THE POTENTIAL ECONOMIC IMPLICATIONS OF BLACK LOCUST (*ROBINIA PSEUDOACACIA* L.) ON AGRICULTURAL PRODUCTION IN SOUTH AFRICA.

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ABSTRACT

Black Locust (*Robinia pseudoacacia* L.) is an invasive deciduous, strongly suckering, broad-leaved tree that has the potential to be widely distributed across a large portion of South Africa. *Robinia pseudoacacia* has invaded all nine of South African provinces, with large infestations found in the Eastern Cape, Kwa Zulu-Natal, Free State and Gauteng provinces. The invasive tree has the potential to spread into livestock grazing lands in South Africa. Because *R. pseudoacacia* has the ability to spread and thrive in a variety of habitats and resists control, the distribution of the invasive tree into grazing land poses a problem for landowners. The potential economic impacts of *R. pseudoacacia* on agricultural production stem from the trees ability to reduce the carrying capacity of livestock.

This study estimated the potential economic implications of *R. pseudoacacia* on agricultural production in South Africa, specifically looking at the livestock sector. The prevalence of *R. pseudoacacia* potential distribution was calculated by using a maximum-entropy predictive habitat model, MAXENT. The distribution of livestock, based on grazing capacity (ha/LSU), in South Africa was then determined. The potential direct economic impacts were estimated by assessing the impact of the potential distribution of *R. pseudoacacia* on the carrying capacity of livestock. The results showed that an infestation of *R. pseudoacacia* has the potential to reduce the gross margin in the livestock sector by between approximately R130 million and R961 million, dependent on the probability of invasion.

Therefore, the potential invasion of *R. pseudoacacia* can have detrimental effects on the livestock sector in South Africa. The potential high levels of foregone income and business activity found in this study reaffirm the need to devote resources to develop a viable, economical and effective control method, such as biological control.

Keywords: invasive alien plants, Black locust (*Robinia pseudoacacia*), livestock, grazing capacity, MAXENT, economic impact.

DECLARATION

This thesis has not been submitted to a university other than Rhodes University, Grahamstown, South Africa. The work presented here is that of the author, unless otherwise stated.

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Luke Humphrey Date

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LIST OF ABBREVIATIONS

ARC-Agricultural Research Council

AUC-Area under curve

BGIS-Biodiversity Geographic Information System

CBD-Convention on Biological Diversity

CEM-Climatic envelope model

EPWP-Expanded Public Works Programme

GARP- Genetic algorithm for rule set prediction

GBIF-Global Biodiversity Information Facility

IAP-Invasive alien plants

IAS-Invasive alien species

NLC-National Land Cover

NPP-Net primary production

NPV-Net present value

QDS-Quarter degree squares

RWF-Rand Water Foundation

SANBI-South African National Biodiversity Institute

SAPIA-Southern African Plant Invaders Atlas

SDM-Species distribution modelling

U.K.-United Kingdom

U.S.-United States

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CHAPTER 1

INTRODUCTION

1.1 INVASIVE ALIEN PLANTS

Invasive alien plants (IAPs) are a problem of global significance and hence have recently received much attention (Van Wilgen *et al.*, 2001b; Brunel *et al.*, 2013). Perrings *et al.* (2000) stated that this is due to an increased recognition of the occurrence and the severity of the consequences associated with IAPs. However, the problem is not recent in nature as the movement of organisms, which according to Mack *et al.* (2000), is a human driven phenomenon that is believed to date back 500 years (Shaughnessy, 1980; Zimmermann *et al.*, 2004). De Lange & Van Wilgen (2010) suggested that the growth in human population and the expansion of global trade has led to the widespread distribution of species beyond their native ranges. IAPs are recognised as invasive species and are deemed the most environmentally problematic of all invasive species (Perrings *et al.*, 2000). Furthermore, IAPs pose significant threats to the biodiversity and functioning of the world's ecosystem and the services they provide (Mack *et al.*, 2000; Pimentel *et al.*, 2005; Forsyth *et al.*, 2012). The Convention on Biological Diversity (CBD) broadly defines an invasive alien species as:

> "a species whose introduction and spread threatens ecosystems, habitats or species with socio-cultural, economic and/or environmental harm, and/or harm to human health" (CBD, 2002: 2).

