

**Woody species increase and development on the
Southern end of Disaneng Village, Ratlou Municipality,
North West Province, South Africa**

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DECLARATION

I, Thabo MacDonald Madibo (20953933), hereby declare that the dissertation titled: **Woody species increase and development on the southern end of Disaneng Village, Ratlou Municipality, 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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DEDICATION

First and foremost I would like to give thanks to Modimo Poho Tau ea khale hence without HIM none of this would be possible and to my grandparents, the late Kamohelo and MmaNnuku Madibo, for the words of wisdom and courage.

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LIST OF ACRONYMS

TE: Tree Equivalent

CO₂: Carbon Dioxide

SAWS: South African Weather Service

GIS: Geographic information System

SPOT: Satellite Pour l'Observation de la Terre (French)

SANSA: South African National Space Agency

GCPs: Ground Control Points

CA: *Consumer's Accuracy*

PA: *Producer's Accuracy*

GPS: Global Positioning System

mm: millimetre

m: metre

ha: hectare

ERRMAT: error matrix

KIA: Kappa Index of Agreement

ABSTRACT

Woody species were quantified at a selected study area near Disaneng (Ratlou municipality) in the North West Province (NWP) of South Africa. Woody plant development was also assessed, using both land surveys and remote sensing techniques. This was compared to woody plants in a selected benchmark (reference) site. The encroaching woody species (*Grewia flava*, *Senegalia mellifera*, *Vachellia tortilis* and *Ziziphus mucronata*) were divided according to height classes so that specific densities could be determined. The total woody species density in the benchmark was 421 TE ha⁻¹, where one tree equivalent (TE) represents a tree of 1.5 m high. The three selected study sites in the same ecological zone as the benchmark recorded woody species densities of 9 640 -, 6 461 - and 7 820 TE ha⁻¹ respectively with an average of 7 974 TE ha⁻¹. Woody plants were the densest within the 0.5 - , 1 – and 2 m height classes. *Senegalia mellifera* (50%) was the dominant woody encroacher and it was found in all the study sites. *Vachellia totilis* contributed 18% of the total woody plant density whereas *Grewia flava* and *Ziziphus mucronata* were recorded at 16% of total woody plant density. This thickening is termed ‘bush thickening’ and was conspicuous in the communally managed rangelands where the study area was located. Remote sensing imagery was used where Spot 5 images was degraded to 20 m in order to make it homogeneous with Spot 2 and 4. The area encroached with woody species in 2006, 2010 and 2014 were 293 -, 307- and 433 ha respectively, a clear indication that the problem of bush thickening is worsening. The woody species densities increased by 4.72% and 41.22% from 2006-2010 and 2010-2014. The overall accuracy assessment (the measure of how accurate the assessed values from ground truthing are in relation to the aerial photographs or satellite images) values for 2006, 2010 and 2014 were 85, 91 and 94%. The classification method used was texture analysis (Mean Euclidean Value) and the correlation coefficient between land survey data and remote sensing was 0.99. Woody encroachers are ‘threatening’ communally managed rangelands in the study area as it reduces the herbaceous layer and thus lowering the grazing capacity of the rangelands.

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CHAPTER 1: INTRODUCTION

1.1 Background

This study was carried out in the Ratlou Municipality of the Molopo District (Recently referred to as the Ngaka Modiri Molema District) in the North West Province of South Africa (Figure 4.1, Chapter 4). The study area is located within the semi-arid Savanna, classified as Thornveld by Acocks (1988). This area is located within in the Savanna Biome (Mucina and Rutherford, 2006). The southern African Savanna is an extensive area with dispersed thorn (*Senegalia* and *Vachellia* spp.) trees and tall grass typifying landscape. Although, a savanna is an unstable and makeshift environment, the grasses and trees defining it are in competition with each other (Versfeld *et al.*, 1998). Savannas contain areas of degradation between the grassland and woodland biomes. Savanna, in this study, is an intermediate form, called bushveld with many high bushes among grasses and trees (Grossman and Gandar, 1989). The vegetation consists of a herbaceous layer usually dominated by grass species and a discontinuous to sometimes very open tree layer (Mucina and Rutherford, 2006). It is a semi-arid area with a rainfall of 600 mm a⁻¹ (Snyman, 1991; Van Wyk *et al.*, 2011). Such low precipitation had innumerable impacts on land-use, making farming a risky enterprise and low in productivity. The low rainfall has been more favourable to pastoralism which has a long tradition in the area.

Bush thickening thickening is defined as a directional increase in the cover of indigenous woody species in savannas (O'Connor *et al.*, 2014). Generalized evidence suggests that the presence of high bush densities and pioneer grasses, causes plants to colonise a region and thus start the processes of succession in the thornveld (Snyman, 1991). There has been no detailed survey of the ratio of trees to grasses in the region and course of botanical variation in this particular area of the Savanna. Trees, especially encroaching species also occur along dolerite dykes, rock intrusions where sub-surface water collects and in areas where long term overgrazing was common. It is especially in communally managed rangelands, where encroaching woody species are causing dense thickets that eventually limit grazing opportunities, as the herbaceous layer is out competed. Communally managed areas, such as where the study area was located (Figure 4.1), are subjected to heavy grazing and there is virtually no control or any form of grazing management (Hoffman and Ashwell, 2001). Communal land tenure is often quite secure,

individuals have few rights to own or sell the land. In case of South Africa, this right is usually vested in the provincial government structures. Areas managed under communal tenure are broadly equivalent to the former homelands (Bophuthatswana) and self-governing territories. Heavy grazing is thought to be unavoidable in communal rangelands because of the problems inherent in communal ownership of a resource where individual benefit is exploited at the expense of the community (Hardin, 1968). In Africa, it is also widely assumed that pastoralists keep larger numbers of livestock than they actually need, as livestock represents wealth and status (Lamprey, 1983). This was understood to be exacerbated by the growing population densities in many of these areas together with a lack of economic development, which makes people strongly reliant on natural resources for their livelihoods (Vetter *et al.*, 2006)

Besides overgrazing of the herbaceous layer by cattle, the hoof action of animals leads to loosening of the topsoil layer that could cause erosion of the top layer and it also compacts the soil especially the secondary layers (Jacobs, 2000). Encroaching bushes generally outcompete the herbaceous layer, thus lowering the grazing capacity. The plants that replace climax species during or after encroachment, are mainly pioneer grasses and unpalatable woody encroaching species. The most prominent encroachers included *Senegalia mellifera* (blackthorn) (Mogodi, 2009), *Vachellia tortilis* (umbrella thorn) (Molatlhegi, 2008), these two species are good browsing species because leaves, pods and young shoots are nutritious and make fodder for livestock and wild animals (Hines *et al.*, 1993; Orwa *et al.*, 2009; Nonyane, 2013; FAO, 2014), *Euclea undulata* (small-leaved guarri), *Grewia flava* (raisin bush) and *Ziziphus mucronata* (buffalo thorn). Overgrazing and consequent variations in veld composition reduce grazing capacity (Otsamo and Maua, 1993). The woody encroachers are of lesser usage to cattle, which are grazers although browsing goats will utilize twigs and shoots. However, cattle will also browse under dry circumstances if grass is limited (Sanon *et al.*, 2007). Grazing capacity of the area is assumed to fall further because the pioneer grasses are less nutritive and less palatable than those ‘demanding’ more optimum conditions (Pasiecznik *et al.*, 2001; Jacobs, 2003). Selective overgrazing allows the proportion of poisonous plants including *Geigeria ornativa* (moremoshumi) and *Orthnoglossum viride* (sekanane) to rise. The most serious concern of woody plant succession in this area, is held to result from the indigenous woody encroacher, *Senegalia mellifera*, which is an aggressive-growing (Donaldson, 1969; Hagos and Smit, 2005), shallow-rooted, thorny woody species (Smit,

1999). *Senegalia mellifera* is a prominent woody encroacher in the study area and competes intensively with the herbaceous layer, causing a general shortage of available grazing land. *Senegalia mellifera* has a rounded to spreading flat crown, arising in the bushveld and semi-desert areas, frequently on Kalahari sand and forming impenetrable thickets in areas where overgrazing is persistent (Jacobs, 2000).

Assessments in the Molopo District showed the yield of grass deteriorating by approximately 75% as the density of *Senegalia mellifera* increased from zero to 1 071 TE ha⁻¹ (Tree Equivalents per hectare, see later, Chapter 5) (Otsamo *et al.*, 1993). Despite state endorsement, the declaration that grasslands are a natural climax in this area (Molopo District) (Dougill and Trodd, 1999), and that the appearance of woody plants is less natural (Jacobs, 2000). There are strong reasons to question arguments that an increase in bushes is ‘grazing induced’ degradation (Linstädter *et al.*, 2014). In range management as well, there has been an improvement of non-climax models that ecosystems vary between states, depending upon disturbances (Jacobs, 2000; Brooks *et al.*, 2004). The acknowledgement of some change as a natural phenomenon and an appreciation of variations in ecological systems have made scientists hesitant to attribute all vegetation changes to herding and to label these changes as ‘degradation’ (O’Connor *et al.*, 2014). Such rationale gives emphasis to insects, fire regimes, rainfall levels, soil conditions and other non-human factors that may cause changes in bush and grass levels (Jacobs, 2000). Recognising spatial variation is critical. Rainfall can also be a factor, supporting bush growth at the expense of grasses. Wet periods can give bushes a competitive advantage over grasses that will last through dry cycles, thus giving bushes the advantage in competing with grasses in the long run (Brooks *et al.*, 2004).

A presumably anthropogenic change involves the rise in the level of atmospheric CO₂ (O’Connor *et al.*, 2014). According to O’Connor *et al.* (2014), the increase in atmospheric CO₂ affected grass-bush ratios worldwide (Jacobs, 2003). In the past two centuries, CO₂ in the atmosphere has risen from approximately 275 parts per million (ppm) to the current level of approximately 363 ppm (Polley *et al.*, 1997). Presumably, under these circumstances, C-3 photosynthesis is less of a disadvantage even on dry lands, and woody plants are better able to succeed against C-4 grasses. The emphasis on biological and physical factors enhancing bush growth does not constitute a denial that grazing affects veld composition but it demands a nuanced understanding of the many influencing

factors (Jacobs, 2000), including natural and global anthropogenic factors (Reynolds *et al.*, 2001). The definition of environmental degradation requires strict qualification. The occurrence of vegetation change cannot be taken as evidence for a degradation of the natural state (Tongway and Ludwig, 1994). According to Hoffman and Ashwell (2001) land degradation refers to a loss or reduction, in arid, semi-arid and dry sub-humid areas, of the biology or economic productivity and complexity of rained cropland, irrigated cropland, or range, forest pasture and woodlands resulting from form of a process or land uses (Verheye, 2009; Global Environment Facility, 2011), including processes arising from human activities such as:

- (i) Soil erosion caused by wind and/or water,
- (ii) Deterioration of the biological and chemical or economic and physical properties of soil (Mark and Lindsay, 2016; Nkonya *et al.*, 2016) and
- (iii) Long-term loss of natural vegetation.

1.2 Problem statement

Woody plant invasion is a process which encompasses the steady replacement of grazing grasses by encroaching woody plants, which are unpalatable to cattle or sheep. The grazing capacity is believed to fall further because the pioneer grasses are less palatable than those requiring more optimum circumstances. Spikey woody thickets become established which can become impossible to move through. The land, then, becomes unsuitable for farming, especially cattle farming, where grasses are of utmost importance for economic prosperity.

1.3 Justification of the study

Bush thickening has adverse effects on livestock farming, as it results in the decline of livestock production, due to the loss of grass production on grazing lands (Smit, 1999). Bush thickening has also reduced the carrying capacity of South African rangelands, which has resulted in the concomitant loss of income, which aggravates poverty and unemployment (De Klerk, 2004). Bush thickening continues to cause substantial losses in communal farms, resulting in lower food security and nutrition. It is especially in

communal areas, where grazing lands are unprotected and continuously under the influence of anthropogenic factors.

1.4 Aim and objectives

1.4.1 Aim

The aim of the study is to monitor and quantify the extent of encroaching woody species and the rates of expansion in the selected study sites.

1.4.2 Objectives

- To quantify encroaching woody species densities in selected sites
- To evaluate the growth rates of woody species in selected sites
- To use satellite images to detect change of woody plant succession over time

1.5 Thesis outline

Chapter 1: Introduction

In Chapter 1, a general overview of the Savanna in the North West and woody species invasion, with a closer look at the communal area. The effects of livestock grazing practices are also explored in this chapter. The aim and specific objectives are outlined. An outline of the dissertation chapters is also provided.

Chapter 2: Literature Review

This chapter reviews previous relevant studies that were done before in line with the aims and objectives of this study.

Chapter 3: Study Area

This chapter provides a background to the geographical characteristics of the study area. Physical characteristics, viz. soil, geology and vegetation and climatic conditions also presented.

Chapter 4: Materials and Methods

This chapter describes the various methods and techniques employed to ensure that the aims and objectives of the study are achieved. The methods utilised include land survey, GIS and remote sensing.

Chapter 5: Bush densities and development of encroaching woody species

In Chapter 5 the results obtained using the land survey methods described in Chapter 3 are presented. An analysis of vegetation condition trends is presented in the form of tables and graphs.

Chapter 6: Remote Sensing

In Chapter 6, the results obtained using the remote sensing methods described in Chapter 3 are presented. An analysis of vegetation condition trends is presented in the form of maps, tables and graphs.

Chapter 7: Discussion and Conclusion

This chapter explains the results obtained in chapter 5 and 6, and the relation between the two using the correlation co-efficient.

CHAPTER 2: LITERATURE REVIEW

2.1 Introduction

2.1.1 The ecology of the savannas

The Savanna Biome is regarded as one of the most prominent biome in southern Africa and also provides a livelihood to a major part of the human population of Africa (Scholes and Walker, 1993). According to Scholes (1997), savannas amount to 54% of the total area of southern Africa (1 435 713 km²) (Knoop and Walker, 1985). Savannas may be envisaged as biomes largely dominated by woody vegetation and grasses. Commonly, there are at least two-layers of above-ground structures viz., a discontinuous crown cover of the tree layer (2 – 10 m) and a grassy layer (0.5 – 2 m) (Scholes, 1997). Savannas generally consist of tropical vegetation in which C-4 grasses often dominate the herbaceous stratum and a woody stratum which is usually fire resistant and which ranges from a low aerial cover to a closed woodland (Baruch and Bilbao, 1999). The former constitutes open savannas and the latter, closed savannas. Leguminous trees and shrubs, some of which have been shown to be nitrogen fixers, dominate the tree layer in many parts of southern Africa (Van Auken, 2009).

A savanna environment could be described as hot, containing seasonally dry grassland with scattered trees and is mostly found to be intermediate between grassland and a forest (Seymour, 2008). Southern African savannas range from tall, moist woodlands receiving up to 1 800 mm rainfall per annum, in northern Angola to sparse grasslands with scattered thorn bushes on the margins of the Namib Desert where rainfall might be as low as 50 mm a⁻¹ during drought years (Scholes, 1997; Nxele, 2010). Rainfall usually occurs in the warmer, summer months with a dry period of between two to eight months duration (Huntley and Walker, 2012). During the dry period, fires are a typical phenomenon at intervals varying from one to fifty per year (Huntley, 1982). Included within this concept are the Miombo–Mopane grasslands, the tall grass ‘derived savannas’ bordering the Guineo–Congolese rainforests, the shrublands of the Kalahari and the Khomas Hochland, the grassy dambos and chanas of central Africa and the succulent thickets of the valley bushveld of the Eastern Cape (Huntley, 1982; Scholes, 1997; Huntley and Walker, 2012).

Such a diversity of physiognomy, flora and environmental conditions has tended to mask otherwise clear relationships between constituent ecosystem relationships that indicate the existence of distinctive arid and moist savanna biomes in southern Africa (Scholes, 1997; Nxele, 2010; Huntley and Walker, 2012). Arid and moist savannas differ significantly in terms of their climatic, dynamics, fauna, flora, physiognomy and soil conditions. The differences are recognized in parts of central Africa which merge increasingly towards the south and south-east, ultimately forming a small scale vegetation mosaic separated by subtle soil and climatic changes (Huntley, 1982; Scholes, 1997; Nxele, 2010). According to Scholes *et al.* (2002), on the western front, where savannas gradually merge into deserts, it appears there is a clear gradient in woody biomass which might correlate with a south to north gradient in rainfall (i.e. from 200 to 1000 mm mean annual precipitation). The structural variation noted within savannas and the differences observed between them, illustrate the role and importance of these biomes in the ecosystem in that they provide diverse environments for diverse species (Teague and Smit, 1992; Van der Vijver *et al.*, 1999).

Savannas provide a livelihood to many organisms, primarily through supplying grazing areas, wood fuel, timber and other resources to the informal and subsistence economies (Scholes, 1997). Savannas provide the feed for livestock and enable the ecotourism industry. Savannas also have a global contribution through their emissions of trace gases from soils, fires, vegetation and animals (Otter *et al.*, 2002). They sequester carbon in soils and biomass (Hernández-Hernández and López-Hernández, 2002) and host a wide biodiversity. Southern African savannas are regarded as part of the Sudano-Zambezian phytochorion as they display many common genera and species with the savannas of Central and East Africa (Scholes, 1997). With comparison to the savannas of West Africa, they share many plant families but very few species (Scholes, 1997).

Despite this continental variation, some common features do exist between southern African savannas and those of the Indian Peninsula, though fewer with those of America, Australia or South-East Asia (Johnson and Tothill, 1985). However, this floristic differentiation does not take away the common essence of a savanna. In essence, the Savanna is a biome which contains vegetation consisting of both a tree and a grass layer with complex interactions between these two structural layers (Scholes, 1997). The Savanna Biome is well developed over the lowveld and Kalahari regions of South Africa

and its vegetation is dominant in Botswana, Namibia and Zimbabwe (Low and Rebelo, 1996).

The sensitivity of savannas to mismanagement, their global distribution and the amount of biodiversity they nurture, as well as the number of human populations they support, is good enough motivation to consider savannas as worth protecting. Land-use change is, without doubt, one of the most important factors affecting ecological systems and also interacts with other components in causing global change (Vitousek, 1992; Henderson, 1995; Ringrose *et al.*, 1998; Manlay *et al.*, 2002). As such, savanna systems that are subjected to some form of anthropogenic activity are prone to disruption.

In most cases, savannas have an extended dry season and a rainy season (Yarnell *et al.*, 2007; Young *et al.*, 2009). Because of this pronounced seasonal variation, the animals that are found in savannas can be seen to have adapted to a great deal of variability in the food supply throughout the year. This adaptation could be because there are times of plenty (during and after the wet season) and times of scarcity (during the dry season). In order to cope with life in savannas, some animals have opted for migration during the dry seasons. Prominent animal taxa found in savannas, range from invertebrates (like grasshoppers, termites, and beetles) to mega herbivores and subsequently, predators and from this, it can be noted that savannas of the world, as different as they are, also support a significant amount of faunal diversity (Roques *et al.*, 2001). In Africa, savannas cover most parts of the continent and have considerable amount of structural variation in terms of tree–grass densities and also a significant amount of biodiversity. This variation may be affected by factors such as rainfall frequency and disturbance for example, grazing, fires, habitat destruction (in the case of elephants) (Van der Vijver *et al.*, 1999; Sankaran *et al.*, 2005).

Grasses, which have shallow root systems and thus can utilize topsoil nutrients and water (Walter, 1939; Huston, 1994), usually outcompete the woody plants (which have deep penetrating roots and are slow growing). The massive germination of woody plants transforms open, diversity-rich savannas into closed, unusable savannas, which have low biodiversity, this is shown by different species of insects, spiders and birds etc. The potential of the land to sustain both humans and their livestock is thus reduced and biodiversity is negatively affected (Asner *et al.*, 2004). In order for species to survive, it

becomes imperative for them to adapt to their environment. This creates habitats for other species, which might be associated with the species undergoing local adaptations. Therefore, one species' response to an environmental condition might result in the creation of a habitat, which might later result in inter- and intra-competition between and within species because of the resources that might be available in that particular habitat (De Jong, 1990). Extensive empirical work has demonstrated that local adaptation exists across different taxa, both in the animal – and plant kingdoms (Raven and Johnson, 1992). A sound understanding of local adaptation might be a tool for understanding why and how speciation occurs. Local adaptation may be viewed as a genetic response of a species to its environment which enables it to respond to an environmental stimulus appropriately in response to the specific environmental condition (Schlichting, 1986; De Jong, 1990) rather than the phenomenon which is due to resource availability. Savannas host sensitive ecosystems and are sensitive to changes such as climate changes (rainfall, temperature etc.) and also anthropogenic factors such as overgrazing of natural pasture, improper applications of fires etc. One of the most important changes taking place in savannas is the successional process of bush thickening, seemingly a natural process.

2.1.2 Effects bush thickening

Humans, however, appear to be the main cause to the problem of bush thickening (Ward, 2005). Barbour *et al.* (1987), however, stated that disruptions of biological control mechanisms have resulted in the current problem of bush thickening. Bush thickening refers to the increase in woody plants while bush thickening refers to invasion of aggressive undesired woody species (O'Connor *et al.*, 2014) According to Joubert (1997), rangelands are encroached with woody species, because of poor management practices. Inappropriate fire management, poor understanding of savanna ecosystem models and poor understanding of the phenology and physiology of the encroaching species, have accelerated the bush thickening problem (Joubert, 1997). The energy released by a fire, has been found to have no effect on the grass sward. However, the intensity of fire, influences the effects of fire on woody plants (Trollope and Tainton, 1986; Tainton, 1999; Walter *et al.*, 2004; Govender *et al.*, 2006). Fire intensity is positively related to the mortality of woody plants (Trollope and Tainton, 1986). The mortality of woody plants due to fire alone, however, is low (Trollope and Tainton, 1986; Midgley *et al.*, 2010). Increased fire intensities induce a proportionally greater topkill of woody plants (Trollope

and Tainton, 1986). Taller plants that have increased stem diameters, compared to shorter plants are, however, considered to be less vulnerable to stem death (Balfour and Midgley, 2006). Relatively small amounts of topkill have been observed in trees (≥ 4 m) after intense fires (Trollope and Tainton, 1986). To maximise grass yield, burning has typically been done when grasses are dormant. In fire-climax grasslands this has typically been done in late winter to early spring (four weeks before the first rains or two weeks after the first rains) (Everson, 1985; Tainton, 1999). Despite the negative effects of burning savanna in early-to-mid winter on the grass sward, it is still widely practiced (Tainton, 1999).

Walter's Two-layer model (Nefabas and Gambiza, 2007) indicates that if the grass layer is over utilised, it loses its competitive benefit and can no longer use water and nutrients efficiently. This causes a greater water and nutrient infiltration rate into the subsoil, thus favouring the development of deeper-rooted woody plants. Some empirical evidence in support of the Two-layer Hypothesis (Kgope *et al.*, 2010), however, does exist (Hesla *et al.*, 1985; Knoop and Walker, 1985; Sala *et al.*, 1989) while a few studies rejected it (Weltzin and Coughenour, 1990; Belsky, 1994; Sieghieri, 1995). Teague and Smit (1992) regard this model as an oversimplification of the interaction between trees and grasses. The State and Transition Model (Jacobs, 2000), recognises the dynamic nature of savanna ecosystems, which are event driven where rainfall and its variability plays a more important role in bush thickening. These have a direct role in vegetation growth than the intensity of grazing. Thus, bush thickening is not a permanent occurrence and a savanna could be changed to its grass dominated state by suitable management or environmental conditions.

Veld fires in spring enable better moisture availability for improved recovery of the grass sward. However, spring fires may expose soil surfaces to the direct negative impact of heavy rains, because erosion would occur (Everson, 1985). Repeated annual burning of continuously or heavily grazed savannas systems in early-to-mid winter in wildlife systems is also not conducive to healthy grass swards. Grazing in these areas will be continuous, as the nutritious regrowth is attractive to herbivores (Tainton, 1999). Such repeated continuous grazing without rest periods will reduce grass vigour and consecutive fuel loads (Van Langevelde *et al.*, 2003). In fire climax grasslands, savannas burnt in

early-to-mid winter, with a two-year resting interval, have caused a decrease in rangeland palatability (Everson, 1985). The preferred, palatable grass species are replaced by less palatable grasses under this regime (Everson, 1985). When using fire to control woody species invasion, it is therefore essential to contemplate the event dependent (e.g. intensity of fire) and 'interval-dependent' (i.e. mean return period and grazing regime) factors that may complicate responses of woody plants to fire frequency (Pratt and Knight, 1971; Higgins *et al.*, 2007; Munkert, 2009).

Some of the causes of bush thickening can be summarised as:

- human impact (Jeltsch *et al.*, 1998, 2000)
- fire (Higgins *et al.*, 2000)
- temperature and precipitation (O'Connor *et al.*, 2014)
- absence of browsers (Pringle *et al.*, 2014)
- drought (Scholes and Walker, 1993)
- nutrient heterogeneities (spatial) in water and seed distribution (Jeltsch *et al.*, 1996).
- livestock grazing (O'Connor *et al.*, 2014)
- increased atmospheric CO₂ (O'Connor *et al.*, 2014)

The encroachment of invasive bush species in Namibia has drastically reduced habitat productivity (Karuaera, 2011). Invasive alien species woody species have the ability to outcompete indigenous species and grasses in the area, resulting in habitats dominated by encroaching especially thorny, species. Human interference has prevented regular occurrence of fires that regulates savanna vegetation, maintaining a balance between the grasses and woody plants. Furthermore, the replacement of wildlife with livestock (grazers) has diminished the growth rates of grasses and thus enhanced bush thickening. Human interference has prevented regular occurrence of fires that regulates savanna vegetation, maintaining a balance between the grasses and woody plants (February *et al.*, 2013). O'Connor *et al.* (2014), however, concluded that an increase in atmospheric CO₂ is the main driver of bush thickening.

