To determine the extent of bush encroachment with focus on *Prosopis* species on selected farms in the Vryburg district of North West Province

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DEDICATION

The study is dedicated to my sons Neo and Katlego who gives me extra strength for every endeavour that I pursue, also to his lovely mother Cecilia Mathilda who stood by me through thick and thin of studying this Masters of Environmental Science, undertaking of the research and the whole part of my life.

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ABSTRACT

The study was undertaken to determine the woody component with the focus on *Prosopis* species at selected farms in the of Vryburg district, Naledi municipality. The study also tested differences of plant density, size (structure) and bush equivalents between previously controlled and non-controlled plots. The study was conducted on the farms of Orsets, Trent 1, Trent 2, Mimosa, Eensaam, Mooidraai and Werda situated South East of Vryburg in Veld type A16 of Kalahari thornveld and shrub bushveld. A belt transect of 400 m^2 was used to carry out the vegetation survey. The woody component was recorded according to species type; height class and bush equivalents were calculated through pre-determined factors for each height class. The results showed that no major differences of bush equivalent exist in previously controlled and non-controlled plots. Species such as *Prosopis glandulosa, Acacia mellifera, Ziziphus mucronata, Grewia flava, Diospyros lyciodes* and *Ehretia rigida* were identified. The sampled sites are highly invaded by *Prosopis glandulosa* with lower abundances of other indigenous species, such as *Acacia karroo, Ziziphus mucronata, Diospyros lyciodes, Ehretia rigida and Grewia flava.* It is recommended that satellite data should always be verified and complemented by field survey in order to have accurate bush density and species type. The control of *Prosopis glandulosa* should integrate various options to have long-term good results. It is concluded that the study areas are highly encroached with *Prosopis glandulosa.* The MODIS satellite remote sensing data are unreliable to study sites on limited scale owing to its reflection of large scale. The use of SPOT images and landsat data provide fair analysis of vegetation growth and extent of bush density. The undertaking of aftercare is mandatory to attain the required control of *Prosopis glandulosa* in the study sites.

OPSOMMING

Die studie was gedoen om voorkoms van houtagtige spesies op geselekteerde plase in die Vryburg distrik (Naledi munisipaliteit) te bepaal, met die fokus op *Prosopis.* Die studie het ook verskille in plantdigtheid, grootte en bos-ekwivalente met betrekking tot vorige beheerde en onbeheerde persele getoets. Die ondersoek was gedoen op die plase Orsets, Trent 1, Trent 2, Mimosa Eensaam Mooidraai en Werda , suidoos van Vryburg dorp, in die A16 veldtipe van die Kalahari doring- en bossieveld. 'n Strook gedeelte van 400 vk meter was gebruik om die plantegroei opname to doen. Die houtagtige komponent was opgeteken ten opsigte van spesietipe, hoogteklassifikasie en bos-ekwivalente was bereken deur middel van voorafopgestelde faktore vir elke hoogteklas. Die resultate het getoon dat geen belangrike verskille van bosekwivalent bestaan in vorige beheerde en onbeheerde persele nie. Spesies soos *Prosopis gladulosa, Acacia mellifera,Ziziphus mucronata, Grewia flava, Diospyros lyciodes en Ehretia rigida* was geidentifiseer. Die monsterpersele was grotendeels ingeneem deur *Prosopis glandulosa* met minder digthede van ander inheemse spesies soos *Acacia mellifera, Ziziphus mucronata, Diospyros lyciodes Ehretia rigida en Grewia flava.* Die aanbeveling is dat sateliet data altyd met veld observasies geverifieer en aangevul moet word om akkurate bosdigtheid en spesietipes vas te stel.

Tydens die beheer van *Prosopis glandulosa* moet verskillende opsies geintegreer word om goeie langtermyn resultate te verseker. Daar is tot die gevolgtrekking gekom dat die studiearea swaar geinfesteer is met *Prosopis glandulosa.* Die data van die MODIS sateliet, se afstandswaarneming onbetroubaar is vir 'n studiegebied met 'n beperkte skaal vanweë groot skaal waarop dit inligting reflekteer. Die gebruik van SPOT satelietbeelde en Landsat data voorsien 'n redelike ontleding van plantegroei en die omvang van bosdigtheid. Die verbintenis tot nasorg is gebiedend noodsaaklik om die nodige beheer van *Prosopis glandulosa* in die studiearea te verkry.

CHAPTER 1

INTRODUCTION

1.1 Problem of bush encroachment

1.1.1 Background of bush encroachment

Bush encroachment is a process whereby the density of woody plants e.g. trees and shrubs, increase in the area (Tainton, 1999). The process is deemed to be a result of demand above ground or below ground competitive capacity of grasses subjected to grazing (Brown & Archer, 1999). According to Ward (2005), bush encroachment is the suppression of palatable grasses and herbs by encroaching woody species often unpalatable to domestic livestock.

Invasion which contributes to bush encroachment refers to colonisation of species in new or pre-existing ecosystems and dominates otherwise intact pre-existing native ecosystems (Pyke & Knick, 2003a). Bush encroachment affects the agricultural productivity and biodiversity of 10 to 20 million hectares in South Africa (Ward, 2005). Accumulating evidence indicates that in the past 50 years, savannas throughout the world are being altered by this phenomenon, known as bush encroachment (Ward, 2005).

The reduced agricultural productivity occurs because of the low value of thorn trees to livestock, while reduced biodiversity occurs because a multi-species grass sward is replaced with a single tree species (Ward, 2005). This may be invasion of woody plants in areas where these did not occur previously or an increase and thickening of certain woody plants already in the natural area (Tainton, 1999). Grazing practices in interaction with rainfall variability, determine the structure and functioning of these savannas, resulting in variable production and quality (Archer, 2003). The prolonged overgrazing as influenced by stocking rate, leads to changes in the botanical composition of the veld (Tainton *et al.,* 1999). The plant species compositions are influenced by soil properties such as nutrient status, pH, salinity and texture. The major factor determining the spatial distribution and productivity of savanna is soil moisture balance (Teague & Smit, 1992).

Bush encroachment is one of the most serious results of imbalances in savanna ecosystems (Richter & Meyer, 2001). According to Richter & Meyer, (2001) bush densities in excess of 2500 tree equivalent, depending on the species and the affected area, can suppress grass production by as much as 82% in years of average rainfall. Trees are able to make more effective use of the deep water-table than the grasses with a much shallow root system (Smit & Rethman, 1999). There are negative grass-tree interactions between woody and herbaceous plants that involve available soil-water as the primary determinant of dry matter production (Teaque & Smit, 1992).

However, there is also a positive effect on grass growth as a result of grass-tree interactions, if the larger trees occur in the established open habitat. These trees create sub-habitats, which differ from the open habitat, which can exert different influences on the herbaceous layer (Smit *et al.,* 1999). The advantages of trees are that they act as a biological agent thereby creating islands that differ from those in the open habitat (Hagos & Smit, 2005). The factors that influences grass growth and productivity under trees are related to relatively high nutrient status of soil beneath trees as compared to influence of tree canopies (Hagos & Smit, 2005). The growth and subsequent size of the individual tree is depended on the accessibility of abundant resources, such as water and nutrients, as well as some disturbance factors (Smit *et al.,* 1999).

Bush encroachment accentuates the effects of drought during below average rainfall years, while it also causes pseudo-drought under normal rainfall conditions (Archer, 2003). The bush encroachment phenomenon is usually characterised by fodder shortages, ranging from extremely bad to more moderate, depending on the long or short-term rainfall (Richter & Meyer, 2001). This situation leads to veld to be increasingly drought sensitive.

1.1.2 Causes of bush encroachment

According to regulation 16 of the Conservation of Agricultural Resource Act, Act No. 43 of 1983 there are 44 species regarded as indicators of bush encroachment in South Africa. The thorny and non-thorny woody species of small-leafed and other leguminous species found throughout most of southern Africa are regarded to be problematic bush encroaching species. Some of the encroaching species are indigenous trees such as *Acacia mellifera* subsp *detinens, Acacia nilotica, Dichrostachys cinerea, Terminalia prunioides* and *Terminalia sericea* (Strohbach, 1998). The alien invasive species causing bush encroachment are species such as *Prosopis* species, *Acacia mearnsii,* three hakea species such as *Hakea drupacea, Hakea gibbosa* and *Hakea sericea* (Henderson, 2001).

These species are referred to as alien because they have been introduced through human activities to an area where they did not occur previously (Barac, 2003). The invasive alien species, particularly tree species such as *Prosopis*, have increased water usage compared to native vegetation (Versveld *et al.,*1998). Most of these alien invasive species produce copious numbers of seeds, are wind or bird dispersed or have highly efficient means of vegetative reproduction, which leads to aggressive encroachment (Brooks *et al.,* 2004).

1.1.2.1 Rangeland management

- **Livestock**

The conventional wisdom that bush encroachment only occurs after grasses are removed by overgrazing is somewhat simplistic and may not be a general explanation of the phenomenon (Ward, 2005). The use of the above mentioned simplistic model is exposed because recruitment in *Prosopis glandulosa*, a bush encroaching tree in North America and other parts of the world, is unrelated to herbaceous biomass or density (Brown & Archer, 1999). It indicates that release from competition with grasses is not required for mass tree recruitment to occur (Brown & Archer, 1999). The overall rangeland management and utilization by livestock has unintended impact contributing to bush encroachment.

The use of rooting niche separation used in justifying initiation of bush encroachment owing to overgrazing cannot be a general mechanism explaining tree-grass coexistence (Ward, 2005). The reasons for not accepting root niche separation in initiation of bush encroachment is that young trees use the same subsurface soil layer as grasses in the sensitive early stages of growth (Ward, 2005). Although Ward (2005) indicates that bush encroachment is not only just caused by overgrazing, Moleele (2005) stresses that the increase in density and cover of woody plant species is coincidental with the introduction of cattle in Southern Africa. The empirical result of a study undertaken by Moleele (2005) confirms that cattle density is responsible for bush encroachment. Also Drewa *et al.,* (2002) support this idea namely that cattle have been directly responsible for increased abundance and expanded distribution of honey mesquite (common name for *Prosopis* spp) through consumption and dissemination of seed. A favourable microenvironment provided by cattle dung may facilitate germination of *Prosopis* plant (Drewa *et al.,* 2002). According to Low & Rebelo (1996) the shrub-tree element comes to dominate the vegetation in areas, which are being overgrazed. According to Ward (2005) the factors causing bush encroaching are poorly understood. This viewpoint is supported by an indication of Smit (2004) that bush encroachment is not understood at fundamental level by both scientists and landowners who have to deal with the problem at practical level. In trying to understand the bush encroachment phenomenon, it is also important to understand interactions between different levels of soil nitrogen and the population genetic structures of the trees (Smit, 2004). This viewpoint will help in determination to make predictions about which areas are most susceptible to bush encroachment (Brown & Archer, 1999).

- **Increase of bush encroachment as a result of impact on biodiversity**

The reasons for an increase in the density of woody plants in any vegetation type are diverse and complex (Smit *et al.,* 1999). There are also secondary factors that promote bush encroachment, such as the decrease in endemic browsers owing to the replacement with cattle (Richter & Meyer, 2001). This entails the replacement of multi-level herbivores with primarily grazers. The eradication of a once widespread native herbivore, such as *Cynomys ludovicianus* (black-tailed prairie dog) coincides with bush encroachment. The field experiments indicate that prairie dog, herbivores and granivores associated with their colonies are likely to maintain the savanna by preventing woody species such as *Prosopis glandulosa* from establishing or attaining dominance. The prevention of *Prosopis glandulosa* encroachment by these small mammals is due to their seed and pod removal abilities (Weltzin *et al.,* 1998 a). The impact owing to extinction of some species indicates how strongly biodiversity in ecosystem has on stability, resilience and domain of attraction, thus linking to bush encroachment. The impact on soil biodiversity and fertility may also be assumed to be positive, particularly in comparison with bare land. Since vegetation cover of *Prosopis* reduces erosion by wind and water, stabilizes dunes and increases soil fertility through nitrogen fixation and litter fall (Geesing *et al.,* 2005).

- **Fire**

Fire appears to promote invasive species in a number of arid and semi-arid ecosystems (Pyke & Knick, 2003b). The change in the fire regime is mostly the result of elimination of burning as veld management practices. Also the reduction in fuel material by either over-grazing or wood harvesting induced encroachment due to impact on fire intensity (Van Vuuren, 2003). The changes in fire regime characteristics, such as frequency, intensity, extent, type and seasonality of fire, promote the dominance of the invaders (Brooks *et al.,* 2004). The lack of sporadic, hot, brush-killing fires, or the misuse of fires such as prevention of natural veld fires, contributes to bush encroachment (Barac, 2003).

According to Trollope & Aucamp (1981) high intensity fires kill tree seedlings, saplings and small trees and damage the above-ground parts of large woody plants, thereby retarding their growth and reproductive organs. In the case of *Prosopis glandulosa* the reductions in fire frequency and intensity, resulting from reductions in fine fuel mass and continuity associated with heavy, continuous livestock grazing, influence the spreading of *Prosopis glandulosa* (Brown & Archer, 1999). The impact would have allowed established, but suppressed, woody plants such as *Prosopis glandulosa* to increase in stature, express dominance over the surrounding herbaceous vegetation and attain seed bearing size (Brown & Archer, 1999). Fire remains effective in top killing shrubs of honey mesquite (*Prosopis glandulosa*) and in so doing may interfere with its development toward reproductive maturity and its ability to set seed (Drewa *et al.,* 2002). Although management of fire has been mentioned to be a causal factor of bush encroachment, it is amazing to note that bush encroachment occurs in many arid regions where fuel loads are insufficient for fires to be an important causal factor (Ward, 2005).

1.1.2.2 Disturbances

Disturbances have been mooted as major determinants of savanna structure, with savanna being portrayed as an inherently unstable ecosystem that oscillate in an intermediate between those of stable grasslands and forest (Ward, 2005). The varying disturbances include the impact of anthropogenic influences and environmental causes e.g. grazing, herbivory, and fire, drought, flood, spatial heterogeneities in water and nutrients and seed production. The impact of above mentioned disturbances incorporates not only their introductions, but also the elimination of them (Pyke & Knick, 2003b). Human beings influence the secondary factors such as fire and herbivory in contributing to bush encroachment (Wiegand *et al.,* 2006). Invasive species may respond to human induced environmental changes and these species in turn initiate changes through their dominance on the landscape (Pyke & Knick, 2003b). Disturbances such as poor grazing practices and injudicious use of fire are seen as factors that affect loss in the resource allocation, therefore stimulating bush encroachment (Smit, 2004). Invasive organisms have equal probability of spread regardless of the distribution of the disturbance when disturbance extent exceeds the threshold area (Pyke & Knick, 2003b).

Rapid colonisation and increase in density of invasive plants are often tightly linked to disturbances (Invasion plants in SA, 2005). Increases in human populations, advances in technology and transportation and shifts toward global economies have created human activities that have transformed land uses. These human activities have modified the earth's biogeochemistry and have influenced the biological resources on the planet (Pyke & Knick, 2003a). The lowering of historic biogeographic barriers that formerly restricted the spread of organisms into new landscapes are contributing in creating an opportunity for species to colonise and, in some cases, dominate new environments (Pyke & Knick, 2003b).

Occasional favourable episodes occur, though and it is during these episodes that tree populations increase (Ward, 2005). The invasion of the semi-arid lands by *Prosopis* species is causing large areas to be uneconomic. Rodents and other mammals also support the increase in *Prosopis* species encroachment by aiding disseminations of *Prosopis* seeds (Drewa *et al.,* 2002). Although Drewa *et al.,* (2002) indicates that rodents and other mammals serve as dispersal agents of *Prosopis* plant spreading, the argument differs with Brown & Archer (1999) who attribute the spread of *Prosopis glandulosa* to introduction of horses, cattle and sheep. They regard limited *Prosopis glandulosa* spreading as the result of lack of effective dispersal agents (Brown & Archer, 1999). Although invasion of woody plants poses serious threat to savannas, the trees are essentially part of this biome ecosystem.

1.1.2.3 Soil nutrients

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, phosphorus, series of anions and cations and various trace elements, are essential to the nutrition of plants and act as determinants of the composition, structure and productivity of vegetation (Hagos & Smit, 2005).

In structured 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 herbivore species (Hagos & Smit, 2005). On the other side, 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 (Smit & Swart, 1994). The improvement of soil potential leads to the development of structured savanna vegetation (Smit, 2004).

1.1.2.4 Soil layer

The conventional wisdom about bush encroachment is questioned, because there are larger areas of bush encroached areas in Southern Africa where there is only a shallow soil layer with insufficient depth for the stratification of grass and tree roots into different layers (Ward, 2005). The differences in soils are important to vegetation structure and species composition (Teague & Smit, 1992). This is evident on heavier soils whereby there are large variations in yield from year to year and marked species changes with time after clearing. According to Smit (2004) much of the spatial heterogeneity in woody vegetation is correlated with various physical and chemical soil properties, therefore contributing to extend bush encroachment in some areas.

1.1.2.5 Seedling recruitment

Seedling recruitment also serves an alternative idea about the causes of bush encroachment (Hurt & Tainton, 1999). This idea is substantiated because it is argued that tree abundance varies, depending on demographic bottlenecks during seedling recruitment or the sapling release stage (Bond *et al.*, 2003). In arid savannas, rain may be too little or too intermittent for successful tree seedling establishment. In mesic savannas, seedling establishment may be much more frequent but higher rainfall is likely to produce higher grass biomass and more frequent fires. Therefore saplings may be stuck in the fire-trap, thus making the sapling release to be the key bottleneck (Bond *et al.*, 2003).

Seedling recruitment is the most critical stage in the life history of the woody plants with potentially long life spans and low post-establishment mortality rates (Brown & Archer, 1999). According to Brown & Archer (1999), continuous recruitment of shrubs in relatively arid systems may be more important than the dogma of event driven or episodic recruitment would suggest. Their study shows that size and age class of distribution of *Prosopis glandulosa* have no indication of episodic establishment or mortality (Brown & Archer, 1999). According to Brown & Archer (1999) their study confirms that recruitment of *Prosopis glandulosa* could have been relatively continuous over the last 100 years. Herbaceous plants have little effect on *Prosopis* species recruitment. *Prosopis glandulosa* successfully emerged and established across a broad range (185-453 g/[m.sup.2]) of aboveground herbaceous biomass levels achieved by clipping and by reducing plant density (Brown & Archer, 1999).

1.1.2.6 Climate

The effects of livestock as primary determinants in bush encroachment are also complemented by factors such as climatic changes in historical atmosphere, e.g. carbon concentration (Brown & Archer, 1999). Changes to temperatures are severely limiting at certain times of the year to woody components influencing productivity and species distribution (Teague & Smit, 1992).

Moisture availability is an important determinant of species composition (Hurt & Tainton, 1999). Variable rainfall in arid areas influences bush encroachment because 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). It has been noted that encroachment of *Prosopis* species may increase as a result of long-term changes in patterns of precipitation. The prolonged dry conditions result in perennial grass mortality and may foster *Prosopis* species invasion (Drewa *et al.*, 2002). The soil profile dries progressively from top to downwards, so that soil water potentials in the upper layers will become more negative than those in the sub-surface regions that normally have lower root densities and are buffered against evaporative losses. The herbaceous plants that concentrate their roots in the topsoil maintain positive turgor, despite the high negative water potentials and are consequently subjected to strong water deficits, desiccation and the death of most of their green material as they enter a state of forced dormancy. The woody plants will be able to continue normal metabolic processes during periodic mid-summer drought periods, even when the herbaceous plants have been forced into a dormant phase during difference of moisture availability with depth in the profile (Hurt & Tainton, 1999).

The global changes such as elevated CO_2 levels may provide advantages to cool season invasive plants (Archer, 2003). The nitrogenous emissions may elevate available nitrogen on a regional basis, therefore favouring fast-growing invasive species such as *Prosopis glandulosa* (Pyke & Knick, 2003a). Climatic change, historic atmospheric CO_2 enrichment and exotic species introductions are potentially important contributing factors to bush encroachment. The current trends in atmospheric CO_2 enrichment may exacerbate shifts from grass to woody plant domination, especially where the invasive trees or shrubs are capable of symbiotic N_2 fixation (Archer, 2003). Changes in CO_2 directly affect growth rates by altering photosynthetic rates.

These changes in atmospheric carbon dioxide could affect the probability of woody plants growing to fire resistant size and therefore alter the tree/grass balance (Pyke & Knick, 2003b). CO_2 effects are likely to be influential for plants recovering from disturbance since light, water and nutrients are least likely to be limiting growth after a burn, therefore facilitating maximum CO_2 responsiveness of photosynthesis and carbon fixation (Bond *et al.*, 2003).

The different responses of tree and grasses to CO_2 are determined mainly by differences in carbon demand for structural allocation, not differences in carbon gain between CO_4 grasses and CO_3 trees. The potential interactive effects of CO_2 and fire on trees would operate regardless of whether the grasses were CO_3 or CO_4. There is a prediction about responsiveness of elevated CO_2 where CO_4 species will respond less than CO_3 species (Walker & Steffen, 1997). Carbon dioxide is an essential requirement for plant growth obtained from the earth's atmosphere. Elevated CO_2 generally increases the allocation of photosynthate to roots, which increases the capacity and or activity of belowground carbon sinks (Walker & Steffen, 1997).

The elevated CO_2 could be having a widespread and pervasive effect on savanna vegetation by tipping the balance in favour of trees (Bond *et al.*, 2003). The tree cover is sensitive to atmospheric CO_2 with large decreases at low CO_2 level and massive increase from pre-industrial conditions to today's levels (Bond *et al.*, 2003). Atmospheric CO_2 enrichment may have in way a facilitated invasion by reducing soil water depletion by grasses (Polley *et al.*, 2002).

1.1.2.7 Patch dynamics

The patch-dynamics approach to savanna dynamics is one of the emerged hypotheses in the debate regarding the need for shift in paradigm about the causes of bush encroachment (Wiegand *et al.,* 2006). According to Wiegand *et al.,* (2002), bush encroachment in many semi-arid and arid environments is a natural phenomenon occurring in ecological systems governed by patch dynamic processes. Woody plant encroachment is part of a cyclical succession between open savanna and woody dominance and is driven by two factors namely rainfall and inter-tree competition (Wiegand *et al.*, 2006). Rainfall in savanna regions is often patchily distributed, both in time and space. The patchiness of rainfall leads to patchy vegetation patterns often in several hectares within an intermediate range of long-term rainfall levels only (Ward, 2005). The soil moisture to support tree growth is insufficient when the average rainfall is too low, while above a certain quantity of rainfall dense woodlands with mixed age distribution will develop.

The patches induced by grass-tree competition as a result of grazing contribute to bush encroachment. Different management practices and selective grazing habits of animals lead to uneven utilisation of rangelands, thereby resulting in the development of a mosaic of patches (Wiegend *et al.,* 2006). Each of the developed patches tends to have a different floristic composition (Kellner & Bosch, 1990). Grazing effectively weakens the suppressive effect of the grass layer on young trees in a patch of a few hectares, leading to the conversion effect of an open savanna patch into bush encroachment. The established encroached bush may take decades to revert to an open savanna. According to Kellner & Bosch (1990), a lack of stocking rate adaptation on the imbalances of vegetation as a result of a mosaic of patches or application of different management strategies, will enhance the degradation of the management unit.

1.1.3 Invasion of alien *Prosopis* species

The nitrogen-fixing genus *Prosopis* species is estimated to have 44 species native to North and South America, Africa and Asia (Ehrhorn, 1996). These nitrogen-fixing *Prosopis* species range from 1 m tall shrubs to 18 m tall trees (Ehrhorn, 1996).The *Prosopis* species are regarded to be invasive because they are non-native to South African ecosystem and thus causing economic and environmental harm (Geesing *et al.,* 2005). As already mentioned the genus of *Prosopis* plant consists of 44 recognized species, of which 40 are native to the Americas, distributed within a wide ecological range (Geesing *et al.,* 2005). Several species of *Prosopis* species such as *Prosopis juliflora, Prosopis velutina, Prosopis glandulosa* var. *glandulosa and Prosopis* var. *torreyana* have been imported into South Africa from various sources in the USA, Hawaii and Mexico as early as 1880s (Klein, 2002).

Only one species, *Prosopis africana*, is native to Africa, occurring in the Sahelian zone from Senegal to the Sudan, Uganda and Ethiopia. An invasive species is characterised by rapid growth rates, extensive dispersal capabilities, large and rapid reproductive output and broad environmental tolerance. The farmers were encouraged to grow *Prosopis* plants in large numbers to provide shade, fuel wood and fodder in the form of nutritious pods in the arid regions, where few other trees will survive (Klein, 2002). *Prosopis* also became a common ornamental tree in many towns and homesteads (Zimmermann & Pasiecznik, 2005). *Prosopis* plants were introduced also for use in sand stabilization projects, soil improvement and for hedges to contain livestock (Wittenberg & Cock, 2001).

