Monitoring of bush encroachment along selected sites of Disaneng, North West Province, South Africa

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DECLARATION

I, Funanani Patricia Begwa (24703648), hereby declare that the dissertation titled: Monitoring of bush encroachment along selected sites of Disaneng, North West Province, South Africa, is my own work and that it has not previously been submitted for a degree qualification to another university.

Signature:...**Date :** 18/10/2017

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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

I dedicate this work to the Almighty God for his protection, his anointing and for blessing me with the gifts of the Holy Spirit, Wisdom, Knowledge and Understanding. Thank you Lord for your tender mercies and favor.

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LIST OF FIGURES

LIST OF TABLES

List of Acronyms

SPOT: Système Pour l'Observation de la Terre

SANSA: South African National Space Agency

GIS: Geographic information system

GCP: Ground control points

AOI: Area of interest

ETTE.ha^{-1}: Evapotranspiration Tree Equivalents per hectare

ABSTRACT

Bush encroachment has been frequently documented in arid and semi-arid environments compared to temperate regions. In these regions, bush encroachment results in dense thickets, mostly composed of thorny or unpalatable bushes, which have negative impact on the carrying capacity of an area and thus the rangelands economic value. Encroachment by woody vegetation into grass dominated landscapes is common in savanna environments. The study area is located at selected sites of Disaneng which is situated at the North West Province, in South Africa, the area is a semi-arid Savanna biome and this biome supports a large population that depends on grazing livestock.

Bush encroachment is currently regarded as a major threat to the agricultural production in savannas as it suppresses grass production. It is not easy to reverse the process of bush encroachment but to control changes in abundance of Savanna trees is possible. The use of Geographical Information System (GIS) and Remote Sensing techniques has become useful in monitoring and quantifying trends in vegetation state.

In this research, the woody vegetation was analyzed following the variable quadrat method by Coetzee and Gertenbach (1977) and using the remote sensing technique. The extent of woody plant encroachment was quantified at the selected sites of Disaneng and in the benchmark. The canopy diameter of each woody species was measured and recorded. The tree densities for each species at different height levels were calculated and expressed as Evapotranspiration Tree Equivalent per hectare (ETTE. ha^{-1}). The data collected using the variable quadrat method was used as baseline data to confirm the remote sensing data to monitor change over time. The prominent species identified in both the study area and the benchmark included *Senegalia mellifera*, *Vachellia tortilis*, *V. erioloba*, *Grewia flava* and *Ziziphus mucronata*. The

study area produced woody plant density of more than 2 000 ETTE. ha^{-1} that according to Moore and Odendaal (1987), the woody species almost totally suppressed grass growth.

Key words: Bush encroachment, Remote sensing, Rangelands, Overgrazing.

TABLE OF CONTENTS

CHAPTER 1

1.1 INTRODUCTION

Monitoring the quality and condition of native vegetation is a critical component of ecological studies and planning processes (Parkes *et al.*, 2003). Vegetation transformation and deterioration are the main problems experienced in rangelands (Saco *et al.*, 2006). Vegetation status can change over time, responding to different land-use practices, management systems and soil erosion (Oztas *et al.*, 2003). According to Economics of Land Degradation (ELD) Initiative and United Nations Environment Programme (UNEP) (2015), land degradation and desertification are among the world's greatest environmental challenges. It is estimated that desertification affects about 33% of the global land surface, and that over the past 40 years erosion has removed nearly one third of the world's arable land from production. Land degradation and desertification are common throughout the continent of Africa, and it is the most severely affected region (Hoffman & Ashwell, 2001). Desertification affects approximately 45% of Africa's land area, with 55% of this area at a very high risk of further degradation (ELD Initiative & UNEP, 2015). Each year, approximately 20 000 000 hectares of agricultural land become degraded due to overgrazing for crop production (UNEP, 2006)

Globally, economic losses due to rangeland vegetation degradation are estimated to be approximately US $7 per hectare (Arntzen, 1998). In South Africa, the value of the fynbos ecosystem have reduced by over US $11.75 billion due to rangeland vegetation degradation, that the total cost of encroachment would be US $3.2 billion on the Agulhas Plain alone and that the cost to clear encroaching species is around US $1.2 billion (Van Wilgen *et al.*, 2001). According to Saco *et al.* (2006), approximately 16% of the land in Africa experienced landscape transformation from natural vegetation to bare and degraded soils. In South Africa, roughly 91% of the land is already prone to land degradation

(Hoffman & Ashwell, 2001), a large part of which corresponds with the distribution of communal rangelands (Lesoli, 2011).

Rangeland alteration and deterioration is intricate, varying in line with spatial and sequential components; thus an assessment is imperative from ecological and anthropogenic perspectives, on both areas and temporal bases (Saco *et al.*, 2006). According to Palmer & Ainslie (2002), rangelands are defined as natural or semi-natural ecosystems mainly characterized by indigenous, natural vegetation mainly savannas which cover large land areas in Africa. Bush encroachment is identified as an indicator of land degradation (Gibbens *et al.*, 2005; Maestre *et al.*, 2009 & Van Auken, 2009). Land degradation is defined as the reduction or loss in the biological or economic productive capacity of the land resource base and is generally caused by human activities, natural processes and often increased by climate change and biodiversity loss (ELD Initiative & UNEP, 2015). According to Hoffman & Ashwell (2001), land degradation can mainly be ascribed to factors such as; poor grazing practices, incorrect use of fire, the clearing of woody plants, poor soil conditions such as erosion, mining industry and urbanization.

Bush encroachment is experienced worldwide (Oldeland *et al.*, 2010 (a)). It has been reported as a major land management issue affecting conservation agencies, and both public and private landowners (Ward, 2005). It has also been of concern to land managers in grasslands and savannas, although Archer *et al.* (2009) concentrated more on the effects of woody plants on grass production, instead of the underlying ecological mechanisms driving encroachment. According to Oldeland *et al.* (2010 (b)), bush encroachment is a conversion of a landscape dominated by grasses to a landscape dominated by trees through plant succession. In this case, tree and shrub cover increase in the area at the expense of grass cover (Tainton, 1999). An expansion in woody plant density suppresses herbaceous plant production, because of high competition for available soil water. Consequently, a shift in location of the vegetation from a grass dominated ecosystem to a woody dominated ecosystem occurs (Schlesinger & Pilmanis, 1998). Species richness and diversity change as herbaceous species (mainly grasses) are replaced by woody plants (Wiegand *et al.*, 2006). This procedure is considered to be a result of ground or below the surface competitive capacity of grasses that are subjected to grazing (Brown & Archer, 1999). The canopies of these trees produce shade that begins to stunt and kill grasses (Belsky, 1990). However, Smit (2004) stated that the soil enrichment under the canopies of trees may have a positive effect on grass growth. Trees only have positive effect on grass growth when found in low densities. It was indicated by Skarpe (1991) that there is intra-species competition between woody species, causing density-dependent mortality and leading towards regular spacing between individuals.

Despite the recognition of woody plant encroachment as a worldwide management problem, information about the rates and dynamics of this circumstance, or its impact on fundamental ecological processes of ecosystems related to energy flow, biogeochemical cycling and biodiversity is still very sparse (Archer *et al.*, 2000). Brown and Archer (1999) also stated that there is little knowledge regarding changes, rates, patterns or successional processes involved in this process. Grass-dominating ecosystems become colonized by hardy, pioneer tree species. Bush encroachment results in habitat degradation (Ward, 2005) by causing reduction in the diversity of habitats, and this in turn reduces biodiversity as a whole (Meik *et al.*, 2002).

Globally, an increase in woody species changes carbon and nitrogen sequestration and nutrient cycling substantially (Archer *et al.*, 2000), which, in turn, have serious consequences for climate change. An increase in carbon dioxide concentration has recently been considered a key global change impact for describing an increase in woody species worldwide (Polley *et al.*, 1992). In South Africa, the role of increased atmospheric carbon dioxide concentration on woody vegetation growth has been noted (Bond 2008; Bond & Midgley, 2012). Bush encroachment also affects agricultural productivity and savanna biodiversity, with 10 to 20 million hectares in South Africa being affected (Ward, 2005).

According to Grossman & Gander (1989), bush encroachment has made 1.1 million hectares of South African savanna unusable, posing a threat to another 27 million hectares, and has, consequently, reduced the grazing capacity throughout the country by up to 50% (Hudak, 1999). The process of bush encroachment has widely been documented and has serious implications for livestock production systems, wildlife habitat etc. (Ward, 2005).

Bush encroachment has been frequently documented in arid and semi-arid environments, compared to temperate environments (McGlynn & Okin, 2006). In these regions, bush encroachment results in dense thickets, mostly composed of thorny or unpalatable bushes, which have a negative impact on the carrying capacity and thus the rangeland economic value (Mampholo, 2006). In arid and semi-arid environments, bush encroachment can be exacerbated by uneven and unreliable rainfall patterns (Schlesinger & Pilmanis, 1998). In studies conducted by Behnke & Scoones (1993) in semi-arid rangelands it was noted that developing sufficient agriculture depends on enhanced understanding of the spatiotemporal variation distinguishing semi-arid ecosystems.

Encroachment by woody vegetation into grassland dominated landscapes is common in savanna environments (Wiegand *et al.,* 2006). The study area (Figure 3.1) is located within the Savanna Biome and this biome supports a large population that depends on grazing livestock. This reduces the carrying capacity of the rangelands. Symeonakis and Higginbottom (2014) also noted that some savanna regions have already been entirely encroached by woody vegetation. Ward (2005) indicated that, in the past 50 years, the Savanna Biome has been altered by bush encroachment worldwide. This is currently having a negative impact on wildlife and the sustainability of pastoral, livelihood and commercial livestock grazing. This has allowed a large proportion of savanna landscape, important for livestock farming, to shift towards being forested area (Symeonakis & Higginbottom, 2014). This situation makes the area increasingly drought sensitive. According to Richter & Meyer (2001) it is not easy to reverse the process of bush encroachment but to control adjustments in abundance of savanna trees is possible. Hence, the capacity to quantify the spatial distribution of plants could specify the current state of change from one state to another in inclusion to procedures of degradation in semiarid mixed-shrub grasslands.

Over the last few decades, researchers have substantially benefitted from Earth Observation data (Symeonakis & Higginbottom, 2014). Accordingly, Earth Observation data, accompanied by its substantial spatial coverage and historical catalogue of data has been regarded as an information source that could greatly expand our understanding of vegetation dynamics in semi-arid rangelands by making it possible for us to test ecological theory on land degradation (Trodd & Doughill, 1998). Geographical Information System (GIS) and remote sensing (RS) approaches have become prevalent in monitoring and quantifying trends in vegetation state, merging to form a powerful information extraction and analysis tool for monitoring both vegetation quantity and state. Remote sensing, particularly, has become the most useful tool in analysing the biometrical properties of vegetation, utilising different wavelengths of the electromagnetic spectrum (Jarocinska & Zagajewski, 2009) and evaluating vegetation status. According to Adam *et al.* (2009), remote sensing techniques provide timely, up-to-date and accurate data, which allow sustainable and effective management of rangeland vegetation.

The advance in sensor's spatial and spectral properties similar with plant cover and landscape composition have presented a better mechanism for assessing bush encroachment over a large area within a short period of time (Okin *et al.*, 2001; Okin & Painter, 2004; Bradley & Mustard, 2005).

1.2 PROBLEM STATEMENT

Communal rangelands in Disaneng Village play an important part in the livelihoods of local inhabitants. The capability of these communal rangelands to meet the requirements of supporting or garnering livelihoods is placed on the integrity of soils and ecological conditions (Ward, 2005). However, these rangelands have undergone severe transformation in recent years (Dougill & Thomas, 2001). Grass-dominating areas have been intensively invaded and degraded by encroaching woody species. This situation was discovered in other studies conducted in the Molopo area by Molatlhegi (2008), Mogodi (2009) and Comole (2014). As a result, the natural flow and biodiversity has been reduced.

Because of this encroachment, grass development is limited, and, consequently, livestock are severely affected. If this problem is not addressed, the wellbeing and economic development of this village will be under threat. Bush encroachment transforms the agricultural capacity and biodiversity of 10-20 million hectares of South Africa and causes the carrying capacity to decline. Grazing capacity in this area by far exceeds the stocking rate of the area. The carrying capacity reduced from one Large Stock Unit (LSU) per 16 hectares to 1 LSU per 18 to 20 hectares (Department of Agriculture, Conservation, Environment and Tourism, 2002). Against the above mentioned background, the current study sought to investigate the condition and extent of bush encroachment in the study area. According to Joubert *et al.* (2008), unsuitable fire management, poor understanding of savanna ecosystem models and poor understanding of the phenology and physiology of the encroaching species, have increased the bush encroachment problem. The substitution of wildlife with livestock has reduced the growth rates of grasses and consequently increased the rate of bush encroachment. Grazing pressure has predictable results. The normal ration of unpalatable to palatable grasses for each different plant community will change and favor the increase of unpalatable speies if the grazing pressure is high (Walker, 1987).

1.3 HYPOTHESIS OF THE STUDY

The hypothesis of the study states that "encroachment of woody plants into grasslands poses a threat to grass development and farming activities are continuously under threat".

1.4 AIM OF THE STUDY

The aim of this research was to investigate the extent to which bush encroachment has occurred over time at selected sites of Disaneng.

1.5 THE OBJECTIVES OF THE STUDY

The purposes of this research were:

- To establish woody species composition in different height classes and bush density in the study area compared to the benchmark site.
- To monitor woody plant changes in the study area and to detect change over time using satellite images ranging from 2000-2009.

CHAPTER 2

LITERATURE REVIEW

2.1 RANGELAND CONDITION

Rangelands are natural or semi-natural ecosystems mainly characterized by indigenous (species that are not introduced to the area but originate from that area), natural vegetation which cover large areas of land (Palmer & Ainslie, 2002). It consists of native or naturalized species wherein the management is restricted to grazing, burning and control of woody species (Ward, 2005). In most African countries a large population that depends on livestock grazing is supported by half the land area of rangelands. The larger part of rangeland ecosystems are situated in grasslands, shrub lands, savannas and even deserts. The vegetation is characterized by an inherent arid climate that experience large daily and seasonal temperature extremes (Vetter *et al.*, 2006; Archer & Tadross, 2009).

It is difficult to establish the causes of change in rangeland state due to site-specific interactions among ecological features such as soil type, climate, competition among plant species and humans use such as grazing their herds and burning of vegetation. Controversy has focused on whether climate or consumers are primarily responsible for vegetation dynamics encountered in arid and semi-arid Africa (Milton & Dean, 1995). Continued heavy grazing contributes to the decline of palatable species, especially grasses, and the subsequent dominance by less palatable ones (McNaughton, 1985). According to O'Connor *et al.* (2014), rainfall and soil type are considered to be the primary determinants, with fire and herbivory regarded as secondary determinants of the physical nature of the African savannas and its functioning.

For or from whatever cause, range condition is low when desirable species are replaced by unpalatable species. It occurs when reduced soil cover exposes excessive bare surfaces, when erosion increases; when production of forage and animals gets reduced or when any combination of these effects occurs (FAO, 2000).

2.2 RANGELAND DEGRADATION

Rangeland degradation is identified as an expansion in biomass and abundance of woody plant species, often unpalatable plants, coupled with the suppression of herbaceous cover (Ward, 2005). Change in the pattern and condition of vegetation, as defined by patchiness and biodiversity in semi-arid regions, are the main indicators of the condition of land degradation (Saco *et al.*, 2006). Ayoub (1998) suggested that excessive grazing causes change in pasture composition, invasion of woody weeds, a decrease in total vegetative cover and an increase in soil erosion. Total absence of grazing decreases the biodiversity value because an abundant canopy of shrubs and trees establish which prevents light and moisture from reaching the soil and results in over-protected plant communities, which are affected by natural disasters such as veld fires (Vetaas, 1992). This expresses that one of the major aspects of rangeland degradation is the decrease in the capacity of the ecosystem to support livestock production and productivity (Saco *et al.*, 2006).

Oba and Kotile (2001) reported that the major threats in rangelands were results of loss of perennial herbaceous cover and an increase in encroaching woody species. Leopold (1989) verified the growth of less palatable grasses, shrubs and weeds and the emergence of unstable equilibrium in rangelands due to overgrazing. This is because of the suppression of palatable grasses due to woody species that are unpalatable to domestic livestock (Ward, 2005). According to Abate and Angassa (2016), the threats to rangelands include overgrazing, invasive plants, human population increase, soil erosion and desertification.

Concerns in Africa's rangelands due to overstocking have persisted over a long period of time (Mack, 1996). Thus, if the rangelands are not properly managed, the rangelands are suspected to eventually become degraded. Cheng *et al.* (2011) stated that land degradation is associated with overgrazing which promotes an increase in undesirable herbaceous species and bush encroachment. Reed *et al.* (2011) stated that pastoralists play a major role in rangeland degradation since they rely on the savanna ecosystem for their livestock grazing; therefore, their contribution towards the management of rangelands is vital since they have extensive ecological knowledge of the environment.

Many of the world's poor areas benefit from the rangelands on resources such as food, water and livelihoods, it is therefore vital to be concerned about rangeland conditions. Bedunah & Angerer (2012) stated that the causes of rangeland degradation are complex and associated with interactions between pastoralists, governance and policy, as well as environmental determinants wherein it is difficult to separate the interaction between climate-induced and human-induced decline. Overgrazing, unsustainable wood fuel use, mining and ploughing of rangelands with subsequent loss of soil productivity are considered causes of rangeland degradation. Policies, socio-economic changes and interactions of socio-economic and governance factors with climatic stressors such as drought are considered the primary drivers of rangeland degradation (Bedunah & Angerer, 2012). According to Wiley (2008), conflicts over collective assets such as communal grazing lands appear to occur because of interethnic and religious differences but often the more fundamental conflict is between people and their governments associated with rights and powers over property.

2.3 BUSH ENCROACHMENT AS A FORM OF LAND DEGRADATION

The permanent loss of a rangeland's biological or economic productivity in an arid and semi-arid environment is regarded as degradation and bush encroachment is known to be a form of rangeland degradation globally (Kgosikoma *et al.*, 2012). According to Oba *et al.* (2000) and Richter *et al.* (2001) bush encroachment is presently regarded as a greater threat to the agricultural production in savannas as it suppresses grass production. Bush encroachment was also noted to have emerged during 1952-1960 in Borama (Ethiopia) as major range degradation characterized by invasion of undesirable woody species and unpalatable species, loss of grass layer and increased soil erosion (O' Connor *et al.*, 2014). According to Wigley *et al.* (2010), increases in woody plant cover in savanna ecosystems have a consequence on land-use and conservation.

Archer (1994) stated that, during the past century, the balance among plant life forms has shifted to favor woody species in several tropical savanna ecosystems. Possible factors contributing to these life form transitions include changing land-use practices such as high stocking rates and associated excessive grazing (Fensham *et al.*, 2005), incorrect burning strategies practices (Briggs *et al.*, 2005), elimination from fire and grazing (Oba *et al.*, 2000), altering climate and rainfall (Fensham *et al.*, 2005), atmospheric nitrogen deposition (Brown & Archer, 1989) and elevated carbon dioxide (CO_2) (Hoffmann *et al.*, 2000; Wigley *et al.*, 2010). Elevated CO_2 decreases the transpiration rate of grasses, resulting in deeper infiltration of water to the sub-soil and, thus, favoring woody plants (Bond & Midgley, 2000). Carbon dioxide concentration also influences the photosynthetic rates of plants, light and nutrient-use efficiency (Drake *et al.*, 1997). According to Polley *et al.* (1992) and Archer *et al.* (1995), increasing atmospheric carbon dioxide concentration has emerged, in recent decades, as a key feature resulting in changes impacting on bush encroachment across the globe.

The suppression of grass production is due to microclimatic adjustment and extreme competition for attainable soil water and nutrients (Richter *et al.*, 2001; Hudak *et al.*, 2003; Satti *et al.*, 2003). Bush encroachment is the invasion and thickening of encroaching woody species, resulting in the reduction of palatable grasses and herbs (Ward, 2005).

According to O'Connor *et al.* (2014), bush encroachment is the increase in the cover of indigenous woody species in savannas. It results in habitat degradation and loss of resource productivity. Woody species reduce grass cover as they compete with grass species for accessible water and nutrients, thereby reducing the sunlight from reaching the grass layer (Thurow, 2000). In extension to competing with grasses, most of the woody plants encroaching into rangelands results in the extreme reduction of grazing capacity (Alemayehu, 2005).

According to Donaldson (1967), bush encroachment is an extreme economic and ecological predicament in many arid and semi-arid parts of the world. It is a process whereby the woody layer of a savanna increases in density and cover to an extent that grass production is negatively affected and eventually leads to reduction in grazing capacity. The increase of woody species is due to competition between woody species and grasses for accessible water and nutrients and woody species reducing light reaching the grass layer (Thurow, 2000). This was illustrated in Walter's (1970) root niche separation model based on soil water as the limiting factor wherein grasses are shallow-rooted and trees are deep-rooted. Grasses are superior competitors for water in the upper soil layer, trees out compete the grasses due to the fact that they have been restricted access to water in the deep soil layers (O'Connor *et al.*, 2014). Buba (2015) stated that the functions of soils which include decomposition, nutrient cycling, soil respiration, invasion resistance and ecosystem stability supports the vegetation. Through cycling of dead biomass, soil biodiversity provides many ecosystem services essential to mankind and the environment such as the support of primary production, control of pests and diseases. Trees tend to affect properties of soil around them through litter-fall input which are relative to the tree species and sizes (Isichei & Muoghalu, 1992, Moody & Jones, 2000, Zhang *et al.*, 2011). According to Scholes & Archer (1997) as well as Ludwig *et al.* (2004) tree size is a factor that determines the extent of the impact around the tree's environments.

According to Skarpe (1990), the absence of grass allows woody encroachers like *Senegalia mellifera* to encroach and get established in savannas. Skarpe (1990) also stated that bush encroachment is generally discovered in areas that are excessively grazed wherein grasses are removed. Grazing is the most extensive form of land-use in southern Africa (Smit & Rethman, 1998). High grazing pressure is not the only cause of the transformation of areas dominated by grasses to areas dominated by woody species, the amount and distribution of rainfall also play a vital role. In cases or seasons where the rain is insufficient for the establishment of woody species seedlings, encroachment by woody species will not occur as seedlings are dependent on the shallow water. The establishment of woody seedlings and transformation of grasslands into savannas may only occur during high or average rainfall seasons (Joubert *et al*., 2012). This is especially true in the savanna areas where sandy soil is abundant. Water infiltration in sandy soils takes place fast and grasses have a limited root system and can only manage to extract water from the upper soil layers. Woody seedlings can then take advantage of the deeper water.

Howden *et al.* (2001) demonstrated that bush encroachment can also emerge in carbon appropriation, even though it reduces grass growth and livestock production, which may result in encroachment of woody species which becomes a new source of income to pastoral people in the making of charcoal. O'Connor *et al.* (2014) interpreted bush encroachment in association to key conceptual models including traditional explanations of fire and grazing, explicitly in terms of climate regime and soil type.

High grazing intensity decreases the amount of grass as ecological opponent to woody plants, thereby resulting in bush encroachment. In most situations, woody encroachment is considered to be because of lack of foraging by livestock and lack of fire (Herlocker, 1999), thus both overuse and under use have been suspected in influencing vegetation dynamics. Environmental changes leading to an increase in bush cover are presently correlated with episodic climatic events including recent and historical land use and water development (SORDU, 1990). According to O'Connor *et al.* (2014), the submerging of bore holes, ring fencing of farms, the removal of veld fires, together with the overutilization of valuable grass species have, within a period of 20 years, diverted the balance in favour of woody plants in southern African savannas.

According to Van den Berg (2007), bush encroachment is, in theory, reversible over a relatively short period, given sufficient knowledge, financial and other resources, such as fencing and the removal or reduction of livestock. However, the majority of land users especially in communal areas do not have sufficient resources, and cannot remove their livestock for periods long enough to allow the rangeland to recover, which leads to a decrease in the condition of the rangeland to an ecological state that has passed the threshold of natural recovery (Reed, 2004). If this change has occurred, active interventions are necessary to restore the degraded rangeland to a better state or system.

2.4 CAUSES OF BUSH ENCROACHMENT

A number of factors can cause bush encroachment, and these may range from naturally occurring causes to human-induced causes. The identification of these causes could provide a guide for management's responses. Rainfall, soil type and natural fire regime are considered the main naturally occurring features which influence bush encroachment (Kraaij & Ward, 2006). However, poor land management in the form of overgrazing, is the main activity which encourages the spread of woody encroacher species (Oldeland *et al.*, 2010 a). In South Africa, overgrazing has caused the replacement of palatable grass species by less palatable bushes and shrubs (Hudak, 1999). However, controlling bush encroachment remains a challenge, and this is exacerbated by insufficient information on the factors that cause bush-encroachment (Moleele *et al.*, 2002). Perpetual encroachment by *Senegalia mellifera* has been reported in many parts of savannas (Smit, 2004), while its reversal has not been reported. However, the necessity for monitoring the state of encroaching woody vegetation for proper management is imperative (Zerger *et al.*, 2009).

O'Connor *et al.* (2014), also stated that, if the effects of atmospheric [CO_2] is dependent upon other factors such as grazing, then management can perhaps influence other factors. The process of controlling bush encroachment relies on whether atmospheric carbon dioxide concentration [$CO2$] is a primary or secondary driver of bush encroachment.

Bush encroachment in rangelands (Ward, 2005) has been widely reported in southern Africa. The driving factors of bush encroachment in rangelands are often correlated with overgrazing because a constructive mutual relationship between grazing pressure and woody species cover has been observed in certain studies (Oba *et al.*, 2000; Buba, 2015). According to Smit (2004) an increase in the biomass of already established vegetation growth and an increase in density of the establishment of seedlings resulted in an increase in woody plant abundance.

Causes of bush encroachment include: improper grazing practices, lack of or misuse of fire, absence of browsing animals and lack of impact of huge animals such as elephants and elevated concentration of carbon dioxide levels in the atmosphere (O'Connor *et al.*, 2014). However, humans are regarded as major catalysts to the problem. People rely on the savanna ecosystem to graze their livestock and mismanagement of the area leads to loss of the palatable herbaceous layer (Kgosikoma *et al.*, 2012)

2.4.1 Anthropogenic factors

In arid and semi-arid African savannas, pastoralists are blamed for contributing to rangeland degradation and bush encroachment (Kgosikoma *et al.*, 2012). According to Long *et al.* (2010) and Morris (2010), anthropogenic activities have the potential to change a landscape structure and ecological function of the landscape over time. Several studies (Roba & Oba, 2008; Dabasso *et al.*, 2012) have suggested that integrating the knowledge of local communities would improve the current understanding of the mechanisms and causes of bush encroachment and rangeland degradation wherein the information will assist in providing a better understanding of the environment from the perspective of resource utilization.

Smit *et al.* (1999) and Smit (2004) defined huge cattle densities, the elimination of burning, the limitation of movement of herbivores by fences, poor land use and grazing management and the artificial watering points that their herds use for drinking water as examples of anthropogenic activities while Moleele *et al.* (2002) included kraals, dipping tanks (the animals trample and kill the herbaceous layer) and settlements as examples of anthropogenic activities.

