# AN ASSESSMENT OF THE IMPACTS OF INVASIVE AUSTRALIAN WATTLE SPECIES ON GRAZING PROVISION AND LIVESTOCK PRODUCTION IN SOUTH AFRICA

by

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# DECLARATION

By submitting this thesis electronically, I declare that the entirety of the work contained therein is my own, original work, that I am the sole author thereof and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

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# ABSTRACT

I investigated the impacts of the invasive wattle species (*Acacia mearnsii, A. dealbata, A. decurrens*), on the ecological function and productivity of rangelands in South Africa and their ability to sustain livestock production. More specifically, this study set out to: (1) assess grazing areas at a national scale; (2) identify evidence of progressive impacts of these species on livestock production across a selection of magisterial districts; (3) determine the effects of *A. mearnsii* density on growth form dominance of indigenous plant species, and highlight how this translates into impacts in forage quality and quantity; (4) determine the effects of *A. mearnsii* invasion on soil resources and conditions (key determinates of ecological function) required to support grazing production; and finally (5) determine to effects that clearing operations have had on the provision of grazing resources.

My results indicate that the level of overlap between wattle invasion and areas with a high grazing potential, at the national scale, is relatively low (2.9%). However, at a magisterial district scale, areas with a high grazing potential have a moderate (approximately 35.1% high grazing potential area under wattle invasion for highest invaded district) level of invasion. Scattered and moderate levels of invasion currently dominate or characterise these species invasion patterns. Both light and dense *A. mearnsii* invasions reduce grazing capacity. Dense invasion had the most severe effect reducing grazing capacity from 2 ha, required to support one large stock unit, to 8 ha. The clearing of this species was found to improve grazing capacity within 5 years. The reduction in grazing capacity following invasion was largely due to reduced basal cover and herbaceous biomass. The clearing of

invaded sites was found to increase both basal cover and herbaceous biomass to pre-invasion levels. *Acacia mearnsii* invasion was found to affect certain soil properties. Under an *A. mearnsii* canopy, plant litter, carbon content of the soil and nitrogen concentrations were all found to be elevated. Overall, this study demonstrates that *Acacia* invasions have a significant effect on grazing resources at a local or site scale, and that for the eastern region of South Africa, these effects are ultimately scaled up to a district level. This is likely to have important financial and human well-being repercussions.  However clearing and restoration programs are clearly able to reverse or mitigate these effects. This study therefore lends support to or complements other ecosystem service provision arguments for clearing alien invasive species.

# OPSOMMING

Die impak van die indringer- wattelbome spesies ( *A. mearnsii , A. dealbata , A. decurrens*) , op die ekologiese funksie en produktiwiteit van weivelde in Suid-Afrika en hul vermoë om vee produksie te onderhou was ondersoek. Meer spesifiek, is hierdie studie uiteengesit om: (1) ondersoek in te stel na die mate van oorvleueling tussen indringing deur wattelbome spesies en belangrike weidingsgebiede op 'n nasionale skaal; (2)  bewyse te verskaf van die progressiewe impak van hierdie spesies op vee-produksie oor 'n seleksie van politieke distrikte, (3) die gevolge van *A. mearnsii* digtheid op groei vorm en  oorheersing van inheemse plante te wys, en hoe dit voer kwaliteit en kwantiteit affekteer; (4) die gevolge van *A. mearnsii* indringing op grond hulpbronne sowel as die  kondisie (sleutel bepalende faktore van ekologiese funksie) wat benodig word om weiding produksie te ondersteun vas te stel, en uiteindelik (5) om die gevolge van skoonmaak bedrywighede op die voorsiening van weiding hulpbronne te bepaaal.

My resultate dui daarop dat die vlak van oorvleueling tussen die wattelbome indringing en gebiede met 'n hoë weiding potensiaal, by die nasionale skaal, relatief laag is (2'9 %). Maar op 'n landdrosdistrik skaal, het  gebiede met 'n hoë weiding potensiaal 'n matige (ongeveer 35.1 % weiding potensiaal area onder wattelbome indringing vir die hoogste binnegevalde distrik ) vlak van indringing. Verspreide en matige vlakke van indringing  oorheers tans of kenmerk hierdie spesies se indringings patrone. Beide lig en digte *A. mearnsii* besettings verminder dravermoë. Digte inval het die mees ernstige uitwerking en verminder dravermoë van 2 ha, wat nodig is om een grootvee-eenheid te ondersteun na 8 ha.   Bevindinge wys dat die

verwydering van hierdie spesies weiding kapasiteit kan verbeter binne 'n tydperk van 5 jaar.   Die afname in dravermoë na indringing is grootliks te wyte aan die afname in basale bedekking en kruidagtige biomassa.   Na die skoonmaak van indringer persele is bevind dat beide basale bedekking en biomassa weer verhoog tot voor-indringings vlakke*. Acacia mearnsii* indringing beïnvloed sekere grondeienskappe. Onder  *A. mearnsii* blaredak, is alle blaarafval, koolstof-inhoud van die grond en stikstof konsentrasies verhef. In die algeheel, toon hierdie studie dat die Acacia indringings 'n beduidende uitwerking op weidings hulpbronne by 'n plaaslike of perseel skaal het, en vir die oostelike streek van Suid-Afrika, is hierdie effekte ook uiteindelik vertaalbaar tot 'n distrik vlak.  Dit sal waarskynlik belangrike finansiële en menslike welsyn gevolge hê, maar die skoonmaak en herstel programme is duidelik daartoe instaat om die gevolge te keer of te verminder. Hierdie studie leen dus ondersteuning aan of komplimenteer ander ekostelsel dienslewering argumente (soos dié van die Werk vir Water program) vir die verwydering van indringerspesies .

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# CHAPTER 1: INTRODUCTION

## 1.1.  BACKGROUND

South Africa's natural ecosystems have been invaded by many different plant and animal species, resulting in both ecological and economic impacts (Richardson *et al.,* 1997, Richardson and van Wilgen, 2004; van Wilgen *et al.,* 2008). About 13% of the country's surface is invaded to some degree by alien tree species (Le Maitre *et al.,* 2013). These invasive trees as well as shrub species, are a threat to native biodiversity (Vitousek *et al.,* 1997; Sala *et al.,* 2000) and therefore to the structure and function of natural ecosystems (Richardson and van Wilgen, 2004). The most disruptive invasive species alter ecosystems through their excessive resources use, in particular water use, which in turn impacts on stream flow and water availability (Scott *et al.,* 1998; Le Maitre *et al.,* 2000). In addition, invasive alien plants alter ecosystems by adding resources to ecosystems (such as nitrogen) (Yelenik, *et al.,* 2007) and by changing fire frequencies and intensity (D'Antonio and Vitousek, 1992). Some invasive alien plants, including *Acacia* spp. (Blanchard and Holmes, 2008) displace native flora and create an imbalance between different functional groups of species such as between shrub and grass layers in natural ecosystems (Pitmental, 2005).

Natural ecosystems provide a variety of resources for people, particularly those living in rural areas in developing countries (MA, 2005). The link between the invasion of ecosystems and land-use management strategies needs to be acknowledged (Jauni and Hyvonen, 2010). Ecosystems used extensively for livestock production or

rangelands, have at least in part become dominated by invasive alien species (Belnap *et al.,* 2012). The interactions between exotic and indigenous plant species have the potential to lead to a decline in quality and quantity of forage grasses that could negatively affect livestock (Pitmental, 2005; Pejchar and Mooney, 2009; Ndhlovu *et al.,* 2011). Furthermore, the reduction in the size of key resource habitats will reduce the capacity of the environment to maintain herbivore populations, affecting the livelihood of those who rely on grazing livestock (De Haan *et al.,* 1996). Whilst we have an understanding of some of these drivers and effects, there are few documented examples demonstrating actual changes in livestock carrying capacity following invasion, at either a national level, or internationally (Ndhlovu *et al.* 2011).

Despite limited understanding of the consequences of invasion on grazing potential, there has been increasing interest, research emphasis and expenditure on invasive alien species and their management over the last decade. The initiation of the Working for Water Programme in 1995 (Van Wilgen *et al*, 1996; Richardson and Van Wilgen, 2004) saw invasive alien plants receive considerable publicity, and millions of Rands are spent annually on the control of these species. While the justification and continued emphasis of this alien clearing programme is on water resource conservation, the clearing operations managers are interested in determining whether clearing activities have any other ecosystem and natural resource benefits which could lend further support to their alien clearing activities. Grazing resource conservation and restoration have been highlighted as such potential benefit.

### 1.1.1.  Research problem

This study formed part of a three year (2010 – 2013) multidisciplinary Research project titled: "Improving understanding of the rate of spread and impacts of invasive alien plants", conducted by the Council for Scientific and Industrial Research (CSIR). The project was funded by the Working for Water Programme (WfW) of the Department of Water Affairs. The aim of this project was to address the shortcomings and lack of existing spatial data and understanding of the rate of spread and impacts of invasive alien plants on select ecosystem services.

This MSc. study investigated the ecological impacts of invasive alien wattle species (*Acacia dealbata, Acacia decurrens* and *Acacia mearnsii*) on grazing provision and livestock production in South Africa. These wattle species are regarded as the most extensive and problematic invasive species in the country (Le Maitre *et al.,* 2000). The study was focussed at two scales. The first was the national scale where I analysed invasive alien plant distribution, focussing specifically on wattle species, in relation to potential carrying capacity. I used magisterial districts to investigate the correspondence in wattle invasion and important grazing areas across the country, using overlap analysis techniques. The second scale of focus was at a local district level, and was guided by the results for the national level assessment. The district of Stutterheim was identified as having a high degree of overlap between dense wattle invasion and important or high potential grazing areas. Within the Stutterheim district, I undertook a field assessment which investigated the impacts of *A. mearnsii* invasion on grazing capacities and the resources (including: soil nutrients and soil moisture) which underpin this, as well as the subsequent effects of clearing programmes on the reestablishment of these resources.

### 1.1.2.  Research aims and objective

The central aim of this research was to develop a deeper understanding of the ecological impacts of alien plant invasions on function and productivity of rangelands in South Africa and their ability to sustain livestock production.

The objectives of this study were as follows:

- To determine at a national level, where invasive alien wattle species are impacting on livestock production and what is the nature and degree of these impacts.
- To determine at a local scale the effects of *A. mearnsii* density and subsequent clearing effects on growth form dominance of the indigenous vegetation species, and how this translates into impacts in forage quality and quantity.
- To determine the effects of *A. mearnsii* invasion on local level soil resources and conditions required to support grazing production.

### 1.1.3.  Thesis structure

**Chapter 1** provides the background to this study, and outlines my aims and objectives.

**Chapter 2** reviews the literature with specific focus on ecosystem services and the provision of grazing in rangeland ecosystems, the drivers and effects of rangeland degradation with specific focus on alien plant invasions and their impact on ecological processes.

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**Chapter 3** explores the correspondence or spatial overlap between areas invaded by wattle species and nationally important or key grazing areas or resources. This chapter also highlights the progressive impacts of these species on livestock production across selected magisterial districts.

**Chapter 4** quantifies the impacts of different levels of *A. mearnsii* invasion and clearing actions on rangeland grazing capacity. Furthermore, it assesses the effects of *A. mearnsii* invasion on key components or indicators of rangeland function and the ability to sustain grazing provision.

**Chapter 5** provides a synthesis of the key findings of chapters 3 and 4 and provides overall conclusions and recommendations.

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**CHAPTER 2: AN OVERVIEW OF THE RELEVANT LITERATURE**

**2.1.  INTRODUCTION**

The concept of ecosystem services is central to this thesis and this chapter presents an overview of the most recent and relevant ideas in this area. How rangelands are to be understood in this context is considered, and the specific ecosystem services offered by rangelands presented. Linkages are then made between rangeland dynamics and how changes in these provide opportunities for plant invasion. Following this, literature on alien plant invasion in South Africa and then more specifically in the rangeland management context, is presented. In conclusion, the notion of ecosystem services is revisited in the context of the invasion of rangelands, how ecosystem services are impacted, and how they might be restored.

**2.2.  ECOSYSTEM SERVICES**

The history of links between human benefit and natural ecosystems (whether directly or indirectly) dates back thousands of years to ancient civilisations (Gómez-Baggethun *et al.,* 2010). This suggests that humans have long acknowledged the importance of what is now referred to as ecosystem services, despite the late introduction of the 'ecosystem services' concept. The introduction of the term 'ecosystem services' dates back to the early 1980s when it was first used by Ehrlich and Ehrlich (1981). Westman (1977) was the first to refer to these benefits as 'nature's services'. Despite this long history of acknowledgement of ecosystem

services, the Millennium Assessment (MA) (2003; 2005) firmly brought forward this concept by demonstrating the linkages between ecosystem function and human wellbeing. Several definitions for ecosystem services have been introduced by various authors. For instance, the commonly cited definitions include that of Daily, (1997) who refers to ecosystem services as the conditions and processes through which natural ecosystems sustain and fulfil human life (Daily, 1997). Costanza *et al.,* (1997) refers to ecosystem services as the benefits human populations derive, directly or indirectly, from ecosystem functions. And the MA (2005) defined ecosystem services as the benefits people obtain from ecosystems. Drawing exclusively from the latter two definitions, in this study I will refer to ecosystem services as the benefits that people derive, either directly or indirectly, from natural or modified ecosystems (Constanza, *et al.,* 1997; Daily, 1997; MA, 2005; van Jaarsveld, *et al.,* 2005). Ecosystem services are often grouped into four service categories which include: supporting, regulatory, provisioning and cultural services (Daily, 1997). Supporting services are generally a requirement for the production of all other services, and these include soil formation and nutrient cycling. Regulating services control the flow of benefits including air quality maintenance, climate regulation, water regulation and erosion control. Provisioning services may include those natural products that are directly available for human use such as food and fiber, fresh water, fuel, and ornamental resources. Cultural services are benefits that enhance the quality of human life and human wellbeing through spiritual enrichment, cognitive development, recreation, spiritual and religious values, knowledge systems aesthetic values, educational values, recreation and ecotourism, sense of place and cultural heritage values (MA, 2005; van Jaarsveld *et al.,* 2005).  Although it is often classified as an ecosystem service on its own (in cases such as nature based

tourism), biodiversity plays a critical role in underpinning these services and ensuring their delivery. Intact ecosystems with higher levels of biodiversity might be more resistant to biological invasions compared to disturbed ecosystems (Davis *et al.,* 2000; Diaz *et al.,* 2006).

Delivery of ecosystem services is to a large extent a consequence of complex interactions among species and their biotic environment and the utilisation of the benefits by people (Fisher *et al.,* 2008). Ecosystems provide a variety of ecosystem services that occur as bundles, which suggests that the use of one ecosystem service usually affects the provision of other ecosystem services or might result in ecosystem service trade-offs (Raudsepp-Hearnea *et al.,* 2010). Therefore, any unsustainable use of a particular ecosystem service may have damaging effects on the supply of other services provided by that ecosystem (De Fries *et al.,* 2004; Rodriguez *et al.,* 2006; Bennett and Balvanera, 2007) and hence to human wellbeing (Reyers *et al.,* 2009). Humans have advanced through the manipulation of ecosystems with the aim to enhance production of goods and services such as food, fibre, and fuel, and water supplies (Kareiva *et al.,* 2007; Swinton *et al.,* 2007). Although these actions have largely resulted in improved human wellbeing (Levy *et al.,* 2005), the resultant ecosystem changes may affect the sustainability of service delivery in the long run (MA, 2005). These are hard to predict, as they are the outcome of multiple interacting drivers such as land cover change, climate change or invasive alien species (MA, 2005). Whilst multiple investments and technological developments have been put in place to mitigate against the impacts of changing environments, humans are ultimately dependant on the continued supply of these services (MA, 2005). Therefore, effective conservation and management of ecosystem processes is critical to ensure their continued supply.

## 2.3.  PROVISIONING OF SERVICES BY NATURAL RANGELANDS

Rangelands provide an array of ecosystem services to millions of people who are located within rangeland ecosystems and to those connected to these ecosystems around the world through, for example, food and water supply to urban areas (Havstad *et al.,* 2007; Reid *et al.,* 2008). These services, as classified according to MA (2005), include provisioning services (food, fiber, clean water and genetic resources), regulating services (water and air quality), cultural services (recreational space, religious sites, sources of natural medicines, and educational values) and supporting services (primary production, and nutrient cycling).

Provision of ecosystem services by rangelands is a function of interacting ecological elements which in turn determines the structure and function of rangeland ecosystems. Havstad *et al.,* (2007) developed a framework (Figure 2.1) that highlights ecologically interacting elements that determine the characteristics and function of rangeland ecosystems and their capacity to provide ecosystem services. Each element of the framework can directly or indirectly influence ecosystem structure and dynamics with consequences for ecosystem services. The relative importance of these elements in the supply of ecosystem services can vary in both time and space for the same system as well as among systems.
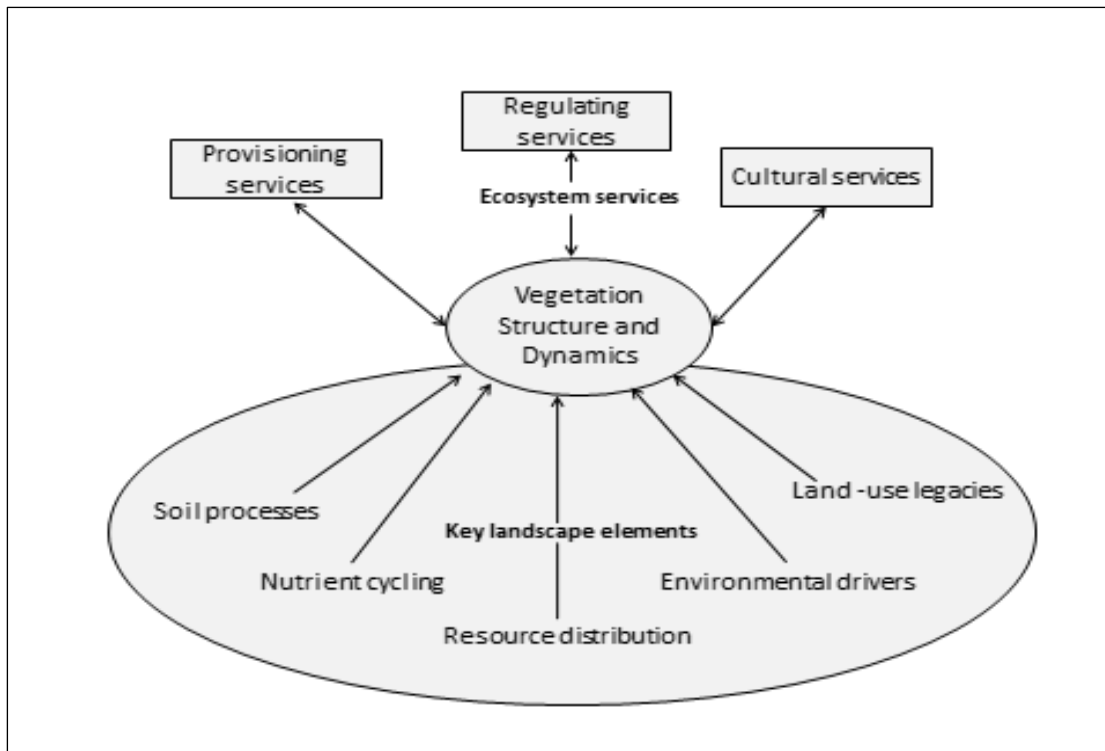
**Figure 2.1:** Key elements of natural landscapes (soil processes, nutrient cycling, resource distribution, environmental drivers, and land-use legacies) interact to determine vegetation structure and dynamics resulting in effects on ecosystem goods and services (adapted from Havstad *et al.,* 2007).

South Africa's rangelands constitute approximately 70% of the country's remaining natural areas, and contribute to human wellbeing and the economy of the country through the provisioning of an array of ecosystem services. Rangelands are also home to a substantial portion of South Africa's population, who depend on the ecosystem services that rangelands provide for their livelihoods and wellbeing (Cousins, 1999; Shackleton *et al.,* 2001). South Africa's rangelands are regarded as the main source of forage for livestock production, which is the mainstay of the South African agriculture sector (DAFF, 2012). Supporting livestock production, which contributes approximately 50% of the gross value of agricultural production, the social and economic value of rangelands to South Africa is clear (DAFF, 2012).

Furthermore, rangelands provide other natural resources, such as water supply, wildlife habitat, fertile soils, fuel wood, thatching grass, wild fruits, edible and medicinal plants, and act as a repository of biodiversity (Dovie *et al.,* 2002; Shackleton *et al.,* 2007).

Rangelands provide a complex array of ecosystem services that support many aspects of human lives.  These range from contributing to national level economy to very localised social customs and traditions (Cousins, 1996; Shackleton *et al.,* 2005; Vetter, 2013). Livestock production is a key livelihood component for many households, particularly those in the communal areas of South Africa (Cousins, 1999; Shackleton *et al.,* 2001; Ainslie, 2002, 2005). Livestock contribute to human wellbeing through a range of ecosystem services including milk, meat, manure, draught power, dung, income security and ceremonial and customary services (Shackleton *et al.,* 2005). Rangelands are also an important wellbeing component even to non-livestock owning households, with cattle used in bride-wealth payments (ilobolo), and loan schemes for co-operative draught power (Cousins, 1996; Shackleton *et al.,* 2005), thereby benefiting broader communities beyond individuals or owners.

### 2.3.1.  Grazing provision in South Africa

### *2.3.1.1.  Vegetation types*

A number of South African biomes (biome is a large eco-region defined by similar growth forms and climatic conditions) support a variety of rangeland vegetation types including grassland, savanna, Nama-karoo, forest, succulent karoo, fynbos and thicket ecosystems (Scholes and Walker, 1993; Murray and Illius, 1996; Scholes and

Archer, 1997; du Toit and Cumming, 1999; Murray and Illius 2000; Mucina and Rutherford, 2006). Grazing provision of natural ecosystems varies across South Africa's landscape, mainly due to variations in vegetation types, which are largely determined by climatic factors (e.g. rainfall) and soil type (Scholes, 1998; Illius and O'Connor, 1999).