An example of how *Prosopis* plants could infest farming land is depicted below (Figure 1.1).

Figure 1.1: *Prosopis* infested agricultural land.

The four species, *Prosopis juliflora, Prosopis velutina, Prosopis glandulosa* var. *glandulosa* and *Prosopis glandulosa* var. *torreyana* have become invasive and naturalised in South Africa, particularly in the Northern Cape and Free State Provinces. The two varieties of *Prosopis* cause most of the problems and are both natives of North America (Klein, 2002). The increase in invasion of *Prosopis* species is assisted by partial reliance on rainfall for their water needs (Geesing *et al.*, 2005). The plant is able to tap ground water supplies with its deep root system or absorb foliar water as mechanism for coping with drought. *Prosopis* species thrive on nutrient poor or degraded and even saline or alkaline soils (Geesing, *et al.*, 2005). The infestation of *Prosopis* plants is also aided by their ability to withstand extremely high temperatures (Ehrhorn, 1996).

The problems that lead to *Prosopis* being a burden have been a century in the making (Zimmermann & Pasiecznik, 2005). Firstly, the unsustainable *Prosopis* species were introduced and widely planted for more than fifty years, starting in the 1900s. There was also early hybridisation between the two dominant species, *Prosopis velutina* and *Prosopis glandulosa* var. *torreyana*. This hybridisation of the two dominant species displayed hybrid vigour and proved to be very invasive (Zimmermann & Pasiecznik, 2005). The invasive trees lost most of their valuable properties and were therefore exploited less.

1.1.4 Reproduction mode of *Prosopis* species

Prosopis plants are the type of tree species that rapidly invade landscapes in the semi-arid and arid lands. *Prosopis* plants, by being phreatophytes, are largely confined to alluvial plains where ground water stores are easily accessed and reliable such as the valleys of the major rivers (Invasion plants in SA, 2005). The invasion of *Prosopis* species in the watercourses is also a major problem and is evident in the study area (Appendix J).

Prosopis species reproduce primarily by seed, which may be highly multiplied. The success of *Prosopis* species reproduction as invaders is largely attributable to the massive number of seeds produced. About 60 million of seeds per hectare per year may be produced and multiplied at faster rate depending on efficiency dispersal methods (Mathews & Brand, 2004). *Prosopis* plants produce their first flowers and seeds when they are between two to five years of age (Csurhes, 1996). The flowering occurs in spring, pods take two to three months to mature and fall in late summer. The mature trees are prolific seeders with estimates of seed-set ranging from 630 000 to 980 000 seeds per tree per annum (Csurhes, 1996).

The *Prosopis* plants' flowers are eaten by numerous bird species (Management considerations, 2005). The recruitment of *Prosopis glandulosa* depends upon plant tolerance of herbivory and or low herbivore abundance, during seedling establishment (Weltzin *et al.,* 1998). All *Prosopis* species are capable of regenerating from basal buds located at or just below the soil, when top growth is removed (Brown & Archer, 1999). The grass-trees associations for effective tree thinning in reducing negative competition, require consideration of the question of how many and which tree species should be removed during clearing operations (Smit & Swart, 1994). The holistic approach in dealing with *Prosopis* species cannot be over-emphasised, because although for decades chemical and mechanical methods have been employed in an attempt to reduce or even eradicate the species rangelands, it has proven to be very difficult to control (Pasiecznik, 2003).

According to Pasiecznik (2003) the invasion of P*rosopis glandulosa* plants is further increased when their seed pass through animals stomachs undigested. The process therefore aids germination and encourages spreading widely by livestock and water (Pasiecznik, 2003). The animals' droppings enhance the infestation of *Prosopis* plants by providing a ready supply of nutrients for the developing seedling (Sastry, 2005). The destruction of surrounding vegetation and exposure of the soil often stimulates mass germination of the soil seedbank, resulting in a sudden infestation of *Prosopis* plants (Mathews & Brand, 2004).

1.1.5 Effects of bush encroachment and invasive alien species

Few studies have in detail quantified the physical invasion process of plants in space and time compared with animals and disease (Invasion plants in SA, 2005). There are some patterns to the invasion process of an area in both space and time, which can be divided into two phases. These invasion pattern phases are expansion and densification. Expansion is the dispersal from the existing patches as an expanding front and by establishing satellite colonies that later become patches. Densification is the increase in the density of population within the colonised patches.

As already mentioned bush encroachment with the densities of 2500 bush equivalents per hectare suppressed phytomass production during years of normal rainfall (Richter *et al.*, 2001). Bush encroachment leads to pseudo-droughts as a result of plant competition (Richter *et al.*, 2001). The effects of high-density trees on drought are the result of their high water use as influenced by their deep root system and evaporation level (Mathews & Brand, 2004). The effect of bush encroachment on soil water is also endorsed by a study of Smit & Rethman (1999) that indicates that there is a low soil water status at high tree densities. The impact of this low soil water status is reflected by the absence of herbaceous plants (Smit & Rethman, 1999).

The invasive plants can indirectly affect native plants and change an ecosystem by altering soil stability, promoting erosion and colonising open substrates (Sastry, 2005). This may affect the accumulation of litter, salt or other soil resources and promote or suppress fire (Brooks *et al.*, 2004). The effects of invaders are particularly dramatic when they alter disturbance regimes beyond the range of variation to which native species are adapted, resulting in community changes and ecosystem level transformation (Brooks *et al.,* 2004). According to Ehrhorn (1996), the diversity of plant communities decline rapidly with increasing aridity.

The *Prosopis* plant is a multipurpose genus, which is biologically diverse, resulting from multiple interbreeding. They are widely adapted to the semi-arid regions of the world (Ehrhorn, 1996). The invasion of the *Prosopis glandulosa* could reduce pasture production by up to 90% in semi-arid regions (Csurhes, 1996). The effect of *Prosopis glandulosa* on the herbaceous biomass and environment is the result of its drought tolerant characteristics.

Prosopis glandulosa requires only 150 to 750 mm rainfall per annum for good growth, while P*rosopis pallida* needs 250 to 1250 mm per annum (Pasiecznik, 2003). The water use of one *Prosopis* plant not only equals the water needs of two non-urban people (60-100L/day), but as a result of its competitiveness, it also seriously threatens the agricultural potential (Versveld *et al.,* 1998). *Prosopis glandulosa* plants are capable of thriving under a wide range of soils and rainfall conditions (Brown & Archer, 1999).

There is some evidence that recruitment in *Prosopis* species depends on good rainfall years (Invasion plants in SA, 2005). Isolated *Prosopis glandulosa* plants have a minor impact on grazing production and may even enhance production in the short term because of the nutritious seedpods and shade they provide (Ehrhorn, 1996). Isolated trees with time, however, reproduce to form dense thickets that replace pasture plants. Dense *Prosopis glandulosa* (honey mesquite) may however interfere with the mustering of stock and the spines also injure animals (Brown & Archer, 1999).

The continued increase in the distribution and density of honey mesquite (*Prosopis glandulosa*), particularly in semi-arid to arid regions, is predicted to result in a physiognomic conversion of open grassland or open woodland to thorned shrublands with a deleterious impact on populations of native flora and fauna (Brown & Archer, 1999). Green pods of *Prosopis* are bitter and can poison livestock in large quantities. The foliage of *Prosopis* is also unpalatable because of the high tannin content, therefore restricting browsing (Mathews & Brand, 2004).

The other evident effects of *Prosopis glandulosa* invasion on rangeland are reduction of carrying capacity as a result of reduced grass production (Smit, 2004, Figure 1.2 & Table 1.1). The loss of biomass production is evident on highly infested land as depicted in Figure 1.2. *Prosopis* trees growing in thickets loose all their useful attributes (Klein, 2002). These reductions in carrying capacity in *Prosopis* invaded land necessitate stock reduction (Pasiecznik, 2003). The *Prosopis* species invasion therefore reduces the profitability of the farm, severely affecting the sustainability of livestock farming (Richter & Meyer, 2001).

Figure 1.2: The loose thickets of *Prosopis* trees impacting on grass production.

The effects of increase in density of above 1500 to 2500 show great reductions of carrying capacity of the veld especially Molopo thorn bushveld. Table 1.1 shows the increase in required hectares per large stock unit with the increase in bush equivalents per hectare in all studied veld types. The Eastern grass shrub veld indicates a low level of change with regard to impact of increase in bush equivalent on carrying capacity as compared to two other studied veld types (Table 1.1.).

Table 1.1: Effects of bush equivalent (BE) on savanna grazing capacity as adapted (Meyer, 1999).

BE/ha	**Grazing capacity ha/LSU in three savanna vegetation types**		
	Molopo thorn bushveld	**Mixed *Tarchonanthus* thorn veld**	**Eastern grass shrub veld**
200	9.0	8.3	8.2
500	11.1	9.7	8.8
1000	15.5	12.5	9.9
1500	21.8	16.3	11.2
2000	30.7	21.1	12.6
2500	43.1	27.4	14.2

Prosopis species have the potential to form dense thickets, excluding native plants, associated animal life and substantially changing community structure (Brown & Archer, 1999). The roots of *Prosopis* species can extend more than 15 metres beyond the canopy and up to 15 metres into the soil profile. The long taproots allow the plants to reach deep water-tables that help to deplete vital ground water reserves in water-scarce environments such as the arid lands of South Africa (Calder & Dyke, 2001). It is generally believed that the loss of ground cover under *Prosopis glandulosa* was caused by increased soil erosion and loss of soil moisture (Csurhes, 1996).

The increase in bush encroachment is not only having major implications for assessment of the sustainability of cattle production, but also on viable livestock management policies (Moleele, 2005).

The phenomenon of bush encroachment intensifies management problems because of the increased risk of fodder shortages and higher feeding costs. Bush encroachment forms part of the range degradation and increase thereof, therefore causing reduction in production potential of rangelands (Meyer, 1999).

1.1.6 Factors that influence the choice of bush and *Prosopis* plant control options

The understanding and correct use of methods that may advances the eradication of bush encroachment as an agricultural issue remain critical. The effectiveness of control methods is important, because the dilemma of this phenomenon is also a biodiversity problem (Ward, 2005). Factors such as bush density, size of area affected, species type and plant communities involved, growth form of dominant species, production potential and financial position of the affected farmer, play a vital role in determining the method of control (Barac, 2003). The intensity of tree thinning with its role in determining the control options, is also influenced by the objectives of bush control (Smit & Rethman, 1999).

The bush control method is not simply a complete removal of woody plants (Smit, 2004). Tree thinning with a view to reduce negative competition effects of grass-tree associations, leads to the question of how many and which tree species should be removed (Smit & Swart, 1994). As mentioned, a holistic approach in dealing with *Prosopis* species cannot be over-emphasised.

Adaptive features that make *Prosopis* plants to control difficult, include abundant, long- lived seed that is disseminated by livestock and wildlife. Other inhibiting characteristics of *Prosopis* plants to control options are, high germination of seed over a wide range of environmental conditions and the ability to resprout following plant damage (Csurhes, 1996). Many of plants resprouted after treatment and developed into multi-stemmed bushes (Csurhes, 1996). As a result of its regenerative capability following damage, control attempts in the past have led to some regions being covered with dense, shrubby thickets that are frequently more detrimental to forage production than the original invasive stands (Management considerations, 2005).

1.1.7 Possible bush control methods for all bush encroaching species with particular reference to *Prosopis* species

1.1.7.1 Introduction

The methods of controlling encroachment have been researched for centuries and yet there is still no clear consensus about the right method or neither any standard of bush control method despite this long history of research (Ward, 2005). All bush control methods, such as chemical, mechanical and biological, are applicable to be used for a variety of encroaching species (Csurhes, 1996). The applicability and use of these methods may differ on plants species because of prevailing conditions, such as extent of infestation, climatic conditions and available resources. The method used should be appropriate for the species concerned as well as to the ecosystem in which they occur (Drewa *et al.,* 2002). One or a combination of all bush control methods may be used to attain the desired results.

The invasion of the woody component in a savanna may be controlled through correct veld management that places emphasis on the prevention of over-utilisation of the grassy layer (Smit, 2004). The application of detailed veld management through grazing and browsing is a control option of invading species and also a follow-up control option on eradicated areas. Good livestock management practices can improve the success of *Prosopis* control programmes (Csurhes, 1996). However, Brown & Archer (1999) indicates that grazing management should not focus on grass-shrub seedling interference, but instead on minimising seed dispersal in the case of leguminous shrubs, where livestock may be primary vectors. Maintaining an effective fire regime, can also assist reduce bush encroachment (Smit *et al.,* 1999).

Land managers often attempt to remove *Prosopis* plants because it reduces grass production. According to Sastry (2005) phase wise removal of *Prosopis* is essential because removing the entire plant at once may cause ecological problems. Phase wise removal entails removal of *Prosopis* species in plots of 1 km x 1 km starting from mature patches influencing invasion (Sastry, 2005). As a result of its good reproductive potential and regenerative capabilities, the plants will probably never be eliminated from sites where it has become established (Management consideration, 2005). The argument is substantiated by Brown & Archer (1999) by indicating that *Prosopis* species are capable of vegetative regeneration within two weeks of cutting. The *Prosopis* plant is also tolerant of repeated top removal during the first growing season and tolerant to hot fires by its second and third year of growth (Csurhes, 1996).

The traditional rangeland management through proper rotation grazing system in coordination with controlled burning may be most effective in managing the spread of *Prosopis* species encroachment dependent on the rate and size of problem (Pasiecznik, 2003). The control options need to be evaluated and chosen on the basis of the likelihood of success, cost effectiveness and any potential detrimental impact on environment (Pyke & Knick, 2003). The reliance on the ecological principles in bush control may influence the ultimate management of the encroaching species in rangelands (Figure 1.3).

The model (Figure 1.3) indicates the three concepts that deal with system dynamics and system domain of attraction. It indicates that stable veld changes little in composition and production when subjected to outside stress (Smit, 2004).

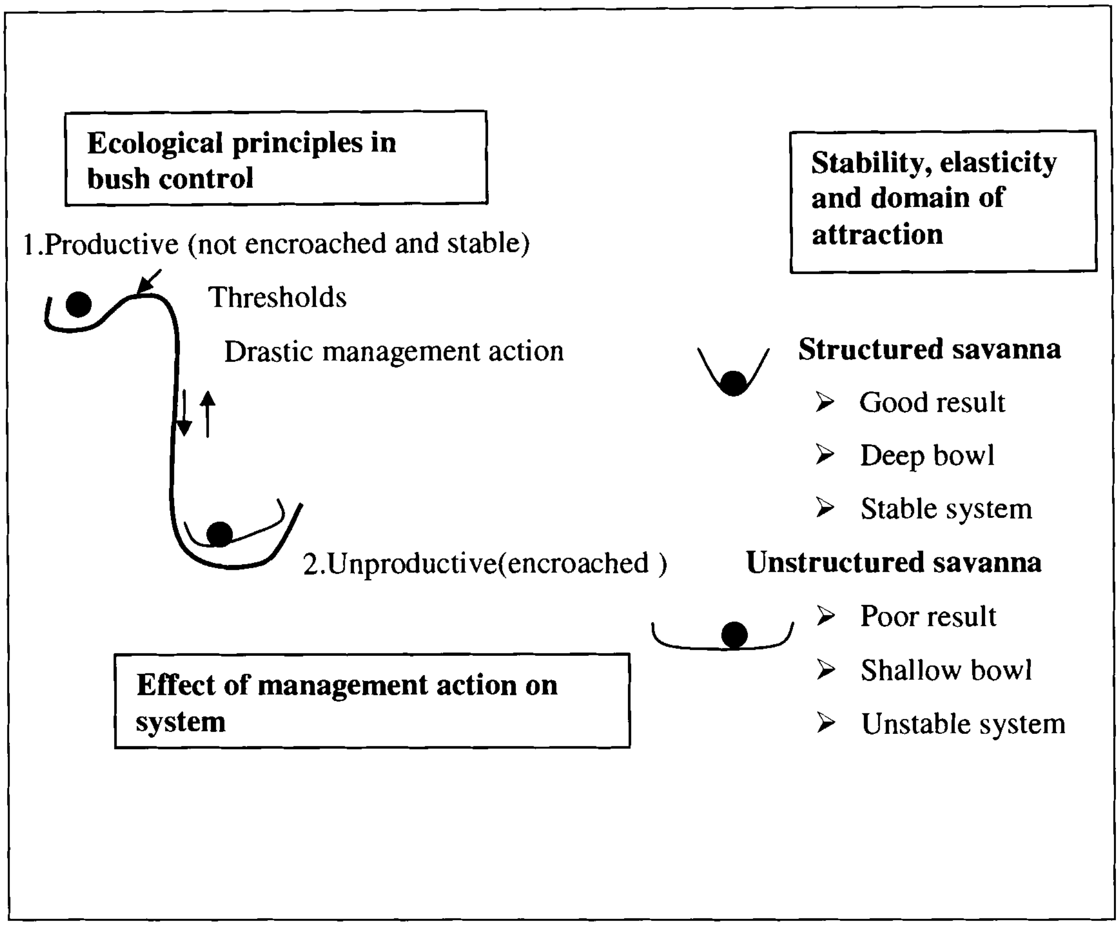

Figure 1.3: Simplified approach to the principle of stability, resilience and domain of attraction as applied to bush encroachment, showing importance of savanna (Smit, 2004).

The resilient system may or may not be stable, but remains attracted towards its equilibrium. There is also a region of a system's state space within which the system is attracted towards an equilibrium termed a domain of attraction. The simplified approach to the principle of stability, resilience (high elasticity) and domain of attraction could be used in restoring encroached savannas with the correct approach to tree thinning (Smit, 2004).

The position 1 on the model indicates that changes in responses to determinants such as drought or grazing may occur depending on its resilience, but as the influence of these changes is removed it will still be attracted towards its original state stays i.e. structured savanna ecosystem (Smit, 2004). These changes must also be within the limits of the ecosystem's domain of attraction and should such state be above domain of attraction and be charged across a certain threshold, the ecosystem will i.e. changes on unstructured and unproductive savanna change to position 2, which may be a stable but an encroached situation, (Meyer, 1999). The change from a stable, structured savanna to an unstable unstructured savanna may occur as a result of management actions or environmental impacts. The model in Figure 1.3 supports what Smit (2004) regards as structured savanna through large trees that are able to suppress the establishment of new seedlings, thereby managing encroachment.

In short grass communities where grasses are less competitive, grazing management is most critical to suppression of *Prosopis* invasion (Csurhes, 1996). Tree thinning or clearing by mechanical or chemical means will result in immediate changes in competition between woody and herbaceous plants, which often determines the growth and structure of savannas (Smit, 2004). The integration of different strategies such as eradication, containment, control and mitigation are vital for effective control of established invasive species (Wittenberg & Cock, 2001). The integration of different strategies is important, as the effective restoration of bush encroached areas should not be considered as a one-off event, but rather as a long-term commitment (Smit, 2004).

1.1.7.2 Mechanical control

This control option is a drastic step that entails the use of machinery and implements, manual methods by hand, felling, hacking or digging out, cable chaining, roller chopping, root plowing, tree grubbing as well as land imprinting (Meyer, 1999). The use of chain-saws (Figure 1.4) for felling is one of the most used implements in controlling bush encroachment through a labour intensive programme such as Working for Water or during cut stump applications. Working-for-Water (WfW) is a national programme, initiated by the Department of Water Affairs and Forestry (DWAF) in 1995 (DWAF, 2003). This programme focuses on the eradication of alien invasive species in South Africa (Barac, 2003). The cut stump control method is usually suitable for the eradication of single-stemmed woody plants. It must be followed by the applications of arboricides on the stumps to kill off the undesirable woody plants completely to prohibit resprouting from the stumps (Csurhes, 1996).

The other important consideration in mechanical operations is to damage or remove dormant buds that occur along the underground stem in order to prevent sprouting and

Figure 1.4: Mechanical bush control methods using a chain-saw.

for mechanical measures to be effective. The damaging of underground parts of the trees is recommended for effective control, because it is noted that problem species do not die after being chopped down (Strohbach, 1998). It is necessary to remove *Acacia* tree species to a depth of about 20 cm underground. The dormant buds at the base of the stem give many of the woody species even *Prosopis* species the ability to resprout from the roots (Strohbach, 1998). Tree grubbing with blades attached to crawler surface and root ploughs, which cut roots, 15 to 30 cm below the soil surface. The root ploughs uproot trees and are very effective control measures, often achieving more than 90% success rate (Csurhes, 1996). The shortcoming of root ploughing is that it disturbs or kills burrowing rodents (Management consideration, 2005).

The land imprinter is a heavy roller, set with pyramid shaped teeth, 10 to 15 cm long attached in an irregular pattern and pulled behind a caterpillar tractor. Hand grabbing of *Prosopis* seedlings, although very labour intensive, is an effective preventive measure used for removing *Prosopis* species during the early stages of invasion (Pasiecznik, 2003). When the roots are cut 10 cm below the soil surface hand grubbing effectively kills plants under 2, 5 cm in stem diameter. The other method that is used for bush control is the cut only treatment. This method is not only labour intensive, but also ineffective, in that it only stimulates vigorous basal re-growth (Coetzee, 2004). The cut only treatment is not regarded as a method that can successfully control *Prosopis* invasions. The success of mechanical bush felling is also influenced by the season of control. It has been noted that re-growth becomes low and mortality rate high for trees felled during the rainy season between January and April (Strohbach, 1998). The eradication of tall, dense infestations, requires uprooting and root ploughing, which must remove the bud zone of the root system (about 30 cm below the surface) to prevent re-sprouting (Geesing *et al.*, 2005)

1.1.7.3 Chemical control

The chemical control method entails plant (foliage or stem) and soil applied chemicals (fluid or granules), including aerial applications. The arboricides applied directly on the plants are soluble in water or diesel. The arboricides such as Tordon super, which is a oil miscible formulation, have the active ingredient picloram/triclopyr (120/240 g/I). It is sprayed directly onto the trunks of smaller trees, e.g. trunk diameter of less than 100 mm (Csurhes, 1996). The trees with a trunk diameter of more than 100 mm should be cut down first and the chemical should then be applied to the stumps. Taller plants may be less susceptible to arboricides than shorter ones. The trees have to be cut down 100 to 150 mm above ground level to obtain good results. At the ground level the arboricides are applied to the cutting plane, as well as the stump and all roots protruding above the ground. The arboricides must be applied as soon as possible after the tree has been cut down, especially during the active growing season for effective results (Csurhes, 1996). The cut stump application and use of arboricides immediately after the cut stump method is very effective against the re-sprouting of many woody plants (Wittenberg & Cock, 2001).

Many multi-stemmed plants are more resistant to foliar applied chemicals than single to few-stemmed plants (Management considerations, 2005). The arboricides called Access, with ingredient picloram (120/240g/I), is a chemical herbicides registered for foliage spraying of a number of indigenous species (Meyer, 1999). It should be mixed at 0,5% with water and it is recommended for spot treatment of isolated or clumps of plants, smaller trees 1 to 2 m, re-growth, as well as the control of seedlings. The foliar control option of bush control depends upon natural and man-made factors for effective control. The climate or season has an impact on the plant vitality such as washing away of chemicals by rain before being adsorbed. The plant vitality effect is observed on plants that cannot efficiently absorb or are in translocation of arboricides due to stress. The humidity and temperature should be suitable for effective chemical control. The method, application and degree of wetting are critical for sufficient coverage of the contact surface of the plants to attain good bush control results.

There are soil-applied arboricides such as Molopo GG with active ingredient tebuthiuron (200 g/kg) that has macro-granule chemical formulations used in bush control. The specific dosage is measured off with a special measuring spoon and onto the soil next to trees. The Molopo GG is in a form of granules and can be applied by hand or aerially with an aeroplane. Soluble concentrate formulation Molopo SC with tebuthiuron (500 g/I) as active ingredient is also recommended for different *Acacia* species (Smit, 1991). Garlon has the active ingredient triclopyr (butoxyl ethyl ester 480 g/I) with emulsifiable concentrate. The toxins of these herbicides are taken up by the roots of the woody plants and inhibit photosynthesis by killing off the leaves. All trees are consecutively killed off until they finally die. The following factors determine the effective dosage of soil applied chemicals, i.e. clay content of the soil, organic matter content of the soil, bush species, size and structure of the bush (Barac, 2003). The effectiveness of soil-applied arboricides is influenced by soil moisture or rainfall. These arboricides require a certain level of moisture and or are washed away into the soil by rain to become active and absorbed by the plant. The type and accuracy of application tools are of importance for effective bush control (Barac, 2003).

The soil-applied herbicides are also adsorbed by clay particles that render them inactive. There is a need for further application or a high dosage rate in a high clay area in order to compensate for any loss (Barac, 2003). The pH of soil affects the rate of herbicides breakdown with impact on residual effect of some chemicals. The humus or organic material content of the soil makes it a stronger adsorber of the chemicals ions, thereby possibly rendering herbicides ineffective.

