report, 2011).

The respondents participating in this study indicated that they do not sell cattle at auctions because they are under the impression that the prices of cattle at auctions are lower than when selling to local residents privately. Auction prices are set at R4500 to R 6000 for a cow, but that the price for selling to local residents is not fixed and may exceed the auction price. Barret *et al.* (2003) & Musemwa *et al.* (2010), who conducted social studies in the Eastern Cape Province of South Africa, however, stated that more than 30% of the respondents preferred selling cattle through auctions.

7.2.1 Eco-rangers

Animals in the Taung area are not kraaled on a daily basis, which makes it “difficult” for communal herders to restrict animal movement or practice good veld management strategies. This has contributed to the continuous selective grazing, which ultimately has led to bush encroachment, thus, leading to a reduction in the grass layer (Chapter 5, Section 5.1 and 5.2) and changes in grass species composition and cover (Chapter 5, Section 5.2). In order to control veld disturbances such as overgrazing and also to reduce stock theft, it is essential that eco-rangers be introduced in Taung area. An eco-ranger is referred to as a person assigned to take care of grazing lands and managing the grazing strategies of the herders taking care of the livestock. Such an eco-ranger is assigned in agreement between the local authorities, the herdsmen, and the local community members. The eco-rangers are remunerated by the WfW programme in the Taung area.

The respondents in favor of the introduction of eco-rangers elaborated that during 1971, there were local residents who volunteered and served as eco-rangers in the grazing lands, to work hand-in-hand with the community and tribal authority. They indicated that members of the community were given terms and conditions by the chief (tribal authority) and herdsmen on how to utilize and divide the grazing areas and should any member of the community bypass these laws and regulation, they would be forced by the chief to either pay a fine or engage in community service. The respondents indicated that, they were willing to support the idea of eco-rangers in their community, however, this decision should be made by the paramount chief tribal authority in the Taung area in collaboration with the community members.

Different findings were reported by Allsop *et al.* (2007) who stated that, communal farmers in the Namaqualand of the Northern Cape Province control livestock movement through herding wherein herders daily select a certain area for grazing, thereby “resting” other areas. Lesilo (2011) indicated that, in the Amakhuze Tribal Authority in the Eastern Cape, South Africa, high crime rates may arise in areas where animals are not kraaled at night or kept to a certain grazing area by the eco-rangers. Farmers are often “forced” to sell their cattle due to the high crime rates. Sive (2016) also indicated that, animals that are not kraaled are more prone to stock theft. Livestock spending long times in the rangelands could therefore, promote overgrazing in the long-term (Lesilo, 2011; Sive, 2016). Some of the woody branches of the spiny trees have to be left on the bare soils to enrich the soils and reduce grazing pressure through protection.