2.4.1.1 Cattle grazing

Bush encroachment was found to occur in the southern Kalahari from cattle grazing beyond a threshold pressure under all rainfall scenarios (Jeltsch *et al.*, 1997; Weber *et al.*, 1998). Low grazing pressure had no effect on woody cover and distribution but an expansion in grazing pressure led to a constant increase in woody cover (O'Connor *et al.*, 2014). Reduced grass competition, combined with some years of relatively high rainfall, favour woody establishment (O'Connor *et al.*, 2014). In arid Kalahari savannas, *Senegalia mellifera* showed the greatest expansion in response to overgrazing (Skarpe, 1990). Overgrazing reduces the supremacy of grass species and favours the growth and multiplication of woody species due to the fact that the woody species can gain or have increased access to soil moisture (Skarpe, 1990). Grazing indirectly contributes towards an increase in encroaching woody species through dispersal of encroacher plant's seeds. The acquired organic matter and soil seed banks introduced by livestock from the grazing lands, which after dispersal, reproduce and grow into woodlands, establish patchy environments wherein grass swards are replaced by virtually impenetrable thickets of thorn trees (Reid & Ellis, 1995). Thomas (2008) stated that overgrazing in rangelands is associated with communal grazing due to mismanagement and uncontrolled grazing regimes since the area used for grazing is not owed by an individual but belongs to the tribal office. It was illustrated by Thomas *et al.* (2000) that there are limited studies to compare vegetation conditions between grazing management systems in the communal and ranching lands. Recent studies by Kgosikoma *et al.* (2012) showed evidence of grazing mismanagement in the communal and ranching lands.

Stuart-Hill (1988) and Scogings (2003) stated that browsing by ruminants stimulates bush growth. Grossman & Gandar (1989) argued that bush growth was kept in check by mega herbivores such as elephants and black rhino. Established bushes are mostly found growing near water establishing into important shade trees wherein animals rest beneath the trees. According to Moleele & Perkins (1998) trees benefit from nutrient enrichment of the soil from the dung of animals. Considering low availability of phosphorus in savanna soils, the dung from animals supplemented with phosphate improves soil fertility.

2.4.1.2 Fire suppression

Fire is regarded as a tool for directly influencing woody plant components of savannas and for managing them. The suppression of fire has a strongly positive effect on increasing woody plants. O'Connor *et al.* (2014) stated that fires were reported or considered to have been frequent and widespread across the savannas and grasslands of South Africa during the nineteenth century. Most fires were considered to be caused by origin and it was a common practice amongst indigenous people which was widely adopted by settlers (O'Connor *et al.*, 2014). Regular burning suppresses plant growth by suppressing the shrubs and juvenile trees, thus preventing their development into mature plants which will be resistant to fire and be out of reach of browsers (Mphinyane *et al.*, 2011). According to O'Connor *et al.* (2014), fire suppression should, therefore, promote an increase in woody vegetation at a rate determined by the growth potential site, which is influenced by mean annual rainfall. Fire exclusion has an unequivocal influence on the increase of the woody component across savanna types with rainfall ranging from 386 to 1300 mm per annum (O'Connor *et al.*, 2014).

Grasses that are overgrazed in savannas provide limited fuel load to enable continual burning at high intensity. Van Langevelde *et al.* (2003) suggested that an increase in the level of grazing leads to reduced fuel loads which make fires less intense to damage and to destroy trees and therefore, results in an increase in woody vegetation. It is therefore necessary to initiate sustainable burning intervals (Fatunbi *et al.*, 2008) and organizations such as Working for Water (WfW) must be involved in the burning programmes.

Fire does not necessarily affect production of the grass layer (Grossman *et al.*, 1981). However, grass species respond differently to fire. Annual burning combined with the effect of fire on soil moisture availability keeps the individual grasses small, leaving space for the colonisation of opportunistic woody species (Yeaton *et al.*, 1988). Fire has a strong negative impact on the survival, growth, adult recruitment and seedling regeneration of woody plants (Bond & Van Wilgen, 1996). Bond & Van Wilgen (1996) stated that fire intensity and the burning season can have different "event-dependent" effects on both the grass layer and woody layer. In a study by Trollope (1980) it was indicated that the burning of the aerial parts of woody species in semi-arid savannas of the Eastern Cape and Kruger National Park, South Africa, was not affected by fire intensity. The importance of fire for creating a demographic bottle-neck for seedling regeneration in open savannas or grasslands has long been emphasized (O'Connor *et al.*, 2014). A single fire can cause high seedling mortality. Joubert *et al.* (2008) recorded a 99% mortality of *Senegalia mellifera* seedlings in a burning experiment in a semi-arid Namibian savanna. It is, however, often necessary to introduce fires into an area for more than one season. Havstad & James (2010) reported that a single fire did not make a difference to woody cover percentage, either of a grass dominated or shrub dominated arid grassland in New Mexico, USA. It is, therefore, evident that fire suppression on its own can account for bush encroachment in mesic and moist savannas and is an important management tool in arid savannas.

2.4.1.3 Soil characteristics

Woody cover is negatively correlated with clay soil (Sankaran *et al.*, 2004; 2005), therefore, bush encroachment is mostly observed in sandy soil with low clay content. Intensive grazing reduces the herbaceous cover on sandy soils, so more water percolates to the subsoil and is available to the roots of woody plants.

According to Sankaran *et al.* (2008) a wide-scale investigation of woody species cover in African savannas disclosed that woody species cover was negatively correlated with soil nitrogen and hence, an increase in nitrogen deposition reduces bush encroachment. In a similar study by Sankaran *et al.* (2008), it was discovered that woody species cover had a complex and non-linear relationship with soil phosphorus. On the contrary, it was indicated by Roques *et al.* (2001) that the soil type had no significant impact on shrub dynamics in African savannas. It was indicated by Doughill & Thomas (2004) that the variability of nutrients in the soil is low but is increased by grazing-induced bush encroachment wherein the development of nitrogen-fixing biological soil crusts under bushes could enhance the competitive advantage of species like *Senegalia mellifera*, favouring further bush encroachment.

Scholes & Archer (1997) suggested that most semi-arid areas on mildly clay soils were "relatively" treeless in pre-colonial times, but were encroached quickly and seemingly irreversibly when grazed by cattle. This is in contrast to semi-arid environments on sandy, low fertility soils which are infrequently treeless. Sandy soils are more easily encroached by woody species than heavy soils because of a greater rate of water infiltration can potentially promote greater percolation to deeper layers (Walker & Noy-Meir, 1982).

2.4.1.4 Increased rainfall

Savanna ecosystems are generally water-limited and bush encroachment is associated with inter-annual rainfall variability (Angassa & Oba, 2007). A changing rainfall pattern from year to year acts as a primary regulator of woody seedling recruitment (O'Connor, 1995). It is, thus, anticipated that years of above average rainfall will support the recruitment of woody seedlings although this period is characterized by increased grass competition and occurrence of high intensity fires. In arid and semi-arid environments, woody species density tends to increase with increasing mean annual precipitation (Sankaran *et al.*, 2004). An unusually high annual rainfall in continuous years leads to an increase in bush encroachment while encroacher species like *Senegalia mellifera* requires at least 3 years of successive pleasant rainfall to grow successfully (Joubert *et al.*, 2008). Poor rainy seasons or droughts followed by years with above-average rainfall with frequent rainfall events have apparently made a considerable contribution to the problem of bush encroachment (Raj, 2005). Moisture availability is an important determinant of species composition (Tainton, 1999). Variable rainfall in arid areas influences bush encroachment because woody plants are more opportunistic, responding closely to rainfall events (Teague & Smit, 1992). Changes in tree abundance have been attributed to deeper rainfall penetration into the soil, favouring deeper-rooted trees (Ward, 2005). According to Tainton (1999), moisture availability is an important determinant of species composition.

Elevated soil moisture availability, specifically when the competition from grasses is limited, permits woody plant seedlings to survive and grow into bush thickets. Drought, through minimal plant development, seed germination and increased competition for limited water with the herbaceous species, leads to mortality of woody species (Roques *et* *al*., 2001) and thus reduces bush encroachment. As a result, bush encroachment is influenced by recruitment and death of encroacher plants in response to rainfall patterns (Wiegand *et al*., 2006). Higgins *et al*. (2000) anticipated that rainfall-driven variation in recruitment is more important in arid savannas, where fires are rarely severe and more frequent.

2.4.1.5 Climate change

Increasing atmospheric carbon dioxide concentration [CO2]and climate change are potential global drivers of bush encroachment (O'Connor *et al*., 2014). Several studies have concluded that atmospheric carbon dioxide concentration [CO2]is the primary cause of bush encroachment in South African savannas (Wigley *et al*., 2010; Buitenwerf *et al*., 2012; Ward & Russell, 2014; Ward *et al*., 2006). The degree of woody response depends on atmospheric carbon dioxide concentration [CO2](O'Connor *et al*., 2014). The evidence for the role of increased atmospheric carbon dioxide concentration [CO2]on bush encroachment derives from physiological understanding and empirical demonstration of its differential effect on the growth of C_3 (plants that go through the Calvin cycle taking carbon dioxide through the leaves 'stomata) versus C_4 plants (plants that reduce carbon dioxide captured during photosynthesis to useable components by first converting carbon dioxide to oxaloacetate, a four-carbon acid) (Bond & Midgley, 2012).

Human activities play a major role in the composition of the atmospheric layer surrounding the earth (Ayoub, 1998). The use of fossil fuels and land clearing increased the level of carbon dioxide in the atmosphere (Jackson *et al*., 1995). According to O'Connor *et al*. (2014) atmospheric carbon dioxide concentration [CO2]has increased from a preindustrial level of 277 ppm to 397 ppm in 2013 and its concentration is expected to continue increasing. It is, therefore, evident that an increase in atmospheric carbon dioxide concentration [CO2]is responsible for climate change. Most of the attention to changes in atmospheric chemistry has been on how greenhouse gases affect climate. There is a more precise and potentially greater impact of carbon dioxide concentration on nutrient cycling and effects on soil (Barrow, 1991).

Brown and Archer (1999) stated that the role of livestock as primary determinants in bush encroachment is also complemented by factors such as climatic changes in the history of the atmosphere, e.g. carbon dioxide concentration. Changes of temperatures at certain times of the year limit woody species productivity and species distribution (Teague & Smit, 1992). O'Connor *et al.* (2014) stated that a monotonic increase in annual rainfall might promote woody vegetation but it would also probably increase the frequency and intensity of fires that shape southern African savannas and grasslands. Climate change may result in a greater frequency of severe droughts and very wet years, without any change in mean annual rainfall.

2.5 IMPACT OF BUSH ENCROACHMENT ON VEGETATION

Changes in natural vegetation where grassland dominated areas are transformed into one dominated by woody species and an increase in unpalatable forbs are regarded as a threat to the condition of rangelands (Oba *et al.*, 2000). Ayoub (1998) verified the growth of less palatable grasses, shrubs and weeds in African savannas, after it has been grazed by livestock. This is because of the suppression of palatable grasses due to woody species encroachment that are unpalatable to domestic livestock (except goats). In some cases, competition from woody plants decreases productivity of the herbaceous layer, thus rendering an environment less suitable for grazers such as cattle and possibly more suitable for browsers such as goats.

Woody plant encroachment and herbaceous biomass production are negatively correlated (Pyke & Knick, 2005) wherein an increase in woody plant abundance is mostly accompanied by reduction in herbaceous production and undesirable shifts in herbaceous composition (Archer, 1990). Woody vegetation reduces grass cover through increasing the competition for available water and nutrients and reducing the sunlight reaching the grass layer (Thurow, 2000). When the grass layer is over utilized, it loses its competitive control and can no longer utilize water and nutrients effectively (Felegeselam, 2006). This results in a higher water and nutrient infiltration rate into the subsoil. Such a situation benefits trees and bushes and allow them to dominate. Prolonged denudation of soils produced by droughts and grazing, followed by above-average rainfall with frequent rainfall events, favour mass tree recruitment (Ayoub, 1998).

2.6 IMPACT OF BUSH ENCROACHMENT ON SOIL

Trees and shrubs have been found to improve the nutrient status of their close surroundings in semi-arid shrub communities, arid grasslands, in tropical and sub-tropical savannas, east African savannas and South African savannas (Belsky, 1990). In a study conducted by Lal (2004) which measured soil carbon, nitrogen and phosphorus, it revealed consistent horizontal patterns in the top soil. The content of these nutrients declined gradually as a function of distance from the tree trunks and was significantly lower in open ground than sub-canopy soil (Hagos & Smit, 2005). This can be ascribed to the interaction between trees, understory plants and symbiotic micro-organisms. According to Hagos & Smit (2005) there are indications that soil enrichment can differ between tree species that grow in the same environment. It was demonstrated by Smit & Swart (1994) that soil under both leguminous trees (mostly *Vachellia erubescens*) and non-leguminous trees (mostly *Combretum apiculatum*) was richer in % total N, % organic C, Ca and Mg, but that nutrients like K and Mg were noted in higher concentrations under leguminous trees compared to non-leguminous trees.

Nutrients found in low concentrations through the soil profile may be taken up by the root system of matured trees and shrubs. Through leaf abscission, nutrients will be concentrated in the sub-canopy area due to litter and decomposition. This has been suggested as a principal explanation by Belsky (1990) and Vetaas (1992). If nutrients are absorbed by tap roots of woody species, a total nutrient supply to the field-layer is altered. The combination of relocation and surface root turnover and shedding of leaves will act together as a nutrient pump (McNaughton, 1985). In this concept, woody species modify the microclimate by interception of solar radiation and rainfall. The trees are intercepting and taking up nutrients the moment they are released by decomposition. Tree litter will lead to accumulation of organic matter under and near the trees (Frost *et al.*, 1986).

The effect of nitrogen on stimulating bush encroachment is evident, because the encroaching species are nitrogen fixers as compared to grasses (Ward, 2005). According to Ward (2005), it is vital to understand interactions between different levels of soil nitrogen and the population genetic structures of the trees in order to make predictions about areas that are susceptible to bush encroachment. Nutrients such as nitrates, phosphates, series of anions and cations and various trace elements, are essential to the nutrition of plants and function as determinants of the composition, structure and productivity of vegetation (Hagos & Smit, 2005).

In savannas, large trees are able to suppress the establishment of new seedlings, while maintaining the other benefits of woody plants, such as soil enrichment and the provision of food to browsing herbivores (Hagos & Smit, 2005). Smit & Swart (1994) stated that the leguminous tree sub-habitat had a marginally higher grass biomass than the non-leguminous tree and the un-canopied sub-habitats at higher tree densities. The improvement of soil water potential (soil water potential is the work water can do as it moves from its present state to the reference state to improve the absorption of water by the herbaceous layer) leads to the development of savanna vegetation (Smit, 2004).

In a study by Dougill & Thomas (2001) in the Mafikeng Bushveld in the Molopo Area, it was observed that significant development of cyanobacteria on sandy soil crusts under the canopies of *Senegalia mellifera* may enable the supply of additional nutrients to plants. Despite similar canopy dimension, soil crust development was found to be greatly reduced under *Grewia flava*, possibly relating to less light reaching the soil surface than with *Senegalia mellifera*. Bosch & Van Wyk (1970) and Kennard & Walker (1973) reported a higher soil pH beneath tree canopies, but lower soil pH values were recorded under *Senegalia mellifera* canopies by Smit (2004). Similar results were noted by Nzehengwa (2013), who also noted a lower pH beneath *Senegalia* and *Vachellia* trees than away from the trunk. The exact reasons for these observations concerning the influence of tree canopies on soil pH are not known.

2.7 GRASS AND TREE INTERACTION

Savanna ecosystems are distinguished by the co-dominance of two contrasting plant life forms, trees and grasses. Competitive-based and demographic-bottleneck models were identified as the two main models for explaining tree-grass co-existence in savannas (Sankaran *et al.*, 2004). Savanna trees affect herbaceous phenology, production and biomass distribution (root/shoot and leaf/stem). Higgins *et al.* (2000) hypothesized that grass-tree coexistence is driven by the limited possibilities for tree seedlings to escape both drought and the flame zone into the adult stage. By this hypothesis, bush encroachment exists due to the expansion of tree recruitment caused by reductions in grass standing crop and, thus, reducing fire intensity.

Competitive-based models emphasise competitive interactions in determining tree-grass co-existence. Trees have historically been viewed as competitors with grasses and are widely regarded as having a negative impact on herbaceous production, particularly where livestock production is a primary land use (O'Connor & Crow, 1999). The competition based model of Walter (1939), is consistent with evidence of the effect of severe grazing on grass competition in semi-arid savannas. Walter's root niche separation model (1939) is the most well-known model based on soil water as the limiting factor with shallow-rooted grasses and deep-rooted trees, having differences in access to soil water. According to Donaldson (1969), root distribution and water use of *Senegalia mellifera* was in accord with Walter's hypothesis. Walter's (1939, 1970) root niche separation (two-layer soil water) model, based on soil water as the limiting factor, with shallow-rooted grasses and deep-rooted trees having differential access to this resource illustrates the grass and tree interaction better. Although grasses are superior competitors for water in the upper soil layers, trees are able to persist because they have exclusive access to water in the deeper soil layers. Tree biomass would increase if the proportion of water in the deeper soil layers increases.

The productivity of soil beneath tree canopies may be enhanced by improved water and nutrient status but be suppressed by competition between trees and grasses for below ground resources. Thus, at the scale, size and density of an individual tree, the net effect on grass production can be negative, neutral or positive and can change with tree age or size and density. As with species composition, the tree-grass relationship is influenced by a variety factors including grazing, browsing and rainfall (Hoffman, 1996). Sustained heavy grazing reduces grass cover, thereby favouring the woody component by allowing more water to infiltrate to deeper soil layers (Walker *et al.*, 1981).

As woody plant cover or density increases, grass production typically declines dramatically (Ward, 2005). Encroachment of trees may further intensify grazing pressure, as landholders destock in response to a decline in grass production corresponding with increases in tree density (Jurena & Archer, 2003). The negative effect of trees on grasses may be an outcome of rainfall interception, litter accumulation, shading, root competition or a composition of these aspects and the effect is determined by the leaf area, canopy architecture and rooting patterns of the tree (Archer, 1990). Demographic-bottleneck models emphasize the impact of climatic variability and disturbance on germination, growth and mortality of trees (Higgins *et al.*, 2000).

The effects of herbaceous species on woody plants are unfavourable during the woody seedling establishment stage, although the effect can be variable. Firstly, herbaceous species can influence woody seedling establishment and recruitment directly by effectively competing for light, water, and nutrients (Knoop & Walker, 1985). The competition can prevent woody seedling emergence, increase the mortality of newly established woody seedlings, and decrease woody seedling growth and recruitment. Even the growth of mature woody plants can be minimized by herbaceous species competition for water in wetter years, when herbaceous biomass is high (Knoop & Walker, 1985).

Herbaceous species can have an indirect effect on woody seedling recruitment (Scholes & Archer, 1997). Herbaceous biomass can increase fine fuel loads, which increases fire frequency and intensity, leading to an expanded mortality of small woody seedlings that are usually vulnerable to fire (Dando & Hansen, 1990; Archer, 1994). Bush encroachment, therefore, depends on growth rate in relation to the fire-return period and would be promoted by those factors that reduce fire frequency or intensity. Increasing density of woody species, such as *Senegalia mellifera*, strongly suppress herbaceous production in the Molopo area (Rutherford, 1978) and suppress grass productivity. *Senegalia mellifera* may, therefore, produce an example of demographic-bottleneck involving seedlings regeneration in a semi-arid savanna (Joubert *et al.*, 2008).

2.8 BUSH ENCROACHMENT AS A FORM OF SUCCESSION

Succession is the process of change in the species structure of an ecological community over time. The time scale can be decades (Bazzaz, 1996). The community begins with relatively few pioneering plants and develops through increasing complexity until it becomes stable as a climax community. Bazzaz (1996) illustrated that the cause of ecosystem change is the impact of established species upon their own environments. It is a process by which an ecological community undergoes more or less orderly and predictable changes following a disturbance or initial colonization of a new habitat.

Plant succession results in the transformation of a grass-dominated ecosystem to a tree dominated system. In a study conducted by O'Connor (2012) on bush clump succession in grassland, it was indicated that clumps formed following establishment of mainly *Vachellia karroo* in grassland through nucleation and propagation by animal. With increasing clump size, there was a corresponding increase in woody basal area, species richness and diversity, but not evenness. O'Connor (2012) stated that the correspondence analysis showed that the pattern of compositional variation was closely related to clump size, differences resulting from clump establishment as clumps developed. Population structure of 20 species was dominated by small individuals, indicating that regeneration of woody species has increased over time (O'Connor, 2012).

2.9 STUDIES OF BUSH ENCROACHMENT IN THE MOLOPO AREA

Several studies were conducted in the Molopo Area on bush encroachment (Molatlhegi, 2008), Mogodi, 2009 and Comole, 2014). The locations where the research were done include areas such as Disaneng, Makgori, Masamane, Loporung, Selosesha and Tshidilamolomo (Refer to Figure 3.1). Woody species identified included *Senegalia mellifera*, *Grewia flava*, *Vachellia erioloba*, *Dichrostachys cinerea*, *Senegalia hebeclada*, *Diospyros lycioides*, *Grewia occidentals*, *Ehretia rigida*, *Searsia dentata* and *Terminalia sericea*. The researchers recorded high densities of woody species in the Molopo Area and the results proved that the area was highly encroached. The results are discussed and compared to the results of this study in Chapter 5.

2.10 *Senegalia mellifera* - A "PROBLEM" PLANT IN THE MOLOPO AREA

The domination of the savanna communities by problematic woody species, mainly *Senegalia mellifera*, which have been favoured by the present conditions in the arid and semi-arid environments, is of major concern wherein the growth and subsequent size achieved by any individual tree is dependent on the resources to which it has access. Fussel (1995) define the term "problem tree" as a conceptual model used as a diagnostic tool to analyse a sequence of events involving the said tree that eventually leads to a problem.

In an estimate of the degree of *Senegalia mellifera* infestation, Ebersohn *et al.* (1960) indicated that over 8 5000 000 ha of veld in the Molopo Area has been encroached by *Senegalia mellifera.* Encroachment by *Senegalia mellifera* has been considered an ecological problem in rangelands of many southern African countries (Donaldson, 1967; Skarpe, 1991). This was also recorded along stock-transport roads in the southern Kalahari as early as the 1860's (O'Connor *et al.*, 2014). The significance of bush encroachment in the Molopo Area has been emphasized by Ebersohn *et al.* (1960) and *Senegalia mellifera* (*Senegalia mellifera*) was identified as the main encroacher (Donaldson, 1969).

According to Richter *et al.* (2001), increasing woody plant density (>2 500 TE. ha^{-1}) decreases the potential grazing capacity by as much as 331% in the Molopo Thornveld, in contrast to sites with tree densities of less than 400 TE. ha^{-1}. According to Hagos and Smit (2005), *Senegalia mellifera* is an "aggressive" invader and is well known for its suppression of the herbaceous layer and is viewed by local inhabitants as a serious threat. *Senegalia mellifera* is well adapted to grow on a variety of soil types, ranging from Kalahari sands to heavy clayey soils. However, it often tends to be associated with calcium-rich soils (Hagos & Smit, 2005). Although soil analysis did not form part of this study, the local soils found were sandy. Both Molatlhegi (2008) and Mogodi (2009) recorded sandy soils (>70% sand) in the Molopo Area. The local soils, found in this study was sandy and this will also favour species such as *Senegalia mellifera*, because of its wide spread root system as well as its deep taproot (Hagos & Smit, 2005). These characteristics will enable *Senegalia mellifera* to be well adapted to and compete with other woody encroachers and the herbaceous component. O' Connor *et al.* (2014) stated that *Senegalia mellifera* is among the 6 most important encroaching legumes. O' Connor *et al.* (2014) indicated changes in *Senegalia mellifera* woody cover, using ground, aerial and satellite based data were done in areas along the Orange River near Okahandjau Namibia, Pniel in the Northern Cape and in Eastern Botswana. The other legumes well known for their invasive characteristics are *Senegalia hebeclada, V. karroo, V. nilotica, V. tortilis* and *Dichrostachys cinerea* (O' Connor *et al.*, 2014).

CHAPTER 3

STUDY AREA AND CLIMATIC CONDITIONS

3.1 STUDY AREA

The study was conducted at selected sites in Disaneng which is situated at the North West Province (NWP), Molopo District, and 46 km west of Mahikeng, South Africa (Figure 3.1). This is a semi-arid Savanna Biome. According to Barnes (2001) a semi-arid savanna is characterized by a range of physiognomic vegetation types of the tropical summer rainfall regions of Africa. The Molopo district is located within the Kalahari Desert ecological zone (Department of Agriculture, Conservation, Environment and Tourism, 2002).

The North West Province (NWP) is situated towards the western part of South Africa. It occupies a total area of 116 320 km^2 (9.5% of the total area of South Africa) and is the sixth largest province in South Africa (State of the Environment Report, 2002:). It is geographically situated between 25 and 28 degrees south of the equator and 22 and 28 degrees longitude east of the Greenwich meridian (Cowan, 1995). The NWP borders the provinces of the Northern Cape in the west, the Free State province in the south while Gauteng in the east and Limpopo (formerly Northern Province) in the north-east. NWP shares an international border with the Republic of Botswana in the north (Figure 3.1).

The study site was located within the coordinates (Figure 3.2):

Point 1 (S25°48.084'; E025°14.269');

Point 2 (S25°48.099'; E025°14.297');

Point 3 (S25°48.106'; E025°14.340');

Point 4 (S25°47.075'; E02514.533');

Point5 (S25°47.067; E025°14.520')

Point 6 (S25°47.986'; E025°13,678').

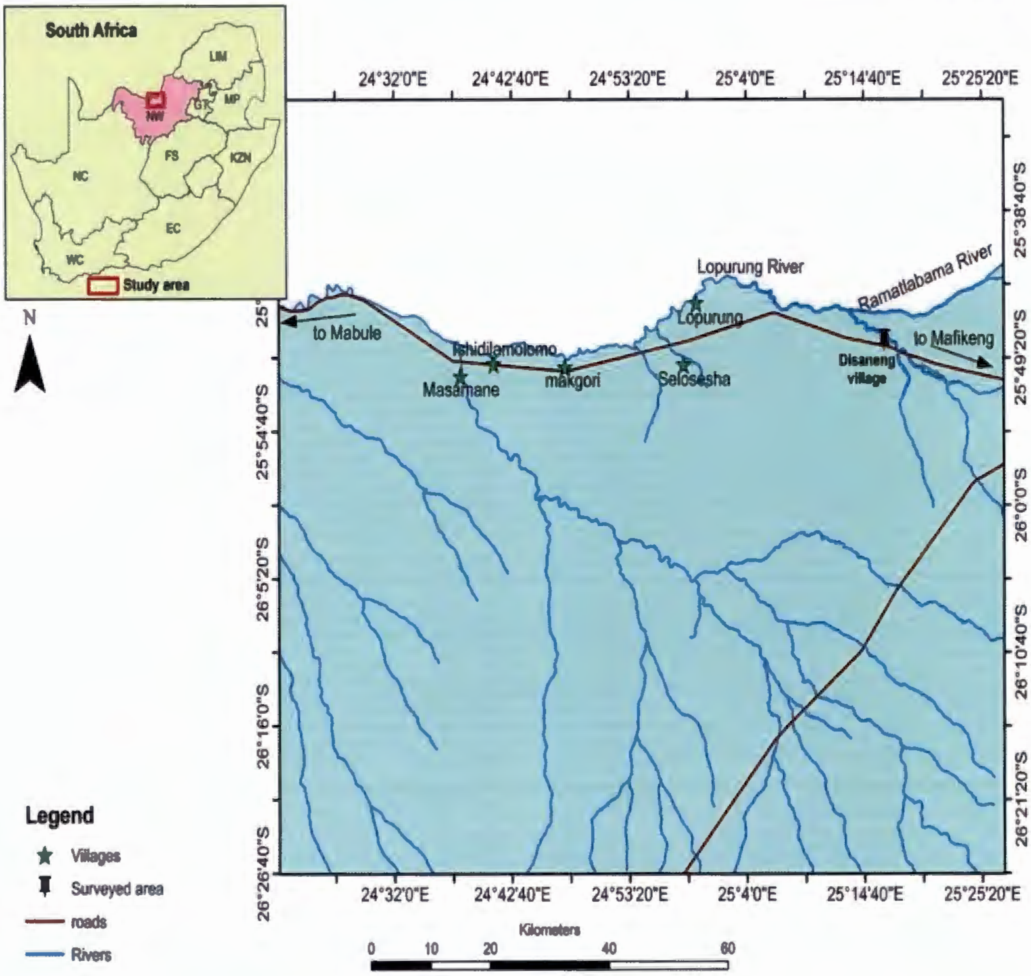


Figure 3.1: Location of the study area

The benchmark is located within co-ordinates S25°48.176' and E25°14.468', one kilometer from the study site (Figure 3.2), with similar environmental conditions such as climate, soil type and veld type to the study site.