Molopo SC and others are mixed with water and applied to the soil at the base of the plant with a dosing syringe. The dosage can also be added to the mixture to mark the trees that have been treated (Smit, 1991). The success of any chemical control methods, both foliar and soil applied, depend on a number of factors (Meyer, 1999). The other system of chemical usage is through aerial application. This method is suitable to handle large areas rapidly where woody plants are very dense with little labour required. The disadvantage of the system is that valuable plants may be adversely affected by the herbicides owing to the non-selective nature of aerial application (Smit *et al.*, 1999).

1.1.7.4 Biological control of encroaching species with reference to *Prosopis* species

Biological control is the intentional use of populations of upper trophic level organisms commonly referred to as natural enemies, or naturally synthesized substances against pest species to suppress pest populations (Wittenberg & Cock, 2001). The use of host specific natural enemies such as insects and disease occurring organism are able to reduce the invasiveness of alien plants (Klein, 2002). There are biological measures such as the use of fires and utilisation systems that remain vital for effective control of *Prosopis* species. Biological bush control involves the application or manipulation of natural factors that control the composition of savannas (Smit, 2004).

The use of the biological bush control is aimed at prevention or initial slowing down of the re-encroachment or re-thickening in the follow-up control programme after the initial drastic reduction of woody plants (Meyer, 1999). The other effective biological method is an old time immemorial environmental factor called fire. Fire is regarded to play an important ecological role in plant communities (Trollope & Aucamp, 1981). The uses of top killing fire as a biological system contribute to inhibit flowering and seed setting of invasive species (Figure 1.5). The prohibition of flowering and seed setting not only aids re-encroachment but also serve in eradication of highly encroached rangelands.

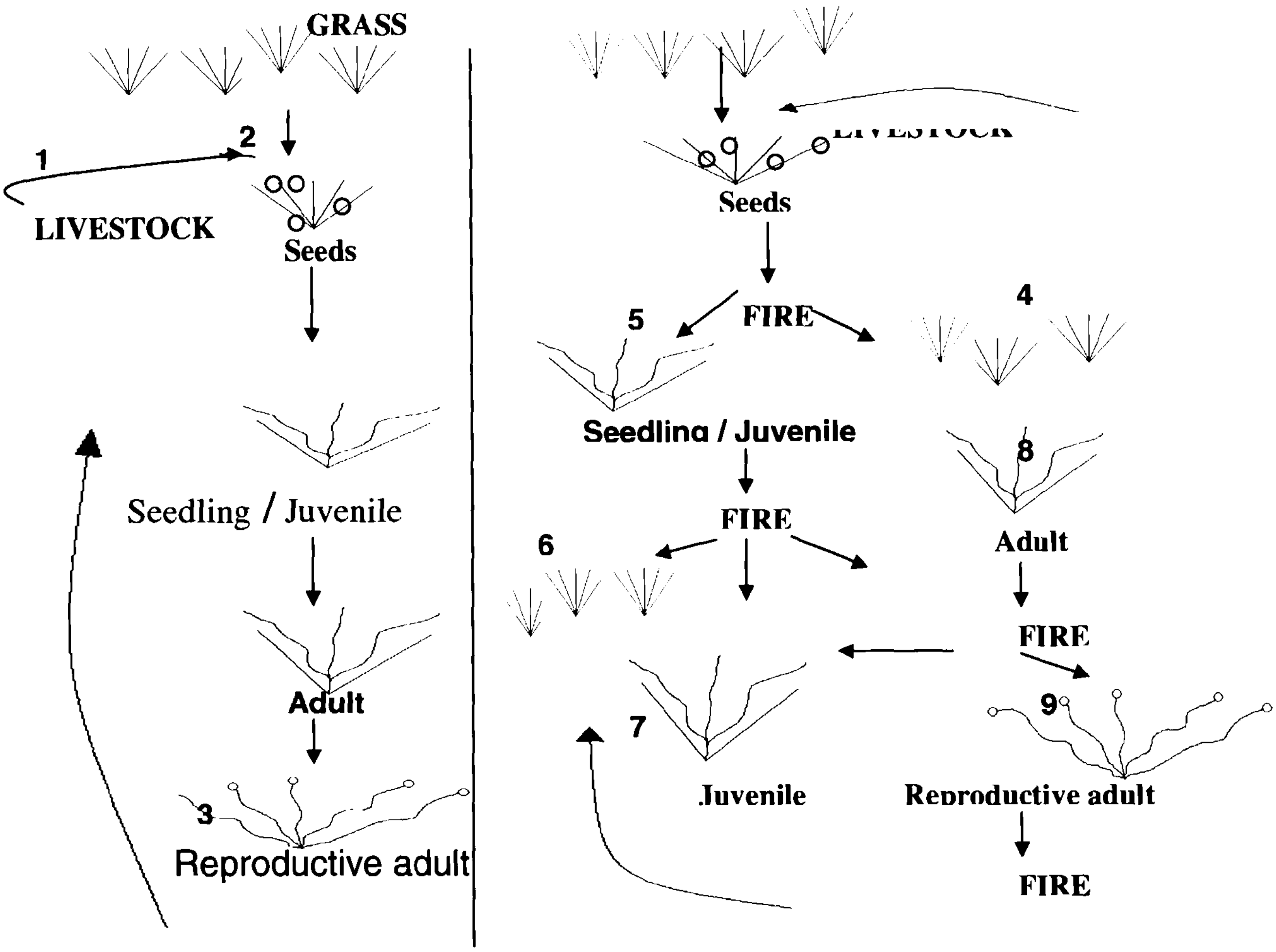

Figure: 1.5: Schematic representation of conceptual model to illustrate how growing season fire might be used as an effective means of deterring honey mesquite invasion (Drewa *et al.*, 2002).

This is advantageous because *Prosopis* plants established from seed may take 10 to13 years before reaching reproductive maturity (Drewa *et al.*, 2002). The model illustrates how prescribed fire might be used as an effective means of deterring further honey mesquite (*Prosopis glandulosa*) invasion in the presence of low and variable herbaceous biomass (Figure 1.5). The models in the left panels (1) indicate the dissemination of honey mesquite seeds into rangeland by cattle initiates' shrub invasion. In the absence of fire at (2) the establishment and growth of honey mesquite plants result in a shrub dominated community over time. Individual plants that reach reproductive maturity (3) provide seed sources for dissemination by livestock and the spread of mesquite across the landscape. In the right panel the mesquite invasion can be decreased in numerous ways. Fire that kills mesquite seeds can allow the persistence of grassland through time (4). In (5) it shows that on some sites seeds of honey mesquite may survive fires and subsequently become established.

An example of the use of fire as a control mechanism for complete kill in bush control is highlighted below (Figure 1.6).

Figure 1.6: An example of the use of fire (stem burning) for complete kill as bush encroachment control mechanism.

Where there is complete kill of young plants, these sites may return to grassland stage (6). Alternatively, fires may be effective in top killing smaller shrubs of honey mesquite to maintain an immature stage (7). For larger shrubs (8), top kill is more likely following fire as well as a subsequent return to an immature stage. Where fires were either of very low intensity or did not occur, other shrubs may reach reproductive maturity (9).

Fire may ultimately top kill these shrubs and return them to a juvenile stage. The effectiveness of fire is expected to be contingent on the quantity and spatial distribution of herbaceous biomass as well as shrub size and status (Drewa *et al.*, 2002). *Prosopis* plants that range from 1. 5 to 2. 5 years old and 9 to 21 cm in height may be vulnerable to complete kill through stem burning (Drewa *at al.*, 2002). The degree of complete kill through stem burning on honey mesquite depends in part on shrub size as well as fire season and intensity (Csurhes, 1996). Stem burning is a technique where a flameless, smouldering and glowing fire is ignited around the base of the undesirable woody plants (Smit *et al.*, 1999). The material used is packed to encircle the entire stem up to a height of 15 to 20 centimeters (Figure 1.6). Stem burning is easy to use for the eradication of large, single stemmed trees, however impractical to use on control of small trees and multi stemmed bushes. The number and distribution of livestock in conjunction with the growing season fire may be an effective strategy for controlling invasions of *Prosopis* plant (Drewa *et al.*, 2002).

Biological control of *Prosopis* species also entails the use of seed-feeding beetles *Algarobius Prosopis* and *Neltumus arizonensis* (Klein, 2002). These biological agents destroy seeds and reduce the invasiveness of the weed without affecting its beneficial attributes (Klein, 2002). The effectiveness of these beetles, that are able to fly long distances to colonise *Prosopis* plants, is compromised by some factors. This includes the fact that livestock and game feed on most pods before the larvae have had a chance to colonise them (Mathews & Brand, 2004). Browsers may also be used as a control option although they can seldom eradicate bush encroachment. This will require efficient goat management as part of the system (Trollope & Aucamp, 1981). Effective bush control by means of domestic browsers, such as Boer goats, can only be successful if the woody plants are completely within browsing reach (lower than 1,5 metres) (Hardy *et al.*, 1999).

1.1.7.5 Aftercare as part of control options

This is a form of maintenance following bush control technologies in rangelands. It is aimed at ensuring that the restored ecosystem recovers fully. The aims of aftercare as part of the follow-up strategy are to prevent or inhibit the re-growth and coppicing of the problem plants and promote the recovery of the herbaceous layer (Barac, 2003). The follow-up programme is necessary because single-stemmed trees that are initially treated by means of certain bush control technologies tend to become multi-stemmed when coppicing of the bush occurs (Brown & Archer, 1999).

The following methods are frequently recommended in after-care programs. These include the application of fire to kill re-growth and young, emerging seedlings. The incorporations of browsers to control or slow down the recovery rate of bush encroaching species (Trollope & Aucamp, 1981). Chemical follow-up controls through the use of selective use of arboricides to inhibit re-growth and the manual control of re-growth by hacking out or pruning back the coppicing plant parts at the base of the stem (Barac, 2003). Repetitive follow-up actions are mandatory until the required control has been achieved (National Department of Agriculture, 2001).

The results of the study sites' bush equivalent at above 650 BE per hectare on previously controlled plots indicates the necessity of planned systematic after-care to deal with bush encroachment. Owing to some problems during applications, none of the chemical control options used on the research trial in the study area are 100%

effective and efficient (Meyer, 2000). The follow-up treatments become necessary after 6 to12 months.

1.1.8 Bush encroachment and legislation

The Conservation of Agricultural Resource Act, Act No. 43 of 1983, (CARA) revised regulation 15 and 16 promulgated by the Minister for Agriculture and Land Affairs on 15 March 2001 is a framework for the regulation of alien and invasive alien and indigenous species. Bush encroachment is a term used for stands of plants of the kinds, where individual plants are closer to each other than three times the mean crown diameter. *Prosopis* species fall within category two of the CARA Act, which indicates that these plants have the proven potential of becoming invasive, but which, nevertheless, have certain beneficial properties (Ehrhorn, 1996).

Category two invaders may not occur on any land or inland water surface other than in an area officially demarcated for that purpose or biological control reserves. These category two plants, such as *Prosopis glandulosa,* have recognised commercial value, provided they are grown under controlled conditions. These plants may not be propagated or sold, unless intended for use in such a demarcated area or in a biological control reserve. All land users shall control plants of these species that occur on any land under their management.

Regulations 15 and 16 of CARA Act makes provision for category two plants to be retained in special areas demarcated for that purpose, but those occurring outside demarcated areas have to be controlled (National Department of Agriculture, 2001). Trading with these plants and their products is allowed, but only when authorised by an executive authority as stated in the revised regulation of CARA Act, Control of category two plants has to adhere to prescribed control methods and inhibit any further spreading.

The regulation of plants contributing to bush encroachment has lately been given an additional boost by the drafting of a new bill and, lately, approval of the National Environmental Management Biodiversity Act No. 10, 2004 (NEMBA). The chapter 5 of NEMBA Act, further tries to give directions to alien and invasive species management. This chapter 5 of the NEMBA Act is also responsible for species and organisms posing potential threats to biodiversity. The purpose of this chapter is to prevent the unauthorised introduction and spread of alien and invasive species to ecosystems and habitats where they do not occur naturally.

The other purpose of chapter 5 of the NEMBA Act is to manage and control alien and invasive species to prevent or minimise harm to the environment and to biodiversity in particular and to eradicate alien and invasive species from ecosystem and habitats where these may harm such ecosystem or habitats. The first part of the chapter 5 of the NEMBA Act, deals with alien species specifically on restricted activities involving alien species, exemptions. Chapter 5 of NEMBA Act, also deals with restricted activities involving certain alien species totally prohibited, amendments of notices and duty of care relating to alien species.

The second part of the chapter deals with invasive species, particularly with the list of invasive species, restricted activities involving listed invasive species, amendments of notices, duty of care relating to listed invasive species, request to competent authorities to issue directives, control and eradication of listed invasive species, invasive species control plans of organs of state and invasive species status reports (NEMBA). These undertakings by Government will in a way assist in management and minimisation of bush encroaching species and alien plants such as *Prosopis glandulosa*.

1.1.9 Importance of remote sensing in determining bush encroachment

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, 2006). The tool is able to detect change and can be aided 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 bio-diversity. Remote sensing will also contribute to 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 process related to those patterns (McGlynn & Okin, 2006).

The benefits of using remote sensing in detecting bush encroachment, is that high spatial resolution remote sensing enables direct imagining of plant individuals that are at least the size of the ground resolution of the remote sensing image (McGlynn & Okin, 2006). This study used various remote sensing techniques in order to determine its applicability and for verification of data regarding bush encroachment. The results with regard to the application of some various satellite remote sensing techniques, indicate positive feedback and contribution for the use in determination of bush encroachment. Although the remote sensing satellite tools have proven to be valuable, the need for ground truthing remains critical. Ground truthing remains relevant as a result of cost implication of technologies necessary for small scale and variations of individual plants. The study area bush equivalent determined through ground truthing is higher compared to remote sensing satellite image results. The results thereby necessitate ground truthing, especially when large-scale remote sensing techniques are used to determine extent of bush encroachment.

The various remote sensing image sensors are classified and analysed through the use of NDVI. NDVI is an index that provides a standardized method of comparing vegetation greenness between satellite images (Avery & Berlin, 1992). The formula to calculate NDVI is: NDVI = (near IR band - red band) / (near IR band + red band). Index values can range from -1.0 to 1.0, but vegetation values typically range between 0.1 and 0.7. Higher index values are associated with higher levels of healthy vegetation cover, whereas clouds and snow will cause index values near zero, making it appear that the vegetation is less green (Avery & Berlin, 1992). Bands from the following satellite sensors can be used to calculate NDVI:

- Landsat MSS -- bands 5 (0.6-0.7 μm) and 6 (0.7-0.8 μm) or 7 (0.8-1.1 μm); bands 2, 3 and 4, respectively, for Landsat 4 and Landsat 5
- Landsat TM -- bands 3 (0.63-0.69 μm) and 4 (0.76-0.90 μm)
- Landsat ETM -- bands 3 (0.63-0.69 μm) and 4 (0.75-0.90 μm)
- NOAA AVHRR -- bands 1 (0.58-0.68 μm) and 2 (0.72-1.0 μm)
- Terra MODIS -- bands 1 (0.62-0.67), 2 (0.841-0.876)

NDVI can be used as an indicator of relative biomass and greenness (Boone & Calvin, 2000; Chen & Brutsaert, 1998). If sufficient ground data is available, the NDVI can be used to calculate and predict primary production, dominant species, and grazing impact and stocking rates (Ricotta & Avena, 1999; Oesterheld & Dibella; 1998, Paruelo & Epsteain; 1997, Peters & Eve, 1997; Diallo & Diouf, 1991). It is also highly correlated with climatic variables, such as the El Niño Southern Oscillation (ENSO) (Li & Kafatos 2000, Boone & Calvin, 2000) and precipitation (Schmidt & Karnieli, 2000).

Simple band ratio images, while very useful, have some disadvantages. First, any sensor noise that is localized in a particular band is amplified by the ratio calculation. The use of image bands that has not been processed to remove such sensor artifacts poses challenges in image data analysis. Another difficulty lies in the range and distribution of the calculated values, which we can illustrate using the NIR / RED ratio (Avery & Berlin, 1992). Ratio values can range from decimal values less than 1.0 (for NIR less than RED) to values much greater than 1.0 (for NIR greater than RED). This range of values posed some difficulties in interpretation, scaling, and contrast enhancement for older image processing systems that operated primarily with 8-bit integer data values (Avery & Berlin, 1992). (Modern image processing allows you to work directly with the fractional ratio values in a floating - point raster format, with full access to different contrast enhancement methods).

A normalized difference index is a variant of the simple ratio calculation that avoids these problems. Corresponding cell values in the two bands are first subtracted, and this difference is then "normalized" by dividing by the sum of two brightness values. The normalization tends to reduce artifacts related to sensor noise, and most illumination effects still are removed. The most widely used example is the Normalized Difference Vegetation Index (NDVI), which is (NIR - RED) / (NIR + RED). Raw index values range from -1 to +1, and the data range is symmetrical around 0 (NIR = RED), making interpretation and scaling easy. Vegetation with a high chlorophyll activity will have high NDVI values and a low chlorophyll activity will have a low NDVI (Chen & Brutsaert, 1998).

1.2 Aims of the study

The study was undertaken to determine:

- The extent of the woody component regarding the
    - Species composition,
    - Size and height classes and
    - Density, with attention to *Prosopis* invasion.
- The bush equivalent of each woody species height class and total species bush equivalent in all height classes.
- The differences of bush equivalent in previously controlled plots compared with non-controlled camps.
- Trends and verify the extent of bush encroachment using different years' satellite remote sensing analysis techniques.

CHAPTER 2

STUDY AREA

2.1 Locality description

The study was undertaken on the following three farms with farm numbers and consisting subdivided farm portions in the Vryburg district of the North West Province (See Table 4.1 p.56) namely, Ventersdwaal 818 on Orsets 27° 10' 57" South and 24° 40' 56" East, Trent 1 27° 11' 18" South and 24° 41' 25" East, Trent 2 27° 11' 45" South and 24° 38' 44" East, Zamenkomst 819 on Mimosa 27° 12' 36" South and 24° 44' 41" East, Eensaam 27° 12' 51" South and 24° 47' 49" East and Mooidraai 27° 35' 48" South and 24° 41' 58" East and Vlaklaagte on Werda 27° 09' 46" South and 24° 48' 57" East (see appendices A-B & Table 4.1). The farms are situated in the South West 20 km of Vryburg town confined to the plateau at 1250 m altitude. The farms are closely situated to each other in Vryburg district with similar environmental conditions such as climate, soil type and veld type. Although study area is homogenous the invasion of *Prosopis* species may differ due to soil properties, rainfall and land use.

The Vryburg town is situated in Bophirima district council of Naledi municipality. There are increasing patches of *Prosopis glandulosa* invasion on some farms. According to Versveld *et al.*, (1998) *Prosopis* species have invaded 210 000 ha in the North West Province. The lack of data regarding the extent of *Prosopis* species in the area and overall bush encroachment prompted the study. The ongoing *Prosopis* research control methods in the area also induced the need to differentiate the rate and extent of *Prosopis* invasion on controlled and non-controlled plots. The need for understanding of results after control measures were implemented on the plots also played a role in choosing the study sites.

2.2 Climate

The Vryburg town is found in the semi-arid area of South Africa. The average annual rainfall in 37 years of the study area is 325 mm and occurs mainly in summer (Appendix C). Arid areas have much less predictable and variable rainfall and plants are more opportunistic, responding closely to rainfall events (Teaque & Smit, 1992). The average minimum and maximum temperature of the study area varies between 9 °C in winter and 42 °C in summer with an average of 18 °C (Van Rooyen & Bredenkamp, 1996). Temperatures are therefore severely limiting at certain times of the year to the woody and grass components, influencing productivity and species distribution (Teague & Smit, 1992). The temperature of 18 to 20 °C shows that a long period of consistently low daily minimum temperatures during the winter provides mesquite-chilling requirement, thus allowing for early budbreak (Duke, 1983). Once the chilling requirement is met, relatively warm daily temperatures can speedup budbreak.

2.3 Geology and soil type

The geology of the area is described as Tweespruit formation consisting of sand as rock type and lithology of sandstone and tillite (State of the environment report, North West Province, 2002). The Kalahari thornveld is characterised by different types of soils such as calcareous tufa, dark brown to red sands and acids gravels, all underlain by dolomite (Van Rooyen & Bredenkamp, 1996). The soil type at the study area is dominantly of a red yellow apedal nature and the soil is mostly calcareous and non-acidic. The soil forms are Mispah and Glenrosa (Appendix D). These soils are subject to wind erosion owing to low clay percentage. The pH of 6 to 8 may contribute to early budbreak 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 area soil forms (MacVicar, 1991). The level of soil carbon concentration has an influence on the rate of soil microbial activity, e.g. the higher the carbon concentrate, the higher the microbial activity (Coetzee, 2004).

2.4 Vegetation and land use type

As mentioned, the study area is situated in the Kalahari thornveld and shrub bushveld of the savanna biome (Acocks, 1988; Van Rooyen & Bredenkamp, 1996; Appendix I). The land cover is particularly thicket and bushveld, with veld type of 16b (Acocks, 1988). Species such as *Tarchonanthus camphoratus*, *Grewia flava*, *Boscia albitrunca* and *Dichrostachys cinerea* are the principal shrubs and trees of the study area (Van Rooyen & Bredenkamp, 1996). Some of the shrub species have a short growth form and are very dense. The Kalahari thornveld and bushveld is characterised by species such as *Acacia tortilis*, *Acacia mellifera* subsp *detinens*, *Acacia haematoxylon* in sparse occurrence.

The herbaceous layer of the study site ranges from a karroid type and grasses adapted to arid environments such as *Stipagrostis uniplumis* and mesophytic conditions containing *Themeda triandra* and *Heteropogon contortus* (Acocks, 1988). The farmers livelihoods depend mainly on livestock and game farming (State of the environment report, North West Province, 2002; Low & Rebelo, 1996; Appendix K). The area is an extensive livestock area with an estimated carrying capacity of 7 LSU /ha on good condition veld (Van Rooyen & Bredenkamp, 1996). The present estimated average carrying capacity of the study area is 14 ha/LSU. This indicates continuing degradation of vegetation (Appendix H) and the concern-causing trend of a decline in the production potential of the area. The impact of previous decades as a result of various allogenic and autogenic factors operating at spatial scales and ranging from small patches to an entire landscape, influenced the degradation of the study area (Ward, 2005).

2.5 Previous control methods at the study sites

Plot 1 and 2 of the farm Orsets were treated previously as part of an ongoing research project on the farm. The trial design was undertaken in plots of 500 m^2. Tordon Super with active ingredient of picloram (120/240 g/I) at 1% and diesel was used as stump applications on Plot 2 after cutting of *Prosopis* plants during March 2000 with a 90% success rate (Coetzee, 2004). The mixture was applied immediately on the cut stumps in order to improve chemical penetration in the plants (Meyer, 2000). The aerial chemical control was carried out on the Plot (1) four years ago (2001) to control *Prosopis glandulosa*. Aerial application of Molopo GG with active ingredient of tebuthiuron (200 g/kg) in three treatments with different concentrations of soil-applied chemicals was carried out in 0.75 ha transects. The different aerial soil applied treatments were Molopo GG with active ingredient of tebuthiuron (200 g/kg) at 5.0 kg/ha, 7.5 kg/ha and 10.0 kg/ha in October 2001.

The Working for Water program of Department of Water Affairs and Forestry to control *Prosopis glandulosa* started with follow-up treatments of bush encroachment in 2005 after the transects to determine the extent of bush encroachment with focus to honey mesquite (*Prosopis glandulosa*) were carried out in the study area. Results as discussed later depicted through remote sensing SPOT 4 image however show how encroached areas have been controlled (Figure 4.19)

CHAPTER 3

RESEARCH METHODOLOGY

3.1 Types of research methods applied in the study

3.1.1 Vegetation sampling

The degree of bush density could have a vital influence on the grass cover with regard to the shading effects and competition for soil moisture, which indicate the condition of the rangeland (Smit, 2004). As already mentioned the increase of the woody component to above 2500 BE/ha has a negative impact on grazing capacity of the savanna (Meyer & Richter, 2001). Changes in species composition and reductions in plant basal area, above and belowground biomass, are known to accompany livestock grazing and bush establishment (Brown & Archer, 1999).

A bush survey of twenty samples was conducted in the different camps on the farms of Orsets, Werda, Trent 1, Trent 2, Mimosa, Mooidraai and Eensaam (Appendix B). The woody species were recorded within the belt transect of 100 m length and within 2 m of each side of the transect representing a total area of (400 m^2). A rod of 2 m was used horizontally at right angles at each side of transect. All woody plants that were noted within the sample area of 400 m^2 of transect area were recorded.