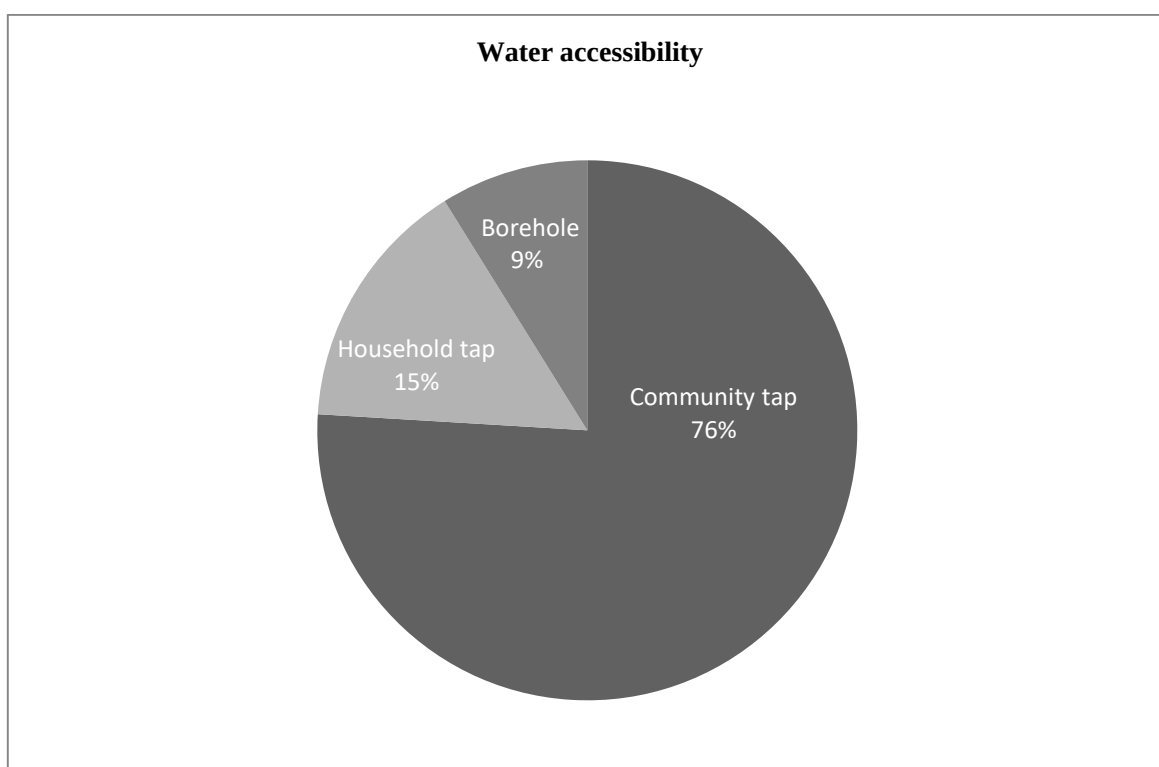
7.3 Water accessibility in Taung

7.3.1 Water used for household purposes

Figure 7.4(a) shows that 76% of the respondents rely on community taps for their household water, while 15% of the respondents collect water from their private household taps and 9% have installed private borehole water systems. The community taps are located some distance away from their homes. Respondents indicated that they have been experiencing water ‘challenges’ which have become more severe since the drought in 2011. Respondents having access to privatized water taps (water taps within their individual residential area)

(Figure 7.4(a)) indicated that, they pay for water at rate of R20.00-R100.00 per month depending on their household needs. The water is supplied by the Sedibeng municipality in Taung.

The high percentage (76%) of the respondents having to rely of the community tap in the Taung area (Figure 7.4) is an indication that the residents have to travel long distances to fetch water for their household need and also for their cattle. This also implies that the livestock in the Taung area also have to walk long distances in order to have access to water, thus, further contributing to soil erosion by hoof action of the animals. Herders in the Taung area indicate that, in 2013 many of their cattle died as a result of drought and the lack of water points in their area.



(a)

Figure 7.4: Water accessibility to the communities in the Taung area.

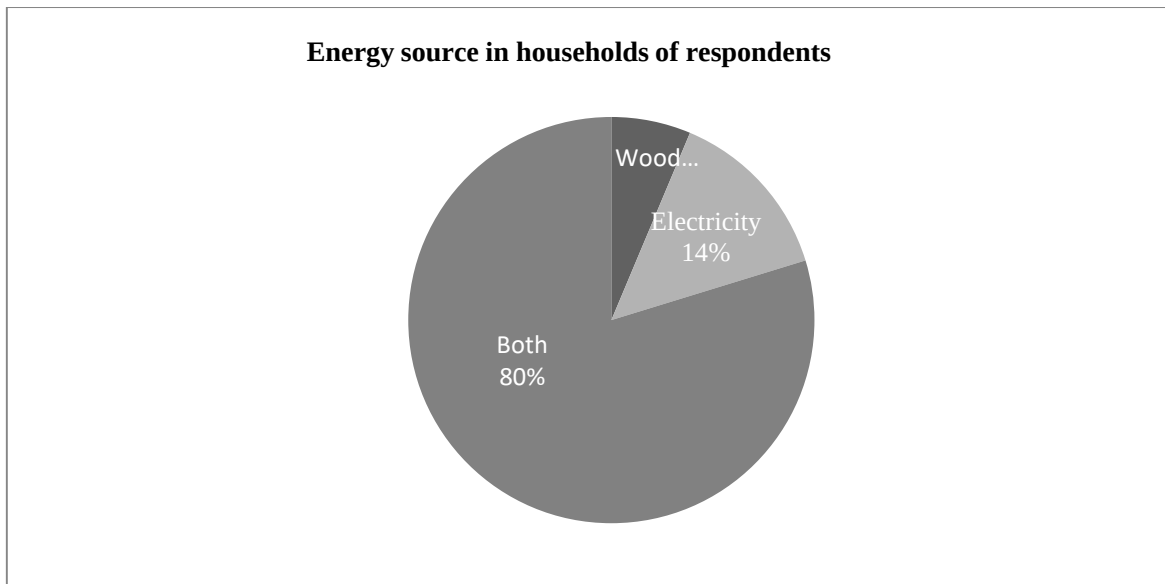
Gender Statistics for South Africa (2011) indicated that, 50.5% of black African households in South Africa were dependent on off-site sources for water. Female members of the household are more responsible for collecting water (Gender Statistics for South Africa Report, 2011). The study revealed that in 1999, 15% of households had to fetch water more