The amount of forage available to livestock is to some extent determined by vegetation structure. Different types of livestock have different grazing requirements that are influenced to a degree by their body sizes (Esler *et al.,* 2006). For instance, in the grassland biome, forage varies in structure between tall and short grasses. Here, smaller herbivores (e.g. sheep) concentrate on more nutritious shorter grasses, while large bodied herbivores (e.g. cattle) graze on higher grass biomass and can be more tolerant to low nutritional value (Murray and Illius, 1996; Prince and Olff, 1998; du Toit and Cumming, 1999; Murray and Illius 2000). Savanna ecosystems are characterised by a mixture of two life forms: trees and grasses (Scholes and Walker, 1993; Scholes and Archer, 1997; du Toit and Cumming, 1999; Mucina and Rutherford, 2006). This creates a suitable environment for browsing (e.g. goats) and grazing (e.g. cattle and sheep) animals (du Toit and Cumming, 1999). In the Nama and succulent karoo biomes, vegetation is characterised by grasses and short shrubs and is more suitable for small grazers such sheep and goats (O'Connor and Roux, 1995).

Within these biomes, different ecological dynamics persist as a result of key ecological drivers, such as rainfall, fire and herbivory. Ecological drivers are the determinants of ecosystem resilience to degrading factors. For instance, many coastal grasslands and savannas of the Eastern Cape show more resistance to

heavy continued grazing (McKenzie, 1982; Duncan, 2010), whereas higher altitude grasslands (Vetter *et al.,* 2006; Vetter, 2007; Vetter and Bond, 2012) and the lowlands of succulent karoo have been shown to be highly vulnerable to degradation (Hendricks *et al.,* 2005; Anderson and Hoffman, 2007). While South Africa's rangelands are broadly defined by the semi-arid nature of the country, on closer examination it is evident that forage production varies significantly, both spatially and temporally and this in turn determines how rangelands can be used.

### *2.3.1.2. Key factors supporting grazing provision*

Plant productivity and nutritional quality are to a large extent dependent upon resources that govern plant growth such as rainfall and soil fertility (Ellery, *et al.,* 1996; Scholes, 1998). Rainfall, particularly plant available moisture, controls plant processes such as plant growth, plant defence and phenology among others, particularly in arid and semi-arid rangelands (Ludwig and Tongway, 1998). Available water plays a vital role in controlling plant internal processes that are essential for plant growth (Scholes, 1993). For instance, available water regulates the stomatal opening which is essential for carbon fixation. Soil moisture enhances soil microbial activity and thereby increases the rate of decomposition of organic matter and in turn nutrient availability. Furthermore, availability of key nutrients such as nitrogen and how these are taken up by plants is strongly related to soil moisture (Scholes, 1993).

Essential components to the functioning and sustainability of an ecosystem occur below ground, hence the physical, chemical and biological functions of soils and processes are a key to productivity and recovery of an ecosystem (Neary *et al.,* 1999; Tongway and Hindley, 2004). Soil physical and chemical properties and

biological processes influence soil water intake as well as nutrient cycling. Most of the organic nutrients from plant material are mineralised in order to be made available for plant root intake (Charley and Cowling, 1968). For instance, organic nitrogen needs to be transformed into ammonium ($NH_4^+$) or nitrate ($NO_3^-$) for absorption by plants. The rate at which these nutrients are converted and made available for plant growth is influenced by biological (e.g. microorganisms) and physical processes (e.g. fire, erosion, leaching of soil nutrients). Processes such as plant uptake of nutrients, growth, decomposition and mineralisation in arid ecosystems are all closely related to climate and most mineralisation of nutrients occurs during wet periods (Singh and Coleman, 1973; Sparling and Ross, 1988). Mineralisation of these nutrients is often achieved through complex interactions of soil organisms (earthworms, termites, microbes), with microorganisms (bacteria and fungi) assuming a major role in this regard, except in some instances where organic matter mineralisation has occurred due to fire (Holt and Coventry, 1990; O'Farrell, *et al.,* 2010). Burning results in rapid mineralisation of organic matter in the soil. However soil nutrients might be lost where there is too frequent burning and high temperature fires (Holt and Coventry, 1990).

Organic matter improves soil structure (Thurow *et al.,* 1986), and thereby enhances water infiltration (Smith *et al.,* 1990) and reduces soil erosion through aggregate stabilisation (Chevallier *et al.,* 2001). The net effect is better water use efficiency by rangelands (Snyman, 2005). Factors affecting plant decomposition affect nutrient cycling and therefore primary productivity (Ekaya and Kinyamario, 2001). Infiltration capacity or run off is linearly related to basal cover of perennial grasses and organic matter content of the soil (Snyman and du Preez, 2005) which can also be affected

by litter cover (Snyman, 2005) and above ground biomass (Scholes and Walker, 1993; Snyman, 2002).

Temporal and spatial variability in production characterises these ecological systems. Seasonal and annual variation in production is a feature of rangeland systems and the failure to adjust management in response to, for example, extremely low production during drought, could lead to degradation (Vetter, 2013).

## 2.4.  RANGELAND DEGRADATION: PAST TO PRESENT UNDERSTANDING

Rangeland degradation is defined as injurious change in the capacity of a rangeland to sustain livestock production by means of supplying a range of related services, including, but not limited to, forage, fresh water and biodiversity (Scholes, 2009). Rangeland degradation comes in various forms including loss of cover, bush encroachment and alien invasions (Hoffman and Todd, 2000). Degradation of South Africa's natural rangelands threatens the livelihood of commercial and communal livestock farmers and game ranchers who are reliant on these ecosystems (Hoffman and Todd, 2000; Shackleton *et al.,* 2001).

Causes of rangeland degradation have been long debated in South Africa (Hoffman and Ashwell, 2001; Ward, 2005; Anderson and Hoffman, 2007). Since the end of the 19th century attempts have been made to address the issue of land degradation. These included the establishment of investigative committees such as the Drought Select Committee of 1914, followed by the Drought Investigation Committee, 1920-1923. Early views put forward by the Drought Investigation Committee in 1923

argued that rangeland degradation was to a large extent associated with land use practices such as overstocking and the kraaling systems (Hoffman and Ashwell, 2001). In response to findings, the government established the Agricultural Extension Service of 1925, followed by the Soil Erosion Advisory Councils of 1930 to combat land degradation. While most of these efforts were focussed on white farming areas, a number of interventions targeted areas termed 'native reserves'. The establishment of native homelands between 1913 and 1936 following the passing of the 1913 Natives Land Act, resulted in over 3 million African people being resettled and restricted to these areas (Christopher, 1994; Fox and Rowntree, 2001). By the 1930's native homelands were reported to experience severe environmental problems including soil erosion and loss of grazing resources. In 1932 the Native Economic Commission called for action which, a couple of years later, lead to the initiation of a 'Betterment' strategy in the homelands, aimed at addressing land degradation. Actions here included gully rehabilitation and scattered rural settlement being concentrated in the villages with the aim to conserve natural resources. This however, had the adverse effects and intensified levels of land degradation and soil erosion as demands for services such as water and firewood led to degradation of these resources around the new villages. Furthermore, the daily movement of livestock between the fields and villages intensified soil erosion around the homesteads (Hoffman and Ashwell, 2001).

To date, rangeland degradation remains a topic of focus due to its adverse effects on soil, water, vegetation and ultimately human wellbeing. To a large extent, degradation has been attributed to unsuitable use of natural resources, particularly in communally managed ecosystems (Hoffman and Todd, 2000; Hoffman and Ashwell, 2001; Vetter, 2005, 2013). The common view on the causes of ecosystem

degradation has been overstocking, with people tending to keep more livestock than they should for a variety of socio-economic reasons (Hoffman and Todd, 2000; Vetter, 2005). Furthermore, the increasing demand for livestock products and services puts these ecosystems under pressure (Illius and O'Connor, 1999; Swanepoel *et al.,* 2008), which in turn translates into overgrazing and less productive ecosystems.

The main debates around rangeland degradation for the past few decades have been based on the dynamics of these ecosystem responses to disturbance. These have become polarised and two very different models have been put forward to describe rangeland function and its key driving variables - the equilibrium and the non-equilibrium model (Briske *et al.,* 2003, Vetter, 2005). The equilibrium model stresses the importance of biotic feedbacks between herbivores and their environment and assumes that the actual carrying capacity of an area at any given time is determined by rangeland conditions, which are a function of species composition, biomass and cover (Trollope, 1990; Vetter, 2005). The management practices involve reduction of stocking rates and encouraging stability, in order to minimise over grazing on certain areas of the landscape. On the other hand, the non-equilibrium model stresses the importance of abiotic factors such as rainfall as the main drivers of vegetation and livestock dynamics. The non-equilibrium model recommends mobility and opportunistic stocking strategies based on the current range conditions. There has been considerable debate as to which model best suits our natural ecosystems. This thesis does not go any further into these debates. Instead it focuses primarily on understanding the ecological drivers of rangeland functions, impacts and grazing services.

## 2.4.1.  Factors promoting rangeland degradation

Heavy stocking of grazing animals and continuous grazing is very common on natural rangelands particularly in areas where livestock production is the main source for livelihood. These factors can have negative impacts on rangeland vegetation as they result in overgrazing of palatable species, as a consequence overgrazing is considered the most common cause of degradation in rangelands (Snyman and du Preez, 2003). Over grazing reduces herbaceous cover and this result in less frequent and low intensity fires. As a result, the effectiveness of fire to control woody vegetation is reduced. High grazing pressure on the herbaceous layer affects the competitive interaction for growth resources between woody and herbaceous species in favour of the woody species (Hoffman and Ashwell, 2001). Furthermore, the reduction of the grass layer exposes the soil surface to trampling by grazers, which in turn exposes the soil to erosion factors such as wind and water (Snyman, 1998). Trampling on bare soil by grazing animals can affect soil structure, reduce water infiltration rates, increase surface water runoff and increase soil moisture loss through evaporation (Snyman, 1998). These biotic and abiotic shifts make ecosystems vulnerable to encroachment and invasion by alien plants and certain indigenous species.

The increase in the relative density and extent of indigenous woody species is referred to as bush encroachment (Ward 2005). Increases in the density of indigenous woody species, which are often unpalatable to livestock due to their chemical and/or physical defense mechanisms, (Rohner and Ward, 1997) have often had a detrimental effect on rangeland productivity. As a consequence bush encroachment reduces carrying capacity of rangelands and hence threatens the livelihoods of commercial and communal farmers. There has been considerable

debate around the causes of bush encroachment and a number of models have been developed to describe the causal factors (Ward, 2005). The main causes of bush encroachment include ecological factors that govern the functioning of rangeland dynamics such as rainfall, nutrients, grazing and fire which affect the tree-grass ratio (Jeltsch *et al.*, 1996; Higgins *et al.*, 2000; Kraaij and Ward, 2006). These factors determine the competitive dynamics of grasses and woody plants for growth resources (Walter, 1971; Walker *et al.*, 1981; Davis *et al.*, 1998). Grasses are assumed to out-compete woody species for resources found in the upper soil layers. In well managed grassland ecosystems, grasses are able to suppress the establishment of tree seedlings (Ward, 2005). However, in areas with shallow soils, where there is no stratification of grass and woody species root systems, bush encroachment can occur. Fire in rangelands is a natural phenomenon and has also been adopted as management tool by ranchers. Fire has been used to control bush encroachment by burning seedlings and by destroying seeds and seedbanks. However, in some instances fire may promote woody species recruitment, acting as a trigger for seed germination (Higgins *et al.*, 2000). This is particularly the case with invasive alien trees, while some species can be controlled by fire, some species, including Australian wattles, are stimulated by fire.

Whilst bush encroachment by indigenous species is a key issue in rangeland dynamics and grazing service provision, my focus is on understanding the closely related issue of invasive alien plants and their impacts on rangelands.

## 2.5.  ALIEN PLANT INVASIONS IN SOUTH AFRICA

South Africa is a country that been severely affected by invasive alien plants and many of the country's ecosystems are regarded as under threat (Macdonald *et al.,* 1986; Richardson *et al.,* 1997; Richardson *et. al.*, 2005; van Wilgen *et al.,* 2008). About 13% of South Africa's total area is invaded by alien trees with many of these invaders already well established, while others are in the early stages of invasion (van Wilgen *et al.,* 2001; Nel *et al.,* 2004; Kotze *et al.,* 2010; van Wilgen *et al.,* 2008; Le Maitre *et al.,* 2013). The introduction of alien species in South Africa dates back to the 19th century with approximately 8750 plants species recorded as having been introduced around that time (van Wilgen *et al.,* 2001; Richardson and van Wilgen, 2004).

Invasive alien plants in South Africa are categorised according to their preferred habitats and or the consequences of their invasion (Swarbrick, 1991; Henderson, 2001). Many species have invaded natural or semi-natural ecosystems and these are referred to as environmental weeds. Whereas some species invade severely disturbed sites such as cultivated land and are referred to as ruderal and agrestal weeds (Swarbrick, 1991; Henderson, 2001). Environmental weed species have been classified into different groups, these include: transformers, potential transformers, special effect weeds, minor weeds. Transformers are typically trees, aggressive climbers, shrubs and dense herbs that often replace any indigenous canopy or sub-canopy layer of a natural or semi-natural ecosystem. Examples here include some species of the *Acacia* genus, such as *A. mearnsii, A. dealbata, A. decurrens*, *Opuntia* species such as *O. aurantiaca* and *O. ficus-indica*, and *Jacaranda*

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*mimosifolia* and *Lantana camara*. Potential transformers are species that have already invaded an ecosystem and have a potential to replace vegetation layer but not yet showing a detrimental effect on the ecosystem or vegetation structure. Here examples include *Pennisetum clandestinum, Eucalyptus cladocalyx* and *E. diversicolor*. Special effect weeds are serious environmental weeds, including poisonous or chemical irritating species, which could degrade an ecosystem without altering vegetation composition, e.g. *Hypericum perforatum*, *Nerium oleander* and *Duranta erecta*. Minor weeds are species that can normally co-exist with indigenous species without dominating the vegetation structure, although increases in abundance could change the native species composition. Examples here include *Solanum sisymbriifolium* and *Cardiospermum halicacabum*.

To date, about 161 are now classified as invasive, and 68% of these are woody species (Nyoka, 2003). This invasion problem is anticipated to increase in its severity and as well as in geographical extent, as human-mediated disturbance makes ecosystems more susceptible to invasion by alien species (Vitousek, *et al.,* 1986; van Wilgen *et al.,* 1996; Richardson and van Wilgen, 2004).

According to several surveys on the distribution of invasive alien plants in South Africa, vegetation types and biomes vary in their susceptibility to invasion (Henderson, 2001; Kotze *et al.,* 2010). So far there have been three national scale surveys in South Africa which examined the abundance and extent of invasion across the country - Henderson (1998); Versveld *et al.,* (1998); and Kotze *et al.,* (2010). In grassland and savanna ecosystems the most prominent invaders include *Acacia* species (such as *Acacia dealbata, Acacia decurrens*, and *Acacia mearnsii*), *Eucalyptus grandis, Jackaranda mimosifolia* and various shrub species (e.g.

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*Chromolaena odorata, Lantana camara*) (Nel *et al.,* 2004; Richardson and van Wilgen, 2004; van Wilgen *et al.,* 2008). The Nama karoo of the Northern Cape is largely dominated by *Prosopis* species (Richardson and van Wilgen, 2004), whereas in the succulent karoo and the thicket biome of the Eastern Cape, the most prolific woody invaders include several *Opuntia* species, and saltbushes (e.g. *Atriplex* species) (Richardson *et al.,* 1997; Milton *et al.,* 1999). The fynbos biome has been regarded the most heavily invaded of all biomes in the country (Richardson *et al.,* 1997; Rouget *et al.,* 2003). Here the principal invaders are trees and shrubs including *Acacia, Hakea* and *Pinus* species (Richardson *et al.,* 1996). The highest levels of invasion are found in riparian zones across all biomes (Richardson and van Wilgen, 2004).

### 2.5.1. Factors promoting alien invasion in natural ecosystem

The factors associated with ecosystem invasions are complex and relate to the invasiveness of the introduced species and invasibility of the new environment (Lonsdale, 1999; Thuiller *et al.,* 2006). Invasions occur under complex conditions with temporal and spatial variation in disturbances and environmental factors making it difficult to isolate the key mechanisms that promote invasions (Fuentes-Ramirez *et al.,* 2011). An ecosystem's vulnerability to invasion can be a result of various factors including changes in disturbance regimes (Crawley, 1987; Prieur-Richard *et. al.,* 2000), climatic variations in a given region (Lonsdale, 1999), fluctuating resources (Davis *et al.,* 2000) and reduced species diversity (Prieur-Richard and Lavorel, 2000).

A positive relationship between invasions and disturbance regimes has been reported as one of the mechanism that promote invasion by alien species (Martin and Marks, 2006). Disturbance is a natural component of many ecosystems and is essential for ecosystem functioning. However, extensive disturbances associated with intentions to increase agricultural productivity alter the natural disturbance regimes (Vitousek, *et al.,* 1986; van Wilgen *et al.,* 1996; Davis *et al.,* 2000). This often creates opportunities for invasion by alien species whose reproductive strategies are linked with such disturbances (Hobbs, 1989; Hobbs and Hueneke, 1992; Pauchard *et al.,* 2008). Disturbances through the overuse or over grazing of plants, which is often the case in natural rangelands, could affect fire regimes and create opportunities for the exploitation of unused resources such as light, nutrients and water. The availability of unused resources is directly related to invasion where invading species have greater success if they do not encounter tight competition for these resources from the indigenous species (Davis *et al.,* 2000). This means that the susceptibility of a community to invasion changes in time as the amount of excess resources fluctuates (Davis *et al.,* 2000).

Invasive alien plants generally invade ecosystems unaccompanied by their natural or biological enemies and their competitors (Williamson and Fitter, 1996; Crawley *et al*, 1997), and indigenous enemies may take many years to colonise and attack alien species. Breaking free from their natural enemies, alien species may increase plant fitness by reallocating resources from other aspects of performance such as defence to growth, longevity and development aspects (Weiss and Milton, 1984; Blossye and Notzold, 1995). This gives the invading species great advantage over the native species.

## 2.6.  INVASIVE AUSTRALIAN ACACIAS

Acacia species have invaded ecosystems in many parts of the world (Castro-Diez *et al.,* 2011; Richardson and Rejmanek, 2011; Richardson *et al.,* 2011). In South Africa, alien acacia species are recognized as some of the most aggressive invaders, extending over a wide range of habitats including forest, plantation margins, riparian zones, savannas, woodland and roadsides (Musil, 1993; Henderson, 2001; Dye and Jarmain, 2004; Nel *et al.,* 2004). They are particularly prominent invaders of disturbed environments, especially following fires (Werner *et al.,* 2010; Morris *et al.,* 2011). They pose a range of ecological and socio-economic impacts on the ecosystems (Le Maitre *et al.,* 2000; De Wit *et al.,* 2001; Marchante *et al.,* 2003; Gaertner *et al.,* 2009; Marchante, 2011).

Australian acacias have traits enabling them to survive nutrient and moisture poor environments and dominate competitive interactions with native species for above and below ground resources (Morris *et al.,* 2011). These include rapid growth rates which give them a competitive advantage for resources such as light (Morris *et al.,* 2011). Furthermore, acacia species have extensive above ground biomass and associated leaf area when compared to indigenous vegetation (Morris *et al.,* 2011). The high leaf biomass is well enriched with higher leaf N concentrations than the indigenous vegetation, in some instances greater concentrations of K, Ca and Mg have been reported (Musil, 1993). With their extended height and large biomass, acacia species shade native vegetation, and change the microclimate below canopy height, hence a shift in native species composition (Holmes and Cowling, 1997). Acacias bear a large number of nutrient-rich seeds, resulting in large persistent seed

banks (Holmes, 1989), which are stimulated by fire (Yelenik, 2007; van Wilgen, 2009; Marchante *et al.,* 2010). Their extensive root biomass enables acacias to acquire nutrients and moisture both from shallow and deep soil depths (Witkowski, 1991; Werner *et al.,* 2010; Morris *et al.,* 2011), which gives them an advantage over native species during water limited periods.

Several species of the acacia group, most notably wattle species (*A. mearnsii, A. dealbata, A. decurrens*), have invaded a wide range of grazing areas in South Africa and are posing a variety of impacts on goods and services provided by these ecosystems (de Wit *et al.,* 2001; Nel *et al.,* 2004; van Wilgen *et al.,* 2008). These species are the focus of this study.

### 2.6.1.  Wattle species: (*A. mearnsii, A. dealbata, A. decurrens*)

Wattle species including *A. mearnsii, A. dealbata* and *A. decurrens* are commonly known as black, silver and green wattles, respectively. They are known as aggressive woody invaders and are classified under the transformers category. Wattle species have invaded various ecosystems including fynbos, grassland and savanna biomes, roadsides and watercourses. *Acacia dealbata* is mostly concentrated in the eastern escarpment including Eastern Cape, Kwa-Zulu Natal, Mpumalanga and eastern parts of Gauteng province with some growing evidence in the Western Cape and Limpopo province*. Acacia decurrens* is widely spread in Mpumalanga, Kwa-Zulu Natal and Eastern Cape provinces. *Acacia meanrsii* is the most prominent of the three wattle species in terms of distribution spreading from the Western Cape along the east escarpment (including Eastern Cape, Kwa-Zulu Natal) to Limpopo province (Henderson, 2001, Kotze *et al.*, 2010).

Wattles are fast growing leguminous trees (Henderson, 2001), reaching about 5-15 meters high and 0.1 to 0.6 in diameter at 3-5 years after planting (DWAF, 1997; Henderson, 2001; Campbell, 2000). Introduced in South Africa in the 19th century (Nyoka, 2003), their primary role was to provide shade, windbreaks, fuel wood and for the tannin content found in the bark (de Wit *et al.* 2001; Henderson, 2001; de Neergaard *et al.,* 2005; Shackleton, 2007b). Wattle species are characterised by bi-pinnate adult foliage and pale to bright yellow flower heads normally arranged in elongated racemes. Wattle species are renowned for threatening indigenous vegetation by outcompeting them for resources such as water, soil nutrients and organic matter. Wattle species often form dense stands, and maintain a high proportion of green leaves throughout the year (Henderson, 2001; Dye and Jarmain, 2004). They have been reported to decimate the grass layer, reduce native species richness and increase water loss from riparian zones (Le Maitre *et al.,* 2000; Dye and Jarmain, 2004; Blanchard and Holmes, 2008).