Non-indigenous species, according to Perrings *et al.* (2000), are successful invaders into new habitats as their natural control mechanisms do not occur in the adventive range. Aggressive invaders have the ability to spread far from parent plants and cover large areas. Invasions into natural ecosystems by non-indigenous species threaten the sustainable use of benefits derived from such ecosystems (Van Wilgen *et al.*, 2001a). Furthermore, IAPs may have economic impacts, cause environmental harm or adversely affect human health (Pimentel *et al.*, 2001; Brunel *et al.*, 2013). IAPs have been recognised globally as the second largest threat to biodiversity (Richardson & Van Wilgen, 2004), leading to a loss in ecosystem value (Lonsdale, 1999). A loss in biodiversity may potentially lead to a loss in ecosystem functioning, as certain species are fundamental to certain ecosystem functions (Perrings *et al.*, 2000). IAPs are known to have other significant environmental impacts, as they spread at alarming rates, alter nutrient cycles and food webs, produce large numbers of seeds, form dense stands and thereby, disrupt various ecosystem processes (Zimmermann *et al.*, 2004; Moran *et al.*, 2005; Eviner *et al.*, 2012). Furthermore, IAPs aggravate land-system changes, influence the quality and quantity of water resources, and displace native vegetation, all of which threaten the delivery of ecosystem services - the foundation of human well-being (Drake *et al.*, 1989; Kaiser, 1999; Richardson & Van Wilgen, 2004; Van Wilgen *et al.*, 2004; Zimmermann *et al.*, 2004; Villamagna & Murphy, 2010; Le Maitre *et al.*, 2014).

A significant number of IAPs, particularly trees and shrubs, have invaded South African ecosystems (Henderson, 2001; Henderson, 2007; Kotzé *et al.*, 2010), many of which are already well established and have negative ecological and economic impacts (Van Wilgen *et al.*, 2004). According to Richardson & Van Wilgen (2004), South Africa is considered one of the most invaded countries in the world, with more than ten million hectares of land being invaded by over 180 IAPs (Van Wilgen *et al.*, 2001a). In recent years, IAPs have consumed excessive volumes of South African water supplies (Le Maitre *et al.*, 2002), they have invaded pristine environments (De Wit *et al.*, 2001; Sheppard *et al.*, 2006) and displaced native vegetation (Higgins *et al.*, 1997; Enright, 2000; Van Wilgen *et al.*, 2001b). This global phenomenon carries a host of environmental and socio-economic implications, costing billions of Rands annually in lost revenue and increasing control costs (Pimentel, 2002; De Lange & Van Wilgen, 2010; Perrings *et al.*, 2010; Van Wilgen *et al.*, 2012).

Invasions are of economic concern because of their impact(s) on human well-being (Emerton & Howard, 2008). Invasions can be classified in economic terms as being economic externalities. Richardson & Van Wilgen (2004) believe that IAPs not only have adverse environmental impacts, but also threaten the economic productivity of a country. Various economic studies conducted have attempted to estimate the economic consequences of IAPs (Pimentel *et al.*, 2000; Van Wilgen *et al.*, 2001a; Cook *et al.*, 2007; Oreska & Aldridge, 2011; Wise *et al.*, 2012). The majority of these studies suggested that the costs of IAPs are significant.

There is an increasing recognition of the importance of the effect that IAPs have on the agricultural sector (Cullen & Whitten, 1995; Leitch *et al.*, 1996; Pimentel *et al.*, 2001; Acquaye *et al.*, 2005; De Neergaard *et al.*, 2005; Eagle *et al.*, 2007; Dube, 2010). IAPs could have substantial impacts on forage quantity and quality, increasing management costs, imposing land use changes, and thereby reducing agricultural production, output and profitability (Eagle *et al.*, 2007). IAPs have the potential to have an impact on the livestock sector, as a reduction in the carrying capacity of livestock disrupts agricultural production. IAPs pose as a threat to livestock production by lowering yield and quality of forage, interfering with grazing patterns, poisoning livestock, restricting access to grazing lands, and increasing costs of managing and producing livestock (Ditomaso, 2000). In South Africa, invasive terrestrial trees are of a particular concern to the agricultural sector and, hence, to the economic development of southern Africa as a whole (Gorgens & Van Wilgen, 2004). The agricultural sector's significance in South Africa is largely because of its potential to create jobs and is a key focus of the New Growth Path (Republic of South Africa, 2013).