2.1.2.1 Impact on agriculture

Bush thickening has serious economic implications in the agricultural sector in the North West Province (NWP). It results in the decline of livestock production due to the loss of grass production and loss of grazing and browsing material in the game ranging industry (De Klerk, 2004). It also results in lower productivity of individual animals (Kruger, 2002). Most encroaching woody species are unpalatable to domestic livestock. Bush thickening has reduced the number of cattle on commercial farms by 47% over the last 30 years (De Klerk, 2004). Bush thickening remains the single most important factor that limits red meat production in commercial farms in NWP and at the national level, bush thickening has resulted in the loss of up to R 4.9 billion in lost meat production each year (De Klerk, 2004).

2.1.2.2 Impact on botanical diversity

Bush thickening is seen as a major threat to botanical diversity (De Klerk, 2004). Species richness and endemism are not high in bush encroached vegetation (De Klerk, 2004) and encroaching species usually have the ability to displace other plant species, resulting in a decline in plant diversity. Bush encroaching species especially thorn trees (*Acacia* species) have extensive and well-developed tap roots and lateral root systems that reach deep into the soil to obtain water and nutrients. Grass roots usually occur in the topsoil layer and are unable to penetrate further into the soil (De Klerk, 2004). The grass-tree competition is vastly influenced by the amount of water retained in the upper soil layer that moves to the lower soil where tree roots predominate. In the case when the grass layer is over-utilized, it loses its competitive edge against the woody species and consequently bush thickening results (De Klerk, 2004). As a result, fewer non-woody species are found to occur in the bush encroached area (De Klerk, 2004).

2.2 Bush thickening and land tenure

According to Hoffman and Ashwell (2001), land tenure plays an important role in bush thickening. Palmer and Ainslie (2005) stated that there are three categories of land tenure (70% commercially, 16% is reserves or freehold industrial and urban and 14% is commercially managed without clear individual boundaries and managed for subsistence)

in South Africa. Production systems in communal areas, based on pastoralism and agro pastoralism, are subsistence-based and labour intensive. Stock rearing is very ancient; cattle predominate but sheep and goats are very important. Fire and browsing have reduced woody vegetation, but bush thickening remains a problem. There are often unclear boundaries, generally open access rights to grazing area and farmers are subsistence oriented. Here, land tenure issues considerably hamper the introduction and adoption of improved management practices (Palmer and Ainslie, 2005). According to Hoffman and Ashwell (2001), 50% of communal districts and 38% of commercial districts in South Africa are affected by woody plant invasion of grazing lands.

2.3 Bush thickening and climate change

The bush thickening problem is also inter-linked with climate change. Previous studies indicate that increased atmospheric carbon dioxide concentrations [CO_2] have an effect on the different floristic components of savanna ecosystems, especially C-3 and C-4 plants (Smit, 1999). According to Wand *et al.* 1999 both C-4 and C-3 species increased total biomass significantly in elevated CO_2 , by 33% and 44%, respectively (Wand *et al.*, 1999). Differing tendencies between types in shoot structural response were revealed: C-3 species showed a greater increase in tillering, whereas C-4 species showed greater increase in leaf in elevated CO_2 (Wand *et al.*, 1999). Increases in atmospheric carbon dioxide (CO_2) improve water-efficiency and increase carbon up-take in thorn tree species. Elevated CO_2 reduces the transpiration rate of grasses, causing deeper infiltration of water to the sub-soil and, therefore, favouring woody plants. CO_2 concentrations also influence the photosynthetic rates of plants as well as light and nutrient-use efficiency. Climate change is also known to prolong the severity of droughts. Such scenarios favour encroaching woody species as they can still extract moisture far below the soil as a result of their developed tap root and lateral root systems (De Klerk, 2004). It is anticipated that the impacts of bush thickening may continue to intensify as increasing atmospheric CO_2 concentrations exacerbate local factors (such as changes in grazing or burning regimes) to further drive bush thickening in the future (O'Connor *et al.*, 2014; Reed *et al.*, 2015). Bush encroached areas have greater biomass (including long tap roots) than grass-dominated areas, implying that they sequester and store more carbon from the atmosphere (O'Connor *et al.*, 2014). During periods of droughts, encroaching woody species have an advantage over grasses and out-compete the grasses for available moisture, nutrients and space. In

the context of global climate change, an increase in woody plant cover has been primarily linked with elevated $[CO_2]$ (Palmer, 2007, O'Connor *et al.*, 2014). Evidence suggests that elevated CO_2 affect individual plant species and community composition in ecosystems, favouring the survival, vegetative growth, seed production and seedling of encroaching species (Dirkx *et al.*, 2008). Although there are many causes of bush thickening O'Connor *et al.* (2014), however, came to the final conclusion that increased atmospheric $[CO_2]$ is the major driver of this process.

2.4 Woody species growth and development

Knowledge of general principles and trends in tree growth and stand development is needed to make sound decisions on these matters (DeBell *et al.*, 1996). Woody species growth increase was measured by measuring the heights of plant in the height classes due to the short period of this study it was difficult to measure height increases over a single growth season. However, according to Table 5.1 woody species are growing (increase in height) at different rates.

2.5 Remote sensing and GIS in land–use and cover change

Remote Sensing in land–use/land cover, as defined by Barnsley *et al.* (2001), is the physical materials on the surface of a given piece of land (i.e. grass, concrete, tarmac, water), and land–use as the human activities that take place on, or makes use of the land (i.e. residential, commercial, industrial). Land–use can consist of varied land covers, (i.e. a mosaic of bio–geophysical materials found on the land surface). For instance, a single–family residential area consists of a pattern of land–cover materials (i.e. grass, pavement, rooftops, trees, etc.), this pattern applies in the selected areas of research interest (i.e. grass, trees, bare area etc.). The aggregate of these surfaces and their prescribed designations (i.e. park) determines land–use (Anderson *et al.*, 1976). Land–use is an abstract concept, constituting a mix of social, cultural, economic and policy factors, which have little physical importance with respect to reflectance properties, and hence has a limited relationship to remote sensing (Treitz and Rogan, 2004). Remote sensing data record the spectral properties of surface materials, and hence, are more closely related to land–cover. Land–use cannot be measured directly by remote sensing, but rather requires visual interpretation or sophisticated image processing and spatial pattern analyses to

derive land–use from aggregate land–cover information and other ancillary data (Cihlar and Jansen, 2001).

Integrated analyses within a spatial database framework (i.e. Geographical Information System (GIS)) are often required to assign land cover to appropriate land–use designations. Success in land-cover and land–use change analysis using multi-temporal remote sensing data (i.e. Spot 2, Spot 4 and Spot 5) is dependent on accurate radiometric and geometric rectification (Schott *et al.*, 1988; Dai and Khorram, 1999). These pre–processing (Chapter 4, Section 4.3.4.) requirements typically present the most challenging aspects of change detection studies and are the most often neglected, particularly with regard to accurate and precise radiometric and atmospheric correction (Chavez, 1996). For change to be identified with confidence between successive dates, a consistent atmosphere between dates must be modelled so that variations in atmospheric depth (i.e. visibility) do not influence surface reflectance to the extent that land-cover change is detected erroneously. This is particularly important in biophysical remote sensing where researchers attempt to estimate rates of primary productivity and change in total above ground biomass (Coppin and Bauer, 1996; Treitz and Howarth, 2000; Franklin, 2001) where change is dramatic, (i.e. conversion of agricultural land to residential), the ‘change signal’ is generally large compared to the atmospheric signal. Here, the accuracy (Chapter 4, Section 4.3.7.; Chapter 6, Section 6.7) and precision of geometric registration influences the amount of spurious change identified where accurate and precise registration of one date to the other is achieved and identified surface changes can be confidently attributed to land conversion. Inaccuracy and imprecise co–registration can lead to systematic overestimation of change, although methods have been developed to compensate for these effects (i.e. spatial reduction filtering).

Typically, the quality (i.e. precision and accuracy) of automated per-pixel classifications in urban areas using remote sensing are poor, compared to communal areas. Also, urban areas present the problem of having logical correspondence between spectral classes and functional land–use classes (Treitz and Howarth, 2000).

It remains difficult to map the point and linear features, particularly digitally, due to the fact that they are not always recognizable at the spatial resolution of the data, nor are they represented at their true location due to sensor and panoramic distortions inherent in

satellite data collection. It has also proven difficult to digitally separate linear features such as road networks from surrounding land-cover and land-use or mixed vegetation in high mountainous areas (Treitz and Howarth, 2000). This is largely due to the complexity of pattern recognition procedures required for tracing specific cultural edge features.

2.6 Bush thickening, tree heights and population dynamics

In order to quantify bush thickening at Disaneng Village, woody species were divided into height classes (See Chapter 4) and expressed in tree equivalents per hectare (TE ha⁻¹). The woody densities (TE ha⁻¹) in each height class within each group (i.e. *Grewia flava*, *Sengalia mellifera*, *Vachellia tortillis* and *Ziziphus mucronata*) were determined for each site in Disaneng (sites I, II and III), (Figure 4.1, Chapter 4) and the benchmark site. Disaneng Site I was near the main road from Mahikeng to Tshidilamolomo (Figure 4.1) and it is easily accessible to livestock. For local communal residents who regularly chop down wood as their source of fuel, the study site is a browsing–grazing area for cattle, sheep and donkeys (Figure 4.1). Site II was more accessible to livestock than humans because it was further away from the road (Figure 4.1) while site III was furthest away from the main road and contained impenetrable thickets of *Senegalia mellifera*, causing limited access to both human and livestock (Figure 4.1).

Plant species composition and productivity within a region, largely reflect the prevailing climate, whereas seasonal and annual variability in rainfall and temperature play a central role in dictating the dynamics of populations over time (Ricardo and Nippert, 2015). However, substantial spatial variability occurs across landscapes and broad–scale climatic variables cannot account for the spatial patterns which shape vegetation form and function on a local scale (Tainton, 1999). Soils and topography exert a strong influence on patterns of plant distribution, growth and abundance over the landscape through regulation of the availability of moisture from precipitation, which also affects nutrient availability (Schmidt *et al.*, 2002). The condition of the soil is another factor which prevents optimum vegetation development. The local soil in the study sites, was sandy with a thin layer of humus (the organic portion of the soil created by partial decomposition of plant or animal matter), which provides the vegetation with nutrients.

Grazing animals affect plants directly and indirectly. The direct effects of grazing are those associated with alterations in plant physiology and morphology, resulting from defoliation and trampling (Anderson and Briske, 1995). Grazing also influences plant performance indirectly by altering microclimate, soil properties (Wilcox and Thurow, 2006) and plant competitive interactions. Grazing reduced total plant cover, especially the herbaceous layer, and substantially altered the species and functional composition at all sites. Over time, the combined direct and indirect effects of grazing on plant growth and reproduction are manifested in plant population dynamics. Herbivores affect the productivity, composition and stability of plant assemblages through mediation of plant natality, recruitment and mortality and may cause directional change in community structure and function (Kraaij and Ward, 2006).

The pathway of succession following relaxation or removal of grazers may differ substantially from the pathway of retrogression, depending on the mobility and availability of propagules (portion of plant or fungus i.e. seed or spores), soil conditions and climatic variables. In addition, the probability of ecosystem recovery to previous states may be greatly reduced beyond certain critical threshold levels of disturbance or change (Archer, 1989). The goal of grazing management for sustained yield is to identify these critical thresholds and manage landscapes so as not to exceed them (Skarpe, 1990). In this area, where communal farming is practiced, fencing and thus a camp system, was absent. This is in contrast to commercially managed areas where a camp system is usually implemented as a management strategy.

Management and manipulation of grazing lands for sustained livestock and wildlife production requires the seasonal integration of information on plant species composition and production across expansive, often heterogeneous areas (landscapes) and over extended planning horizons (decades) (Heitschmidt and Taylor, 1991). The increase of bush cover suppresses the productivity of herbaceous plant species (Hagos and Smit, 2005) and negatively affects the use of rangelands and conservation of biodiversity and eventually reduces its carrying capacity. This eventually implies lower revenue for the farmer, farming on communal land. Communally managed areas, such as where the study area was located, are subjected to heavy grazing and there was virtually no control or any form of grazing management. Communal land tenure is often quite secure and individuals have few rights to own or sell the land.

2.7 How trees grow

The survival and growth of trees depend on adequate amounts of carbon dioxide, water, sunlight and nutrients (Connell, 1978). Factors that influence the availability or use of these elements include tree characteristics, site characteristics, topography and climate (Ebersohn *et al.*, 1960; Chen *et al.*, 1999).

The interrelationship of all tree's parts is complex and especially so is its photosynthetic properties. A tree's root system comprises the important water-collecting mechanism that makes life possible for trees and ultimately for everything on the planet that depends on trees. Trees grow differently in different areas as they respond to environmental variables such as moisture content and soil type in different ways (Hendrik and Oscar, 2000).

An important biologic functionary of the tree root system is the tiny, nearly invisible root hair. Millions of microscopic root hairs wrap themselves around individual grains of soil and absorb moisture along with dissolved minerals (Scholes *et al.*, 2002). A major soil benefit occurs when these root hairs bind soil particles. Gradually, the tiny roots bind to so many soil particles that the soil becomes firmly tied into place. The result is that soil is capable of resisting the erosion of wind and rain and becomes a firm platform for the tree itself (Mashiri *et al.*, 2008).

To take full advantage of finding and obtaining available moisture, tree roots run shallow with the exception of the anchoring tap root. The roots are fragile and any significant soil disturbance close to the trunk can potentially harm a tree's health. The study sites were encroached by *Senegalia mellifera* trees, which generally has a shallow root system and competes directly and indirectly with the herbaceous (especially grasses) stratum (Hagos and Smit, 2005). A tree crown is where most bud formation takes place. In addition to branch growth, buds are responsible for flower formation and leaf production. The crown diameter, especially, is important when applying satellite image data to measure bush density expansion. The combination of field studies of tree parameters like height, trunk diameter, and biomass content (both above and below ground) with remote sensing based estimation of tree cover allows large scale inventories of both, current and future expansion. Tree crown cover is an essential parameter in many applications of remote

sensing data, as it significantly influences surface reflectance and temperature as well as surface anisotropy in the visible and in the thermal domain.

CHAPTER 3: STUDY AREA

3.1 Climate

Climate is a key factor of the geographical distribution of plants and animals (Tainton, 1999; Comole, 2014). Within any area of general climatic uniformity, local conditions of temperature, light, humidity and moisture vary greatly and these factors play an important role in the production and survival of plants (Tainton, 1999).

Precipitation in the Savanna Biome is seasonal (alteration of wet summer and dry winter periods). The Savanna Biome in South Africa does not occur at high altitudes and is found mostly below 1 500 m and extends to 1 800 m on parts of the Highveld, mostly along the southern most edges of the Central Bushveld. Temperatures are higher than those of the adjacent Grassland Biome at higher altitudes (Mucina and Rutherford, 2006).

The mean daily maximum temperature for February seldom drops below 26°C in the Kalahari region and some low altitude parts of the savanna in the east (Schulze, 1997). In July, this temperature remains above 20°C for most of the region, with temperatures at the maximum altitudes dropping to 18°C. The mean daily minimum temperature in February rarely reaches 16°C, with the temperature of substantial parts of the lower lowveld remaining above 20°C (Mucina and Rutherford, 2006). Mahikeng (formerly Mafikeng) falls within the Mafikeng Bushveld (SVK 1) (Mucina and Rutherford, 2006) and has summer rainfall with very dry winters. Frost occurs frequently in winter (Schulze, 1997). The monthly average rainfall and temperature for Mahikeng are indicated in figures 3.1 and 3.2 respectively. The mean monthly maximum and minimum temperatures for Mahikeng are 35.6°C and –1.8°C for November and June (Mucina and Rutherford, 2006). From Figure 3.2, it is clear that the months of October to March are the hottest, while June and July are the coldest. The Molopo District has a semi-arid climate, characterised by high day temperatures during the summer months and cool daily temperatures during winter months.

3.1.1 Rainfall

The North West Province (NWP) ‘offers’ almost year round sunshine, except June and July when cold dry winters prevail (Figure 3.1).

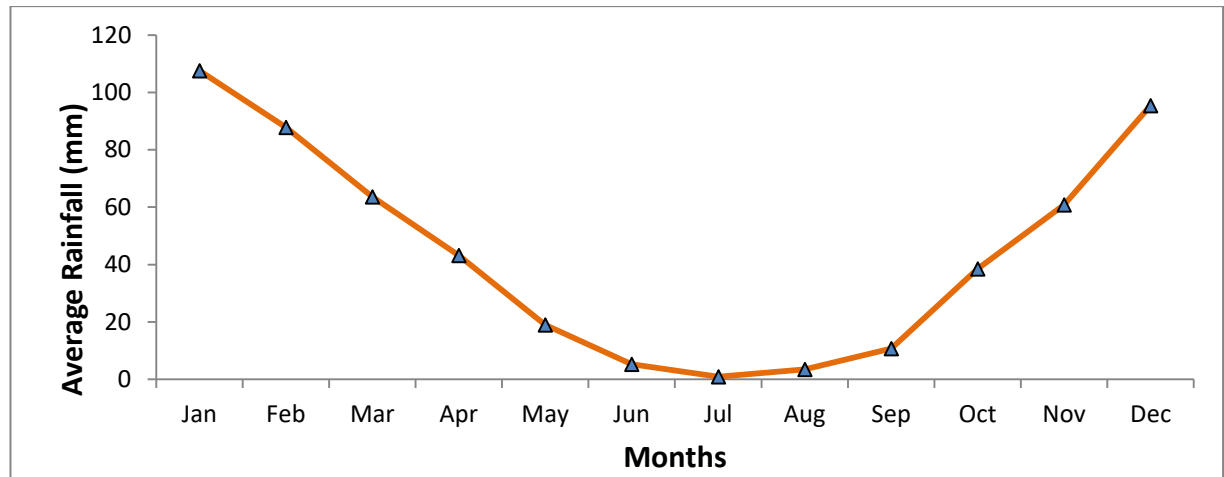


Figure 3.1: Monthly average rainfall in the study area (1985 – 2014) (Source: SAWS, 2015)

Rainfall is variable and concentrated mainly between five and six months of the year. Rain falls most frequently in the form of diurnal thunderstorms of relatively short duration. Figure 3.1 shows the variation in mean monthly rainfall and the number of rainy days per month over the period May 1985 to December 2014. From Figure 3.1 it is clear that there are at least two to three months (June to August) with limited precipitation. Considerable fluctuations in rainfall occur from year to year and, although the mean ranges from 500 to 600 mm a⁻¹, Schulze (1997) indicated that, the two years, 1975 and 1976, experienced 686 mm and 618 mm respectively. This was particularly pronounced during the 1976 to –77 seasons as there was a dry period over December to January, coupled with high temperatures which, had an influence on reproductive activities.

3.1.2 Temperature

Average summer temperatures range between 22°C and 34°C and winter brings with it chilly nights and dry, sunny days (Figure 3.2). The regular winter (May to July)

temperature is 16°C but can range from an average of 2°C to 20°C in a single day (Baldi and Paruelo, 2008).

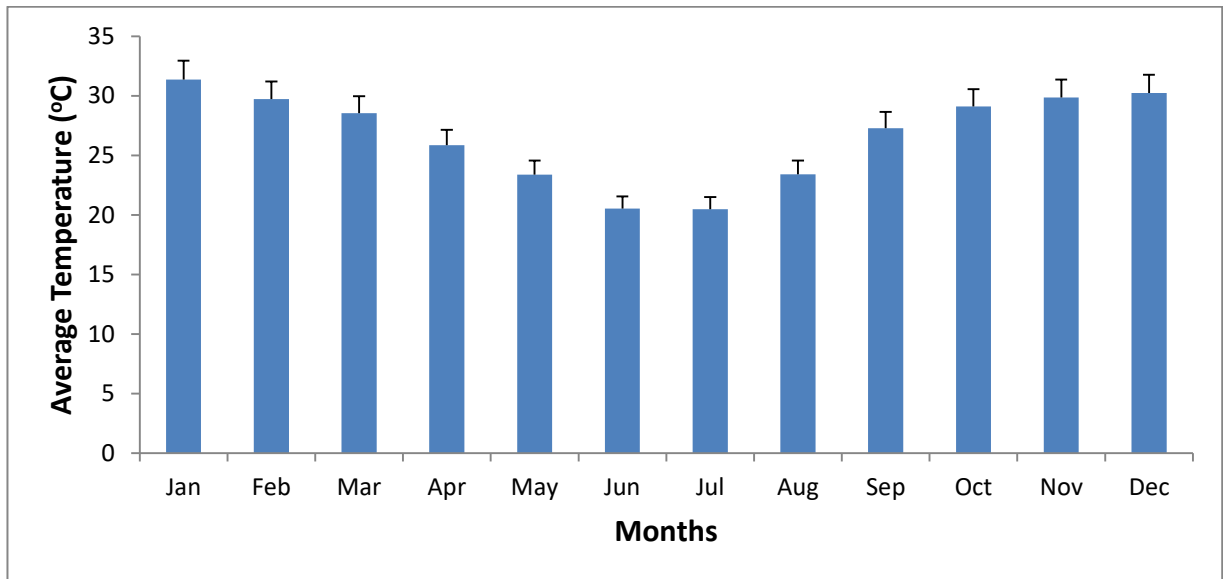


Figure 3.2: Average temperature of the study area (1984 – 2014) (Source: South African Weather Services, 2015)

The summer months are characterized by hot days and warm nights, particularly during the period September to December, after which the increasing rainfall introduces a cooling effect. There are variations, depending on the frequency and occurrence of the rainfall (Figure 3.1). During winter, however, temperatures are moderate during the day and cold at night, while clear cloudless skies are associated with this time of the year, permitting extensive heat radiation into the atmosphere. This accounts for the large daily range in temperature, especially during the months of June, July and August, reaching as much as 17.5°C between the mean maximum and mean minimum temperature (Ben-Shahar, 1991). Daily temperature ranges may even be greater as temperatures may drop to as low as –5°C at night and rise to 24°C during the day in winter. Ground frost is experienced nightly in the lower lying areas of the Disaneng Dam, but is rare in the study area. It is clear that the most frost occurs during the winter months, and the least during the raining season (Figure 3.1). Soil temperatures, however, do not fluctuate as much as environmental temperatures (Giannecchini *et al.*, 2007).

3.2 Vegetation patterns and biomes

The study area is characterised by a grassy ground layer and a distinct tree layer. The three environmental factors playing the most important role in the vegetation composition are (Low and Rebelo, 1996):

- (1) low precipitation, which prevent the upper layer from dominating
- (2) fires
- (3) grazing, which keeps the grass layer dominant

The study sites were mostly used for cattle grazing but other agricultural activities such as maize and groundnut production were also practiced. The Savanna Biome in South Africa is generally better conserved than most of the other biomes, mainly due to the presence of the Kruger National Park and the Kalahari Gemsbok National Park, now recognised as the Kgalagadi Transfrontier Park within this biome (Low and Rebelo, 1996). Conservation areas in the North–West Province, however, include, amongst others, the Borakalalo Nature Reserve, Pilanesberg National Park, Madikwe Game Reserve and Magaliesberg Nature Reserve, which cover a surface area of 193 500 ha (Mucina *et al.*, 2005).

Although the study area was located on the flat areas, the communities of the elevated sandstone regions developed in surface to deep non–calcareous soils which may be litholitic (Wigley *et al.*, 2009). On the lower slopes, a grass savanna is more apparent, with a gradual change in structure to tree savanna on higher slopes with impeded drainage. The upper elevations of the study area, therefore, are covered by grasses, such as *Cynodon dactylon*, *Eragrostis lehmanniana* and *E. rigidior* as well as thorn tree species. The study area is restricted to non–litholitic sandy soils, such as Haplic Arisols (Figure 3.3). These soils of the Molopo District (Mogodi, 2009) have a low nutrient content and consequently the ground cover is relatively open with *Urochloa panicoides*, *Eragrostis lehmanniana* and *Arsitida congesta* the dominant grasses (Personal observation by the author), although a grass survey was not part of the objectives of this study (See Objectives, Chapter 1). Occasionally, sandstone outcrops occurred. There is usually a 5–10% cover of plant litter, mainly leaves from *Senegalia mellifera* and *Vachellia tortilis* but twigs and grass also form a major part. Dead trees were occasionally found and provide

food and shelter for many invertebrates. In winter, the area tends to have large areas bare of vegetation and by August, most trees have shed their leaves. Only *Euclea undulata* appeared to be evergreen. The leaf litter and grass at the base of shrubs, therefore, are important in providing cover for insects and termites.