Despite the documented detrimental effect on ecosystem services supply (De Wit *et al.*, 2001; Van Wilgen *et al.*, 2008; 2011) these species continue to contribute to South Africa's economy and supply an array of services and benefits to local communities (Kull *et al.*, 2011). Wattle species have been the main stay of the forestry industry in South Africa since their introduction, with *A. mearnsii* being the most prominent in this regard. In South Africa plantations of *A. mearnsii* are currently owned by approximately 2700 growers (Forestry South Africa, 2009). These provide employment opportunities in areas with high levels of unemployment and poverty. Approximately 30, 000 people are estimated to be employed by commercial and small scale farmers of *A. mearnsii*, collectively (Forestry South Africa, 2009). The commercial wattle sector produces timber, pulp, woodchips, and charcoal (de Wit *et*

*al.*, 2001). About 1.2 million tonnes of timber was produced in 2009, most of which was exported as woodchips. Furthermore, 180, 000 tonnes of wattle bark was converted to tannin products (Forestry South Africa, 2009).

From a rural household perspective, the benefits these species provide include building and fencing material (in particular poles), supply of fire wood, and green manure due to its high protein content. Furthermore, these species provide a source of income to those who collect and sell the fire wood and building or fencing material to other community members (de Wit *et al.*, 2001; de Neergaard *et al*, 2005; Shackleton *et al.*, 2007b).

## 2.7. IMPACTS OF INVASIVE ALIEN PLANTS ON ECOSYSTEMS SERVICES AND ECOSYSTEM FUNCTIONING

Invasive alien plants present a range of negative impacts on natural ecosystems functioning (Yelenik *et al.,* 2004), indigenous species diversity (Blanchard and Holmes, 2008; Gaertner *et al.,* 2009), and therefore the delivery of ecosystem services (Constanza *et al.,* 1997; Richardson *et al.,* 2000; Pejchar and Mooney, 2009). There is growing evidence that invasive alien trees alter ecological processes (Vitousek *et al.,* 1997; Sala *et al.,* 2000; Richardson and van Wilgen, 2004) that regulate ecosystem functioning (Mack *et al.,* 2000; Hooper *et al.,* 2005) such as species interactions, hydrological regimes (Le Maitre *et al.,* 2000; Obrist *et al.,* 2004), disturbance regimes (including fire frequency and intensity) (D'Antonio and Vitousek, 1992; Davis *et al.,* 2000), and nutrient cycling (Yelenik *et al.,* 2004; Yelenik *et al.,* 2007; Ehrenfeld, 2010; Vila *et al.,* 2011). This often translates into changes in

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species richness, biomass and composition of native flora and fauna (Vila *et al.,* 2006; Gaertner *et al.,* 2009). Impacts of invasive alien plants in ecosystem services are complex, as the effect of invasive alien plants on one ecosystem service may propagate directly or indirectly to other corresponding services (Kinzig *et al.,* 2007; Le Maitre *et al.,* 2011). As a consequence of these changes, the delivery of ecosystem services and human wellbeing is affected (Richardson and van Wilgen, 2004; Pejchar and Mooney, 2009).

### 2.7.1. Impacts on ecosystem services delivery: A human well-being perspective

The impacts of invasive alien plants on national resources, such as water, grazing and biodiversity have been estimated to cost South Africa more than R 4 billion annually, most of which (70%) is attributed to reductions in water resources in the grassland and fynbos biomes (van Wilgen *et al.,* 2008; De Lange and van Wilgen, 2010).  Invasive Australian acacias have been associated with high water intake with negative implications for river flow (Le Maitre *et al.,* 1996; Le Maitre *et al.,* 2000; Le Maitre *et al.,* 2002) which in turn translates to reduced water availability for various socio-economic uses, including agricultural related activities, conservation and for domestic uses (Gorgens and van Wilgen, 2004). About 2.9% (1 444 million cubic meters) of the estimated annual water runoff in South Africa is utilised by alien invaders (Le Maitre, *et al.,* 2013). According to a biome scale study on selected ecosystem services (van Wilgen *et al.,* 2008), invasive alien plants have been estimated to reduce South Africa's grazing capacity at biome scale level by 1% under current levels of invasion, with an anticipated 71% loss in grazing capacity if

invasions are allowed to reach their full potential (van Wilgen *et al.,* 2008). In the Western Cape Province *Prosopis* species have been reported to reduce the grazing capacity in the Nama karoo biome by 65% when compared to uninvaded sites (Ndhlovu *et al.,* 2011). Change in species richness associated with alien plant invasions, measured using the Biodiversity intactness index (Scholes and Biggs, 2005), was reduced by between 71% and 89% (van Wilgen *et al.,* 2008). Furthermore, higher intensity fires fuelled  by high density invasive alien plants resulted in altered soil structure with negative implications for ecosystem services such as water repellency, decreased soil stabilisation, and promoted soil erosion which often leads to the siltation of rivers and dams (van Wilgen and Scott, 2001; Holmes, 2001; Le Maitre *et al.,* 2011). Communities surrounded by heavily invaded ecosystems are exposed to a greater degree of risk of fire and more extensive damage if fires do occur (Smita, 1998; Le Maitre *et al.,* 2014).

## 2.7.2.  Perceived impacts on rangeland structure and productivity

Invasion of natural rangelands by invasive alien plants is considered a major threat to the productivity of these ecosystems and to the overall livestock production systems of South Africa (Milton *et al.,* 2003; Richardson and van Wilgen, 2004; van Wilgen *et al.,* 2008; Ndhlovu *et al.,* 2011). Levels of invasion in these ecosystems vary from one region to another according to disturbance regimes, dominant vegetation types, resource availability as well as past and current land management strategies (Jauni and Hyvonen, 2010). Studies have reported that tree invasion can alter ecosystem dynamics, and reduce quality and quantity of forage for grazers (Pejchar and Mooney, 2009; Ng'weno *et al.,* 2009; Ndhlovu *et al.,* 2011). Below is an

extrapolation of how invasive acacia species can potentially impact on rangelands ecosystem dynamics.

Invasive alien plants can directly or indirectly alter native vegetation composition through alterations in ecosystem properties and processes such as nutrient cycling, water availability, soil stability and fire regimes (Vitousek, 1990; D'Antonio and Vitousek, 1992; Richardson *et al.,* 2000; Mack *et al.,* 2000). Invasive alien tree species such as Australian acacias are fast growing trees with higher above ground and below ground biomass volumes than the native species. This allows them to dominate competitive interactions over indigenous species for both above (including light) and below ground resources (including soil nutrients and water) (Le Maitre *et al.,* 2002; Yelenik *et al.,* 2004; Yelenik *et al.,* 2007; Morris *et al.,* 2011). Acacia biomass can alter habitat suitability for indigenous vegetation by reducing light penetrability which changes the microclimate below the canopy tree canopy (Holmes and Cowling, 1997; Morris *et al.,* 2011). The increased N-rich litter fall rate, as a result of high biomass from acacia species, results in changes in N-cycling regimes which then has a negative feedback on the native plant community structure (D'Antonio and Vitousek, 1992; Holmes and Cowling 1997; Mack *et al.,* 2000). On the ground level, residues from acacia species have been reported to produce strong phenolic compounds that are introduced into the soil, affecting indigenous seed germination and plant growth (Fatunbi *et al.,* 2009).

In natural ecosystems, plant community structure and species composition are related to soil properties, including soil structure and soil nutrients (Richardson *et al.,* 1997). Invasive alien plants, particularly the N2 fixing species change nutrient concentration and soil pH (Musil and Migdely, 1990; D'Antonio and Vitousek, 1992;

Mack *et al.,* 2000; Yelenik *et al.,* 2004; Yelenik *et al.,* 2007). *Acacia mearnsii* has been reported to increase P and N concentrations when compared to native vegetation (Witkowski, 1991; Yelenik *et al.,* 2004) whereas *A. saligna* has been reported to increase $Mg^+$ and $Ca^+$ concentrations in the soil (Yelenik *et al.,* 2007). High leaf biomass, from N2 fixing species, translates into increased plant litter, therefore N-rich plant litter from Australian acacias results in increased N content of the top soil below the tree canopy (Yelenik *et al.,* 2004). The deep rooting system of invasive alien trees has the potential to access unused soil nutrients in the deeper layers of the soil and redistribute them to the soil surface (Yelenik *et al.,* 2004; Morris *et al.,* 2011). Hence the nutrient dynamics of whole ecosystems are affected (Witkowski 1991; Stock and Allsopp, 1992; Yelenik *et al.,* 2004). Plant litter from invasive alien plant species differs in chemical composition from that of the native species. Therefore changes in soil chemical composition could also reduce microbial activity in the soil (Boudiaf *et al.,* 2013).

The high water intake by these species has been associated largely with high leaf biomass when compared to native species (Le Maitre *et al.,* 2011; Morris *et al.,* 2011). Evapotranspiration from *A. mearnsii* stands was estimated to be 13-51% higher than native vegetation (Le Maitre *et al.*, 2002). Previous studies have reported decreased soil moisture content under acacia invaded soils when compared to native vegetation (Dye and Jarmain, 2004; Gaertner *et al.,* 2011).

Fire, as a natural disturbance, forms an integral component of ecosystem functioning. This cyclical phenomenon is to a large extent driven by the availability of fuel material. Therefore, increases or decreases in fuel will determine the intensity and frequency of fires. Large quantities of above ground biomass, such as the

volumes of biomass under dense Australian acacias stands, increase the extent and severity of fires (Mack, 1986). This advantages further invasion, as Australian acacia seeds are stimulated by fire (Vitousek, 1990). Therefore the altered fire regime creates more opportunity for invasive alien trees to exploit ecosystems (Matthew *et al.,* 2004).

## 2.8. RESTORATION OF INVADED ECOSYSTEMS IMPLICATIONS FOR MANAGEMENT

The degradation of ecosystems by invasive alien plants has received considerable attention worldwide (Byers *et al.,* 2002; Mooney, 2005; van Wilgen *et al.,* 2008). Responses to invasion include strategies to mitigate the impacts of invasion (van Wilgen *et al.,* 2011). Ecosystem restoration and the control of alien invasions are fundamental to the sustainable management of natural resources and environment. In South Africa, the issue of invasive alien plants has seen the establishment of the WfW programme.  This is a national scale alien plant control programme which is run by the Department of Environment Affairs and is focussed on controlling the spread of alien species (van Wilgen *et al.,* 2011). The primary goal of the programme and the rationale behind its establishment was to secure threatened water resources through the clearing of invasive alien plants countrywide, and at the same time to address poverty through the creation of employment (van Wilgen *et al.,* 1998; 2011; Le Maitre *et al.,* 2000). The project rests on a further premise that clearing of invasive alien plants will restore ecosystems to their initial conditions. The programme, to a large extent, has employed a combination of mechanical and

chemical methods for clearing. Furthermore, alongside the programme, several developments have emerged, including use of biological control agents, which provide a more sustainable way of controlling invasive alien plants (Holmes *et al.,* 2000; Zimmermann *et al.,* 2004), the introduction of legal requirements for landowners to deal with the problem and payment for ecosystem services approaches that generate funding for the programme (Turpie *et al.,* 2008).

Ecosystem restoration is a complex process and requires one to understand the underlying drivers, invader and ecosystem characteristics and the extent to which an ecosystem is transformed before deciding on the clearing technique to be used (Richardson *et al.,* 2007). Research has shown that while some areas can recover following the removal of invasive aliens without further interventions (Blanchard and Holmes, 2008; Ruwanza *et al.,* 2012) other areas require additional inputs and actions. The application of correct clearing techniques is crucial, as cleared environments can experience secondary problems during the clearing process, which may result in further invasion by the same or other invasive species (Holmes *et al.,* 2000; Holmes, 2001). For instance, Blanchard and Holmes (2008) found that in treatments where *A. mearnsii* was felled and removed, there was a 97% ecosystem recovery rate. Whereas where a felling and burn technique was applied, fire killed the native seed bank, and promoted the resprouting of *A. mearnsii.* Ruwanza *et al.,* (2012) reported a positive trajectory towards recovery of structure and composition after *Eucalyptus camaldulensis* was felled and burnt on the site. It has also been suggested that in other areas active restoration (including planting of native species) is required to enhance ecosystem recovery (Galatowitsh and Richardson, 2004; Beater *et al.,* 2008; Gaertner *et al.,* 2011).

Several studies, including Blanchard and Holmes (2008), Morris *et al.,* (2008), Ndhlovu *et al.,* (2011), and Ruwanza *et al.,* (2012), have demonstrated the benefits associated with clearing of invasive alien plants on ecosystem structure and function, including  the re-establishment of indigenous species and hence the re-initiation of lost ecosystem services. In some instances the legacy of soil nutrients after the clearing of invasive species has been shown to improve grazing capacity by more than 50% when compared to the uninvaded land (Ndhlovu *et al.,* 2011). Such conditions could be explained by increased levels of soil nutrient availability associated with nutrient rich litter from the invasive tree (Witkowski, 1991; Yelenik *et al.,* 2004; Yelenik *et al.,* 2007; Gaertner *et al.,* 2011) which resulted in the dominance of more palatable perennial species. However, in some cases high level of soil nutrients (Witkowski, 1991; Yelenik *et al.,* 2004; Yelenik *et al.,* 2007; Gaertner *et al.,* 2011) resulting from nutrient rich litter, can lead to an undesirable situation of reinvasion by the same invasive species (Beater *et al.,* 2008; Witkowski and Garner, 2008).

Therefore, clearing programmes need a solid scientific basis from which to initiate clearing activities. This requires investigations into the underlying drivers and impacts of invasion, levels of invasion and the types of invading species and measuring success of past clearing interventions.

## 2.9.  CONCLUSIONS

The ecosystem services framework has, over the last two decades, become a popular approach with which to examine both the benefits supplied by natural ecosystems and the consequences of changing natural environments (MA, 2005). A variety of ecosystem services are supplied by natural rangelands, of which the production of grazing is of prime importance. Also acknowledged is the fact that natural ecosystems are being degraded as a result of multiple complex drivers (Hoffman and Ashwell, 2001). Invasive alien plants have been documented as one of these key drivers, or causes, of ecosystem degradation (Richardson *et al.*, 2007; Le Maitre *et al.*, 2011) and a variety of programmes and response measures are currently in place to address invasion in South Africa (van Wilgen *et al.*, 1998; 2011). However, our understanding of invasive alien plants is incomplete, particularly with regard to their impacts on ecosystem services (van Wilgen *et al.*, 2008). Few studies have examined invasion effects on natural rangelands processes and function and ultimately service delivery, or how invasion translates into changes in livestock production (Ndhlovu *et al.*, 2011). Nor do we have a good understand of the benefits of clearing operations within these environments. This study takes steps towards addressing these identified gaps in our understanding.

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**CHAPTER 3**

**Exploring the spatial relationship between high value grazing areas and invasion by Wattle species at a national level**

## 3.0.    ABSTRACT

Natural ecosystems in South Africa provide a variety of ecosystem services. The provision of grazing by natural rangelands has arguably been one of the key ecosystem services in this region for the last 2000 years. However, many of South Africa's natural ecosystems have been invaded by invasive alien plants, which typically results in a decline of these ecosystem goods and services. Australian wattle species (*Acacia dealbata, Acacia decurrens and Acacia mearnsii*) have been described as the most problematic of all invasive aliens in this regard. The focus of this study was on determining the extent of invasion in association with potential carrying capacity of natural ecosystems at both national and magisterial district scales. I used a geographic information system (GIS) to explore the degree of spatial overlap between invasion by Australian wattle species and key grazing assets at both the national and magisterial district scale. I developed a reclassified grazing layer that created four grazing class as follows: *high* (3-6 ha/LSU), *moderate* (7- 15 ha/LSU), *low* (16-30 ha/LSU), and *very low* (>30 ha/LSU). Wattle invasion, expressed in stems per hectares, ($ha_{stem}$), were grouped and classified into four classes indicating level of invasion: *Very scattered* (0 – 225), *scattered* (>225 – 1200), *moderate* (>1200 – 4300), and *dense* (>4300 – 5699). I found the level of overlap between wattle invasion and high grazing potential areas at the national

scale to be relatively low ranging between 0.2 and 8.6%. At the magisterial district scale the extent of wattle invasion into high grazing potential areas across the country ranged from 1.4% to 35% of the total natural grazing land, for the most invaded district. The results indicate that both scattered and moderate levels of invasion contribute the most to total area invaded. The implication of this overlap is that areas that could be used for intensive livestock production are compromised by the presence of invasive alien tree species. Therefore, livestock production, and other services associated with these ecosystems including water resources, and local biodiversity could be negatively affected by invasive wattles species as well. There is a need to advance our level of understanding of the impacts of invasive alien plants at a national scale. In particular, we need to understand how vegetation composition and soil resources that support grazing provision change under different levels of invasion. Priorities for clearing operations should target ecosystem service providing areas.

## 3.1.  INTRODUCTION

Ecosystems provide a variety of services and benefits to people (Constanza *et al.,* 1997). The provision of grazing for livestock by natural ecosystems has arguably been one of the key ecosystem services in the southern African region for the last 2000 years (MA, 2005). Constituting approximately 70% of South Africa's land surface, natural rangelands are a vital source of forage for livestock.  Livestock production contributes approximately 50% of the total value of national agricultural output and thus remains the single most important agricultural activity in the country

(DAF, 2012). Livestock are an important livelihood component of many households in communal areas. They provide households with an array of goods and services including milk, meat, manure, draught power, dung, income security and are used for ceremonial services and customary practices (Cousins, 1996; Shackleton *et al.,* 2005). Livestock is even an important human wellbeing component for households that do not own livestock as they are used in bride-wealth payments (ilobolo), and loaning schemes for co-operative draught power (Cousins, 1996; Shackleton *et al.,* 2005). However, there are numerous challenges facing the livestock production sector in South Africa. These include diseases, drought and other forms of natural disasters (Swanepoel *et al.,* 2008). Furthermore, rangeland degradation and non-sustainable livestock management practices are a critical challenge that threaten the sustainability of the livestock industry (Hoffman and Ashwell, 2001; Mansour *et al.,* 2013).

Invasion by invasive alien plants is associated with and exacerbated by degradation (van Wilgen *et al.,* 2008; Ndhlovu *et al*, 2011). Invasive alien plants have been reported to have negative impacts on natural processes (Vitousek *et al.,* 1997; Sala *et al.* 2000; Richardson and van Wilgen, 2004), such as hydrological regimes (Obrist *et al.,* 2004), fire frequency and intensity (D'Antonio and Vitousek, 1992), and nutrient cycling, which in turn affect species richness, biomass and composition of native flora and fauna. These changes affect ecosystem productivity (Mack *et. al.,* 2000; Hooper *et. al.*, 2005), and as a consequence all organisms sustained by these systems. This includes the provision of grazing for livestock production (Hoffman and Ashwell, 2001; Ndhlovu *et al*, 2011).

South Africa has been invaded by many alien plant species with well recognized ecological and economic impacts (Richardson *et al.,* 1997, Richardson and Van Wilgen, 2004). About 13% of the country's total area is invaded by alien trees (Le Maitre *et al.,* 2013). According to several surveys, vegetation types and biomes vary in susceptibility to invasion by alien invasive plants (Henderson, 2001; Kotze *et al.,* 2010). The grassland and savanna biomes have been extensively invaded by Acacia species (*Acacia dealbata, Acacia decurrens*, and *Acacia meanrsii*), other trees (e.g. *Eucalyptus grandis, Jackaranda mimosifolia*) and various shrub species (e.g. *Chromolaena odorata, Lantana camara*) (Richardson and van Wilgen, 2004, van Wilgen *et al.,* 2008). In the Nama and Succulent Karoo, and the thicket biome of the Eastern Cape, the most prolific woody invaders include *Prosopis* species, and several cacti (*Opuntia* species), and saltbushes (*Atriplex* species) (Richardson *et al.,* 1997; Milton *et al.,* 1999, Ndhlovu *et al.,* 2011).

Despite the recognition of the key invading species across biomes, our understanding of the impacts of these invasions on grazing capacities has not been well documented. Initial studies have suggested that invasions can alter ecosystem dynamics (Pitmental, 2005; Blanchard and Holmes, 2008; Ruwanza *et al.,* 2012). For example, when tree species invade grasslands, this often results in the reduced quality and quantity of forage (Pejchar and Mooney, 2009; Ng'weno *et al.,* 2009; Ndhlovu *et al.,* 2011).

In this study I focused on wattle *species (Acacia dealbata, Acacia decurrens,* and *Acacia mearnsii),* as these species are documented as having invaded a wide range of grazing areas in the country (de Wit *et al.,* 2001; Nel *et al.,* 2004, van Wilgen *et al.,* 2008). The extensive nature of wattle invasion is expected to have concomitant

impacts on the ecosystem services provided by natural ecosystems in South Africa, most notably grazing provision (van Wilgen *et al.,* 2008). To start to understand these impacts, an understanding of where the key grazing areas are in South Africa (Scholes, 1998), and in relation to invasion, is essential. The objectives of this study were to explore the degree of overlap between invasion by wattle species and important or high capacity grazing areas at a national level using an overlap analysis technique. These objectives contribute to the broader aim of this thesis, which is to understand the extent to which the invasion by wattle species is impacting on livestock production in South Africa.

## 3.2.  MATERIALS AND METHODS

### 3.2.1. Sources of data

A range of spatial data sets, captured in a geographic information system (GIS), were used to identify magisterial districts with high grazing potential and those impacted by Wattle invasion. These included the South African magisterial districts map, the National Vegetation Map of South Africa (Mucina and Rutherford, 2006), the South African 1:250 000 maps of areas of homogeneous grazing potential (Scholes 1998) and the South African National Land Cover of 2000 (NLC 2000) (Fairbanks *et al.,* 2000). Magisterial districts were a focus because of their relative long term geographic stability. They also remain a key level at which agricultural policies are enacted, and because they are still used in the collection of agricultural census data, they enable comparisons of historical livestock statistics with current datasets (Hoffman and Ashwell, 2001). The National Invasive Alien Plant Survey (Kotze *et al.,* 2010), was used to determine degree of invasion for Wattle species in

South Africa. All raster layers used in this GIS analysis were converted to a 30 x 30 m grid.