The methodology was followed to determine the species type and the density of all woody plants. The bush density was determined by bush counts and expressed in terms of bush equivalents per hectare, where a bush equivalent is defined as a woody plant with a height of 1.5 m (Tainton, 1999). The density was established at different height classes namely >0. 5, 0. 5-1 m, 1-2 m, 2-3 m, 3-4 m and >4 m.

The uses of change in tree and shrub density, size (height or volume), species composition or the degree of browse available are appropriate applicable parameters in identifying change in the woody layer (Hardy *et al.,* 1999). The identification of a tree takes consideration of the leaf, the flower, the fruit, bark and the overall shape of the tree. The shape and height of the tree is of paramount importance in determination of bush encroachment (Hurt & Tainton, 1999).

The density of a species is defined as the number of individuals per unit area in relation to other species occurring at the given site (Van Vuuren, 2004). The bush equivalent is determined per hectare per structure class to obtain the density of the woody species per hectare. The understanding of bush equivalent will help in decision making about the veld management that should be implemented.

During the study, only monitoring of the woody component was undertaken in order to determine, especially the extent of *Prosopis glandulosa* in the study area. As mentioned, the study area was very homogeneous regarding environmental factors. The environmental factors for the total area such as climate and soil were discussed previously. Environmental factors for each site were therefore not separately distinguished and studied. The follow-up study that aims to understand the interactions and influence of environmental factors on the woody component will require thorough data collection of such environmental factors. The surveys were carried out in February and March 2004 on previously controlled and non-controlled farms.

3.1.2 Remote sensing satellite image

This study used medium resolution Landsat and SPOT data for the woody density assessment. Unfortunately, there is no suitable archive of high-resolution data available for the assessment period for this study. The only option for high-resolution data would be the historical black and white aerial photo graphs taken by the Directorate of Surveys and Mapping for cadastral mapping. To acquire and assess new high-resolution data would also be too expensive for this particular study. The advantage of using medium resolution image data is the cost effectiveness of assessing large size of land such as 20 727 hectares.

The study area and individual field survey transects are shown in Figure 3.1.

Figure 3.1: Cadastral data and sampling transects overlay on Landsat 1991 image. (See Table 4.1 for the names of the 20 sites on Figure 3.1 sampled on the 7 farms in the study area).

The study area is approximately 25 km South West of Vryburg in the North-West Province. The 1:50 000 topo-cadastral data (semi-transparent raster map) is overlaid on a Landsat image of 1991.

Ortho-rectified Landsat TM and ETM scenes have been acquired for 1991 and 2001. SPOT scenes of 2001 and 2005 have been precision registered on the Landsat ETM data. Field sampling transects were plotted on the cadastral and Landsat data. The spatial resolution of the Landsat multi-spectral bands is 28.5m and for the SPOT data it is 20m.

The used remote sensing methods in the study area and their dates are as follows:

- Landsat TM – 1991/03/04
- SPOT 2 – 2001/03/27
- Landsat ETM – 2001/05/26
- SPOT 4 – 2005/03/26

The Landsat 7 satellite, launched in 1999, uses Enhanced Thematic Mapper Plus (ETM+) to observe the Earth. The capabilities new to Landsat 7 include the following:

- 15m spatial resolution panchromatic band
- 5% radiometric calibration with full aperture
- 60m spatial resolution thermal IR channel

The primary receiving station for Landsat 7 data is located in Sioux Falls, South Dakota at the USGS EROS Data Center (EDC). ETM+ data is transmitted using X-band direct downlink at a rate of 150 Mbps. Landsat 7 is capable of capturing scenes without cloud obstruction, and the receiving stations can obtain this data in real time using the Xband. Stations located around the globe, however, are only able to receive data for the portion of the ETM+ ground track where the satellite can be seen by the receiving station (Boone & Calvin, 2000). One type of data available from Landsat 7 is browse data.

Browse data is "a lower resolution image for determining image location, quality and information content." The other type of data is metadata, which is "descriptive information on the image." This information is available via the internet within 24 hours of being received by the primary ground station. Moreover, EDC processes the data to Level 0r. This data has been corrected for scan direction and band alignment errors only. Level 1G data, which is corrected, is also available. Landsat 7 has a swath width of 185 kilometers. The repeat coverage interval is 16 days, or 233 orbits. The satellite orbits the Earth at 705 kilometers (Li & Kafatos, 2000).

Information about the spectral range and ground resolution of the bands of the Landsat 7 satellite is provided in the following table:

Table 3.1: Landsat 7 characteristics

Band number	Spectral range in microns	Ground resolution (m)
1	.45 to .515	30
2	.525 to .605	30
3	.63 to .690	30
4	.75 to .90	30
5	1.55 to 1.75	30
6	10.40 to 12.5	60
7	2.09 to 2.35	30
Pan	.52 to .90	15

The SPOT satellite as used in the study was first developed by the French Centre National d'Etudes Spatiales (CNES), launched in early 1986. The second SPOT satellite was launched in 1990 and the third in 1993. The SPOT 4 satellite was launched in 1998. SPOT is commonly referred to as a pushbroom scanner meaning that all scanning parts are fixed, and scanning is accomplished by the forward motion of the scanner. SPOT pushes 3000/6000 sensors along its orbit. This is different from Landsat which scans with 16 detectors perpendicular to its orbit. The SPOT satellite can observe the same area on the globe once every 26 days. The SPOT scanner normally produces nadir views, but it does have off-nadir viewing capability. Off-nadir refers to any point that is not directly beneath the detectors, but off to an angle. The off-nadir capability ensures one area on the Earth to be viewed as often as every 3 days. This off-nadir viewing can be programmed from the ground control station, and is quite useful for collecting data in a region not directly in the path of the scanner or in the event of a natural or man-made disaster, where timeliness of data acquisition is crucial. It is also very useful in collecting stereo data from which elevation data can be extracted (Li & Kafatos, 2000).

SPOT 4 carries High Resolution Visible Infrared (HR VIR) instruments that obtain information in the visible and near-infrared spectral bands. The SPOT 4 satellite orbits the Earth at 822 km at the Equator. The SPOT 4 satellite has two sensors on board: a multispectral sensor, and a panchromatic sensor. The multispectral scanner has a pixel size of 20 x 20 m, and a swath width of 60 km. The panchromatic scanner has a pixel size of 10 x 10 m, and a swath width of 60 km (Erdas field guide, 1999).

3.2 Data analysis

3.2.1 Ground truthing bush survey analysis

The process of data analysis started by adding up all the individual bushes to determine the number in every height class for all species encountered. The data was analysed using bush equivalent factors at different height class by multiplying the number of individuals recorded by the factor for individual height classes. The bush equivalent is used to express the tree population of different sites in a common currency (Tainton, 1999). The bush equivalent factor of f=0.33 was used for species at heights of less than 0.5 and factor of f=0.50 was used for species height at 0.5 – 1.0 m. The bush factor of f=1.00 was used for species height of 1.0 – 2.0 m, bush factor of f=1.67 used for species at heights of 2. 0 – 3.0 m, f=2.33 used for bush heights of 3.0 – 4.0 and f=3.00 used for bush heights of more than 4 m (Table 3.2 & Appendix G), (Tainton, 1999). The bush factors are analysis techniques used to determine bush equivalent of species height classes in hectare.

According to Tainton (1999) the term tree equivalent is widely used in South Africa to express the tree populations of different sites in a common currency. Tree equivalent is defined as a 1.5 m tall tree or shrub. Although the definition of bush equivalent is not different from tree equivalent Meyer (1999) indicates that bush density is determined by bush counts and expressed in terms of bush equivalents per hectare, where a bush equivalent is defined as a woody plant with a height of 1.5 m. The study makes use of bush equivalent for clarity and as it is easier associated with bush encroachment.

Table 3.2: An example of the calculations of the woody vegetation at the Orsets farm study site.

Species	**Height classes and factors used to calculate bush equivalent**							
	>0, 5m f =0. 33	**0, 5-1m f=0. 50**	**1-2m F=1**	**2-3m f=1. 67**	**3-4m f=2. 33**	**>4m F=3**	**Total**	**%**
Prosopis glandulosa	2. 31	3. 4	2. 4	13. 36	18. 64	33. 5	**73. 61**	**96. 8**
Acacia hebeclada	3. 3	0. 5	1. 1	0	0	0	**4. 9**	**1. 7**
Ziziphus mucronata	0	2. 5	0. 5	0	0	0	**3**	**1. 3**
Total number of BE	**5. 61**	**6. 4**	**4**	**13. 36**	**18. 64**	**33. 5**	**81. 51**	**99. 8**
% of Total	**6. 8**	**7. 8**	**4. 9**	**16**	**22. 8**	**41**		**100**

Bush equivalent is measured in trees/ha, but to compare the different species with each other, the percentage of bush equivalent of each species relative to the other species is calculated. The bush equivalent per hectare for one species is divided by the total BE/ha for all species to calculate the relative percentage of the species contribution to the BE/ha in the plot.

The density was calculated by the number of trees per belt transect (400 m^2) and converted to the number of bushes per hectare (Table 3.2). The calculation was for the total bush density for every species. 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 will assist in understanding the woody species that contribute largely to bush encroachment of the area and also in the management of dominant species. The result was used to calculate the bush equivalent of all species per hectare over the entire area (Tainton, 1999).

The results are presented in graphs that indicate the percentage of bush equivalent per species on different height classes and species, as well as total bush equivalent per hectare. The belt transect method used to record the woody component was applicable in determining the extent of *Prosopis glandulosa* encroachment.

3.2.2 Remote sensing satellite image data analysis

All the images such as landsat TM taken on 1991/03/04, Landsat ETM of 2001/05/26, SPOT 2 of 2001/03/2007 and SPOT 4 of 2005/03/26 were visually analysed using a false colour band combination that displayed actively growing vegetation as red. For the Landsat images the band combination used was 4, 5 and 3 as RGB. For the SPOT images it was 3, 2 and1 as RGB.

A Normalised Difference Vegetation Index (NDVI) was performed on all the scenes [(red-near-infrared)/(red + near-infrared)*scale factor]. The NDVI layers were then classified in 3 NDVI classes. The classifications were performed in such a way that differences in vegetation growth activity along streamlines were accentuated. The NDVI classes used were high, medium and low. The actual median NDVI value was also calculated for every field transect on every scene date. The TNT MIPS is image processing software used in the study to analyses Landsat and SPOT image. TNT MIPS was chosen because it remains one of suitable computerized system for spatial analysis (Mcglynn & Okin, 2006).

The remote sensing satellite MODIS (Moderate Resolution Imagery Spectroradiometer) results of 2003 were also compared with the information obtained from field survey. The remote sensing satellite MODIS results from map (Appendix E) and Table 4.3 were compared throughout the study. The study areas site coordinates were taken in decimal degrees and mean percent of bush encroachment given, based on the MODIS data. The mean of the study area represents the estimated average percentage of bush encroachment as undertaken by MODIS remote satellite image.

MODIS has been designed to provide a spatial resolution of 250 m, 500 m and 1 km and a swath of 23 330 km with 36 spectral channels in the 0.4 um – 1.4 um electromagnetic spectra (Zhao *et al.*, 2005). The continuous fields moderate resolution imaging pectroradiometer land cover products are 500 m-sub pixel representations of basic vegetation characteristics including trees (Hansen *et al.*, 2002). The MODIS platform completes a globe observation every one or two days and receives 6. 1 KB data from land surface, ocean and atmosphere per second during observation. The free to access MODIS data are necessary on geoscience and ecological environmental monitoring (Zhao *et al.*, 2005). One of the annual MODIS land cover products is the vegetation continuous fields layers. Its layers are able to include bare ground, herbaceous and tree covers such as percent evergreen, deciduous, needle leaf and broad leaf (Hansen *et al.*, 2002).

There are differences in terms of MODIS rating of the study area regarding bush encroachment and state of environment report. The state of environment report (Appendix F) regards the study area to be a high priority in terms of the map indicating the priority of the extent of bush encroachment per magisterial district of North West Province as compared to MODIS satellite focus of 10% extent of bush encroachment for the area, which is very low. The MODIS results, although very usable and informative, have been noted to be inaccurate at below 30% bush density owing to the distance of 500 m covered by one pixel (Hansen *et al.*, 2002). The study field results (ground truthing) at above 100% bush encroachment serve as evidence of this inaccuracy of satellite remote sensing to narrow-leafed plants such as *Prosopis* species (Hansen *et al.,* 2002 & Appendix E).

CHAPTER 4

RESULTS AND DISCUSSION

4.1 Introduction

The results presented and discussed in this chapter are based on information obtained from the bush survey through ground truthing and analysis of remote sensing satellite data of landsat 7 and SPOT 2, 4 images. The bush count results are presented by explaining the different species identified in the surveys as well as the density, size and bush equivalent at the different height classes of each species. Graphs are used to indicate the bush equivalent percentage contribution of the individual species at the different height classes. The results of Orsets farm are first presented because the initial survey was carried on that farm, while other farms results are presented chronologically based on the date and sampling sequence.

The following terminology is used to explain the results of the study.

- The "farm" in the study is explained in two distinct ways, i.e. as the farm name as represented in the topographical map and the farm name representing the common name that is frequently used such as Orsets, Trent, Mimosa, Eensaam, Mooidrai and Werda.
- The camps entail partitions inside the above named farms.
- Plot referred to marked areas in the camps where *Prosopis* control research activity was conducted. The latter is only relevant for the farm Orsets.

Table 4.1: The sampling framework of the study area.

Topographical farm name	**Farm common name**	**No. of camps sampled**	**No. of plots sampled**
Verstersdwaal	Orsets	1&2	1&2
	Trent 1	1,2 &3	None
	Trent 2	1,2 & 3	None
Zamenkomst	Mimosa	1,2	None
	Eensaam	1,2 & 3	None
	Mooidraai	1 & 2	None
Vlaklaagte	Werda	1,2 & 3	None

The dense thickets and control programme of *Prosopis glandulosa* followed in the site induced the decision to start previously with this farm. The survey was carried out in controlled and non-controlled camps in order to make accurate comparisons of extent of bush encroachment, effectiveness of treatments and timeliness of resprouting. The study included the determination of the bush density of the site. It also compared the bush equivalent of controlled plots with the uncontrolled camps.

The chemical and mechanical control methods were used in the controlled plot of the farm Orsets. As mentioned previously Tordon Super with active ingredient of picloram (120/240 g/I) at 1% and diesel was used as stump treatments on Plot 2 after cutting of *Prosopis* plants during March 2000 with 90% success (Coetzee, 2004). The bush equivalent, particularly of *Prosopis glandulosa* in non-controlled camps as well as in controlled plots is high, although an effort was made to deal with eradication of *Prosopis glandulosa*. The previous experience of the *Prosopis* species control carried out in controlled plots in the Orsets study sites, indicates that *Prosopis* has the ability to coppice two or even three years after initial treatment (Coetzee, 2004). The remote sensing satellite data results of visual analysis and Normalised Difference Vegetation Index will be presented in Figures 4.14 to 4.20. The remote sensing satellite data image will be presented from landsat picture of 1991, 2001 and SPOT 2 data images of 2001 and SPOT 4 of 2005. The results of the NDVI value calculation per transect will be presented by means of a table.

4.2 Farm Orsets results

The results of Orsets farm are presented by looking first at the camps sampled and then followed by samples undertaken in plots. Two camps in Orsets and two plots were sampled in order to achieve the objective of the study, namely, to determine the extent of bush encroachment through, species type, size and height classes and density and by verifying the differences of previously controlled plots and non-controlled sites.

4.2.1 Orsets camp 1

Site No. 20 (See Table 4.2)

The camp is situated in 27° 11' 16" South and 24° 42' 38" East nearby the town of Vryburg. The camp is used for livestock grazing. It has been alternated with other camps through rotational grazing. The primary objective of this rotational system is to control frequency, at which the plants are grazed, manage the intensity of grazing pressure and reducing the extent of selective grazing. The latter objective is achieved by confining a relatively large number of animals to a small proportion of the veld (Tainton *et al*,.1999). The camp is near the homestead where *Prosopis* was initially planted some years ago to provide shade. Livestock movement and human consumption of *Prosopis glandulosa* pods are related to have made major influence on the increase of the plants in this camp.

The Orsets camp 1 consists mainly of *Prosopis glandulosa* and only a few *Grewia flava* plants. The bush density for all height classes of *Prosopis* is 1 525 trees per hectare with a total bush equivalent of 89%, while *Grewia flava* consisted of 375 trees/ha with total bush equivalent of only 10% (291. 25 BE/ha). The highest bush equivalent was found in the >4 m tree height class (40%) followed by the height class of 3 to 4 m. The total bush equivalent of the site is 2788.25 BE/ha (Figure 4.1).

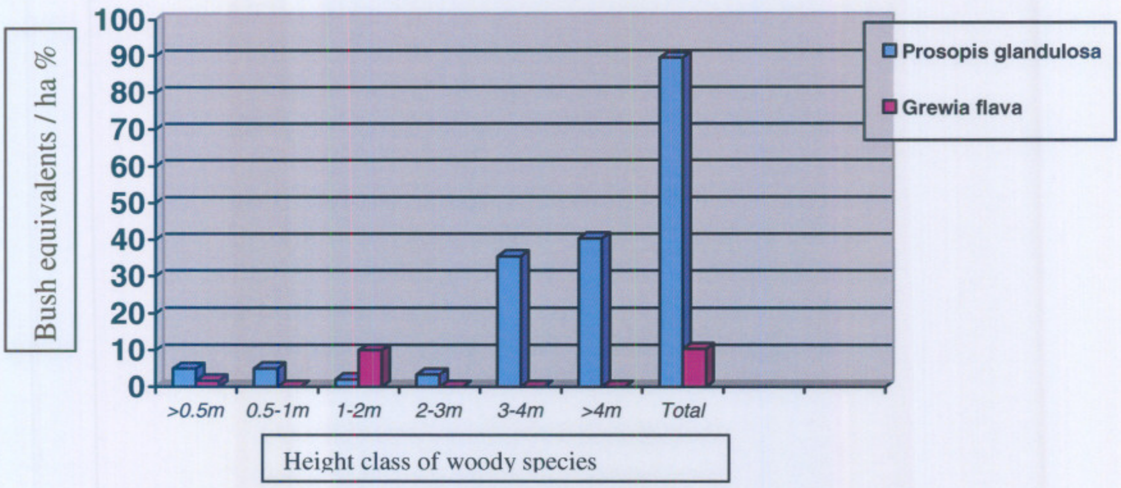

Figure 4.1: Bush equivalents (%) per height class in camp 1 on the farm Orsets.

According to Smit & Rethman (1999) and based on the rule of thumb, which indicates that if the bush equivalent is higher than double the average rainfall for the area then it may be classified as encroachment of bush. The average rainfall for the study area is 325 mm per annum, which clearly indicates that at a bush equivalent rate of 2 788.25 BE/ha, camp 1 on the farm Orsets is highly encroached by especially *Prosopis glandulosa*. The acceptable bush equivalent of the area is estimated at 650 BE/ha per hectare. Since the *Prosopis glandulosa* bush density is 2 497 BE/ha, this woody species should be cleared owing to its invasiveness and the status as declared alien weed.

Grewia flava should not be thinned because of the low level of encroachment in the study site.

4.2.2 Orsets camp 2

Site No. 9 (See Table 4.2)

The results of camp 2 indicate the distribution of three types of tree species, namely *Prosopis glandulosa*, *Acacia hebeclada* and *Ziziphus mucronata*. The *Prosopis glandulosa* plants dominate at height classes ranging from 0.5 - 1m to >4 m height classes (Figure 4.2). The total bush equivalent (BE) of *Prosopis glandulosa* in the camp per hectare is 97%, followed by *Ziziphus mucronata* at 1.7% and lastly by *Acacia hebeclada* at 1.3%.

The camp total bush equivalent is estimated at 13 430 BE/ha. Camp 2 of the farm Orsets is therefore also highly encroached owing to these high bush equivalents, which are above norms based on the average rainfall of the area.

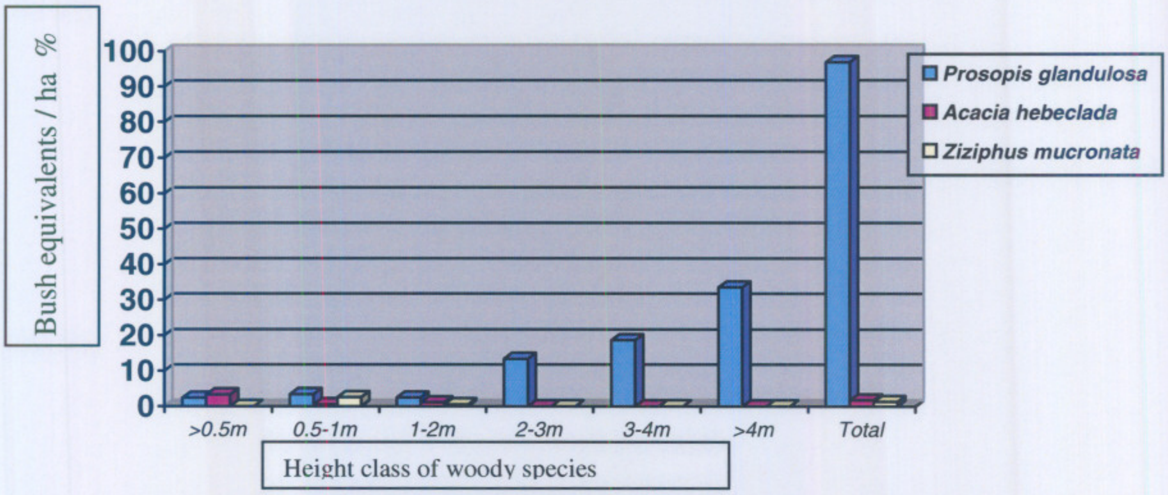

Figure 4.2: Bush equivalents (%) per height class in camp 2 on the farm Orsets.

4.2.3 Orsets Plot 1

Site No. 7 (See Table 4.2)

This plot has undergone some research trial during the past three years for testing of the aerial chemical control method. Aerial application of Molopo GG consisting of tebuthiuron as active ingredient (200 g/kg) applied in three treatments with different concentrations of soil-applied chemicals was carried out on 0.75 ha transects. The different aerial soil applied treatments were Molopo GG at 5.0 kg/ha, 7.5 kg/ha and 10.0 kg/ha in October 2001. The plant species identified during belt transect sampled for this study in March 2004 were *Prosopis glandulosa* with a total bush equivalent of 1 224.5 BE/ha (88.3%), *Grewia flava* 87.25 BE/ha (6.2%), *Acacia mellifera* 62. 25 BE/ha (4. 4 %), *Ziziphus mucronata* 12.5 BE/ha (0. 9%). *Prosopis* plants therefore dominate the bush equivalent in all height classes but mostly in the 3 to 4 m height class (Figure 4.3). The estimated total bush encroachment in Plot 1 is 2 773 BE/ha.

It seems that the chemical control methods carried out in at 2001 were unsuccessful, as these plots are still heavily encroached by *Prosopis glandulosa*.

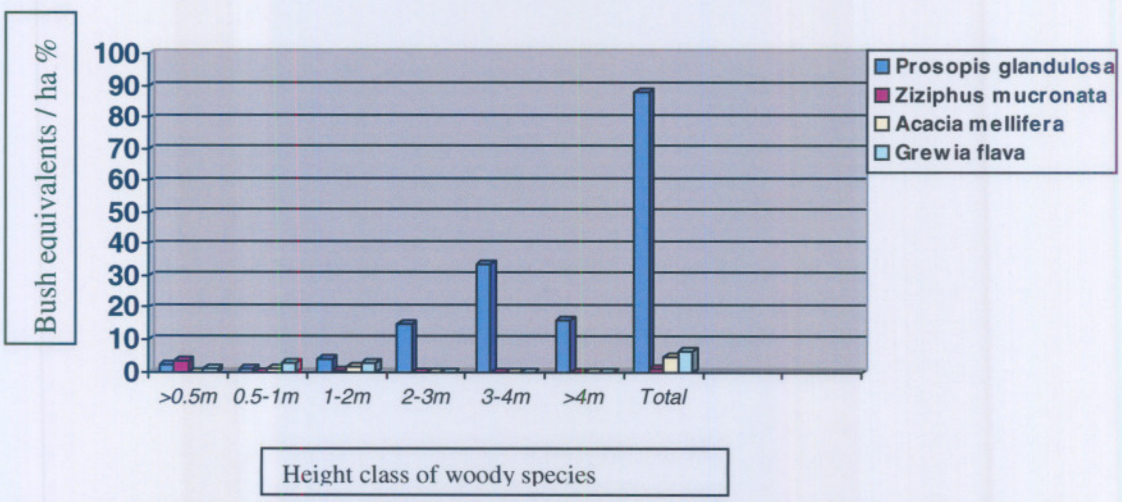

Figure 4.3: Bush equivalents (%) per height class in Plot 1 on the farm Orsets.