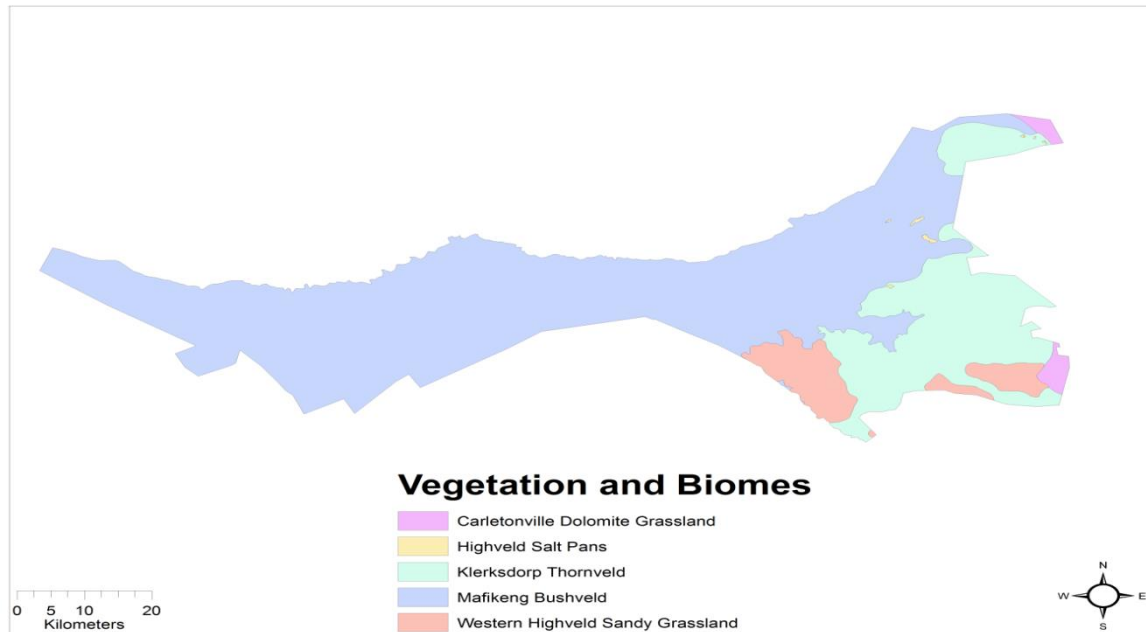


Figure 3.3: Vegetation patterns in the Molopo District (Source: Johnson *et al.*, 2006)

3.3 Physical environment

3.3.1 Geology

Although rainfall is regarded as the primary common source of soil moisture for vegetation, there are some vegetation classes which depend fully on groundwater for growth and health (Eamus, 2009). The geology is divided into Allanridge, Black Reef, Dominion, Granite, Kalahari, Kraaipan, Malmani and Quaternary formations (Figure 3.5) where Quaternary is the least and Granite most evident which forms part of Beaufort Group rocks of the Savanna Supergroup (Johnson *et al.*, 2006). There are two geological formations in the study area, namely Granite and Kraaipan, where the Granite group is the most prominent (Figure 3.5).

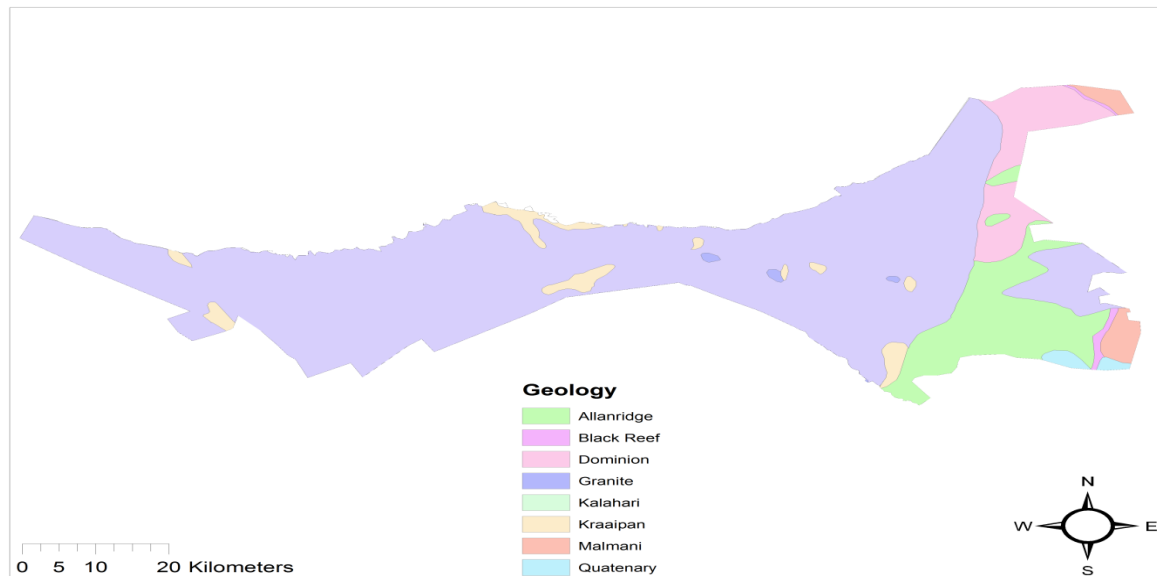


Figure 3.4: Geology map of the Molopo District (Source: Johnson *et al.*, 2006)

3.3.2 Soils

Soil is a mixture of living and non-living organisms upon which most terrestrial life depends (Sampson and Knopf, 1996). Soil structure usually changes gradually with depth and the profile is designated as O, A, B and C horizons (Munsiri *et al.*, 1995). The O, or organic horizon lies at the top of the profile. The most superficial layer of the O horizon is generated from freshly fallen organic matter, including whole leaves, twigs, and other plant parts. The deeper parts of the O horizon consist of highly dis-jointed and partially decomposed organic matter. Fragmentation and disintegration of the organic matter in this horizon are mainly due to the activities of soil organisms, as well as bacteria, fungi and animals ranging from nematodes and mites to burrowing mammals. This horizon is regularly absent in agricultural soils and deserts (Adam, 1984). At its inner levels, the O horizon merges gradually with the A horizon.

The A horizon encompasses a mixture of mineral constituents such as clay, silt and sand and organic material derived from the O horizon (Golchin *et al.*, 1995). The A and O horizons mutually support high levels of biological activity. Burrowing animals, such as earthworms, blend organic matter from the O horizon into the A horizon. The A horizon is generally rich in mineral nutrients. It is progressively leached of aluminum (Al), clays,

iron (Fe), humus and silicate which is partially decomposed organic matter (Berg, 2000). These substances gradually move down from side to side in the soil profile until they precipitate in the B horizon (Walker *et al.*, 1981)

The B horizon contains the clays, humus and other materials that have been transported by water from the A horizon. The deposition of these constituents often gives the B horizon a distinguishing colour and banding pattern (Franzmeier *et al.*, 1983; Lawrence *et al.*, 1995). This horizon is also occupied by the roots of numerous plants. The B horizon steadily assimilates with the C horizon which is the deepest layer of the soil (Imberger and Chiu, 2001). It contains worn parent material which has been worked by the actions of frost, the deeper probing roots of plants and water (Trenhaile, 1987). Weathering bit-by-bit divides the parent material into smaller fragments to yield clay, sand and silt particles (Nye, 1955). Because weathering is fragmentary and less deep than in the A and B horizons, the C horizon may comprise many rock wreckages (Hill and Tedrow, 1961; Van der Meulen and Westfall, 1979). Under the C horizon we find un-weathered parent material, which is often bedrock (Lawrence and Ripple, 1998).

Soil structure is in a constant state of flux as a result of several factors (Lawrence and Ripple, 1998):

- climate
- organisms
- parent material
- time
- topography

Climate affects the rate of weathering of parent material, the rate of leaching of organic–inorganic materials, the rate of erosion and transport of mineral particles and the rate of decomposition of organic material. Climate also impacts the type of vegetation and animals that occupy a region. Topography affects the proportions and direction of water flow and patterns of erosion. Meanwhile, parent constituents such as granite, volcanic rock, wind and water-transported sand, set the stage for all other influences. Soil age affects soil structure (Lees and Ritman, 1991).

Soil is a complex and dynamic entity. It forms the medium in which organisms develop and the activities of those organisms, affect soil structure. Soils in the study area are heterogeneous but are mainly sedimentary (mud – and sandstones) with some distinction when intrusions of igneous rock (doloritic dykes and sheets) result in red soils developing in some areas (De Bruyn, 1998; Slayi *et al.*, 2014). The study area is characterised by Haplic Arisols (Figure 3.4), implying variations in the nutrient amounts available in the soil (Martin and Asner, 2005). Some soils have sufficient nutrient contents while others have minimal. Soils fluctuate from clay in the top soil, of less than 15% deep solonetic soils, originating from the dolerites of the Beaufort Group, to deep loamy soils derived mudstone and sandstone of the Savanna Sequence. This clay content plays a significant role in permeability and it affects the rate at which water moves from the top layer of soil to the lower ones. On steeper slopes, these soil types tend to be shallow, but may also be deep in valley regions (Bredenkamp *et al.*, 1996).

The soils found in this region (Figure 3.4) can be grouped into six broad types (Harmse, 1977):

- (a) Haplic Arisol
- (b) Ferralic Arenosols
- (c) Petric Calcisols
- (d) Chromic Cambisols
- (e) Ferric Luvisols
- (f) Haplic Lixisols

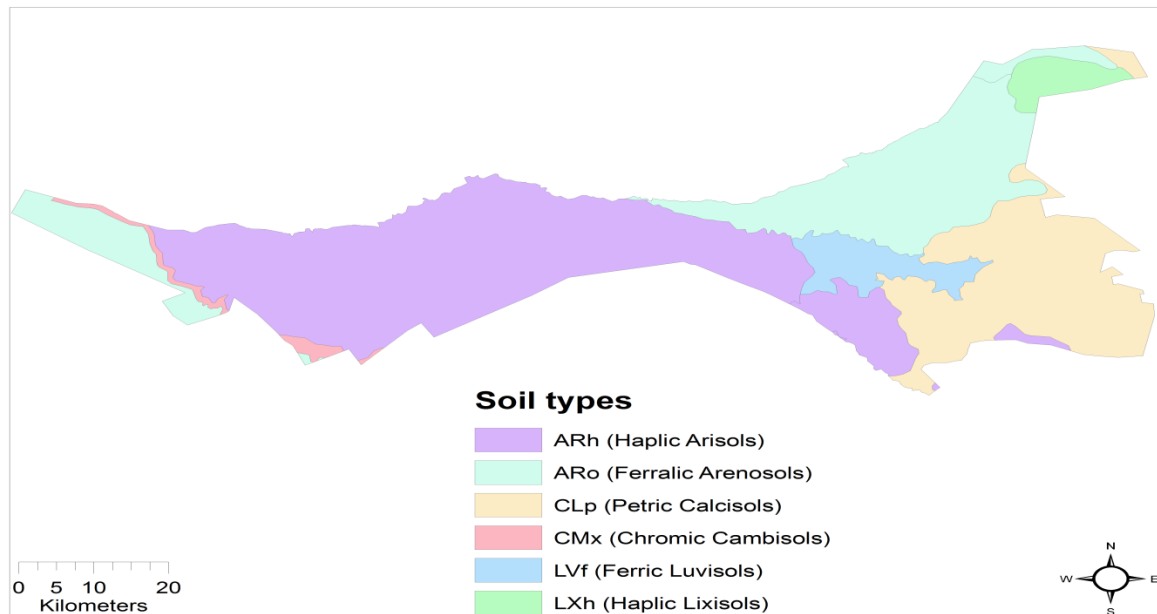


Figure 3.5: Soil types in the Molopo District (Source: Johnson *et al.*, 2006)

Only Haplic Arisols were present in the study area. These soil types are categorised by unconsolidated material or hard rock with a minimum of soil particles. The largest part of the study area is enclosed by non-calcareous well-drained sandy soils, as well as well-drained sandstone mostly of the Hutton and Mispah forms with sequestered rocky outcrops (Shackleton, 2000). The sandy soil is medium to coarse with red B-horizons. This soil is sandy and has a yellow B-horizon. It tends to be more water-logged than the soil higher up the horizons. However, numerous series are illustrious and together form a topo-sequence. This sequence from upper to lower horizons contains meso-trophic and eutrophic Hutton soils, followed by dystrophic and meso-trophic Clovelly soils (Shackleton *et al.*, 2005). The fine-drained soils of the elevated regions of the study sites provide a habitat for many insects which are not found elsewhere (Harmse, 1977; Ullah and Moore, 2009). Plant communities on the elevated sandstone regions developed on deep non-calcareous soils (Shackleton *et al.*, 2005).

3.4 Topography

The western Central Bushveld is found on the extensive, undulating interior plateau of the North West Province. Encircled by the Great Escarpment in the west, south and east, the Plateau gradually loses altitude towards the North West where the Mahikeng Bushveld Basin is found (Wdowinski and Bock, 1994). The vegetation and topography are interrelated to geology, through the nutrients (vegetation) in the soil from the parent rock material through intrinsic geological resistance (topography) to lithospheric-altering agents of erosion, mass wasting and weathering that create various relief features (Munyati and Ratshibvumo, 2009; Munyati *et al.*, 2010).

3.4.1 Evolution of the Mahikeng Bushveld

According to Cowling *et al.* (1997) and Molebatsi (2011), the Mahikeng Bushveld was created by geological events in the Pliocene *via* dropping along marginal faults. The present geomorphology of the western Central Bushveld landscape is closely linked to the geological history of the area (Van der Meulen, 1979). Since the break-up of the Gondwana super-continent in the early Cretaceous period, the African landscape has been denuded and dissected through repeated rotations of erosion, uplift and rifting (Cowling *et al.*, 1997). This created a highly dichotomized landscape with waving topography and persistent occurrence of rugged hills or ‘koppies’ (Nellemann and Fry, 1995).

3.4.2 Land use and history of the area

The land (rangelands) in the Ratlou Municipality area, where the study area is located, is largely communal and is utilised for livestock grazing, with little arable land. Of the minimal arable land available, large portions are abandoned, with the fenced areas remaining as the only cultivated land. The abandoned arable lands have been converted into grazing lands. However, uncontrolled grazing in large parts of the study sites has already caused land deterioration. Although there is electricity in all the villages under study, the villagers are still highly dependent on wood for fuel. Communal land is under the management of a tribal office and the farmers in the area do not own the land. All the farming, political and management decisions are made by the chairman of the tribal office. Traditionally, the farmers are not paid by the chief tribal office, but are allowed to

farm on the land with regard of various terms and conditions (some farmers to donate a cow per annum to the tribal chairman etc.).

Land contains the physical environment including climate, hydrology relief, soils and vegetation to the extent that these influence potential for land use (Zonneveld, 1989; Neameh, 2003). It includes the results of past and present human activities, i.e. vegetation clearance and also adverse results, i.e. soil salinization (Beaulac and Reckhow, 1982).

The study area (three study sites) is located within the communal areas (Figure 3.1) where cattle farming plays an important role. The communal areas occupy about 17% of the total farming area of South Africa and hold 52% of the total cattle population, 72% of the goats and 17% of the sheep (Shackleton *et al.*, 1989; Vetter *et al.*, 2006). They differ markedly from the freehold areas in their production systems, objectives and property rights. Only the cropping areas are normally allocated to individual households, while the grazing areas tend to be shared by members of a community. The communal sector has a substantially higher human population per unit area than the commercial sector and has suffered from lower levels of state intervention (Venter *et al.*, 1989). Investments in infrastructure (access roads, fences, water provision, power supply, dipping facilities) has not kept up with the commercial rangelands (Hoffman and Ashwell, 2001). The production systems in the communal areas are based on pastoralism and agro-pastoralism and the majority of households are subsistence-based and labour intensive, with limited use of technology and external inputs. The outputs and objectives of livestock ownership are much more diverse than in commercial livestock production and include draft power, milk, dung, meat, cash income and capital storage as well as socio-cultural factors. The combination of objectives tends to be met by a policy of herd maximisation rather than turnover, hence even the large herd owners tend to sell only to meet cash needs (Vetter *et al.*, 2006).

Communal area livestock production contributes insignificantly to formal agricultural output and is mainly confined to the eastern and northern part of the country (Hoffman and Ashwell, 2001). However, herd sizes vary considerably between and within regions, and livestock ownership is strongly skewed, with a small number of people owning large herds and the majority owning few animals or none at all.

Stock numbers tend to be less evenly distributed in communal than in commercial areas. There is a tendency for high concentrations of people and livestock near to access roads, towns and infra-structure (schools, clinics, supply stores) and permanent water. Portions of the landscape that are inaccessible (i.e. steep slopes, high mountain plateaux) or far from permanent water remain under-utilised.

Mixed livestock ownership is more common in communal than freehold areas. Cattle are generally the preferred livestock, and are important for draft power, but economic and ecological conditions often limit the possibilities of cattle ownership. Goats and, to a lesser extent, sheep are widely distributed in the communal areas, with a few communities in the high elevation regions of the Eastern Cape focussing on sheep only (Vetter *et al.*, 2006). The pigs and poultry in the communal areas are mainly commercial breeds.

Cattle, sheep and goats are herded during the cropping season in cropping areas, and where there are predator or theft risks in other areas, but herding tends to be relaxed during the dry season during which animals have access to crop residues. In the communal areas of Namaqualand, herd owners have cattle posts away from the village and crop lands, and maintain most of their animals there (Vetter *et al.*, 2006). Pigs and poultry in the communal areas are generally free-ranging and scavenging, although some owners practise housing and feeding.

3.4.3 Land-use

Suitability evaluation involves relating land mapping units to specified types of land-use. The types of use considered are limited to those which appear to be relevant under general physical, economic and social conditions prevailing in an area (Barnes, 1976). These kinds of land-use serve as the subject of land evaluation. They may consist of major kinds of land-use or land utilization types. However, the land-use practices in the study area are agriculture, livestock-grazing and wood supply.

3.4.4 Major types of land use and land utilization types

Livestock farming is a major subdivision of rural land use, such as rain-fed agriculture, irrigated agriculture, grassland, forestry or recreation. The main categories of land-use are

usually considered in land evaluation studies of a qualitative or reconnaissance nature (Department of Agriculture, Conservation, Environment and Tourism, 2002).

Land utilization variations (or farming systems) are described in as much detail and precision as the purpose requires. Detailed or quantitative land evaluation studies, the kinds of land-use considered will usually consist of land utilization types. They are described with as much detail and precision as the purpose requires. Thus, land utilization types refer to any defined use below the level of the major kind of land-use (Midgley *et al.*, 1994).

A land utilization type consists of a set of technical specifications in a given physical, economic and social setting. This may be the current environment or a future setting modified by major land improvement i.e. an irrigation and drainage scheme. Attributes of land utilization types include data or assumptions on (Midgley *et al.*, 1994):

- Produce, including goods (i.e. crops, livestock timber), services (i.e. recreational facilities) or other benefits (i.e. wildlife conservation)
- Market orientation, including whether towards subsistence or commercial production
- Capital intensity
- Labour intensity
- Power sources (i.e. man's labour, draught animals machinery using fuels)
- Technical knowledge and attitudes of land users
- Technology employed (i.e. implements and machinery, fertilizers, livestock breeds, farm transport, methods of timber felling)
- Infrastructure requirements (i.e. sawmills, factories, agricultural advisory services)
- Size and configuration of land holdings, including whether consolidated or fragmented
- Land tenure, the legal or customary manner in which rights to land are held, by individuals or groups
- Income levels, expressed per capita, per unit of production (i.e. farm) or per unit area.

The study area is under communal tenure and livestock farming was remain common but there were no recreational facilities and wildlife conservation.

Management practices on different areas within one land utilization type are not necessarily the same. For example, the land utilization type may consist of mixed farming, with part of the land under arable use and part allocated to grazing. Such differences may arise from variation in the land, from the requirements of the management system, or both.

Some examples of land utilization types are (Johnson *et al.*, 2006):

- Rain-fed annual cropping based on groundnuts with subsistence maize, by small-holders with low capital resources, using cattle drawn farm implements, with high labour intensity, on freehold farms of 5–10 ha.
- Farming similar to the point above in respect of production, capital, labour, power and technology, but farms of 200–500 ha operated on a communal basis.
- Commercial wheat production on large freehold farms, with high capital and low labour intensity, and a high level of mechanization and inputs.
- Extensive cattle ranching, with medium levels of capital and labour intensity, with land held and central services operated by a governmental agency.
- Softwood plantations operated by a government Department of Forestry, with high capital intensity, low labour intensity, and advanced technology.
- A national park for recreation and tourism.

Where it is wished to relate agricultural land utilization types to a general classification, the Typology of World Agriculture of the International Geographical Union may be considered (Scholes and Walker, 1993). The topology discussed above is not static but a change is inevitable with the change of environmental conditions. The modification may change the agricultural characteristics but with the inclusion of a new element the change in agricultural topology is impossible.

3.4.5 Multiple and compound land-use

Two terms, ‘multiple’ and ‘compound’ land utilization types, refer to situations in which more than one type of land-use is practiced within an area. A multiple land utilization type

consists of more than one use simultaneously undertaken on the same area of land, each use having its own inputs, requirements and produce, an example is a timber plantation used simultaneously as a recreational area (Scholes and Walker, 1993). The study area is mainly dominated by woody species (*Senegalia* and *Vachellia* species), which are removed and used as an energy source creating patches of land used for crop (i.e. maize) production and kraal construction. Thus, multiple land–use is prominent in the study area. A compound land utilization type consists of more than one type of use undertaken on areas of land which for purposes of evaluation are treated as a single unit. The multiple and compound land–use may occur in time sequence (i.e. as in crop rotation) or simultaneously on different areas of land within the same organizational unit (Tainton, 1999). Mixed farming involving both arable use and grazing is an example. Land utilization types are defined for the purpose of land evaluation. Their descriptions need not comprise the full range of farm management practices, but only those related to land management and improvement (Ramakgwale, 2006). At detailed levels of evaluation, closely–defined land utilization types can be extended into farming systems by adding other aspects of farm management. Conversely, farming systems that have already been studied and described can be adopted as the basis for land utilization types. Overgrazing and sometimes an inappropriate land utilization type can be identified by the levels of grass layers. However, overgrazing can be avoided with livestock management which combines grazing on uplands in the rainy season and on seasonally flooded lowlands in the dry season. However, this process is not practised in the communal areas compared to the commercial areas.

CHAPTER 4: MATERIALS AND METHODS

4.1. Trial layout

Disaneng Village (Figure 4.1) is a communally managed area in the North West Province, South Africa. It is located in the semi-arid region of the North West Province (NWP), west of the city of Mahikeng (previously Mafikeng) and east of Logageng, in the former Molopo District (Figure 4.1) and is inhabited by Tswana speaking people. The northern part of Disaneng, which is the focus of this study, lies in the geographical position of 25°48'13.02 S, 25°15'39.57 E. The study area comprises a benchmark (Figure 4.1) area and three sites in relation to five other Villages and was chosen because it has similar vegetation to the study sites, namely, Lopurung (25°45'22.50 S, 24°59'24.62 E), Makgori (25°50'01.24 S, 24°47'29.45 E), Tshidilamolomo (25°49'53.26 S, 24°41'04.58 E), Masamane (25°50'41.88 S, 24°37'57.54 E) and Logageng (25°55'57.60 S, 24°41'15.82 E) (Figure 4.1).

The basis of veldt condition assessment is to compare a site with one in excellent condition (in terms of defined management objectives) in the same ecological zone. The first requirement is to define such excellent veldt (Tainton, 1999). The basis of veldt condition assessment is to compare a site with veldt which is in an excellent condition in the same ecological zone. The extent to which the above task is accomplished will greatly affect the robustness of inferences regarding the ecological condition of the study area (Figure 4.1). The benchmark is located in Disaneng Village and is in same region with the study area (Figure 4.1).

Molatlhegi (2008) and Mogodi (2009) made use of two benchmark (BM) sites for comparison purposes each in a communal and commercial area. For this study, the commercially managed benchmark sites in these studies were omitted, as the current research was restricted to communally managed areas.

Comole (2014) made use of three benchmark sites (reference sites), in Lopurung, Makgori and Tshidilamolomo Villages (Refer to Figure 4.1). The Tshidilamolomo benchmark would not be used because the reference site was close to riparian habitats and the recorded dominant species (*Prosopis velutina*) were not found in this study (Figure 4.1).

The study area was located within the Savanna Biome, which extends from Molopo District to Botswana and it is mainly found in other provinces such as Kwa–Zulu Natal, Limpopo and Northern Cape. The Savanna Biome is the largest biome in Africa and also in South Africa (Scholes, 1997), occupying approximately 65% of the continent (Huntley and Walker, 1982). The Savanna Biome represents 32.78% of South Africa (399 600 km²) (Mucina and Rutherford, 2006). This biome extends beyond the tropics to meet the Nama–Karoo Biome on the central plateau. Areas of the Savanna are largely tropical and occupy the greater area of the southern continents (Huntley and Walker, 1982) as well as some parts of the northern continents (Mucina and Rutherford, 2006).