### 3.2.2. Extracting natural areas

The focus of this study was on understanding grazing provision by natural ecosystems. Therefore, I restricted the analysis to the remaining natural areas within South Africa's magisterial districts utilisable for grazing. All areas identified as natural in the NLC 2000 database were extracted and overlain with the South African magisterial districts. This resultant map provided the basis for the invasive wattle overlap analysis.

### 3.2.3. Identifying high grazing potential magisterial districts

The Homogenous Grazing Potential layer developed for South Africa by Scholes (1998) was used to identify the carrying capacity of natural vegetation. Scholes (1998) developed the layer by integrating a number of variables that govern livestock and plant relationships. These included the relationship between climate, soil type and vegetation with long term observed stocking rates for both livestock and African wildlife systems. The homogenous areas are defined as units of land area with the potential to support a similar number of foraging animals under current and previous climatic conditions in a natural state without any human intervention (Scholes, 1998).

Data were converted to express the number of hectares needed to support a Large Stock Unit (LSU). The LSU concept is defined in the section below. I then grouped and reclassified the Scholes (1998) grazing class data layer into four categories:

*high* (3-6 ha/LSU), *moderate* (7- 15 ha/LSU), *low* (16-30 ha/LSU), and *very low* (>30 ha/LSU). This map was then overlaid with the derived natural land cover map for South Africa's magisterial districts. The amount of natural grazing land (ha) for each of the four categories within a district was calculated and expressed as a percentage of the district's natural area. The end result indicated the grazing potential of natural areas within South Africa's magisterial districts.

### 3.2.4.  Identifying magisterial districts invaded by wattle

The National Invasive Alien Plant Survey (NIAPS) database (Kotze *et al*, 2010) was used to map the distribution and extent of invasion by three Australian *Acacia species: Acacia dealbata, Acacia decurrens* and *Acacia mearnsii*. These species are collectively referred to as wattle in the NIAPS database since they were very difficult to tell apart during the mapping exercise. I extracted the occurrence of these wattle species from the NIAPS database and created a new wattle map that could be used to assess the extent of wattle across South Africa's magisterial districts. The wattle values, expressed in stems per hectares, ($ha_{stem}$), were grouped and classified into four classes, indicating level of invasion, based on Le Maitre and Versfeld (1994). The classes were: *very scattered* (0 – 225 $ha_{stem}$), *scattered* (>225 – 1200 $ha_{stem}$), *moderate* (>1200 – 4300 $ha_{stem}$), and *dense* (>4300 – 5699 $ha_{stem}$). The natural land cover map for South Africa's magisterial districts was overlaid with the re-classed wattle map using a GIS. The amount of wattle invasion into natural areas, for each level of invasion, was calculated and expressed as the percentage of the district's natural area. The resultant map showed the extent and spread of wattle across South Africa's magisterial districts for the remaining natural areas.

### 3.2.5.  Grazing areas impacted by invasion

In order to identify areas of grazing importance that are invaded by wattle, I combined the output of both processes described above. The combined output reflected the amount of high grazing potential land invaded by wattle. These areas were expressed as the percentage of the total natural potential grazing area for each district.

## 3.3.  RESULTS

### 3.3.1.  Identifying high grazing potential magisterial districts

The districts with the highest grazing potential were found to be situated in the eastern and northern half of the country (Figure 3.1) (Table 3.1). The Wakkerstroom and Vrede districts, with approximately 70% and 50% of the total natural area, respectively, have high grazing potential. Districts with the greatest amount of natural vegetation in the high grazing potential class, (Figure 3.2), fall within the grassland biome. King William's town, Mqanduli and Ngqeleni are exceptions, falling predominantly into the savanna biome. The Stutterheim district has virtually equal proportions of grassland and savanna (Figure 3.2).
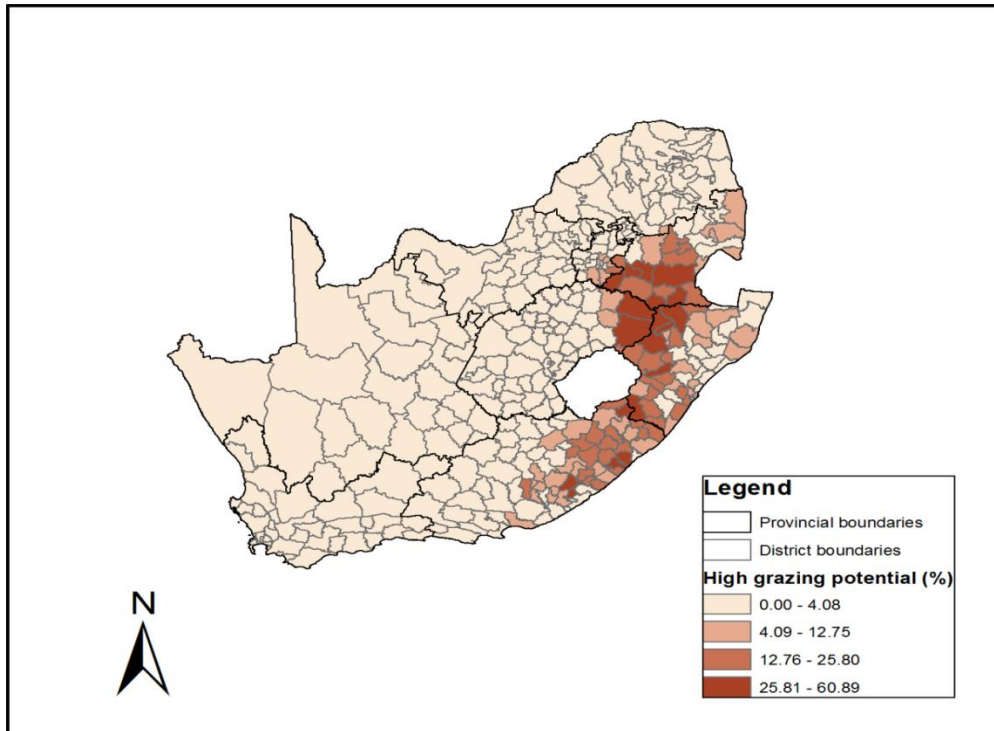
**Figure 3.1:** Magisterial districts expressed according to their percentage area of remaining natural vegetation with high grazing potential (Derived from Scholes, 1998).

**Table 3.1:** Selected magisterial districts with the greatest proportion of land area in the high potential grazing class. These high potential grazing classes are expressed as a percentage of the total district area. Also shown are the percentage area contributions of the range: low, very low potential grazing classes and the total grazing is the sum of different classes and highlights the percentage natural area as a proportion of the district area.

| District name | High | Moderate | Low | Very Low | Total grazing |
|---|---|---|---|---|---|
| Kliprivier | 33.2 | 56.9 | 0.5 | 0.0 | 90.7 |
| King William's Town | 35.1 | 42.8 | 11.7 | 0.0 | 89.6 |
| Utrecht | 30.7 | 54.7 | 0.2 | 0.0 | 85.5 |
| Wakkerstroom | 60.9 | 15.7 | 0.8 | 0.0 | 77.4 |
| Vrede | 50.9 | 26.1 | 0.0 | 0.0 | 77.0 |
| Mooirivier | 27.7 | 44.4 | 4.7 | 0.0 | 76.9 |
| Stutterheim | 27.4 | 41.3 | 8.2 | 0.0 | 76.9 |
| Newcastle | 42.2 | 28.1 | 0.0 | 0.0 | 70.2 |
| Mount Currie | 39.4 | 28.7 | 0.8 | 0.0 | 69.0 |
| Dannhauser | 32.9 | 34.8 | 0.0 | 0.0 | 67.7 |
| Harrismith | 42.2 | 21.4 | 0.4 | 0.0 | 64.0 |
| Ngqeleni | 28.2 | 23.6 | 5.8 | 0.0 | 57.5 |
| Ermelo | 28.0 | 21.7 | 1.8 | 0.0 | 51.4 |
| Balfour | 37.4 | 13.8 | 0.0 | 0.0 | 51.3 |
| Mqanduli | 32.3 | 18.6 | 0.0 | 0.0 | 50.9 |
| Highveld Ridge | 39.7 | 2.1 | 0.0 | 0.0 | 41.8 |
| Volksrust | 31.3 | 3.9 | 0.0 | 0.0 | 35.2 |
| Bethal | 29.3 | 2.7 | 0.0 | 0.0 | 32.1 |
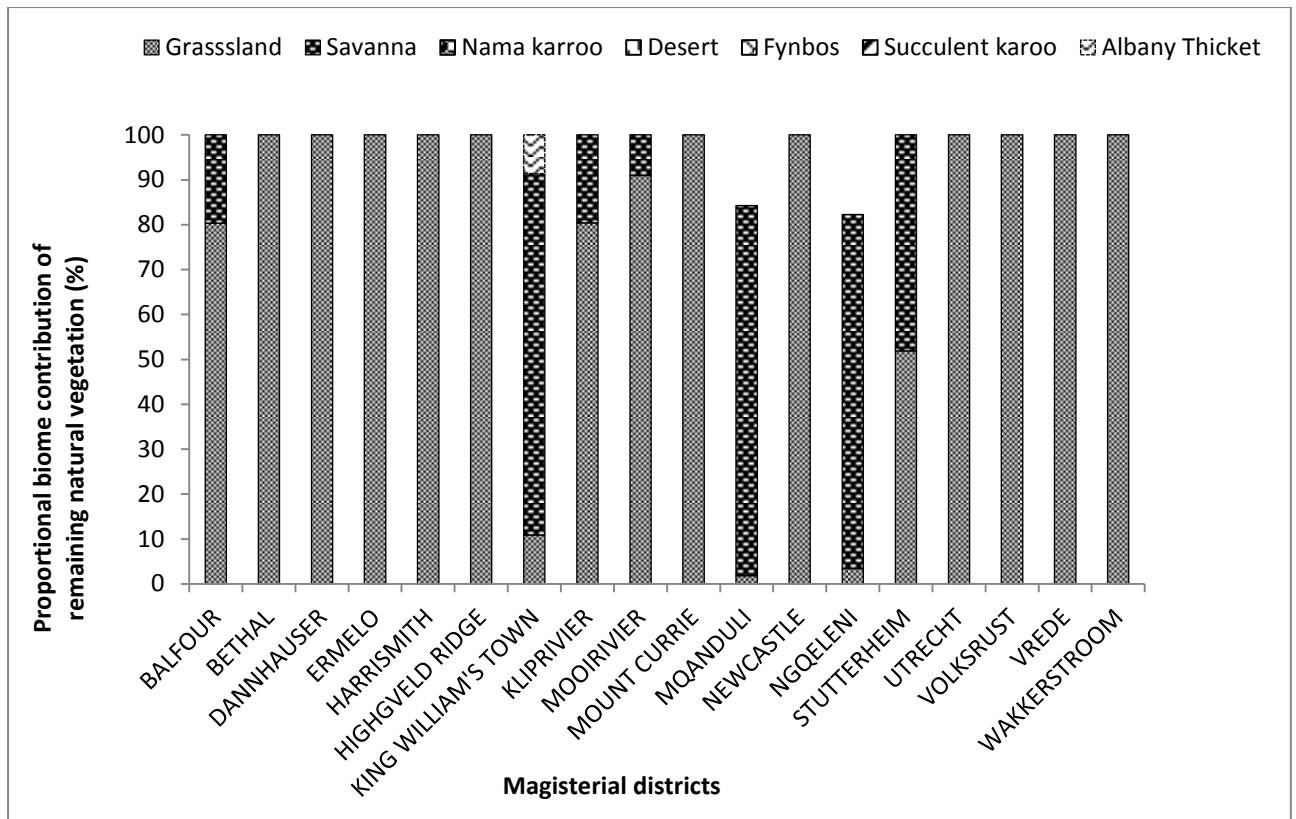
**Figure 3.2:** Magisterial districts with the greatest proportion of high potential natural grazing, expressed according to their biome contributions.

### 3.3.2. Identifying districts invaded by wattle

The distribution and density of wattle invasion into the remaining natural areas is presented as a percentage of a magisterial district area in Figure 3.3. The magisterial districts that are most prone to wattle infestation are mostly in the eastern and northern parts of the country, and appear closely aligned with the moderate and high grazing areas (see Figure 3.3).
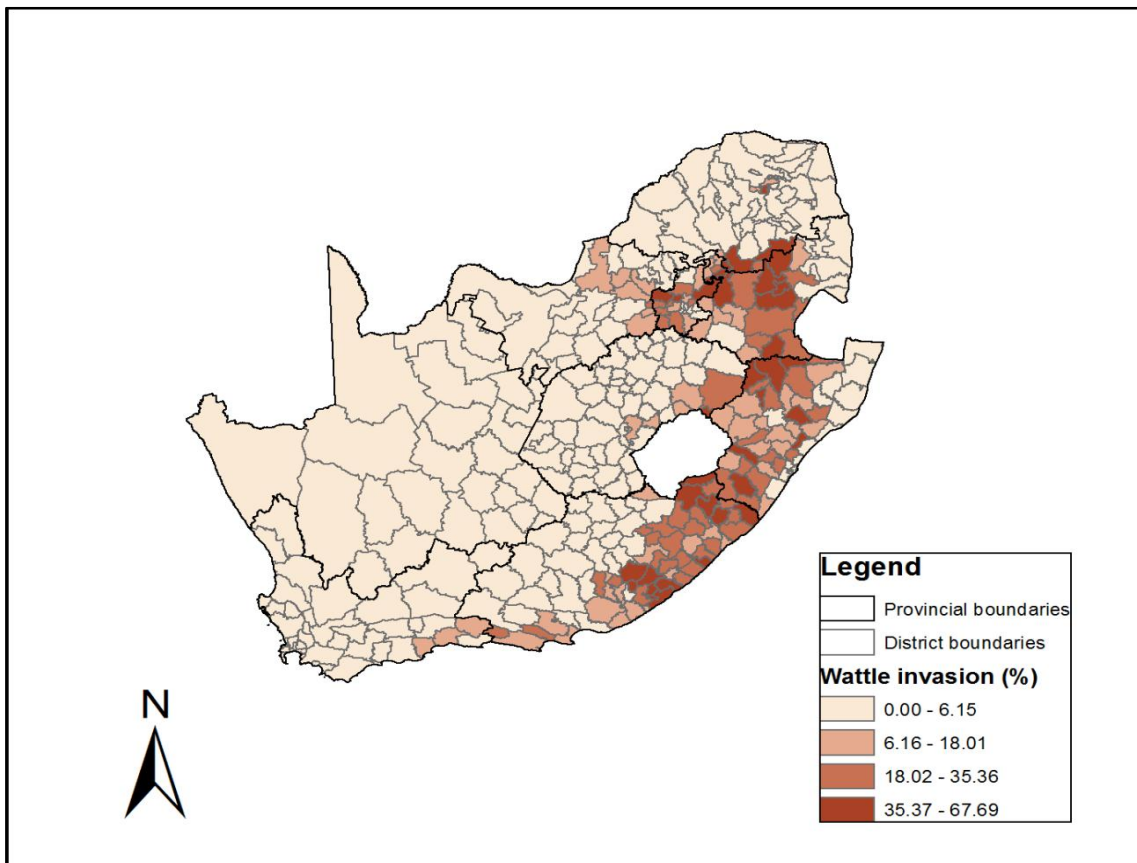
**Figure 3.3:** Magisterial districts expressed according to their percentage wattle invasion (based on Kotze, *et al.,* 2010).

Thirty five magisterial districts were found to have more than 35% of their total remaining natural ecosystems invaded by wattles when invasion classes were combined. The extent of invasion expressed according to each class (very scattered, scattered, moderate and densely) for each of the 35 districts is presented in Table 3.2. Four districts have dense wattle invasions, the extent of which varies from over 6% of the remaining natural area to less than 0.2% of the percent area.

**Table 3.2:** Magisterial districts with more than 35% of the remaining natural ecosystem areas invaded by wattle, expressed according to four invasion classes (dense, moderate, scattered and very scattered). Proportional areas of different classes were summed to give total invaded areas for each magisterial district. The districts are ranked according to the total invaded natural areas.

| Districts | Dense | Moderate | Scattered | Very Scattered | Total invaded |
|---|---|---|---|---|---|
| King William's Town | 0.0 | 24.7 | 37.0 | 6.0 | 67.7 |
| Mkobola | 0.0 | 25.4 | 30.1 | 8.8 | 64.3 |
| Belfast | 0.0 | 24.2 | 20.7 | 18.5 | 63.4 |
| Kwabhaca | 4.7 | 22.3 | 25.3 | 6.2 | 58.4 |
| Mdantsane | 0.0 | 1.8 | 33.0 | 21.8 | 56.6 |
| Waterval Boven | 0.0 | 23.3 | 14.6 | 17.3 | 55.3 |
| Moutse | 0.0 | 6.8 | 40.4 | 6.6 | 53.8 |
| Komga | 0.0 | 18.4 | 24.5 | 9.2 | 52.1 |
| Stutterheim | 6.2 | 13.0 | 28.0 | 5.0 | 52.1 |
| Maluti | 0.0 | 22.5 | 28.3 | 0.5 | 51.3 |
| Carolina | 0.0 | 18.9 | 22.3 | 8.8 | 50.1 |
| Keiskammahoek | 0.0 | 7.5 | 0.3 | 41.9 | 49.7 |
| Witzieshoek | 0.0 | 0.0 | 48.9 | 0.2 | 49.1 |
| East London | 0.0 | 26.8 | 15.6 | 6.7 | 49.0 |
| Mount Fletcher | 0.0 | 17.1 | 25.1 | 3.5 | 45.8 |
| Lydenburg | 0.0 | 9.2 | 16.0 | 20.4 | 45.6 |
| Utrecht | 0.0 | 18.0 | 16.3 | 10.1 | 44.4 |
| Mpendle | 0.0 | 0.0 | 30.6 | 13.1 | 43.7 |
| Wakkerstroom | 0.0 | 20.7 | 17.2 | 5.5 | 43.3 |
| Tabankulu | 0.0 | 1.5 | 26.3 | 15.1 | 43.0 |
| Groblersdal | 0.0 | 0.3 | 34.6 | 6.7 | 41.5 |
| Newcastle | 2.5 | 20.8 | 16.4 | 1.9 | 41.5 |
| Krugersdorp | 0.0 | 22.6 | 4.3 | 13.9 | 40.7 |
| Glencoe | 0.0 | 0.0 | 29.3 | 10.8 | 40.1 |
| Nkandla | 0.0 | 6.9 | 32.4 | 0.8 | 40.1 |
| Cathcart | 0.0 | 12.2 | 22.3 | 4.5 | 39.1 |
| Randburg | 0.0 | 0.0 | 19.3 | 19.2 | 38.5 |
| Bronkhorstspruit | 0.0 | 9.7 | 22.0 | 6.7 | 38.4 |
| Xhora | 0.0 | 7.1 | 23.5 | 7.2 | 37.8 |
| Paul Pietersburg | 0.0 | 8.4 | 19.3 | 9.4 | 37.2 |
| Witbank | 0.0 | 7.8 | 24.0 | 5.4 | 37.2 |
| Maphumulo | 0.0 | 0.0 | 15.4 | 21.5 | 36.9 |
| Soekmekaar | 0.0 | 0.0 | 1.6 | 34.8 | 36.5 |
| UMzimkhulu | 0.0 | 6.2 | 19.4 | 10.9 | 36.5 |
| Bizana | 0.0 | 16.9 | 16.6 | 2.8 | 36.2 |

### 3.3.3.  Grazing areas impacted by wattle invasion

To quantify the potential loss of grazing resulting from invasion by wattle species, the area of invasion in each of the invasion classes was calculated for each of the grazing classes for the country (Table 3.3).

The overlap results, presented in Figure 3.4, show the distribution and extent of total wattle invasions, in remaining natural areas, with high grazing potential. The total proportional overlap at a magisterial district level ranges from 35% to down to 1.4%. The high proportional overlap is explained by scattered and moderate levels of invasion (Table 3.4). Magisterial districts with more than a 10% proportional overlap are listed in Table 3.4, and only two of these districts (i.e. Newcastle and Stutterheim) are densely invaded.

**Table 3.3:** Percentage overlap between wattle invasion (expressed according to each invasion class) and grazing potential areas (expressed according to each class) expressed according total natural area.

| Grazing | Wattle invasion | | | | |
|---|---|---|---|---|---|
| | **Dense** | **Moderate** | **Scattered** | **Very scattered** | **Total** |
| **High** | 0.03 | 0.80 | 1.52 | 0.59 | 2.94 |
| **Moderate** | 0.03 | 1.95 | 4.34 | 2.27 | 8.58 |
| **Low** | 0.00 | 0.21 | 0.82 | 0.49 | 1.52 |
| **Very low** | 0.00 | 0.03 | 0.08 | 0.04 | 0.15 |
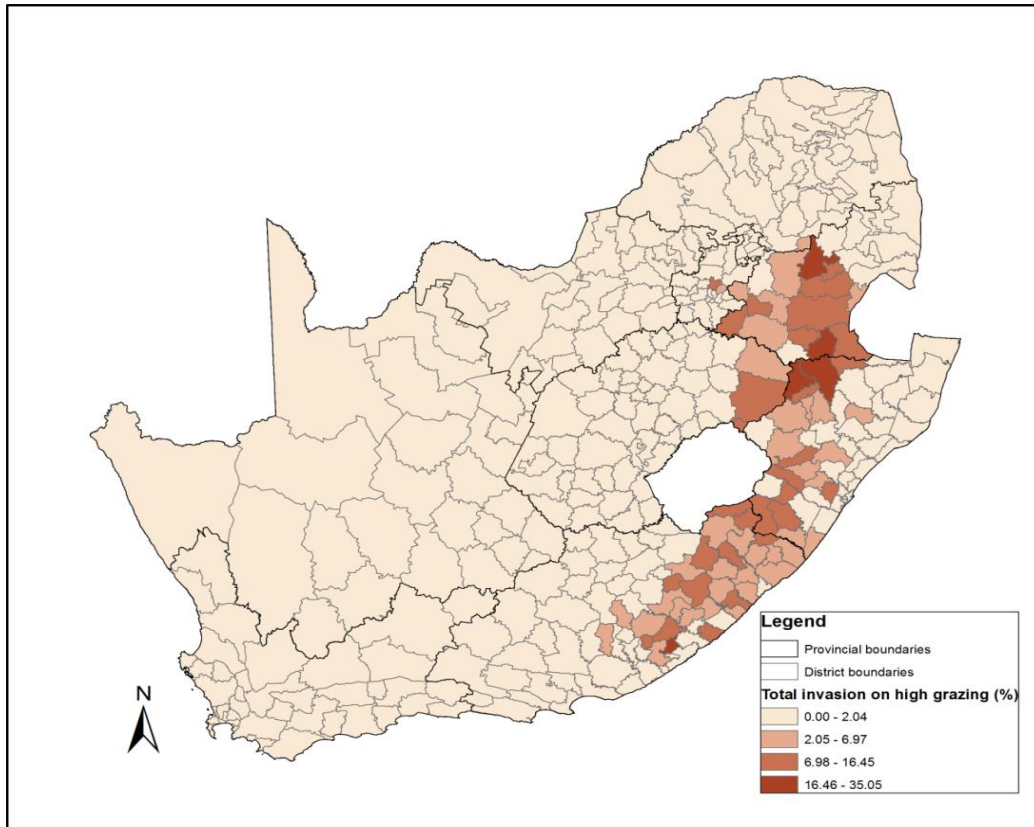
**Figure 3.4**: The percentage overlap between total invasion by wattle and high grazing potential areas expressed according to magisterial districts.