4.2.4 Orsets Plot 2

Site No. 8 (See Table 4.2)

This plot has undergone some research trial during past three years for testing of mechanical (stump cut) combined with the chemical control method. Tordon Super herbicides at 1% with active ingredient of picloram (120/240 g/I) and diesel was used on the stumps during March 2000. According to Coetzee (2004) 90% control success was obtained. After sampling the same plot in March 2004, the following species at different height classes were however recorded i.e. *Prosopis glandulosa*, *Acacia hebeclada* and *Ziziphus mucronata*. The total bush equivalents of identified species were as follows: *Prosopis glandulosa* (92%), *Acacia hebeclada* (3.9%) and *Ziziphus mucronta* (3.6%). *Prosopis glandulosa* is out-competing other species mostly at the 1 to 2 m height class with high bush equivalents (32.5%). The Plot 2 total bush encroachment is estimated at 2 304 BE/ha.

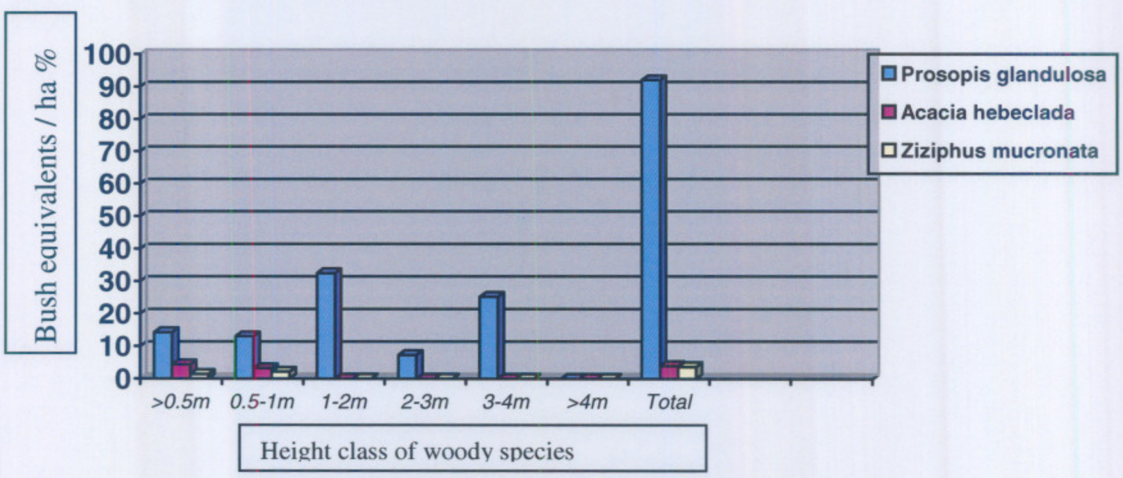

Figure 4.4: Bush equivalents (%) per height class on Plot 2 on the farm Orsets.

The results indicate the coppicing ability of *Prosopis glandulosa* on a previously controlled plot. According to Csurhes, (1996), the cutting of *Prosopis glandulosa* influences vigorously its vegetative reproduction. All *Prosopis* species are capable of regenerating from basal buds located at or just below the soil surface, when top growth is removed (Csurhes, 1996). Subsequent growth can either be expressed as single stemmed or multiple stemmed shrubs. The control methods were therefore not 100% successful, which means that follow-up procedures are necessary to prohibit the smallest trees to grow taller. Although a controlled grazing strategy is used on the farm aftercare treatments are vital for effective control of *Prosopis glandulosa.*

4.2.5 Summary of Orsets farm results

All the camps and plots on Orsets farm are dominated by *Prosopis glandulosa* species in all height classes, but mostly in the 3 to 4 m and > 4 m height class. The average bush equivalent for the farm Orsets is estimated at 5323.81 BE/ha, based on the four-bush survey samples taken on the farm Orsets during this study. The bush equivalent of this specific study site is higher than the MODIS satellite data, which places the estimate of the bush equivalent at around 15% (Table 4.3).

Plants such as *Ziziphus mucronata*, *Grewia flava*, and *Acacia mellifera* were identified to have the lowest level of bush equivalent at the Orsets study site. The aerial treated plot of Orsets is dominated solely by *Prosopis* at height classes 3 to 4 and above 4 m. This result indicates the possibility that the aerial chemical control method on higher height classes was unsuccessful. It is taken that chemical control of *Prosopis* becomes difficult at height classes above 2 m (Coetzee, 2004). The non-eradication of *Prosopis glandulosa* plants above the 2 m height class were repeatedly identified as from year 2000 till the research was carried out in 2004 on the controlled plots of the Orsets farm (Coetzee, 2004). The bush equivalent at treated plots is lower at 2773 BA/ha in aerially treated as compared to untreated camps. The MODIS satellite remote sensing data of Orsets farm is estimated at 0-5% (Appendix E) while the survey results reflect above 100% encroachment based on the rule of thumb.

4.3 Farm Trent results

Trent farm is divided into two portions, which are identified as Trent 1 and Trent 2. Three samples were taken in each of the Trent 1 and 2 farms. The results of Trent 1 will be presented first followed by Trent 2 results. Trent 1 and 2 are owned by different farmers and therefore also managed differently.

4.3.1 Farm Trent 1

Site No. 4, 5 and 6 (See Table 4.2)

On Trent farm 1, vegetation surveys were conducted at three sample sites. In one of the sample sites, up to six different species were surveyed, whereas in the remaining samples only three species were dominating, namely *Prosopis glandulosa, Grewia flava* and *Acacia karroo* (Figure 4.5).

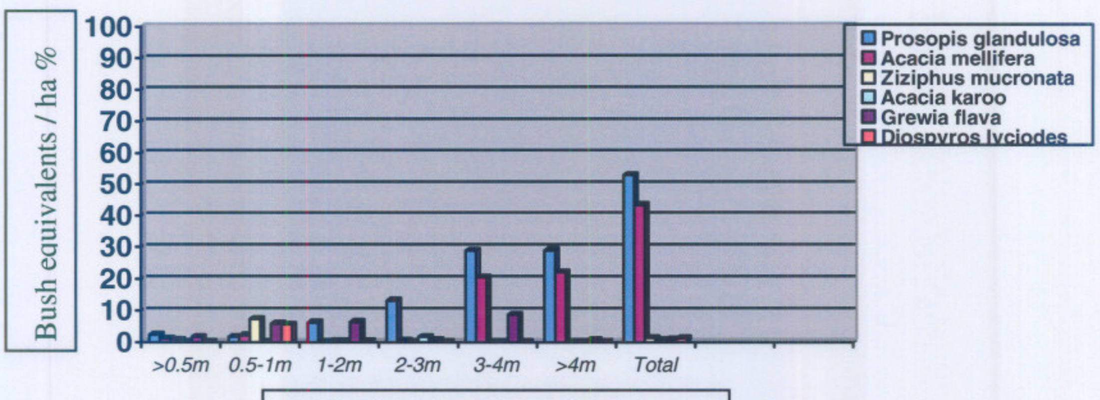

Figure 4.5: Bush equivalents (%) per height class in camp on the farm Trent 1.

The average for the three sample sites will be discussed. A variety of species were observed in Trent 1, namely *Acacia mellifera, Prosopis glandulosa, Ziziphus mucronata, Acacia karroo, Grewia flava and Diospyros lyciodes. Prosopis glandulosa* is the dominant species in the camp (52%) of the total area of bush followed by *Acacia mellifera* (43%) and *Grewia flava* (1.3%). Other species type iditified in the belt transect have very low bush equivalent rates below 1% (Figure 4.5). The study sites in Trent are highly infested with *Prosopis glandulosa,* thus justifying recommendation to control and owing to regulation 15 of CARA (Conservation of Agricultural Resources Act, No. 43 of 1983). CARA dictates that category two plants need to be eradicated, unless they are used with a permit granted for economic purposes.

The bush equivalents especially of *Prosopis glandulosa* is high in the 3 to 4 m height class followed by the height class of > 4 m. The average total bush equivalent on the farm is estimated at 12494.33 BE/ha. Based on the rule of thumb that the BE/ha must be at least equal to double the average rainfall for moisture competition. This farm is also highly encroached at a rainfall of 650 mm per annum.

4.3.2 Farm Trent 2

Site No. 1, 2 and 3 (See Table 4.2)

The three samples were taken on the farm Trent 2 in site 1, 2 and 3. In all the samples *Prosopis glandulosa* was the highest contributor to bush density and bush equivalents with a total of 82%. The highest bush equivalent of *Prosopis glandulosa* is in the 3 to 4 m height class which is then followed by >4 m with (Figure 4.6).

The lowest contribution of *Prosopis glandulosa* is at a height class of >0.5 with only 5% (Figure 4.6). Other woody plants what were recorded at this site, included, *Acacia karroo*, *Ziziphus mucronata* and *Grewia flava*).

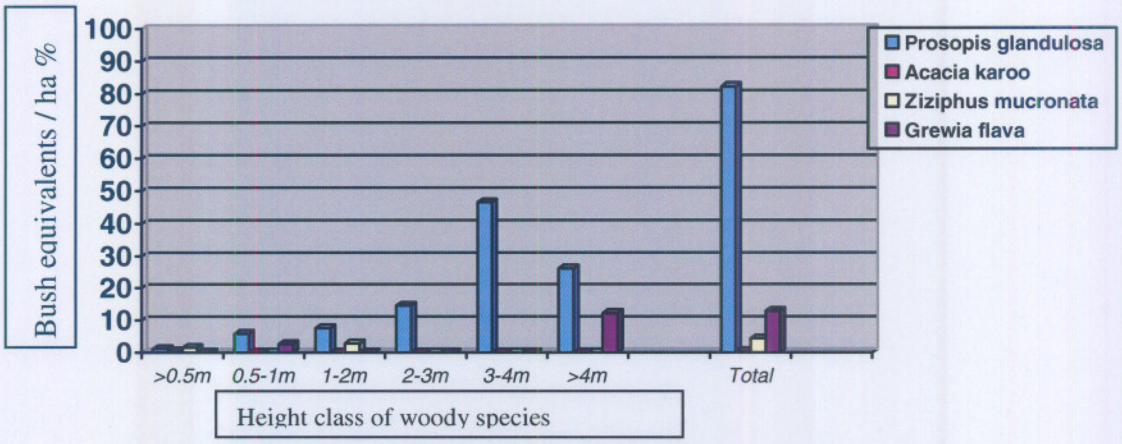

Figure 4.6: Bush equivalents (%) per height class on the farm Trent 2.

The total average bush equivalent on the farm Trent 2 is estimated at 6 680.9 BE/ha. This area is therefore also highly encroached by especially *Prosopis glandulosa*.

4.3.3 Summary of Trent farms results

The average bush equivalent of all sampled sites in Trent 1 and 2 is estimated at 9587.37 BE/ha. All of the identified woody species contribute little in bush height of <0.5 m except *Prosospis glandulosa*. *Prosopis glandulosa* is heavily dominating in all sites and remains a threat to the farm productivity. The bush equivalent of the study is over the estimated 11% of MODIS satellite remote sensing data (Table 4.3).

4.4 Farm Mimosa

Site No. 10 and 14 (See Table 4.2)

On the Mimosa farm two samples were surveyed in site 10 and 14 (Table 4.2). Woody species, such as *Prosopis glandulosa*, *Acacia mellifera*, *Grewia flava* and *Ziziphus mucronata* were identified at these sites. The bush equivalents of the species in ranking order were as follows: *Prosopis glandulosa* (87%), *Acacia mellifera* (7%), *Ziziphus mucronata* (3.5%) and *Grewia flava* (1.8%). The bush equivalent of *Prosopis* is the highest of all species (Figure 4.7). *Prosopis glandulosa* dominates in all height classes with the biggest contribution at height classes 3 to 4 m, and >4 m.

The least contribution of *Prosopis glandulosa* to bush equivalent is in the <0, 5 m height class. The study sites on Mimosa also has an increasing invasion of *Acacia mellifera*, thus causing a threat to grazing land, owing to its ability to deplete water resources (Smit & Rethman 1999).

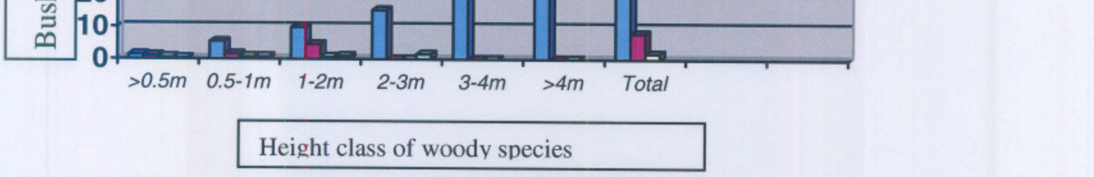

Figure 4.7: Bush equivalents (%) per height class in Mimosa farm.

The average bush equivalent in the farm is 11527 BE/ha, which is higher than the estimated MODIS satellite remote sensing data of 5% BE/ha (Table 4.3). Based on the rule of thumb that the area must at least have a number of trees equal to double the average annual rainfall of 325 mm, this camp can also be regarded as highly encroached by *Prosopis glandulosa.*

4.5 Farm Eensaam

Site No. 11, 12 and 13 (See Table 4.2)

The survey on farm Eensaam was carried out in three camps. Species such as *Prosopis glandulosa*, *Ziziphus mucronata* and *Acacia hebeclada* were identified. *Prosopis glandulosa* plants contribute 97% of the bush equivalent in the sites, while indigenous *Ziziphus mucronata* is contributing very low (2%) and *Acacia hebeclada* at below 1% (Figure 4.8).

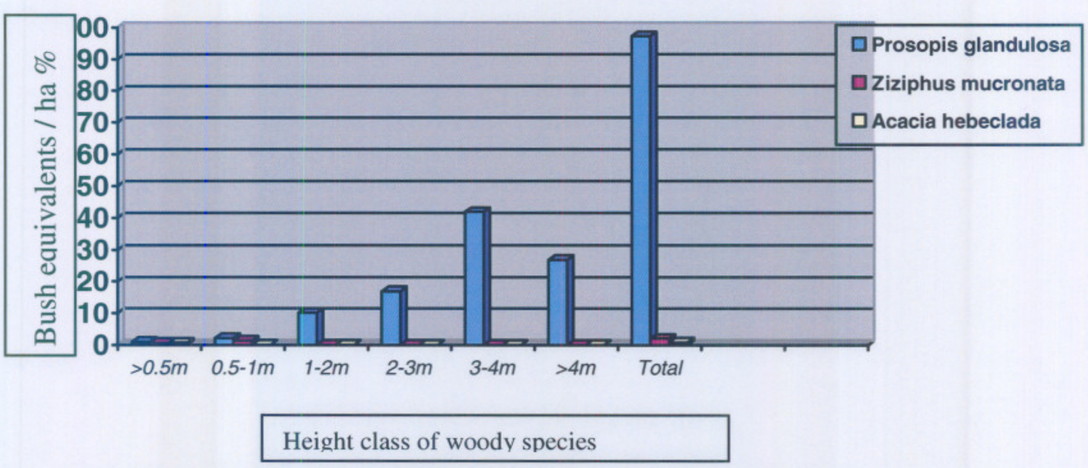

Figure 4.8: Bush equivalents (%) per height class in all camps on the farm Eensaam.

The highest bush equivalent of *Prosopis glandulosa* is in the 3 to 4 m tree height class, followed by the height class of >4 m. The average farm bush equivalent is estimated at 9 892 BE/ha more above the estimated MODIS satellite remote sensing data for the area between 11 to 20% (Table 4.3). Based on the rule of thumb that the area must at least have a number of trees equal to double the average annual rainfall of 325 mm, this camp can also be regarded as highly encroached by *Prosopis glandulosa*

4.6 Farm Mooidraai

Site No. 15 and 16 (See Table 4.2)

On the farm Mooidraai, surveys were carried out in site 15 and 16 (Table 4.2). Two species, namely *Prosopis glandulosa* and *Ziziphus mucronata* were identified on both sites of the farm Mooidraai. The dominant species in all the height classes was *Prosopis glandulosa* (Figure 4.9). *Prosopis glandulosa* contributes 99% of total bush equivalent, with only (0,8%) of *Ziziphus mucronata* (Figure 4.9).

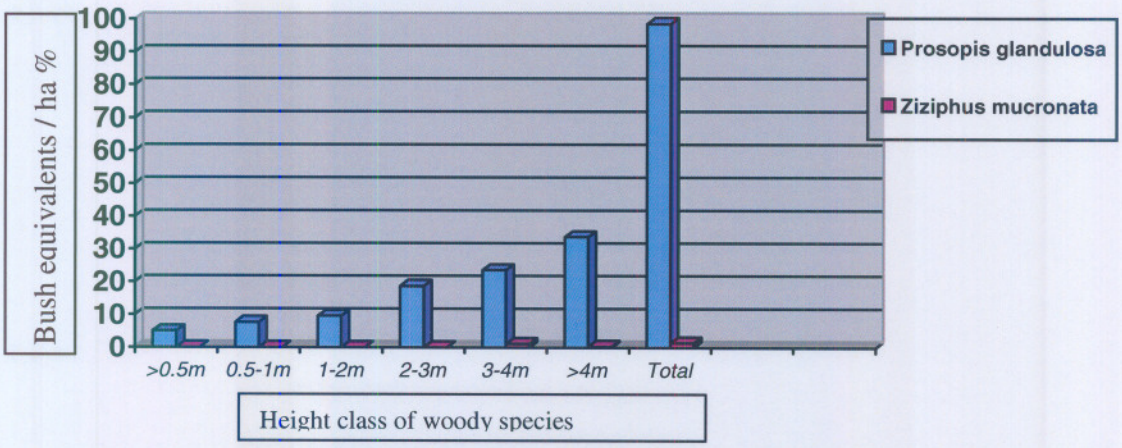

Figure 4.9: Bush equivalents (%) per height class in camps A&B on the farm Mooidraai.

The highest bush equivalent is in the class of >4 m height class, followed by 3 to 4 m tree height class (Figure 4.9). The average bush equivalent is estimated at 11064.62 BE/ha, that is above satellite MODIS data estimation of between 11 to 20% BE/ha (Table 4.3).

4.7 Farm Werda results

On the farm Werda, three camps were surveyed. The results for the three camps are discussed separately because of different utilisation rates. Rotational grazing was practised in previous years in camps 1 and 3.The watering points are located in camp 2 next to the residential houses. This camp is heavily utilised because it is adjacent to the kraal and watering points. Camp 3 was also burned accidentally during the previous season.

4.7 .1 Werda camp 1

Site No. 17 (See Table 4.2)

Camp 1 of the Werda farm consists mainly of *Prosospis glandulosa*, *Acacia karroo*, *Grewia flava* and *Ziziphus mucronata*. The *Prosopis* plants contribute 97% of the total

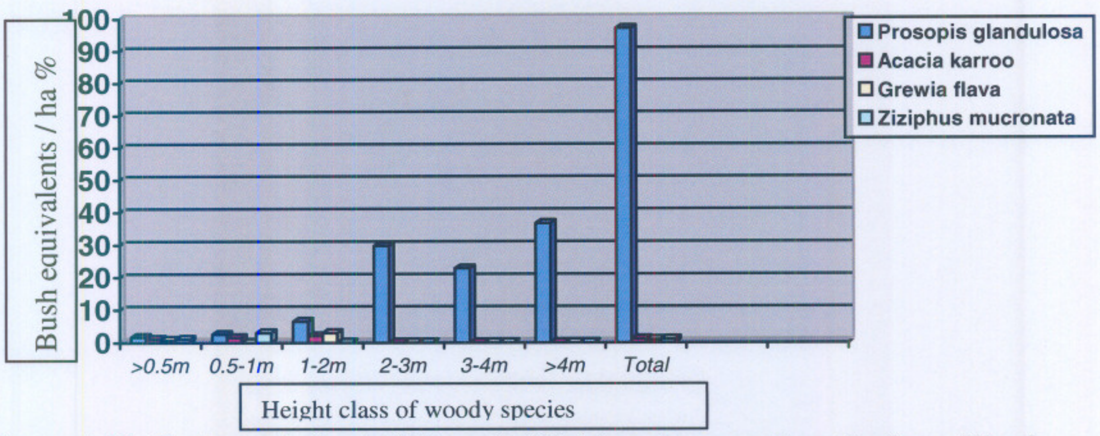

Figure 4.10: Bush equivalents (%) per height class in camp 1 on the farm Werda.

bush equivalent and bush encroachment (Figure 4.10).

The highest bush equivalent is in >4 m height class followed by the 2 to 3 m and 3 to 4 m height classes (Figure 4.10). The Werda farm camp 1 study site bush equivalent is estimated at 10 210.5 BE/ha. Based on the rule of thumb that the area must at least have a number of trees equal to double the average annual rainfall of 325 mm, this camp can also be regarded as highly encroached by *Prosopis glandulosa.*

4.7.2 Werda camp 2

Site No. 18 (See Table 4.2)

Camp 2 of the Werda farm consists of *Prosospis glandulosa*, *Acacia karroo*, *Ehretia rigida* and *Grewia flava*. *Prosopis glandulosa* dominates at 94% of the total bush equivalent. The dominance of *Prosopis glandulosa* is significant in height classes from 3 to 4 and >4 m (Figure 4.11). Based on the rule of thumb that the area must at least have a number of trees equal to double the average annual rainfall of 325 mm, this camp can

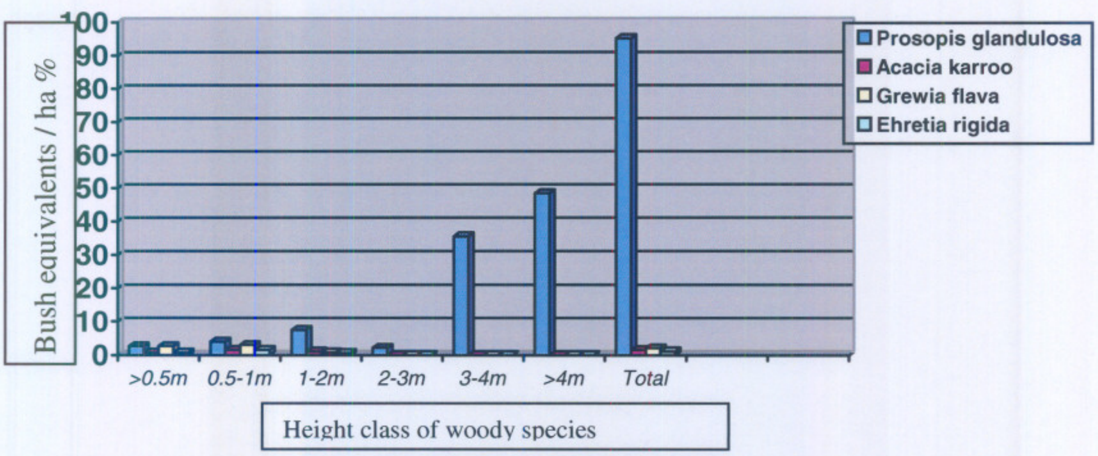

Figure 4.11: Bush equivalents (%) per height class in camp 2 on the farm Werda.

also be regarded as highly encroached by *Prosopis glandulosa.* The site has higher bush equivalent of 13 101.75 BE/ha compared to estimated MODIS satellite results of 11 to 20% bush density (Table 4.3).

4.7.3 Werda camp 3

Site No. 19 (See Table 4.2)

In camp 3 of the Werda farm, the following trees can be found, namely *Prosospis glandulosa, Acacia karroo, Ehretia rigida, Ziziphus mucronata* and *Grewia flava. Prosopis glandulosa* plants contribute 74% of the total bush equivalent, followed by 14% of *Acacia karroo.* The least contributor to bush encroachment in this camp is *Ehretia rigida* with a total bush equivalent of 1.3 % (Figure 4.12).

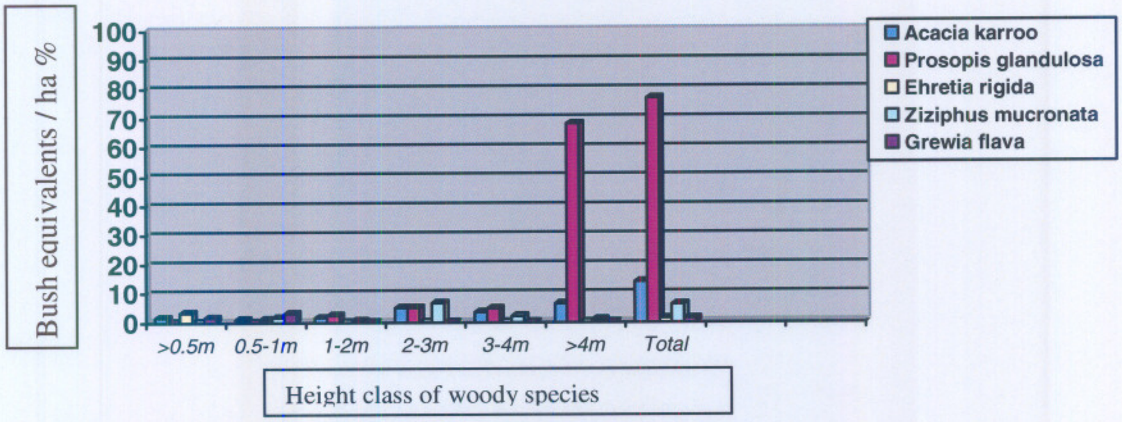

Figure 4.12: Bush equivalents (%) per height class in Werda camp 3.