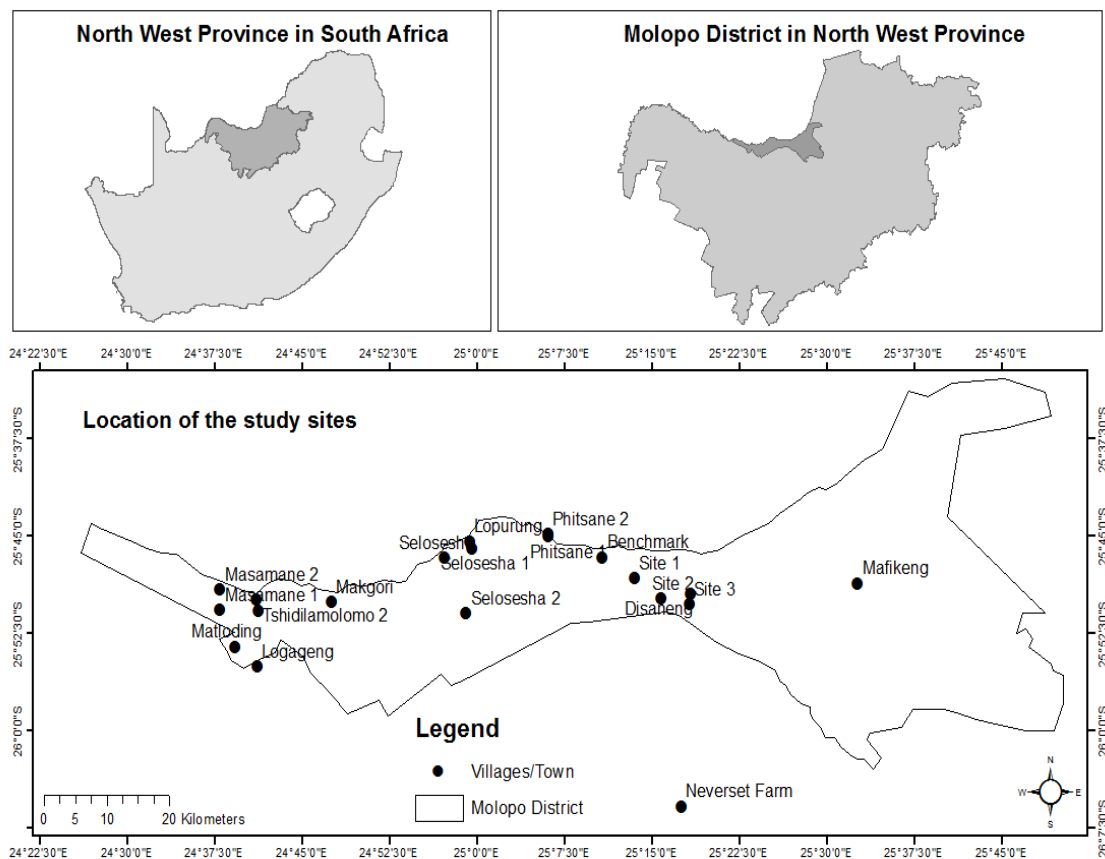


Figure 4.1: Location of the study sites in the Molopo District

This chapter outlines the techniques employed in the achievement of the objectives of this research study. The objectives were accomplished by applying GIS, remote sensing methods and on-site field measurements and observations. Satellite images, taken in 2006,

2010 and 2014, were utilized to analyze vegetation condition trends. The selection of the satellite imagery was influenced by season, availability and quality of images.

4.2 Field observations and data collection procedure

4.2.1 Woody species structure

Woody structure and composition were recorded to provide for calculating per species, stem height increase form and heights class (Figure 4.2) (Coetzee and Gertenbach, 1977). Ranging poles were used to construct a square cross with four equal arms of 25 m lengths, calibrated at 5 m intervals. For each height class, four assessment squares were investigated, one in each of the quadrants delimited by the cross: 5 m x 5 m; 10 m x 10 m; 15 m x 15 m; 20 m x 20 m and 25 m x 25 m (Figure 4.2). All woody species inside the plot were identified and structurally assessed. In this way, the quadrant size for each height class of woody plants was determined.

Further, the square plot coincides with the square shaped remotely sensed image pixels, which enhances computational efficiency: all the woody species' height classes of interest were marked with an aerosol spray in 2014 at a specific height, each with a specific symbol (i.e. a circle for 0.5 m –, a triangle for 1 m –, rectangle for 1.5 m high trees etc.). The following year (2015) all marked trees' heights were measured using a vegetation measuring stick and only the woody species whose height increases (classes range were) more than 500 mm were recorded. This method was used to determine the woody species developments (height increase rates). Woody species were identified and divided according to height classes (0.5, 1, 1.5, 2, 3, 4, 5 and 6 m). Woody plant density (plants ha⁻¹) per species was determined for the different height classes (Trollope, 2011). For each height class, the phytomass (Tree Equivalents (TE) ha⁻¹) was calculated, using the formula, $\text{phytomass} = n \times h \times 1.5^{-1}$ (n, number of plants ha⁻¹; h, mean height of the shrub or tree (m) (Dreber *et al.* 2014). According to Smit (1989), 1 TE is equal to 1 tree of 1.5 metres tall. Thus 2 TE is equal to a 3 metre high tree etc. The total phytomass was calculated as the sum of the phytomass in each height class (Trollope, 2011; Dreber *et al.*, 2014) and expressed in TE ha⁻¹. All densities supplied refer to the phytomass

densities. The data was taken in 2014 (April and May) and 2015 (September and October).

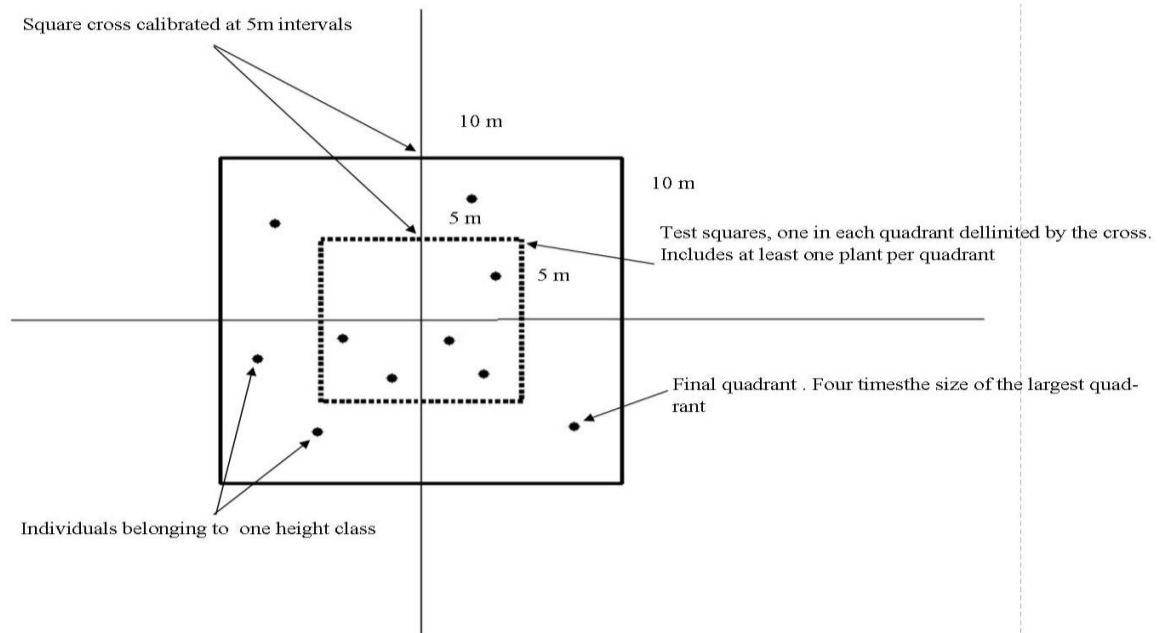


Figure 4.2: Procedure by Coetzee and Gertenbach (1977) for determining quadrant size for a height class, i.e. 1 m high plant. Test squares are enlarged in steps until at least one plant is included

4.2.2 Height increase rate of woody encroachers

According to Carr (1976), *Acacia* trees do not grow at the same rates and will also grow at different rates in different environments. It was also mentioned by Van Wyk and Van Wyk (2013) that the height increase of plant species, especially woody species are determined by its specific habitat and anthropogenic factors affecting it.

Senegalia mellifera shrubs, however, limit their own height increase rate by intense density-dependent inter-shrub competition (Smit, 2003) and often grow very slowly. Joubert (2014) noted that *Senegalia mellifera* saplings increased in height by an average of only 3.19 cm per annum, due to inter species competition.

An insignificant relationship between density and tree size existed for *Senegalia mellifera* trees in the Highland savannas of Namibia was reported by Ngaruka (2011), implying that, tree density does not affect tree size (Ngaruka, 2011). However, since the density of plants does not seem to change as the plant height increases the increased crown diameter leads to crowns overlapping (Ngaruka, 2011). In bush thickening management, thinning mostly targets size classes of 2.5 m and below, where it has been indicated by De Klerk (2004) that only a 10% of their density must be maintained, whereas larger trees are maintained at least at 90% of cover, to create open woodland. Ngaruka (2011) also stated that, on average, most thickets formed are of trees mostly of 2.5m in height, or below (Ngaruka, 2011). Such shrub-sized trees are the most problematic in encroachment since their canopies are likely to spread outwards or overlap to neighboring trees and thus closing off access, mainly for foraging animals. In this light, it was decided to measure the increase in height of woody encroachers in the different height classes as well as to determine the density of woody encroachers per height class.

Woody encroaching species at each height class were marked with an aerosol spray in 2014 at a specific height, each with a symbol (i.e. circle for 0.5m a triangle for 1m and rectangle for 1.5m high trees etc.). In 2015 the marked trees were measured again, using a vegetation measuring stick. Woody species whose heights increased with more than 500 mm were recorded. In this way, the height increase rates of different species of woody plants were measured under natural encroaching conditions.

The densities of all woody plants in different height classes were determined and expressed in tree equivalents per hectare (TE ha⁻¹). The term tree equivalent is widely used in South Africa to express the woody plant population in a common unit (Smit *et al.*, 1999; Hagos and Smit, 2005). A tree equivalent is defined as a 1.5 m tall tree or shrub (Smit *et al.*, 1999; Hagos and Smit, 2005).

Therefore, a plant of 3 m in height represents 2 TE etc. Previous experiments also recommended that plot size ranging from 20 m × 20 m up to 50 m × 50 m be used for woody vegetation studies in Savannas (Demisse, 2006). The 25 m × 25 m (Figure 4.2) plot was considered large enough to represent the surrounding woody plant properties. The total woody plants in each height class were thus calculated separately. The grand total density of all woody species was then determined and expressed in TE ha⁻¹. All

encroaching woody plants were identified. A summary of sampled variables is given in Table 4.1.

Table 4.1: List of field variables sampled

Number	Types of woody vegetation structural traits
1.	Woody species diversity.
2.	Woody species ring marks (using aerosol paint) according to their height class.
3.	Height of individual woody species per quadrant.
4.	Number of woody species (Density per quadrant).

The guidelines adopted by the Department of Agriculture used to describe the extent of bush thickening are shown in Table 4.2, adapted from Moore and Odendaal (1987) and Bothma (1989).

Table 4.2: Indication of the extent of bush thickening (TE.ha⁻¹) (Moore and Odendaal, 1987; Bothma, 1989)

Tree equivalents per hectare (TE/ha)	Extent of bush thickening
200–457	Encroached (grass production declines linearly)
458–714	Highly encroached (grass suppressed further)
715–1099	Severely encroached (more grass suppression)
>2000	Grass height increase almost totally suppressed

4.3 Remote sensing and GIS techniques

The common image processing functions available in image analysis systems can be characterized into the following seven categories:

- Image acquisition
- Importing images

- Geometric corrections
- Pre-processing
- Enhancement
- Classification and Analysis
- Accuracy Assessment

These steps were employed in this study, using ERDAS Imagine 2015 software.

4.3.1 Image acquisition

Three SPOT (*Satellite Pour l'Observation de la Terre*) images (see Table 4.3) were used.

4.3.2 Importing images

The raw images were obtained in a tiff format which cannot be processed in ERDAS Imagine 2015. They were imported using the ERDAS Imagine 2015 software so that they could be converted into a usable format (.img).

Table 4.3: The SPOT images that were used

Image	Date	K/J	Spatial Resolution (m)
SPOT 2	04-09-2006	127-402	20
SPOT 4	21-08-2010	127-402	20
SPOT 5	19-11-2014	127-402	10

4.3.3 Geometric corrections

Geometric rectifications include correcting for distortions (i.e. clouds and smoke) due to sensor-Earth geometric variations and change of the data to real world coordinates (i.e. latitude and longitude) on the Earth's surface. The predictable distortions can be amended by precise modelling of the sensor and platform motion and the geometric relationship of the platform with the terrain. Therefore, to correct other unsystematic or random errors,

geometric registration was implemented on the imagery to a known ground coordinate system (UTM, Zone 35 and WGS84 S).

The geometric registration process had the following steps:

- Recognizing the image coordinates (i.e. row, column) of several clearly discernible points, ground control points (GCPs).
- Root mean square (RMS) error residuals were determined. Residuals are the difference between the actual values and the predicted values.
- Polynomial equations were used to convert the source coordinates to corrected coordinates, by means of first order transformation.
- The coefficients of the polynomials were premeditated by the least square regression technique that helped in relating any point on the map to its corresponding point in the image.
- Out of the three common methods for resampling the nearest neighbour resampling method was used (Lillesand *et al.*, 2008).

4.3.4 Spatial resolution processing

The SPOT 5 image in Table 4.3 had the spatial resolution of 10 m but was degraded to 20 m so that it could be of the same spatial resolution as the SPOT 2 and 4 spatial resolutions. The images were then subset in order to extract the study area in Disaneng.

4.3.5 Enhancement

In order to improve the appearance of vegetation on the imagery to assist in image classification and analysis, the images were enhanced using texture analysis. The Mean Euclidian distance operator was used to enhance the woody plant species.

4.3.6 Classification and analysis

Image classification refers to operations used to digitally detect and classify pixels in the information. Grouping is usually performed on multi-channel data sets and this method assigns each pixel in an image to a specific class or theme based on statistical

characteristics of the pixel brightness values. There is a diversity of approaches taken to perform digital classification. There are generally two forms of classification: supervised and unsupervised. For the purpose of this research, an unsupervised classification was used on the texture enhanced image.

4.3.7 Accuracy assessment

The accuracy of information extraction, by image exploration, measures the agreement between a standard assumed to be correct and a classified image of unknown quality. Precision defines the level of detail found within the classification. It is possible to increase the accuracy of a classification by reducing the amount of detail or by generalizing to broad classes rather than very specific ones (Demisse, 2006). For example, a scheme which generally categorizes *trees* versus *crops* presents less opportunity for classification miscalculation than one that differentiates many types of trees and many types of crops. In this case, lower precision provides the potential for higher precision. However, the user of the map that offers only general classes cannot make accurate statements about any given point on the map.

Classification error occurs when a pixel (or feature) belonging to one category is assigned to another category. *Inaccuracies of omission* occur when a feature is left out of the category being evaluated while *errors of commission* occur when a feature is incorrectly included in the category being evaluated (Babykalpana and ThanushKodi, 2010). An error of omission in one category will be counted as an error in commission in another category.

Accuracy assessment was performed by comparing the map created by remote sensing analysis to a reference map based on a different information source. One might ask why the remote sensing analysis is needed if the reference map to compare it to already exists. One of the primary purposes of accuracy assessment and error analysis in this case is to permit quantitative comparisons of different interpretations. Classifications done from images acquired at different times, classified by different procedures, or produced by different individuals can be evaluated using a pixel-by-pixel, point-by-point comparison. The results must be considered in the context of the application to determine which is the most correct or most useful for a particular purpose.

One simple method of comparison is to calculate the total area assigned to each category in both maps and to compare the overall figures. This type of assessment is called the *non-site-specific-accuracy* (De Fries and Belward, 2000). On the other hand, *site-specific accuracy* is based on a comparison of the two maps at specific locations (i.e. individual pixels in two digital images). In this type of comparison, it is obvious that the degree to which the pixels in one image spatially align with the pixels in the second image contribute to the accuracy assessment result. Errors in classification should be distinguished from errors in registration or positioning of boundaries. Another useful form of site-specific accuracy assessment is to compare field data or training data at a number of locations within the image, similar to the way spatial accuracy assessment using ground checkpoints is performed for digital ortho-photos and terrain models. The second method was used in this study.

Accuracy of image classification is most often reported as a percentage correct. The *consumer's accuracy* (CA) is computed using the number of correctly classified pixels to the total number of pixels assigned to a particular category (Cihlar, 2000). It takes errors of commission into account by telling the consumer that, for all areas identified as category X, a certain percentage are actually correct. The *producer's accuracy* (PA) informs the image analyst of the number of pixels correctly classified in a particular category as a percentage of the total number of pixels actually belonging to that category in the image. Producer's accuracy measures errors of omission.

CHAPTER 5: BUSH DENSITIES AND DEVELOPMENT OF ENCROACHING WOODY SPECIES

5.1 Introduction

5.1.1 Plant growth patterns

The growth rates of woody species varies from the slow, medium and fast category, as indicated by *Grewia flava* (400–500 mm.a⁻¹) to *Ziziphus mucronata* (500–1500 mm.a⁻¹) (Table 5.1).

Table 5.1: Woody species growth rates

Woody species	Growth rate (mm.a ⁻¹)	Growth category
<i>Grewia flava</i>	400–500 (Trollope <i>et al.</i> , 1998)	No info
<i>Senegalia mellifera</i>	300–500 (Carr, 1976)	No info
	319 (Joubert, 2014)	Slow ⁵
<i>Vachellia tortilis</i>	300–400 (Anderson and Krzywinski, 2007)	Slow ^{1,2,3,4}
<i>Ziziphus mucronata</i>	500–1500 (Jacana Enterprise, 2012)	Fast ^{1,2,3}

References on growth category: ¹Coates Palgrave (2000), ²Van Wyk (1990), ³Palmer and Pitman (1973), ⁴Anderson and Krzywinski (2007), ⁵Joubert (2014)

5.1.2 Fast growing woody species

Fast growing woody species allocate relatively small portions of their resources to defence (Hogberg, 1986) and rely upon the ability to outgrow invaders. Yet, as these trees grow large, growth rate declines and they can no longer add more tissue than decay organisms remove. At this point, structural failure becomes likely (Rasmussen *et al.*, 2011). The fast growing woody plant in the study area was *Ziziphus mucronata*, however, there was no information about *Grewia flava* and *Senegalia mellifera* while *Vachellia tortilis* was grouped in the ‘slow’ growth category (Table 5.1).

5.1.3 Woody plants with slow height increase rates

Ziziphus mucronata, allocate a high proportion of resources to defence and these trees tend to be long-lived (Mace *et al.*, 2003). According to Carr (1976), most drought restricted thorn trees have a slow growth rate, but can grow fast if the environmental conditions are favourable.

5.1.4 Tree characteristics

The genetic characteristics of a tree determine its ability to tolerate shade and competition from other plants and will influence how well it will grow in different environments (Herrick *et al.*, 2006). However, as individuals are quite similar in their resource requirements, such competition may be particularly intense, intra-specific competition is a major force in ecology and is responsible for phenomena such as dispersal and territoriality, as well as being the primary cause of population regulation *via* density-dependent processes (Tainton, 1999; Mackenzie *et al.*, 2001). If the crown is narrow, the tree will grow slowly, whereas when the crown is large or wide, there will be much wood in the limbs and little in the more usable main stem (O'Connor and Crow, 1999).

Tree species differ in their tolerance for shade and competition and are classified as very tolerant, tolerant, intermediate, intolerant, and very intolerant to shade (Dyksterhuis, 1949). Some species tolerate more shade as seedlings than as large trees. Species that are very tolerant will reproduce and grow in deep shade beneath a dense canopy. Species that are very intolerant will survive only if their seeds sprout in wide openings that receive direct sunlight (O'Connor and Crow, 1999). The size of canopy openings, ranging from the loss of a single tree to removal of many acres of trees, greatly influences which tree species will reproduce and survive (Williams *et al.*, 1987). A species such as *Senegalia mellifera* can be classified as shade tolerant and does readily grow beneath each other and other woody species (Hagos and Smit, 2005). This implies that high (less than 2 m tall), *Senegalia mellifera* trees will rather expand than to develop beneath other trees. However, juvenines are effective to establish beneath higher trees (Smit, 2003).

5.1.5 Effects of site characteristics

Site characteristics that affect tree growth include soil depth, texture, moisture, fertility, pH and topography (Glen, 2004). On the whole, deep soils are better suited for tree growth than shallow soils because they potentially have a greater nutrient supply and water-holding capacity. Sandy

soils generally have a great leaching capacity and generally lead to nutrient and water leaching in semi-arid areas (Molatlhegi, 2008). Rooting depth may be restricted by bedrock, coarse gravel, a hardpan layer or excess soil moisture. Tree roots that absorb most nutrients and water usually are found in the top 600 mm of the soil profile (Hassan and Styles, 1990).

Soil texture refers to the relative proportions of sand, silt and clay in a mass of soil. Soils with a high percentage of sand have large pore spaces between soil particles (Lidicker, 1975). Woody plants absorb and drain water fast and they are dry unless there is a shallow water table. Clay soils have a large water-holding capacity, but they absorb water slowly and water adheres so tightly to the soil particles that much of it is unavailable for plant use (Huntly and Inouye, 1988). Soils with a high percentage of silt have the most favourable texture for moisture absorption and drainage. Although soil analyses did not form part of this study the soil in the study sites were sandy. Mogodi (2009) reported that sites in the nearby Selosesha (Selosesha 1 and Selosesha 2, indicated on Figure 4.1) and Disaneng (Refer to Figure 4.1) had sandy soils (> 70% sand).

Soil fertility is based largely on the type of parent material from which the soil originated. Some of the most fertile soils originated from limestone, shale, and windblown deposits, whereas some of the least fertile soils originated from sandstone and granite (Barnard and Ischer, 1972). On the whole, fine-textured (clay) and medium-textured (silt) soils have a greater nutrient supply than coarse-textured (sandy) soils (Lidicker, 1975). Most tree species grow well when the soil remains moist much of the year, but only a few species tolerate very dry or very wet conditions for long periods (Moleele *et al.*, 2002). Soil may be too dry for normal tree growth where the soil is sandy, rocky, or shallow while soil may be too wet where the soil is clay and the area has high rainfall or groundwater close to the surface (Barnard and Ischer, 1972).

Most tree species grow best in a slightly acid soil, but the preferred pH varies depending on the species (Barnard, and Ischer, 1972). The pH of soil is one of the most important properties involved in plant growth (Foth, 1990, Sangha *et al.*, 2005; Blaser *et al.*, 2014). Soil pH affects absorption of minerals by plant roots and solubility of mineral nutrients is greatly affected by soil pH (Tisdale *et al.*, 1993; Kidd and Proctor, 2000; Maestre *et al.*, 2009). Acidic soils (pH 4.0 – 5.0) can have high concentrations of soluble aluminium (Al^{3+}) and manganese (Mn^{3+}) which may be toxic to plant growth (Nweke and Nsoanya, 2013). Moreover, Miller and Donachue (1992) and Tisdale *et al.* (1993) indicated that, a pH range of approximately 6–7 promotes the most readily available plant nutrients, while a pH above 7 (alkaline) reduces the ability of plants to absorb nutrients such as manganese, boron and other trace elements.

Hagos and Smit (2005) indicated that soil in sites encroached by *S. mellifera* recorded a lower ($P > 0.05$) pH than in the open (non-encroached area). Likewise, Murovhi and Materchera (2006) recorded a significantly lower ($P < 0.05$) soil pH under *Vachellia erioloba* (Previously *Acacia erioloba*) trees than in the open. Elridge *et al.* (2011), however, did not find any difference between soil pH in encroached sites, compared to non-encroached sites. The exact reasons for these differences regarding the influence of tree canopies on soil pH are not known, however, based on the positive association between increases in exchangeable cations and soil pH (higher base saturation), a higher pH under canopies of savanna trees conforms more logically to the higher content of exchangeable cations in the sub-habitat (Hagos and Smit, 2005).

5.1.5.1 Topography

Topography (graphic representation of the surface features of a region on a map, indicating their relative positions and elevations) affects tree growth largely because of its influence on soil depth, moisture availability, and sunlight exposure. Because gravity pulls soil particles and water downhill, soil depth, nutrient supply, and water supply usually are greater on lowlands, hillside benches, and coves than on steep hills (Midgley *et al.*, 1994). For similar reasons concave slopes are better for tree growth than convex slopes.

5.1.6 Effects of climate

Trees are genetically programmed to grow within certain climatic conditions. The length of the frost-free growing season, temperature extremes, precipitation amounts and the duration of droughts are a few components of climate that influence tree growth (Midgley and Bond, 2001). A single tree species also varies in its climatic requirements across its natural range. For example, *Populus tremuloides* (quaking aspen) naturally occurs from northern Alaska to northern Illinois but a quaking aspen transplanted from Illinois will not flourish in Alaska (Nielsen, 1986). Native trees have evolved and adapted to specific climates.

Tropical grasslands, savannas and forests (the savanna complex), are expected to respond sensitively to climate and atmospheric changes. This is because the main players, such as grasses and trees, differ fundamentally in their response to temperature, carbon dioxide supply and fire and are in an unrelenting struggle for the dominance of the savanna complex (Virginia and Delwiche, 1982). The outcome of this struggle determines whether vast portions of the globe's tropical and sub-tropical regions are covered with grasslands, savannas or forests.

Experimental studies have generally shown that plants do not show a large response to CO₂ fertilization (Sankaran *et al.*, 2005). CO₂ fertilization means the larger amount of carbon dioxide in the atmosphere that has resulted from rising anthropogenic emissions induce the growth of plants, which use carbon dioxide during photosynthesis. However, most of these studies were conducted in northern ecosystems or on commercially important species (Roques *et al.*, 2001).