**Table 3.4:** Percentage overlap between wattle invasion (expressed according to each invasion class) and high grazing potential area expressed according to magisterial districts. Shown are the magisterial districts with more than 10% area overlap. The districts are ranked according to total invaded natural areas.

| District | Dense | Moderate | Scattered | Very scattered | Total invaded |
|---|---|---|---|---|---|
| Wakkerstroom | 0.0 | 15.2 | 14.6 | 5.3 | 35.1 |
| King William's Town | 0.0 | 11.7 | 10.8 | 1.6 | 24.1 |
| Newcastle | 1.6 | 14.6 | 7.1 | 0.6 | 23.8 |
| Utrecht | 0.0 | 9.3 | 6.0 | 5.4 | 20.6 |
| Belfast | 0.0 | 5.4 | 8.5 | 4.3 | 18.1 |
| Paul Pietersburg | 0.0 | 3.2 | 8.8 | 4.4 | 16.5 |
| Dannhauser | 0.0 | 0.0 | 14.5 | 0.6 | 15.2 |
| Stutterheim | 1.2 | 2.6 | 9.6 | 1.5 | 15.0 |
| Mount Currie | 0.0 | 5.6 | 5.7 | 3.1 | 14.3 |
| Harrismith | 0.0 | 3.3 | 9.4 | 0.4 | 13.1 |
| Ermelo | 0.0 | 2.2 | 8.7 | 1.0 | 11.9 |
| Carolina | 0.0 | 4.4 | 6.0 | 1.5 | 11.8 |
| Mount Ayliff | 0.0 | 0.0 | 10.1 | 1.6 | 11.7 |
| UMzimkhulu | 0.0 | 4.3 | 4.7 | 2.2 | 11.2 |
| Waterval Boven | 0.0 | 5.0 | 2.7 | 3.2 | 10.8 |
| Mqanduli | 0.0 | 2.3 | 8.2 | 0.1 | 10.6 |
| Maluti | 0.0 | 2.4 | 8.0 | 0.1 | 10.5 |
| Engcobo | 0.0 | 7.2 | 2.7 | 0.6 | 10.5 |
| Mooirivier | 0.0 | 1.5 | 8.5 | 0.5 | 10.5 |
| Highveld Ridge | 0.0 | 0.0 | 10.0 | 0.3 | 10.3 |
| Amersfoort | 0.0 | 0.1 | 4.9 | 5.2 | 10.1 |

## 3.4.  DISCUSSION

### 3.4.1.  Grazing provision in the South African context

The results of this study show that areas with the highest carrying capacity and grazing potential are concentrated in the eastern half of the country, falling into the grassland and savanna biomes. The vegetation structure of both these biomes enables them to support large and small grazers, making these natural rangelands significant sites of ecosystem services provision underpinning this economically and

culturally important national agricultural activity. Grasslands are characterised by short to tall grasses, whereas savannas are defined by a mixture of trees and grasses (Scholes and Archer, 1997; Scholes, 1998; Mucina and Rutherford, 2006). The productivity of these areas is to a large extent enhanced by the availability of plant growth resources including soil water and fertile soils.  Both grassland and savanna biomes are underlain by higher soil moisture content when compared to the Nama-karoo and succulent karoo biomes which are the other biomes used for grazing in South Africa (Schulze, 1998; Mucina and Rutherford, 2006).

### 3.4.2. Impacts of invasive alien plants on grazing provision and implications for livestock production

There is a certain degree of overlap between areas of high grazing potential and those invaded by wattle species. This overlap appears to be relatively low when viewed as the proportion of the country's total remaining natural areas. However, only approximately 13% of the country's surface area has some degree of invasion, with most invasions concentrated in the riparian areas (van Wilgen *et al.,* 2001; Nel *et al.,* 2004; Kotze *et al.,* 2010; Le Maitre *et al.,* 2013). Despite the seemingly low level of overlap at national scale, some magisterial districts show a remarkable degree of overlap between high grazing potential and scattered to moderate levels of wattle invasion. The overlap between high grazing potential land and wattle invasion could be explained by ecological conditions including, high rainfall, that are dominant in the grassland and savanna biomes (Scholes, 1998; Henderson *et al.,* 2001; Dye and Jarmain, 2004; Nel *et al.,* 2004; Kotze *et al.,* 2010).

My approach in analysing the level of overlap between high grazing potential areas and wattle invaded areas does not give any measureable losses in terms of grazing capacities. However, the implications of this overlap, when viewed in light of available evidence from previous studies (van Wilgen *et al.,* 2008; Ndhlovu *et al.,* 2011; Le Maitre *et al.* 2011), are that areas that could be used for high livestock production are under threat due to invasion by invasive alien tree species. Therefore, livestock production, and other services associated with these ecosystems including water resources, and local biodiversity, could be negatively affected by invasive wattles species (de Neergaard *et al.,* 2005; Shackleton *et al.,* 2007b; van Wilgen *et al.,* 2008). Furthermore, the different densities of invasion could be used to prioritise action. At present, alien clearing is prioritised on the basis of water and biodiversity. Given the importance of livestock production (e.g. for livelihood security and cultural practices), it too should inform prioritisation decisions. Enhancing the understanding of the ecology and dynamics of different densities of invasion would refine interventions.

The invasion of large alien acacia trees into natural rangelands, particularly in biomes such as grasslands which have few trees, interferes with ecological process that regulate the function and structure of these ecosystems including hydrological regimes and nutrient cycling (Davis *et al.,* 2000; Mack *et al.,* 2000; Sala *et al.* 2000; Yelenik *et al.,* 2004, 2007; Vila *et al.,* 2011). This often transforms the conditions of the environment to those that are more favourable for the establishment of the invading species, which may ultimately result in an invasive tree dominated ecosystem. As a consequence, indigenous species richness, biomass and composition of native flora and fauna are often impacted  (Vila *et al.,* 2006; Gaertner *et al.,* 2009). This is in turn likely to pose negative consequences for rangeland

services, including the current and future carrying capacity where palatable forage is lost and livestock access and mobility reduced (Kinzig *et al.,* 2007; van Wilgen *et al.,* 2008; Le Maitre *et al.,* 2011; Ndhlovu *et al*, 2011). Thickening of woody vegetation or the shift from scattered invasion to dense invasion also restricts and reduces access to other rangeland services such as water and habitat for wild herbivores (de Neergaard *et al.,* 2005; Shackleton *et al.,* 2007b). Furthermore, invasion that results in a general increase in ecosystem biomass, which is often the case with invasive *Acacia spp*, can lead to fast spreading, high intensity fires resulting in the loss of ground vegetation cover and the exposure of soil to  water and wind erosion (D'Antonio and Vitousek, 1992; van Wilgen and Scott, 2001; Le Maitre *et al.,* 2011). In addition to these impacts some species, such as *Lantana camara*, have been found to be poisonous to livestock and humans, while some invasive plants have morphological features that can harm livestock, such as sharp barbs (Binggeli *et al.,* 1998).

### 3.4.3.  Degradation as a driver: implications for management

The invaded areas highlighted in the results of this study were closely aligned to the areas that were classified as degraded in the National Review of Land Degradation (Hoffman and Todd, 2000). The National Review of Land Degradation focussed on soil degradation and veld degradation across the magisterial districts of South Africa (Hoffman and Todd, 2000). Here the most prominent areas in terms of soil and veld degradation (combined indices) were found in the communal areas of the former Ciskei, Transkei and KwaZulu-Natal, which were characterised by steep slopes, high mean annual temperature, and rural populations with low income (Hoffman and

Todd, 2000).  The areas identified in the overlap analysis as being of high grazing value and invaded, closely align with the areas that are more vulnerable to land (soil and rangeland) degradation as identified in this National Review (Hoffman and Todd, 2000). One cannot simply neglect the possibility of the linkages between the common causes of land degradation in these areas and invasion by alien plants. However, causes of land degradation are complex; therefore one cannot simply attribute the invasion of natural rangelands to a single degradation factor.  Predicted future precipitation changes, increasing for the grassland and savanna biomes (Meadows and Hoffman, 2003), coupled with other climate variables in South Africa, may exacerbate land degradation in areas that are already severely impacted. This study suggests that wattle species favour higher rainfall areas, and these predicted changes may promote further invasions and or increase the density of current invasions. A further factor for consideration is the increasing levels of $CO_2$ which could result in the increase of C3 trees and shrubs relative to a grass layer (Scholes and Archer, 1997; Scholes, 1998; Bond *et al.,* 2003; Meadows and Hoffman, 2003). This implies that as climate change becomes more evident, these areas will be more vulnerable to degradation through, among others things, invasion by alien plants, considering the fact that climate and rainfall are key drivers of rangeland vegetation dynamics (Scholes and Archer, 1997; Scholes, 1998; Bond *et al.,* 2003; Meadows and Hoffman, 2003).

There is also considerable evidence indicating that the unsustainable use of natural rangelands, which is mostly associated with communal areas, including heavy stocking, coupled with continuous grazing, may lead to the degradation of ecosystems (Hoffman and Todd, 2000; Snyman and du Preez, 2005; Scholes, 2009; Palmer and Bennett, 2013). Communal areas, created under apartheid, tend to be

heavily populated and heavily stocked, and this limited relative land availability results in overgrazing. In communal areas livestock are kept for a variety of reasons, and heavy stocking puts pressure on natural rangelands in these areas, and hence overgrazing is regarded as the main cause of rangeland degradation (Hoffman and Todd, 2000; Vetter, 2013). Overgrazing, as a results of heavy stocking, reduces the grass layer and therefore exposes the soil surface to trampling by grazers, which exacerbates the impacts of wind and water as drivers of erosion. Heavy grazing, particularly in savanna ecosystems, interferes with the competitive interaction for growth resources between woody and herbaceous plants by reducing herbaceous vegetation (Scholes and Archer, 1997). In addition to these impacts, the reduction of the herbaceous layer, due to heavy stocking, may result in less frequent and low intensity fires and as a result the effectiveness of fire to control woody vegetation is reduced. Therefore, where there is an onset of invasion by alien plants, changes that come with soil degradation and disturbance, could stimulate or increase an ecosystem's vulnerability to further invasions, increase levels of degradation and yet more losses in carrying capacity (Richardson, *et al.,* 1997; Scott *et al.,* 1998; Davis, *et al.,* 2000; Jauni and Hyvonen, 2010).

Therefore, if natural rangelands are going to provide continued grazing resources, then the sustainable use of this service must be applied in order to reduce disturbances that may lead to ecosystem degradation. For effective control of the spread and impacts of invasive alien plants on rangelands, the underlying factors that have resulted in ecosystem degradation need to be addressed. Furthermore, the restoration of degraded ecosystems should be coupled with the critical management of rangelands to ensure successful recovery and the prevention of further invasion by alien plants.

## 3.5.  CONCLUSIONS

South Africa's high grazing potential areas mainly occur on the eastern and northern parts of the country. My study found that invasive alien plant distributions overlap with these high potential grazing areas. This implies that invasive alien plants have the potential to threaten highly productive land if invasions continue unabated. The more rapid the invasion the more quickly South Africa's livestock production will be under threat. I believe that invasive alien plant control, which to date has been led or guided by water and biodiversity conservation concerns, should also be guided by the distribution of high potential agricultural areas. To do this we need to advance our level of understanding of the impacts of invasive alien plants at both national and local scales. Furthermore, we need to understand how vegetation composition and soil resources that support grazing provision change under different levels of invasion. Future research should focus on the careful selection of case studies that are critical for service delivery and are threatened by invasion. Such studies will allow us to in gain a better understanding of the impacts of invasive plants on services and benefits from an ecological perspective. This research needs to be carried out in such a way that allows us to up-scale the findings and use them in policy development, as well as in prioritising alien clearing. This chapter provides a starting point for identifying important areas for grazing services that are threatened by wattle invasion.

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**CHAPTER 4**

**Alien tree invasion into grassland ecosystems: impacts on rangeland condition and livestock production**

**4.0.   ABSTRACT**

This study investigated the ecological impacts resulting from the invasion by *Acacia mearnsii* (black wattle, an introduced Australian tree species) on rangelands, and the subsequent condition of these environments following clearing restoration activities. I located uninvaded, lightly invaded, densely invaded and cleared sites in a grassland ecosystem in the Eastern Cape, South Africa, and examined the impacts of these treatments on forage quality and quantity, and on soil resources. Invasion by *A. mearnsii* reduced grazing capacity by 56% and 72% on lightly and densely invaded sites respectively, whereas clearing improved grazing capacity by 66% within 5 years. Loss of grazing capacity during invasion was largely due to reduction in total basal cover (by up to 42%) and herbaceous biomass (from 5200 to 1200 kg ha$^{-1}$). Subsequent clearing of invaded sites allowed both basal cover and biomass to return to pre-invasion levels. With the exception of less coarse sand on cleared sites, there were no differences in soil physical properties (clay, silt, fine, medium). Soil moisture content was found to be significantly lower on densely invaded sites compared with lightly invaded and cleared sites. Following invasion by black wattle, plant litter increased from 1.3 to 4.2%, the carbon content of the soil increased from 2.0 to 4.0%, and nitrogen concentrations increased from 0.1 to 0.2% in response to nitrogen fixation by the acacia trees. Overall, these changes reduced grazing

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capacity, from 2 to 8 ha required to support one large livestock unit on uninvaded and densely invaded sites respectively. My findings provide a strong motivation both for preventing further invasions and for clearing existing invasions.

## 4.1.  INTRODUCTION

Invasive alien plants pose a significant threat to all components of biological diversity, from genes to whole ecosystems (Constanza *et al.,* 1997, Richardson *et al*, 2000). Invasive alien tree species and shrubs are known to threaten ecosystem functioning in various ways, for example through the reduction in stream flow and water availability (Vitousek *et al*, 1997; Le Maitre *et al.,* 2000, Pitmental, 2001; Pejchar and Mooney, 2009), loss of potentially productive land for cropping, and loss of grazing potential in rangelands (Scott *et al.,* 1998, van Wilgen *et al.,* 2008; Ndhlovu *et al.,* 2011). High densities of alien plant species can alter habitat suitability for native species, increase fire hazards as well as the extent of damage caused by fire, the costs of fire protection, and soil erosion following fires. Some invasive species are also toxic to humans and livestock (Smita, 1998; Binggeli, *et al.,* 1998).

Invasion of rangelands by alien plants is considered a threat to the agricultural production systems of South Africa (Milton *et al.,* 2003; Richardson and van Wilgen, 2004; Moyo and Fatunbi, 2010, Ndhlovu *et al.,* 2011). Livestock production in the communal areas and certain commercial areas is rangeland-based and therefore dependant on the state or condition of the rangeland for the provisioning of forage. Fast-growing *Acacia* species use excessive amounts of water and nutrients, potentially reducing the ability of these ecosystems to provide this service (de

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Neergard *et al.,* 2005; Moyo and Fatunbi, 2010). Invasive alien *Acaci*a species have the potential to convert rangelands into dense woodlands through the displacement of grasses and herbs (Nyoka, 2003; de Neergard *et al.,* 2005). However, little has been done to quantify these impacts on livestock production, particularly at local scales (Ndhlovu *et al*, 2011). This is important because it has clear implications for both land management and human wellbeing in rural areas.

Despite our lack of understanding relating to many of the impacts invasive alien species have on South African ecosystems (Richardson and van Wilgen, 2004; van Wilgen, 2008),  measures are currently being taken at a national level to control these species. The Working for Water (WfW) programme is a national scale alien plant control programme run by the Department of Environment Affairs focussed on controlling the spread of these species. The primary goal of the programme and the rationale behind its establishment was to secure threatened water resources through the clearing of invasive alien plants countrywide, and at the same time it addressed poverty through the creation of employment (van Wilgen *et al.,* 1998; Le Maitre *et al.,* 2000). The project rests on a further premise that clearing of invasive alien plants will restore ecosystems to their initial conditions.

*Acacia mearnsii* (an introduced tree from Australia) is one of the most prolific and extensive invaders of rangelands in the more mesic areas of South Africa particularly in the Eastern Cape, KwaZulu-Natal and Mpumalanga Provinces (Nel *et al.,* 2004; de Neergard *et al.,* 2005) (See chapter 2). Several studies have reported that the removal of *Acacia* species can facilitate re-colonisation by indigenous species and improve basal vegetation cover (Blanchard and Holmes, 2008; Ruwanza *et al.,* 2012). However, while some areas can recover following the removal of invasive

aliens without further interventions, in other areas active restoration is required to enhance ecosystem recovery (Beater *et al.,* 2008; Gaertner *et al.,* 2011). In some cases the high level of soil nutrients (Witkowski, 1991; Yelenik *et al.,* 2004; Yelenik *et al.,* 2007; Gaertner *et al.,* 2011) derived from the nutrient rich litter after clearing, can lead to the undesirable situation of reinvasion by the same species (Beater *et al*, 2008; Witkowski and Garner, 2008). Investigating the impacts of invasive alien plants and the recovery of rangelands from clearing of invasive alien plants will help quantify the damage caused by and benefits of clearing invasive alien plants (Ndhlovu *et al.,* 2011). While there are several studies reporting on the benefits of clearing and the recovery of some ecosystem components (Witkowski, 1991; Yelenik *et al.,* 2004; Yelenik *et al.,* 2007; Blanchard and Holmes, 2008; Gaertner *et al.,* 2011) these have not been well described or documented in the context of grazing capacities.

The aim of this chapter was to quantify the impacts of *A. mearnsii* invasions on range condition and livestock production. The objectives of the study were: (1) to determine the effects of *A. mearnsii* density on growth form dominance of indigenous vegetation species, and how this translates into impacts in forage quality and quantity; (2) to determine the effects of *A. mearnsii* invasion on soil resources and conditions (key drivers of ecological function) required to support grazing production; and (3) to determine the effects that clearing operations have had on the grazing resources.

## 4.2.  STUDY AREA

The study was conducted in the Stutterheim (32° 34' 0.84"S, 27° 25' 0.12"E) region of the Eastern Cape Province in South Africa. This area has a 40-year mean annual rainfall of 752 mm, with most of the rain falling in summer (October-March). Average minimum and maximum temperatures range from 14 to 25 $^o$C, in summer, and 7 to 18 $^o$C, in winter.

Deep, freely drained, highly weathered soils of the Karoo basin group typify the region. Soils are characterised by weakly developed lithosols, solonetzic soils interspersed with red clay (Mucina and Rutherford, 2006). Soil features associated with luvisoils, planosols, plintosols, vertisols and cambisols can be found in some places (IUSS Working Group, 2006).

Stutterheim's dominant vegetation type is Amathole Montane Grassland (Mucina and Rutherford, 2006). It is characterised as short grassland with a high species richness of forbs (especially *Helichrysum* and *Senecio* species). The dominant grass species include *Themeda triandra, Sporobolus africanus, Eragrostis chloromelus, E. curvula, Herteropogon contortus* and *Tristachya leucothrix* (Mucina and Rutherford, 2006), while the low shrubs *Chrysocoma ciliata*, and *Felicia filifolia* are also prevalent. The most prominent invaders into this vegetation type include *Acacia mearnsii* and *Acacia dealbata* (Mucina and Rutherford, 2006).

Land use within this region is dominated by livestock production with numerous cattle and sheep stud farms, commercial livestock farms as well as communal livestock areas. Stocking rates in this area range between 2 to 40 ha.LSU$^{-1}$ under optimal and poor conditions, respectively. The short grassland structure that

characterises some areas is a result of heavy grazing by livestock. The area also supports a forestry industry which includes large pine, gum and wattle plantations that are established on the slopes of the Kologha Mountains. More than 10 % of the landscape has been transformed to plantations and cultivation. In Stutterheim some of the farms which were previously under the ownership of commercial- and market-oriented farmers with better means have been transferred to previously disadvantaged famers, through the government initiated, Land Redistribution for Agricultural Development programme.  The LRAD programme was initiated to help improve the livelihoods of previously disadvantaged citizens through agricultural activities.

## 4.3.  METHODS

### 4.3.1.  Site selection

I investigated the effects of two levels of invasion, lightly invaded and densely invaded, and compared these with uninvaded and cleared sites. For each of these four classes or treatments I examined five sites (n = 20 sites) on commercial farms within the same vegetation type, based on Mucina and Rutherford's (2006) national vegetation map. The uninvaded sites were used as a control to which the invaded and cleared sites were compared. The clearing of *A. mearnsii* in the cleared treatments was done five years prior to assessment (cleared in 2006). Control operations involved the complete mechanical removal of *Acacia mearnsii* by cutting and the application of herbicide on cut stumps to prevent resprouting. Follow-up treatments at these sites were applied every year after the initial clearing, and sites

were burnt after clearing to stimulate germination from the soil stored seed bank. Field surveys were undertaken during the period December 2011 to February 2012.