than one kilometre from their water source compared to only 8% in 2011 (Gender Statistics for South Africa Report, 2011). This means that more water resource points were installed closer to the households over the last two years.

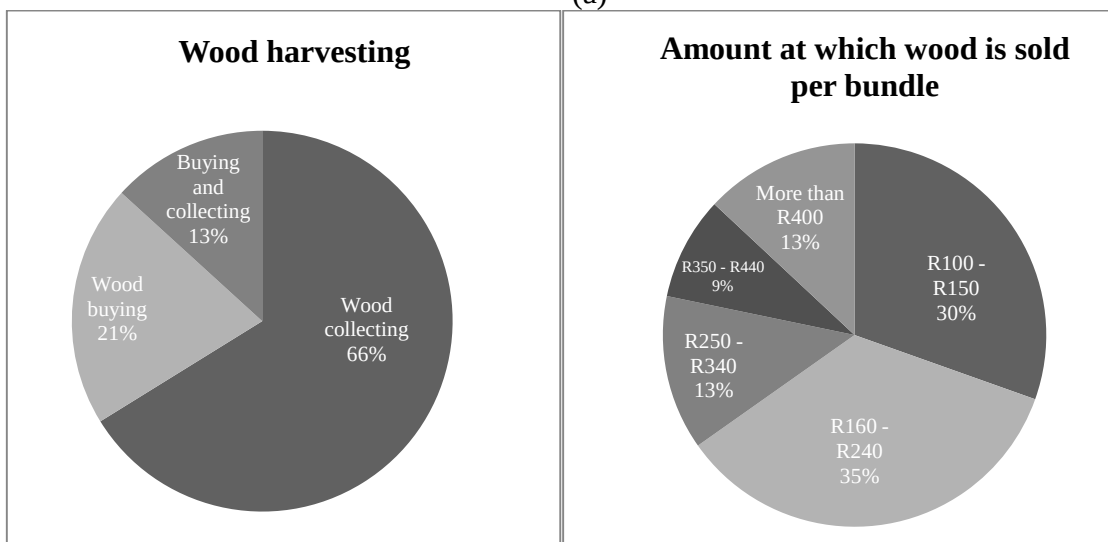
Communal farmers in the Taung area indicated that they had no water points for their cattle and animals have to travel long distances to get access to water. Where kraaling is practiced, cattle owners indicated that they water their animals on a daily or weekly basis, depending on the number of animals. Farmers further indicated that they are faced with huge water shortages for their animals during the winter season (Sive, 2016). Lesilo (2011) stated that communal farmers perceive factors such as lack of water points, decrease of grazing capacity and continuous grazing due to poor grazing management as the major challenges the rangelands.

7.4 Energy usage and wood harvesting in the communal Taung villages

Figure 7.5 (a & b) show that 66% of the respondents participating in this study indicated that they make use of fuel wood as their primary source of energy if they do not have electricity. According to Figure 7.5(a), 80% of the respondents make use of both wood and electricity as their household energy source. The respondents indicated that, they had resorted to using fuel wood instead to of electricity, mainly due to the high costs of electricity. The residents in the study area also use woody plants for the building of kraals.