5.2 The growth of trees in the Savanna

5.2.1 The general growth of trees

The cumulative pattern of tree growth is well described by a sigmoid model (Zeide, 1993). The Sigmoid model represents a pattern of growth in which, in a new environment, the population density of an organism increases slowly initially, in a positive acceleration phase, then increases rapidly, approaching an exponential growth rate as in the J-shaped curve, but then declines in a negative acceleration phase until at zero growth rate the population stabilizes (Belsky and Amundson, 1992). This decline reflects increasing environmental resistance which becomes proportionately more important at higher population densities. This type of population growth is termed density-dependent, since growth rate depends on the numbers present in the population (Holdo and Timberlake, 2008). Each stand in the Savanna has an ecological trajectory, a natural pathway for change which is heavily influenced by site characteristics. This pathway can be altered and the species composition of a stand changed *via* natural disasters (such as a fire or windstorm) or human activities (such as harvesting, thinning, or introducing invasive exotic species) (O'Connor, 1991). Each tree species has a natural life span that may be shortened on a poor site or lengthened on a good site. Competition for nutrients, moisture, and sunlight leads to the death of individual trees. Fire, wind, insects, and disease can weaken and eventually kill trees. Woody species found in the study area, included *Grewia flava*, *Senegalia mellifera*, *Vachellia tortilis* and *Ziziphus mucronata*, all having different growth rates (Refer to Table 5.1).

5.2.2 Growth of local encroaching woody species

Senegalia mellifera (Black thorn), a drought resistant and aggressive encroacher, has a moderate growth rate of 300–500 mm a⁻¹ (Table 5.1) and it is a commonly occurring shrub on rangelands throughout the savanna in Western, Eastern and Southern Africa (Milton, 1986). Joubert (2014), however, recorded a slow growth rate of only 319 mm a⁻¹ in the Highland savanna of Namibia. Its terrain preference is rocky hillsides with rainfall along seasonal watercourses, mixed with other trees. These soils are typically more than 300 mm in depth, with a very high schist and quartz gravel content (Joubert, 2014). If left unattended, especially if grazing is heavy and no fires check it, its spread may form dense, impenetrable thickets higher than 3 metres and sometimes hundreds of metres across, slowly taking over good grazing land (Hassan and Styles, 1990). It may grow on a variety of soil types, ranging from Kalahari sands to heavy, clayey soils. In sandy areas, it often tends to be associated with more calcium-rich soils on drainage lines (Hagos and Smit, 2005). It is usually found in arid areas where it is well adapted to dry conditions with its shallow, wide spreading root system, additional to having a deep taproot. In South Africa, *Senegalia mellifera* is found widely distributed in the drier western parts. It also occurs in Angola, Namibia and Botswana, extending northwards to Tanzania (Smit, 1999).

Invasion by *Senegalia mellifera* in the savanna is an indicator of overgrazing (Coates Palgrave, 1990) and is characterised by a concentration of a number of plants in clumps or thickets, which appear to be contagiously distributed (Coates Palgrave, 1990). In the initial stages of invasion, these thickets generally consist of one or more large plants surrounded by younger plants of two or more height classes (Donaldson, 1967; Coates Palgrave, 1990). *Acacia mellifera* subsp. *detinens* occurs as a multi-stemmed shrub of up to 3 m high, or occasionally a tree that can grow to a height of 7 m (Coates Palgrave, 1990; Smit, 1999). It has a spreading, rounded to flattened crown which may reach down to the ground level. As a tree, it usually branches low down with a substantial horizontal spread that can well exceed its height (Smit, 1999). These trees spread very rapidly, both from seed and vegetatively so much so that they can be a menace, forming impenetrable, tangled thickets (Coates Palgrave, 1990).

Vachellia tortilis (Umbrella thorn) occurs on rocky outcrops and alluvial valley bottoms, avoiding seasonally waterlogged sites. The umbrella thorn grows in areas with annual rainfall as low as 40 mm and as high as 1 200 mm, with dry seasons of 1–12 months (Kraaij and Novellie, 2010). The tree favours alkaline soils but will also colonize saline and gypseous soils. It is fairly slow growing and in cultivation reaches a final height of between 3 and 5 m, with a lateral spread of 8–13 m (Smit, 1999). In extremely arid conditions, it may occur as a small, wiry bush. This is a widely distributed species and occurs in Botswana, Namibia, Swaziland (Venter and Venter, 1996). The plant is known to tolerate high alkalinity, drought, high temperatures, sandy and stony soils, strongly sloped rooting surfaces and sand blasting. Plants older than two years have been observed to be somewhat frost resistant (Smit *et al.*, 1993). It occurs in deciduous woodland, thornveld and bushveld. It is found from sand dunes and rocky scraps to alluvial valley bottoms, avoiding seasonally waterlogged sites. *Vachellia tortilis* grows in areas with annual rainfall as low as 40 mm per annum and as high as much as 1 200 mm a⁻¹ with dry seasons of 1–12 months with the medium *growth rate* of 500–600 mm per year, but it can be speeded up by cutting away the lower branches (Carr, 1976). The tree favours alkaline soils but will colonize saline and gypseous soils (Milton and Hoffman, 1994).

Grewia flava (Raisin bush) grows at a rate of 400–500 mm a⁻¹ (Trollope *et al.*, 1998). It is found throughout the North West Province and Northern Cape in South Africa, the South Western areas of Zimbabwe, in Namibia, Botswana and as far north as Zambia (Wiegand *et al.*, 2006). This rather untidy multi-stemmed, low growing shrubby plant is easy to find with its distinctive greyish green hairy leaves. The underside of the leaves has three prominent veins beginning at the leaf base. Another distinctive feature is that the leaves grow upright or horizontal and do not droop downwards (Wiegand *et al.*, 2006).

Ziziphus mucronata (Buffalo thorn) has a slow growth rate of between 4–6 m in four years (De Klerk, 2004). It is a hardy tree which is drought resistant and also deciduous tree growing up to 17 m in height (Hudak and Weissman, 1998). The buffalo thorn is distributed throughout the summer rainfall areas of sub-Saharan Africa, extending from South Africa northwards to Ethiopia and Arabia. *Ziziphus mucronata* grows in areas dominated by thorny vegetation in both temperate and tropical climates (Carr, 1994). It is also found in a wide range of habitats such as woodlands, open scrubland, on rocky koppies, open grasslands, on a variety of soils along streams, nutrient-rich valley bottoms

and forest margins (Glen, 2004). It reaches its largest size on the margins of scrub forest and on deep, alluvial soils near water. Its presence is said to indicate the presence of underground water.

5.3 Data analysis and discussion

5.3.1 Data analysis

The woody plant densities for each height class were determined using MS Excel software showing the woody plant height classes of *Grewia flava*, *Senegalia mellifera*, *Vachellia tortilis* and *Ziziphus mucronata*. The relative change in woody plants was compared to the satellite image values which were then correlated using ANOVA. The woody plant development (growth rate) from 2014 and 2015 were recorded so that the difference in growth rates could be analysed.

Findings of the density of woody plants in this research were compared to findings by Molatlhegi (2008), Mogodi (2009), and Comole (2014) who did similar studies in the Molopo Area, except for the determination of the growth rates of the woody encroachers.

5.3.2 Woody plant density in Disaneng Village (Benchmark)

The site revealed a few large scattered woody species and a layer of *Themeda triandra* (Red grass), *Bothriochloa insculpta* and *Urochloa panicoides*. *Themeda triandra* is an indicator of the veld being in a good condition (Leistner, 2000) and is also known to be resistant to fire and the resistance increases when burnt regularly, but only if allowed to recover after fire and if overgrazing does not occur. This grass is not favoured by animals because, although it has good leaf production, the aromatic taste deters animals (FAO, 1999).

It colonises open areas quickly and is, therefore, useful for controlling erosion, but is a strongly competitive subclimax grass and an indicator of disturbance (Milton *et al.*, 1994). *Urochloa panicoides*, is a fodder grass originating in southern Africa, is a grass of

disturbed ground, such as cropping, overgrazed pastures, roadsides, gardens and paths (Zacharias, 1990) and is also used as an indicator of habitat disturbance in any given area (Okello and Young, 2000). It has poor drought tolerance and grows most vigorously during wet summers.

Senegalia mellifera, *Vachellia tortilis* and *Ziziphus mucronata* were the only woody species present in this site with densities of 165 –, 120 – and 127 TE ha⁻¹ respectively. The total woody species density was recorded as 412 TE ha⁻¹ (Figure 5.1).

This corresponds (species composition) with the results of Molatlhegi (2008) at the benchmark site (Table 5.2) in Disaneng recorded at 1 007 TE ha⁻¹, with *Vachellia tortilis* and *Ziziphus mucronata* at densities of 100 – and 75 TE ha⁻¹ respectively. Mogodi (2009), however, recorded a woody plant density of 407 TE ha⁻¹ (Table 5.2).

Neither Molatlhegi (2008) nor Mogodi (2009) recorded woody plant densities at different height classes, but only reported on the total woody density in their respective reference sites.

Table 5.2: Densities of woody plants in benchmarks of other researchers

Researcher	Location	Total woody plant density (TE ha ⁻¹)	
		Communal area	Commercial Farm
Molatlhegi (2008)	Disaneng	1 007	
Mogodi (2009)	Disaneng	407	
	Neverset Farm		2 243
Comole (2014)	Lopurung	1 077	
	Makgori	297	

The woody species composition within this reference site closely correspond to the reference sites surveyed by Mogodi (2009) and Comole (2014) at Makgori Village (Table

5.2). The total woody plant density in the benchmark site found by Mogodi (2009) added up to 407 TE ha⁻¹, (benchmark site) while the reference site in Makgori added up to a total woody plant density of 297 TE ha⁻¹ (Comole 2014). Similar densities for *Senegalia mellifera* were recorded in this benchmark site (165 TE ha⁻¹, Table 4.3) as compared to the density of *S. mellifera*, recorded by Comole (2014) at 201 TE ha⁻¹.

Senegalia mellifera is an aggressive encroacher and was present in this reference site in all the height classes (Figure 5.1). *Grewia flava* was absent in the benchmark sites of Molatlhegi (2008) and Mogodi (2009). Figure 5.2 shows that 1 m high *Ziziphus mucronata* and 2.5 m high *Senegalia mellifera* trees were the most prominent in this site.

The woody plants are imposed by the interactive effects of fire, herbivory and competition with grasses while the tree is still within the grass layer (De Klerk, 2003). Plant competition starts when any of the resources (water, soil, nutrients or sunlight) become available in insufficient amounts for individuals, and as stated by Archibold (1995), the most successful competitors become the common members of the community. The domination of some savanna communities by problematic woody species (i.e. *Senegalia mellifera*) may have been favoured by the present conditions. The growth and subsequent size that any individual tree can achieve is dependent on the resources to which it has access (Johnston, 1972). This, in turn, is a function of available water and nutrient levels but also of the proximity and size of neighbouring individuals. The larger the tree, the larger the area (volume) of resource depletion around it, and the greater the competitive effect it may have on its neighbours (Smith and Goodman, 1986). As individuals die, due to competition or to factors extrinsic to the population such as fire, herbivory, drought, or even old age, the resulting gaps will allow either the increased growth of neighbouring individuals or the establishment of new individuals (De Klerk, 2004). According to Tainton (1999), the distance between a tree and its nearest neighbour of the same type is not determined purely by chance, but that tree spacing is evenly distributed. The larger the individual, the greater is the distance between it and the nearest individual of the same type. Spacing in *Acacia* communities has been influenced by roles of inter- and intra-species competition (Smith and Walker, 1983).

Woody plant densities in the text refer to phytomass density and are expressed in TE ha⁻¹ (Dreber *et al.*, 2013). Table 5.3 illustrates the densities of woody species from 2014 and 2015. The recorded woody species showed different densities due to the different growth rates of each woody species (Table 5.1).

Senegalia mellifera was found at height classes 0.5 – , 1 – , 1.5 – and 2 m in 2014 and was recorded at densities of 60 – , 37 – , 22 – and 70 TE ha⁻¹ respectively and developed (increased in height) to 1 – , 1.5 – , 2 – and 2.5 m high trees at respective densities of 55 – , 29 – , 10 – and 59 TE ha⁻¹ in 2015 (Table 5.3). The recorded densities for *Senegalia mellifera* in 2014 and 2015 were 189 – and 153 TE ha⁻¹ respectively (Table 5.3). Similarly, *Vachellia tortilis* was found at height classes and 1 m at densities of 80 and 55 TE ha⁻¹ in 2014 and advanced to 1 – and 1.5 m high trees at densities of 60 – and 43 TE ha⁻¹ in 2015 (Table 5.3). *Vachellia tortilis* was recorded at a total density of 140 – and 98 TE ha⁻¹ in 2014 and 2015 respectively. *Ziziphus mucronata* (0.5 – and 1 m high trees) was present at densities of 98 – and 30 TE ha⁻¹ in 2014 and developed to 1 – and 1.5 m high trees in 2015 at densities of 90 – and 26 TE ha⁻¹ (Table 5.3). *Ziziphus mucronata* was recorded at a total density of 131 – and 120 TE ha⁻¹ in 2014 and 2015 respectively. Similar growth patterns were observed for *Vachellia tortilis* and *Ziziphus mucronata* for the same period (Table 5.3). The maximum heights of *Vachellia tortilis* and *Ziziphus mucronata* observed in 2015 was 1.5 m, indicating that the wood of larger *V. tortilis* and *Z. mucronata* was utilized by the local inhabitants for kraal building and fencing purposes.

Table 5.3 illustrates the growth rates of *S. mellifera*, *V. tortilis* and *Z. mucronata* in different height classes from 2014 to 2015. It was taken into consideration that the woody species grow at different rates (Table 5.1), implying that for example a 0.5 m *Senegalia mellifera* tree in 2014 will now be a tree in the height class (0.75–1m) in 2015. Phytomass densities (TE ha⁻¹) for 0.5 m high *S. mellifera* trees in 2014 will thus now be reflecting densities of 0.75 – 1m *S. mellifera* trees in 2015. Similar results were obtained for *V. tortilis* and *Z. mucronata* and are presented in Table 5.3.

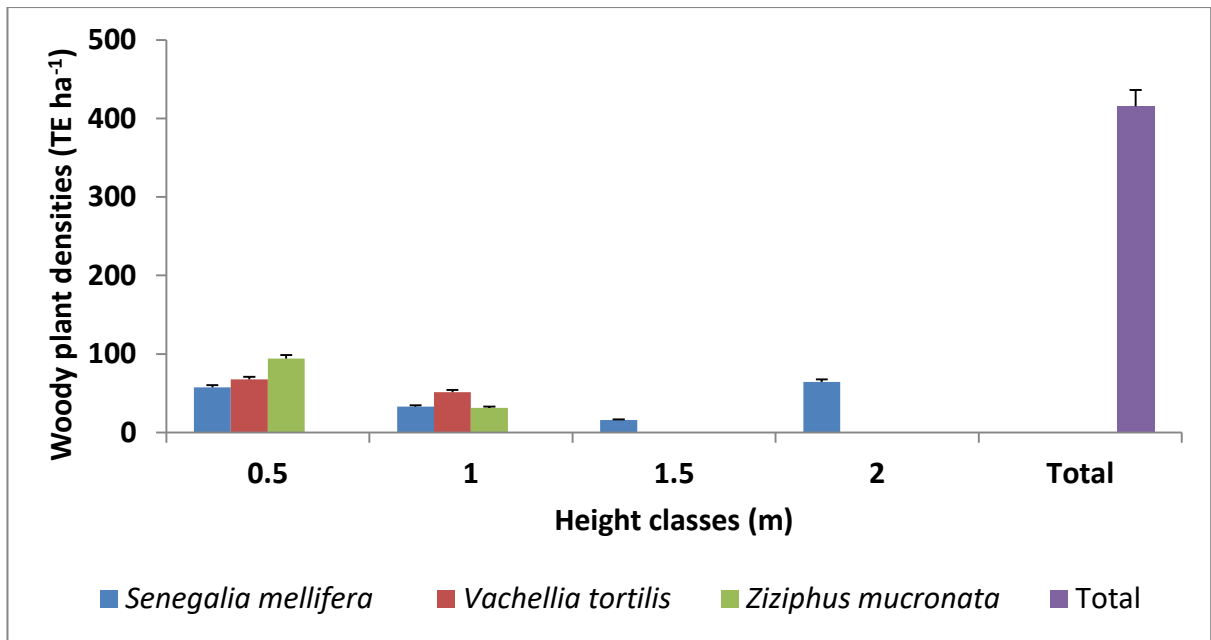


Figure 5.1: Woody species densities in the reference site (2014)

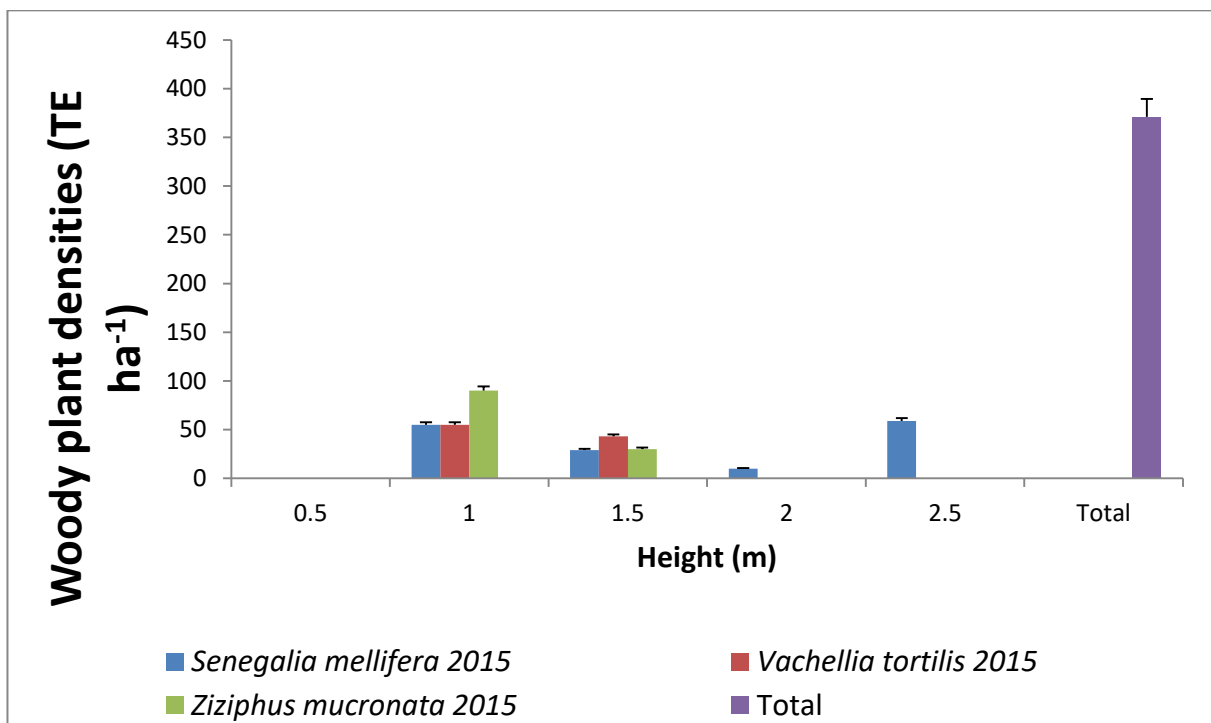


Figure 5.2: Woody species densities per height class in the reference site (2015)

Note that in Table 5.3, trees of a particular species (*Senegalia mellifera*, *Vachellia tortilis* and *Ziziphus mucronata*) in a specific height class in 2014 and in the next height class in 2015 were indicated by the same colour code (Table 5.3).

Table 5.3: The growth of woody species per height class in the benchmark site (2014–2015)

Height class (m)	<i>Senegalia mellifera</i> 2014 (TE ha ⁻¹)	<i>Senegalia mellifera</i> 2015 (TE ha ⁻¹)	<i>Vachellia tortilis</i> 2014 (TE ha ⁻¹)	<i>Vachellia tortilis</i> 2015 (TE ha ⁻¹)	<i>Ziziphus mucronata</i> 2014 (TE ha ⁻¹)	<i>Ziziphus mucronata</i> 2015 (TE ha ⁻¹)
0–0.5	60		80		98	
0.5–0.75				55		
0.75–1	37	55	60		33	
1–1.25				43		90
1.25–1.5	22	29				
1.5–1.75						30
1.75–2	70	10				
2–2.25						
2.25–2.5		59				
Total:	189	153	140	98	131	120

5.3.3 Woody plant densities in the study sites

Similar studies were conducted by Molathlegi (2008), Mogodi (2009) and Comole (2014) in the Molopo Area. The woody plant densities (phytomass) are presented in Table 5.4.

5.3.3.1 Study Site I (Disaneng I)

Disaneng I (Figure 4.1) was near the main road from Mahikeng to Tshidilamolomo and it was easily accessible to livestock and was a browsing–grazing area for cattle, sheep and donkeys. This site (Refer to Figure 4.1) had dense woody species and a very limited grass layer of *Cynodon dactylon* and *Urochloa panicoides*. *Grewia flava*, *Senegalia mellifera*, *Vachellia tortilis* and *Ziziphus mucronata* were the only woody plants present and occurred at densities of 3 388 –, 3 907 –, 1 699 – and 646 TE ha⁻¹ respectively (Figure 5.4). The total woody species density for this site was recorded at 9 430 TE ha⁻¹ (Figure 5.3). *Grewia flava* was recorded in height classes of 0.5 –, 1 –, 1.5 –, 2 – and 2.5 m while *Senegalia mellifera* was present in all height classes except 3.5 m. One, 2 – and 3 m high *Vachellia tortilis* trees were present, whereas *Ziziphus mucronata* occurred in the 0.5 –, 1 – and 3 m height classes (Figure 5.3).

According to Molatlhegi (2008), *Senegalia mellifera* (2 314 TE ha⁻¹) was the most abundant woody species in Disaneng I (Table 5.4) and occurred at a density of 6 667 TE ha⁻¹ in Selosesha. The total woody plant density recorded by Molatlhegi (2008) in the bench mark site was 1 007 TE ha⁻¹ (Table 5.2). Mogodi (2009), however, recorded a woody plant density of 407 TE ha⁻¹ in the communal area of Disaneng and a density of 2 243 TE ha⁻¹ in the commercial area (Table 5.4). This, however, confirms that woody plant density in communal areas is not always higher than in the commercial areas.

Mogodi (2008), recorded *Senegalia mellifera* at a density of 1 886 TE ha⁻¹ in Disaneng, 932 TE ha⁻¹ in Setlhabaneng, 2024 TE ha⁻¹ in Logageng, 6967 in Selosesha and 12649 TE ha⁻¹ in Neverset (Table 5.4). The highest density of *Senegalia mellifera* noted by Comole (2014) was 3447 TE ha⁻¹. Comole (2014) recorded *Grewia flava* at a density of 1 535 TE ha⁻¹ in Lopurung and 2087 TE ha⁻¹ in Makgori while *Vachellia tortilis* was present at a density of 1 693 TE ha⁻¹ in Lopurung. Refer to Figure 3.2 (Chapter 3) for the locations of the various sites.

The total woody species density within Disaneng I showed an increase compared to sites surveyed by Molatlhegi (2008), Mogodi (2009) and Comole (2014). The total woody plant density in Disaneng in this study site was 9 430 TE ha⁻¹, implying that grass growth was virtually totally suppressed (Table 4.3) and was substantially higher than the total

woody plant density of 412 TE ha⁻¹ (Figure 5.1) in the benchmark site. Large *Senegalia mellifera* trees (higher than 6 m) were more evident in 2015 (Figure 5.4) as compared to 2014 (Figure 5.3), a clear indication that large trees are substantially contributing to phytomass density.

Table 5.4: Woody plants densities of *Senegalia mellifera*, *Grewia flava*, *Vachellia tortilis* and *Ziziphus mucronata* in the study sites of other authors, conducting studies in the Molopo Area (* indicates a commercially managed area). Locations of all the sites are indicated on Figure 4.1 (Chapter 4).

Author	Year	Location	Woody species	Phytomass Density (TE ha ⁻¹)
Molatlhegi	2008	Disaneng 1	<i>Senegalia mellifera</i>	2 314
		Phitsane 2	<i>Grewia flava</i>	250
		Seloshesha 1	<i>Senegalia mellifera</i>	667
Mogodi	2009	Disaneng 1	<i>Grewia flava</i>	178
			<i>Senegalia mellifera</i>	1 886
			<i>Ziziphus mucronata</i>	54
		Logageng	<i>Senegalia mellifera</i>	2 024
		*Neverset Farm	<i>Senegalia mellifera</i>	12 649
			<i>Ziziphus mucronata</i>	430
		Seloshesha	<i>Senegalia mellifera</i>	6 967
		Setlhabaneng	<i>Grewia flava</i>	248
			<i>Vachellia tortilis</i>	932
Comole	2014	Lopurung 1	<i>Grewia flava</i>	1 535
			<i>Senegalia mellifera</i>	1 839
		Lopurung 2	<i>Vachellia tortilis</i>	1 693
		Lopurung 3	<i>Senegalia mellifera</i>	3 008
		Makgori 1	<i>Senegalia mellifera</i>	3 447
		Makgori 2	<i>Grewia flava</i>	2 087
			<i>Senegalia mellifera</i>	2 143
		Makgori 3	<i>Senegalia mellifera</i>	3 029

Nutrients, such as nitrates, phosphorus, a series of anions and cations and various trace elements, are essential to plant nutrition (Hagos and Smit, 2005) and act as determinants of the composition,

structure and productivity of vegetation. While the base richness of the parent material is initially important in determining soil fertility, biological activities are important in the creation and maintenance of localized areas of enhanced soil fertility, often on base-poor substrates (Hagos and Smit, 2005). Trees may act as such a biological agent, creating islands that differ from those in the open. The results of this study confirm this to be true in the case of *Senegalia mellifera*.