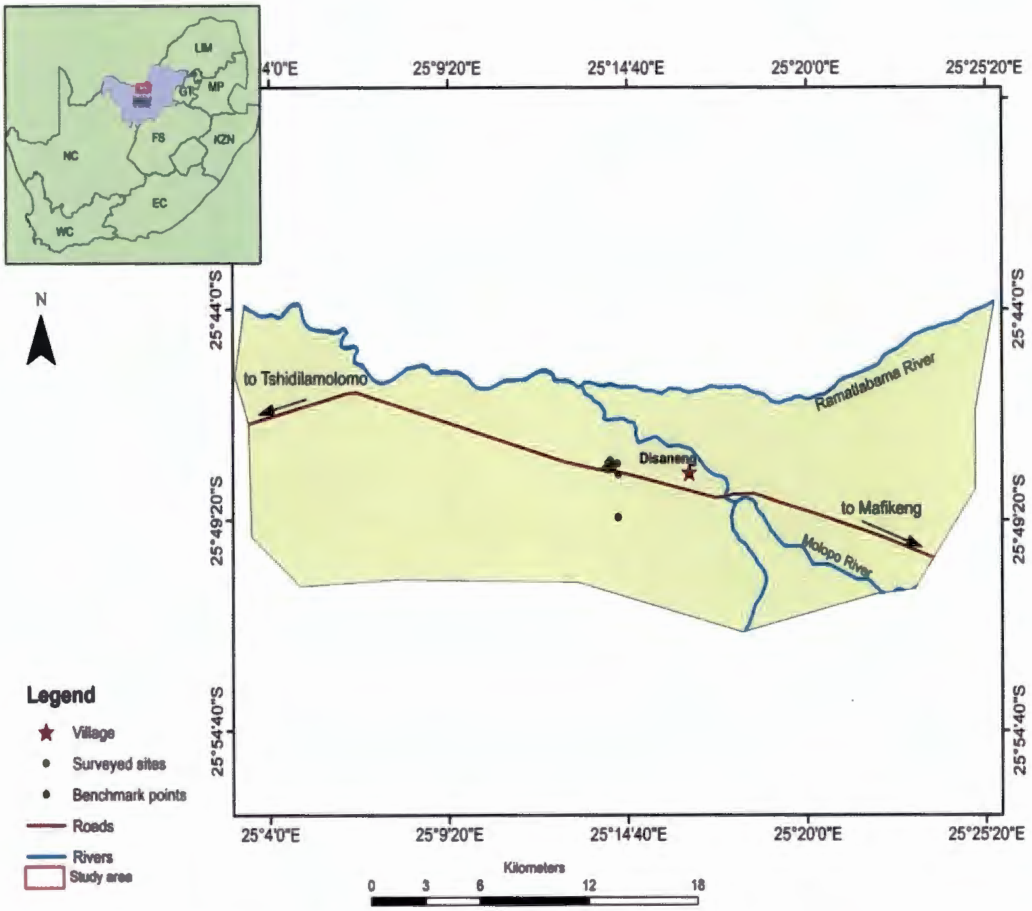


Figure 3.2: Study site and benchmark points

The mean monthly maximum and minimum temperatures for the NWP are 35.6°C and 1.8°C for November and June respectively (Mucina & Rutherford, 2006). Annual rainfall is approximately 360 mm, with the highest rainfall during the summer months, between October and April. Due to the low precipitation levels, the NWP is considered to be an arid region. A large amount of precipitation occurs as thunderstorms, which are associated with events such as heavy gusts of wind, lightning, hail and flash-floods. Water drains through three primary river systems: The Vaal in the south, Molopo in the West and Crocodile/Marico in the east (Department of Agriculture, Conservation, Environment and Tourism, 2002).

3.2 CLIMATIC CONDITIONS

The major determinant of the geographic distribution of plants and animals is climate (Comole, 2014). Tainton (1999) stated that within any area of general climatic conditions, local state of temperature, light, humidity and moisture differ and these factors play an important role in the production and survival of plants. Temperature and rainfall are the most important climatic variables.

Precipitation in the Savanna Biome is seasonal (alteration of wet summer and dry winter periods) (Mucina & Rutherford, 2006). 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, mainly along the southern most edges of the Central Bushveld (Mucina & Rutherford, 2006). Temperatures are higher than those in the adjacent Grassland Biome at higher altitudes (Mucina & Rutherford, 2006). The mean daily maximum temperature for February rarely drops below 26°C in the Kalahari region and some low-altitude parts of the Savanna in the east. In July this temperature remains above 20°C for the most of the area, with some temperatures at the highest altitudes dropping to 18°C (Mucina & Rutherford, 2006). The mean daily minimum temperature in February is rarely 16°C, with the temperature of substantial parts of lower low-veld remaining above 20°C (Mucina & Rutherford, 2006).

Mahikeng falls within the Mahikeng bushveld (Mucina & Rutherford, 2006) and has a summer rainfall with very dry winters. The mean annual precipitation (MAP) varies from

350 mm in the west to 520 mm in the east (Figure 3.3). Frost occurs frequently in winter. According to Mucina & Rutherford, (2006) the mean monthly maximum temperature is 35.6°C and minimum temperature is -1.8°C in Mahikeng for November and June respectively.

3.2.1 Rainfall

The NWP is in a summer rainfall area with more sunshine days and warm temperatures than winter season. On average, the western region receives less than 350 mm rainfall per annum, while the eastern and south-eastern region receives over 550 mm rainfall per annum (Mucina & Rutherford, 2006). The dominant rainfall season in the central region (Where the study site is located) is mid-summer (peaking in January). The western parts of the NWP receive precipitation in the late summer (peaking in February), while the eastern parts reach the maximum level in early summer (December) (Dirkx *et al.*, 2008).

Annual precipitation over the country as a whole is relatively low and evaporation losses are high. Moisture stress is a factor which exerts a major influence on plant survival (Tainton, 1999). Rainfall is, therefore, the factor which most clearly determines the distribution of plant communities in the study area, as well as the potential productivity of these communities. Bush encroachment in the arid areas is influenced by variation of rainfall because plants respond to rainfall events (Teague & Smit, 1992). Changes in vegetation type and amount have been attributed to deeper rainfall penetration into the soil, favoring deeper-rooted trees (Ward, 2005).

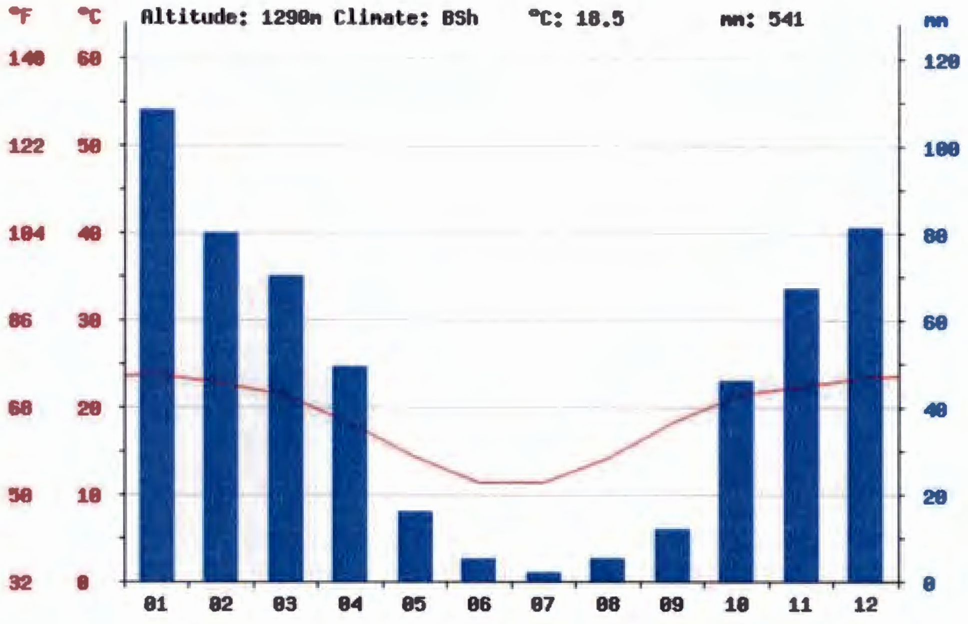


Figure 3.3: Mahikeng climate (South African Weather Service). The X-axis presents the months of the year and y-axis presents the temperature in ℃ on the left and rainfall measurements in mm on the right

Figure 3.3 shows the highest mean rainfall of 110.7 mm during the month of January, followed by 90 mm in December, for the period of 28 years. Mahikeng receives a relatively low rainfall during the winter season (June to August, Figure 3.3). This limited rainfall and the long-term overgrazing may contribute to the proliferation of woody plant species.

Climatic conditions in the NWP vary significantly from west to east. The far western region is arid (receiving less than 300 mm of rainfall per annum), encompassing the eastern reaches of the Kalahari Desert (Kent, 1980). The central region of NWP is dominated by typically semi-arid conditions, with the eastern region being predominantly temperate (Kent, 1980).

3.2.2 Temperature

The NWP is characterised by great seasonal and daily variations in temperature, being very hot in summer (daily average maximum temperature of 32°C in January) and mild to cold in winter (average daily minimum in July of 0.9°C) (Mucina & Rutherford, 2006) with an average annual temperature of 18.5°C (Figure 3.3). Mean seasonal temperatures fluctuations between the warmest and the coldest months exceed 15°C in the western region, while the central and eastern regions encounter a range between 12°C and 15°C (Dirkx *et al.*, 2008). The mean monthly maximum and minimum temperature for Mahikeng is 35.5°C and -1.8°C for November and June respectively (Mucina & Rutherford, 2006). Frost is frequent in winter. According to Figure 3.3, the months of October to December and January to March are the hottest, while June and July are the coldest. The Molopo District has a semi-arid climate, characterized by high day temperatures during the summer months and cool daily temperatures during winter months.

Relative humidity is also typically low throughout the NWP, being below 28% in the northern part of the province in July and between 28-30% for the central and eastern regions, in February which is the month with the highest relative humidity. The relative humidity in eastern and northern parts range between 66-68% and the rest of the province ranges between 64-66% (Palmer *et al.*, 2007). This gives rise to high potential evapotranspiration rates, influencing the resident flora of the region (Dirkx *et al.*, 2008).

The predominant wind direction is from a northerly direction. There is a trend for the windy months to occur between August and November (O'Connor, 1991).

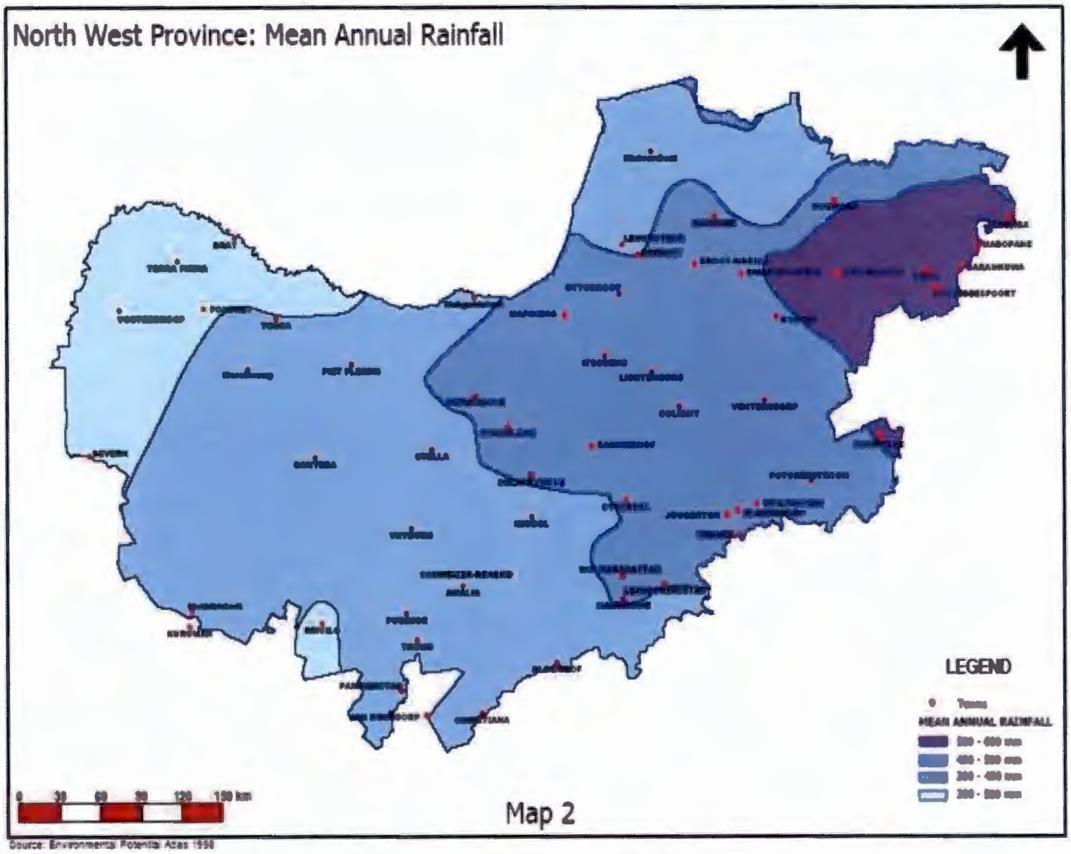


Figure 3.4: Mean Annual Rainfall of the North West Province (Department of Agriculture, Conservation, Environment and Tourism, 2002)

3.3 GEOLOGY AND SOIL TYPES

The condition of the soil is another factor which may prevent the optimum development of the vegetation. A terminal community determined by soil condition is known as an edaphic climax. An edaphic climax could result from a permanent limitation of soil depth, as in the iron-pan soils of many parts of South Africa, or from an inherent lack of soil nutrients. This is another situation where the climax community may not provide any clear indication of the area (Tainton, 1999).

3.3.1 Geology

The geology of an area influences the topography and thus influencing the climate, present materials, soil and vegetation (Kent, 1980). The condition of the soil is another important factor which may prevent the optimum development of the vegetation. In South Africa, the Savanna Biome is located mostly in the north-eastern part of the country (Mucina & Rutherford, 2006). The geology of this area is dominated by a very stable block of ancient continental crust, known as the Kaapvaal Craton (Mucina & Rutherford, 2006). The Kaapvaal craton began to form by a process of accretion over 3.5 billion years ago (bya) and has been largely unaffected by crustal processes for the past 2 giga years, except on its fringes (Mucina & Rutherford, 2006). The craton also hosts a number of significant sedimentary basins and igneous intrusions, thus preserving a geological record spanning most of geological time (Mucina & Rutherford, 2006).

The NWP has an interesting and ancient geological heritage, rich in minerals and paleontological artefacts. Igneous rock formations are prominent in the north-eastern and north-central regions of the province as a result of the intrusion of the Bushveld Complex. The Ventersdorp geological formations originated more than 2 000 million years ago and appear to be the prominent formations in the western, eastern and southern regions of the province. Sedimentary rocks dating back to the Quaternary period (65 million years) occur in the north-western corner of the Province (Kent, 1980) (Figure 3.5).

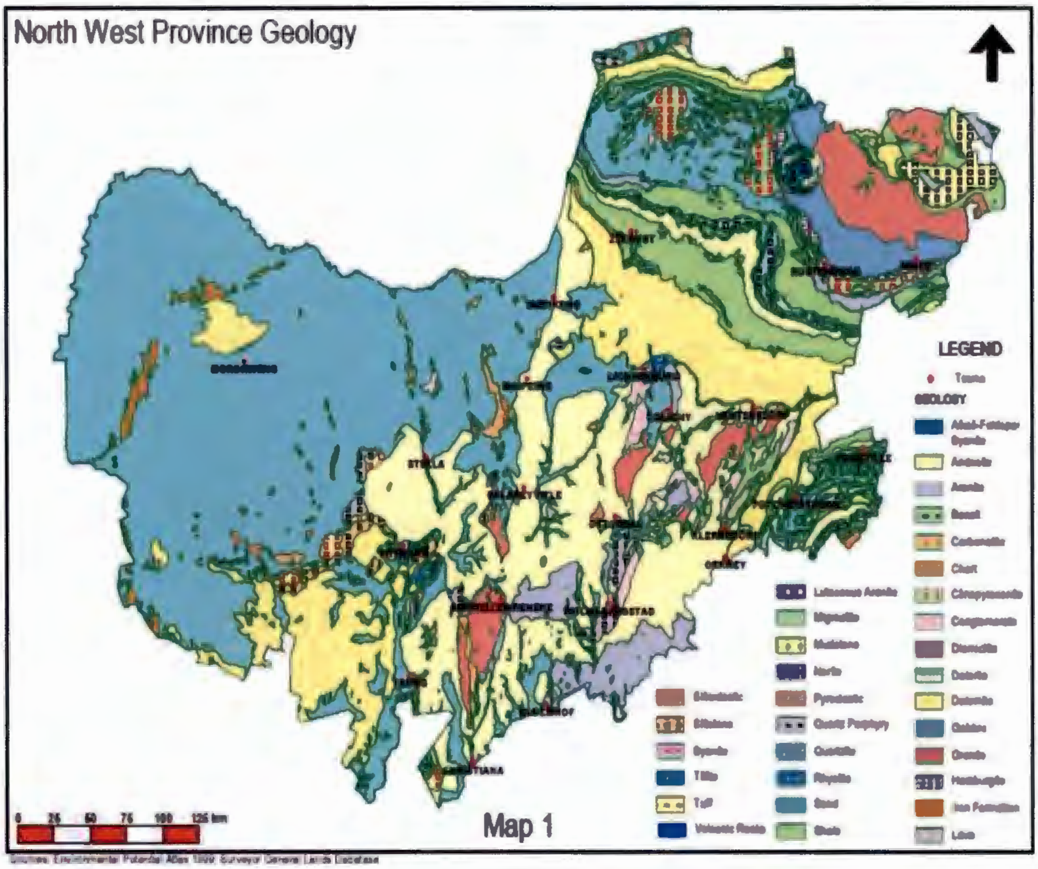


Figure 3.5: Geology of the North West Province (Department of Agriculture, Conservation, Environment and Tourism, 2002)

The study site is situated next to Mahikeng town wherein its morphology is described as plains (Figure 3.6). The landscape is predominantly flat with scattered trees and grassland.

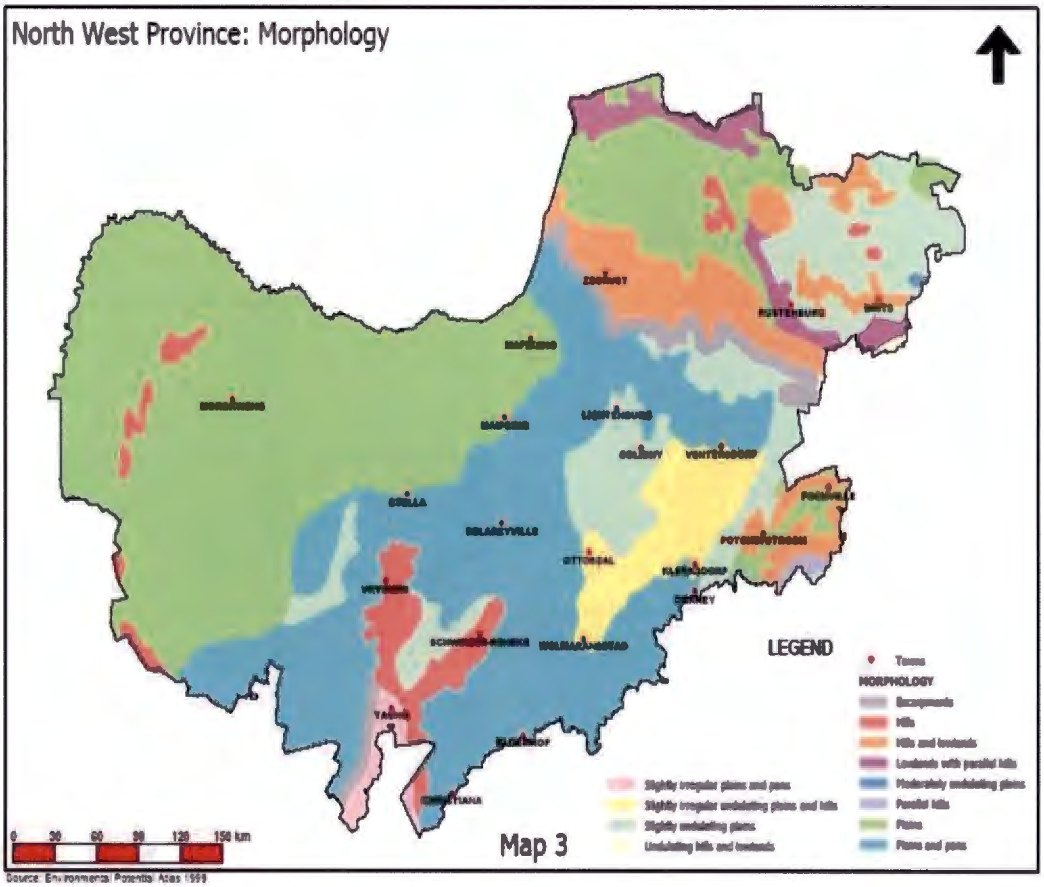


Figure 3.6: NWP map presenting the morphology of different towns (Department of Agriculture, Conservation, Environment and Tourism, 2002)

3.3.2 Soils

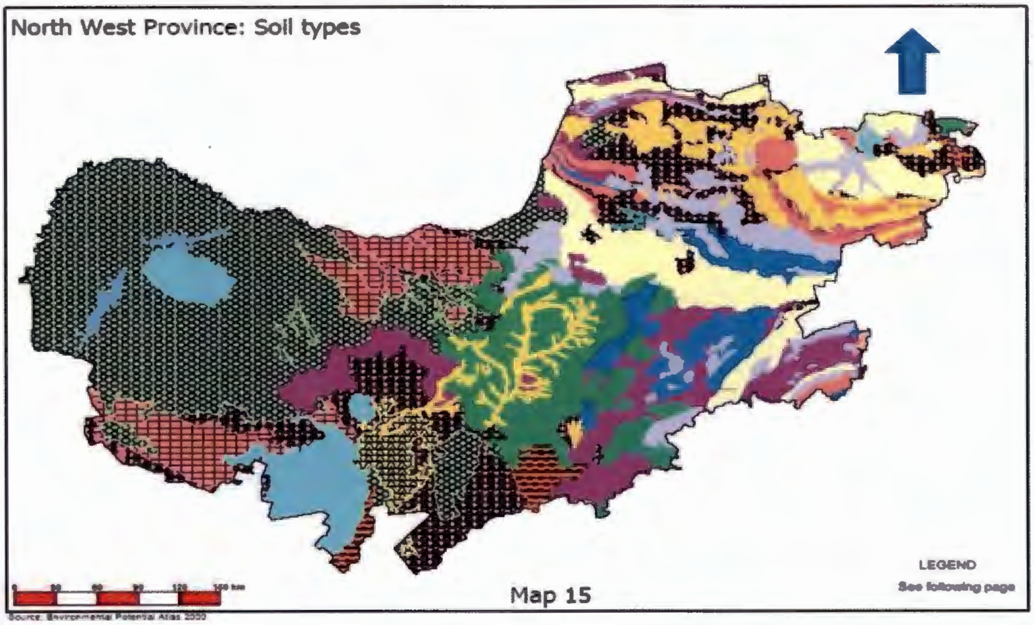
Different soil types as well as soil depths determine the productivity and the palatability of grazing on at long term bases. Soil colour, texture and structure are the most important characteristics of the soil (Moleele, 1998). Due to the dynamic nature of soils, soil types are constantly evolving and degrading by means of natural and man-induced processes. The weathering of rocks (one of the main processes responsible for soil formation) in deserts and semi-arid areas tends to be superficial; therefore, these soils tend to be shallow and stony. Erosion and deposition by the agents of wind and water are responsible for the transportation of soils from one environment to another.

Soils in the NWP are deemed to be only slightly leached over much of the western region due to the low rainfall generally experienced. With high evaporation rates, there is a tendency of upward movement of moisture in the soils. This often leads to high concentrations of salts such as calcium and silica in soils, which occasionally lead to the formation of hard pans or surface duri-crusts. As a result, high levels of salinity or alkalinity may establish in these areas. Levels of organic matter tend to be low, governing the vegetation types present.

Large areas of yellow shifting sands occur in the north-western region of the NWP, while a plinthic catena of yellowish-brown sandy loam is characteristic of the south and eastern region. The central region areas are covered by red or brown sands with rock. This region also has weakly developed lime soils similar to the dolomite limestone establishments. The south-western region also has areas characterized by undifferentiated rock and lithosols. According to Nel *et al.* (1995) lithosols are shallow soils containing coarse fragments and solid rock at depths less of than 30cm.

The north-eastern part of the NWP has been shown to have lithosols of arenaceous sediments (State of the environment report, North West Province, 2002). The southern and central regions have black and red clays with ferrisiallitic soils of sand, loam and clay. The predominant soil characteristics of the study site are red-yellow apedal, free drained soils (usually$<15\%$ clay) (Figure 3.7) and the soil is mostly calcareous and non-acidic. The soil forms are Mispah and Glenrosa. These soils are subject to wind erosion owing to low clay percentage. The pH of 6 to 8 may contribute to early bud break of *Prosopis* plant. The soil of the area has relatively low biological activity (State of the environment report, North

West Province, 2002). The low level of biological activity is the result of very low organic matter content, associated with the soil forms of the area (MacVicar *et al.*, 1991).



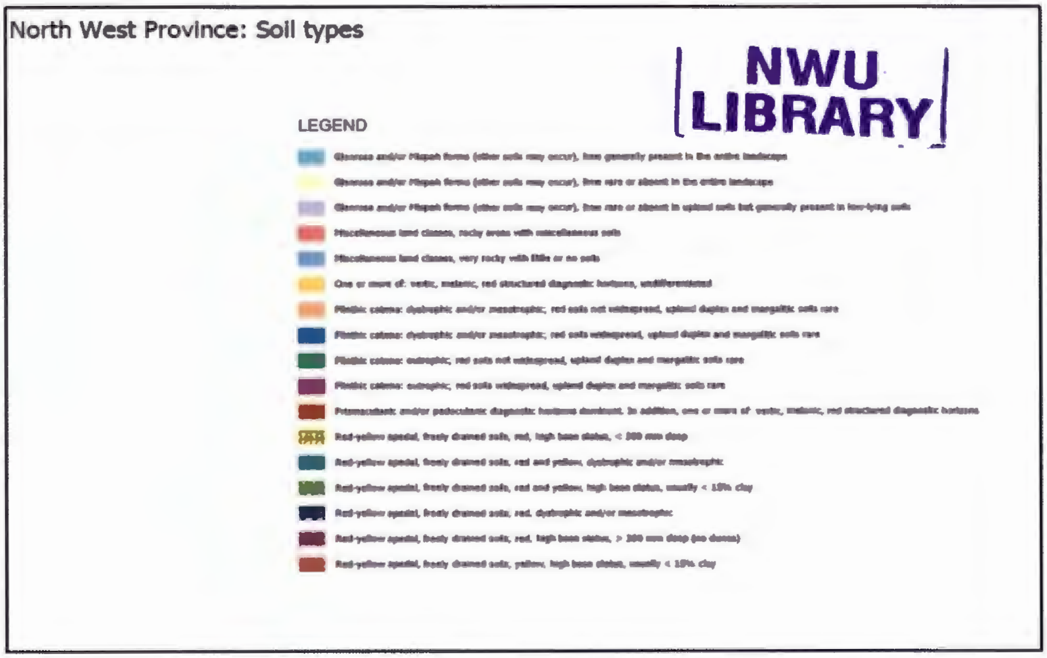


Figure 3.7: NWP map presenting the soil types of different regions of the province (Department of Agriculture, Conservation, Environment and Tourism, 2002)

3.4 VEGETATION OF THE STUDY SITE

The NWP falls within the Savanna and Grassland Biomes. Most of the NWP (71%) falls within the Savanna Biome with its associated bushveld vegetation (Low & Rebelo, 1998). The remainder falls within the grassland biome comprising a broad diversity of grasses particular of arid areas. Considering the arid and semi-arid conditions of the western half of the NWP, the vegetation of this region largely comprises plants that are adapted to these conditions (known as xerophytes). The outcome of this situation is that, low biomass, productivity and a wide range of plant species are found in this region. With the east-west difference in climate and rainfall, there is an equivalent gradation in the vegetation types from xerophytic in the western region, to open grassland and savanna in the central region and Bushveld in the eastern region.