(a)



(b)

(c)

Figure 7.5: (a) Energy source in households; (b) people harvesting wood; (c) amount in South African Rands at which wood is sold per bundle.

Wood is a secondary resource and unlike livestock, easily accessible and also easy to transport from one village to the next, especially in bush encroached areas. There is also no restriction as to how much wood an individual can collect in this communally managed area. Uncultivated land in communal areas is used more extensively by rural households for purposes other than livestock, but rather for fuelwood, selling as raw material for house construction (especially in the construction of informal settlements) or processed for natural products to generate other income needs (Twine *et al.*, 2003; Shackleton & Shackleton, 2004; Babulo *et al.*, 2008; Shackleton *et al.*, 2008). Woodland products are very important

for the poorest members of the communities, as they provide almost 30% of total income (Frost *et al.*, 2007). According to Cousins (1999) the extra income made from selling wood is referred to as ‘hidden capital’. It, therefore, seems that bush encroachment is only a problem for livestock keepers who are dependent on the grass composition and higher grass biomass cover and production for the grazing animals, but that other people who benefit from wood as a ‘hidden capital’, are not as worried about the bush encroachment problem. On average, 84% of households in North West Province have access to electricity. In the Dr. Ruth S. Mompoti District municipality, 82% of local residents have access to electricity. Of the 82% households with access to electricity within the district, Taung local municipality recorded the highest (NWP Integrated Development Plan, 2015). This is in correspondence with the results reported in Figure 7.5 (b) who indicate that although, other respondents indicated that they make use of electricity, 80% indicate that they use both wood and electricity). The only problem of wood collection is, that the organic material that is required to enrich the soils that have been degraded by bush encroachment, is being removed.

Figure 7.5(c) indicates that 35% of the respondents buy wood at R160.00 to R240.00 per bundle, followed by 30% of the respondents buying the wood for R100.00 to R150.00 per bundle and 13 % at R250.00 to R340.00 per bundle. The residents in the Taung area indicate that this size of the bundle depends on the seller and also the amount at which it is sold (eg. a bundle bought at R100-R200 usually last for a week, whereas, that sold for R340 to more than R400 will usually last for atleast a month or two). It is, however, important to note that the duration (time period) that the bundle of wood would last differs between households as some respondents use both electricity and woody and others rely (depend) only on wood as their household energy source (Figure 7.5 (a)). This opens an opportunity for the use of wood as a bio-fuel product in the Taung area, which would benefit the local residents. The cleared areas by the WfW programme in the Taung communal areas can be used to support the bio-fuel initiative. However, it should be the local residents who are assigned to collect the wood who benefit from such an initiative which might lead to an additional income and contribute to the increase in human well-being of people in the area. This wood is usually collected by women, either in bundles on their heads or in wheelbarrows.

According to Vedeld *et al.* (2007), income from secondary resources such as fuelwood in the form of cash, savings and provisions contributes an average of 22% of total household income in rural areas, however, according to Babulo *et al.* (2008) this figure may be two or three fold higher among poorer households. In the Taung communal areas, most of the respondents reported that they have resorted to selling firewood as a means of generating income because of the low employment rates (Figure 7.1(f)). This finding is supported by Shackleton *et al.* (2005) and Madubansi and Shackleton (2007) who reported that, while 61% of households in Bushbuckridge (Mpumalanga, South Africa) do not own cattle, 94% of the people use firewood. Due to the low formal qualification (schooling) status which resulted in high unemployment rates (Figure 7.2 (a)), it is not surprising those local residents have resorted to selling wood in order to generate some income (Figure 7.5 (c)). This is supported by Mothata (2002), Twine (2005) and Al-Subaiee (2014) who reported that rising unemployment rates have motivated a growing number of commercial harvesters of wood for fuel. Resource scarcity often results in an inflation of local prices, thus enticing increasing numbers of new traders into the market (Twine, 2005).