### 4.3.2. Vegetation assessment

At each site, three 100 m parallel transects were laid out 10 m apart (Fig 4.1).
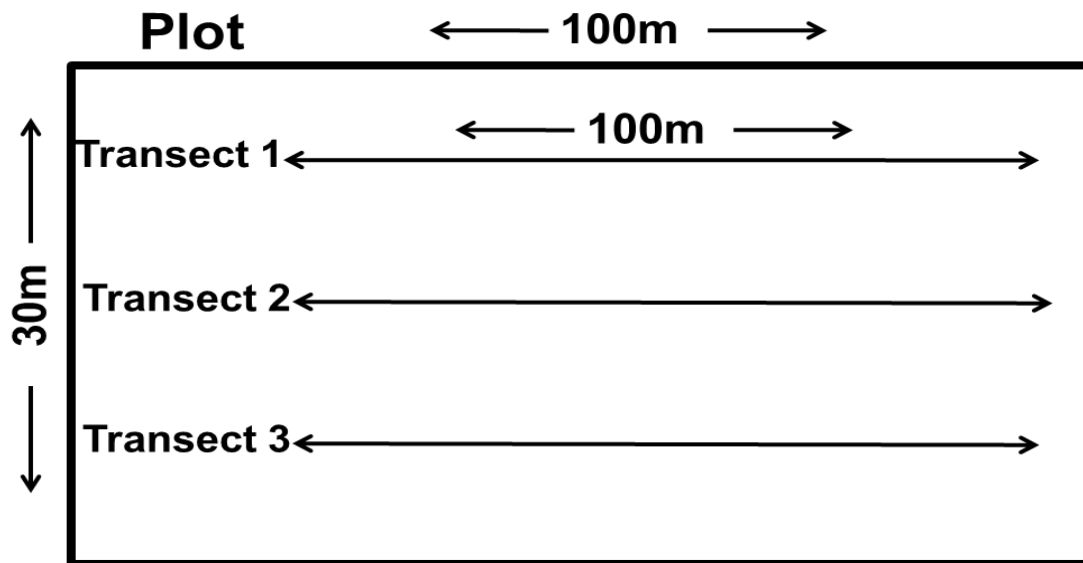


**Figure 4.1:** A diagram showing the sampling plot layout with its associated transect lines used to survey vegetation under various levels of invasion.

### *4.3.2.1. Botanical composition and rangeland condition assessment*

Botanical composition and the basal cover of the herbaceous layer were assessed using the point centred quadrat method described by Trollope (1986). Along each 100 m transect, 50 recordings of species composition were made at 2 m intervals. At each 2 m interval one herbaceous species at or nearest to the point within a 30 cm radius was identified and recorded. If there was no vegetation within the 30 cm radius, the point was recorded as bare ground, so as to determine basal cover.

### 4.3.2.2. Classification of grasses

Grass species were grouped as being Decreaser and Increaser species according to their response to grazing (Trollope, 1986; Van Oudtshoorn, 2006). This grouping system was then used to determine rangeland condition at each site (Van Oudtshoorn, 2006). Decreaser grass species are those that dominate in rangelands that are optimally utilized, but that decrease in relative abundance in rangelands that are over- or underutilised. Increaser species are the indicators of the poor range condition, and are divided into 3 classes: Increaser I, II and III. Increaser I species are generally unpalatable climax grasses that are abundant in underutilised rangeland and under conditions of little or no herbivory. Increaser IIs are grasses that dominate in over utilized rangeland. These include pioneer and sub-climax species that produce much viable seed and can thus quickly establish on new exposed ground. Increaser III species increase in abundance with selective grazing, are generally unpalatable dense climax grasses that are common in overgrazed rangeland. These are strong competitors and increase because the Decreasers are weakened by overgrazing. It is also possible that they are stimulated by light grazing during overgrazing. Herbaceous species that did not fall in the above categories (including alien grasses, and sedges and forbs) were listed as "other".

### 4.3.2.3. Herbaceous biomass assessment

Herbaceous biomass was measured on the same transects (Fig 4.1), using a calibrated disc pasture meter (Trollope, 1983; Trollope and Potgieter, 1985; Bransby and Tainton, 1997). Mean disk heights from each plot were converted to estimate the amount of standing herbage mass using the generalised calibration equation

developed for the area (Trollope, 1983; Trollope and Potgieter, 1985; Bransby and Tainton, 1977). The equation was:

Y = 340 + 388.3x

Where Y = the Herbage biomass or mean standing crop of grass (kg ha$^{-1}$), and x = mean disc heights (Trollope, 1983).

### 4.3.2.4. *Rangeland condition scores and grazing capacity assessment*

Rangeland condition scores were determined by multiplying the percentage cover of each species from each plot with its forage factor, which is an index of the forage production potential for each species (Trollope, 1986). Forage factors are expressed on a scale of 0 (low potential) to 10 (high potential), (Trollope, 1986). Rangeland condition scores for species grouped into a given ecological group were summed to give that ecological group's contribution per plot and the sums of all the ecological groups were added to give the rangeland condition score for each plot. Ecological groups whose rangeland condition scores differed significantly between treatments were examined to establish which species were most affected and therefore responsible for changes in grazing capacities following *A. mearnsii* invasion and clearing.

Current grazing capacities (CGC) for each plot were determined using the estimates of rangeland condition scores (RCS) obtained for each plot, as presented by Danckwerts, (1989) and Hurt, (1989). Rangeland condition scores obtained from the sample plots were each divided by the condition score of the benchmark (CSB). The proportions obtained were multiplied by the potential grazing capacity of the

benchmark (PGC) to drive current grazing capacity for each plot in hectares per large stock unit (ha LSU[1]). The formula was:

CGC = PGC x (1÷ (RCS/CSB)) (Danckwerts, 1989; Hurt, 1989).

### 4.3.3. Soil assessment
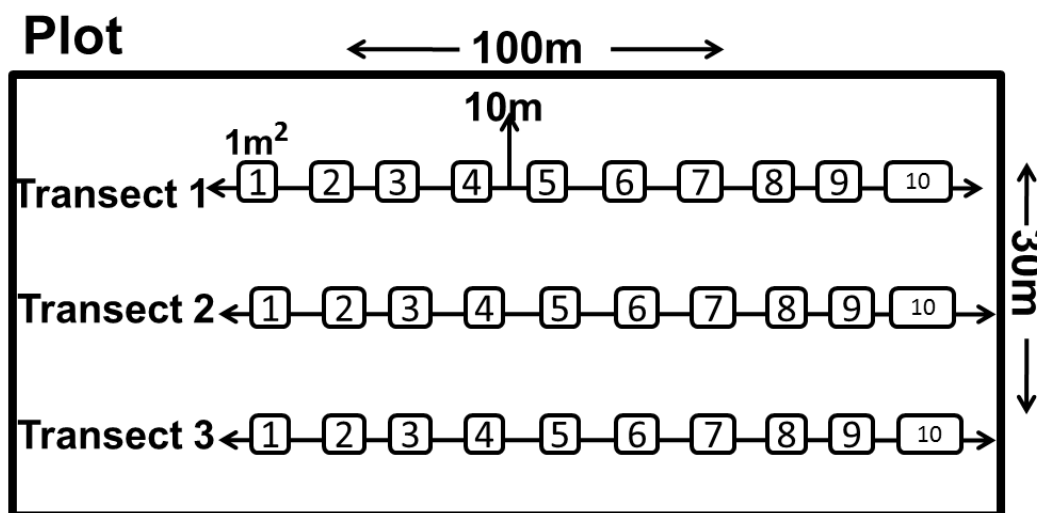


**Figure 4.2:** A diagram showing a sampling plot with associated transect lines and quadrats used to survey soil under selected levels of invasion.

The same plots and 100 m transects that were used for the vegetation assessment were used to survey soils; Ten 1 m$^2$ quadrats were laid out at 10 m intervals (Fig. 4.2). A visual soil surface assessment was conducted in each quadrat using a method derived in part from the Landscape Functional analysis method developed by Tongway and Hindley (2004). This method uses several indicators to assess the biogeochemical function of the landscape. The selected indicators for this study include plant litter, crust brokenness, deposited material and soil surface roughness.

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The amount of dung found in each quadrat was recorded as a proxy for grazing intensity (Thrash *et al.,* 1993; Hodgins and Rogers, 1997; Barnes, 2001).  Soil infiltration rate was measured in each quadrat to assess the soil's ability to partition rainfall into soil water and runoff water. Using a single ring infiltrometer, the amount of time taken for 25 ml of water to infiltrate the soil surface was recorded in seconds.

### 4.3.3.1.  Description of soil surface indicators

Litter

To assess the amount and degree of decomposition of plant litter, the observable amount of plant litter was recorded in the following categories: < 10% (1), 10-25% (2), 25-50 % (3), 50-75% (4), 75-100% (5). When litter cover was  100%, the depth was assessed by compressing it with the flat of hand to remove the air gaps and the values given were: 10 - 20 mm thick (6), 21-70 mm (7), 70-120 mm (8), 120-170 mm (9), > 170 mm (10).

Crust brokenness

The purpose of this measure was to assess to what extent the surface crust is broken, leaving loosely attached soil material available for erosion. Crust brokenness was recorded in each quadrat, by allocating a score based on the extent of observable cracks on the soil crust. The given values were: no crust present (0), crust present but excessively broken (1), crust present but moderately broken (2), crust present but slightly broken (3), curst present but intact, smooth (4).

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Deposited material

To assess the amount of alluvium transported to and deposited on the query zone (i.e. the area being examined), the observable amounts of deposited soil and litter material were assessed based on their percentage cover. The values were given as: more than 50% (1), 20-50% (2), 5-20% (3), 0-5% (4). Averages of means from maximum and minimum values obtained along each transect were used to estimate the amount of deposited material at each site.

Soil surface roughness

Soil surface roughness is related to the ability of the surface to capture mobile resources such as water, propagules, topsoil and organic matter therefore increasing nutrient cycling and infiltration rate. Indicators of surface roughness such as microtopography (depressions) and high grass plant density were used to assess surface roughness. A score for each quadrat was based on the extent of depressions and standing grass tussocks. The given scores were as follows: < 3 mm relief in soil surface (1), shallow depressions 3-8 mm (2), deeper depressions 8-25 mm or dense tussock grasslands (3), deep depressions (4) and very deep depressions or cracks (5).

### 4.3.3.2. Soil collection and analysis

Within each quadrat, a soil sample was taken to a depth of 10 cm. Soils samples for each transect were grouped, providing three bulked samples per site. Samples were

sent to BemLab, Somerset West, for moisture content, chemical and physical analysis.

For physical analysis sand fractions were determined through sieving as described in The Non-affiliated Soil Analyses Work Committee (1990). Silt and clay were then determined using sedimentation rates at 18$^o$C, using a hydrometer. Chemical dispersion was done using sodium hexametaphosphate (calgon). The soil was air dried, sieved through a 2 mm sieve and analysed for pH (1.0 M KCl), P (Bray II) and total extractable cations, namely K, Ca, Mg and Na (extracted at pH = 7 with 0.2 M ammonium acetate) and organic carbon (C) by means of the Walkley-Black method. The extracted solutions were analysed with a Varian ICP-OES optical emission spectrometer (The Non-affiliated Soil Analyses Work Committee, 1990). Total N content of soil was determined through total combustion using a Leco Truspec® CN Nitrogen Analyzer. The soil's CEC was determined using 0.2 M ammonium acetate method as described by The Non-affiliated Soil Analyses Work Committee (1990), where after the soil was leached with 0.2 M $K_2SO_4$. The total $NH_4^+$ was then determined using a Varian ICP-OES optical emission spectrometer as indication of CEC.

## 4.4.   STATISTICAL ANALYSIS

Mean grazing capacities were compared between treatments (uninvaded, cleared, lightly invaded and densely invaded). Impacts of *A. mearnsii* invasion and clearing on soil physical properties (soil texture, resistivity, infiltration rate and moisture content), soil surface properties (litter cover, crust brokenness, deposited material,

surface roughness and dung density), and chemical properties (pH, P, N, C, Na, $K^+$, $Ca^+$, $Mg^+$ and CEC) were evaluated by comparing mean concentrations in different treatments. Significant differences in the above variables between the treatments were taken to represent the impacts of invasion and clearing of *A. mearnsii.* Standard errors were used to express variation around means. The effects of different treatments on means were compared using the one-way analysis of variance (ANOVA) and Kruskal-Wallis multiple comparisons, after normality tests were completed using the Shapiro-Wilk test in Statistica v 10 (Statsoft Inc, 2010). Differences between means were considered significant at $p \leq 0.05$.

## 4.5.  RESULTS

### 4.5.1.  Vegetation assessment

#### 4.5.1.1.  Impacts of A. mearnsii invasion and clearing on herbaceous biomass

The estimated mean herbaceous biomass declined from 4000 to 1224.05 kg ha$^{-1}$ as *A. mearnsii* invasion levels increased (Fig. 4.3). Cleared sites had significantly higher herbaceous biomass (5200 kg ha$^{-1}$) compared to the densely invaded sites ($H=$ 14.60, $p < 0.005$). Although not significantly different from the uninvaded and lightly invaded sites, there was a clear trend showing that an increase in the density of *A. mearnsii* reduced herbaceous biomass. On the other hand, clearing increased herbaceous biomass to levels similar to those of uninvaded sites.
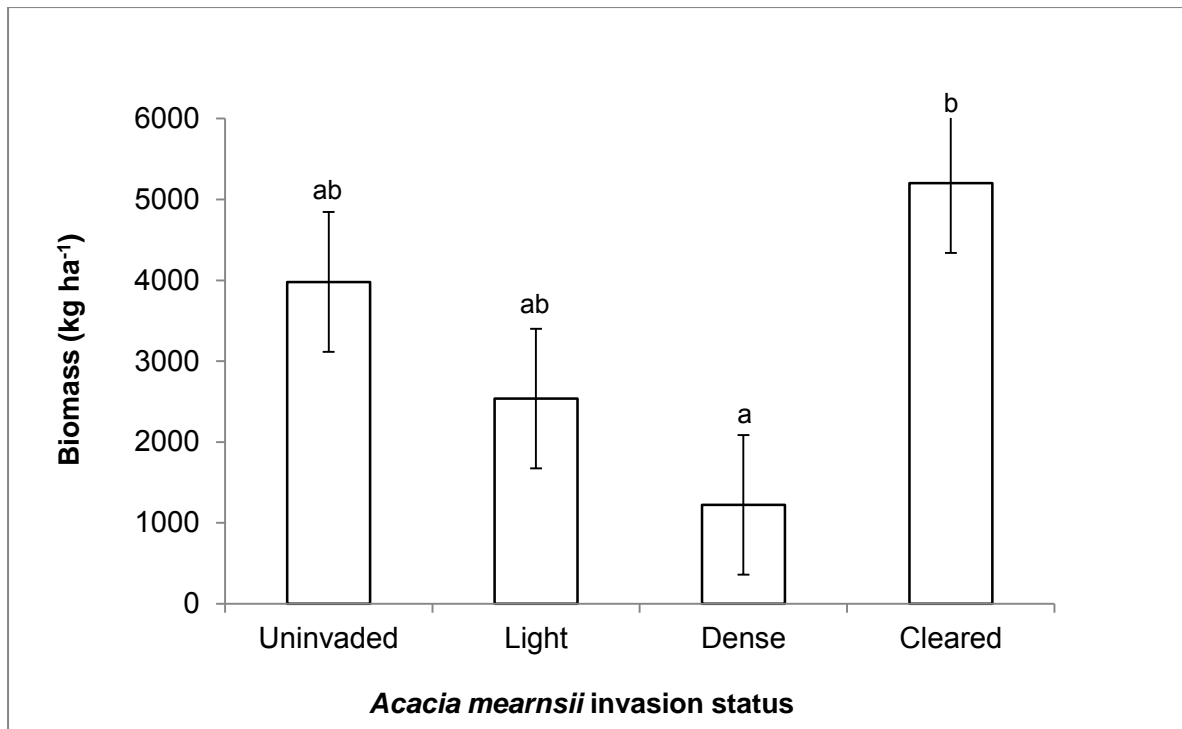
**Figure 4.3**: Mean biomass values ± SE for uninvaded (n=5), lightly invaded, (n=5), densely invaded (n=5) and cleared (n=5) sites in Amathole Montane Grassland of Stutterheim, Eastern Cape. Superscript values denote significant differences at p ≤ 0.05 (one-way ANOVA; H = 14.69, p < 0.05 followed by Kruskal-Wallis multiple comparisons).

### 4.5.1.2. Impacts of A. mearnsii invasion and clearing on botanical composition and basal cover

Invasion by *A. mearnsii* reduced basal cover by 15% and 42% on lightly and densely invaded sites, respectively (Fig. 4.4) when compared to uninvaded sites. There was no significant difference between the mean basal cover on uninvaded and lightly invaded sites. Mean herbaceous cover was improved by clearing of *A. mearnsii* and both cleared and uninvaded sites had higher (H = 18.03, p < 0.001) basal cover than densely invaded sites (Fig. 4.4).

**Figure 4.4:** Mean percent cover ± SE for ecological groups in uninvaded (n=5), lightly invaded, (n=5), dense invaded (n=5) and cleared (n=5) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape.

No significant variation in the proportional species composition under different treatments was found. Increaser II and "other'' groups (Fig. 4.4) were the only two groups whose mean percent cover differed significantly (p ≤ 0.05) between treatments. Mean percent cover of Increaser II species on cleared sites was higher than that on densely-invaded sites (H = 15.59, p < 0.001). Cover of "other" was higher on lightly invaded sites when compared to cleared site (H = 15.59, p < 0.05) (Table 4.1). The uninvaded and cleared sites had no bare areas recorded during the study. Bare area constituted 15% of the lightly invaded site area and nearly 60% of the area of the densely invaded sites. Decreaser species, or species that are preferred by livestock because of their high forage quality, were abundant in the uninvaded and cleared sites.

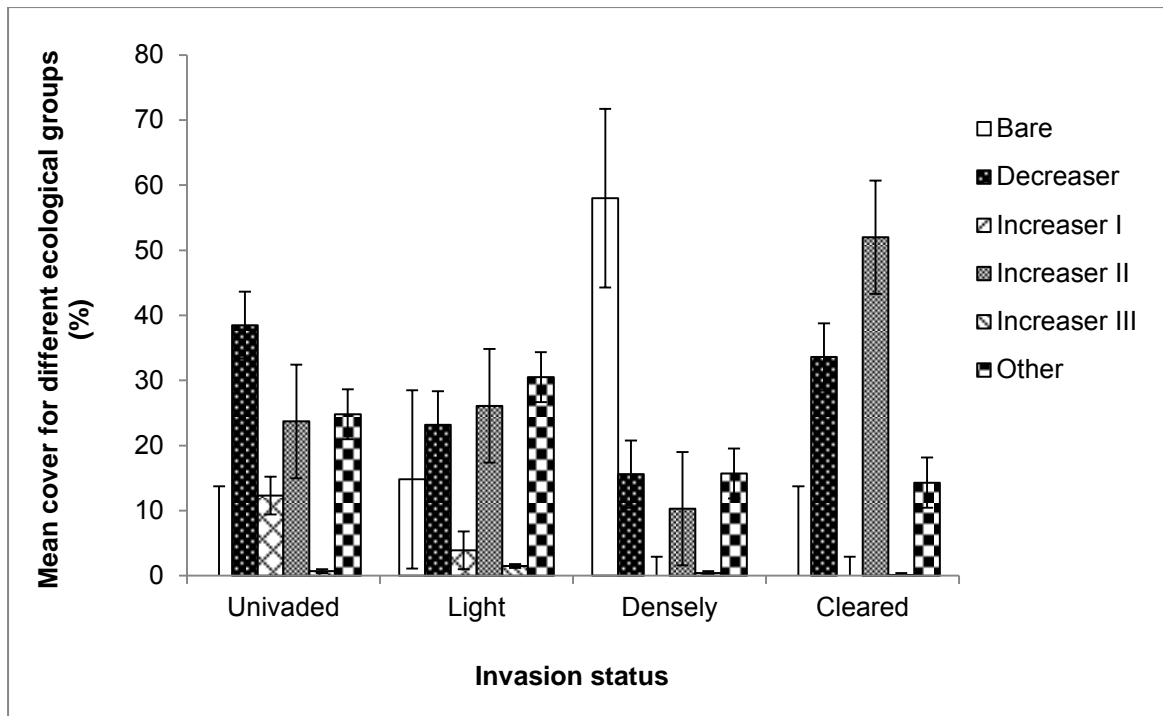**Table 4.1:** Mean percent cover (and ± SE) of grass species in uninvaded (n=5), lightly invaded, (n=5), dense invaded (n=5) and cleared (n=5) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape. Superscript values denote significant differences at p≤ 0.05 (one-way ANOVA; H = 14.69, p < 0.05 followed by Kruskal-Wallis multiple comparisons)

| Species cover | Uninvaded | Lightly Invaded | Densely Invaded | Cleared |
|---|---|---|---|---|
| **Decreaser** | | | | |
| *Cenchrus incertus* | 2.0 ± 1.23[a] | 0.9 ± 0.58[a] | 0.5 ± 0.53[a] | 0.0 ± 0.00[a] |
| *Panicum maximum* | 0.0 ± 0.00[a] | 1.5 ± 1.47[a] | 1.9 ± 1.39[a] | 0.0 ± 0.00[a] |
| *Sporobolus fimbriatus* | 22.5 ± 6.89[a] | 20.4 ± 4.94[a] | 13.2 ± 4.41[a] | 32.0 ± 1.87[a] |
| *Themeda triandra* | 14.0 ± 2.90[a] | 0.4 ± 0.40[b] | 0.0 ± 00[b] | 1.6 ± 1.17[ab] |
| **Increaser I** | | | | |
| *Cymbopogon plurinodis* | 0.1 ± 0.13[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Hyparrhenia hirta* | 0.3 ± 0.27[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Miscanthus capensis* | 0.0 ± 0.00[a] | 3.9 ± 3.87[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Tristachya leucothrix* | 11.9 ± 4.91[a] | 0.0 ± .000[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| **Increaser II** | | | | |
| *Cynodon dactylon* | 0.8 ± 0.65[a] | 6.1 ± 1.18[ab] | 5.1 ± 0.86[ab] | 27.3 ± 5.88[b] |
| *Eragrostis capensis* | 2.7 ± 2.67 [a] | 4.2 ± 3.77 [a] | 0.1 ± 0.13 [a] | 1.1 ± 0.69 [a] |
| *Eragrostis chloromelas* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 1.1 ± 0.54[a] |
| *Eragrostis racemosa* | 2.3 ± 1.95[a] | 8.1 ± 4.34[a] | 0.8 ± 0.49[a] | 0.0 ± 0.00[a] |
| *Eragrostis trichophora* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.1 ± 0.13[a] |
| *Eragrostis viscosa* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.8 ± 0.80[a] |
| *Hyparrhenia hirta* | 0.1 ± 0.27[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Panicum schinzii* | 0.0 ± 18.0[a] | 0.0 ± 7.60[a] | 0.0 ± 4.27[a] | 1.1 ± 19.60[a] |
| *Sporobolus africanus* | 18.0 ± 3.13[a] | 7.6 ± 3.19[a] | 4.3 ± 2.50[a] | 19.6 ± 5.24[a] |
| **Increaser III** | | | | |
| *Merxmuellera stricta* | 0.7 ± 0.67[a] | 1.5 ± 1.31[a] | 0.4 ± 0.27[a] | 0.1 ± 0.13[a] |
| **Other** | | | | |
| *Cyperus rotundus* | 0.0±0.00[a] | 0.9±0.40[ab] | 0.4±0.40[ab] | 0±00[b] |
| *Forb* | 5.5±1.77[a] | 28.3±5.47[b] | 13.7±3.34[ab] | 2.3±1.02[ab] |
| *Kyllinga erecta* | 8.1±2.24[a] | 0.1±0.13[ab] | 1.1±0.91[ab] | 0.0±0.00[b] |
| *Paspalum dilatatum* | 1.1 ± 1.07[ab] | 1.2 ± 0.61[ab] | 0.0 ± 0.00[a] | 12.0 ± 3.26[b] |
| *Paspalum notatum* | 10.1±5.23[a] | 0.0±0.00[b] | 0.5±0.53[ab] | 0.0±0.00[b] |

### 4.5.1.3. Impacts of A.mearnsii *invasion and clearing on rangeland condition and grazing capacity*

Significant differences between ecological groups with respect to different treatments and species mean rangeland condition scores and treatments were taken as an indication of changes in grazing capacities following *A. mearnsii* invasion and clearing.