According to Low & Rebelo (1998), nine different vegetation types are present in the NWP, belonging to the following categories: clay, Kalahari, Kimberley, mixed bushveld and highveld grassland. There is a predominance of Kalahari deciduous *Acacia* (*Senegalia* and *Vachellia*) thornveld (open savanna of *Vachellia erioloba* and *Vachellia haematoxylon* as well as desert grasses) and shrub bushveld in the dry western part of the Province. The rocky soil is valuable to *Tarchonanthus* veld on the dolomite Ghaap Plateau (Acocks, 1975).

The northern and eastern regions reflect the greatest variability of vegetation types in the Province. Vegetation types contain sourish mixed bushveld which is an open savanna dominated by *Senegalia caffra* and grasses of the *Cymbopogon* and *Themeda* types, grass thornveld and isolated patches of Kalahari thornveld and shrub bushveld. The Rustenburg and Marico districts have mixed vegetation types dominated by *Senegalia caffra* and *Vachellia erioloba*, with the latter tree being found growing on deep sand overlying limestone bedrock (State of the Environment report, North West Province, 2002). South Zambezian undifferentiated woodland and grassland covered by woody species occur in the central region. Highveld grassland of the *Cymbopogon-Themeda* veld type and *Vachellia erioloba* savanna predominate in the central and southern regions (Acocks, 1975).

The broad patterns of geology, soil types and climate are the major governing factors in the distribution of the province's vegetation types (Department of Agriculture, Conservation, Environment and Tourism (DoACET), 2002). The description of the vegetation patterns (Figure 2.8) is based on Low & Rebelo (1998). However, approximately 60 % of natural vegetation types in the North West Province have been transformed through anthropogenic activities. Approximately 72% of the NWP falls within the Savanna Biome, with the following major vegetation types (percentage cover shown in parentheses) (Department of Agriculture, Conservation, Environment and Tourism, 2002). The study area was situated within the Sourish Mixed Bushveld (Figure 3.8) as described by Mucina & Rutherford (2006). Thorny shrubs, mostly *Vachellia* and *Senegalia* spp. (Previously *Acacia* spp.) such as *Senegalia mellifera* and *Vachellia tortilis*, *Grewia flava* and *Ziziphus mucronata* are the main encroachers in the study area and may form dense stands with little understory growth. However, these shrubs have a valuable browsing value for, especially goats. *Senegalia mellifera* was the most prominent encroacher species in the study site as a result of heavy grazing.

3.4.1 Sourish Mixed Bushveld

Combretum apiculatum, *Senegalia caffra*, *Dichrostachys cinerea, Lannea discolor*, *Sclerocarya birrea*, and various *Grewia* spp. form the main tree and shrub components within this vegetation type (Mucina & Rutherford, 2006). Local thorn trees include *Senegalia hebeclada, V. karroo, V. tortilis* and *Senegalia mellifera* as well as *Tarchonanthus camphoratus*. The dominant grasses are *Digitaria eriantha*, *Schmidtia pappophoroides*, *Anthephora pubescens*, *Stipagrostis uniplumis* and various *Eragrostis* spp. and *Aristida* spp.

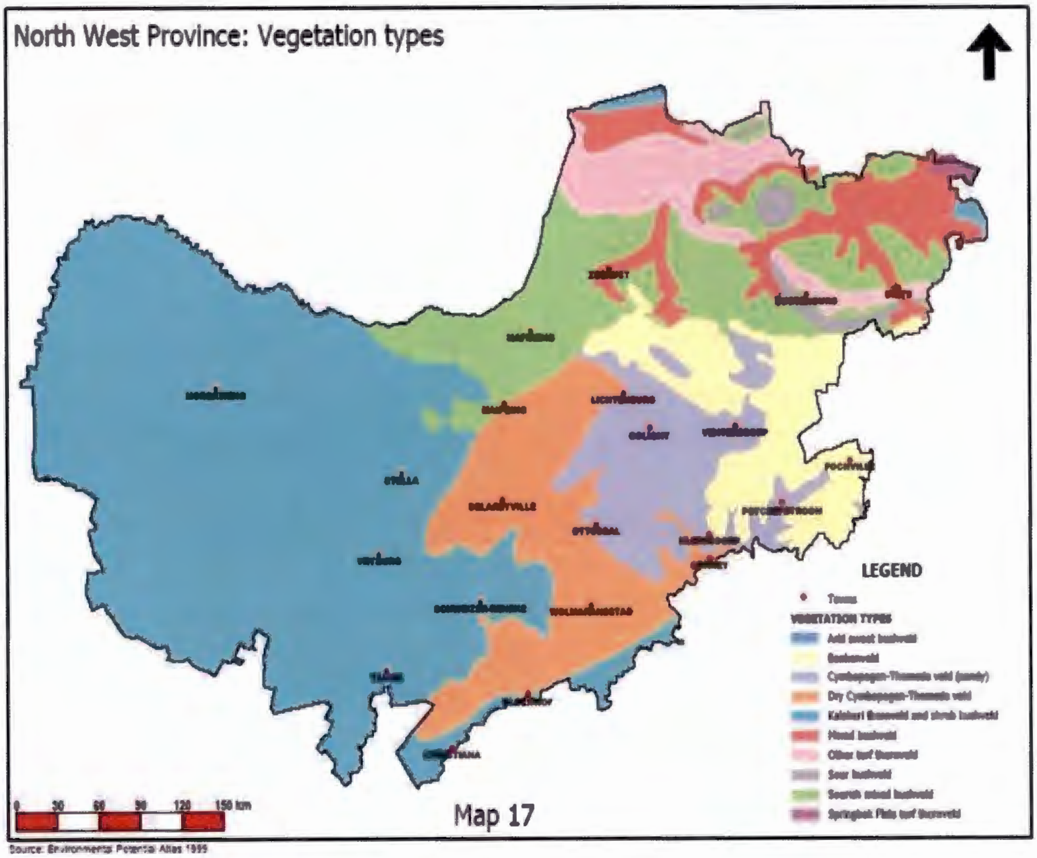


Figure 3.8: Vegetation Types of the North West Province (Department of Agriculture, Conservation, Environment and Tourism, 2002)

3.5 MAIN LAND USE IN THE NORTH WEST PROVINCE

The rapid rate of population growth is consistent with the trends in environmental degradation and rapid depletion of natural resources (Comole, 2014). The North West Province is the sixth largest of South Africa's provinces, and has the third lowest population density (North West Province Vegetation Types, Department of Agriculture, Conservation, Environment and Tourism (DoACET), 2002). Most of the rural communities are found in the Bophirima District followed by the Central Eastern, Rustenburg and Southern Districts. Rural villages in the west and north are small and scattered, and the majority of which are located in the former Bophuthatswana areas (currently the North West Province) (Department of Agriculture, Conservation, Environment and Tourism (DoACET), 2002). Most of these communal areas are also over-crowded, giving rise to squatter settlements. The study area is located within the communal areas (Figure 3.10) where livestock farming (Figure 3.9) plays an important role.

Most of the farmers keep livestock for barter-trading, milk-production and slaughter. Most farmers keep cattle but do not take part in any grazing program. There was no sign in the study area where farmers invested in any type of fencing. This can be ascribed to security concerns.

To fence an area is expensive and newly erected fences are stolen virtually overnight (Department of Agriculture, Conservation, Environment and Tourism (DoACET), 2002). Communal areas are not privately owned by the farmers, but are under the control of the tribal authority (chief). The latter has the final say as to what can and cannot be done in an area.

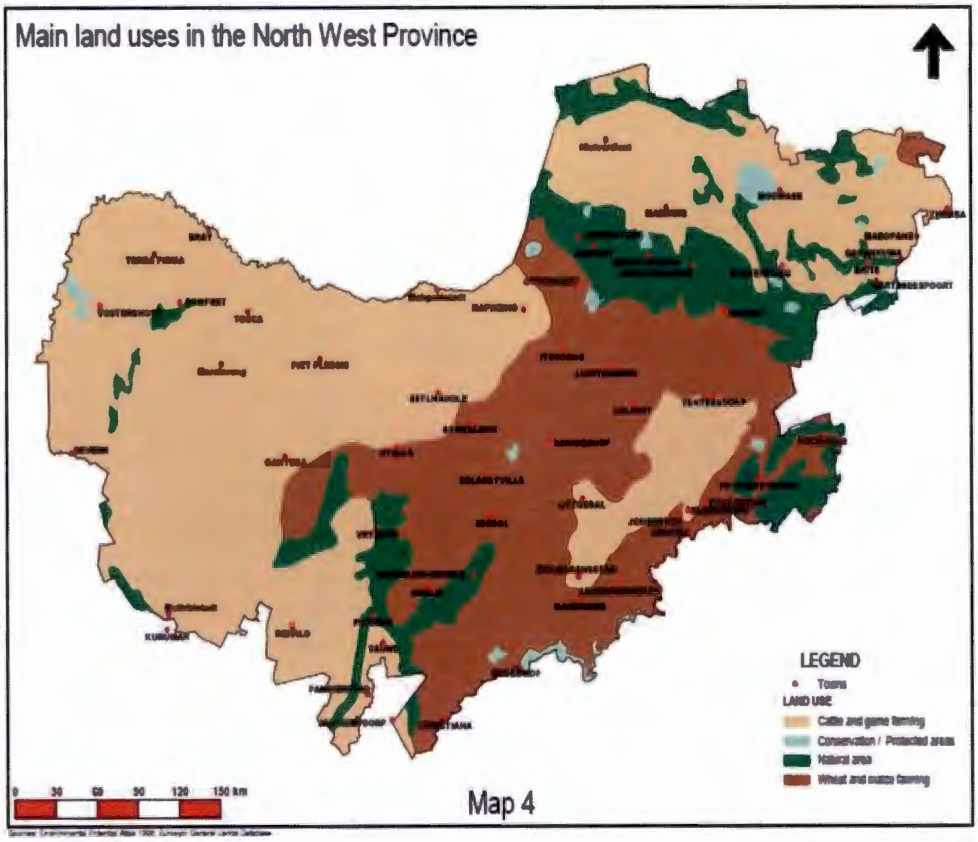


Figure 3.9: Main land uses of the North West Province (DoACET, 2002)

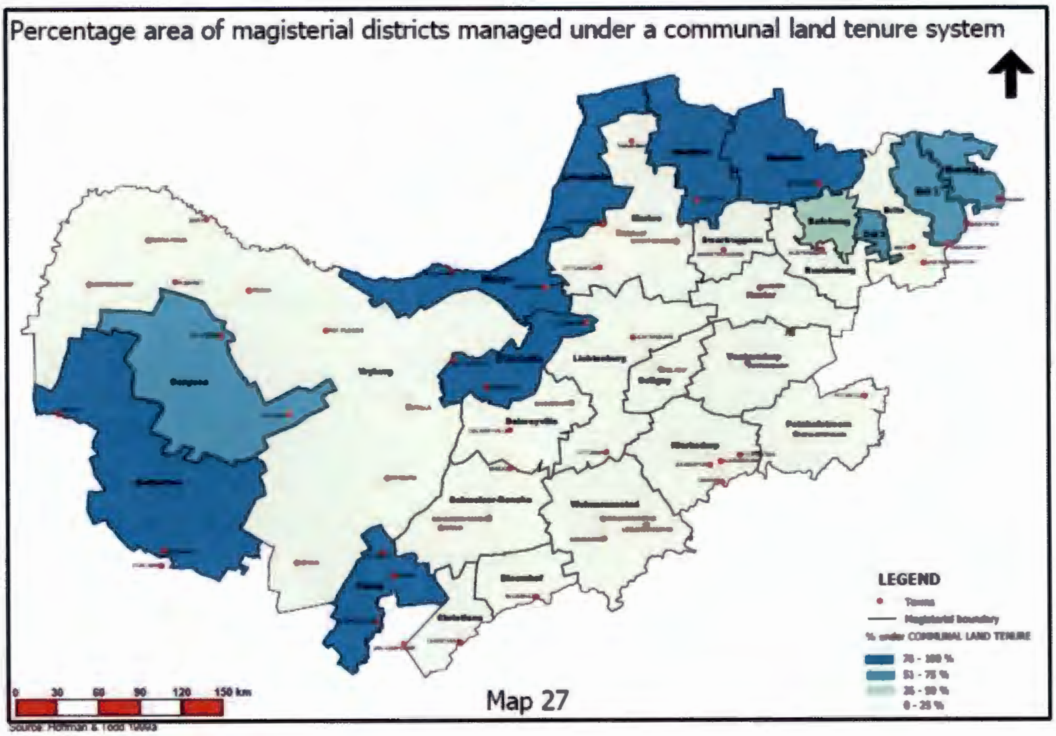


Figure 3.10: Percentage area of magisterial districts managed under a communal land tenure system in the North West Province (Department of Agriculture, Conservation, Environment and Tourism, 2002)

CHAPTER 4

METHODOLOGY

4.1 VARIABLE QUADRAT METHOD

4.1.1 Introduction

Bush density has a critical role on the grass cover with regards to the shading effect and competition for soil moisture, which indicate the condition of the rangeland (Smit, 2004). An increase of the woody component to above 2 500 $TE.ha^{-1}$ has a negative influence on grazing capacity of the savanna (Richter & Meyer, 2001). Information on the abundance of certain species is fundamental to most questions in Ecology. Due to time and resource challenges, ecologists rarely do a complete study of all species in the area of interest. They usually have to estimate the abundance of species by sampling or counting a subset of the population of interest.

In this research, the woody vegetation was studied following the variable quadrat method by Coetzee & Gertenbach (1977) (Figure 4.1). This method provides the following outcomes per species and height class:

1. Canopy regime at different height levels,
2. Total estimated canopy cover, and
3. Density of woody species.

The method provides a site description of woody vegetation composition and structure. The variable quadrat method has been applied successfully by numerous vegetation scientists and researchers (Molatlhegi, 2008; Mogodi, 2009; Comole, 2014).

4.1.2 Materials and Methods

4.1.2.1 Materials

We made use of the following materials (Figure 4.2):

1. Ranging poles
2. Measuring tape
3. Measuring stick
4. GPS apparatus

4.1.2.2 Methods

The variable quadrat method by Coetzee & Gertenbach (1977) was used to collect data in both the survey site and the benchmark. A benchmark is a site whose veld condition is considered to be outstanding (Chapter 5) in terms of management objectives (Tainton, 1999). A benchmark site is used for comparison with the veld in the same ecological zone and with the same macroclimatic conditions to differentiate between the influences of climate and management (Tainton, 1999). The benchmark was located a kilometer away from the survey site (Figure 3.2, Chapter 3). A handheld GPS device with an accuracy of five meter was used for position location in the field. Woody structure and composition were recorded per species, height class and canopy regime at different height levels. Quadrant size was determined independently at each height class to describe the density and distribution of plants (Figure 4.1). Figure 4.1 illustrates the procedure for 1 m high plants and shows that 5 m x 5 m squares were large enough to include a 1 meter high plant in three quadrants but in the fourth, a 10 m x 10 m square was required to include such a plant. The quadrant size for 1 m high plants was therefore four contiguous, nested, 10 m x 10 m squares, i.e. one square of 20 m x 20 m with an area of 400 m². The procedure was repeated to determine a suitable quadrant size for each height class. A specially designed data sheet was used to record the data.

The canopy diameter of each woody species was measured and recorded. This was due to the fact that species of different height levels do not produce the same canopy cover and do not affect the herbaceous layer in the same manner. This was indicated by Tainton (1999) wherein he stated that the larger the tree, the larger the area of resource consumption will be and the greater its competition with its neighbors. Ngaruka (2011) stressed that *Senegalia mellifera* trees of heights between 0.6 m and 2.5 m have canopies where the maximum diameter exists at the top of the trees and that smaller or equal to 2 m high trees are the most problematic in bush encroachment.

The surveys were conducted in 2013 and 2014 and the presented data is the average between the different years of sampling. GPS readings ensured that the same spots were resampled. The identified species were counted and recorded.

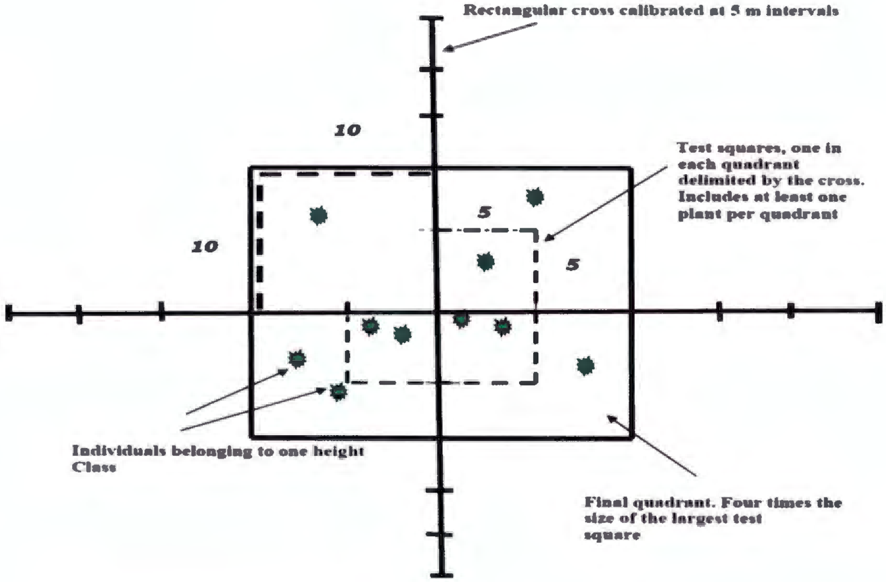


Figure 4.1: Technique by Coetzee & Gertenbach (1977) for discovering quadrant size for a height class, e.g. 1 m high plant. Test squares are expanded in steps until at least one plant is included

Figure 4.2: Materials used when collecting data in the selected sites in Disaneng

Every woody plant species canopy diameter was measured and recorded because woody species of different height classes do not produce the same canopy cover. The total of each class was thus calculated separately. Observed disturbances that led to the acquired results were also noted and recorded.

The data collected by using the variable quadrant method (Coetzee & Gertenbach, 1977) was used as baseline data to confirm the remote sensed data to identify and map the different categories of bush encroachment (Chapter 6). The recommendations acquired by the Department of Agriculture used to describe the extent of bush encroachment are shown in Table 4.1.

Table 4.1: Demonstration of the extent of bush encroachment (TE ha^{-1}) (Moore & Odendaal, 1987; Bothma, 1989; National Department of Agriculture, 2000)

Tree equivalents per hectare (TE ha^{-1})	Extent of bush encroachment
200-457	Encroached (grass production declines linearly)
458-714	Highly encroached (Grass suppressed further)
715-1099	Severely encroached (More grass suppression)
>2000	Grass growth almost totally suppressed

4.1.3 Woody plant density

Data analysis started by adding up all the individual woody (bushes/trees) species to determine the total number of woody individuals in every height class for all woody species encountered. The bush/tree equivalent is used to express the tree population of different sites in a common currency (Tainton, 1999). The tree densities were expressed as Evapotranspiration Tree Equivalents per hectare (ETTE.ha^{-1}) wherein 1 ETTE is equivalent to mean leaf volume of a single stemmed tree with a height of 1.5 meter equal to 500 cm^3 leaf volume (Smit, 1989). According to Rothauge (2011) an ETTE is the amount of water required by a woody species of 1.5 meter in height. The number of ETTE thus gives an indication of how much soil moisture is extracted from the soil by woody plants (Smit, 1989). Rothauge (2011) stated that 1 ETTE will be 1 TE (Tree Equivalent). Thus implies that 1 $ETTE.ha^{-1}$ equals to 1 $TE.ha^{-1}$. Dreber *et al.* (2014) expressed woody plant density in woody plants per hectare (woody plants ha^{-1}). Phytomass is expressed in $TE.ha^{-1}$ (Tree Equivalents ha^{-1}) where for every height class, the number of 1 TE= $n \times h \times 1.5^{-1}$, where n is the number of plants ha^{-1} and h is the mean height of the tree (m). The densities of woody species in this document will thus be expressed as $ETTE.ha^{-1}$ as stated by Smit (1989).

The density of woody species was calculated as the number of woody species per test square and converted to the number of (tree equivalent) woody species (trees or bushes) per hectare. In this way, total woody plant density for every species could be determined as per height class. It is ideal to attain bush density for every species in order to determine and compare the extent of bush density for the different woody species. This assists in understanding the diversity of woody species that contribute largely to bush encroachment of the area and also in the management of dominant encroaching species.

4.2 REMOTE SENSING

4.2.1 Introduction

The use of remote sensing techniques in the estimation of land degradation has been indicated in a number of researches (Hudak & Wessman, 1998; O'Connor & Crow, 1999; Hudak & Wessman, 2001; Roques *et al.*, 2001; Laliberte *et al.*, 2004; Martin & Asner, 2005). Hudak & Wessman (1998) and O'Connor & Crow (1999) used aerial photographs while Hudak & Wessman (2001) used satellite imagery.

A combination of aerial images and satellite images have also been used (Hudak & Wessman, 2001; Laliberte *et al.*, 2004) as well as airborne hyperspectral images (Martin &Asner, 2005). Multi-temporal panchromatic aerial images were used before the satellite images age (Laliberte *et al.*, 2004) where changes in image quality as woody cover expands have been used as key indicator of bush encroachment (Roques *et al.*, 2001).

The categorizing algorithms utilized in distinguishing and mapping bush encroachments on remotely sensed images have been found to be varied, along with varied associated classification accuracies (Munyati *et al.*, 2011). Multi-temporal remote sensing offers possibilities of expanding techniques to quantify the proportions and monitor the increase of land degradation due to the availability of historic imagery in archives (Munyati *et al.*, 2011). In this study the data acquired by using the variable quadrant method (Coetzee & Gertenbach, 1977) was used as training data that was used to confirm the remote sensed data to identify and map the different categories of bush encroachment (Moore & Odendaal, 1987; National Department of Agriculture, 2000).

4.2.2 Materials and methods

4.2.2.1 Materials

The research made use of the following materials:

1. SPOT 4 and SPOT 5 images

2. ERDAS Imagine 8.4
3. GPS device

4.2.2.2 Methods

ERDAS Imagine 8.4 was used to process satellite images and the supporting mapping was undertaken using ArcGIS 10.2. A handheld GPS device with an accuracy of 5 m was also used for position location in the field.

4.2.2.2.1 Image data

Système Pour l'Observation de la Terre (SPOT) satellite images were used for the research because of the necessity for high spatial resolution and availability in a long-term archive. SPOT images were acquired in a digital format, from the South African National Space Agency (SANSA).The oldest image acquired dated back to 2000 and the latest to 2009 (Table 4.2). The image dates transverse across the late rain season (autumn) to early spring period for the area. This was a perfect time of choosing the images to be processed for studying encroachment because the grass was in senescence whilst the tree species had fully grown leaves which enabled the mapping of locations with woody vegetation per date and the monitoring of bush encroachment over the time span represented.

Table 4.2: List of SPOT images used in the research

Date	K/J Reference	Satellite	Mode	Spatial resolution
21 September 2000	127-402	SPOT 4	Multispectral (XS)	20 m
03 May 2004	127-402	SPOT 4	Multispectral (XS)	20 m
11 May 2009	127-402	SPOT 5	Multispectral (XS)	10m

4.2.2.2.2 Geometric correction

Geometric correction of images is a vital requirement which must be implemented prior to using images in geographic information system and other processing programs (ERDAS Field Guide, 1999). The images were geographically registered to a common projection (UTM WGS 84) using the geometric correction in ERDAS. The geometric registration of images was done using the ground control points (GCP) wherein the registration error (RMSE) in the process was less than a pixel. GCP were identified between two images in recognizable locations. These points were static relative to temporal change. In this case road intersections and bridges over water were the best GCP and their coordinates were noted. Features that move through time were avoided.

Using a minimum of 4 GCP's, the Root Mean Square (RMS) error was computed as the distance between the source location of a GCP and the converted position for the same GCP. RMS error was calculated when a minimum of three GCP points were collected to define the relationship between the source and the reference coordinate system. After each computation of a transformation and RMS error, GCP's with higher RMS error were removed since the GCP was least accurate. The total RMS error for SPOT 4 images was equal to 0.0005 m and for SPOT 5 image was equal to 0.0003 m.

4.2.3 Mapping woody vegetation on image data

Supervised classification was used to map woody vegetation of the study site. The following classes were used for the classification:

1. Woody vegetation,

2. Senescent grass, and
3. Bare dry surface.

Representative samples for each class were selected. These sample land cover classes are called "training sites". The image classification software used the training sites to identify the cover classes in the entire image. The classification of land cover is established on the spectral signature interpreted in the training set. The digital image classification software modifies each class on what it resembles most in the training set.

In this study, maximum likelihood was used when classifying the images, because it is relatively more accurate. Three training area signatures were added. The signatures were created by drawing a polygon at an area representing the class of interest then the signature from the area of interest (AOI) was created. The actual classification was then performed using the maximum likelihood algorithm.

Change in woody species abundance (as indicator of the bush encroachment process) was detected by classifying the area of cover by the woody vegetation class from the earliest to the newest image. Therefore, the method of post classification comparison was used to quantify change in woody cover (i.e. encroachment). The results are presented and discussed in chapter 6.

CHAPTER 5

WOODY PLANT ENCROACHMENT IN THE STUDY AREA

Results and Discussion

5.1. RESULTS

5.1.1 Bush encroachment in Disaneng (Benchmark)

The benchmark site was chosen as a natural veld reference (Figure 3.1, Chapter 3) because of "good" veld conditions and experienced limited veld disturbances. The benchmark was located within the same ecological zone as the study site (Figure 3.1, Chapter 3). This site had a few large scattered trees and a well-established grass layer, with *Eragrostis lehmanniana* and *E. rigidior* as the only grasses present. A benchmark site is used for comparison with the veld in the same ecological zone and with the same macroclimatic conditions to differentiate between the influences of climate and management practices on the vegetation.