Wood is collected through cutting down of usable woody species such as *V. tortilis* for fuel. A total number of 21% of the respondents resort to buying wood from their local areas due to the strict rules and regulations of the tribal authorities regarding wood harvesting (Figure 7.5(b)). The local residents indicated that they first have to obtain permission from the tribal authority or the headmen before cutting down any tree for wood products. Figure 7.5(b) also indicates that, 13% of the individuals buy and collect the wood. Due to increasing population numbers, the demand for natural resources for sustaining livelihood systems is also increasing, as wood is still the major source for cooking and heating in average rural household (Al-Subaiee, 2014). It can be in excess of 3 tons of fuel wood per annum (Banks *et al.*, 1996; Shackleton & Shackleton, 2000; Giannecchini, 2001; Twine *et al.*, 2003; Davids *et al.*, 2005).

CHAPTER 8

GENERAL CONCLUSION AND RECOMMENDATIONS

8.1 General conclusion

8.1.1 Effect of woody species control on woody and grass species abundance

The results indicated high coppicing (re-growth) of *Senegalia mellifera*, *Vachellia tortilis*, *V. karroo* and *Tarchonanthus camphoratus* respectively. Although the control methodologies implemented in the sixteen study sites were successful with regard to Magogong, Manthe and Taung Dam (102), the woody encroacher species still continued to be a problem. This was mainly as a result of high coppicing due to no follow-up strategies being implemented in all the survey sites. Due to the communal status of the Taung area, the survey sites were subjected to continued non-selective grazing pressures. This eventually led to failure of the WfW programme to successfully control the encroacher species in sites where control procedures were implemented.

The increase in abundance of encroacher species, especially *V. tortilis* and *S. mellifera*, could also be attributed to factors such as tree size/age at the time of control, a reduction in closed canopies which led to high coppicing rates, the timing of control, herbicide used for control, soil structure and depth as well as reduced competition for water and nutrients due to high grazing pressure. It is also important to take into consideration, global factors such as increased levels of CO₂ concentrations, levels of atmospheric nitrogen concentrations and rainfall regime as well as drought. These factors are also observed to be important determinants in increased woody plant densities. Prolonged herbicide application, immediately after cutting, must be followed up with another round of chemical application. Soil disturbance must be limited as it gives the opportunity for “dormant” seeds to germinate.

There was no significant difference in terms of grass establishment observed in Taung Dam controlled and uncontrolled sites. Both perennial and annual grass species were observed in these survey sites. The implementation of both manual cutting and chemical control methods by the WfW programme contributed to the establishment of palatable grasses, such as *Cynodon hirsutus* and *Digitaria eriantha*. This is an indication that, due to the above-mentioned control methodologies, ecosystem recovery and rehabilitation could be achieved

if proper follow-up strategies were applied. Proper follow-up strategies will eventually promote the growth and establishment of more palatable grass species. However, re-seeding of grasses should also be considered after physical and chemical control is done on the woody component.

8.1.2 Effect of woody species control on soil chemical characteristics

The pH values of the soils sampled in the different study sites in Taung area were above 5.0, therefore, soil pH did not result in root injury to plants. Leguminous trees, such as, *Vachellia* species have deep root systems and are able to access more available water and soil nutrients for their growth and development. Vigorous root growth is important for nutrient and water uptake and consequently for plant productivity and persistence.

The available soil organic carbon content in the soils in the study sites was low. This could have been as a result of overgrazing, soil trampling and soil compaction. Carbon (C) easily escapes the soil with constant removal of grasses. There was unavailable soil organic nitrogen (N) in the soil and this eventually affected the C: N ratio. The soil C: N ratio of most sites could not be determined because of the unavailability of soil N.

Soil calcium (Ca) content was higher in controlled sites when compared to the uncontrolled sites. Soil nutrient accumulation is positively related to soil pH and the high Ca content. The observed Ca accumulation in the Magogong controlled site also led to high soil exchangeable Ca and this was also true for the Taung Dam (98) uncontrolled site which indicated lower soil Ca content and, therefore, lowest soil exchangeable Ca content. The differential soil nutrient availability and soil chemical analysis can be attributed to the effects of bush encroachment on soil in these study sites.