The highest bush equivalent is in the >4 m tree height class with similar contributions of lower bush equivalent % in all height classes. Based on the rule of thumb that the area must at least have a number of trees equal to double the average annual rainfall of 325 mm, the camp is also taken as highly encroached by *Prosopis glandulosa*. The site has a high bush equivalent at 11 448 BE/ha above satellite MODIS results of 5 % (Table 4.3).

4.8 Remote sensing satellite image results

The results of the visual analysis of remote sensing satellite image data are displayed in Figures 4.13, 4.15, 4.17 and 4.19. The results of the NDVI classifications are displayed in Figures 4.14, 4.16, 4.18 and 4.20. The results of the NDVI value calculation per transect are shown in Table 4.2.

Table 4.2: Median NDVI values of each sampling transect with bush equivalents (BE) of all species measured in the field per sample site.

Site No	Sample of FEB and MAR 2004	Landsat TM 1991	ETM 2001	SPOT 2001	SPOT 2005
	BE per sample	Median NDVI	Median NDVI	Median NDVI	Median NDVI
1. Trent 2	4562.5	20	21	21	14
2. Trent 2	9216.25	31	18	29	23
3. Trent 2	6264	33	28	30	22
4. Trent 1	13 099	26	16	27	10
5. Trent 1	11 653	21	23	23	12
6. Trent 1	12731	14	22	21	7
7. Orsets plot 1	2773	20	19	22	10
8. Orsets plot 2	2304	20	18	26	6
9. Orsets camp 2	13 430	22	21	24	12
10.Mimosa camp 1	7 876	27	20	25	11
11.Eensaam camp 1	8 591	19	20	31	16
12.Eensaam camp 2	8 745	38	33	23	7
13.Eensaam camp 3	12 340	24	19	23	12
14. Mimosa camp 2	15 179. 25	44	29	33	17
15. Mooidraai B	6 578. 50	44	28	36	22
16. Mooidraai A	15 550. 75	54	26	28	16
17. Werda camp 1	10 210. 5	27	22	22	12
18. Werda camp 2	13 101. 75	31	25	23	10
19. Werda camp 3	11 448	24	21	23	13
20. Orsets camp 1	2 788.25	15	22	20	11

Bush control treatments were carried out since March 2000, with follow-ups in 2005 by Working for Water. The understanding is that treatments were only carried out along streamlines and low-lying areas that were encroached by *Prosopis* spp. According to the image in Figure 4.13 with dense encroachment it is correct to assume that the image was recorded before starting bush control. It is clearly possible to see a reduction of actively growing woody species in all subsequent images (Figure 4.15, 4.17 and 4.19).

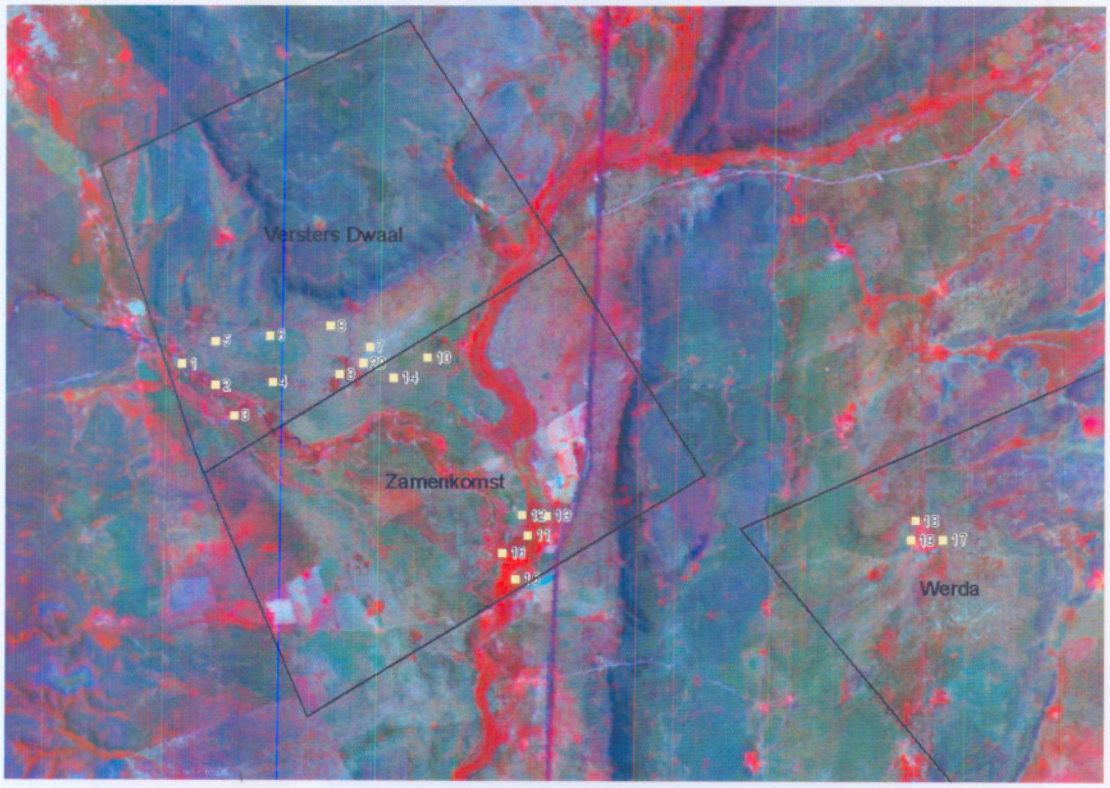

Figure 4.13: Landsat TM 1991 image (vegetation with a high growth activity is shown as red). The number of sites can each be seen in Table 4.2.

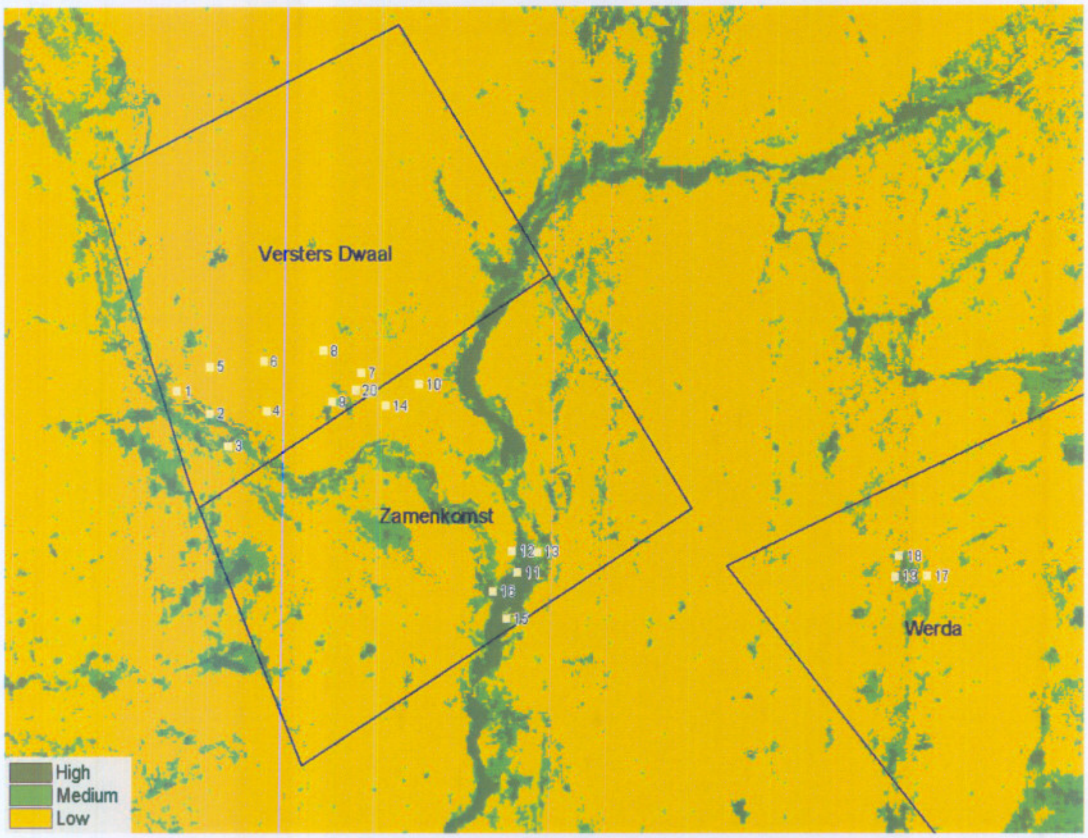

Figure 4.14: NDVI classification of Landsat TM 1991.

NDVI is a MODIS product that land research communities depend on for classification and presentation of remote sensing data images (Palmer & Fortescue, 2006). Different satellite image of the study area is presented then followed by NDVI map that is explained in Table 4.2. NDVI values are a function of chlorophyll activity and do not necessarily correspond exactly to woody density but can serve as a broad guideline (Zhao *et al.*, 2005). Seasonal and rainfall differences have also a marked influence on the NDVI value. There is a high NDVI value in Landsat TM of 1991 at most sites depicted by Dark green colour that shows dense bushes with a high chlorophyll activity especially along the streamlines (Table 4.2 & Figure 4.14).

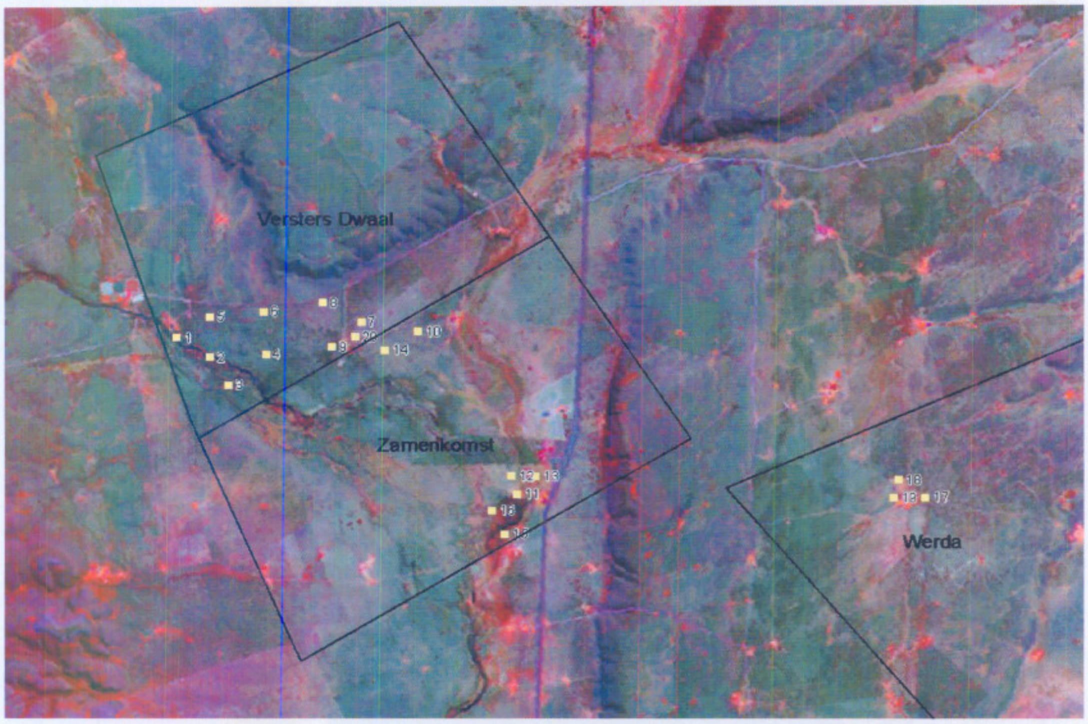

Figure 4.15: Landsat ETM 2001 image. (Vegetation with a high growth activity is shown as red.

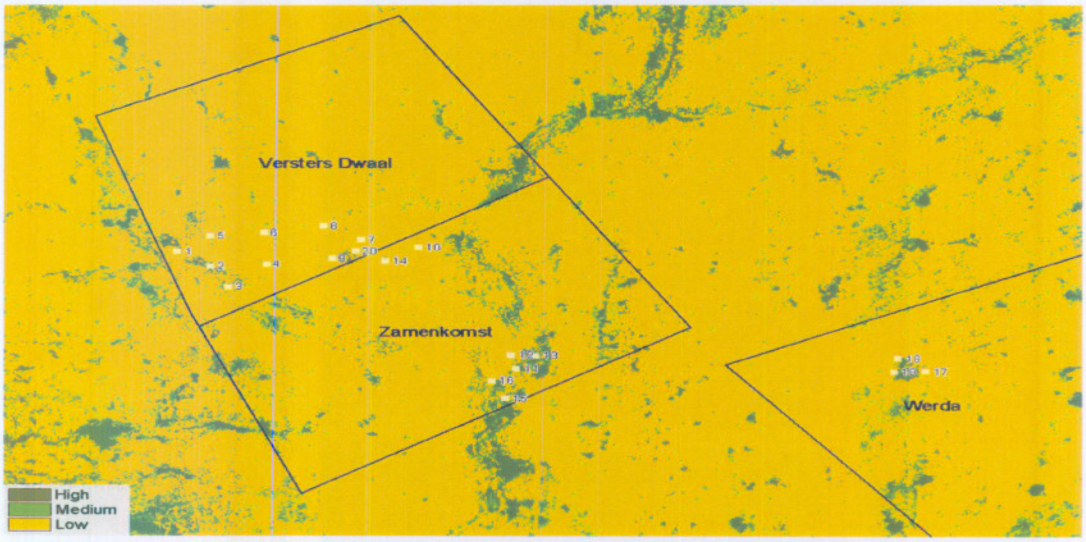

Figure 4.16: NDVI classification of Landsat ETM 2001.

Figure 4.16 of Landsat ETM 2001 shows reduction of dark green areas compared to Figure 4.14 showing less actively growing bushes than in 1991 (Table 4.2). The reduction may be attributed to bush treatments undertaken in 2000.

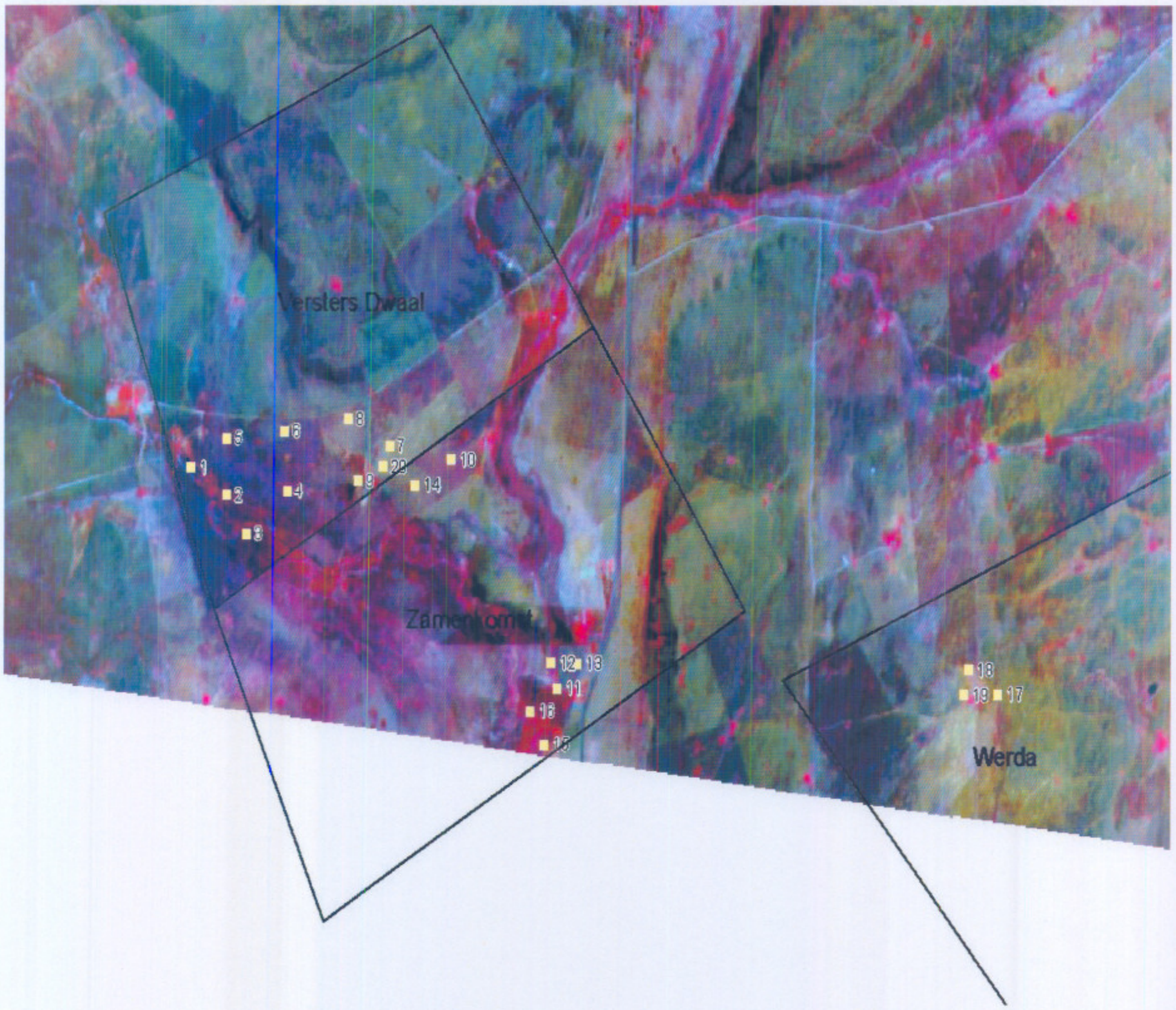

Figure 4.17: SPOT 2001 image (vegetation with a high growth activity is shown as red).

The Figure 4.18 of SPOT 2001 shows similar NDVI pattern with Landsat ETM 2001 NDVI values depicted in Figure 4.16. The time difference of images for landsat ETM 2001 and SPOT 4 image is three months. The main differences in NDVI values may be attributed to rainfall.

Figure 4.18: NDVI classification of SPOT 2001.

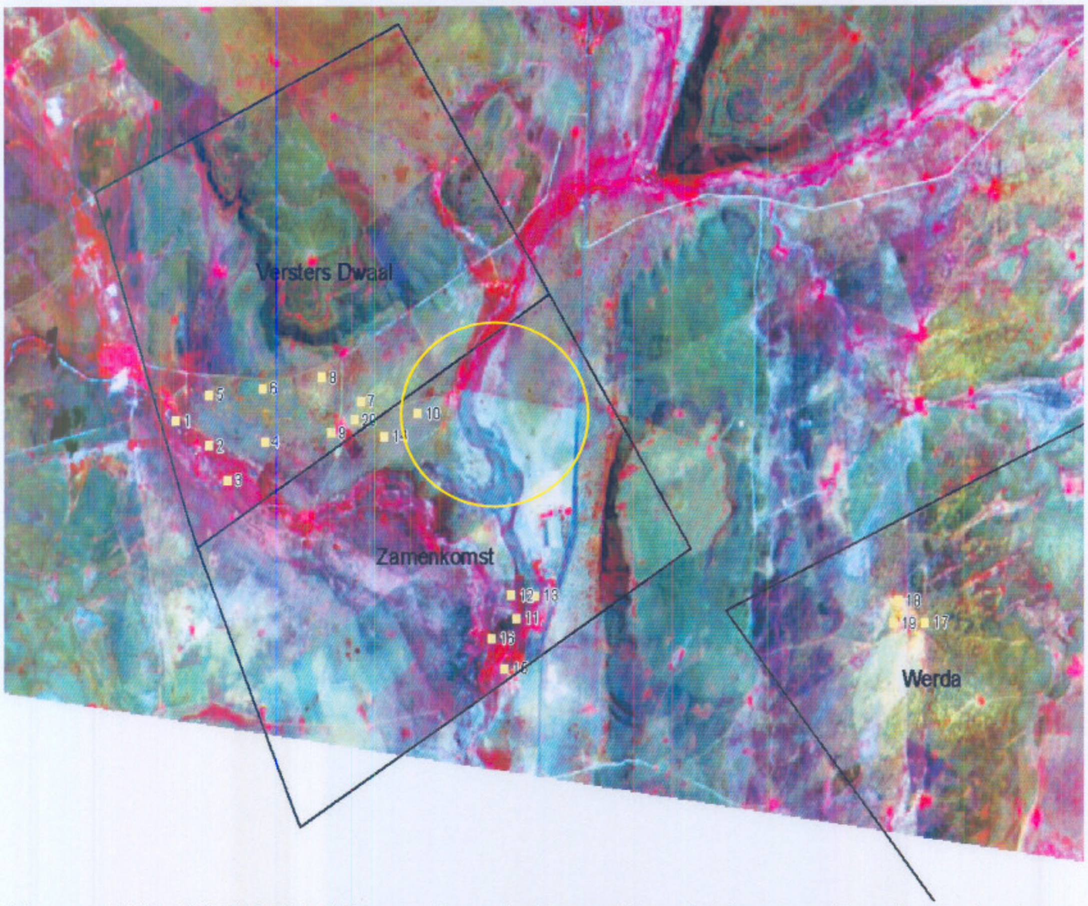

Figure 4.19: SPOT 2005 image (vegetation with a high growth activity is shown as red). See cleared area in yellow circle.

On the SPOT 2005 it is possible to observe patches where bushes are completely cleared in the steam flow area on the farm Zamenkomst in site No. 10 (Figure 4.19). Table 4.2 shows that there is also a significant reduction in NDVI values from 1991 to 2005 in all the sites. The reduction in NDVI values indicates positive trends in bush control treatments on the study area. The observed cleared area as indicated in SPOT 2005 image in Figure 4.16 and reduction of NDVI are as a result of Working for Water program that started treatments of *Prosopis* species in 2005. It is recommended that high-resolution data should also be used in areas where there is a high capital layout in terms of bush control. It is possible to register different high-resolution datasets to each other to the extent that tree on image be the same as in the field (Mcglynn & Okin, 2006). The correct identification of tree through the image with similar field observation will greatly reduce the degree of fieldwork necessary for quantitative work.

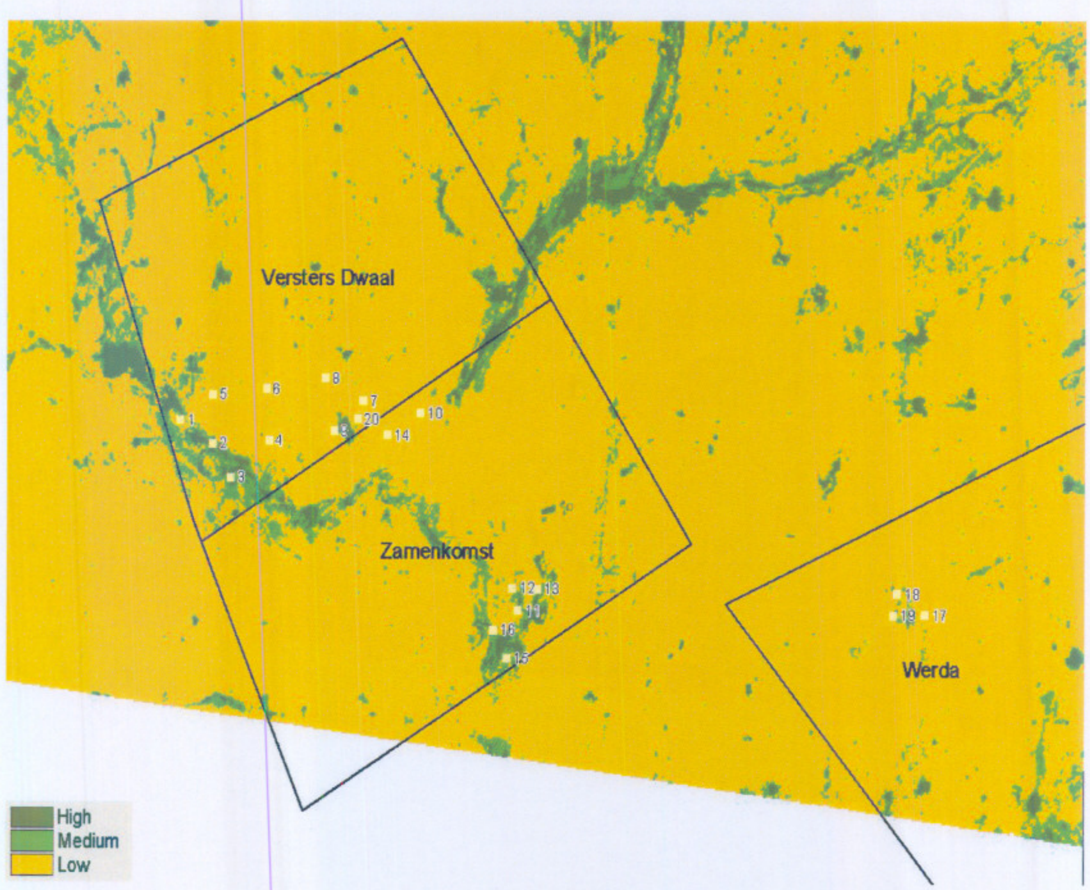

Figure 4.20: NDVI classification of SPOT 2005.