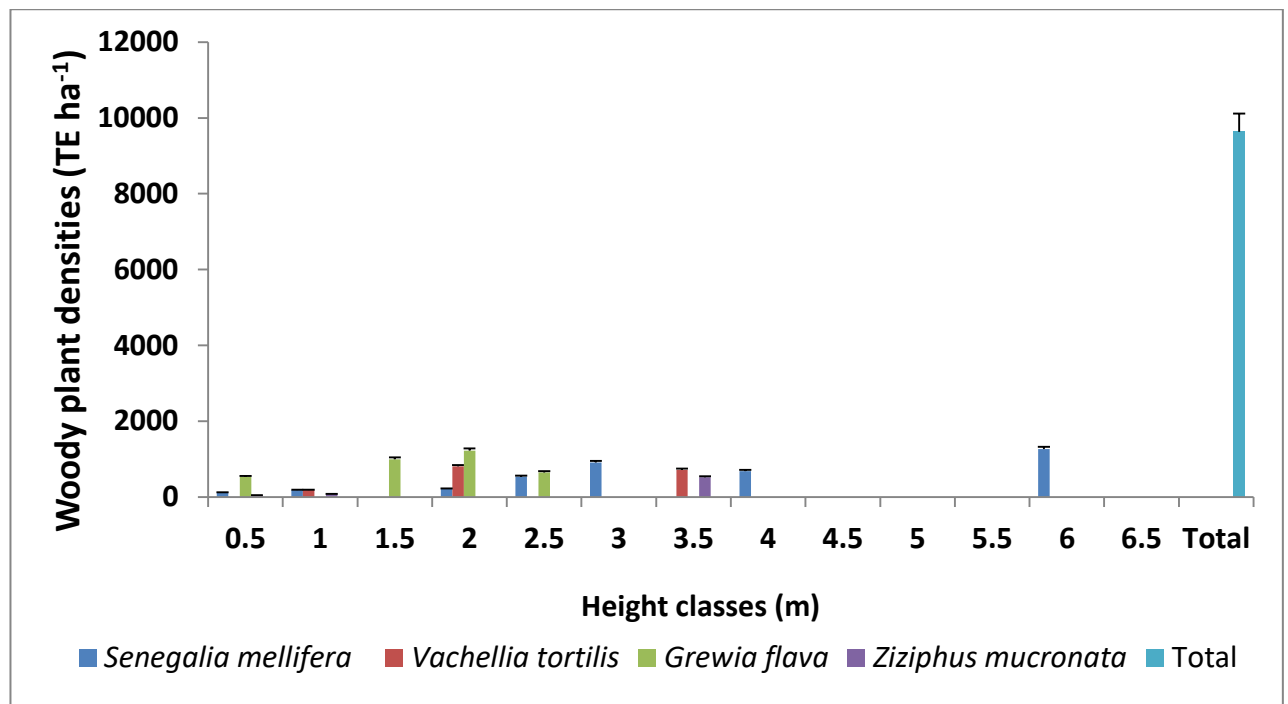


Figure 5.3: Woody species densities in Disaneng I (2014)

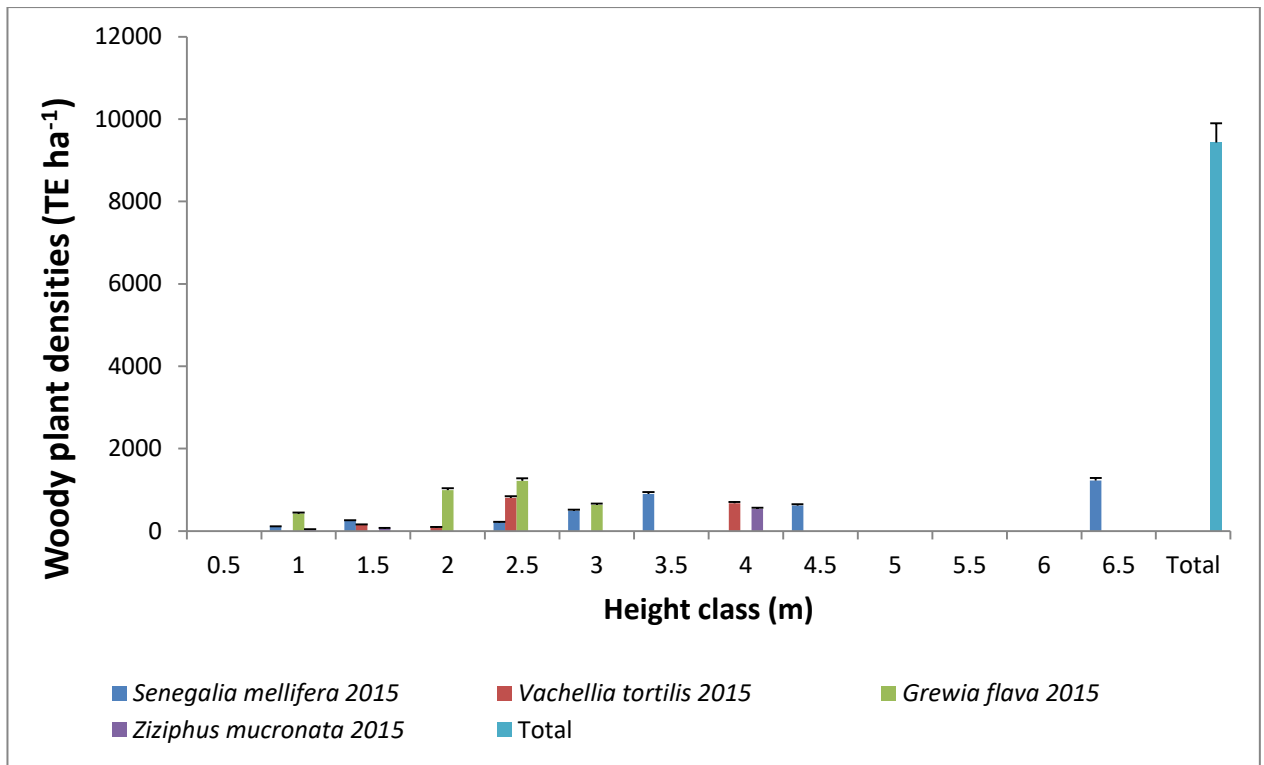


Figure 5.4: Woody species densities per height class in Disaneng I (2015)

5.3.3.2 Study Site II (Disaneng II)

This site was more accessible to livestock than humans because it was further away from the road (Figure 4.1). This site had a moderately dense woody species composition and a limited grass layer of *Bothriochloa insculpta* and a few patches of *Themeda triandra*.

Similar to study site I, *Grewia flava*, *Senegalia mellifera*, *Vachellia totilis* and *Ziziphus mucronata* were the only woody species present and occurred at densities of 1 263 –, 4 051 –, 511 and 636 TE ha⁻¹ respectively. The total woody species density was recorded at 6 459 TE ha⁻¹ in 2014 (Figure 5.5) and increased to a density of 7464 TE ha⁻¹ in 2015 (Figure 5.6).

Senegalia mellifera was present in all height classes except for 2.5 m high trees (Figure 5.8). Other woody encroachers were present as follow (Figure 5.3):

Grewia flava: Height classes 0.5–, 1– and 2.5 m

Table 5.5: The growth of woody species per height class in Disaneng I (2014–2015)

Height Classes (m)	<i>Senegalia mellifera</i> 2014 (TE ha ⁻¹)	<i>S. mellifera</i> 2015 (TE ha ⁻¹)	<i>Vachellia tortilis</i> 2014 (TE ha ⁻¹)	<i>V. tortilis</i> 2015 (TE ha ⁻¹)	<i>Grewia flava</i> 2014 (TE ha ⁻¹)	<i>G. flava</i> 2015 (TE ha ⁻¹)	<i>Ziziphus mucronata</i> 2014 (TE ha ⁻¹)	<i>Z. mucronata</i> 2015 (TE ha ⁻¹)
0–0.5	130				623		55	
0.75–1	114	110	210			427	89	
1–1.25								40
1.25–1.5		250	90	150	1 000			
1.5–1.75								70
1.75–2	214		804	90	1 220	990		
2.25–2.5	586	214		804	665	1 220		
2.75–3	911	490				630		
3.25–3.5		900	760					
3.5–3.75							500	
4–4.25	752			670				
4.25–4.5								535
4.5–4.75		615						
6–6.25	1 300							
6.5–6.75		1 225						
Total:	4 007	3 804	1 864	1 714	3 508	3 267	644	645

Vachellia tortilis: Height classes 0.5– and 1.5

Ziziphus mucronata: Height classes 0.5–, 3– and 5.5.

Molathlegi (2008) recorded a similar density of 4 250 TE ha⁻¹ for *Senegalia mellifera* in Masamane (Figure 4.1). *Grewia flava* was recorded at a density of 2 000 TE ha⁻¹ at Tshidilamolomo (Figure 4.1) and was the only site where *Grewia flava* occurred at a high density (1 263 TE ha⁻¹, Mogodi, 2009). It is thus clear that woody plant encroachment is a continuous process.

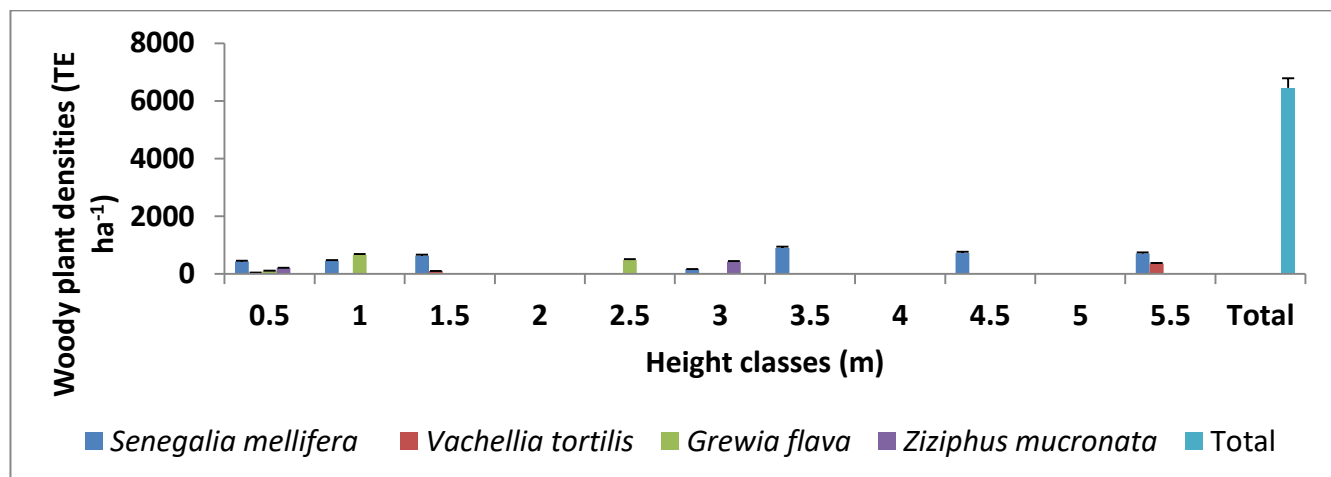


Figure 5.5: Woody species densities in Disaneng II (2014)

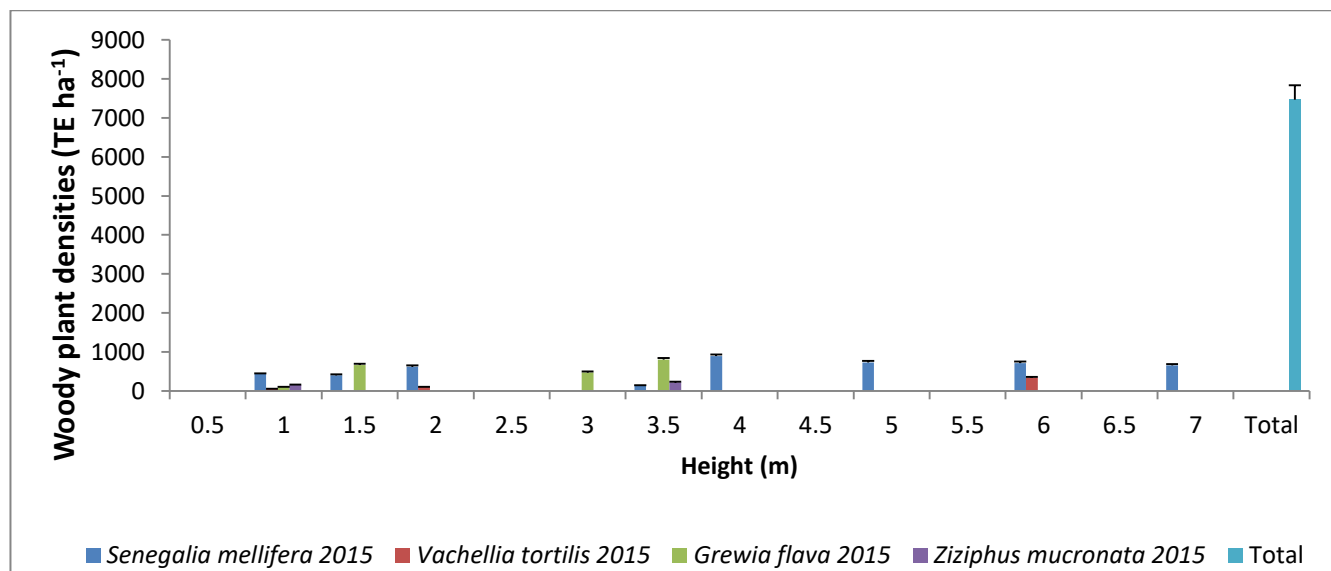


Figure 5.6: Woody species densities per height class in Disaneng II (2015)

The reasons for this increase in total woody densities from 2014 to 2015, is especially the occurrence of 5 m 6 m and 7 m high *Senegalia mellifera* trees (Figure 5.6).

Table 5.6: The growth of woody species per height class in Disaneng II (2014–2015)

Height Classes (m)	<i>Senegalia mellifera</i> 2014 (TE ha ⁻¹)	<i>Senegalia mellifera</i> 2015 (TE ha ⁻¹)	<i>Vachellia tortilis</i> 2014 (TE ha ⁻¹)	<i>Vachellia tortilis</i> 2015 (TE ha ⁻¹)	<i>Grewia flava</i> 2014 (TE ha ⁻¹)	<i>Grewia flava</i> 2015 (TE ha ⁻¹)	<i>Ziziphus mucronata</i> 2014 (TE ha ⁻¹)	<i>Ziziphus mucronata</i> 2015 (TE ha ⁻¹)
0–0.5	455		52		130		264	
0.75–1		430		50		100		150
1.25–1.5	650		100					
1.5–1.75					660			
1.75–2		619		100		660		
2.25–2.5					500			
2.75–3	200				800	475	637	
3–3.25								
3.25–3.5						800		220
3.5–3.75	911	136						
3.75–4		890						
4.25–4.5	730							
4.75–5		730						
5.25–5.5	714		380					
5.75–6		714		340				
6.25–6.5	650							
6.75–7		650						
Totals:	4 830	4 569	532	490	2 090	2 035	901	370

5.3.3.3 Study Site III (Disaneng III)

Disaneng III was the furthest from the main road and an impenetrable thicket of *Senegalia mellifera*, causing both livestock and human accessibility to be limited was evident (Figure 4.1). Similar to site II, this site had a dense layer of woody species and a limited grass layer of *Bothriochloa insculpta* and a few patches of *Themeda triandra*.

Grewia flava, *Senegalia mellifera*, *Vachellia tortilis* and *Ziziphus mucronata* were the only woody species present with densities of 690 –, 4 244 –, 1 696 – and 1 139 TE ha⁻¹ (Figure 5.8) respectively. The total woody species density was recorded at 7 818 TE ha⁻¹ (Figure 5.7 in 2014 and decreased to a density of 7464 TE ha⁻¹ in 2015. *Senegalia mellifera* was present in all height classes (0.5 – 5 m), except for 2 m high plants (Table 5.7).

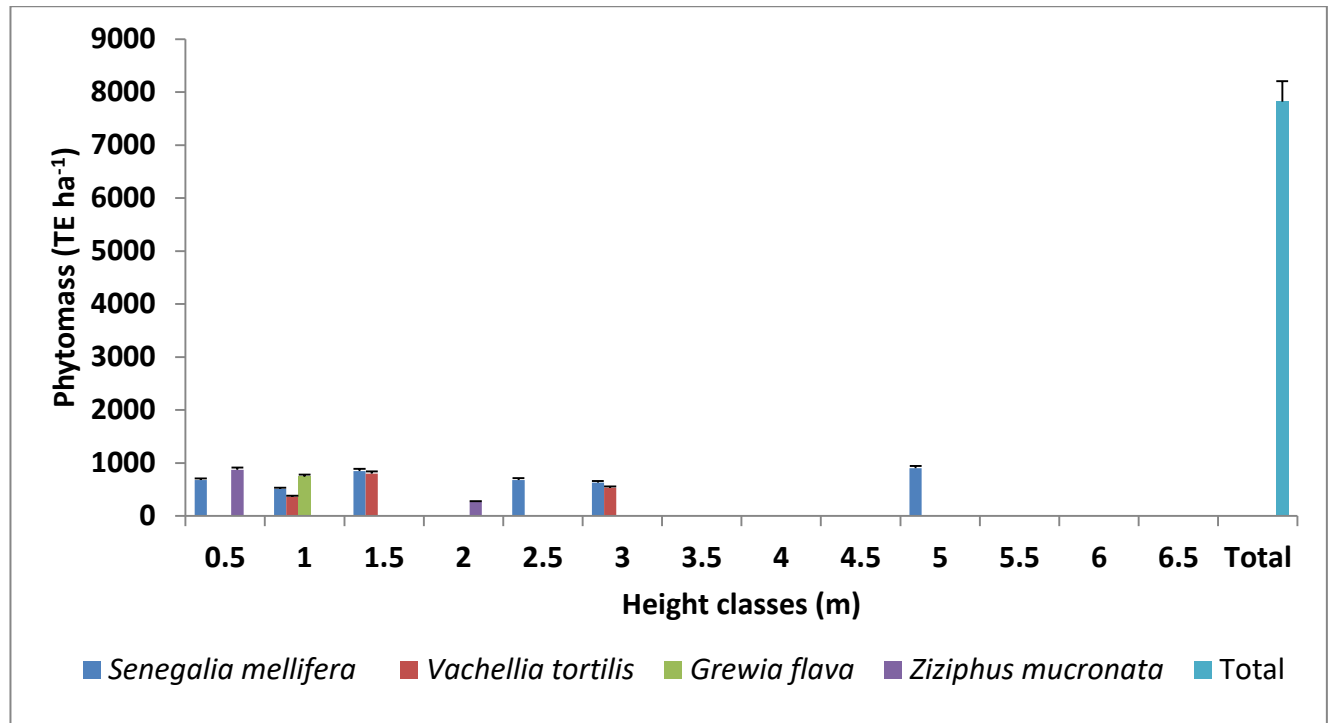


Figure 5.7: Woody species densities in Disaneng III (2014)

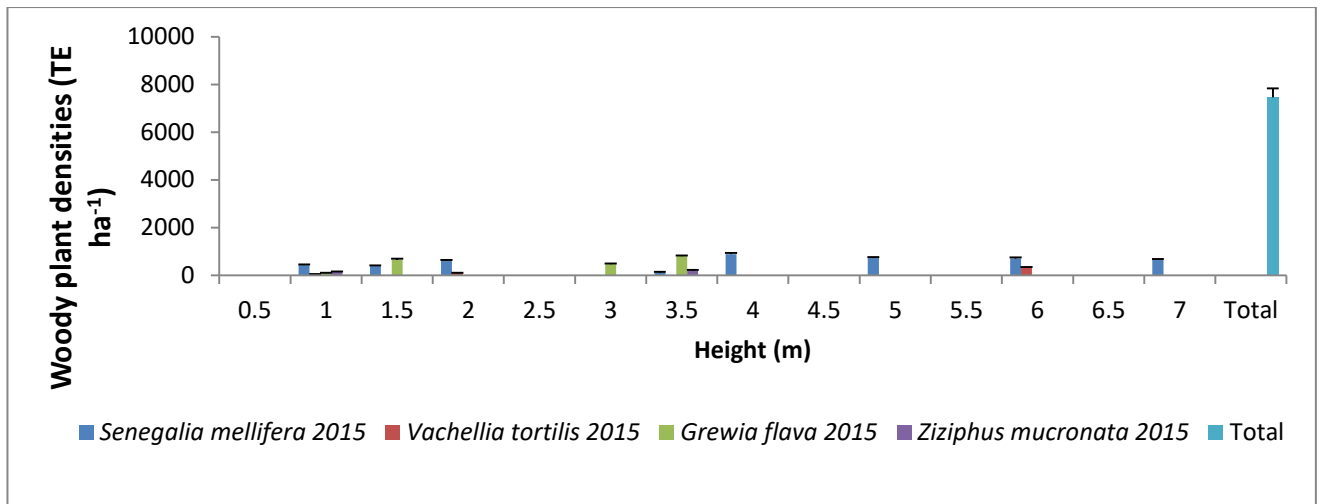


Figure 5.8: Woody species densities per height class in Disaneng III (2015)

Table 5.7: The growth of woody species per height class in Disaneng III (2014–2015)

Height Classes (m)	<i>Senegalia mellifera</i> 2014 (TE ha ⁻¹)	<i>Senegalia mellifera</i> 2015 (TE ha ⁻¹)	<i>Vachellia tortilis</i> 2014 (TE ha ⁻¹)	<i>Vachellia tortilis</i> 2015 (TE ha ⁻¹)	<i>Grewia flava</i> 2014 (TE ha ⁻¹)	<i>Grewia flava</i> 2015 (TE ha ⁻¹)	<i>Ziziphus mucronata</i> 2014 (TE ha ⁻¹)	<i>Ziziphus mucronata</i> 2015 (TE ha ⁻¹)
0–0.5	700						891	
0.75–1	510	655	363		690			
1.5–1.75	865	510	804	363		690		850
2–2.25		830		800			268	
2.5–2.75								268
2.75–3	630		532					
3–3.25								
3.25–3.5		630		530				
5–5.25	899							
5.5–5.75		899						
Totals:	3 604	3 524	1 699	1 693	690	690	1 159	1 118

Only one metre high (1 m height class) *Grewia flava* shrubs were recorded (Table 5.7) whereas *Vachellia tortilis* was recorded at height classes of 1 –, 1.5 – and 3 m. *Ziziphus mucronata* occurred at the height class of 0.5 –, 3 – and 5.5 m. This was substantially different from benchmark (BM) site, which recorded a woody density of 412 TE ha⁻¹ (Table 5.3).

The total woody plant density showed that the grass was totally suppressed in this area. Several explanations for the persistence of tree-grass mixtures in savannas have been advanced thus far. These either concentrate on competition-based mechanisms, where niche separation with respect to limiting resources such as water, lead to tree-grass coexistence, or demographic mechanisms, where factors such as fire, herbivory and rainfall variability promote tree-grass persistence through their dissimilar effects on different life history stages of trees. Tests of these models have been largely site-specific and although different models find support in empirical data from some savanna sites. This lack of consensus on determinants of savanna structure and function arises because different models focus on different demographic stages of trees; focus on different limiting factors of tree establishment; and emphasize different subsets of the potential interactions between trees and grasses.

Grewia flava was found in the three study sites except in the reference site. Thus the species diversity in the study area and in the benchmark was similar.

5.4 Discussion

It is clear that there was no woody plant invasion (bush encroachment) in the benchmark site. *Senegalia mellifera* was the dominant woody encroacher, followed by *Ziziphia mucronata* and *Vachellia tortilis* respectively.

The woody species density in the benchmark site was recorded at 412 TE ha⁻¹, thus enabling the herbaceous stratum to develop.

Woody species densities in the three study sites Disaneng I, 9 430 TE ha⁻¹, Disaneng II 7 464 TE ha⁻¹ and Disaneng III 7 818 TE ha⁻¹ were more than ten times higher than in the benchmark site (412 TE ha⁻¹), implying that the grass species in these sites were totally suppressed. *Senegalia mellifera* was dominant at all three study sites. *Senegalia mellifera* is known to cause impenetrable thickets and is also an indicator of overgrazing in the savanna (Coates Palgrave, 2000; Comole, 2014). The shallow root system of *Senegalia mellifera* enables it to compete directly with grasses (Smit, 1999 and Comole, 2014) and

can completely suppress grass growth (Moore and Odendaal, 1987; Belsky and Amundson, 1992; Richter *et al.*, 2001). The total woody plant density in site II increased from 2014 to 2015, while it decreased in sites I and III.

Senegalia mellifera was described as acting like windmills, pumping large amounts of water into the air every day (De Klerk, 2003). Smit (1999) stated that, many thorn trees fail to grow beneath canopies (shade intolerant) of established individuals, irrespective of its species, while *Senegalia mellifera* is able to grow under canopies of other *Senegalia mellifera* trees (Tainton, 1981; Comole, 2014). Moreover, *Senegalia mellifera* encroaches regularly in open savannas and has a great re-sprouting ability after intense fires although little is known regarding the relationship between total shrub height and regrowth after fires (Meyer *et al.*, 2005; Comole, 2014).