The highest recorded magnesium (Mg) content was recorded in the Magogong uncontrolled site and the lowest was recorded in the Moretele uncontrolled site. The high values obtained for soil Mg in the Magogong and Moretele sites were as a result of high bush densities recorded in these sites. Potassium (K) content was highest in the Magogong uncontrolled site and the lowest in Myra (76) uncontrolled site. Soil available K also contributed to the high soil exchangeable K^+ . These results, therefore, indicated that there was a positive correlation between soil exchangeable K^+ and bush encroachment. With regard to other

study sites in this study, the observed soil exchangeable K^+ was influenced by the soil texture and depth at which the soils were sampled. In this study it was revealed that available soil sodium (Na) concentrations did not have a significant effect on the availability of K concentration in the survey sites.

Sodium concentrations in soil sampled in the Taung study area were generally high in the controlled sites as compared to the uncontrolled sites. The highest recorded soil Na content was in the Taung Dam (97) controlled site and the lowest in the Taung Dam (98) uncontrolled site. The extent of availability of soil Na content in the soil sampled in the different study sites was generally influenced by the range in which the soil pH values of these study sites were recorded. The variation with regard to the differences in exchangeable soil Na^+ was generally low in all survey sites.

There was generally no soil phosphorus (P) in both controlled and uncontrolled sites. The Magogong controlled and uncontrolled study sites had limited P concentrations. The results indicated that, the soil pH of the soils in the survey sites were mostly acidic, excluding that of the Magogong survey site.

Soil chemical properties such as soil pH, exchangeable base cations such as Ca^{2+} , Mg^{2+} , K^+ , Na^+ , CEC, EC and the percentage base saturation indicated that the Magogong and Moretele study sites expressed more soil fertility as compared to the other study sites.

8.1.3 Social surveys

The respondents in this study were mostly male. The results indicated that the majority of the respondents were in position of a primary level education, followed by secondary and tertiary respectively. Most of the respondents were; however, unemployed. This, therefore, posed a challenge to their socio-economic status. The marital status of the respondents was high among the respondents who were 35-67 years of age and declined with a decline in age of the respondents. The number of respondents who had children was higher as compared to the number of married individuals interviewed in this study. This indicated that there is increasing population numbers and, therefore, even higher demand of natural resources such as wood and water to sustain household livelihoods, therefore, enhancing the process of bush encroachment.

In this study, several factors such as low marital status, high unemployment rates and poor education status of the respondents were observed to be the main contributors, inevitably leading to high stocking rates in the Taung area. The existence of a high number of cattle and poor land management led to overgrazing and, therefore bush encroachment. The respondents indicated that, they were aware of the effects of bush encroachment locally and had extensive knowledge of their environment. Although they are aware of environmental constraints such as drought and overgrazing, they felt that they had no choice but to resort to high stocking rates as this plays a major role in contributing toward their income flow. This cash flow is used to pay for other household needs.

8.2 Recommendations

- The prerequisite for woody plant control is to cut the stem at a height of 0.5 m above ground and apply chemical control immediately after cutting.
- Mechanical eradication of woody plants generally produces short-term changes in plant community, which often stimulates coppice shoots, and leads to the eventual persistence and dominance of woody plants. The application of follow-up treatments is, therefore, important to ensure no further growth or coppice of woody plants.
- In survey sites where *Senegalia mellifera* is mostly persistent, it is recommended that appropriate veld fires followed by the introduction of Boer-goats at the seedling stage must be implemented.
- It is recommended that follow-up strategies be implemented in the first five years after control (yearly) to ensure that no coppicing takes place.
- Proper follow up strategies will also eventually lead to the establishment of desired palatable grass species.
- Grazing areas must be regulated by appointed eco-rangers as this will assist in reducing cattle theft and promoting proper veld management practices.
- Local residents must be more knowledgeable on how to combat or control bush encroachment in their area to ensure the success of bush control in their area.

- The woody component is an integrated part of the ecosystem. Attention should be focused on reducing woody plant encroachment, rather than total eradication.