Woody plant densities were recorded in variant height classes and represented as Evapotranspiration Tree Equivalents per hectare (ETTE.ha^{-1}) (Chapter 4). This was done, because small and large trees do not have the same canopy regimes and thus will influence the understory vegetation differently. The woody plant density in the benchmark (reference site) was determined at 1 297 ETTE.ha^{-1} (Figure 5.1). *Grewia flava* was the most abundant woody species at a density of 827 ETTE.ha^{-1} (Figure 5.1). *Vachellia erioloba* was determined at a density of 354 ETTE.ha^{-1} and *Senegalia mellifera* 111 ETTE.ha^{-1} (Figure 5.1). Other woody species observed, included *Ziziphus mucronata* and was noted at a density of only 5 ETTE.ha^{-1} (Figure 5.1). According to Table 4.1 (Chapter 4), *Ziziphus mucronata* was at a "non-threatening" state. Woody species densities within different height classes (0.5 m, 1 m, 2 m, 3 m, 4 m, 5 m, 6 m) in the benchmark site were compared with the densities (per height class) in the study area (refer to chapter 4).

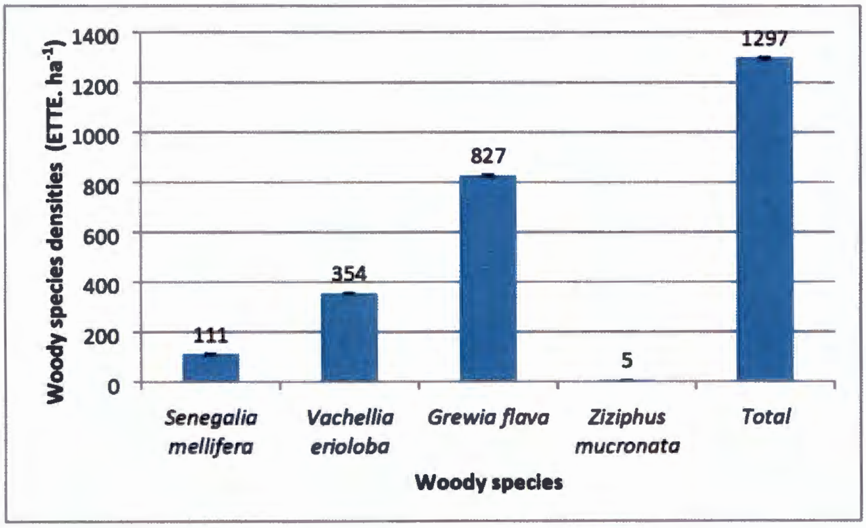


Figure 5.1: Density of woody species in the benchmark

Woody plants within the 0.5 m height class (Maximum height of 0.5 m) were limited and were noted at a total density of 103 ETTE.ha^{-1} (Figure 5.2). *Senegalia mellifera* was determined at a density of 47 ETTE.ha^{-1} while *Grewia flava* was the dominant species and occurred at a density of 51 ETTE.ha^{-1} while *Ziziphus mucronata* was present at a density of 5 ETTE.ha^{-1} (Figure 5.2). According to Table 4.1 (Chapter 4), the juveniles (plants smaller than 0.5 m) did not encroach in this site and the grasses were not suppressed due to low competition between grasses and woody juveniles, especially for water and nutrients.

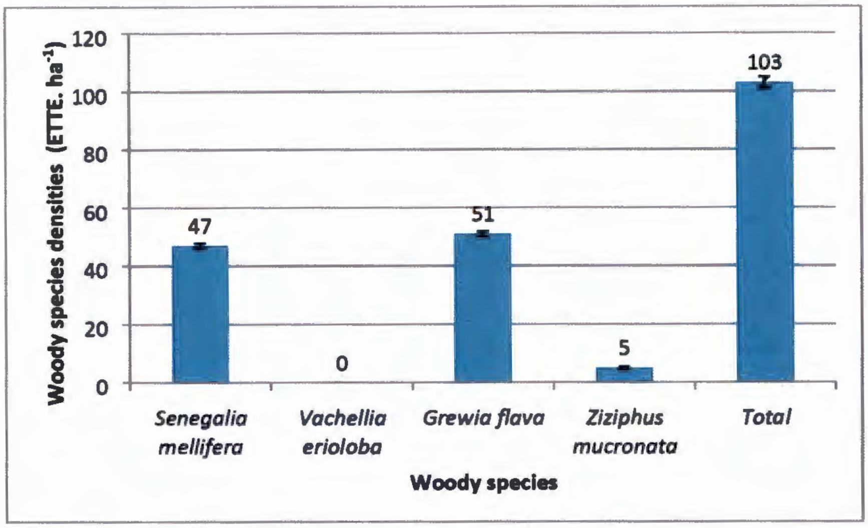


Figure 5.2: Density of woody species within the 0.5 m height class in the benchmark

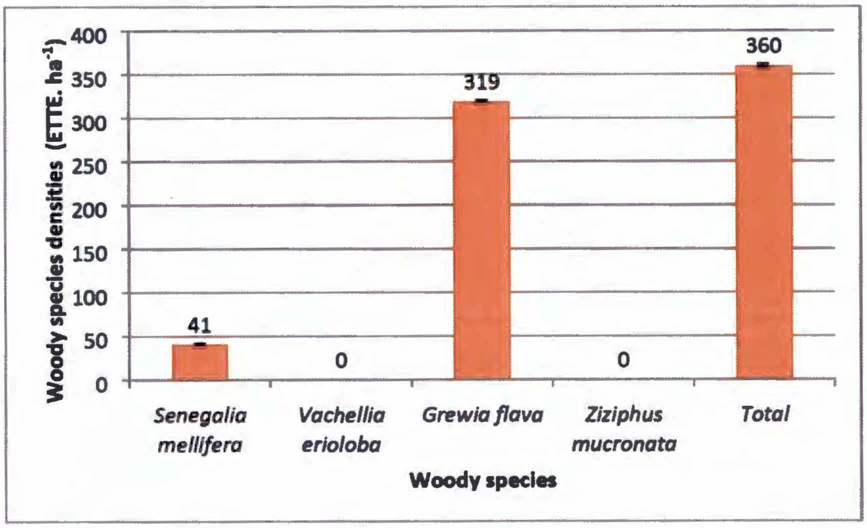


Figure 5.3: Density of woody species within the 1 m height class in the benchmark

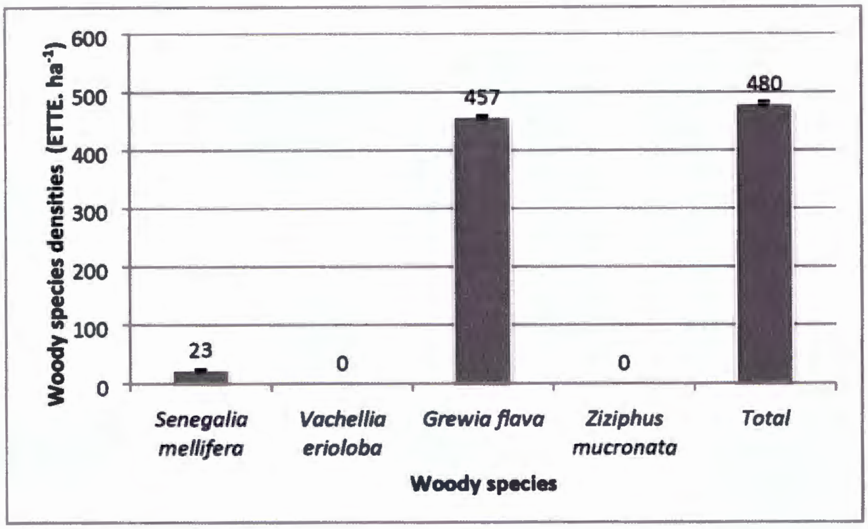


Figure 5.4: Density of 2 m high woody species in the benchmark

One meter high (Figure 5.3) and two meter high woody plants (Figure 5.4) were the most abundant in the reference (benchmark) site. The highest woody species density was present for two meter high trees (two meter height class) with a total density of 480 ETTE.ha^{-1} where *Grewia flava* was the dominant species (Figure 5.4).

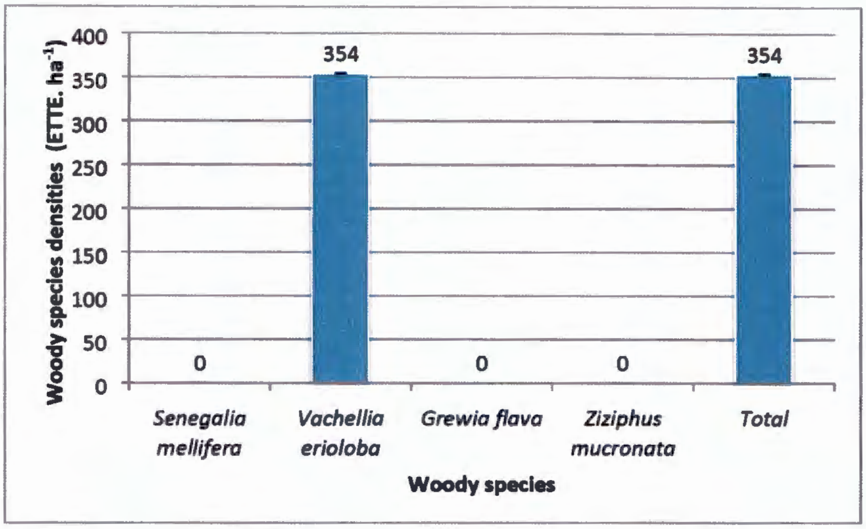


Figure 5.5: Density of woody species within the 6 m height class in the benchmark

The maximum height of woody species in this site was six meters (Figure 5.5). No woody species within the 2 m, 3 m, 4 m, and 5 m height classes were encountered. *Vachellia erioloba* was the only woody species present and was present at a density of 354 ETTE.ha[1] (Figure 5.5).

Figure 5.6 provides canopy diameter information of trees of different heights. *Vachellia erioloba* trees produced the widest average canopy diameter of 4.84 meters (Figure 5.6) when compared to other woody species. *Ziziphus mucronata* had the lowest canopy diameter of only 0.23 meter (Figure 5.6) and did not affect the herbaceous layer in a substantial way. According to figure 5.7 *Grewia flava* produced the widest canopy diameter of 3 772 m in the area.

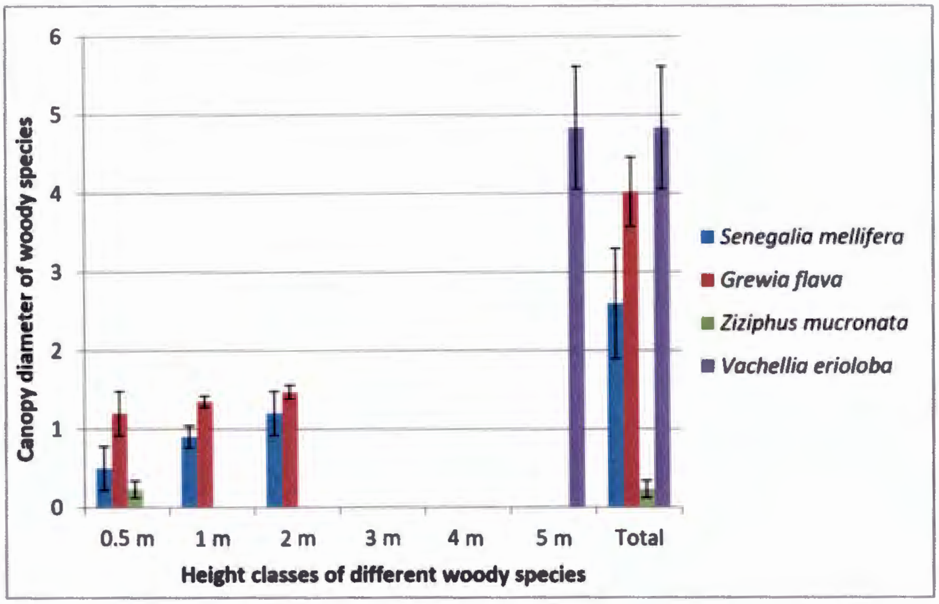


Figure 5.6: Average canopy diameter (m) of woody species at different height levels in benchmark

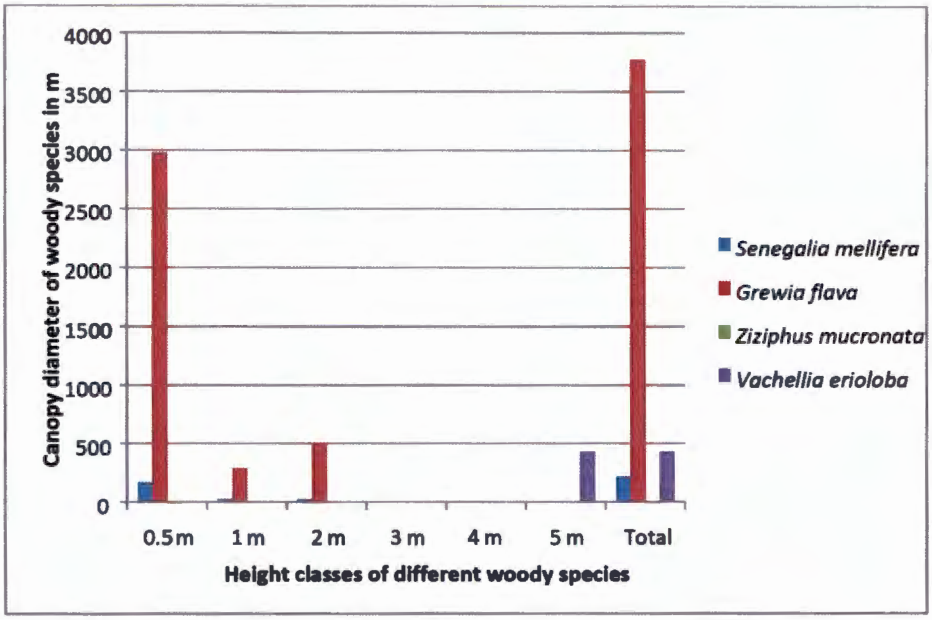


Figure 5.7: Total canopy diameter (m) of woody species at different height levels in the benchmark

5.1.2 Bush encroachment in selected sites of Disaneng

5.1.2.1 Total woody plant encroachment

The total density of all woody species in the research site was noted at 13 547 $ETTE.ha^{-1}$ (Figure 5.8) which will almost completely suppress grass growth (Table 4.1, Chapter 4) since the density of woody encroachers exceeds 2 000 $TE.ha^{-1}$ (Moore & Odendaal, 1987). Encroaching woody species included *Senegalia mellifera*, *Vachellia tortilis*, *Grewia flava* and *Ziziphus mucronata* (Figure 5.7). *Senegalia mellifera* was the most abundant woody encroacher at a density of 7 106 $ETTE.ha^{-1}$ while *Grewia flava* was present at a density of 4 234 $ETTE.ha^{-1}$ (Figure 5.7). *Vachellia tortilis* was the only species present, which did not occur in the benchmark site (Figure 5.1).

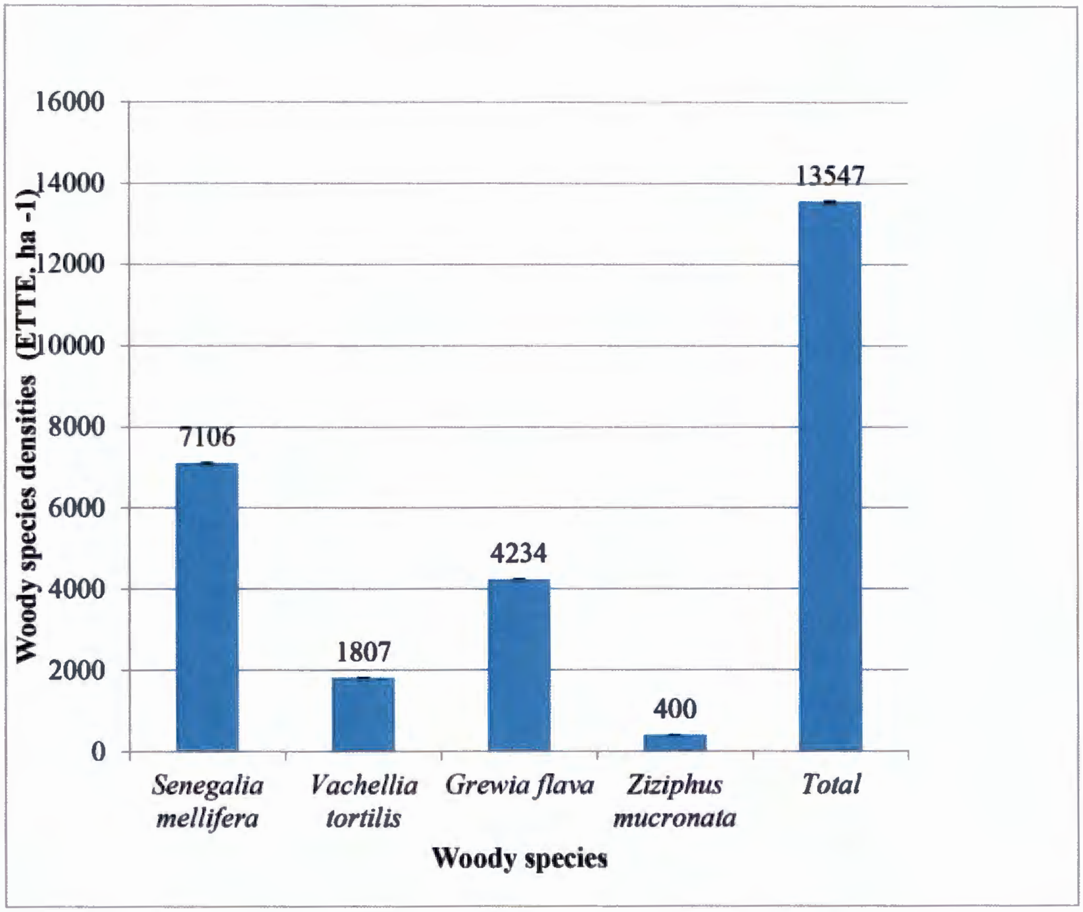


Figure 5.8: Total density (ETTE.ha^{-1}) of woody species in Disaneng

5.1.2.2 Woody plant encroachment per height class in the study area

The density of woody juveniles (0.5 m high trees) was recorded at a density of 3 743 ETTE.ha-1 (Figure 5.9). *Senegalia mellifera* (1 886 ETTE.ha^{-1}) juveniles contributed most to bush encroachment and according to Table 4.1(Chapter 4), woody species occurring at this density will completely suppress grass production. Small woody species, mainly of *Senegalia* and *Vachellia* species, were observed growing away from the parent trees while a herbaceous layer was limited and only individuals of *Eragrostis lehmanniana* and *E.rigidior* were noticed.

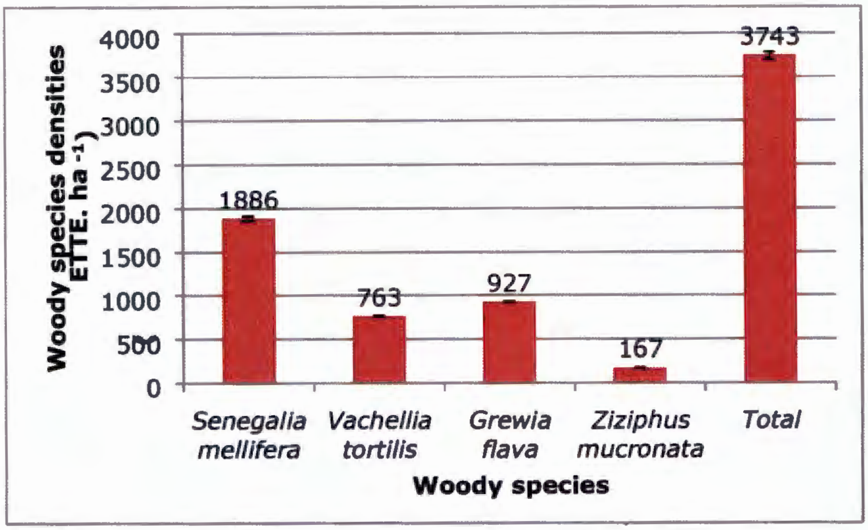


Figure 5.9: Density (ETTE.ha^{-1}) of woody species within the 0.5 m height class in Disaneng

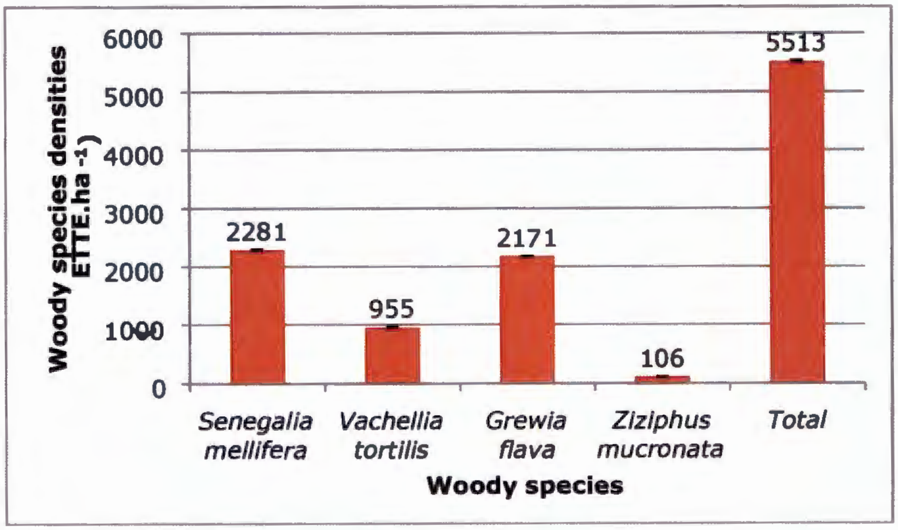


Figure 5.10: Density (ETTE.ha^{-1}) of woody species within one meter height class in Disaneng

The density of the one meter high woody plants (plants between 0.5 m and 1 m in height) was determined at a total density of 5 513 ETTE.ha^{-1} (Figure 5.10), compared to a density of 360 ETTE.ha^{-1} in the benchmark (Figure 5.3). Plants within this height class contributed most to woody plant density, as compared to plants in the other height classes (Figure 5.10). This is an indication of intra-species competition wherein trees of similar species compete for accessible resources such as water, nutrients and location. Their spread could also restrict the movement of animals in the bush area.

One meter (1 m) high *Senegalia mellifera* trees occurred at a density of 2 281 ETTE.ha^{-1} (Figure 5.10), while it was observed at a density of only 41 ETTE.ha^{-1} in the benchmark (Figure 5.3). Due to the density of *Senegalia mellifera* trees no grasses were observed growing beneath these trees. *Grewia flava* was present at a density of 2 171 ETTE.ha^{-1} (Figure 5.10) locally, compared to a density of 319 ETTE.ha^{-1} in the benchmark (Figure 5.3).

(Figure 5.10), while it was observed at a density of only 41 ETTE.ha^{-1} in the benchmark (Figure 5.3). Due to the density of *Senegalia mellifera* trees no grasses were observed growing beneath these trees. *Grewia flava* was present at a density of 2 171 ETTE.ha^{-1} (Figure 5.10) locally, compared to a density of 319 ETTE.ha^{-1} in the benchmark (Figure 5.3).

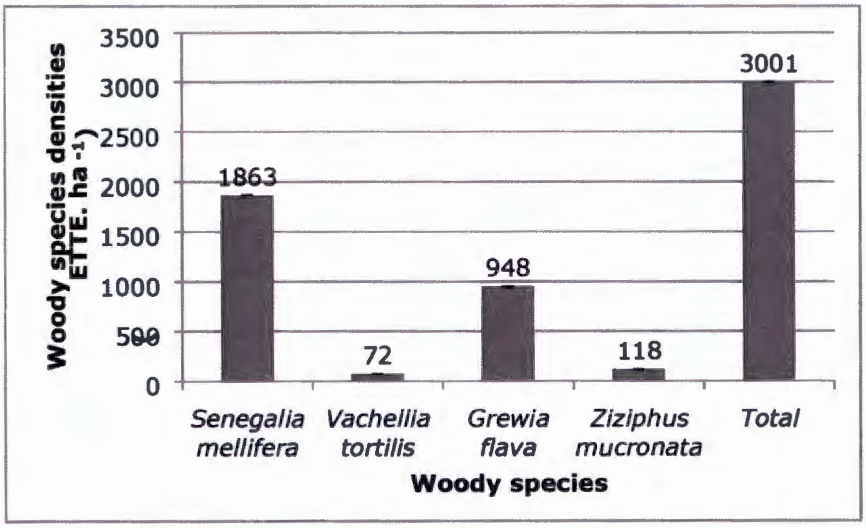


Figure 5.11: Density (ETTE.ha^{-1}) of two meter woody species in Disaneng

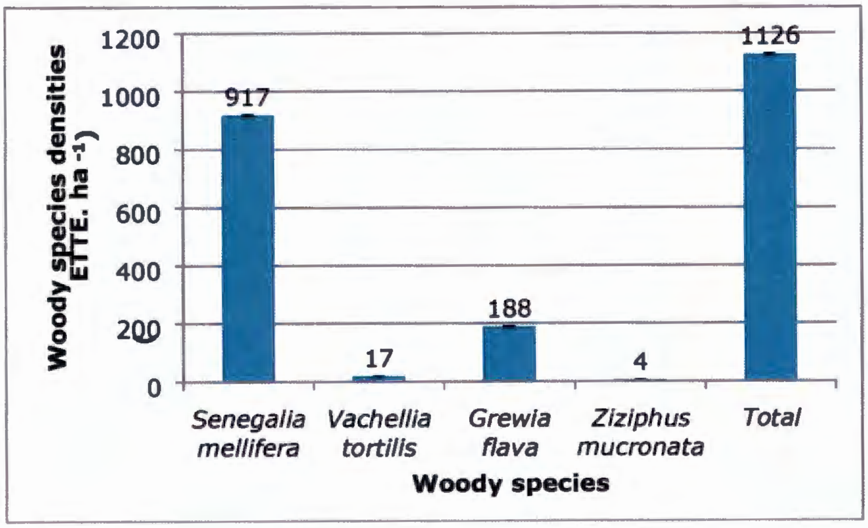


Figure 5.12: Density (ETTE.ha^{-1}) of three meter high woody species in Disaneng

According to Figure 5.11 and Figure 5.12, *Senegalia mellifera* was noted at high densities (1 863 ETTE.ha^{-1} for 2 m and 917 ETTE.ha^{-1} for 3 m high trees). In the benchmark, the highest woody species density was represented by two meter high trees (Figure 5.4) where *Senegalia mellifera* was present at a low density of only 23 ETTE.ha^{-1} (Figure 5.4). *Grewia flava* was the sole dominant woody species in the benchmark (Figure 5.1). *Grewia flava* was identified up to three meter high plants, with the highest density noted at 2 171 ETTE.ha^{-1} for one meter high individuals (Figure 5.10). *Vachellia tortilis* was only present up to three meter high plants and occurred at a density of 955 ETTE.ha^{-1} for one meter high trees (Figure 5.10) and 72 ETTE.ha^{-1} and 71 ETTE.ha^{-1} for two meter and three meter high individuals respectively (Figure 5.11 and 5.12), *Vachellia tortilis* was absent in the benchmark (Figure 5.1).