Figures 5.4, 5.6 and 5.8 show that there is a significant relationship ($P < 0.05$) between the density of *Senegalia mellifera* trees and the heights of these trees. Figure 5.4 shows that 3.5 m, 5 m and 6.5 m high *Senegalia mellifera* trees were the most evident in site I, Figure 5.6 indicates that 4 to 7 m high *S. mellifera* trees were dominant in site II, while Figure 5.8 indicates that 2 m, 4 m and 6 m high *S. mellifera* trees were dominating in site III. Ngaruka (2011) indicated most *Senegalia mellifera* thickets in the highlands of Namibia were of trees mostly of 2.5 m in height, or below. Such shrub-sized trees are the most problematic in encroachment since their canopies are likely to spread outwards or overlap to neighbouring trees and thus closing off access, mainly for foraging animals (Ngaruka, 2011). Therefore, this result is not in agreement with Ngaruka (2011) who indicated that density of *Senegalia mellifera* does not change with plant height.

Both Ngaruka (2011) and Joubert (2014) did studies on *Senegalia mellifera* in the highland savanna of Namibia where the soils were typically more than 300 mm in depth, with a very high schist and quartz gravel content (Joubert, 2014), while the soil in this study was well-drained sandy soil, as well as well-drained sandstone mostly of the Hutton and Mispah forms. The sandy soil is medium to coarse with red B-horizons (Refer to Chapter 3).

Hagos and Smith (2005) concluded that *Senegalia mellifera*, like so many other woody species, act as a biological agent that facilitate the enrichment of the soil beneath their canopies. Global increases in atmospheric CO₂ may play a greater role in this than previously thought (Bond and Midgley, 2000).

Ungulate browsers are probably ineffective in preventing a transition towards bush thickening; rather, they regulate the structure of existing thickets (Teague and Smit, 1992). Domestic goats utilise *Senegalia mellifera* leaves and pods but it is not their preferred forage species (Rothauge *et al.*, 2003). Small browsers such as lagomorphs may be far more important regulators of thicket densities (Ostfeld *et al.*, 1997; Weltzin *et al.*, 1997).

Ziziphus mucronata is a small to medium-sized deciduous tree, up to 10 metres in height, with an irregular spiky canopy (Shackleton *et al.*, 2005) and did not occur at high densities in this study.

Vachellia tortilis was evident in study site III. *Vachellia tortilis* is adapted to dry conditions and is often one of the first plants to establish on areas where the vegetation is disturbed (Coates Palgrave, 2000 and Comole, 2014). This species also occurred along dry riverbeds and is cold tolerant (Coates Palgrave, 2000; Smit, 1999 and Comole, 2014).

In many cases, *Grewia flava* is harvested to make brushwood fencing and shelter for cooking in communal areas and also used as laths for roofing traditional houses (Mainah, 2006) and in some cases, *Grewia flava* is uprooted to obtain the roots, which are considered to have medicinal value (Moleele *et al.*, 2002). This species is browsed on mainly by donkeys and goats, both of which are considered less selective feeders (Molatlhegi, 2008; Mogodi, 2009; Comole, 2014).

Molatlhegi (2008) recorded *Grewia flava* at a density of 250 TE ha⁻¹ in Phitsane Refer to Figure 4.1). *Grewia flava*, is also an encroacher but it occurred at non-threatening densities in this study, probably due to its uses by local inhabitants. Although being used by the local inhabitants that is keeping it in lower numbers, it is a plant well-known to coppice after being cut and also in becoming dense.

The major products obtained from *Grewia flava* include berries (fruits), roots, stems and leaves (Comole, 2014). Different parts of the plant are used for direct consumption, commercial trading, beer brewing, firewood (kindling), stimulant and/or for building fences and houses. Leaves are mainly used to a relatively lower scale for making ‘tea’ when boiled (Mainah, 2006). The use of berries was reported as the most important product of *Grewia flava* for direct consumption, commercial selling and making beer (Mainah, 2006).

Although the woody plants (in the different height classes) increased in height, the total density of the woody plants in all the sites slightly decreased. This could be due to various environmental variables such as expansion of the villages for housing purposes. It seems that the increase in height of the measured woody species in each height class is not a suitable evaluation tool for woody plant growth over a period of a single year, but that the measurement period be a minimum of 2 years or more would be more appropriate.

5.4.1 Woody species cover and density

The woody plant density in all the survey sites indicated that few trees/shrubs grew above 2 m high probably due to browsing pressure since all of the dominant woody plants are excellent browse species (Ringrose and Matheson, 1991; Moleele, 1998) and also due to intra species competition between individuals of the same species for available resources such as nutrients, water and space. Subsequently, browsing could also be responsible for proliferation of woody species through seed dispersal by livestock (Tews *et al.*, 2004). *Senegalia mellifera* seedlings are shade tolerant (Joubert, 2014) and thus grow in the shade of higher trees. Shrubs limit their own growth rate by intense density-dependent inter-shrub competition (Smit, 2003).

5.4.2 Woody vegetation diversity

Browsing pressure in overstocked rangelands could also increase mortality, but palatable woody species can reduce interspecific competition, thus allowing a range of less dominant woody species to establish. This could eventually lead to an increase in woody plant species diversity (Reyes *et al.*, 2010). The lowest woody plant species richness was

observed at site I. From 2014 to 2015, *Ziziphus mucronata* were completely eradicated from this site, due to roadworks and kraal building by inhabitants.

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

5.4.3 Rooting systems of encroacher and non-encroacher woody vegetation

The difference in root characteristics of woody vegetation facilitates coexistence with other plant species because they allow the use of various resources at different soil depths (Smith and Smith, 2001). The dominant woody plant species at all study sites differed significantly in root type and most of them, particularly *Grewia flava* had several lateral roots spreading within 500 mm depth from the soil surface. According to Hipondoka *et al.* (2003) both the grass and woody vegetation roots occupy the surface soil layers in the Kalahari sandveld of Botswana. The dominance of shallow rooted woody plants such as *Senegalia mellifera* has been attributed to their ability to absorb nutrients (Tolsma *et al.*, 1987) and soil moisture (Cole and Brown, 1976; Hipondoka *et al.*, 2003) in the surface soil layers. This contradicts the two layer hypothesis, which suggests that woody plants monopolise the use of soil moisture at deeper soil layers (Walker *et al.*, 1981).

The variation in the rooting systems did not follow any clear pattern (i.e. rainfall gradient) and therefore could be attributed to individual plant response to micro habitat condition such as soil moisture, fertility and competition (Hipondoka and Versfeld, 2006). For example, *Vachellia tortilis* had taproots reaching the water table (Hipondoka and Versfeld, 2006; Holdo and Timberlake, 2008). However, the growth of *Vachellia tortilis*

roots is known to be adaptive to local conditions and had exhibited taproots in other savanna ecosystems across Southern Africa (Hipondoka *et al.*, 2003) as observed in this study. The rooting depth and distribution of tree species could be strongly influenced by edaphic factors (Holdo and Timberlake, 2008) and probably in the case of these results, sandy soils allowed *Grewia flava*, *Sengalia mellifera*, *Vachellia tortilis* and *Ziziphus mucronata* to develop taproots capable of accessing soil moisture that escapes the lateral roots in the surface soil.

5.4.4 Relationships between woody vegetation dynamics and environmental variables

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

A multiple factor analysis showed that edaphic factors, particularly clay content was the most important predictors of woody vegetation cover, density and diversity in a non-linear model (Laurance *et al.*, 2001).

The study area, located in the savanna, showed higher woody vegetation cover than in the reference (benchmark) site. These results are consistent with observation in other African savanna systems (Sankaran *et al.*, 2008), identified rainfall as the key determinant of

woody plant cover. Sankaran *et al.* (2008), however, observed a negative relationship between woody plant cover and soil clay content. Water infiltration in sandy soils is higher than in clay soils, even in the absence of vegetation, because the soil absorbs rainwater well and uniformly (Rietkerk *et al.*, 1997). In semi-arid environments, where soil water limits plant growth, good infiltration enhances the development of a continuous vegetation cover during good rainy years (Rietkerk *et al.*, 1997). As a result, Sankaran *et al.* (2005) suggested that the Kalahari sandy soils sustain higher woody vegetation cover. The results of this study also showed that the increases in woody plant cover lead to a decline in herbaceous biomass.

The reduction in herbaceous biomass in relation to an increase in woody vegetation cover has also been reported in studies conducted elsewhere (Dalle *et al.*, 2006) and is consistent with bush encroachment perception to be discussed in Chapter 7. Herbaceous biomass reduction through grazing create space and reduces competition for resources, which allows tree seedlings to develop (Kraaij and Ward, 2006) and eventually accelerates bush encroachment. The reduction in herbaceous biomass threatens the sustainability of the livestock industry, especially cattle that are dependent on grass productivity in the rangelands. However, introducing browsers such as goats in bush encroached rangelands could help reduce the woody plant cover (Augustine and McNaughton, 2004) and would also diversify livelihood of pastoral farmers.

5.4.5 Herbaceous composition

The herbaceous vegetation compositions were highly heterogeneous spatially, both at local scale and between sites. The variation in herbaceous composition between sites could be explained by differences in soil texture, and rainfall. Our rangeland assessment showed that the study area had relatively high bare ground and presence of unpalatable herbs associated with heavy grazing (Makhabu and Marotsi, 2011; Tessema *et al.*, 2011), which suggests that these rangelands are experiencing ecological stress (Whitford *et al.*, 1998). In addition, the study sites were also dominated by unpalatable herbaceous plants, and subsequently the condition of both rangelands was classified as ‘poor’ based on a range condition guide (Mosley *et al.*, 1986). The reference site was in ‘fair’ condition, characterized by a grass cover, limited bare ground and abundance of palatable herbaceous plants. The reference site also had a balanced composition of decreaser,

increaser I and II species, which indicated that they were not overgrazed (Trollope *et al.*, 1989).

However, our observations at the other sites showed no visible differences in palatable herbaceous plant cover, which are supported by other studies in African savannas (Parsons *et al.*, 1997; Tefera *et al.*, 2008). The herbaceous vegetation composition in savanna ecosystems is generally undergoing changes. This is suggested by the high cover of Increaser II herbaceous plants, particularly at site I and site III. These rangelands were declining in rangeland condition as indicated by decrease in grass cover, biomass and increase of herbs cover as was clearly observed in site II. This observation is consistent with the literature (Oba *et al.*, 2001).

5.4.6 Herbaceous biomass

Although a herbaceous biomass study was not part of the objectives of this study (see Chapter 1) the results of the current study revealed that herbaceous biomass was highly variable at local spatial scale and between Disaneng site I and the other two sites. This observed variability in herbaceous biomass is recognised as an attribute of arid and semi-arid ecosystems (Mashiri *et al.*, 2008). Herbaceous biomass variability between the study sites reflected rainfall differences as shown in Chapter 3. Bush encroachment was shown to be another local factor that could have contributed to herbaceous biomass variation.

Hagos and Smit (2005) have shown that bush encroachment suppresses herbaceous plant productivity. The highly encroached site II had low herbaceous biomass, while site I, characterized by low woody cover, had high herbaceous biomass. The herbaceous biomass was also significantly affected by grazing management systems. Herbaceous biomass differences between grazing management systems at a local scale are due to the higher grazing intensities of the communal grazing lands reflecting greater consumption of herbaceous biomass in the communal lands responsive to grazing pressure as also suggested in other studies (Sternberg *et al.*, 2000; Mphinyane *et al.*, 2008).

The surface plant litter was also significantly lower at all sites, which further confirmed higher consumption of herbaceous plants such that there is less biomass accumulating and degenerating into litter.

5.5 Conclusion

Staver *et al.* (2001) showed that grass dominance, in general, negatively impacts on woody seedling survival and growth, which can suppress woody vegetation even if resource availability is adequate to support adult shrubs and trees. With grasses virtually absent in the study sites, competition for resources was among encroaching woody species.

Woody plants and grasses in southern Africa grazing lands appear to compete primarily for water and it is found that, because of a deep penetrating root system, they are consistently stronger competitors for water and nutrients (Scholes and Walker, 1993). This competition encourages bush encroachment, especially in arid and semi-arid communally managed areas where overgrazing is evident. High woody plant densities suppress the grass cover and allow encroaching woody species to thicken.

Higher *Senegalia mellifera* trees contributed more to woody plant density than lower trees.

By monitoring bush densities by species and by height class over a single growing season (2014 to 2015) seems to be inaccurate. Longer time intervals (5 years or even more) should be more appropriate to monitor bush density development (change) over time. According to tables 5.3, 5.5, 5.7 and 5.8 dieback of woody plants rather than growth took place from 2014 to 2015. This is because of local use of wood for building and fire purposes by local inhabitants.

Hagos and Smit (2005) stated that *Senegalia mellifera* is of important economic value, if it is not of a high density and can be used as a valuable fodder tree on game farms in semi-arid areas. The favourable effects of *Senegalia mellifera* and the other invading species discussed on the soil nutrient status should thus be considered when a bush control

program is implemented in the area. It is thus recommended that the encroacher species be controlled by chemical arboricides (Hagos and Smit, 2005). In order to reach this goal, it is important for the Department of Agriculture to intervene by training local inhabitants to use arboricides, in determining which individuals should be removed and how many. Inhabitants should also be taught about proper grazing strategies. The Department of Agriculture should aid communities by supplying chemical arboricides and assist them with the control process. This, however will only be successful if the local people gets property rights and own land 'of their own'. In this way they will accept responsibility of 'proper grazing management structures' and the eventual rehabilitation of degraded grazing land.

CHAPTER 6: TEXTURE ANALYSIS USING REMOTE SENSING

6.1 Introduction

Texture classification was applied to the images that were used (see Section 4.2.5). The objective was to segment the input image into homogeneous textured regions and identify each region, as one of a pre-learned library of textures, i.e. tree area and grass area distinction. The classification of remote sensing imagery is of importance in many applications, such as navigation, surveillance and exploration. It has become a very complex task spanning a growing number of sensors and application domains. Much attention has been given to the use of the spectral signature for the identification of region types (Wharton, 1987; Lee and Philpot, 1991). Only recently has the idea of adding on spatial information been presented (Ton *et al.*, 1991). Texture can play a major role in segmenting the images into homogeneous areas and enhancing other sensors capabilities, such as multispectral analysis, by indicating areas of interest in which further analysis can be pursued. Fusion of the spatial information with the spectral signature will enhance the classification and the overall automated analysis capabilities. Most of the work in the literature focuses on human expert-based rules with specific sensor data calibration.

6.2 Remote sensing of vegetation density estimation in the savanna

Increasingly, the application of remotely sensed data has proven to be the most efficient means to assess vegetation cover or density. Remote sensing-based vegetation estimation methods and assessment of existing issues influencing vegetation estimation are inevitable for improving estimation accuracy. However, vegetation canopy cover or density estimation remains a difficult task especially in areas with complex vegetation stand structures and heterogeneous environmental conditions. Various techniques based on integration of field measurement and remote sensing have been applied for vegetation modelling (Olsson, 1984; Yang and Prince, 1997; Skidmore *et al.*, 1997; Moskal and Franklin, 2001; Hudak and Brockett, 2004). Savannas are described as tropical vegetation consisting of a continuous grass layer usually with a sparse canopy of woody vegetation. Savanna ecosystems worldwide occupy about 20% of the Earth's land surface (Stott, 1991) and support a diverse community of species. At the regional or landscape level the

savanna biome is estimated to occupy 46% of southern Africa and over one third of South Africa, making it the largest biome in southern Africa (Low and Rebelo, 1998).

6.3 Savanna woody cover estimation and remote sensing

Increasingly savanna environments transformed by a diverse range of land management systems and naturally occurring disturbances are causes of potential concern for ecosystem managers and researchers alike. Savanna landscapes set aside for biodiversity conservation play an important role in their capacity to function as carbon sinks for the reduction of global atmospheric carbon emission (Hudak *et al.*, 2003), and their influences on the density, diversity and distribution of wildlife species. Woody vegetation cover is specifically an important component and a variable relating to the savanna ecosystem composition (Hudak, 1999). Hence, the ability to quantify woody vegetation structural parameters can provide a measure of the underlying variation in savanna areas and enhance our understanding of the factors driving variability in the savanna environment. Remotely sensed vegetation data is an efficient means to obtain information on savannas. Remote sensing has been utilised as a primary source of spatial data to characterize patterned woody vegetation density and canopy structure in the southern African savanna ecosystems (Hudak and Wessman, 1998; Yang and Prince, 2000; Hudak and Wessman, 2001; Wessels *et al.*, 2006). Remotely sensed data has also been used to develop updated vegetation maps and establish statistics about percentage woody canopy cover in semi-arid savanna environment (Stuart *et al.*, 2006; Wessels *et al.*, 2006).

The various remote sensing techniques applied to woody cover assessment produced varying significant levels of correlation with the woody vegetation cover and their estimation accuracy (Hudak and Wessman, 1998; Yang and Prince, 2000; Hudak and Wessman, 2001; Wessels *et al.*, 2006). Although several image analysis techniques have been evaluated for vegetation remote sensing, one of the main challenges of using remotely sensed data to estimate woody vegetation cover in savanna ecosystems has been the identification of the best remote sensing derived vegetation variables that significantly relate with the ground surveyed vegetation data. In addition, unsatisfactory levels of estimation error arising from the use of traditional correlation or ordinary regression models that ignore the spatial autocorrelation factor in both field and remote sensing data

(Atkinson *et al.*, 1994; Mutanga and Rugege, 2006) limit the full potential of remote sensing for vegetation resources estimation.

In this regard, the application of remote sensing techniques that explore the maximum use of information contained in remotely sensed vegetation coverage is crucial for the accurate delineation of the patches of vegetation cover (woody cover, a continuous variable) from other scene elements. The utility of spatial information such as auto and cross correlation factors, using applications of spatial statistics such as kriging interpolators which reduces interpolation error (Atkinson *et al.*, 1994; Mutanga and Rugege, 2006) can be beneficial for the estimation of woody vegetation cover. It is important to evaluate the utility of spatial statistical approaches in combination with remote sensing because fundamentally vegetation and its spectral measure (radiation) are spatially correlated, both to themselves (auto correlated) and to one another (cross correlated) (Atkinson *et al.* 1992; Acharya, 1999).

In this respect, the spatial dependence between vegetation and its radiation obtained from remote sensing (i.e. spectral vegetation indices or image texture measures) can enhance the utility of using remote sensing techniques to extract woody density parameters. Given that it is imperative for savanna remote sensing studies to identify critical variables for modelling vegetation resources, the following sections introduce satellite remotely derived vegetation data for the estimation of woody vegetation cover.

6.4 Image analysis

The classified images shown in Figures 6.1–6.3 show change in woody cover over time. These changes of woody plants species over time (2006, 2010 and 2014) have an inversely proportional relationship with the grasses. The legend in all these classified remote sensing images depicts trees (green colour) and different grass types (Figures 6.1–6.3). The increase in woody plant cover between 2006 – 2010 was 4.72% whereas between 2010 – 2014 was 41.22 % (Figure 6.4).

6.5 Field validation

Hudak and Wessman (1998) determined that image texture correlates significantly to two structural parameters measured in the field: woody plant densities and the region covered by woody species each year. While woody plant densities proved the better correlate in the analysis (Figure 6.5), percentage of the area measurements validate this analysis because it can be measured much more quickly using the variable quadrant (see Section 4.3).

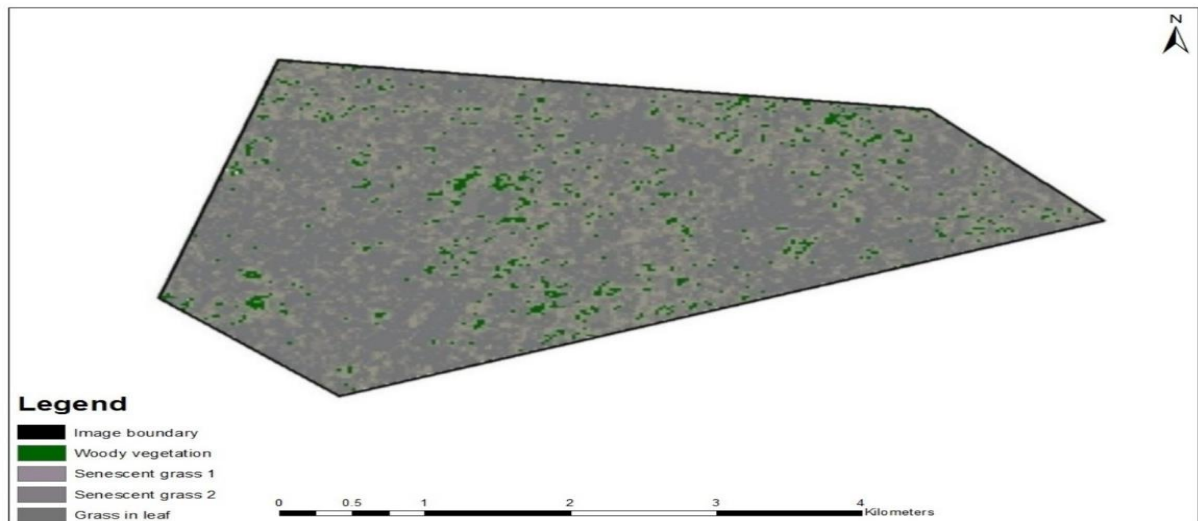


Figure 6.1: Classification of the 2006 texture image of the Disaneng study area

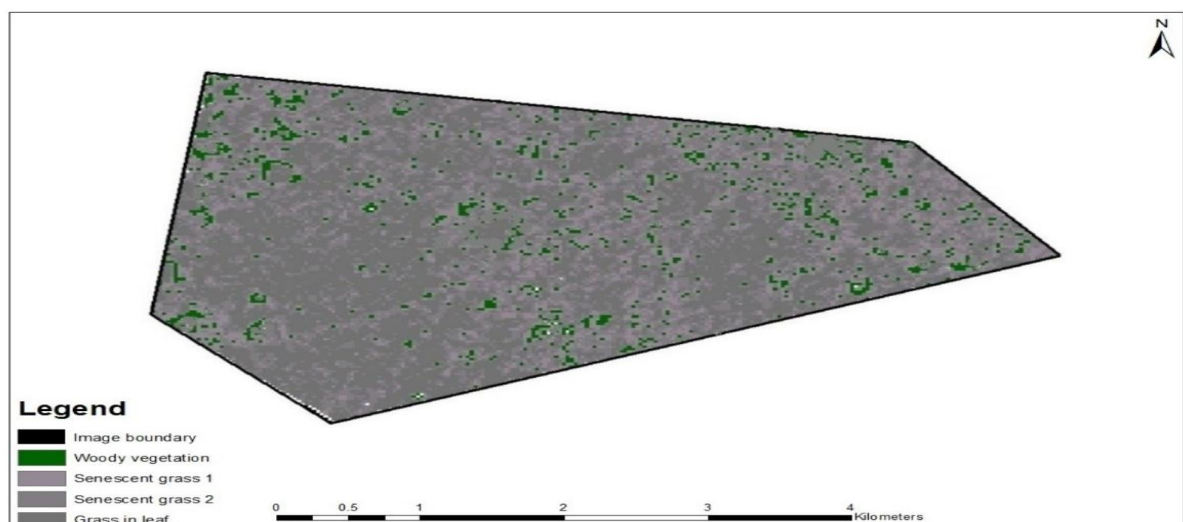


Figure 6.2: Classification of the 2010 texture image of the Disaneng study area

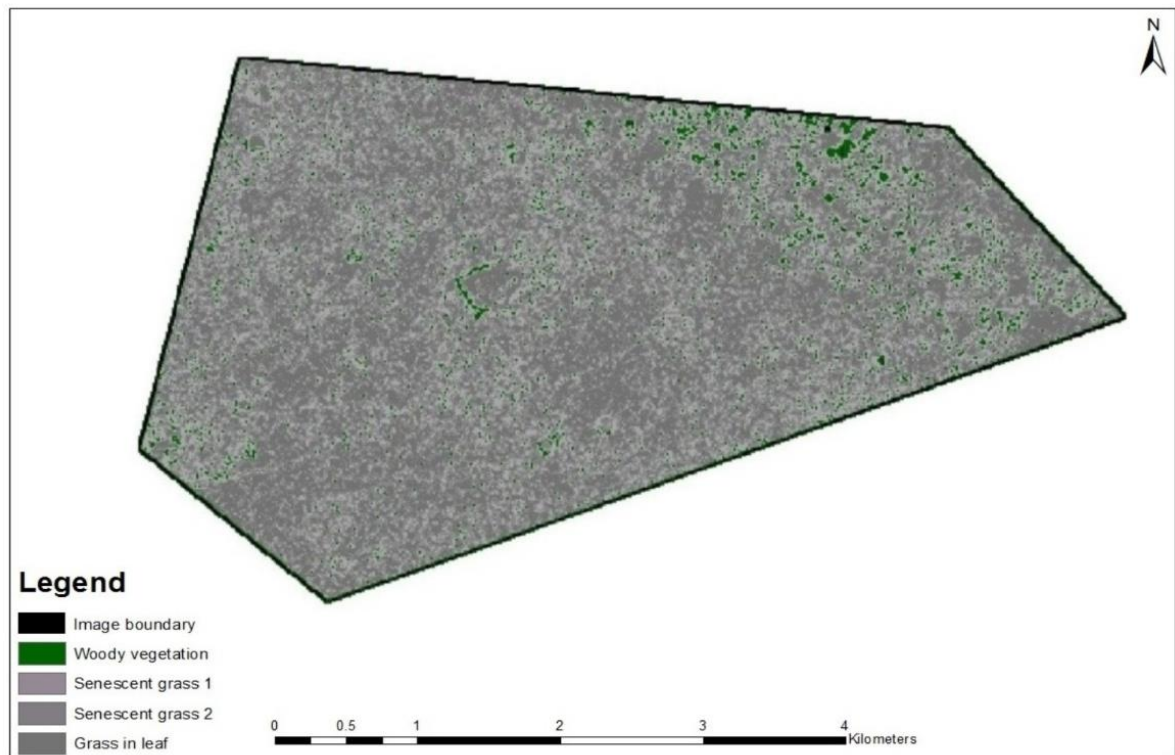


Figure 6.3: Classification of the 2014 texture image of the Disaneng study area

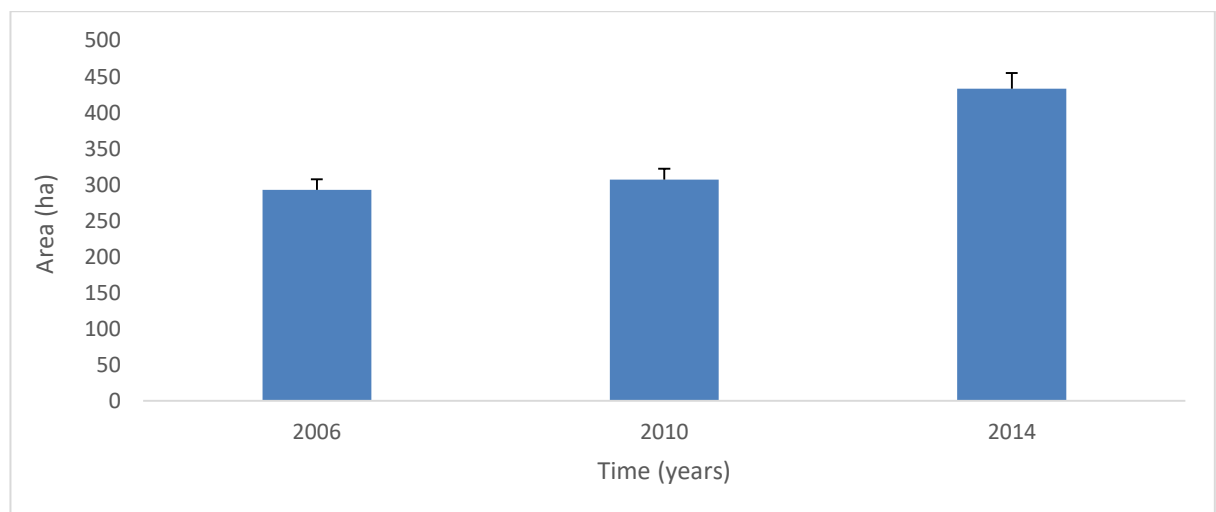


Figure 6.4: The woody cover increase from 2006 – 2014

Field sampling sites were geo-located with a global positioning system (GPS) and corrected to an accuracy of ± 5 m via post-processing. Mean textural values from the SPOT image were calculated within a 25 m radius of each corrected position (approximately 70 pixels) and compared to the field measurements. The correlation

coefficient was recorded at 0.99, which implies that there was a direct proportional relationship between the two variables (Figure 6.5).