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APPENDIX A

FIXMOVE DATASHEET FOR WOODY COMPONENT LAYER

Site ID:	Date: 2014 /...../	Time
----------	--------------------	------

	Species	Growth form	Control (Y/N)	Control %	Coppicing (1 to 5)	Height Tree (m)	Canopy 1	Canopy 2
1								
2								
3								
4								
5								
6								
7								
8								
9								
10								
11								
12								
13								
14								
15								
16								
17								
18								
19								
20								
21								
22								
23								

APPENDIX B

FIXMOVE DATASHEET FOR GRASS LAYER

Fix site ID		GPS	S	Date	20		
			E				
Nearest Town	KM			Time	:		
Observer			Institution				
Photo nr.	S	E	N	W	Other:		

Herbaceous Survey (1m interval over 100 m transect)

Point to Tuft Distance (DIST) & Disk Pasture Meter (DPM)
--

m	Species	Dist.	DPM	Ecosystem Integrity/Patch	m	Species	Dist.	DPM	Ecosystem Integrity/Patch
1					23				
2					24				
3					25				
4					26				
5					27				
6					28				
7					29				
8					30				
9					31				
10					32				
11					33				
12					34				

APPENDIX C

SOCIAL SURVEYS



Restoration Ecology
Department of Natural Sciences

PERSONAL INFORMATION

1. Name of respondent

Male	
Female	

2. How old are you?

1. 20-30 years	
2. 30-40 years	
3. 40-50 years	
4. Older than 50	

3. Are you a permanent resident in this area?

Yes		No	
-----	--	----	--

4. If yes, how long have you been residing in this area?

1. 1-5 years	
2. 5-10 years	
3. More than 10 years	

5. What is your highest qualification?

Primary		Secondary		Tertiary	
---------	--	-----------	--	----------	--

6. Are you employed?

Yes		No	
-----	--	----	--

7. If yes, what is your career/job description?

.....

.....

8. Do you have children, if yes how many?

1. 1-2	
2. 2-3	
3. More than 3	

9. Where do you get water from?

1. Municipality	
2. Community taps	
3. Rivers or Dams	
4. Other	

10. What do you use the water for?

1. Washing	
2. Cooking	
3. Other	

11. Do you have a vegetable garden?

Yes		No	
-----	--	----	--

12. If yes, please elaborate

.....

.....

13. What mode of transport do you use?

1. Bus	
2. Taxi	
3. Donkey	
4. Other	

14. If donkeys or horses do you use them for transport purposes only, if no please elaborate

.....

.....

ENVIRONMENTAL INFORMATION

1. What was the land used for in the previous years (since you have been resident)?

1. Vegetable farming	
2. Cattle farming	
3. Wood harvesting	
4. Other	

2. Do you have any cattle?

Yes		No	
-----	--	----	--

3. If yes, what type and how many? (Please estimate)

1. Cows	
2. Sheep	
3. Pig	
4. Donkeys	
5. Horses	
6. Other	

4. What do you use them for?

1. Keeping	
2. Selling	
3. Food production (milk or meat)	
4. Transport	
5. Other	

5. Are you of the opinion that you might have a negative impact on the environment?

Yes		No	
-----	--	----	--

6. Are you aware of any negative constraints within the environment, such as the following:

1. Overgrazing	
2. Droughts	
3. Wood harvesting	
4. Sanitation	
5. Other	

7. Has the Working for Water Program on bush clearing improved the current condition in your area?

Yes		No	
-----	--	----	--

8. If yes, please explain

.....

.....

9. If no, please explain

.....

.....

10. What role do you think you can play to make the project successful?

1. Attend and participate in workshop meetings on bush clearing	
2. Maintain proper land management practices in your area	
3. Take care of you area and encourage others to do the same	
4. Other (specify)	

11. What recommendation can you give for the project to be successful?

1. Leave the area as open as it is	
2. Introduce different grazing management practices in the area	
3. Erect fences	
4. Other (please specify)	

12. What input can you give to make the area more attractive upon the successful completion of the project on bush clearing, please explain?

.....

.....

13. What is your opinion in regard to the introduction of eco-rangers in your area? Please elaborate.

.....

.....

Thank you for your cooperation