The study area tree density is estimated to be approximately 10 to 20% as derived from the MODIS satellite data launched by NASA (See appendix E). MODIS is known as the "Moderate Resolution Imaging Spectroradiometer". The MODIS result of the study area is shown below (Table 4.3). The extent of bush encroachment results of MODIS is very low compared to ground truthing results as depicted by graphs in Figure 4.1 to 4.12. The figures show high extent of bush encroachment as compared to MODIS results at high resolution for the study area.

Table 4.3: MODIS remote sensing satellite image results of study sites.

Farm name	**Area in decimal degree**	**Mean in %**
Orsets	249998.8594	15 %
Trent	249998.8594	11 %
Mimosa	249998.8594	5 %
Eensaam	249998.8594	4 %
Mooidraai	249998.8594	11%
Werda	249998.8594	5%

CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

Woody plant encroachment will always be part of savannas and management measures will not be able to eradicate woody plants completely, especially invasive alien *Prosopis* species. Invasion species is defined as a species that is non native to the ecosystem under consideration and whose introduction or colonisation causes or is likely to cause economic or environmental harm or harm to human health. Woody plant encroachment is an increase in woody plant density that results in impenetrable thickets. The highest bush equivalents, particularly of *Prosopis glandulosa* are on the farms of Orsets, Trent 1, Trent 2, Mimosa, Eensaam, Mooidraai and Werda. This concludes that the farms are highly encroached by *Prosopis glandulosa.* In all the study sites on farms where vegetation sampling was carried out, the bush equivalent was above 2 500 BE/ha, which is very high for this rainfall area. The bush equivalent of the farms is regarded to be very high as based on the rule of thumb, the bush equivalent should not exceed its equal to double the rainfall of the area which is 325 mm. The study area bush equivalent can be regarded as nearly four times higher than the norm, which indicates a high rate of bush encroachment.

The area can only accommodate about 650 bush equivalent per hectare as determined by the average rainfall of 325 mm per annum. At an encroachment of 2500 BE/ha, the high tree density will have a negative impact on the herbaceous production. *Prosopis glandulosa* was found to be dominant in most of the height classes, particularly at height classes namely, 3 to 4m and above 4 m, while density of *Acacia mellifera* was only high in two sites. The trees such as *Ehretia rigida, Acacia karoo, Diospyros lyciodes, Ziziphus mucronata, Grewia flava* and *Acacia hebeclada* were found in lower densities. The bush encroachment in the study area is also higher at above the norm in the entire site than the estimated 10 to 20% value of bush encroachment as depicted by the MODIS satellite image of 2004 (Appendix E).The bush encroachment of the study area is therefore above 100% based on the rule of thumb. The impact of 2 500 bush equivalent per hectare in rangeland is mostly its high reduction of grass production. The Plots 1 and 2 of Orsets, which were controlled four years ago, show no significant difference compared with the other non-controlled camps. The difference in the degree of *Prosopis glandulosa* invasion on the two controlled plots is insignificant.

The high invasion of *Prosopis glandulosa* in the study area is probably owing to its ability to spread easily, compete with herbaceous plants and can be ascribed to poor veld management and ineffective control methods. *Prosopis glandulosa* therefore also has a negative impact on livestock production and biodiversity. This tree also has some positive characteristics, which are often overlooked by its negative impact and its ability to spread quickly (Brown & Archer, 1999). The persistence of *Prosopis glandulosa* in the study area indicates that eradication of the plant is often not an answer also due to high input costs during control methods. Adoption of integrated management that can turn the weedy stands into profitability is emphasised.

The positive impact of the plants entails the significant advantage of *Prosopis glandulosa,* as this plant is a nitrogen fixer, which enhances the soil nutrient content (Pasiecznik, 2003). The plant remains valuable, because it has great historical precedence for providing food for humans and domestic livestock. The provision of foods is favoured as management option of *Prosopis* species, because its pods have a high sugar content of 30% and protein (12%) (Ehrhorn, 1996).

Prosopis glandulosa can also be exploited for production of flooring, fine furniture (Figure 5.1) and artisanal products, because the timber of the plants is dimensionally stable or has lower volumetric shrinkage than any other types of measured to date.

Figure 5.1: An alternative usage of *Prosopis* for immediate action to manage invasion.

Using *Prosopis glandulosa* for timber production will create an economic control strategy of the plants on affected farms rather than relying on real travesty of destruction of an entire ecosystem from the honey mesquite eradication programme implemented by means of chemicals. The commercial exploitation of *Prosopis* should also consider a variety of other uses, such as firewood, charcoal and livestock feeding (Figure. 5.2). These new initiatives provide some hope, following the failure of earlier control efforts and results of early utilisation programmes (Zimmermann & Pasiecznik, 2005).

The results of the study sites' bush equivalent at above 650 BE/ha on previously controlled plots, indicate the necessity of planned systematic after-care to deal with bush encroachment. Owing to some problems during applications, none of the chemical control options used on the research trial in the study area are 100% effective and efficient (Meyer, 2000). The follow-up treatments become necessary after 6 to12 months.

Figure 5.2: Sheep feeding on *Prosopis* pods.

To minimise the impact of *Prosopis glandulosa* a combination of control options should be integrated into the grazing management system and farm plan that includes careful control of stocking rates, strategic use of fire, biological control agents, holding of livestock for some time in the paddocks in order to allow seeds held in animals' digestive tracts to be expelled. Other methods include chemical control of highly infested and isolated areas, mechanical control removal of dense infestation and the strategic use of fire. The incentive provision for control of *Prosopis glandulosa* should be developed by National Government in order to keep the growing density at a reasonable level.

It is also concluded that MODIS satellite remote sensing data are unreliable to study sites on limited scale owing to its reflection of large scale. The uses of SPOT images provide fair analysis (trends) of vegetation growth and extent of bush density. Although the SPOT images are assisting in the determination of bush encroachment through NDVI classification, the necessity for ground truthing is unquestioned for accurate planning in terms of extent of invasion and control methods. There is a marked declining in the rate of bush encroachment, as depicted by remote sensing techniques applied in the study area after 2004 (Table 4.2). The declining bush encroachment especially after survey transects is attributed to Working for Water project that started since 2005. The undertaking of planned aftercare is mandatory to attain the required control of *Prosopis glandulosa* in the study sites.

5.2 Recommendations for future management of *Prosopis* encroachment and research projects

- The concentration should be on integrated management through farsighted and sustainable control of *Prosopis* species by combining prevention of spread, selective eradication and full exploitation of the resource.
- Future research on bush control should assess animal's management and usage, because of their immense contribution to bush encroachment and success of control.
- The assessment of area-specific environmental factors is vital for area-specific interaction and influence on bush encroachment. This will also aid ground truthing activities when using satellite images to determine trends of encroachment on a larger scale.

- It is recommended that alternative control options be used rather than the traditional approach of eradicating P*rosopis glandulosa.* The control options should incorporate chemical, mechanical and biological means as well as, proper livestock management in single strategy because total eradication of the species is costly and not simple.
- The control programme should be diverted to economic spin-off options that may be created by using the timber of *Prosopis glandulosa* plants for charcoal, furniture and fire wood.
- The initial control programme should target trees above 2m because they prohibit movement and complicate management. This means that initial control methods should be carried out in the early stages of problem and that aftercare be implemented if the trees are still small after resprouting.
- The control programme should form part of farm management with great emphases on aftercare treatment of controlled areas.
- The short-term rehabilitation strategy should be aimed at eradication through labour intensive methods such as soil-applied chemicals and or mechanical methods. This

will contribute to job creation, capacity building and livelihood improvement initiatives.

- The biological control strategy that reduces seed germination should then be incorporated in the after care of eradicated camps.
- The Government through Conservation of Agricultural Act, Act No. 43 of 1983 and National Environmental Management Biodiversity Act, Act No. 10 of 2004 should provide incentives to these affected farmers for stimulation of *Prosopis glandulosa* control and avoid further spreading of this plant to other farms.
- The influence of demographic conditions such as soil and climatic impacts, as well as land type use on spreading and control success of *Prosopis glandulosa* should be further researched.
- With regard to remote sensing analysis, it is recommended that high-resolution data should also be used in areas where there is a high capital layout in terms of bush control. It is possible to register different high-resolution datasets to each other to the extent where a tree digitized on the image will be a tree on the ground such as SPOT 5. This can greatly reduce the volume of fieldwork necessary for quantitative work.

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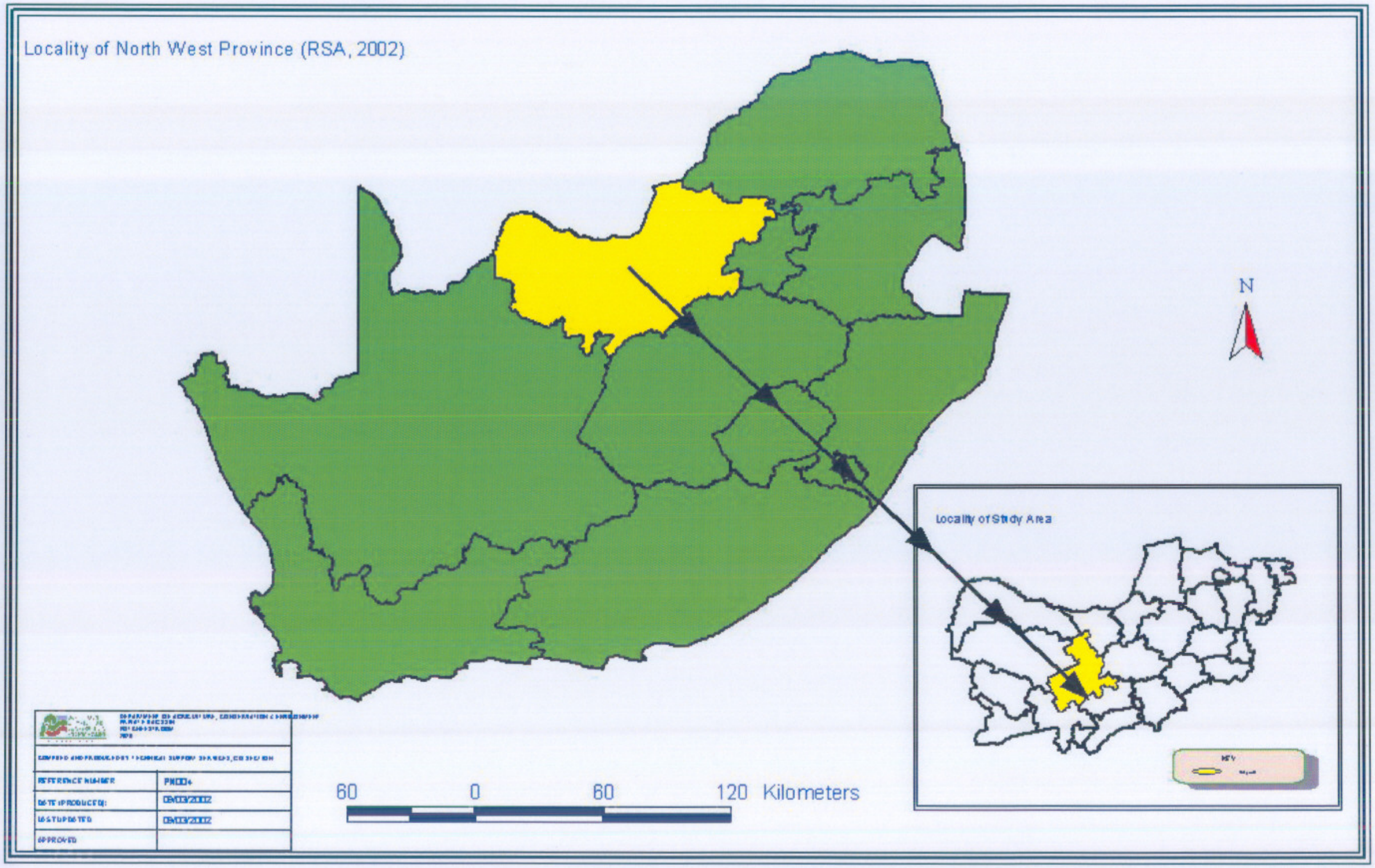
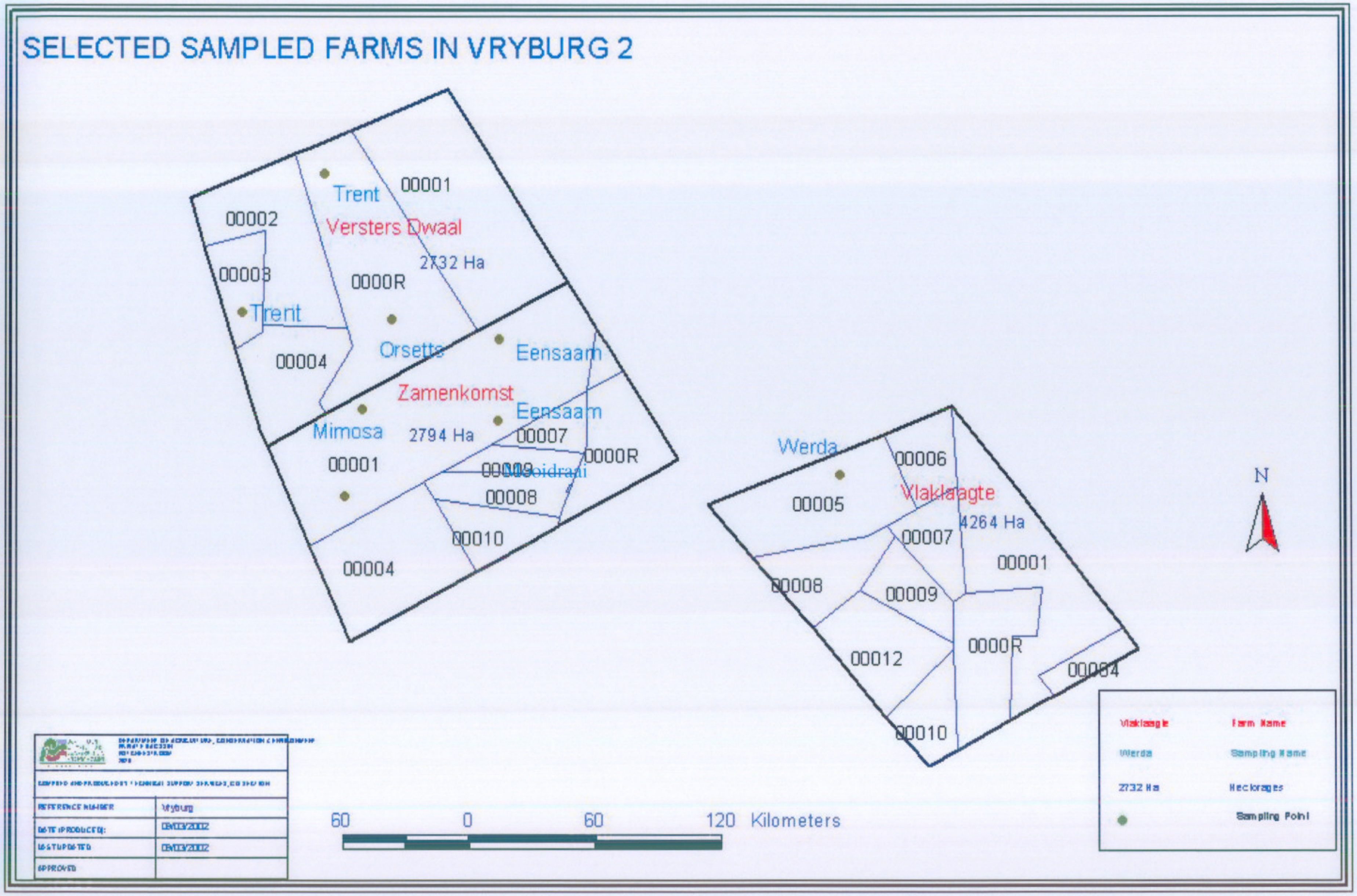
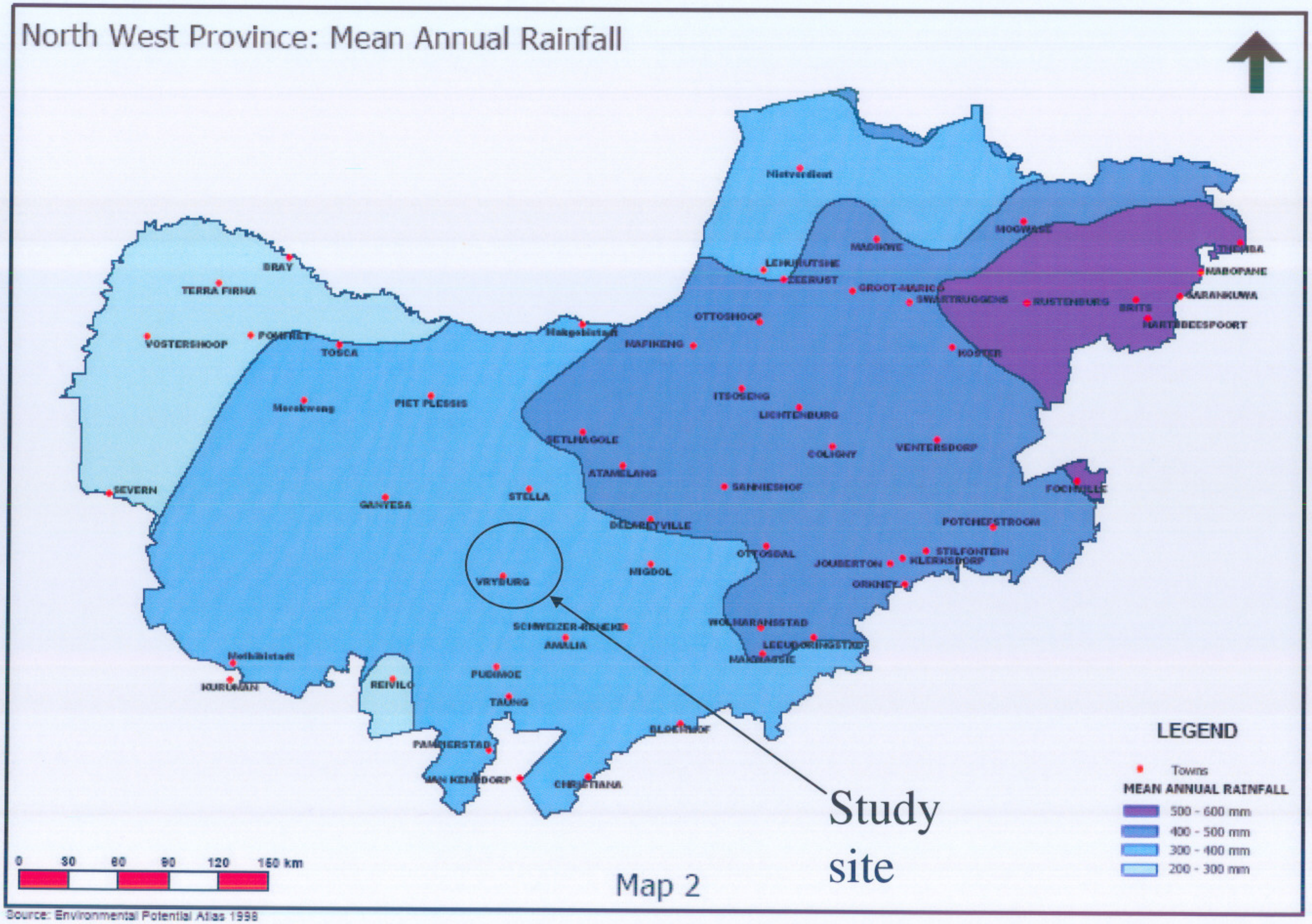
Appendix A: Locality of study area depicted inside South Africa