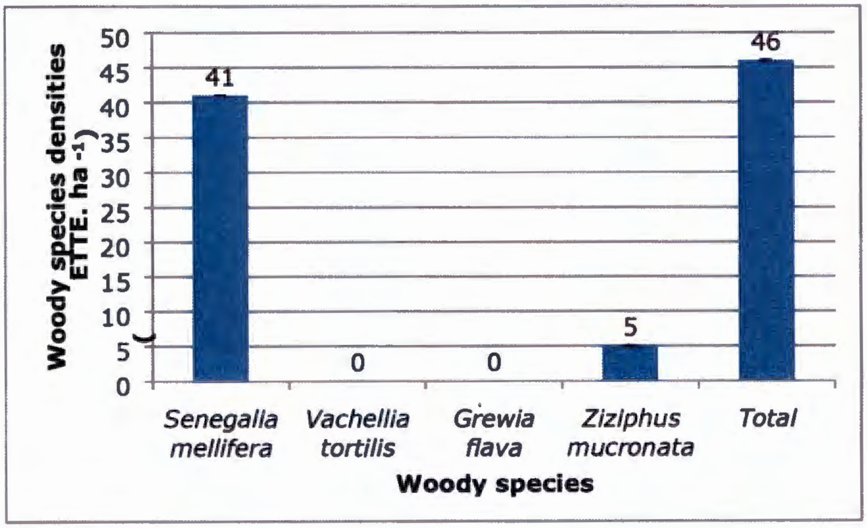


Figure 5.13: Density (ETTE. ha^{-1}) of woody species within the four meter height class in Disaneng

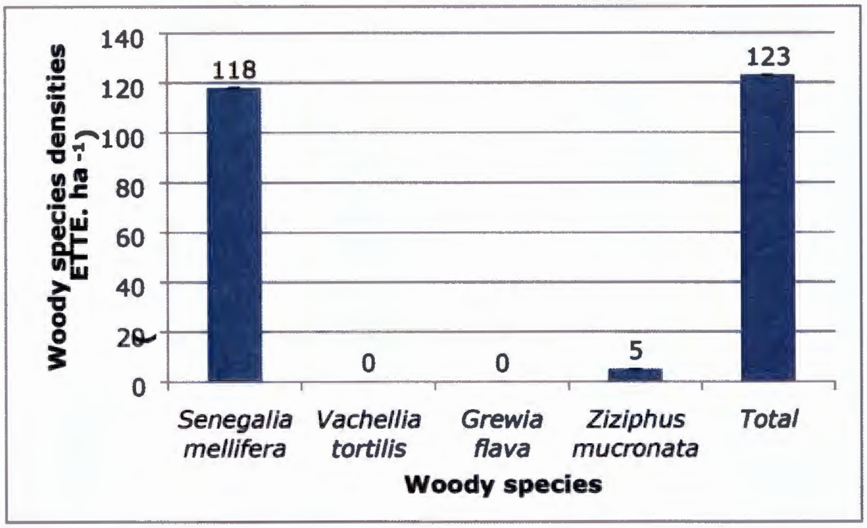


Figure 5.14: Density (ETTE.ha^{-1}) of woody species within the five meter and six meter height class in Disaneng

The maximum height of woody species in both the study site and the benchmark was six metres (Figure 5.5 and Figure 5.14). Five meter high *Senegalia mellifera* trees were limited and occurred at a density of 118 ETTE.ha^{-1} in the survey sites. Woody species within a maximum height of four meter and five meter were limited and were determined at a density of 46 ETTE.ha^{-1} (Figure 5.13) and 123 ETTE.ha^{-1} (Figure 5.14) respectively, showing that high trees do not necessarily cause the major encroachment problems.

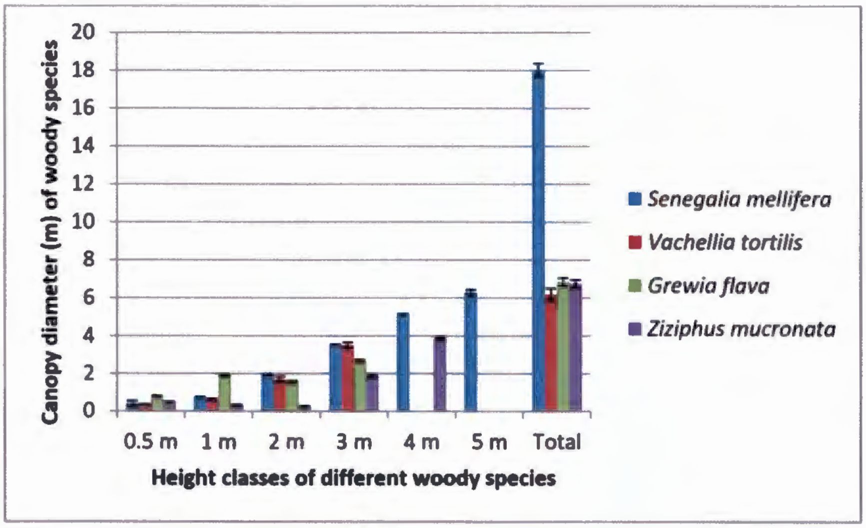


Figure 5.15: Average canopy diameter (m) of woody species at different height levels in Disaneng

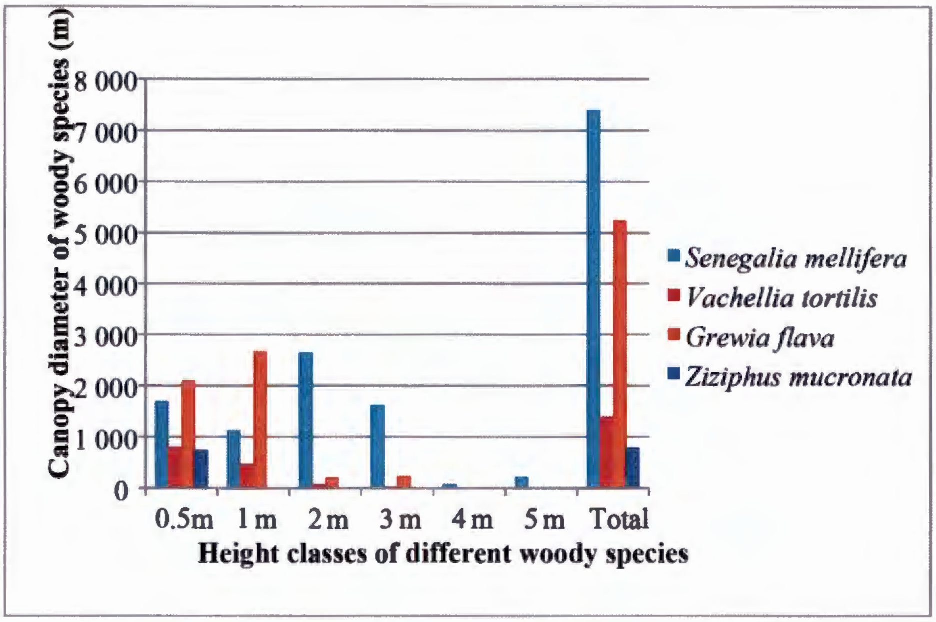


Figure 5.16: Total canopy diameter (m) of woody species at different height levels in Disaneng

Senegalia mellifera had the widest canopies of all the woody encroachers and recorded an average canopy diameter of 18.03 meters, followed by *Grewia flava* at an average canopy diameter of 6.85 meters, *Ziziphus mucronata* and *Vachellia tortilis* at 6.76 meters and 6.18 meters respectively (Figure 5.15). Four meter high *Senegalia mellifera* trees recorded the widest average canopy diameter of 5.14 meters (Figure 5.15) when compared to *Senegalia mellifera* trees in other height classes. Juveniles did not record wide canopies. Figure 5.16 indicates that species under height class 2 meter and 3 meter high woody species had the widest canopies and *Senegalia mellifera* recorded a total canopy diameter of 7 389 meters in the area (Figure 5.16).

5.2. DISCUSSION

5.2.1 Benchmark site

It was indicated by Buba (2015) that the shading produced by trees, rainfall interception and interference with light penetration affect their environment while the physical and chemical properties of the soil are changed by the quantity and quality of litter they produce. Sharma and Sharma (2013) stated that the amount of the above mentioned impacts through interference with sunlight availability and soil nutrient amplification also depends on tree species and size.

The results of this study show that the benchmark site was not encroached by woody species as a low woody plant density of (1 297 $ETTE.ha^{-1}$) was observed in this site. *Senegalia mellifera* was present, but at a low density.

Joubert (2014) defined "natural" veld as a natural veld where the grassy state is dominated by 2-3 meter high single stemmed woody species wherein *Senegalia mellifera* is typically rare. According to Coates Palgrave (1990), *Senegalia mellifera* is an indicator of overgrazing. The presence of this woody species at this density will not have a negative effect on grass production (Table 4.1, Chapter 4), implying that grass will be able to grow in the benchmark site. Although the benchmark is also encroached with woody species, it was the only site in the visinity that had a substantial lower density of woody species especially *Senegalia mellifera*. Other "potential" sites for refering were simply too degraded in terms of soil and plant degradation. Richter *et al*. (2001) indicated that a bush density higher than 1 500 trees/ha can be considered as a possible threshold between an encroached and a non-encroached site.

Table 5.1: Woody plant densities in benchmark (Reference) sites in the Molopo Area

Reference	Location	Woody plant density (ETTE. ha^{-1})	Dominant woody species
Molatlhegi (2008)	Disaneng	1 007	*Senegalia hebeclada, V. erioloba, Ziziphus mucronata*
Mogodi (2009)	Disaneng	407	*Senegalia hebeclada, V. erioloba, V. tortilis, Grewia flava , Ziziphus mucronata*
	Tshidilamolomo	1 110	*Vachellia erioloba, Senegalia mellifera*
Comole (2014)	Loporung	1 077	*Vachellia erioloba, Senegalia hebeclada, Tarchonanthus camphoratus*
	Makgori	297	*Senegalia mellifera*

Similar studies in the Molopo Area were conducted by Molatlhegi (2008), Mogodi (2009) and Comole (2014). Table 5.1 provides a summary of these results.

Molatlhegi (2008) noted a total woody species density of 1 007 ETTE.ha^{-1} in a reference site in Disaneng with *Senegalia hebeclada* (471 ETTE.ha^{-1}) and *Vachellia erioloba* (370 ETTE.ha^{-1}) as the prominent woody species. *Ziziphus mucronata* was the only common woody species found by Molatlhegi (2008) with this benchmark site (Figure 5.1). Molatlhegi (2008) recorded *Ziziphus mucronata* at a low density of 40 ETTE.ha^{-1} which is similar to low density noted in this study for *Ziziphus mucronata*.

Mogodi (2009) recorded a woody density of only 407 ETTE.ha^{-1} in the reference (benchmark) site in Disaneng (Table 5.1). Woody species present included *Senegalia hebeclada* (17 ETTE.ha^{-1}), *V. erioloba* (278 ETTE.ha^{-1}), *V. tortilis* (46 ETTE.ha^{-1}), *Ziziphus mucronata* (44 ETTE.ha^{-1}) and *Grewia flava* (22 ETTE.ha^{-1}). *Senegalia mellifera* was not observed within the reference site (Mogodi, 2009), while *Vachellia tortilis* and *V. hebeclada* were not found during this survey (Figure 5.1). According to Condit *et al.* (1998), size dispersal interpretation of *Vachellia tortilis* showed absence of juveniles which is an indication of very low recruitment and high death rate of small and large trees. Absence of juveniles suggested that *Acacia* (*Vachellia* and *Senegalia*) trees are facing population depression. These results were high when compared to other studies conducted in Disaneng.

Two metre high woody species were the most abundant in the benchmark site (Figure 5.6), while 0.5 meters high trees were the most abundant in Disaneng and were recorded at a density of 1 380 ETTE.ha $^{-1}$ (Molatlhegi, 2008). The woody species density noted by Molatlhegi (2008) was low when compared to the current results (Figure 5.1) which is an indication that there is an increase in woody species density in the area (Disaneng) over time.

Mogodi (2009) reported that large trees (6 meter in height) were the most prominent during 2009 in the reference (benchmark) site, also in Disaneng. Common woody species encroachers encountered by Mogodi (2009) and encountered in this benchmark site include *Grewia flava*, *Vachellia erioloba* and *Ziziphus mucronata* (Figure 5.1 and Table 5.1).

In another study conducted in the Molopo District by Comole (2014), a woody plant density of 1 110 ETTE.ha^{-1} (Table 5.1) was encountered in the benchmark site in Tshidilamolomo (Refer to Figure 3.1, Chapter 3). *Vachellia erioloba* was the most abundant woody species and was present at a density of 367 ETTE.ha^{-1} while *Senegalia mellifera* was present at 321 ETTE.ha^{-1}. One metre high woody species were the most abundant in Tshidilamolomo (Comole, 2014). These findings were not consistent with the results found in the benchmark of this study, where *Grewia flava* was the dominant encroacher (Figure 5.1). Comole (2014) found woody plants at a density of 1 077 ETTE.ha^{-1} (Table 5.1) in the benchmark in Loporung (Refer to Figure 3.1, Chapter 3) where *Senegalia hebeclada* (324 ETTE.ha^{-1}), *V. erioloba* (267 ETTE.ha^{-1}) and *Tarchonanthus camphoratus* (326 ETTE.ha^{-1}) were prominent. *Vachellia erioloba* trees with a maximum height of 0.5 meter and 3 meter high *Tarchonanthus camphoratus* trees were the most abundant (Comole, 2014). Comole (2014) found woody species at a density of 297 ETTE.ha^{-1} in Makgori Village (benchmark 2 of Comole (2014)) (Refer to Figure 3.1, Chapter 3) where *Senegalia mellifera* was the most prominent encroacher. In this research, *Senegalia mellifera* was measured at a density of 111 ETTE.ha^{-1} with 2 meter high plants being the most abundant (Figure 5.4).

Human activities such as unsupervised grazing of cattle, creation of footpaths and trampling by both animals and human were not observed in the benchmark. Although this benchmark site was located within a communally managed area the grazing of cattle was well supervised by cattle herders. The soils of this site were covered by a herbaceous layer, mainly grasses (*Eragrostis lehmanniana* and *E. rigidior*), which prevented soil erosion. *Grewia flava* was the most prominent woody species (Figure 5.1) and was observed growing beneath *Vachellia erioloba* trees. In open Kalahari savannas, *Grewia flava* typically grows below *Vachellia erioloba* trees (Dean *et al.*, 1999) whereas under cattle grazing areas it may also encroach in tree interspaces (spaces between encroaching trees) (Skarpe, 1990).

Tews *et al.* (2005) stated that cattle are the main vehicle in the spread and establishment of *Grewia flava* in savannas as they consume large quantities of *Grewia flava* and play an important role in seed distribution. According to Schurr (2001) their dung pads are known to contain large scale of *Grewia flava* seeds. Skarpe (1990) reported *Grewia flava* to severely encroach and form dense shrub thickets in savanna regions. According to Tews *et al.* (2005), *Grewia flava* encroachment in rangelands should be limited through the use of frequent fires but requires a dense grass stand to be implemented.

According to Scholes and Archer (1997) and Stuart-Hill *et al.* (1987), large trees are able to promote grass growth under their canopies but the net result is dependent on tree density. Canopy morphology of woody plants contributes towards their proliferation (Tainton, 1981), because of light interception as well as nutrients that build up directly below their canopies (Belsky & Amundson, 1992). Large trees in semi-arid regions have also shown to be creating "islands of fertility" (Hagos & Smit, 2005) below their crowns which could facilitate grass growth. According to Esteban *et al.* (2004) nutrient and water uplift in plants can affect the vertical distribution of soil nutrients, counterbalancing the downward transport of nutrients by leaching and weathering. If soil erosion rates are high, nutrient and water uplift in plants may favour soil nutrient content by depositing nutrients to the soil surface (Esteban *et al.*, 2004). Large trees have three times more Ca and Ca: K ratios five times higher in their leaves than in small trees. It was indicated by Esteban *et al.* (2004) that soils around large trees contained more nutrients compared to soils beneath small trees wherein nutrients under the small plants were depleted in intermediate soil layers. In the benchmark woody species promoted grass growth underneath their canopies.

Grewia flava is harvested to make brushwood fencing and shelter, fuel for cooking, use of the fruits for making beer and is also used as laths for roofing traditional houses in communal (Mainah, 2006). *Grewia flava* plants were uprooted to obtain the roots, which are considered to have medicinal value (Mainah, 2006). This species is browsed by donkeys and goats, since they are less selective feeders (Mainah, 2006).

Mainah (2006) reported that, when green palatable vegetation cover becomes scarce, *Grewia flava* becomes a more preferred browse. Products obtained from *Grewia flava* include berries (fruits), roots, stems and leaves (Mainah, 2006). The leaves are 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 (Kgadi).

Vachellia erioloba was noted in the reference sites of Molatlhegi (2008) and Mogodi (2009) in Disaneng and Comole (2014) in Tshidilamolomo and Loporung (Table 5.1). The presence of *Vachellia erioloba* in the benchmark, renders the site ecologically important due to its ability to increase soil nutrient levels beneath their canopies (Materechera & Materechera, 2001) and mitigate soil erosion (Barnes, 2001) due to its potential to suppress growth of other *acacias* (previous botanical name for thorn trees) (Tainton, 1999). This was illustrated by Rothauge *et al.* (2003) indicating higher soil N, P, K, Mg, Na and organic matter in Namibia's camel-thorn savannas.

Vachellia erioloba has a nutritional value as fodder for livestock and is used for various purposes by local people (Smit, 1999). The pods are useful fodder for cattle and are favoured by wild animals, especially elephants which chew the pods and disperse the seeds in their dung. The timber is strong and highly prized for firewood (Milton *et al.*, 2002).

Dry powdered pods are used to treat ear infections (Palgrave, 1990). The gum is also eaten by human as well as animals (Smit, 1999) and can be used for the treatment of gonorrhea while the pulverized, burnt bark is used to treat headaches. The roots are used to treat toothaches. To treat tuberculosis, the roots are boiled for a few minutes and the infusion is swirled around in the mouth and spat out. The seeds can be roasted and used as a replacement for coffee.

According to the National Department of Agriculture (NDA) (2000), *Vachellia erioloba* is not considered to be an encroaching species. It is a slow growing woody species (Palgave, 1990) that ranges from a 2 meter thorny shrub to a 16 meter flourishing tree. It grows well in poor soils and in harsh environmental conditions. *Vachellia erioloba* has one or two stems and its upright form and small canopy when young, allows grass and other herbaceous plants to grow below it (Belsky, 1994). It has a deep taproot (Smit, 1999) that makes it easy to interact positively with grasses because it does not compete directly with the shallow roots of grasses for available resources.

Vachellia erioloba is one of the protected trees in South Africa in terms of Section 12 of the National Forest Act, 1998 (Act No. 84 of 1998). Under this act, No person may:

a) Cut, disturb, damage, destroy or remove any protected tree; or

b) Accumulate, remove, transport, export, purchase, sell, donate or in any other manner acquire or dispose any protected tree, except under a license granted by the Minister of Water affairs and Forestry.

This Act does not distinguish between dead and living trees. This implies that removal of dead specimens without a permit is illegal. *Vachellia erioloba* (previously known as *Acacia erioloba*) has the potential of suppressing growth of other acacias. This is because most acacias have been shown not to grow under the canopies of other acacias (Tainton, 1999). *Senegalia mellifera* the most "problematic" encroaching species in the Molopo District for decades (Donaldson, 1967) is the exception. It is capable of growing beneath canopies of other acacias (previous botanical name of thorn trees) and was identified growing underneath *Vachellia erioloba* trees at this site.

A positive interaction of trees with herbaceous plants beneath their canopies is of interest for both economic and scientific reasons (Materechera & Materechera, 2001). Belsky *et al.* (1993) stated that the effects of tree species on the understory species which may be expressed by cover productivity and diversity are considered to be of economic value. In this case, *Vachellia erioloba* is regarded as the species of high economic value since it is harvested for fuel and its tendency to out-compete other woody species such as *Senegalia mellifera* for soil water and nutrients.

5.2.2 Study area

Figure 5.17: Bush encroachment in the study area

High densities of woody species were noted in the study area with *Senegalia mellifera* being the most prominent. As in the reference site, densities of woody encroachers per height class were recorded (0.5 m, 1 m, 2 m, 3 m, 4 m and 5 m). Plants of different heights do not have the same effect on the vegetation, due to usage of water, nutrients and shading as the canopy of a small tree does not have the same effect on the underneath vegetation as large trees (Joubert, 2007).

The only woody species noted in the research site were *Senegalia mellifera*, *Vachellia tortilis*, *Grewia flava* and *Ziziphus mucronata*. These encroaching species vary extensively with respect to phenological and physiological aspects of their life history, producing different pathways of bush thickening (Joubert *et al*., 2008). According to Kraaij and Ward

(2006), *Senegalia mellifera, Vachellia tortilis, Vachellia karroo* and *Dichrostachys cinerea* are considered the main encroaching woody species in African savannas. They are known to contain very high levels of phenolic compounds such as tannins in their leaves which are capable of reducing the digestibility for livestock (Mzezewa, 2009). Jacobs (2000) states that the combination of the species thorniness and low digestibility d their accessibility to livestock.

Senegalia mellifera dominated the vegetation (Figure 5.7). *Senegalia mellifera* has a shallow root system and out-competes grasses for water and soil nutrients (Smit, 1999) and grasses are limited beneath the canopies of this encroacher (Donaldson, 1967). Rothauge (2011) stated that the increase in bush density and cover causes a substantial decline in grass production, mainly through competition for soil moisture. A dense stand of *Senegalia mellifera* (*Senegalia mellifera*) dries out the soil, lowers the ground water table and severely restricts the amount of water for use for other rangeland plants. *Acacias (Vachellia* and *Senegalia*) are C_3-plants that use more water during photosynthesis and transpire more water in hot climates than C_4 plants (Rothauge, 2011). De Klerk (2003) reported that a 2.5 meter high *Senegalia mellifera* with a canopy spread of six square meters uses up to 64.8 litres of water during the day in the growing season, therefore, ane plant of 1.5 meter will thus use 38.9 liters of water daily. The total density of *Senegalia mellifera* in the study area was 7 106 ETTE.ha^{-1} (Figure 5.8) and the transpiration rate of 276 423 litres of water per day (according to the calculations of De Klerk, 2003). The high rate of transpiration rate leads to the reduction in soil water and it is evident that *Senegalia mellifera* plants will therefore outcompete more intensively for soil water.

Figure 5.18: The stem of *Senegalia mellifera* with leaves and thorns

Senegalia mellifera is a thorny tree with an extensive rounded to flatted crown which may reach down to the ground level and can grow to a height of up to 7 meters (Hagos & Smit, 2005). It is mostly found in sandy soils (Coates Palgrave, 2002) and also plays a important role in soil enrichment (Hagos & Smit, 2005). According to Hagos and Smit (2005), the positive effects (soil nutrient enrichment) of *Senegalia mellifera* need to be evaluated first before any woody control programme is initiated. Woody species, however, do not only have negative effects upon the understory grass productivity. According to Amundson *et al.* (1995) and Cadwell *et al.* (1998), trees have the ability to improve grass productivity through hydraulic lift, reduction in evapotranspiration and increasing nutrient availability. Belsky (1992) conducted a study on *Acacia* species (currently known as *Senegalia* and

Vachellia) and noted that there was higher forage beneath trees than in the open. It was also indicated by Cadwell *et al.* (1998) and Ludwig *et al.* (2004) that woody species may improve the water status of understory species while in some cases the grass productivity was high beneath tree canopies (Maranga, 1984). Leaf water potential of grasses growing k2beneath woody species canopies was higher than for those growing away from the canopies even under circumstances wherein there was less rain water reaching the ground under tree canopies and competition between trees and grasses for the water (Knoop & Walker, 1985).This was also illustrated by Joubert (2007) that increases in plant height and size may also stimulate a higher diversity of understory plants because of the patch effect and the increase in habitat heterogeneity beneath tree canopies.

Senegalia mellifera is the member of Fabaceae, capable of fixing nitrogen. In the study of estimating nitrogen fixation by trees on an aridity gradient in Namibia, Schultz *et al.* (1955) recorded nitrogen fixation capabilities of *Senegalia mellifera* trees. This nitrogen fixation fertilizes poor soils where it is found, allowing for the establishment of other herbaceous species in naturally nutrient-poor areas. This species is suspected to enrich soil N with *Rhizobium*-legume symbiosis (Young & Johnston, 1989). Hagos & Smit (2005) identified the availability of nutrients under *Senegalia mellifera* tree canopies and limited nitrogen in the open deep Kalahari sandy soils in South Africa. This can be ascribed to nitrogen fixation which improves the understory by enhancing soil fertility.

The presence of *Senegalia mellifera* is not just associated with soil enrichment, but can be useful in some communities. The twigs and pods of the plant are nutritious to livestock and game. It is browsed by animals such as Black Rhino (*Diceros bicornis*), Kudu (*Tragelaphus strepsiceros*), Eland (*T. oryx*) and Giraffe (*Giraffa camelopardalis*) (Smit, 1999). The twigs are also chewed and used by humans as toothbrushes. The heartwood of the plant is termite-resistant and used as fencing posts. The pods, young twigs, leaves and flowers are all highly nutritious (Van Wyk & Van Wyk, 1997).

Figure 5.19: The stem of *Grewia flava* with leaves and fruits

Grewia flava is a shrub or a small dense multi-stemmed tree that grows up to three meter tall that is widely distributed (Mainah, 2001). The encroachment of this species in the study site is encouraged by grazing through the dispersal of its seeds. *Grewia flava* is extremely palatable and thus, largely consumed by livestock. Cattle promote the dispersal of seeds by browsing on the foliage of *Grewia flava* and dispersing seeds into the grassland matrix (Mainah, 2001). The edible shoots and foliage of *Grewia flava* are important fodder for domestic livestock (Watt & Beyer-Brandwijk, 1962). Its berries are taken as snacks and for making of traditional beer (locally called kgadi in Setswana) (Tews *et al.*, 2004).

Figure 5.20: The stem of *Vachellia tortilis* showing leaves and the two types of thorns

Vachellia tortilis is a medium to large canopied tree that grows up to 21 meter high (Corby, 1974). The tree caries leaves that grows to approximately 2.5 cm in length with pinnae of between 4 to 10 pairs each with up to 15 pairs of leaflets. The seeds are produced in pods which are flat and coiled into a spring-like structure and it has a combination of paired hooked thorns (Coates Palgrave, 1990). *Vachellia tortilis* is not only considered one of the main encroachers in the world, it is useful to most communities and to the environment. It is known by its ability to fix Nitrogen in soils. N, K, P, Ca and Na were identified under the canopies of this species on rich volcanic soils. In studies conducted by Ludwig *et al.* (2004), *Vachellia tortilis* did not increase grass productivity, but had a positive effect on the grass quality for herbivores. Ludwig *et al.* (2004) reported that grasses growing under *Vachellia tortilis* have higher nutrient and protein concent due to soil enrichment by *Acacia* trees. It was also stated by Ludwig *et al.* (2004) and Abdallah *et al.* (2008) that grass N content was higher due to soil enrichment. According to Richter *et al.* (2001), fluctuations in seasonal rainfall appear to be the most vital factor governing changes in species composition of the grass component. According to Nzehengwa (2013) there was no significant difference in soil pH between soils under *Senegalia mellifera* and in the open; although research by Hagos & Smit (2005) indicated that the soil pH was lower under *Senegalia mellifera* than in the open while other studies showed higher soil pH under the trees than in the open (Bosch & Van Wyk, 1970; Kennand & Walker, 1973; Palmer *et al.*, 1985).

The timber from the *Vachellia tortilis* tree is used for furniture, wagon wheels, fence posts, cages and pens (Kyalangaliwa *et al.*, 2013). The pods and foliage are used as fodder for desert grazing animals. The bark is often used as a string. The gum from the tree is palatable and is used as a "poor man's" chewing gum. The roots, shoots and pods of the plant are also used by native people for decorations, weapons, tools and medicines. *Vachellia tortilis* is also known as a vital species in the battle to green the desert as it is one of few trees to tolerate very harsh and, arid environments (Kyalangaliwa *et al.*, 2013). In a study conducted by Abd El-Wahab *et al.* (2013), human interference such as cutting of *Acacia* trees for different purposes (fuel wood, charcoal production and construction), overgrazing and uprooting of *Acacia* trees for cultivation or urbanization purposes were observed. Beyene (2013) concluded that there were expressions of bush encroachment in savannas by woody species such as *Vachellia tortilis*, regardless of the harvest of the species by communal people for different purposes. It is harvested to make brushwood fencing and shelters for cooking in communal areas and also used as laths for roofing traditional houses. The plant is uprooted to obtain the roots which are considered to have some medicinal value (Smit, 1999).