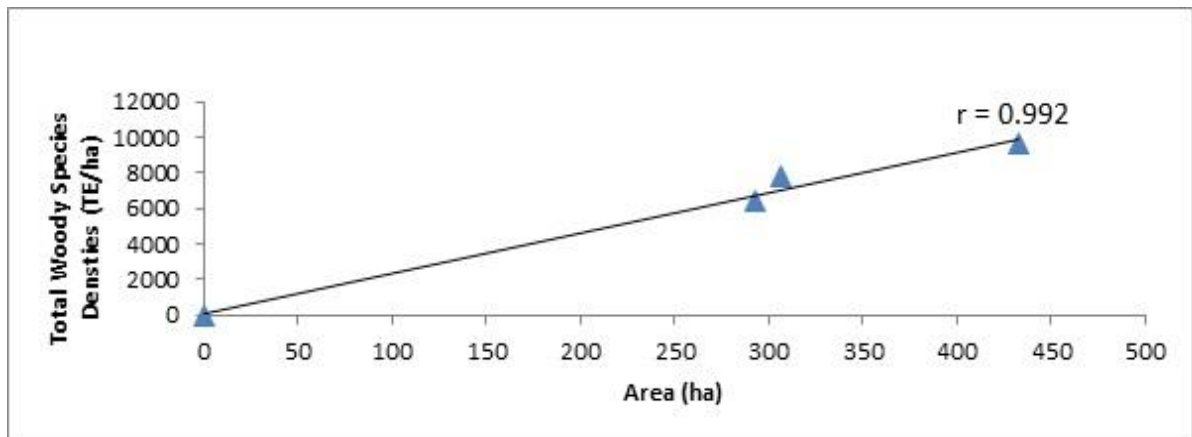


Figure 6.5: Correlation between woody densities derived in the field and area of woody vegetation derived by image classification

Aerial photography and other remote sensing techniques detect gradual increases in cover, rather than sudden increases in density, reinforcing the impression that bush thickening in the Highland Savanna is a continuous or creeping process, yet at a microsite level, it follows a sleep then leap mode of regression. Parent trees ‘sleep’ (do not recruit successfully) until three consecutive years of above average rainfall occur, the resultant establishment or ‘leap’ typically only being metres from the parent trees (Joubert, 2014). Given the size of seedlings and mature *Sengalia mellifera* plants in general (Joubert, 2014), cover changes over 13 years might not even be detectable.

Woody density and overall cover, mean percentage cover density of woody plants per hectare were estimated 31% and 1 995 ha, respectively. Coppock (1994) reported that woody plants’ cover exceeds 40% of Borona rangelands. Gemedo, Mass and Issestein (2006) also reported 52% and 3 014 woody plant cover and density, respectively (Table 6.1).

Table 6.1: Mean percentage cover and density of woody plants

Percentage %	Phytomass (plants ha ⁻¹)
31	1 995
40	2 400
52	3 014

Bush cover of 40% and/or 2 400 (woody plants ha⁻¹) was considered as borderline between non-encroached and encroached conditions (Roques *et al.*, 2001), and 2 500 (tree equivalent ha⁻¹) was considered as a highly encroached condition (Ritchter *et al.*, 2001).

The woody plant densities in the 3 study sites were;

Site 1: 9 640 TE ha⁻¹

Site 2: 6 461 TE ha⁻¹

Site 3: 7 820 TE ha⁻¹

In light of what was indicated by Coppock (1994) (Mean % cover of 31% represents a woody plant density of 1 995 plants ha⁻¹), Roques *et al.* (2001) (Mean % cover of 40% represents 2 400 plants ha⁻¹) Gemedo *et al.* (2006) (Mean % of 52% represents 3 014 plants ha⁻¹) it is clear that the woody plants densities in the three study sites were severe. Roques *et al.* (2001) indicated that a woody plant density of 2 500 TE ha⁻¹ is the borderline between an encroached and a highly encroached condition. From figures 6.1, 6.2 and 6.3 it is thus clear that bush encroachment in the study sites were severe and an urgent bush control programme in these areas is thus essential.

6.6 Validating remotely sensed data by ground truth data

Remotely-sensed data necessitates ground based validation. Data received from the remote sensors are generally not perfectly correlated with the actual condition of the earth's surface (Park, 1984). There is no map which provides completely accurate information, mapping and interpretation of vegetation conditions is limited to certain adjacent scales of observation. Arieira *et al.* (2011) recommend integrating remotely-

sensed data and ground observations, to reduce uncertainties and increase the reliability of the results. Their study revealed that an improvement of vegetation mapping could be achieved by increasing the number of field observations. Fuller *et al.* (1998) integrated field surveys and remote sensing data for a biodiversity assessment in the tropical forests and wetlands of Sango Bay, Uganda. They observed that an integration of field observation and remote sensing assists in the generation of biodiversity maps useful for conservation planning. Traditionally, a correction of digital data was done using photo-interpretation (Congalton, 1991). This was because photo-interpretation was accepted as accurate without confirmation, based on the assumption that it was 100 % correct (Congalton, 1991).

Zavoianu *et al.* (2001) define accuracy assessment as the degree of closeness of results to the value accepted as true or correct. The main objective of performing accuracy assessment is to provide an index of the degree of correctness of a map or classification (Foody, 2002). In remote sensing, this is achieved by overlaying a reference image and simulated image (Pontius, 2000). Accuracy assessment in remote sensing is achieved by using the error matrix (ERRMAT) method and the Kappa Index of Agreement (KIA). The error matrix measures the relationship between image classification results and measured ground conditions (Collingwood *et al.*, 2009). It is regarded as standard procedure for accuracy assessment (Foody, 2002). On the other hand, the KIA is a discrete multivariate technique which is used to statistically evaluate the accuracy of classified data (Collingwood *et al.*, 2009). It is commonly used to quantify the agreement between two images on a categorical basis. For this agreement to be obtained, two images must be categorized into equal groups. It considers the effect of chance agreement in the error matrix (Collingwood *et al.*, 2009). The advantage of the KIA over the standard error matrix is that it measures both overall and individual class accuracy (Collingwood *et al.*, 2009). Several studies that evaluated the efficiency and effectiveness of both the error matrix and KIA for accuracy assessment have recommended the KIA for accuracy assessment (Rossiter, 2004; Pontius, 2000). Against this background, the present study also used the KIA for accuracy assessment.

According to Amiri and Tabatabaie (2009), in order to determine the suitable indices for vegetation cover and production assessment, a contemporaneous analysis of remotely-sensed and field data should be undertaken. True, actual ground measurements are

required for several steps in the image processing procedure (Clark *et al.*, 1999). Field based measurements are important for the establishment of a gradient of conditions, to test ecological indicators and the sensitivity of remote sensors. Studies focused on biomass estimation, using remote sensing have attempted to couple ground-based vegetation quantification to satellite remote sensing (Roy and Ravan, 1996). Roy and Ravan (1996), employed remotely sensed measurements and compared them with field samples to determine rangeland stress patterns.

6.7 Classification accuracy assessment

An accuracy assessment process was undertaken to validate the vegetation classes. One hundred sample points were generated by ERDAS Imagine 2015 using the stratified random sampling method. Their GPS readings were taken using the centimetre level precision Ashtech®ProMark2™ Global Positioning System (GPS) receiver and then visited in the field. The accuracy of classification of the random points was verified based on the field visits, utilising the error matrix analysis algorithm built into the ERDAS Imagine 2015 software. Estimates of error were provided by module results, i.e. Error of Omission - occurs when pixels of class 1 were wrongly assigned to class 2, and from class 1 perspective, these pixels should have been assigned to class 1 but were omitted. Error of Commission - occurs when pixels of class 2 were wrongly assigned to class 1, and from class 1 perspective, these pixels should not have been assigned to class 1 but were included. The pixels classified correctly are found along the diagonal of the confusion matrix table, which lists the number of pixels that were classified into the correct ground truth class.

For the 2006 image class 4 (grass in leaf) had the highest user's accuracy (100 %) due to the least inter-class confusion, while the highest producer's accuracy generated by class 2 (senescent grass 1) was recorded at 93.1 % (Table 6.1). The overall accuracy and Kappa coefficient were 85.0 % and 75.9 %, respectively.

Table 6.1: Error (confusion) matrix for classification of the 2006 image

Classification data	Reference data					User's accuracy (%)
	Class 1	Class 2	Class 3	Class 4	Total	
Class 1 – Trees	44	2	3	1	50	88.0
Class 2 – Senescent grass 1	6	27	2	0	35	77.1
Class 3 – Senescent grass 2	1	0	11	0	12	91.7
Class 4 – Grass in leaf	0	0	0	3	3	100.0
Total	51	29	16	4	100	
Producer's accuracy (%)	86.3	93.1	68.8	75.0		
Overall accuracy = 85.0 %						
Overall Kappa coefficient = 75.9 %						
Producer's accuracy = error of omission						
User's accuracy = errors of commission						

Table 6.2: Error (confusion) matrix for classification of the 2010 image

Classification data	Reference data					User's accuracy (%)
	Class 1	Class 2	Class 3	Class 4	Total	
Class 1 – Trees	32	7	0	0	39	82.1
Class 2 – Senescent grass 1	1	31	2	1	35	88.6
Class 3 – Senescent grass 2	1	1	16	0	18	88.9
Class 4 – Grass in leaf	0	0	1	7	8	87.5
Total	34	39	19	8	100	
Producer's accuracy (%)	94.1	79.5	84.2	87.5		
Overall accuracy = 86.0%						
Overall Kappa coefficient = 79.8%						
Producer's accuracy = error of omission						
User's accuracy = errors of commission						

For the 2010 image class 3 (senescent grass 2) had the highest user's accuracy (88.9.0 %) compared to other classes, while class 1 (trees) showed the highest producer's accuracy (94.1 %) (Table 6.2). The overall accuracy was at 86.0 %, and the overall Kappa coefficient was 79. 8 % (Table 6.2).

For the 2014 image Class 3 (senescent grass 2) had higher user's and producer's accuracies compared to the 2006 and 2010 images and this implies that the inter-class confusion were lower (Table 6.3).

Table 6.3: Error (confusion) matrix for classification of the 2014 image

Classification data	Reference data					
	Class 1	Class 2	Class 3	Class 4	Total	User's accuracy (%)
Class 1 – Trees	36	3	0	0	39	92.3
Class 2 – Senescent grass 1	3	27	1	3	34	79.4
Class 3 – Senescent grass 2	1	0	21	0	22	95.5
Class 4 – Grass in leaf	0	0	1	4	5	80.0
Total	40	30	23	7		
Producer's accuracy (%)	90.0	90.0	91.3	57.1		
Overall accuracy	= 90.0%					
Overall Kappa coefficient	= 85.6%					
Producer's accuracy = error of omission						
User's accuracy = errors of commission						

The 2014 image also had the highest overall accuracy and Kappa coefficient, recorded at 90.0 % and 85.6 %, respectively.

6.8 Discussion and conclusion

This section reviews the aim and set objectives of the study, evaluate what has been achieved as regarding the set goals and provides a synthesis of the approaches adopted for this study. The conclusion as well as limitations and recommendations for future applications of remote sensing and geostatistical techniques for the estimation of savanna woody vegetation cover will also be provided. The good correlation results obtained from texture analysis of the 20 m resolution SPOT 5 multispectral images further confirm the recommendations made by Hudak and Wessman, (1998). In the study, the authors tested the performance of several image resolutions and found that texture measures calculated on simulated 20 m resolution SPOT image were capable of characterizing savanna woody cover more accurately. Texture measures calculated on the 10 m resolution SPOT 5 multispectral image yielded the highest correlation with the field observations. Hence, the performance of texture measure is dependent on the spatial resolution of the original imagery that the texture index is calculated from.

Several potential sources of uncertainty exist in this analysis include a non-uniform atmosphere, variable sun/target/sensor geometry, and temporal dynamics in land cover

properties themselves. The unquantified uncertainty underlying our assumption of minimal vegetation change at the study site is most problematic but the use of the benchmark site is not entirely unprecedented. In another study, Hudak and Wessman (2000) used a control site to standardize vegetation indices between multi-temporal SPOT satellite images and obtained reasonable estimates of woody cover change. However, further tests clearly are still needed to assess the broader applicability of this technique. Univariate linear relationships between land survey and remotely sensed data are often the most practical validation method.

On the other hand, the bush density patterns reflect the distribution of the dominant species types in Disaneng which are the major determinant of vegetation composition across this landscape. These same patterns were evident in the 2006, 2010 and 2014 as indicated in Figures 6.1–6.3, respectively, serving, as a form of replication that strengthens the confidence in image texture as a good indicator of relative bush densities across a landscape. Raw textural values were lower in the SPOT 2 images than in the SPOT 4 – and 5 simulation images at the same 10 m resolution. Atmospheric scattering effects on SPOT imagery probably explain this reduced contrast in SPOT data relative to the land survey. Hudak and Wessman (1998) concluded that the radiometry of multi-temporal SPOT panchromatic images should be approximately equivalent and should, therefore, allow monitoring of canopy structure. Indeed, raw textural values from the 2006 and 2014 SPOT images used in this analysis proved very similar (Figures 6.1–6.3). The decline in woody density along the road than further inwards away from the road from 2006 to 2014 suggests differential change in vegetation cover, which may relate to extensive veld unrest in 2006–2014 to control bush encroachment.

If one assumes that canopy structure along the road has changed little between 2006 and 2014, then the larger decrease in woody cover inside the study site translates into an absolute loss in percent woody cover of 0.75% based on the above calculations. Although far too minor to consider definitive, this is a reasonable gesture given that the average woody plant mortality rate is 1.3% (Brockett, 2014). If the bush encroachment rate is linear and extrapolated back one hundred years, when white settlers arrived and current grazing practices are believed to have commenced, then it appears that bush densities have approximately doubled during the past century. However, we have no evidence that bush encroachment is a linear or nonlinear process. Two earlier studies in neighbouring

Botswana by Moleele *et al.* (2002) demonstrated the potential for higher initial rates of bush encroachment. In these studies overgrazing experiment on a 10×10 m plot and was conducted and a doubling in percent woody cover was induced in recent years. Hudak (1999) estimated a tripling in woody biomass from 1950 to 1975 over a 108 km² area, beginning in a savanna with very little woody cover.

CHAPTER 7: GENERAL DISCUSSION, CONCLUSION AND RECOMMENDATIONS

7.1 General discussion and conclusion

Senegalia mellifera (50%) was the dominant woody encroacher and it was found in all the study sites. *Vachellia totilis* contributed 18% of the total woody plant density whereas *Grewia flava* and *Ziziphus mucronata* were both recorded at 16% of total woody plant density. *Senegalia mellifera* is an aggressive encroacher that is shade intolerant, and thickens easily when overgrazing of the herbaceous layer occurs. This gives *Senegalia mellifera* the ideal prospect to thicken and expand in the region, especially where there are deep, sandy soils. These woody plants grow well in deep sandy soils where they not only use the surface waters but also the sub surface moisture. The study sites recorded woody plant densities of 9 640 -, 6 461 - and 7 820 TE ha⁻¹ respectively with an average of 7 974 TE ha⁻¹. The reference site (benchmark site) recorded a woody plant density of only 412 TE ha⁻¹.

The findings of this study compare well with findings by Molatlhegi (2008), Mogodi (2009) and Comole (2014) where *Senegalia mellifera* also prevailed as the most abundant encroacher. According to Moore and Odendaal, (1987) the average woody plant density of 7 974 TE ha⁻¹ shows that grass growth was almost totally suppressed (Table 4.2).

The woody plant development strategy was not successful, due to changing environmental factors. Site 2 (II) showed an increase in total woody plant (especially *Senegalia mellifera*) density from 2014 to 2015, while sites 1 (I) and 3 (III) indicated a general decrease in woody plant density during the same time interval. This can be because of various factors such as tree cutting by local people, browsing by goats, removal of woody plants, especially along the main roads etc. Coppicing of browsed woody plants was not measured during this study. Although the coppicing of different woody species would be different the coppicing species would restore the canopy cover but the reducing of trees in different height classes would be reduced. Monitoring changes in bush thickening over a single season (2014 to 2015) seems to be too short a period and a longer time interval is needed to monitor changes in the woody density. The growth rate of woody species is too generalized, depending on the soil conditions. *Senegalia mellifera*, growing on sandy soil

(with a greater leaching ability) where the study sites were located, will probably grow faster in clay or rocky soil.

Remote sensing imagery shows that from 2006, 2010 and 2014, there was a high (41.22%) woody species change between 2010 and 2014 compared to the lower (4.72%) between 2006 and 2010. This could be indicative of the increase in atmospheric [CO₂]. However, the correlation coefficient ($r = 0.99$) between remote sensing and land survey shows that woody plant invasion occurred. The technique to measure tree heights and then compare the same trees' development the following year, seems to be impractical due to 'changing environmental factors such as droughts, higher than expected rainfall etc.

Various factors such as overgrazing, mismanagement, the question of land tenure and improper farming activities are contributing to bush thickening, while it was concluded by O'Connor *et al.* (2014) that increased atmospheric [CO₂] is the main driver of bush thickening. Woody plant encroachment will always be part of savannas and management measures will not be able to eradicate woody plants completely, especially invasive *Acacia* (*Senegalia* and *Vachellia* spp.) species.

An invasive species is defined as a species that is non-native to the ecosystem under attention. All the encroachers in this study (*Senegalia mellifera* and *Vachellia tortilis*) were, however, indigenous to the area.

7.2 Recommendations for future management of bush thickening

Hagos and Smit (2005) proposed a partial eradication of *Senegalia mellifera*. This is because a part of the advantages of woody species, especially their ability to enrich the soil nutrient status, must be taken into consideration before any bush clearing program is initiated. Another management option would be to introduce more browsers, such as boer goats to the area in order to eradicate the dense vegetation. Goats however, do also graze and contribute to overgrazing (one of the causes of bush thickening), especially on woody plants lower than 1.5 m that is being considered as the browsing height of goats. Land users should be encouraged to preserve large trees (structured savanna), which is key to finding long term solution to the problem of bush thickening. Large trees are able to suppress establishment of new seedlings, while maintaining other benefits of woody

plants such as soil enrichment and the provision of food browsing herbivore species. This needs to be done on a large scale and in areas where the woody plants are still at a height which would be accessible to browsers. This management should be integrated with management through farsighted and sustainable control of thorn tree species by combining prevention of spread through eradication and full management of the resource. Future research on bush control should assess animal management and usage, because of their immense contribution to bush thickening and success of control.

The assessment of area-specific environmental factors is vital for area-specific interaction and influence on bush thickening. This will also aid ground truthing activities when using satellite images to determine trends of encroachment on a larger scale.

It is recommended that alternative control options be used rather than only the traditional approach of eradicating thorn species. The control options should incorporate chemical, mechanical and biological means as well as proper livestock management strategy because total eradication of the species is costly and complex. However, for this to realize, the people of the land must gain property rights.

The initial control program should target trees above 2 m because they prohibit movement and complicate management. This means that initial control methods should be carried out in the early stages of the problem and that aftercare be implemented if the trees are still small after re sprouting. The control program should form part of farm management with great emphases on aftercare treatment of controlled regions. The short-term rehabilitation approach should be aimed at eradication through labor intensive methods such as soil-applied chemicals and or mechanical methods. This will contribute to job creation, capacity building and livelihood improvement initiatives. The biological control strategy that reduces seed germination should then be incorporated in the after care of eradicated camps. This, however, will be challenging due to the fact that inhabitants do not have property rights in a communal system.

The Government through Conservation of Agricultural Act, Act No. 43 of 1983 and National Environmental Management Biodiversity Act, Act No. 10 of 2004 should provide incentives to these affected farmers for removal of *Acacia* species (*Senegalia* and *Vachellia* spp.) control and avoid further spreading of these plants to other farms. The

influence of demographic conditions such as soil and climatic impacts, as well as land type use on spreading and control success of *Acacia* and other opportunistic encroachers should further be researched.

With regard to remote sensing analysis, it is recommended that high-resolution data should also be used in areas where there is a high capital outlay in terms of bush control. This can greatly reduce the volume of fieldwork necessary for quantitative work.

Bush thickening as described earlier generates different benefits and consequences. Below is a list adopted from Wigley *et al.* (2009). Bush thickening advantages and disadvantages can be summarized as follow:

Advantages:

- a) Increase in woody biomass produces more building materials,
- b) Increase in woody biomass produces more fencing materials,
- c) Increase in woody biomass produces more firewood resources,
- d) Increase in woody biomass produces more browse for browsers at the expense of grazers,

Disadvantages:

- a) Increase in woody biomass produces less grass for grazers,
- b) Increase in woody biomass lead to loss of biodiversity,
- c) Increase in woody biomass creates poor visibility for game viewing,
- d) Increase in woody biomass leads to loss of grass layer and exclusion of fire and**
- e) Increase in woody biomass increases the threat of invasion by opportunistic alien invaders.**

It, however, seems that bush thickening as a whole, can be considered to be negative succession as the disadvantages outweigh the potential advantages. It is, especially, the negative effect it has on the herbaceous development that shifted the scale into the negative. It can thus be concluded that bush thickening needs to be monitored and controlled on a regular basis.

Bush thickening in communal land tends to generate mixed feelings. Due to human population increase in communal land, woody species provided much needed building materials and heat energy resources, browse for goats and cattle. Goat rearing has become more important in communal areas since it is relatively less energy intensive and requires less capital investment. Poorer households are able to farm with goats with ease.

In both commercial and communal land, bush thickening has been seen as a form of land degradation. Bush thickening has reduced farming income and field carrying capacities. In order to implement the best possible control strategy for the specific bush encroacher, managers need to understand that an increase in woody plant species abundance is influenced by two processes:

- (i) A vegetative growth through existing biomass and,
- (ii) An increase in density through reproduction (Smit, 2004).

Depending on how bush thickening control tools and strategies are used, these tools and strategies can impact the species abundance processes simultaneously or separately. The most efficient bush control tool and strategy should be the one that aims to manage both the vegetative and reproduction processes simultaneously and should be able to be deployed in different seasons. A combination of control strategies and tools are likely to be most efficient. Different tools and strategies have been used over the years to manage bush thickening. These methods *inter alia* include the use of fires, introduction of browsers, chemical control and mechanical clearing. Fires application is required to reduce the height of woody plants to be within the reach of browsers especially goats but due to the lack of enough grass (fuel) hot fires are unlikely in this area. Selective thinning /harvesting *via* thinning or by fire application would eventually reduce competition between trees and grasses and also among remaining trees. However, in order to implement any control mechanism, property rights in communal areas must be restored, so that **‘who ever owns the land, must rehabilitate the land’**.

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