Invasion by *A. mearnsii* reduced grazing capacity by 56% and 72% on light and densely invaded sites respectively, whereas grazing capacity improved by 66% five years after clearing (Fig. 4.5). The mean grazing capacity on the uninvaded site (2.33 ± 0.16 ha LSU$^{-1}$) was 56 % higher (H = 14.84, p < 0.05) than that on lightly invaded sites (5.27 ± 0.88 ha LSU$^{-1}$) and 72 % higher (H = 14.84, p < 0.05) than that on densely invaded sites (8.64 ± 0.10 ha LSU$^{-1}$).
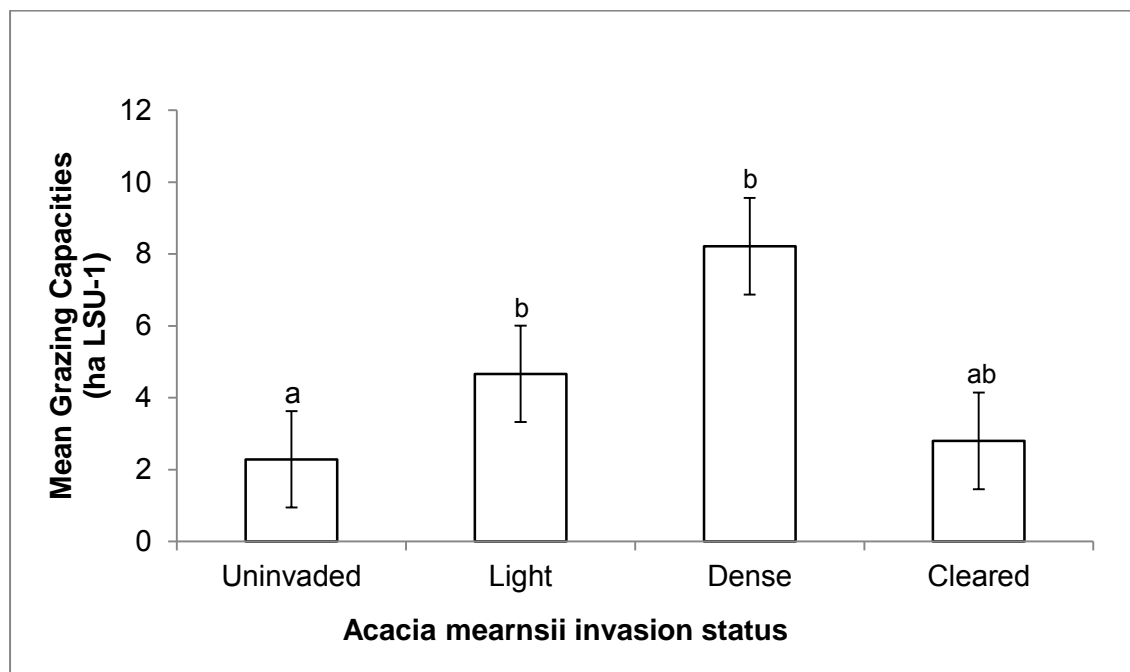


**Figure 4.5:** Mean grazing capacities ± SE for uninvaded (n=5), lightly invaded, (n=5), densely invaded (n=5) and cleared (n=5) sites in the Amathole Montane Grassland. Superscript values denote significant differences at p ≤ 0.05 (one way ANOVA; H = 14.84, p < 0.05, followed by Kruskal-Wallis multiple comparisons).

### 4.5.1.4. Ecological groups and species condition scores underlying changes in grazing capacity

Decreasers (H = 8.49, p < 0.05) and Increaser I (H = 15.12, p < 0.05) were the only ecological groups that showed significant differences in rangeland condition scores for the treatments uninvaded, densely invaded sites and cleared sites (Fig. 4.6). Of the four Decreaser grass species present in the study sites (Table 4.2), *T. triandra* made a significant contribution to the higher rangeland condition score in uninvaded when compared to lightly (H = 14.55, p < 0.05) and densely (H = 14.55, p < 0.05) invaded sites. The contribution of other ecological groups (Increaser II, III and ''other'') to the overall rangeland condition scores in the uninvaded, lightly and densely invaded sites were not significantly different (Table 4.2). However, higher proportions of Decreaser and Increaser II groups (Fig. 4.6) accounted for the improved grazing capacity on the cleared sites. Of the Decreaser group, *S. fimbriatus* had the highest contribution to the rangeland condition score. Increaser II was the only ecological group that contributed significantly (H = 12.63, p < 0.01) to the higher mean grazing capacity of the cleared sites, with *C. dactylon* showing significant contributions (H = 13.67, p < 0.01).

**Figure 4.6:** Mean range condition scores ± SE for different ecological groups in uninvaded (n=5), lightly invaded (n=5), densely invaded (n=5) and cleared (n=5) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape.
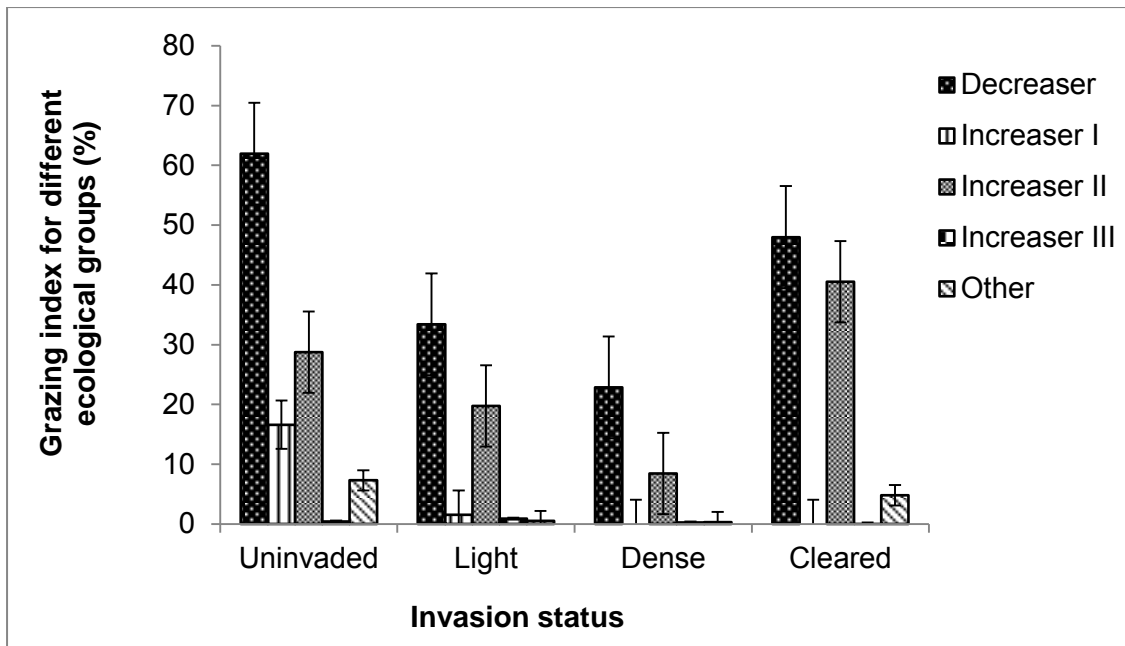
**Table 4.2:** Mean range condition scores ± SE for different species in uninvaded (n=5), lightly invaded, (n=5), densely invaded (n=5) and cleared (n=5) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape. Superscript values denote significant differences at p ≤ 0.05 (one way ANOVA, followed by Kruskal-Wallis multiple comparison).

| Species | Uninvaded | Lightly Invaded | Densely Invaded | Cleared |
|---|---|---|---|---|
| **Decreasers** | | | | |
| *Cenchrus incertus* | 2.4 ± 1.48[a] | 1.1 ± 0.7[a] | 0.6 ± 0.64[a] | 0.0 ± 0.00[a] |
| *Panicum maximum* | 0.0 ± 0.00[a] | 2.9 ± 2.93[a] | 3.7 ± 2.78[a] | 0.0 ± 0.00[a] |
| *Sporobolus fimbriatus* | 31.6 ± 9.65[a] | 28.6 ± 6.91[a] | 18.5 ± 6.17[a] | 44.8 ± 2.62[a] |
| *Themeda triandra* | 28.0 ± 5.80[a] | 0.8 ± 0.80[b] | 0.0 ± 0.00[b] | 3.2 ± 2.33[ab] |
| **Increaser I** | | | | |
| *Cymbopogon plurinodis* | 0.1 ± 0.11[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Hyparrhenia hirta* | 0.2 ± 0.21[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Miscanthus capensis* | 0.0 ± 0.00[a] | 1.6 ± 1.55[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Tristachya leucothrix* | 16.6 ± 6.87[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| **Increaser II** | | | | |
| *Cynodon dactylon* | 0.3 ± 0.26[a] | 2.5 ± 0.47[ab] | 2.0 ± 0.34[ab] | 10.9 ± 2.35[b] |
| *Eragrostis capensis* | 2.3 ± 2.13[a] | 3.4 ± 3.02[a] | 0.1 ± 0.11[a] | 0.9 ± 0.55[a] |
| *Eragrostis chloromelas* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.9 ± 0.43[a] |
| *Eragrostis racemosa* | 0.9 ± 0.78[a] | 3.3 ± 1.74[a] | 0.3 ± 0.20[a] | 0.0 ± 0.00[a] |
| *Eragrostis trichophora* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.1 ± 0.11[a] |
| *Eragrostis viscosa* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.3 ± 0.32[a] |
| *Panicum schinzii* | 0.0 ± 0.00[a.] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Sporobolus africanus* | 25.2±4.38[a] | 10.6±4.47[a] | 6.0±3.50[a] | 27.4±7.33[a] |
| **Increaser III** | | | | |
| *Merxmuellera stricta* | 0.4 ± 0.40[a] | 0.9 ± 0.78[a] | 0.2 ± 0.16[a] | 0.1 ± 0.08[a] |
| **Other** | | | | |
| *Cyperus rotundus* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Forb* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Kyllinga erecta* | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] | 0.0 ± 0.00[a] |
| *Paspalum dilatatum* | 0.4 ± 0.43[ab] | 0.5 ± 0.24[ab] | 0.0 ± 0.00[b] | 4.8 ± 1.30[a] |
| *Paspalum notatum* | 6.9 ± 3.14[a] | 0.0 ± 0.00[b] | 0.3 ± 0.32[ab] | 0.0 ± 0.00[b] |

### 4.5.2.  Soil assessment

#### 4.5.2.1.  *Soil physical properties and moisture content*

**Table 4.3**: Mean percent content ± SE for different soil types in uninvaded (n=5), lightly invaded (n=5), dense invaded (n=5) and cleared (n=5) sites in the Amathole Montane Grassland of the Stutterheim, Eastern Cape. Superscript values denote significant differences at p≤ 0.05 (one way ANOVA, followed by Kruskal-Wallis multiple comparison).

| Variable | Uninvaded | Lightly Invaded | Densely Invaded | Cleared |
|---|---|---|---|---|
| Clay | 8.4 ± 0.55[a] | 8.8 ± 0.54[a] | 8.6 ± 0.61[a] | 7.9 ± 0.39[a] |
| Silt | 11.3 ± 0.82[a] | 10.8 ± 0.90[a] | 10.1 ± 0.93[a] | 8.9 ± 0.57[a] |
| Fine Sand | 74.9 ± 0.75[a] | 75.9 ± 1.21[a] | 76.1 ± 1.25[a] | 78.0 ± 1.00[a] |
| Medium Sand | 2.7 ± 0.64[a] | 1.9 ± 0.37[a] | 2.4 ± 0.64[a] | 4.0 ± 0.97[a] |
| Coarse Sand | 2.7 ± 0.40[a] | 2.7 ± 0.25[a] | 2.8 ± 0.48[a] | 1.2 ± 0.23[b] |

No significant differences in soil physical properties were found between treatments (Table 4.3). However, the exception was noted for the lower (H = 15.83, p < 0.001) coarse sand content on the cleared sites.

 Soil water infiltration rate was significantly faster on lightly (H = 346, p < 0.001) and densely (H = 346, p < 0.001) invaded sites when compared to both uninvaded and cleared sites (Fig. 4.7). There was no significant difference between uninvaded and cleared sites (Fig. 4.7). There was no significant difference on soil moisture content between uninvaded and both light and dense invaded sites (Fig. 4.8). However, percentage soil moisture was significantly higher (H = 16.35, p < 0.05) on lightly invaded sites than dense invaded sights. Soil moisture content was higher (H = 16.35, p < 0.001) on the cleared sites than on dense invaded sites (Fig. 4.8).
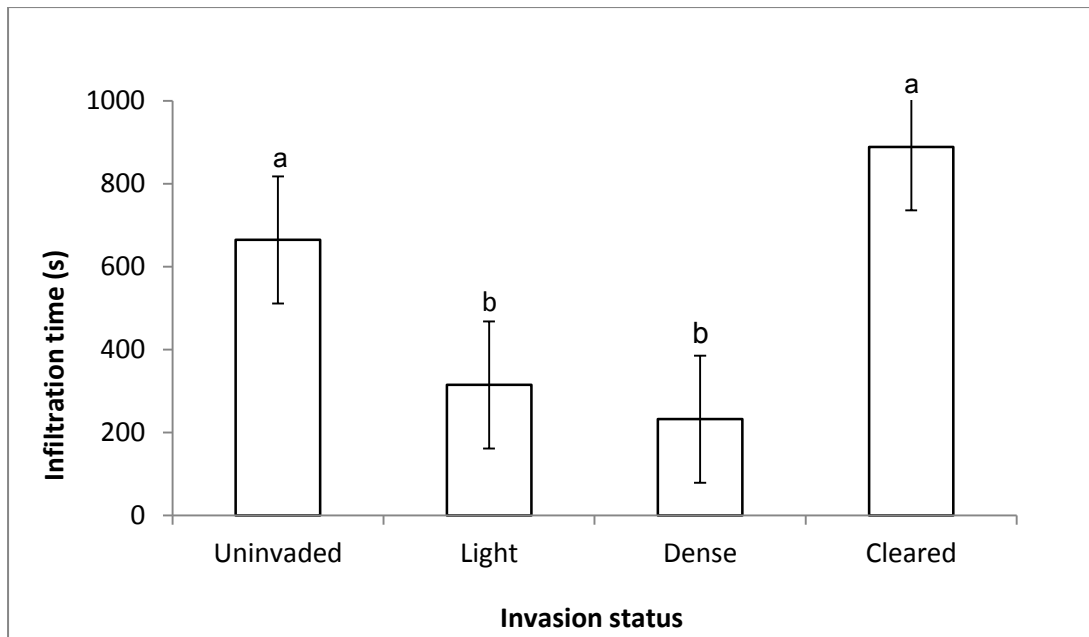
**Figure 4.7:** Mean infiltration time ± SE in seconds, for uninvaded (n=5), lightly invaded (n=5), densely invaded (n=5) and cleared (n=5) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape. The error bars are times on SE. Superscript values denote significant differences at p≤ 0.05 (one-way ANOVA; H = 346, p < 0.05 followed by Kruskal-Wallis multiple comparisons).



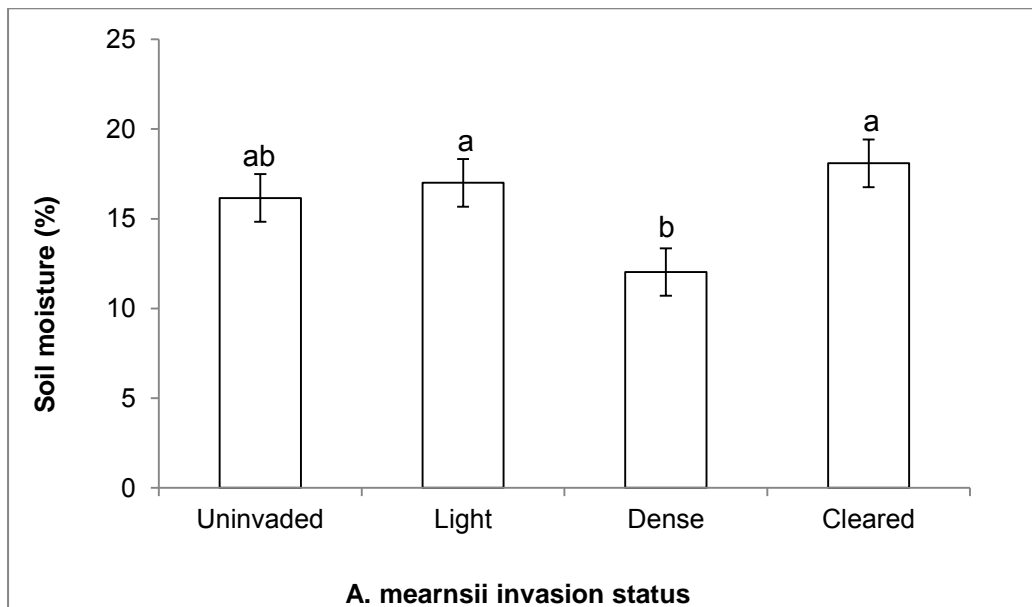**Figure 4.8:** Average percent moisture content ± SE for difference sites: for uninvaded (n=5), lightly invaded, (n=5), densely invaded (n=5) and cleared (n=5) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape. Superscript values denote significant differences at p≤ 0.05 (one way ANOVA; H = 16.36, p < 0.05 followed by Kruskal-Wallis multiple comparisons).

### 4.5.2.2. Impacts of A. mearnsii invasion and clearing on soil surface properties

Plant litter cover increased with increasing density of *A. mearnsii* (Table 4.4). Litter cover was higher on both lightly (H = 328.28, p < 0.001) and densely invaded (H = 328.28, p < 0.001) sites when compared to uninvaded and cleared sites. There was no evidence of crust brokenness in uninvaded and cleared sites, whereas, both lightly (H = 368.63, p < 0.001) and densely invaded (H = 368, p < 0.001) sites showed significant evidence of crust brokenness (Table 4.4). There was no evidence of deposited material in uninvaded and cleared sites. Between lightly and densely invaded sites, deposited material was more (H = 342.37, p < 0.001) on densely invaded sites (the lower values indicate higher amount of deposited material and vice versa). Soil surface roughness was lower (H = 144.71, p < 0.001) on dense invaded sites when compared to all other treatments. Dung density was lowest (H = 45.75, p < 0.001) in densely invaded sites when compared to uninvaded, lightly invaded and cleared sites (Table 4.4).

**Table 4.4:** Means ± SE for layer properties in uninvaded (n=5), lightly invaded (n=5), dense invaded (n=5) sites and cleared (n=5) in the Amathole Montane Grassland of Stutterheim, Eastern Cape. Different letters in a row denote significant values at p ≤ 0.05 (one way ANOVA, followed by Kruskal-Wallis multiple comparison).

| Treatment | Uninvaded | Lightly Invaded | Densely Invaded | Cleared |
|---|---|---|---|---|
| Litter | 1.3 ± 0.04[a] | 2.2 ± 0.13[b] | 4.7 ± 0.10[c] | 1.4 ± 0.09[a] |
| Crust brokenness | 0.0 ± 0.00[a] | 2.9 ± 0.13[b] | 2.9 ± 0.12[b] | 0.0 ± 0.00[a] |
| Deposited material | 4.0 ± 0.00[a] | 3.2 ± 0.07[b] | 2.8 ± 0.06[c] | 4.0 ± 0.00[a] |
| Soil surface roughness | 3.0 ± 0.00[a] | 3.2 ± 0.12[a] | 1.9 ± 0.09[b] | 3.0 ± 0.01[a] |
| Dung density | 0.8 ± 0.08[a] | 0.9 ± 0.10[a] | 0.3 ± 0.06[b] | 1.1 ± 0.10[a] |

### 4.5.2.3. Impacts of A. mearnsii invasion and clearing on soil chemical properties

Sites that were densely invaded by *A. mearnsii* were slightly more acidic ($p \leq 0.05$) when compared to the uninvaded ($H = 15.56$, $p < 0.05$) and cleared sites ($H = 15.56$, $p < = 0.05$) (Table 4.5). There was a significant ($p < 0.05$) increase on soil pH after 5 years since removal of *A. mearnsii*. Densely invaded sites had higher concentrations of P ($H = 26.56$, $p < 0.001$), C ($H = 10.83$, $p < 0.01$) and N ($H = 17.23$, $p < 0.01$) when compared to uninvaded sites (Table 4.5), whereas, Na and $Mg^+$ ($H = 14.55$, $p < 0.01$) showed a relative decrease in densely invaded sites. Lightly invaded sites were intermediate between uninvaded sites, i.e. there were no significant variation between either uninvaded or densely invaded and lightly invaded sites (the exception was for N and Na which decreases significantly during light invasion). Potassium and exchangeable cations (CEC) concentration showed no significant response to *A. mearnsii* invasion and clearing (Table 4.5).