Figure 5.21: ***Ziziphus mucronata*** **stem showing leaves and thorns**

Ziziphus mucronata is a small to medium sized tree of 3-10 meter high with a spreading canopy (Brunken *et al*., 2008). The main stem is green and hairy when young. The leaves are simple, alternate, and ovate and vary enormously in size from tree to tree. The leaves turn yellow in autumn and they are borne in impenetrable clusters in leaf axils and they turn from green to yellow. The fruits are smooth, shiny, and reddish-brown when ripe and the pulp of the fruit is dry (Shackleton *et al*., 2005). It is distributed throughout the subSaharan Africa areas which are characterized by the summer rainfall, spreading from South Africa northwards to Ethiopia. It grows in areas dominated by thorny woody species in both temperate and tropical climates. It is also found in advanced habitats such as woodlands (Brunken *et al*., 2008). The leaves and fruits are desired by birds, wild animals and domestic stock. It's inconspicuous, green to yellow flowers produce sufficient nectar and often yields good honey (Venter & Venter, 1996).

The densities of woody species in Disaneng (Figure 5.8) were recorded at "threatening" levels (Table 4.1, Chapter 4). According to Smit & Rethman (1999) an expansion in woody species densities above "critical-density" will lead to the suppression of the herbaceous layer, mainly grasses due to competition for available soil water. Similar results were observed in Disaneng wherein there was an increase in woody species density (especially *Senegalia mellifera*) and the grass layer was virtually completely removed. Woody species locally in Disaneng observational or experimental site exceeded the density in the benchmark (Figure 5.1). Similar studies in the Molopo Area were done by Molatlhegi (2008), Mogodi (2009) and Comole (2014). The locations where the researches were done include areas such as Disaneng, Makgori, Masamane, Loporung, Selosesha and Tshidilamolomo wherein high densities of encroaching species such as *Senegalia mellifera* were observed (Refer to Figure 3.1 (Chapter 3) for the locations of these site).

It was estimated by Coppock (1994) that a shrub cover of 31% expresses density of 1 860 woody plants ha^{-1}. According to Roques *et al.* (2001), a shrub cover of 40% expresses a woody plant density of 2 400 $TE.ha^{-1}$ and has been considered a borderline between nonencroached and encroached conditions while a density of 2 500 TE ha^{-1} has been considered as a highly encroached condition. Gemedo *et al.* (2006) indicated that a shrub cover of 52% is equivalent to 3 014 $TE.ha^{-1}$. It is thus clear that, according to the calculations done by Roques *et al.* (2001), the study area can be considered as highly encroached with the density of 13 547 $ETTE.ha^{-1}$ (Figure 5.8).

Woody species density (especially by *Senegalia mellifera*) was high when compared to other studies conducted in the Disaneng area. It was reported by Molatlhegi (2008) that in Tshidilamolomo, a total woody plant density of 7 980 $ETTE.ha^{-1}$ was noted with *Senegalia mellifera* (5 980 $ETTE.ha^{-1}$) and *Grewia flava* (2 000 $ETTE.ha^{-1}$) the dominant encroaching species. *Senegalia mellifera* and *Grewia flava* were also the dominant woody species in this site in Disaneng (Figure 5.8). Other woody species identified by Molatlhegi (2008) included *Vachellia erioloba, V. hebeclada, Dichrostachys cinerea, Diospyros lycioides, Grewia* *occidentals* and *Terminalia sericea.* None of these species were identified in this research. Molatlhegi (2008) noted a total woody species density of 4 511 ETTE.ha^{-1} in Masamane (Refer to Figure 3.1, Chapter 3), where *Vachellia erioloba* (65 ETTE.ha^{-1}), *Senegalia mellifera* (4 210 ETTE.ha^{-1}), *Dichrostachys cinerea* (75 ETTE.ha^{-1}) and *Grewia flava* (161 ETTE.ha^{-1}) were the most prominent woody species with *Senegalia mellifera* the sole dominant. Molatlhegi (2008) determined a woody plant density of 3 441 ETTE.ha^{-1} in Disaneng with *Senegalia mellifera* (2 314 ETTE.ha^{-1}) and *Dichrostachys cinerea* (909 ETTE.ha^{-1}) noted as the sole dominant woody species. Other woody species identified by Molatlhegi (2008) included *Grewia flava* (141 ETTE.ha^{-1}), *Vachellia erioloba* (27 ETTE.ha^{-1}), *Ehretia rigida* (25 ETTE.ha^{-1}), *Searsia dentata* (17 ETTE.ha^{-1}) and *Ziziphus mucronata* (8 ETTE.ha^{-1}). It is thus clear that the dominance of *Senegalia mellifera* in the study area in Disaneng is in similar to the findings of Molatlhegi (2008).

In other similar studies conducted in the Molopo District in the Disaneng Village, Mogodi (2009) recorded high woody plant densities of 3 527 ETTE.ha^{-1} as compared to the woody density of 407 ETTE.ha^{-1} in the benchmark site (Table 5.1). Similar results were noted in this research. The dominant woody encroachers noted by Mogodi (2009) were *Senegalia mellifera* (1 886 ETTE.ha^{-1}) and *Dichrostachys cinerea* (1 109 ETTE.ha^{-1}). *Dichrostachys cinerea*, however, was not identified in this research. Other woody species identified by Mogodi (2009) included *Senegalia hebeclada* (76 ETTE.ha^{-1}), *V. karroo* (100 ETTE.ha^{-1}), *Ehretia rigida* (91 ETTE.ha^{-1}), *Grewia flava* (178 ETTE.ha^{-1}), *Ziziphus mucronata* (54 ETTE.ha^{-1}) and *Searsia dentata* (33 ETTE.ha^{-1}). The total woody plant density in this research was 13 547 ETTE.ha^{-1} (Figure 5.8) which was almost three times higher than the density recorded by Mogodi (2009) and Molatlhegi (2008). This is an indication of an increase in bush densities in Disaneng. *Vachellia erioloba* was not observed in Disaneng, similar to the findings of Mogodi (2009).

Mogodi (2009) indicated that 0.5 meter high woody encroachers were the most prominent in Disaneng and were recorded at a density of 2 065 ETTE.ha^{-1}. Five meter high woody species were few and found at a density of only 52 ETTE.ha^{-1} (Mogodi, 2009).

Large trees were limited in the survey site. Four meter high (Figure 5.13) and 5 m high woody species (Figure 5.14) were present at densities of 41 ETTE.ha^{-1} and 118 ETTE.ha^{-1}, respectively. Mogodi (2009) identified *Senegalia mellifera* as the sole encroaching species occurring at a density of 7 000 ETTE.ha^{-1} in Selosesha Village (Refer to Figure 3.1, Chapter 3), with *Vachellia erioloba* the only other woody species present. According to the National Department of Agriculture, *Vachellia erioloba* is not considered an encroaching species. *Vachellia erioloba* was only identified in the benchmark (Figure 5.1) and not in the research sites (Figure 5.8) Comole (2014) noted woody plant encroachment in Tshidilamolomo Village (Figure 3.1, Chapter 3) at "threatening" levels with woody plants reaching a density of 3 175 ETTE.ha^{1} . The exotic, *Prosopis velutina*, was the most prominent woody species at a density of 3 089 ETTE.ha^{-1}. *Senegalia hebeclada* and *Senegalia mellifera* were determined at low densities. This was not the case in this research where *Senegalia mellifera* was recorded at high densities (Figure 5.8) and was the most prominent encroacher. *Prosopis velutina* and *Senegalia hebeclada* were not identified in this research.

In Loporung (Figure 3.1, Chapter 3) however, a woody plants density of 3 382 ETTE.ha^{-1} was observed and the vegetation was completely dominated by *Senegalia mellifera* (1 788 ETTE.ha^{-1}) (Comole, 2014). *Vachellia tortilis* was the second most abundant woody encroacher occurring at a density of 641 ETTE.ha^{-1} (Comole, 2014). Other woody species encountered included *Grewia flava* and *Ziziphus mucronata* (Comole, 2014). These results are consistent with the findings of this research where similar woody species were recorded and the suppression of grasses by these woody species was also common.

Comole (2014) recorded a woody plant density of 4 945 ETTE.ha^{-1} in Makgori (Figure 3.1, Chapter 3), where *Senegalia mellifera* was the most abundant. Other species identified included *Vachellia erioloba, V. hebeclada, V. tortilis, Grewia flava* and *Ziziphus mucronata* (Comole, 2014).

Comole (2014) recorded *Senegalia mellifera* and *Grewia flava* in Loporung and Makgori, while in Tshidilamolomo, *Prosopis velutina* was the only species within the 2 and 3 m height classes. Four meter and five meter high *Senegalia mellifera* trees were the most abundant in Loporung and Makgori (Comole, 2014).

It is, therefore, clear that *Senegalia mellifera, Vachellia tortilis, Dichrostachys cinerea Grewia flava* and *Ziziphus mucronata* were the common encroaching species in the Molopo District. *Prosopis velutina* is evident, but virtually restricted to the riparian areas of the Molopo River.

Work by Wiegand *et al.* (2006) indicated that intraspecific competition between trees plays an important role in opening up encroached savannas and creating grassy patches in a cyclical succession. Competition among trees may reduce their densities, sizes and distribution (Smith & Walker, 1983; Jeltsch *et al.*, 2000) but according to Scholes and Archer (1997), Bruno *et al.* (2003) and Miriti (2006), positive intraspecific interactions may lead to aggregation and increased densties.

Large trees (Four metre and five meter high) were noted in low densities. This can be ascribed to intraspecific competition between woody species in the area. According to Smit and Grant (1986), Bonan, (1991) as well as Teaque & Smit, (1992) intraspecific competition between woody species, plays a major role in the savanna structure and productivity. According to Table 4.1 (Chapter 4), the density of these woody species was at a non-threatening state and did not suppress grass growth. However, shrubs and other small trees were observed, growing beneath large trees. According to Joubert (2007), large trees of 4-5 meter support the growth of shrubs and other herbaceous species beneath their canopies. The density of woody species has been reduced in this height class but canopy cover tends to remain high. Figure 5.15, however, shows that the canopy diameter of 4 meter and 6 meter high plants were lower than plants in the other height classes while figure 5.16 shows that 1-2 meter high trees produced the widest canopy diameter in the area. Canopy morphology of woody plants contributes towards their proliferation (Tainton, 1981) because of light interception as well as nutrients that build up directly below the canopies (Belsky & Amundson, 1992). Hagos & Smit (2005) stated that woody species, especially *Acacia* species are canopy intolerant, implying that woody species are generally not growing beneath the canopy of other woody species, but will rather expand in an outward direction. Smit and Swart (1994) stated that tree-on-tree competition is species specific, thus indicating that woody species, especially *Acacia* species will expand and thicken rather than develop beneath currently existing trees.

Ngaruka (2011), however, indicated that there is no significant relationship between size and density of *Senegalia mellifera* in the highland savanna of Namibia and based on the results, the tree density does not affect tree size. Intra- or single species competition among woody plants is induced by an increased demand for limited resources such as water, soil nutrient and sunlight (Ngaruka, 2011).

Rothauge (2004) stated that broad-leaved forbs growing in the nutrient-rich sub canopy habitat, followed by shade tolerant grass species are commonly associated with savanna trees. Lemenih *et al.* (2004) stated that a strong correlation exists between canopy characteristics and understory species richness, density and tree height. It was confirmed by Vadigi & Ward (2013) and Linstadterk *et al.* (2016) that canopy shade is an important driver of ecosystem structure and function. Canopy shade promotes tree sapling survival. According to Vadigi & Ward (2013) sapling survival was not effected by grass competition in a humid savanna in the north-eastern coastal region of Kwazulu-Natal, but shade exerted a negative effect on sampling growth.

Donaldson (1967) stated that the establishment of *Senegalia mellifera* shrubs mainly occurs close to large trees because seed dispersal is generally slow. It has also been implied that woody seedlings recruitment takes place under the canopies of savanna trees because they increase resource availability under their canopies. *Vachellia tortilis* and *Senegalia mellifera* have been shown to increase herbaceous productivity, lower soil temperatures and increase soil fertility under their canopies (Belsky *et al*., 1989; Hagos & Smit, 2005). Elevated nitrogen levels, typically found under *Acacia* trees in savannaa (Smit & Swart, 1994; Hagos & Smit, 2005), give grasses a competitive edge over woody seedlings and forbs (Kraaij & Ward, 2006). *Ziziphus mucronata* tree represented up to 4 meter high trees. The highest density of these trees was, however, for 0.5 meter high trees (Figure 5.9).

Ngaruka (2011) showed that there is a significant relationship between size (height) and crown form of *Senegalia mellifera* trees. Ngaruka (2011) indicated that trees of heights between 0.6 meter and 2.5 meter mostly develop a B crown form (incomplete cone shape) where the maximum canopy diameter occurs at the top of the tree, while trees taller than 3 meter mostly develop the canopy D type, which is a cone shape, where the maximum canopy is not at the top of the tree. Although no effort was made to determine canopy type at different height levels for different trees (see chapter 4), *Senegalia mellifera* trees demonstrated the B crown form (Ngaruka, 2011).

From figures 5.9, 5.10 and 5.11, it is clear that *Senegalia mellifera* revealed the highest densities for 0.5 m, 1 m and 2 m high plants while 1 m high *Vachellia tortilis* trees were most abundant at a density of 955 ETTE.ha^{-1}. Joubert (2014) indicated that *Senegalia mellifera* seedlings are shade tolerant. It was also demonstrated by Hagos (2001) that under adequate soil and water conditions *Senegalia mellifera* seedlings grow better in soil extracted from under tree canopies due to the higher soil nutrient status although the reduced water infiltration below the parent tree is the inhibitor of seed survival beneath the parent tree. Donaldson (1967) stated that *Senegalia mellifera* is known to effectively funnel rainwater to the base of the stem, thus reducing water penetration under other parts of the canopy. No *Vachellia tortilis* plants higher than 3 meter were observed. According to Condit *et al.* (1998), size distribution analysis of *Vachellia tortilis* showed absence of juveniles which is an indication of very low recruitment and high mortality of small and large trees. This absence of juveniles suggested that *Acacia* trees are facing population decline in tropical savannas (Condit *et al.*, 1998).

Figure 5.15 and Figure 5.16 indicate that *Senegalia mellifera* trees had the widest canopies for 2 meter and 3 meter high trees. This is in agreement with Ngakura (2011), indicating that *Senegalia mellifera* trees of medium size (<2 meter) were the most problematic in encroachment, since their canopies are likely to spread outwards or overlap neighboring trees and thus closing all access (Ngakura, 2011). According to Smit (1999), these wide canopies of especially *Senegalia mellifera* had a negative influence upon grassland development beneath the canopies. Joubert (2014) indicated that an increase in plant height and size may also promote a higher diversity of understory plants because of the patch effect and the increase in habitat heterogeneity beneath tree canopies. Ludwig *et al.* (2004) stated that the reduction in soil water moisture in areas where rain mostly falls in small events is caused by rainfall interception by tree canopies which lead to competitive effects beneath tree canopies. Trees, therefore, reduce grass growth by competing with grasses for water, light and nutrients (Scholes & Archer, 1997; Anderson *et al.*, 2001). Joubert *et al.* (2013) indicated that *Senegalia mellifera* seed production showed a positive correlation with rainfall. In the Molopo area, South Africa, Donaldson (1967) showed that *Senegalia mellifera* fruited profusely after a season of above average rain.

Ludwig *et al.* (2004) stated that established trees create sub-habitats that differ from open habitat and exert different influences on the herbaceous layer. The factors that influence grass growth and productivity under trees are related to high nutrient status of soil beneath these trees as compared to the influence of tree canopies (Hagos & Smit, 2005). Smit *et al.* (1999) and Buba (2015) stated that the growth and subsequent size of the individual tree is dependent on the accessibility of abundant resources as well as disturbance factors. The varying disturbances include grazing, herbivory, fire, drought and floods. According to Joubert *et al.* (2012) the virtual exclusion of planned fires on an opportunistic basis is probably the most important controllable variable which has been neglected in semiarid rangeland management and when the juveniles have matured, fire offers a much smaller window of opportunity because of the suppression of grass growth by the woody encroachers.

The presence of *Senegalia mellifera* and *Grewia flava* in the study area in such high densities (Figure 5.8), made the species suppress the herbaceous layer, mostly grasses. According to Richter *et al.* (2001), *Senegalia mellifera* is regarded as a serious threat by most land owners and farmers due to its invasive habitats wherein, when found in high densities, it definitely suppresses the herbaceous layer, mainly grasses. This was also demonstrated in another study conducted in the Kalahari Thornveld, wherein a negative correlation between the density of *Senegalia mellifera* trees and grasses was found (Richter *et al.*, 2001). Pienaar (2006) indicated that rainfall proves to be a very important determinant in the response of woody plants to competition from the herbaceous layer and grazing.

According to Hagos & Smit (2005), the Kalahari Thornveld is water-limited and an increase in woody plant density invariably results in the suppression of herbaceous plants, mainly grasses. An ecosystem is considered water limited when the annual precipitation is typically less than annual potential precipitation, such that the ratio of precipitation to annual potential precipitation ranges from 0.03 to 0.75 and because extreme temporal variability results in extended periods with little to no precipitation (Guswa *et al.*, 2004).

Other woody species observed in our study, included *Vachellia tortilis* at a density of 1 807 ETTE.ha^{-1} and *Ziziphus mucronata* was recorded at a density of 400 ETTE.ha^{-1} (Figure 5.8). *Vachellia tortilis* was not found in the benchmark (Figure 5.1) while *Ziziphus mucronata* was recorded at a low density (Figure 5.1).

Any form of disturbance such as soil erosion and overgrazing, destroys the herbaceous layer and creates an environment, conductive for water and nutrients to be available for woody trees to germinate. An increase in bush densities in the selected sites of Disaneng was influenced by human activities and disturbances such as unsupervised cattle grazing which led to overgrazing. Poor distribution of livestock in the site also led to overgrazing. According to Skarpe (1991), overgrazing is considered to be a primary cause of bush encroachment and soil degradation in many savanna ecosystems all around the world. This was also observed in studies conducted by Smit & Rethman (1998), wherein bush encroachment was considered to be associated with overgrazing by livestock, the elimination of herbivores and exclusion of fires. However, according to O'Connor *et al.* (2014), the main drivers of bush encroachment in southern Africa were direct fire suppression, followed by livestock grazing and indirect fire suppression, an increase in atmospheric carbon dioxide concentration and directional change in temperature and precipitation. Pienaar (2006) stated that the recruitment of *Senegalia mellifera* seedlings was "aggressive" during a bush control program in the Marakele Park in the Limpopo Province.

Trampling by humans and animals, together with fragmentation of footpaths and fire suppression were also observed in the study area. The trampling was not observed in the benchmark and the herbaceous layer was not disturbed. According to Daniel *et al.* (2000), trampling by animals and people on the vegetation, compacts the soil and increases soil bulk density, thereby reducing organic matter and significantly reduce water infiltration capacity. These activities were responsible for the conversion of a grass dominated ecosystem to a dense woody species thicket with no grass cover in the survey site.

According to Smit & Rethman (1998), the grazing of grass cover by livestock has a major impact on the herbaceous layer and can eventually lead to overgrazing. Overgrazing is believed to contribute substantially to the transformation of *Senegalia mellifera* dominated savannas to dense thickets (Smit & Rethman, 1998). When heavy grazing occurs, grasses are removed and topsoil moisture then becomes more readily available to the trees. This allows the trees to encroach the area and provides a good explanation of the absence of grasses in the survey site. According to Quan *et al.* (1994), this was also identified as a concern on commercial rangelands in central Namibia. The area affected by bush encroachment in Namibia is estimated to be 260 000 km^2 and nearly 50% of the commercial rangelands are affected by bush encroachment, mainly by *Senegalia mellifera* (De Klerk, 2003). Joubert (2014) stated that in commercial rangelands of Namibia, overgrazing promoted an increase in woody plants densities by increasing competition between grasses and woody species for water and soil nutrients, thus, allowing woody plants to grow and thicken up.

High grazing intensity restricts primary production and suppresses the dominance of grass species in many savanna ecosystems by removing the vegetal cover, resulting eventually in a decline in soil depth when the exposed soil gets compacted (Branson *et al.*, 1981). This was clearly visible in the study area. According to Skarpe (1990), this situation favours the growth and multiplication of woody species because they have increased access to available soil moisture. Loss of palatable species such as grasses and increase in woody species was observed in the study sites due to heavy grazing by livestock. The area was under communal management (Chapter 3) indicating that farmers are not land owners. Skarpe (1990) reported that *Senegalia mellifera* and *Grewia flava* increased significantly within five years in areas subjected to heavy grazing. According to Joubert (2014) the major reason that bush encroachment is observed in areas that are heavily grazed is that, the fuel for an effective fire is removed and this creates space for woody species such as *Senegalia mellifera* to establish well. *Senegalia mellifera* suppresses grasses to the extent that fire is unlikely since they develop an extensive lateral root system (Smit, 1999). According to Smit (1999), *Senegalia mellifera* roots can extend linearly up to seven times the extent of the canopy-spread and can thus accumulate nutrients from well outside its canopy. Although soil analysis did not form part of this study, sandy soils are evident in the Molopo Area (Mogodi, 2009). The sandy texture of the soil in the study area (>70% sand, Mogodi(2009) further aids in the development of woody species, due to the deeper infiltration of water in sandy soils.

The removal of the herbaceous layer in the study area through heavy grazing by livestock such as cattle and goats reduced the fuel load to allow frequent burning at high intensity. No fire activities were observed in the survey site due to the total suppression of grasses. According to Fatunbi *et al.* (2008), fire plays an important role in maintaining savanna ecosystems. It is, therefore, necessary to establish sustainable burning intervals in such environments. There is a positive feedback between fuel load and fire intensity wherein an increase in grazing capacity reduces grass biomass, making fires less intense and less damaging to trees (Ludwig *et al.*, 2004). These results in the transformation of a grass dominated ecosystem to a woody dominated one. Trollope (1980) showed that there is a negative impact of fire on established tree seedlings while Schultz *et al.* (1955), proved that there is a beneficial effect of burning in reducing above-ground grass biomass, thus reducing competition with tree seedlings which is an indication in the enhancement of woody species encroachment. Donaldson (1967) stated that the chief causes of the failure of woody plants to survive the seedling stage would, therefore, include drought or absence of moisture in the root zone, lack of success in competition with grasses and damage by veld fires. Joubert *et al.* (2008), however, stated that in semiarid savannas with a mean annual rainfall of less than 400 mm, it is more crucial to coincide the fire application with grass seedling establishment.

Kraaij and Ward (2006) stated that, where vegetation has been exposed to overgrazing, livestock such as cattle have a major negative impact on the soil of the rangelands. The soil becomes loose and exposed when the grass layer is removed, as grass roots are good binders of soil, it therefore, gets easily eroded by strong winds and rainfall (Ward, 2004). Soil texture is a determinant of the tree-grass ratio due to its effects on plant growth, soil moisture and nutrient availability in the soil (Wiegand *et al.*, 2007).

Senegalia mellifera seedlings grow slowly, typically only reaching a stem diameter of 5 centimeters after six years and require a good rainfall season for their establishment (Joubert, 2007). According to Donaldson (1967), 0.5 meter high *Senegalia mellifera* trees transpires 12.8 litres of water per day, implying that the total loss of water through transpiration by this species alone in the study area will be approximately 7 000 litres of water daily. According to Skarpe (1990), the increase of shallow rooted *Senegalia mellifera* (Smit, 1999) and *Grewia flava* in the savanna is favoured by an increase in water availability in the surface soil, following overgrazing of the grass layer.

Smit (2004) stated that an increase in woody plant abundance is primarily brought by an increase in the biomass of already established plants and mainly by an increase in density, from the establishment of seedlings. There is widespread concern over the high mortality and low recruitment of *Acacia* trees in arid and hyper arid ecosystems (Moustafa *et al.*, 2000; Noumi *et al.*, 2010) while the absence of juveniles suggests that *Acacia* trees are suffering from a population decline (Condit *et al.*, 1998). According to Abd-Wahab *et al.* (2013) the mortality of *Acacia* species is possibly caused by anthropogenic disturbances. These disturbances are mainly related to human impact including high intensity grazing, cutting of trees and urbanization. It has been suggested that direct competition between grasses and woody seedlings for soil water can suppress the recruitment of woody species (Obot, 1988), although studies conducted by Kraaij and Ward (2006), showed no direct competition. It was illustrated by Kraaij and Ward (2006) that the grass sward in low rainfall savannas is unable to out-compete woody species seedlings. The effect of preemergence fires on recruitment of woody species was also investigated. The negative influence of dense grass on early *Senegalia mellifera* establishment should thus be ascribed to below ground competition for water and nutrients rather than above ground competition for light (Obot, 1988).

According to Skarpe (1990), *Senegalia mellifera* seedlings establish themselves with great difficulty in a dense grass sward and that competition from grasses suppresses the growth of *Senegalia mellifera*. Joubert (2014) stated that the denser the grass sward, the more likely a fire can be sustained that will kill the seedlings while climatic conditions such as the amount of rainfall plays a role in the establishment of *Senegalia mellifera*. It was, therefore, concluded by Joubert (2014) that the competitive effects of the grass sward in arid and semiarid savannas may have been over-emphasized in the past and the effect of grass competition on seedlings, samplings and shrub mortality is currently being tested.

Figure 5.9 indicates that grasses will be completely suppressed and the development of woody seedlings will be highly favored due to high densities of species under 0.5 meter height class. Therefore, the predominance of small, shrubby woody trees leads to the reduction of grass production and palatable species although this may also be a response to grazing. The high density of juveniles in the study site is an indication of the future regeneration trend and intense intra-species competition will reduce woody species diversity

According to Joubert *et al.* (2008), grass accumulates under the open thicket canopy of *Senegalia mellifera*, providing sufficient fuel for fires that may kill senescing trees.

According to Hagos & Smit (2005), the understanding of the potential role of *Senegalia mellifera* in enhancing and maintaining elevated levels of soil nutrients in the Kalahari Thornveld, which consist of nutrient poor sandy soils must be taken into consideration during land management practices. Richter *et al.* (2001) and Hagos & Smit (2005) recommended that *Senegalia mellifera* must be controlled with the use of chemical arboricides as there is a decline in the grass production due to the encroachment of *Senegalia mellifera*. It was concluded by Hagos & Smit (2005) that, in encroached areas in the Kalahari Thornveld, total clearing of all woody species, especially *Senegalia mellifera,* is often the practice. Hagos & Smit (2005), however, proposed selective thinning of *Senegalia mellifera* while conserving some of the beneficial effects of the tree in terms of soil enrichment.

The transformation of certain savannas to bush thickening states is a progressive succession that takes decades. *Senegalia mellifera* requires three years of above average rainfall to establish (Coates Palgrave, 1990, Joubert *et al.*, 2008, Moustakas *et al.*, 2009). Several researchers have reported that recruitment of woody species into savannas is limited by soil nutrients (Kellman, 1979), frequent fires (Bond *et al.*, 2005) and drought (Knoop & Walker, 1985). According to Smit *et al.* (1999), it is generally accepted that bush encroachment is encouraged by long-term overgrazing of the herbaceous layer, the elimination of browser herbivore species and the exclusion of sporadic fires. According to Smit (1999), many *Acacias* (*Vachellia* & *Senegalia*) fail to establish beneath canopies of established individuals, irrespective of its species. *Senegalia mellifera* is among the few exceptions which have been shown to be able to grow under canopies of other *Senegalia mellifera* trees (Tainton, 1981; Hagos & Smit, 2005).