Appendix B: Map of study area indicating marked sample sites

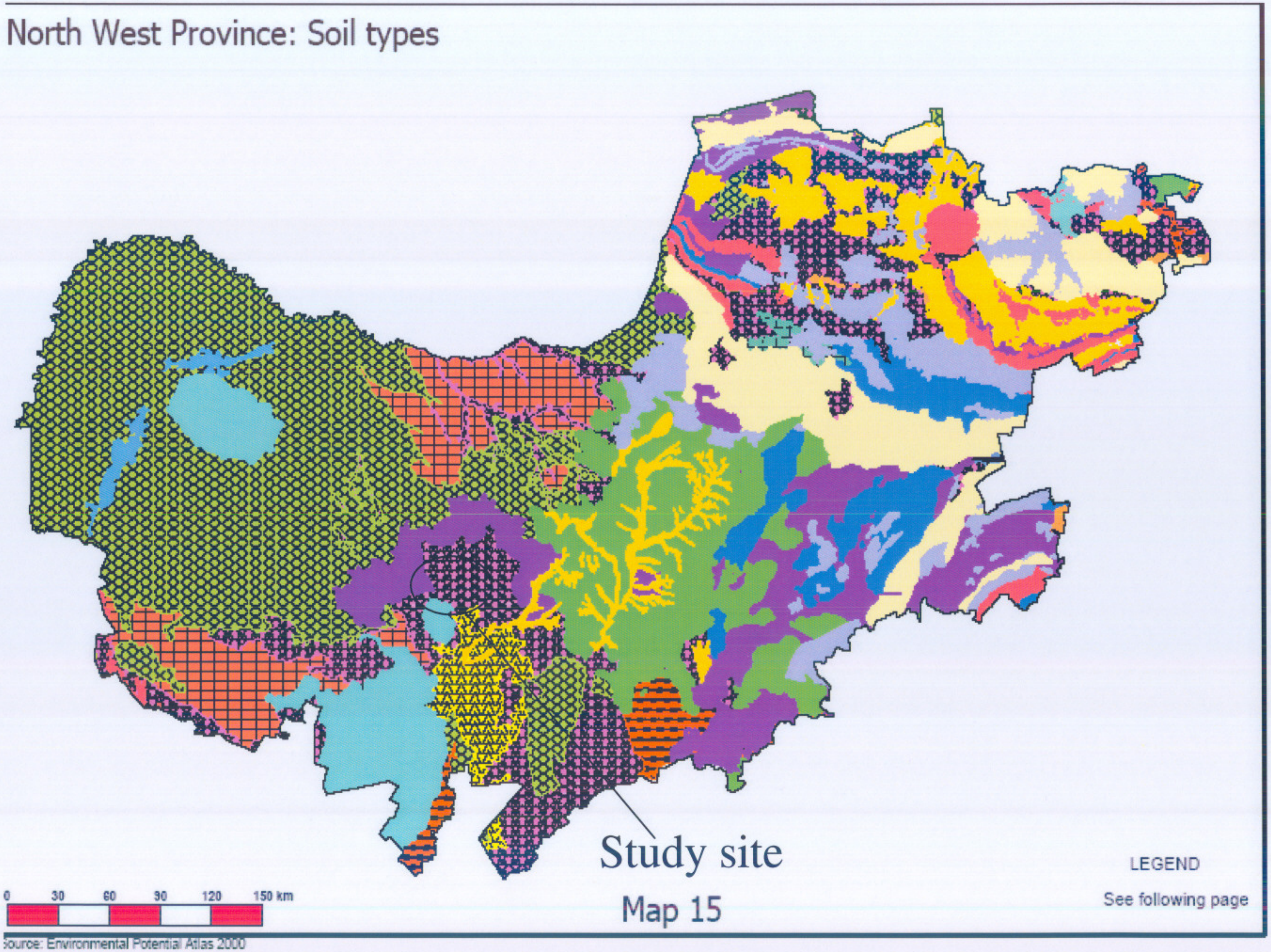


Appendix C: North West Province: Mean annual rainfall



Source: Environmental Potential Atlas 1998

Appendix D: North west province: Soil types



Map 15

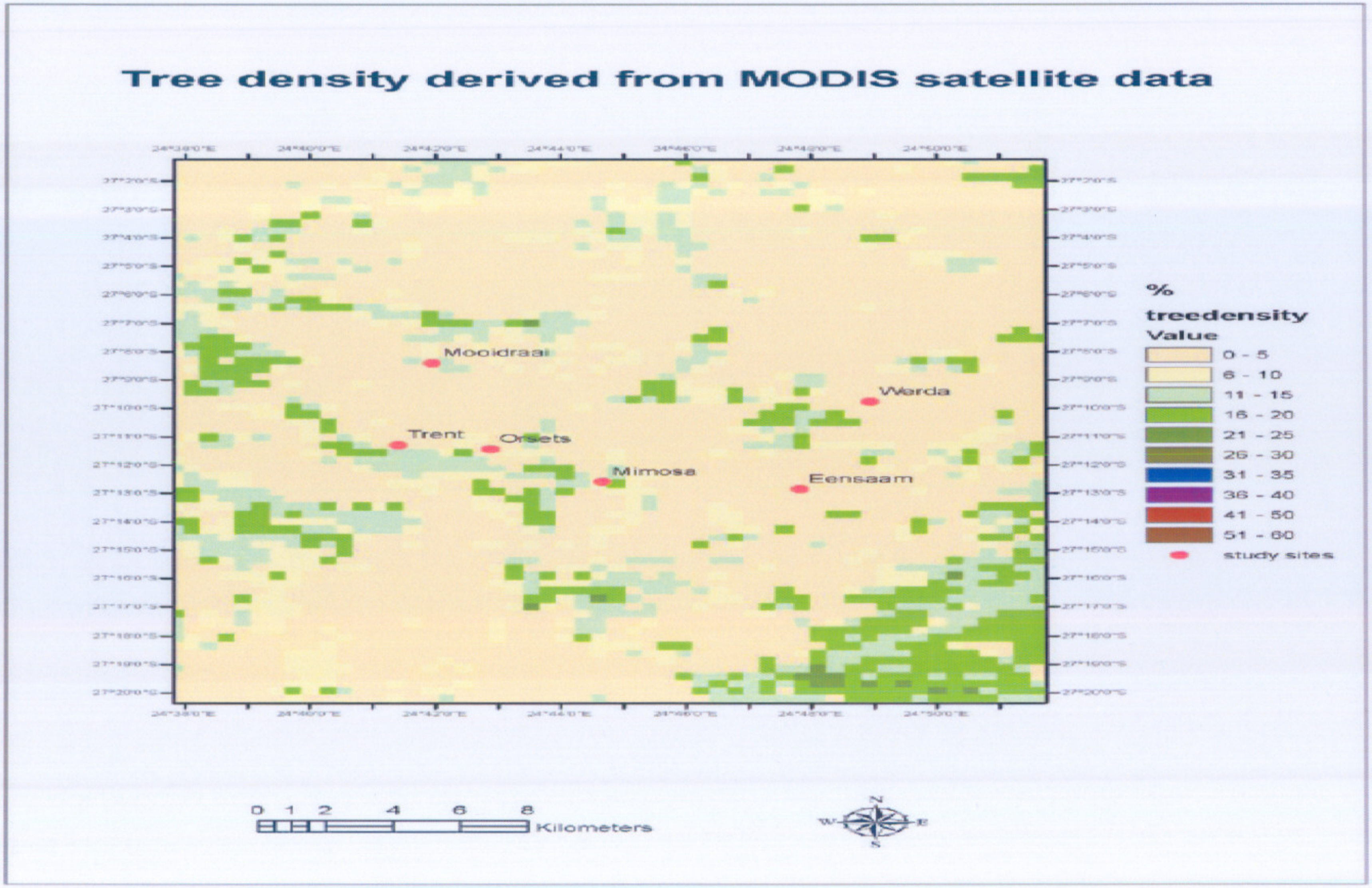
Legend of appendix D

North West Province: Soil types

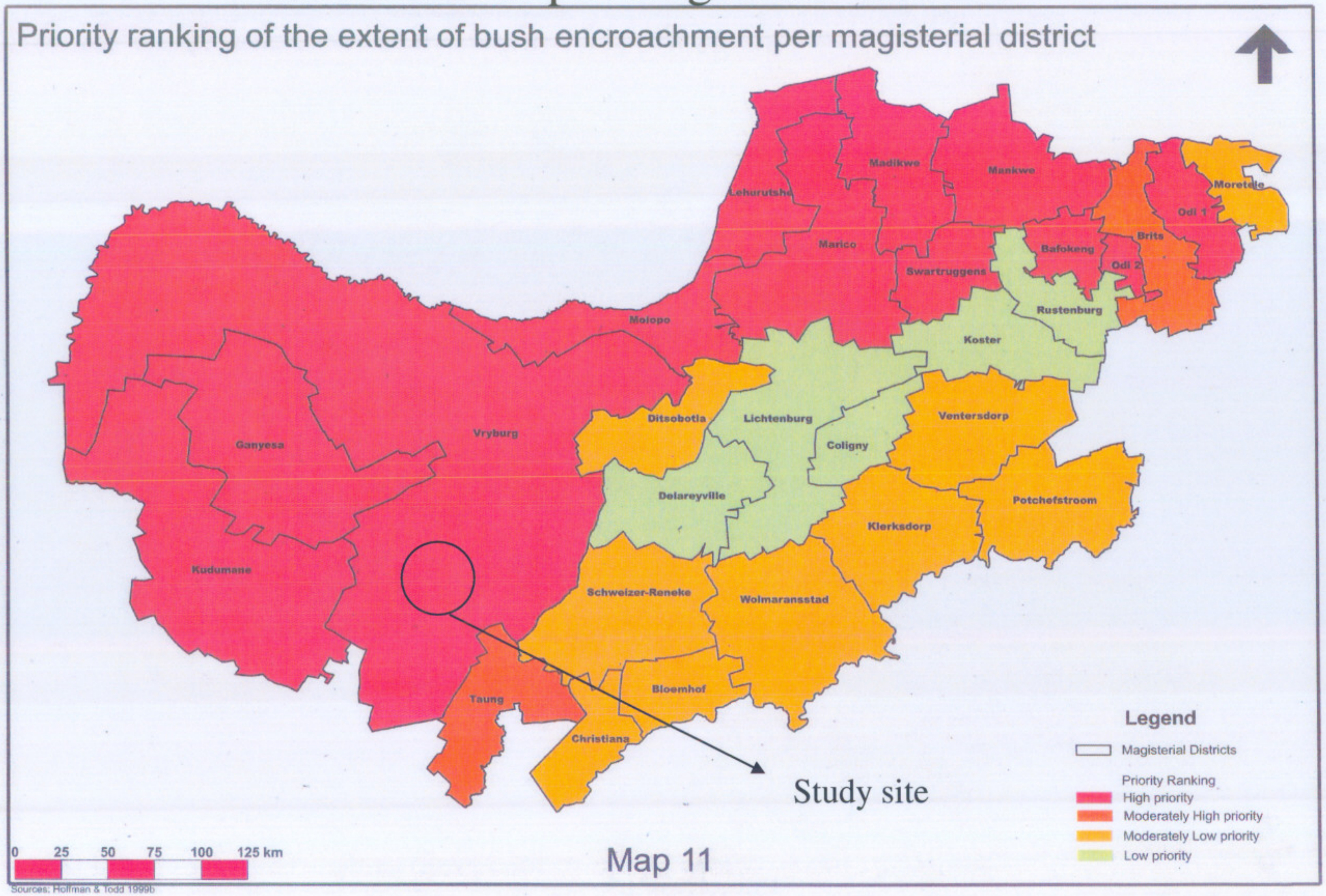
LEGEND

- Glenrosa and/or Mispah forms (other soils may occur), lime generally present in the entire landscape
- Glenrosa and/or Mispah forms (other soils may occur), lime rare or absent in the entire landscape
- Glenrosa and/or Mispah forms (other soils may occur), lime rare or absent in upland soils but generally present in low-lying soils
- Miscellaneous land classes, rocky areas with miscellaneous soils
- Miscellaneous land classes, very rocky with little or no soils
- One or more of: vertic, melanic, red structured diagnostic horizons, undifferentiated
- Plinthic catena: dystrophic and/or mesotrophic; red soils not widespread, upland duplex and margalitic soils rare
- Plinthic catena: dystrophic and/or mesotrophic; red soils widespread, upland duplex and margalitic soils rare
- Plinthic catena: eutrophic; red soils not widespread, upland duplex and margalitic soils rare
- Plinthic catena: eutrophic; red soils widespread, upland duplex and margalitic soils rare
- Prismacutanic and/or pedocutanic diagnostic horizons dominant. In addition, one or more of: vertic, melanic, red structured diagnostic horizons
- Red-yellow apedal, freely drained soils, red, high base status, < 300 mm deep
- Red-yellow apedal, freely drained soils; red and yellow, dystrophic and/or mesotrophic
- Red-yellow apedal, freely drained soils; red and yellow, high base status, usually < 15% clay
- Red-yellow apedal, freely drained soils; red, dystrophic and/or mesotrophic
- Red-yellow apedal, freely drained soils; red, high base status, > 300 mm deep (no dunes)
- Red-yellow apedal, freely drained soils; yellow, high base status, usually < 15% clay

Appendix E: Tree density derived from Modis satellite data



APPENDIX F: Priority ranking of the extent of bush encroachment per magisterial district

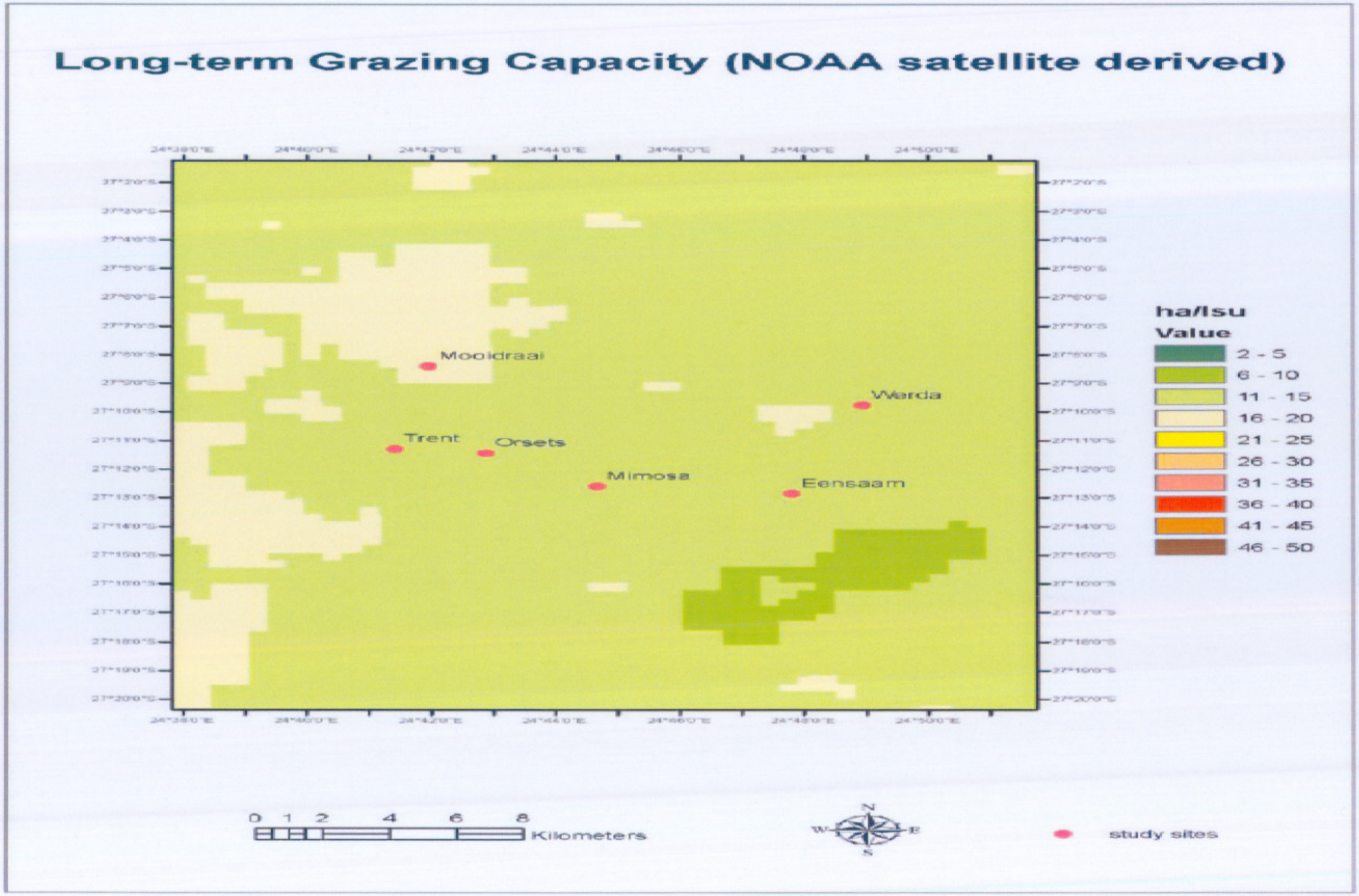


Appendix G: Bush survey field form

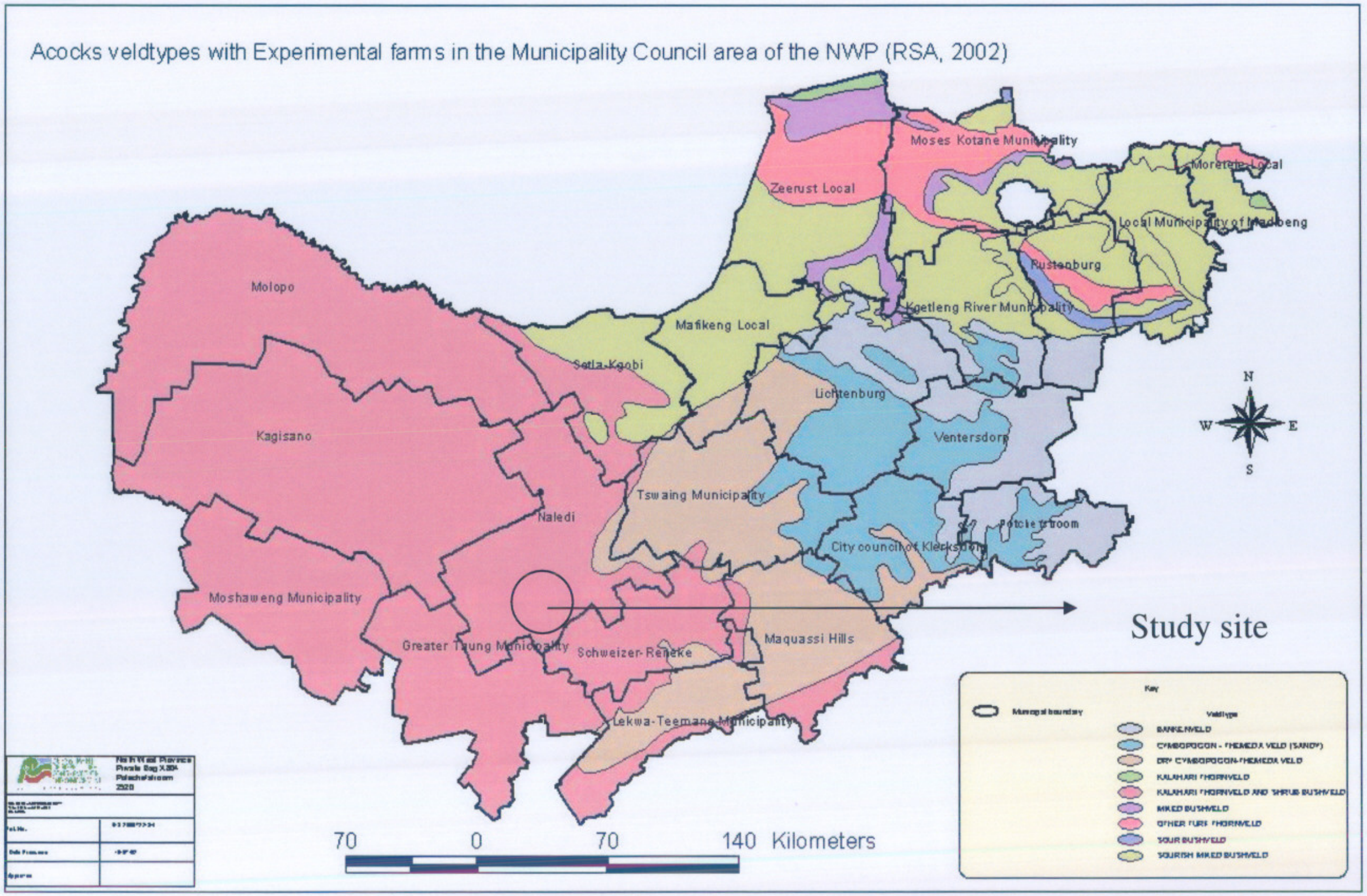
Locality………………… Plot no:……………… Date………………

Height class & BE factors	<0.5 m f=0.33	0.5-1 m f=0.50	1-2 m f=1.00	2-3 m f=1.67	3-4 m f=2.33	>4 m f=3.00	Total
Species							

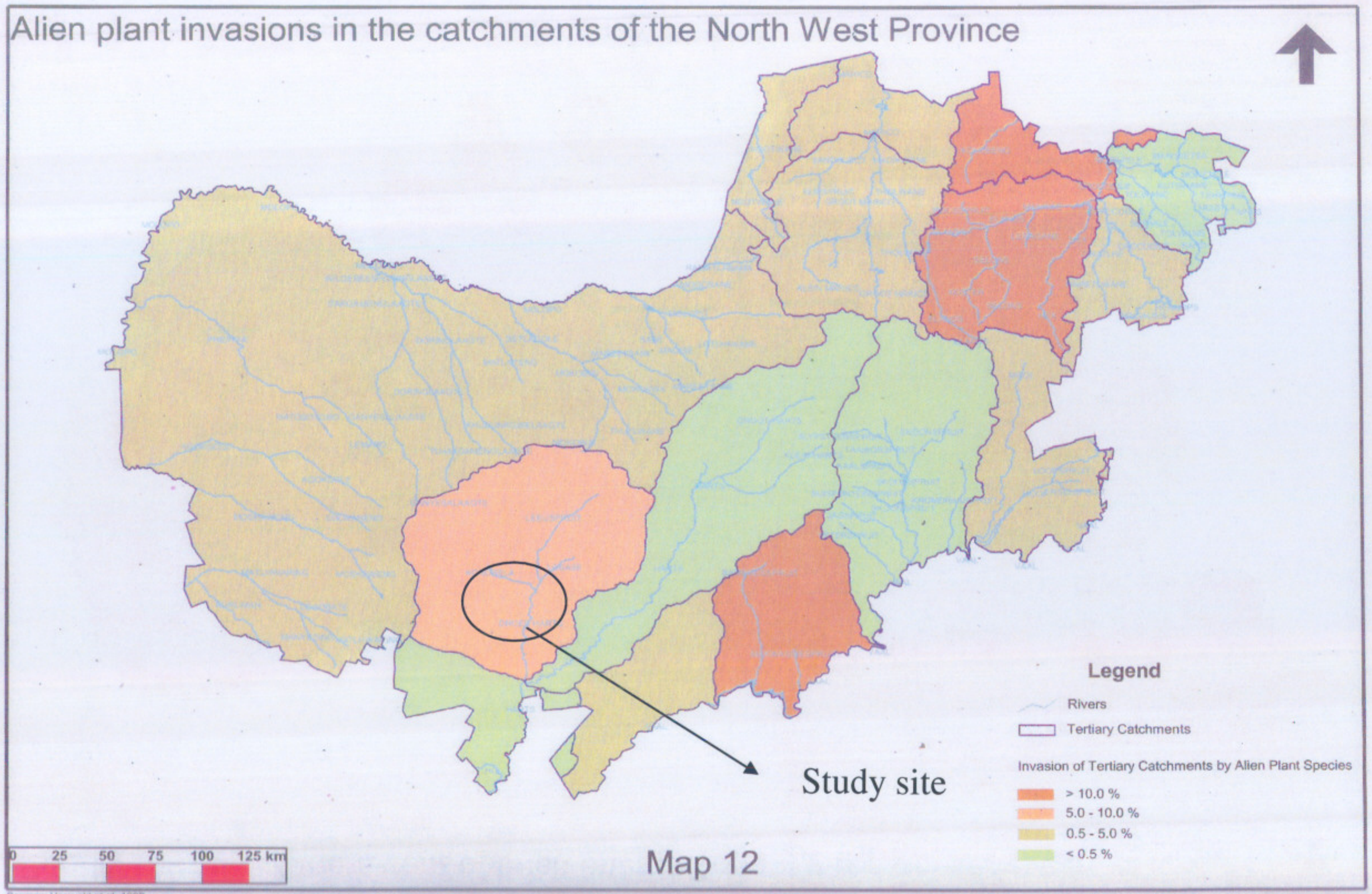
Appendix H: Long-term grazing capacity (NOAA Satellite derived)



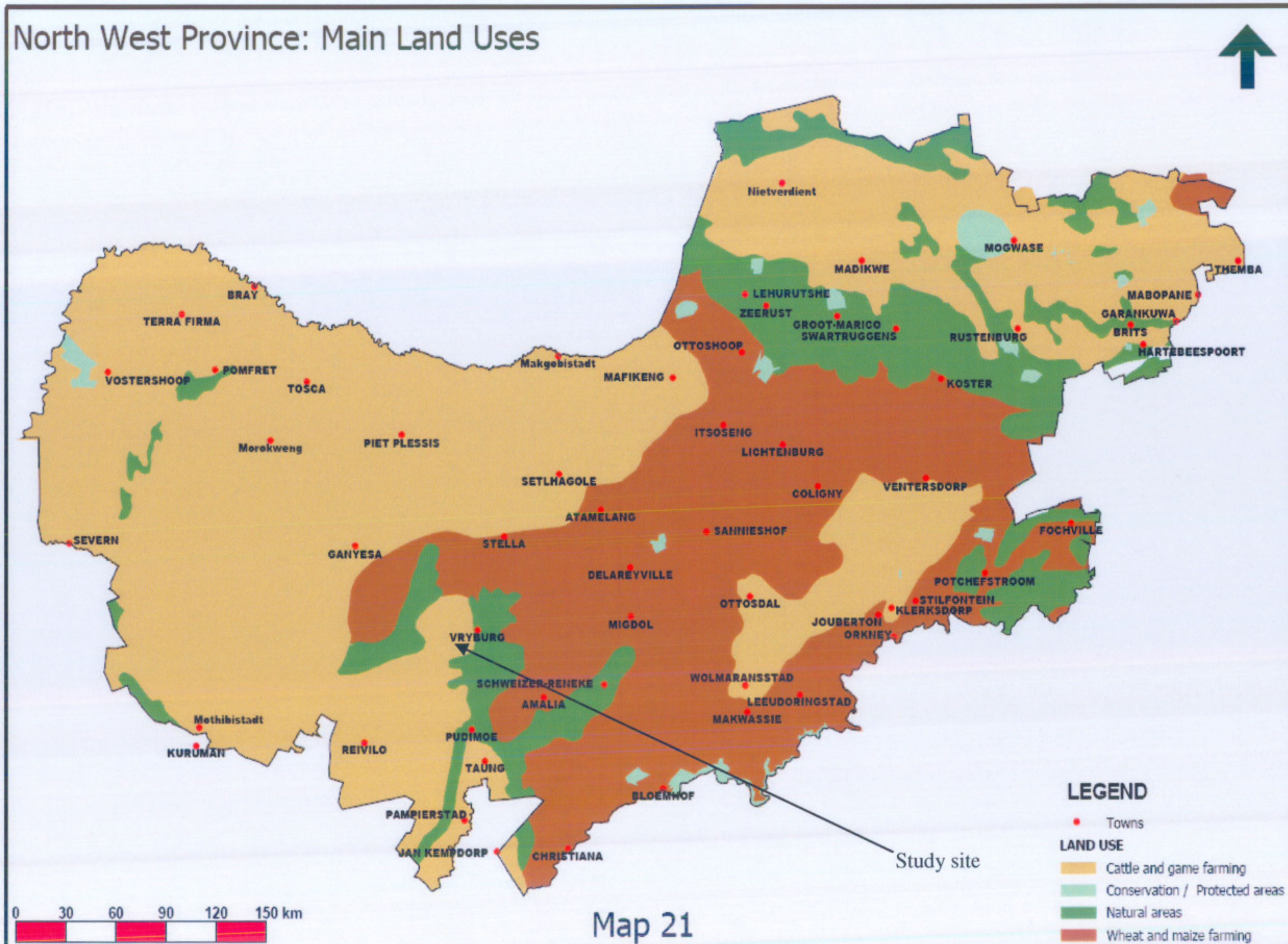
Appendix I: Veld types of South Africa



Appendix J: Alien plant invasion in the catchments of the North West Province



Appendix K: Main land uses in North West Province



Source: Environmental Potential Atlas 1998