Cleared sites had higher P and K concentrations ($p < 0.05$) when compared to uninvaded and dense invaded sites, whereas Na and $Mg^+$ had decreased significantly ($p < 0.05$) in 5 years since the removal of *A. mearnsii*. Clearing of *A. mearnsii* resulted in a slight decrease of C, N, $Ca^+$ and CEC in 5 years; therefore, cleared sites were intermediate between dense and uninvaded sites (Table 4.5).

**Table 4.5:** Mean percent content ± SE for soil chemical properties in uninvaded (n=5), cleared (n=5), lightly invaded (n=5) and dense invaded (n=5) sites in the Amathola Montane Grassland of the Eastern Cape. Different superscripts in a row denote significant values at p ≤ 0.05 (one way ANOVA, followed by Kruskal-Wallis multiple comparison).

| Chemical properties | Uninvaded | Lightly Invaded | Densely Invaded | Cleared |
|---|---|---|---|---|
| pH(KCl) | 4.4 ± 0.04[a] | 4.3 ± 0.05[ab] | 4.2 ± 0.04[b] | 4.4 ± 0.06[a] |
| P (mg/kg) | 2.0 ±0.20[a] | 5.2 ± 1.78[ab] | 4.2 ± 0.59[b] | 7.3 ± 1.57[b] |
| C% | 2.0 ± 0.10[a] | 3.0 ± 0.41[ab] | 4.0 ± 0.51[b] | 3.2 ± 0.46[ab] |
| N% | 0.1 ± 0.01[a] | 0.2 ± 0.02[b] | 0.2 ± 0.01[b] | 0.2 ± 0.02[ab] |
| Na% | 5.1 ± 0.18[a] | 4.0 ± 0.18[b] | 3.4 ± 0.14[b] | 3.7 ± 0.25[b] |
| K% | 4.1 ± 0.22[a] | 3.9 ± 0.20[a] | 4.2 ± 0.12[a] | 5.7 ± 0.37[b] |
| Ca% | 33.4 ± 1.53[a] | 35.3 ± 1.31[a] | 36.3 ± 0.82[a] | 35.9 ± 1.25[a] |
| Mg% | 22.8 ± 0.63[a] | 21.0 ± 0.68[ab] | 19.6 ± 0.66[bc] | 17.0 ± 0.94[c] |
| CEC (cmol (+)/kg) | 8.0 ± 0.79[a] | 10.7 ± 0.79[a] | 10.8 ± 0.53[a] | 8.6 ± 0.58[a] |

## 4.6. DISCUSSION

### 4.6.1. Consequences of *A. mearnsii* invasion on rangeland condition and grazing capacity

The results of this study demonstrate an alarming decline in grazing capacity as a result of *A. mearnsii* invasion. The reduction in grazing capacity, from 2 to 8 ha required to support one large livestock unit on uninvaded and densely invaded sites respectively has important socio-economic and human wellbeing implications. Similar results were reported by Ng'weno *et al* (2009) who found a significant decline of forage quantity on the *Acacia* invaded sites of Lake Nakuru National Park, Kenya. Despite the insignificant differences on percentage species cover between uninvaded and invaded treatments in my study, the displacement of grasses was coupled by loss of palatable species including Decreaser and Increaser II species,

and these changes were most obvious when comparing uninvaded and densely invaded sites. Therefore, the loss in grazing capacity in invaded sites is attributed to displacement of high gazing value Decreaser species including *Themeda triandra* and *Sporobolus fimbriatus* which were more prominent in uninvaded sites. The reduction in herbaceous and grass cover resulting from invasion by *Acacia* has also been reported by several studies (Holmes *et al.,* 2005; Blanchard and Holmes, 2008; Morris *et al.,* 2008; Ruwanza *et al.,* 2012). This could be due to the highly competitive nature of *Acacia* species for growth resources such as water, soil nutrients and light (Dye and Jarmain, 2004, Yelenik *et al.,* 2007, Morris *et al.,* 2011).

Previous studies have shown that *Acacia* species alter soil properties such as moisture content and nutrient status (Musil, 1993; Werner *et al.,* 2010; Gaertner *et al.,* 2011). In this study, soil moisture was reduced under densely invaded sites. Because of the reported comparatively excessive water use by *Acacia* species (Dye and Jarmain, 2004; Gaertner *et al.,* 2011), I anticipated that all invaded sites would have significantly less soil moisture. However, local conditions and factors play an influencing role and other studies such as that by Yelenik (2004) confirm this, finding higher soil moisture content under dense *Acacia* stands compared with their fynbos reference sites. Soil moisture at my sites may also have been affected by rainfall events that occurred during the survey period.

High water infiltration rates noted on the invaded sites are attributed to degraded soil surface conditions and poor herbaceous cover which translates into broken root networks. This could imply that while water infiltration is faster here, soil may also dry out faster. Whereas in uninvaded sites, good cover of standing biomass slows water runoff and allows time for water to infiltrate into the soil. Similar to the results of this

study, Mills and Fey (2004) found high crust brokenness under invaded sites and associated this with low vegetation cover. Vegetation protects the soil surface from damaging effects caused by animal trampling, for example. Increased litter content found in *A. mearnsii* invaded sites, when compared with uninvaded sites, has been reported by previous studies (Witkowski, 1991; Yelenik *et al.,* 2006; Gaertner *et al.,* 2011). Evidence of greater volumes of deposited material in *Acacia* invaded sites could also be associated with low basal level vegetation cover. Water runs freely on bare soil and carries plant debris and other light material to the lower slopes or until it is trapped by branches of other material that may have fallen from trees. High soil surface roughness on uninvaded sites was explained by presence of standing grass tussocks.

The lower dung content found under densely invaded sites is an indication of low grazing animal activity (Hodgins and Rogers, 1997; Barnes, 2001). Low grass cover at these sites further supports the notion that these environments are undesirable for grazing animals. The presence of some palatable grass species in the lightly invaded sites and higher dung content of these sites could suggest that these sites are being utilised to a degree by animals. *Acacia* trees at this density do not create a barrier to animal movement and may instead possibly provide shade and for shelter during extreme weather events.

Soil biological indicators and properties are linked to soil biological processes that are responsible for plant growth. For instance, soil organic matter, which to some extent relates to decomposing plant litter, is essential to ensure soil aggregate stability which facilitates a higher rate of infiltration. Water infiltration rate determines soil moisture content which is essential for nutrient cycling, which in turn is essential

for plant growth (Mills and Fey, 2003). Furthermore, organic matter and soil moisture are essential for the functioning of soil microbes which are responsible for nutrient cycling (O'Farrell *et al.,* 2010).

The consequences of *A. mearnsii* invasion on soil nutrient status were more evident in the densely invaded sites than lightly invaded sites. Musil and Midgley (1990) reported high concentrations of soil pH under *Acacia saligna* stand, whereas Montgomery (2001) reported otherwise. Findings of this study concur with those of Montgomery (2001), where densely invaded sites were more acidic than uninvaded soils. In agreement with previous studies (Witkowski, 1991; Yelenik *et al.,* 2004), *A. mearnsii* invaded sites had higher concentrations of P, C, and N. Yelenik *et al.,* (2007) found no differences in $K^+$ and Na concentrations under *A. saligna* canopy but reported higher levels of $Mg^+$ and $Ca^+$ when compared to the fynbos species dominated soils. Results of this study showed low levels of $Mg^+$ and Na concentrations under *A. mearnsii* canopy, whereas there were no differences in $Ca^+$ and $K^+$ concentrations under all invaded treatments. Witkowski (1991) and Yelenik *et al* (2007) associated higher levels of nutrients with the redistribution of soil nutrients by *Acacia* trees from deeper layers of the soil. *Acacia* species have been associated with high nutrient intake when compared to native vegetation, and this might be the cause of low concentrations of some nutrient levels under *A. mearnsii* canopy in this study. Increased soil N concentrations under Acacias canopy has been reported by several studies (Musil and Migdely, 1990; Yelenik *et al*, 2004; Yelenik *et al.,* 2007) and have been associated with high levels of nitrogen-rich plant litter. *Acacia mearnsii* is a leguminous tree with higher $N_2$ fixing capability (Yelenik *et al*, 2004) and hence the capacity to increase soil N concentrations. High concentrations of soil N in *Acacia* invaded sites, in this study, could be associated with higher levels of N-

rich litter from *Acacia* leaves which will result in more N being returned from the plant material to the soil (Witkowski, 1991; Stock, 1995; Yelenik *et al.,* 2004). The same reason was assumed to be the cause of increased N concentration on *A. mearnsii* invaded sites, in this study. The findings from other studies, however, conflict with these, as Marchante *et al* (2008) and Gaertner *et al.,* (2011) found insignificant differences of available N in *Acacia* invaded sites compared to their native reference sites and argued that high nutrient intake by *Acacia* trees could reduce quantities of N in the soil.

## 4.6.2. Changes in rangeland condition and grazing capacity following the clearing of *A. mearnsii*

The results showed a considerable increase in rangeland condition and gains in grazing capacity five years after the clearing of *A. mearnsii*. These improvements could be attributed to growth and general increased cover of the herbaceous and grass layers since the removal of *A. mearnsii.* The results revealed an increased composition of Decreaser and Increaser II species on cleared sites which probably, due to their palatability and resulting high grazing index values (Trollope, 1986; Van Oudtshoorn2006), contributed positively to the grazing potential of the rangeland. The improvement of vegetation cover after *Acacia* clearing has been reported by several studies (Blanchard and Holmes, 2008; Morris *et al.,* 2008; Ruwanza, 2011). The improvements in vegetation cover in cleared sites could also be attributed to the legacy effects of *Acacia's* invasion on soil nutrients status. It might be suggested that clearing of *A. mearnsii* would have released the pressure on soil nutrients and enhanced availability of these growth resources to native grasses (Yelenik *et al.,*

2004). Morris, *et al.,* (2008), Ndhlovu *et al.,* (2011) and Ruwanza *et al.,* (2012) reported an increase in species richness, cover and abundance of native vegetation after complete removal of alien invasive species. They found that careful monitoring and proper follow-up treatments after clearing contributed significantly to the recovery of indigenous vegetation.

High rainfall during the field survey may have masked any longer-term effects of invasion on soil moisture. Thus, while the results showed that soils in cleared sites could hold soil moisture much longer than densely invaded sites, such findings may not be typical of drier spells. This was also shown by a slower water infiltration rate on cleared sites. High moisture content and therefore slow infiltration rates could be attributed to good herbaceous cover on cleared sites. Clearing of trees reduces water loss from the soil and therefore improves soil water availability to the grass species. This could explain why cleared sites and uninvaded sites had similar levels of vegetation cover.

Plant litter was significantly lower on cleared sites than on invaded sites, and this could be explained by the absence of litter-producing *Acacia* species (Milton, 1981; Witkowski, 1991; Yelenik, 2004). Similar to uninvaded sites, low proportions of crust brokenness, and the absence of deposited material in cleared sites could be attribute to build up of plant material and soil following removal of *A. mearnsii* (Mills and Fey, 2004).

The legacy of *A. mearnsii* had positive impacts on soil fertility of cleared sites. Here concentrations of soil nutrients on cleared sites were similar to that of dense invaded sites. Similar results have been reported by previous studies (Yelenik *et al.,* 2004, Yelenik *et al.,* 2007, Gaertner *et al.,* 2011). Soil pH on cleared sites was higher than

on densely invaded sites and similar to that of uninvaded sites (Montgomery, 2001). The same effect was noted for P and K concentrations. Soil organic carbon and nitrogen content after clearing of *A. mearnsii* 5 years ago remained the same as before clearing. However, Corbin and D' Antonio (2004) suggested that *Acacia spp.* has an extended legacy after clearing as a result of lag effect that causes slow release of available N from the higher N pool in the soil.

## 4.7.   CONCLUSIONS

This study clearly demonstrates that significant grazing resources are lost under conditions of *A. mearnsii* invasion. This study has also shown that clearing of *A. mearnsii* can actually help to restore the decline in grazing capacity within 5 years, but this is conditional on the autogenic recover of the grass layer. Increased soil nitrogen levels, which can remain for an extended period after clearing of *A. mearnsii,* would have had significant impact on improving grazing condition. Even though in most cases these changes were not picked up by the analysis, impacts of *A. mearnsii* can be evident even under light invasion. In this study, there were slight differences in nutrient concentrations between uninvaded and lightly invaded sites, but these were escalated under dense invasion.

The successful recovery following alien removal in the study area is largely due to well-timed-follow up treatments. Seed bank sizes, disturbances during clearing, and the potential abundance of nutrients, increase the potential reinvasion of the area by the same species. Therefore careful monitoring of sites associated rapid response is essential for ecosystem recovery. In particular such actions need to be coupled

closely with land restoration actions. The cost involved in clearing alien invasive plants is completely restrictive and it is highly unrealistic to assume that poorly resourced emerging farmers will be able to undertake such interventions. Government interventions that assist emerging farmers in this process of clearing invasive alien plants will both enhance the speed of recovery and the chances of creating sustainable rural economies.

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# CHAPTER 5: CONCLUSIONS AND FUTURE RESEARCH RECOMENDATIONS

## 5.1.  INTRODUCTION

Invasive wattle species (*A. mearnsii, A. dealbata, A. decurrens*) dominate large areas of South Africa which are used for livestock production. The degree to which these species affect livestock production has received little attention, both nationally and globally. This study aimed to address this research gap. My objectives were: to explore the degree of overlap between invasion by wattles species and important grazing areas at a national scale; to develop an understanding of temporal aspects associated with invasion and the associated loss or decline in carrying capacity; to determine the effects of *A. mearnsii* invasion on growth form dominance of indigenous vegetation species, and how this translates into impacts in forage quality and quantity; to determine the effects of *A. mearnsii* invasion on soil resources and conditions  required to support grazing production; and to determine the effects that clearing operations have had on the grazing resources. I discuss each of these in turn below and reflect on the implications of these changes as well at future research needs that are highlighted by this study.

## 5.2.  KEY FINDINGS AND CONCLUSIONS

### 5.2.1.  National scale grazing priority areas and Acacia invasions

My analysis indicated that the magisterial districts with the highest grazing potential are concentrated in the eastern and northern half of the country, falling into the grassland and savanna biomes (see Chapter 3). The results suggest that there is a

certain degree of overlap between these areas of high grazing potential and those invaded by wattle species. Whilst this overlap appears small when viewed as a proportion of the country's total remaining natural areas, the impacts can be substantial at the magisterial district level (see Chapter 3). These fast growing trees dominate interactions and competition with indigenous species for growth resources, displacing palatable grazing species and reducing grass biomass and grazing capacity (see Chapter 4; Morris *et al.,* 2011). Furthermore, these species are known to spread extensively over a short period of time (DWAF, 1997; Henderson, 2001), suggesting that additional critical grazing areas will soon be lost, and those already invaded will become further degraded as densities of these invasive alien species increase. Their spread and impact is likely to affect currently degraded areas more severely, including those communal areas concentrated in the eastern and northern provinces of South Africa, exacerbating degradation there.

Grassland and savanna biomes are a cornerstone of natural grazing resources in South Africa and contribute extensively to the country's agricultural economy. A substantial portion of these areas are communal areas that are home to the majority of low income people who live in rural areas. Rangelands provide for essential components of human well-being, especially for people who depend on certain ecosystem services (fuel wood, food and water) for their livelihoods (Cousins, 1999; Shackleton *et al.,* 2001). Invasion is likely to ultimately negatively impact on livelihoods.

### 5.2.2.  Trends in time

While at the national scale invasion might appear insignificant, closer inspection at a districts level shows areas of severe invasion. This is the scale at which the impacts and losses of ecosystem services would be felt, as well as being the level at which land-use management decisions are implemented, thus making it an important level to focus on. Given what we know of invasion rates from the literature (Le Maitre *et al.,* 2004; van Wilgen *et al.,* 2008), these local-levels of invasion present a critical problem. Carrying out temporal-scale studies is important in showing shifts over time. I conclude that invasive alien plants have the potential to severely threaten highly productive land if invasions continue unabated. The more rapid the invasion the more quickly South Africa's livestock production will be under threat.

### 5.2.3.  The effects of invasive acacia density on grazing provision

My exploration of ecological responses to alien plant invasion shows the specific nature of the threat of acacia invasion. Invasion by *A .mearnsii* is characterised by a shift in species composition from high grazing value species in uninvaded sites to low grazing value species dominated by forb species (see Chapter 4). The loss of grazing capacity under densely invaded sites is to a large extent explained by displacement or suppression of the grass layer (see Chapter 4). Invasive wattle species are highly competitive. Wattle species have high biomass and so can reduce light penetrability which changes the microclimate below the canopy (Holmes and Cowling, 1997; Morris *et al.,* 2011). These combined effects result in dense invasions reducing grazing capacities by more than 200% at the local scale.

### 5.2.4.  The effects of invasive acacia density on biophysical properties

This study shows clear biophysical environmental changes with invasion by *A. mearnsii*. Invasion by *A. mearnsii* was found to increase the concentration of phosphorus, nitrogen and carbon as well as soil acidity (see chapter 4). The results show that changes in soil nutrients were coupled with changes in soil layer properties during invasion by *A. mearnsii*. Invasion by *A. mearnsii* was found to increase plant litter content (see Chapter 4). This explains the high water infiltration rate that I found on the invaded sites. Furthermore, I also found that invasion by *A. mearnsii* has to some extent reduced grazing animal activity on densely invaded sites due to very low forage availability.

### 5.2.5.  Changes resulting for clearing acacia

My study demonstrated that clearing *A. mearnsii* enhanced forage quality and quantity and that grazing capacities could be re-established in the relatively short term. Whilst no substantial changes in the composition of ecological groups was identified as a result of invasion by *A. mearnsii*, I found that clearing increased the percentage cover of increaser II grasses, and more generally herbaceous biomass and basal cover (see Chapter 4). I also found that on cleared sites the legacy of *A. mearnsii* had positive impacts on soil fertility as soil nutrient concentrations were higher than on uninvaded sites. I conclude that the well-timed follow-up treatments contributed to successful recovery of the sites after clearing of *A. mearnsii*. Broadly put, this study brings us one step closer to understanding the complex nature of invasions and how these in turn can influence grazing availability.

## 5.3.  FUTURE RESEARCH RECOMMENDATIONS

This study has demonstrated the geographical focus of invasion by acacia species in relation to priority grazing areas. Trends over time on the local scale suggest a growing problem of significant impact in terms of ecosystem service delivery for some communities reliant on rangeland services with likely livelihood and well-being implications with respect to cultural and provisioning ecosystem services. This study points to several obvious next steps in terms of research into the little explored area of invasive aliens and ecosystem services.

In South Africa management of invasive alien plants has to a large extent been guided by water and biodiversity conservation concerns (Le Maitre *et al.,* 2000). Considering the socio-economic value of the livestock industry (DAAF, 2012; Shackleton *et al.,* 2001; Ainslie, 2002, 2005) and the potential impact resulting from invasion by alien plants, such as *A. mearnsii,* as demonstrated by this study (see chapter 3), I recommend that future investments into alien plant management, particularly alien plant clearing prioritisation also be guided by the location and distribution of high potential agricultural areas. This requires directing research into areas that are critical for ecosystem service delivery as well as those areas being threatened by, or already affected by invasive alien plants. Furthermore, understanding the dynamics between multiple invasive alien species and a variety of services important to South Africans also need to be investigated in this regard. Such studies would allow us to gain a better understanding of impacts of invasive plants on ecosystem services and to assess benefits from both a social and ecological perspective. This research needs to be carried out in such a way as to

enable us to up-scale these findings and use them in policy development, as well as in the prioritisation of alien clearing at national and provincial levels.

Previous studies on restoration of invaded ecosystems have demonstrated mixed successes with regards to clearing operations (Blanchard and Holmes, 2008; Morris *et al.,* 2008; Ndhlovu *et al.,* 2012). These varying successes have been attributed to the variety of clearing methods and follow up treatments employed, site specific factors and characteristics of the invaded ecosystem, the extent of invasion, and the characteristics of the invading species (Blanchard and Holmes, 2008; Morris *et al.,* 2008). While some areas can recover following the removal of invasive aliens without further interventions (Blanchard and Holmes, 2008; Ruwanza *et al.,* 2012) other areas require additional inputs and actions. Extensive further research into clearing practices associated with these variables is required so as to determine what works where and why, especially on relation to grazing as a driver of land alien plant invasions. Detailed monitoring programs that capture information on key variables enabling us to make these determinations regarding drivers, effects and outcomes are vital, and will enable us to optimise these activities in relation to expenditure.

The high cost associated with clearing operations and even follow up treatments are extremely restrictive. It is not always possible for individual landowners to take these actions on their own. This is particularly true when these happen to be poorly resources emerging farmers that are beneficiaries of the Land Redistribution for Agricultural Development (LRAD) program. In such cases government assistance to emerging farmers in this process of clearing invasive alien plants will both enhance the speed of recovery and the chances of creating sustainable rural economies.

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Here too, a detailed understanding of appropriate clearing approaches will affect the amount of time it takes for these farmers to succeed.

Rangeland ecosystems are exposed to various drivers that can result in their degradation (Hoffman and Todd, 2000; Hoffman and Ashwell, 2001). Many of these drivers, including climate change (Richardson *et al.,* 2000) and injudicious use and management of natural resources (Hoffman and Todd, 2000; Hoffman and Ashwell, 2001; Reyers *et al.,* 2009) have the potential to facilitate degradation by invasive alien plants. We also need to start thinking of novel ways to address these factors that may indirectly facilitate invasion by alien plants.

In conclusion, the negative impacts of invasive alien plants on grazing resources as an ecosystem service that supports livestock production and associated livelihoods are substantial and warrant the same focus that impacts of invasive alien plants on water resources have received. While the clearing of invasive alien plants in catchments and along water courses has been the responsibility of the Department of Water Affairs, the Department of Agriculture, Fisheries and Forestry may need to take responsibility for the clearing of areas of high agricultural potential.

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