On the contrary, lower herbaceous productivity under tree canopies than in the nearby open was indicated by Dye & Spear (1982). Similar results were recorded by Ludwig *et al.* (2004), wherein water limited the grass productivity under the *Acacia* trees due to lower soil moisture contents and soil water potential than soils away from the tree canopies.

5.3 CONCLUSION

It was indicated by Buba (2015) that different tree species or members of the same tree species with different sizes have different environmental requirements and different environmental conditions creating microhabitats wherein different microenvironments created by trees of different sizes will be occupied by combinations of different species with different degree of abundance. According to Trollope (1980), bush encroachment is a serious veld management problem as was clear in this study wherein the selected study area in Disaneng was highly encroached, especially by *Senegalia mellifera*. The findings of this research are consistent with similar studies where it was found that dense woody trees, resulting from overgrazing, sparse rainfall patterns and other factors have a competitive advantage over grasses. This study confirms findings of Molatlhegi (2008), Mogodi (2009) and Comole (2014) in the Molopo District, where similar findings were recorded, especially regarding the dominance of *Senegalia mellifera*.

The density of encroaching species in the study area was recorded at a density of 13 547 ETTE.ha^{-1} which far exceeded a density of 2 000 TE.ha^{-1} which was indicated by Moore & Odendaal (1987) that as the upper limit to totally suppress grass growth. *Senegalia mellifera, Grewia flava, Vachellia tortilis* and *Ziziphus mucronata* were identified as the encroaching species with *Senegalia mellifera* and *Grewia flava* being the most abundant. *Vachellia erioloba* was only noted in the benchmark while *Vachellia tortilis* was absent in the benchmark. The study area was highly encroached by *Senegalia mellifera* as this was found to be dominant in all height classes, particularly height classes 0.5 m, 1 m and 2 m and was recorded at tree densities of more than 2 000 $TE.ha^{-1}$. The distribution of *Senegalia mellifera* in the study sites resulted in the suppression of grasses. Species such as *Ziziphus mucronata* were recorded at low densities in all height classes. It is clear that in Disaneng, at the evaluation sites, both woody species cover and density have crossed the threshold level and entered the encroached condition.

The good state of savannas is of economic importance for both farmers and non-farmers. It is of importance for the Department of Agriculture to intervene in the control of unwanted woody encroachers by providing the communities concerned with chemical arboricides and assisting them with the control process. Tainton (1999) stressed that a good grass sward is essential to implement fire control. In the survey sites of this research, fire cannot be used as a management tool in controlling undesirable woody species, since the grasses are totally suppressed. Therefore, mechanical methods can be applied in the survey site. These methods imply the use of machinery or implements and even manual methods such as stem burning, hacking or digging out of trees. It can thus be concluded that restoration of the selected sites of Disaneng must be applied since local people depend on the area for grazing their cattle. From this study it is clear that, in cases of increased bush densities, there are no other alternatives but to reduce the densities of encroachers, especially *Senegalia mellifera*. The distribution of *Senegalia mellifera* needs to be monitored and chemical control methods are advised. Further research in this regard is necessary and local communities must be involved in the decision making process and should be educated too. However, for this to take place, the challenges of the communal system in this area have to be addressed first, because the farmers often are not owners of the land.

CHAPTER 6

REMOTE SENSING

6.1 INTRODUCTION

Bush encroachment is classified as a natural phenomenon resulting in the conversion of a grass dominated ecosystem to a tree dominated ecosystem through of plant succession (Archer *et al.*, 1988). According to Brown & Archer (1999) this phenomenon has been widely recognized and has implications for livestock production systems, wildlife habitats and biogeochemistry. Bush encroachment is one of the broadest changes in land-cover in semi-arid rangelands and a predicament worldwide, rapidly decreasing the productivity of the rangeland (Ward, 2005). Despite the severity of these consequences the complete and detailed valuation of bush encroached areas is still limited.

A number of factors can cause bush encroachment, however rainfall and soil type are the primary causes while high grazing intensity and fire suppression are secondary causes (Kraaij & Ward, 2006). Increasing atmospheric carbon dioxide concentration has become known in recent decades as key global change impact for clarifying bush encroachment across the world (O'Connor *et al.*, 2014). There is strong evidence in the South African context for the effect of atmospheric carbon dioxide concentration on increased woody growth (O'Connor *et al.*, 2014). A number of southern African researchers have concluded that landscape-level bush encroachment, in which the potential role of other factors has been investigated, can be attributed to increasing atmospheric [CO_2] (Wigley *et al.*, 2010; Buitenwerf *et al.*, 2012; Ward & Russell, 2014; Ward *et al.*, 2014). The role of atmospheric [CO_2] has huge impact in a management context.

Classifying and mapping vegetation is a vital technical responsibility for managing natural resources as vegetation provides a basis for all living things and plays an essential role in affecting global climate change, such as influencing terrestrial CO_2 (Xiao *et al.*, 2004). Vegetation mapping also presents valuable information for understanding the natural and man-made environments through quantifying vegetation cover from local to global scales at a given time or over continuous period. It is critical to obtain current states of vegetation cover in order to initiate vegetation protection and restoration programs (Egbert *et al.*, 2002; He *et al.*, 2005).

One way to assess bush encroachment is the infield assessment (Field surveys) of bush densities in savannas (Chapter 4). However, this approach is time consuming and expensive due to the large amount of work force necessary to execute it. A complementary approach, especially on large scales of study, is remote sensing of vegetation cover. The technology of remote sensing offers a practical and economical means to study vegetation cover changes, especially over large areas (Langley *et al.*, 2001; Nordberg & Evetson, 2003). Remote sensing extends possible data achieved from present time to over several decades back because of the potential capacity for systematic observations at various scales. Many researchers (e.g. Mampholo, 2006; Munyati *et al.*, 2011; Comole, 2014; Sinthumule & Munyati, 2014) applied the remote sensing technique to delineate vegetation cover from local scale to global scale. The approach has been applied successfully for mapping of bush encroachment (Wagenseil, 2008).

The aim of this study was to monitor bush encroachment in Disaneng. In this regard, the utility of remote sensing in the assessment of bush encroachment was applied, as demonstrated by a number of authors (e.g. Hudak & Wessman, 1998; O'Connor & Crow, 1999; Hudak & Wessman, 2001; Laliberte *et al.*, 2004; Martin & Asner, 2005; Munyati *et al.*, 2011).

6.2 LITERATURE REVIEW

Hill (2000) defined remote sensing as the science of collecting information about objects without coming into physical contact with them. This definition applies to recording of electromagnetic radiation by aircraft or satellite-born sensors (Richards & Jia, 1999, De Lange, 2006). Distinguishing between different objects on the images relies on difference in their spectral reflectance behaviour.

Sensors record electromagnetic radiation being reflected by objects on the earth's surface, (e.g. plants, buildings, water bodies), or in the atmosphere, (e.g. clouds). These objects show characteristic patterns of reflectance across the wavelength spectrum allowing for determination of specific types of objects (De Lange, 2006). These patterns, referred to as spectral signatures or spectral fingerprints (De Lange, 2006), differ among specific types of objects, e.g. vegetation, water and soil, but also between similar objects, e.g. different kinds of vegetation and soils (De Lange, 2006).

The spectral signature of a pixel of a satellite image (r_{pixel}) is a function of various factors (Asner, 2004):

$r_{pixel} = f$(geometry, tissue optics, canopy structure, landscape structure, soil optics) where

'geometry' refers to the sun and sensor zenith and azimuth angles, "tissue optics" describes the different reflectance and transmittance characteristics of living (photosynthetic) and senescing (non-photosynthetic) vegetation due to chemical characteristics and structure,

'canopy structure' characterizes the distribution and abundance of biomass tissue, 'landscape structure' refers to shadowing of trees and "soil optics" describes physical and chemical differences of soil types (Asner, 2004).

6.2.1 Importance of remote sensing in determining bush encroachment

Developing methods to monitor the extent of bush encroachment is potentially beneficial for the management of savanna rangelands. Satellite imagery has been used from its earliest times for the preparation of base maps for rangeland inventory. It has been introduced as an important tool for understanding and monitoring various components of rangeland function and health (Palmer & Fortescue, 2004). The tool is able to detect change and to monitor the rangeland conditions by describing deviations from desired state of rangeland health. It is a valuable tool in establishing desired states and can serve as benchmark from which deviations can be evaluated.

The desired state can be defined using a number of critical indices, including production, structure and biodiversity. Remote sensing will also contribute to all the preparation of all these indices. Remote sensing provides an opportunity to monitor and understand the spatial patterns of vegetation and to inform the understanding of biotic and abiotic processes related to those patterns (McGlynn & Okin, 2006). The benefits of using remote sensing in detecting bush encroachment, is that the high spatial resolution of remote sensing enables direct imaging of plant individuals (McGlynn & Okin, 2006).

6.2.2 Remote sensing image classification

One of the most important uses of remote sensing is the production of land-cover maps and this can be done through image classification (Guo *et al.*, 2008). Image classification is a process that converts satellite data into information based on pixel values within the image.

Image classification had made great progress over the past decades in the following areas:

1. Producing land cover map at regional and global scale,

2. Classification algorithms,

3. Use of multiple remote-sensing features including spectral, spatial, multitemporal and multisensory information and;

4. Incorporation of ancillary data into classification procedures, including data such as topography, soil, road and census data.

Guo *et al.* (2008) stressed that accuracy assessment is an integral part in image classification. The success of image classification in remote sensing depends on many factors, such as the availability of high quality remotely sensed imagery and ancillary data, the designer of a proper classification procedure and the researcher's skills and experience (Guo *et al.*, 2008).

According to Aplin & Atkinson (2004), image classification is separated into supervised classification and unsupervised classification. Unsupervised image classification is the process in which image interpreting software separates a large number of unknown pixels in an image based on their reflectance values into classes with no direction from the analyst (Tou & Gonzalez, 1974). The advantages of unsupervised classification are: human error is minimized; no extensive prior knowledge of the region is required and unique classes are recognized as distinct units. Unsupervised classification has the following disadvantages: it identifies spectrally homogeneous classes within the data that do not necessarily correspond to the information categories that are of interest to the analyst.

The supervised technique involves the need for prior knowledge of the ground cover of the study site and has advantages over the unsupervised one. In supervised approach, useful information categories are distinct first and then their spectral separability is examined while in the unsupervised approach, the computer determines spectrally separable classes and then defines their information value (Lillesand & Keifer, 1994).

6.3 RESULTS

Figures 6.1 to 6.3 show the results of the multi-temporal classification of the images. The change in area of the savanna cover classes into which the multi-temporal images were classified is indicated in Table 6.1. The total area under the woody vegetation class indicates an increase in woody species over time. On the 21 September 2000 SPOT 4 image, the woody vegetation covered only a small area amounting to 14.64 ha, and then it increased to 40.64 ha by the 03 May 2004 SPOT 4 image and was highest on the 11 May 2009 SPOT 5 image, covering an area of 56.77 ha.

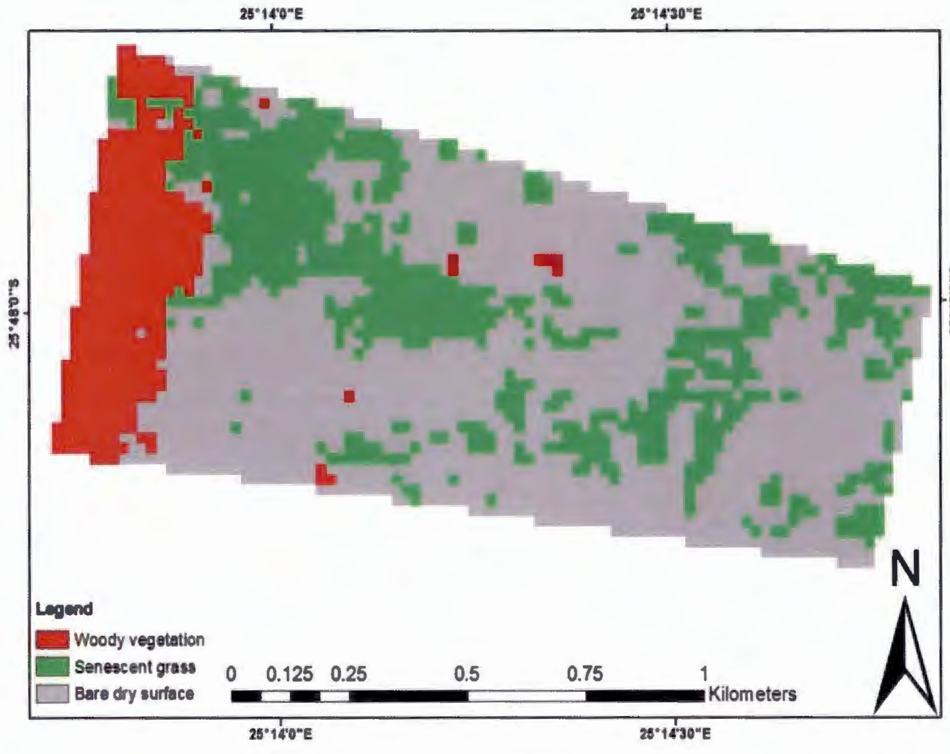


Figure 6.1: Supervised classification of the 21 September 2000 SPOT image of the study area

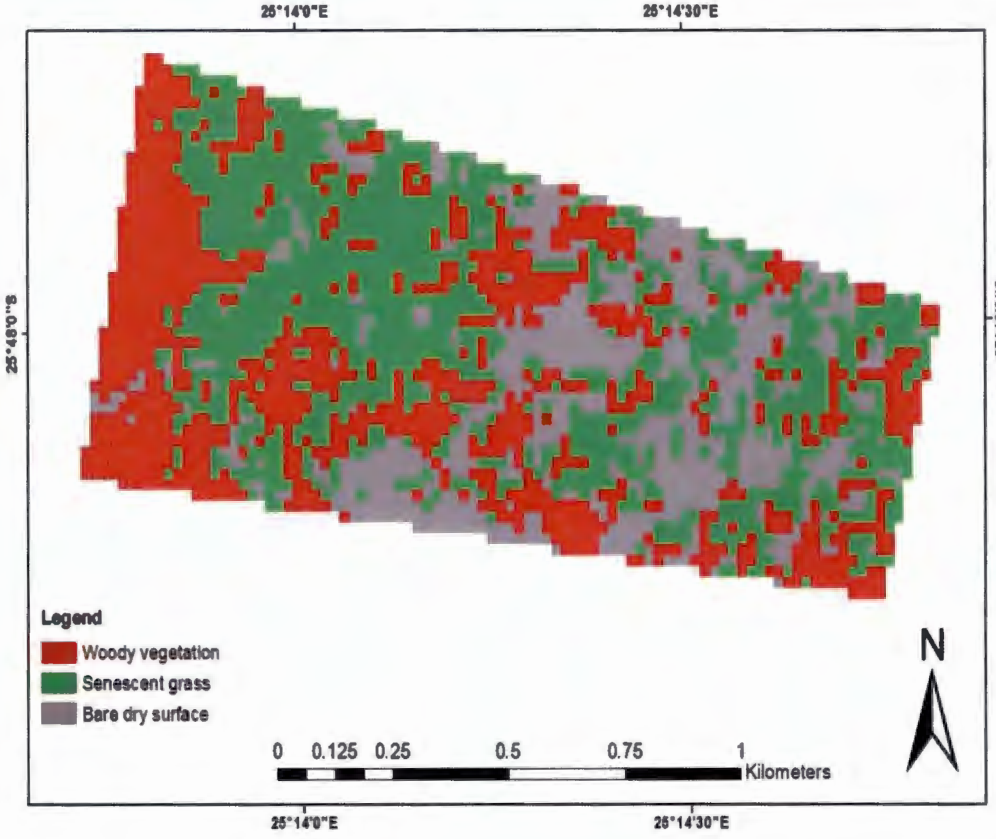


Figure 6.2: Supervised classification of the 03 May 2004 SPOT image of the study area

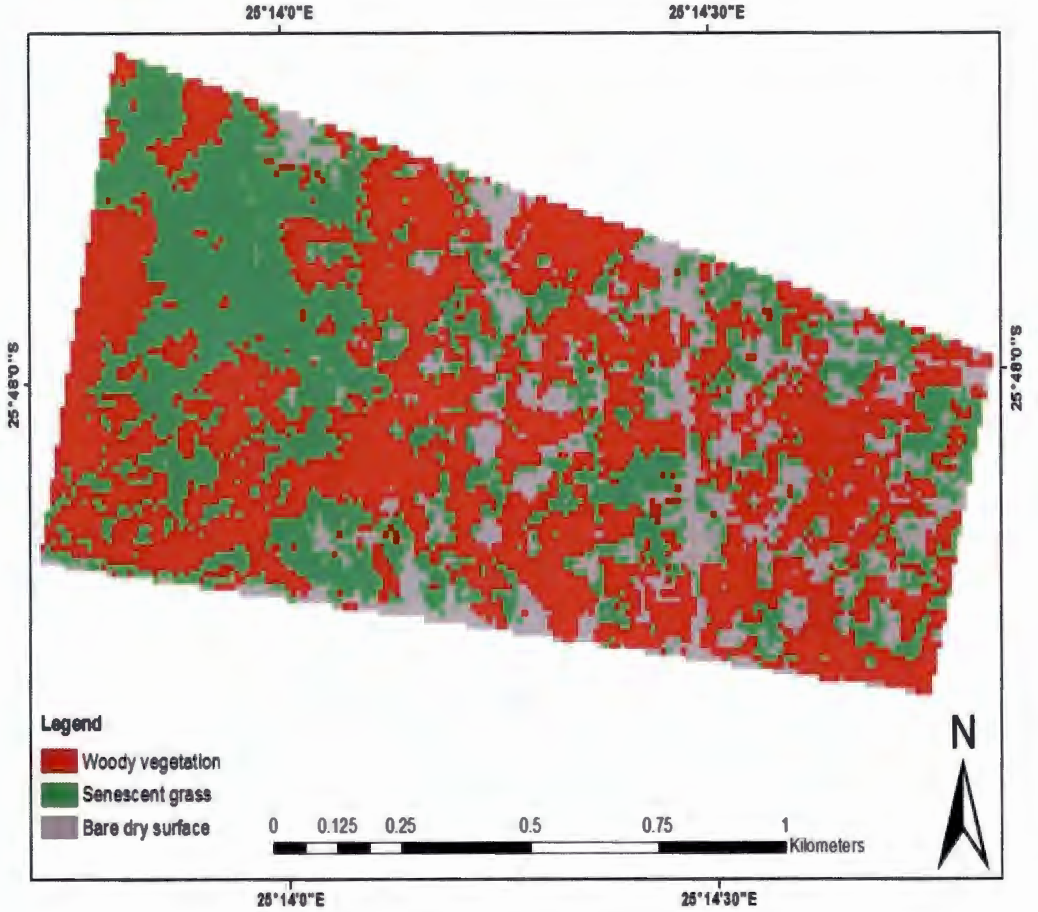


Figure 6.3: Supervised classification of the 11 May 2009 SPOT image of the study area

The change in area of the rangeland cover classes into which the multi-temporal images were classified is shown in Table 6.1.

Table 6.1: Change in area of vegetation cover, senescent grass and dry bare surface for the survey site, derived from image processing. WV – Woody Vegetation, SG = Senescent Grass, DB – Dry Bare area, TA – Total Area.

Class	Disaneng				
	Area (ha)			Percent area change	
	21 September 2000	03 May 2004	11 May 2009	2000-2004	2004-2009
Woody vegetation	14.64	40.64	56.77	177.6%	39.69%
Senescent grass	32.48	47.36	40.05	45.84%	-15.43%
Dry bare surface	71.44	30.56	19.85	-55.22%	-35.05%
Total area	118.56	118.56	116.67		

According to Table 6.1 there was a continuous decrease in the bare soil class and senescent grass which is an indication of the establishment and spread of woody species. All three images detected change in the survey site, which is in agreement with the field data collected in 2013 and 2014 (Figure 5.7). However, the 2013 and 2014 images were not available; the selected images were mainly on the basis of availability.

6.4 DISCUSSION

Mapping vegetation through remotely sensed images involves various considerations, processes and techniques. The use of both field survey of woody species density and remote sensing on image classification methods is important. The results acquired in this study demonstrate the usefulness of remote sensing in detecting, quantifying and monitoring bush encroachment in savanna rangelands. These results will therefore, be useful in the management of savannas. Munyati *et al.* (2011) stated that developing methods to quantify rates and monitor extent of bush encroachment is potentially beneficial for the management of rangelands in savanna rangelands globally. Hence, multitemporal remote sensing offers possibilities in monitoring the extent of bush encroachment due to historic imagery.

Differences in season aspects of the images used in this research had no effect on the results. The images dated (21 September 2000, 03 May 2004 and 11 May 2009) were suitable for detecting woody cover because the grasses were senescence (dry) and the trees were largely still in leaf. Rain seasons (November-March) preceeding the time for the respective images were of little influence in the detection and monitoring bush encroachment because the woody species (encroaching species) identified in the field survey (*Senegalia*, *Vachellia* and *Grewia*) are aridity tolerant.

Careful selection of image dates is an important component of the protocol for detecting and monitoring change in woody vegetation cover on African savannas. This was done to eliminate grasses from the analysis. The phenology of grasses responds to rainfall events since during the rainy season (November-March) the grasses are green as the trees, while during the dry season (April-October) the grasses are dry.

The variable quadrant method which was used avoids visual estimation on the density of woody species in the survey site. The method has the disadvantage of being time consuming, physically demanding and expensive and is, therefore, suitable when there are no time frames, and there are enough funds and resources. The results acquired from this method showed high densities of woody species in the survey site (Figure 5.8, Chapter 5). Although the causes of the increase in woody species densities are not well understood (Ward, 2005), O'Connor *et al.* (2014) stated that an increase in carbon dioxide concentration [CO2]in the atmosphere is the main primary factor leading to bush encroachment. Factors such as humans activities and mismanagement of the area were observed in the survey site. Unsupervised grazing of livestock in the study area led to overgrazing, suppressing the grass layer which led to an increase in woody species.

A possible cause of increase in bush densities in the survey site is mismanagement such as over-grazing by cattle. It was observed in the survey site that animals were grazing more than what the area was providing, wherein grasses were not given the opportunity to reach the maturity stage. This has resulted in high grazing pressure which led to the suppression of grasses. An ideal pasture management practice in communal rangelands is rotation grazing methods. The rotational grazing methods were not applied in the study area. The trampling of the savannas by both animals and people has resulted in the suppression of grasses and creating open spaces for woody species to recruit and encroach in the savannas.

In this case, fire regimes are not considered as a savanna management tool due to the suppression of grasses. According to Skarpe (1992) fire is regarded as a norm in shaping savannas but in the survey site, the suppression of fire led to the increase in bush densities. The suppression of fire seems to explain the slightly higher structural attributes for the encroaching species.

6.5 CONCLUSION

Land degradation is affecting large areas of savannas in South Africa and bush encroachment has been identified as one of the causes. It is, therefore recommended that woody cover field quantification and multispectral woody cover image classification methods should be utilized for this research in similar studies of savannas since the methods can contribute to the monitoring of bush encroachment. A number of studies (Hudak & Wessman, 1998, 2001; McGlynn & Okin, 2006; Pringle *et al.*, 2009; Oldeland *et al.*, 2010(a)) have shown that spatial resolution is a determinant factor in the successful use of remote sensing to monitor bush encroachment. High spatial resolution is vital in detecting bush encroachment because of the small sizes of encroaching shrub canopies in savannas.

According to Wigley *et al.* (2010) not all woody species are a manifestation of bush encroachment given the diverse drivers of change in savanna woody cover. However, in areas that are known to have been open savanna in the past, the occurrence of woody vegetation would be a manifestation of encroachment. The field survey indicated a large amount of encroaching species in the survey site (Figure 5.8) wherein encroaching species such as *Senegalia mellifera* was recorded as the most prominent species (Figure 5.8). This is an indication that the survey site was once an open savanna and an increase in woody species is a manifestation of bush encroachment since woody vegetation showed an increase in the area (Figures 6.1, 6.2, 6.3 and Table 6.1) over time. Approximately 60% of the original area of Disaneng (Figure 3.1, Chapter 3) has been transformed, thus reducing biodiversity. Encroachment of woody plants is continuously threatening the livestock production and challenging the sustainability of the local pastoral system.

Bush encroachment is a problem (Ward, 2005) as it affects the agricultural productivity and biodiversity of 10-20 million hectares of South Africa, threatening the sustainability of livestock production system and humans well-being. It occurs in many arid regions where fuel loads are insufficient for fires to be an important causal factor (Roques *et al.*, 2001). It is therefore vital to properly manage and conserve such encroached rangelands since the quality of rangelands is of economic importance and which makes their degradation of a concern.

CHAPTER 7

CONCLUSIONS AND RECOMMENDATIONS

7.1 CONCLUSIONS

Bush encroachment is not a new phenomenon and it has affected the biodiversity of savanna ecosystems in the South Africa and worldwide. Bush encroachment is the increase in woody species density that results in impenetrable thickets. The highest densities, particularly of *Senegalia mellifera*, were noted in Disaneng Village. The density exceeded 2 500 $ETTE.ha^{-1}$ which is an indication of considerable bush encroachment in the study site and this will have a negative impact on the herbaceous production and livestock production.

Senegalia mellifera was found in high densities in the survey site and its presence in such densities is an indication of overgrazing. According to the Department of Agriculture, Conservation, Environmental and Tourism (2002), the natural vegetation of the central, southern and south eastern regions of the North West Province (NWP) has been transformed by overgrazing. Overgrazing was evident in the selected sites of Disaneng and contributed towards the establishment of woody encroachers such as *Senegalia mellifera*, although this species is not only associated with negative impacts, it is a nitrogen fixer which enhances the soil nutrient content. However, an increase in atmospheric carbon dioxide concentration is believed to be a major or primary driver of woody species encroachment. An assessment from ecological and anthropogenic perspective is vital in the study area. The community of Disaneng benefit from their environment on resources such as food and livelihoods, it is therefore vital to be concerned about the area's conditions. To minimize the effects of bush encroachment in the selected sites of Disaneng, a combination of control options should be integrated into the grazing management.

It is therefore concluded that the use of remote sensing technique provide an accurate analysis of vegetation growth and the extent of bush encroachment. There is an indication of an increase in the rate of bush encroachment in the selected sites of Disaneng as depicted by remote sensing technique using SPOT images applied in this research.

7.2 RECOMMENDATIONS

- In making any decisions on the most appropriate procedures to use two conditions must be met: Ecological responsibility and economic viability.
- Research gap regarding the usage of encroaching species for economic purposes such as selling of wood and fruit production needs to be conducted so that local people can be able to gain direct useful advantage from these actions.
- The selected sites of Disaneng must be restored and managed since the community relies on the sites for stock farming.
- It is thus recommended that the control options should incorporate chemical, mechanical and biological controls together with livestock management.
- The Department of Agriculture, Conservation, Environmental and Tourism (DACET) must educate the villagers regarding bush encroachment through tribal meetings wherein strategies to combat bush encroachment should be provided to the villagers.
- Farmers or livestock owners must be trained on using good grazing practices to prevent overgrazing.
- Villagers and farmers must be educated about protected plant species such as *Vachellia erioloba*.
- Fencing of areas that need to be restored must be considered.
- The initial control method should target trees of one meter height class because they prohibit movement and complicate management. These species were recorded at high densities in the selected sites of Disaneng.

- The control programme should form part of the selected sites of Disaneng management with great emphases on after care treatment.

The findings of this study will be presented to the Department of Agriculture, Conservation, Environmental and Tourism (DACET) so that they can be able to assist in distributing the information regarding bush encroachment in Disaneng and combating the problem, thus, adding more knowledge to the villagers regarding their surrounding environment. The way forward: A research will be conducted regarding restoration of the selected sites of Disaneng.

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