The following chapter provides a brief introduction and overview of Black locust (*Robinia pseudoacacia* L.) (1.2) and discusses agriculture in South Africa (1.3). The chapter further addresses the problem statement (1.4), the aims and objectives of the study (1.5), the method to be followed (1.6), and outlines the structure of the thesis (1.7).

1.2 BLACK LOCUST (*ROBINIA PSEUDOACACIA* L.)

Robinia pseudoacacia an invasive deciduous tree (Cierjacks *et al.,* 2013), is ranked as a problematic invader (Kurokochi *et al.,* 2010; Henderson, pers comm, 2015). Although native to south-eastern United States (U.S.), the broad-leaved tree has been widely planted and become naturalised elsewhere in temperate North America, Europe, Australia and southern Africa (Sheppard *et al.,* 2006). *Robinia pseudoacacia* is known to have a number of negative environmental and socio-economic impacts. The invasive tree is a threat to existing ecosystems as it spreads rapidly from suckering roots and seeds creating monocultures that displace native species (Sabo, 2000). It is a prolific water user, capable of invading pristine environments, and its seeds, leaves and bark are toxic to both humans and animals (Cooper & Johnson, 1984; Cheeke & Shull, 1985; Sabo, 2000; Sheppard *et al.*, 2006). Thus, *R. pseudoacacia* possesses most of the characteristics associated with "weediness" (Sabo, 2000).

In South Africa, *R. pseudoacacia* is causing extensive negative ecological and economic effects: it impacts on native biodiversity (Van Wilgen *et al.,* 2001), fragrant blossoms compete with indigenous plants for essential pollinating bees (PCA, 2005) and the poisonous proteins in the tree affect farmers' livestock (Sabo, 2000). Henderson (pers comm, 2015) believes that "*R. pseudoacacia* could be having as much of an economic impact as some of the other alien Acacia species". The implementation of control measures in order to combat the spread of *R. pseudoacacia* have proven difficult due to its rapid growth and clonal spread (Akamatsu *et al.,* 2014). Hildegard (pers comm, 2015) stated, "I am quite concerned about *R. pseudoacacia* - not so much based on its current distribution or rate of spread, but based on its potential spread".

Several mechanical and herbicidal control attempts have been made in South Africa, but have been proven unsuccessful (Coulsen, 2015).

1.3 AGRICULTURE IN SOUTH AFRICA

Despite contributing only 2.3% to South Africa's primary production in 2013, agriculture is an important employer of labour, is responsible for supplying local and global commodities, has a significant influence on alleviating food scarcity, provides foreign exchange earnings, is a key focus of the New Growth Path, as well as a significant contributing component to South Africa's socio-economic development (Nieuwoudt *et al.,* 2004; Eagle *et al.,* 2007; Republic of South Africa, 2013; De Lange & Mahumani, 2013). In South Africa, there are approximately 100 million hectares of agricultural land, of which 72% is used for extensive grazing. Therefore, agricultural land in South Africa is primarily livestock-based (Meissner *et al.,* 2013). Livestock production not only contributes substantially to food security in South Africa (Meissner *et al.,* 2013), but forms a critical part of South Africa's socio-economic and socio-political stability (Tibane & Vermeulen, 2014). Furthermore, livestock is the primary driver underpinning sustainable rural agriculture (Palmer *et al.,* 2010). The grassland biome is one of the most valuable biomes in South Africa, in terms of agricultural production. Much of the increasing demand for meat and dairy products is supplied from the grassland ecosystems (Boval & Dixon, 2012). The grassland biome is an important form of land use, as livestock farmers owners depend on grasslands for their livelihoods (Bouwman *et al.*, 2005: Suttie *et al.,* 2005).

1.4 PROBLEM STATEMENT

Robinia pseudoacacia has the potential to invade livestock grazing lands in South Africa, disrupting agricultural production. The potential economic impacts of *R. pseudoacacia* on agricultural production stem from the trees ability to reduce the carrying capacity of livestock (Bungsund *et al.*, 1999). *Robinia pseudoacacia* reduces the carrying capacity of livestock in two ways. Firstly, infestations restrict access to grazing lands through encroachment, decreasing the amount of available forage for livestock and interfering with grazing patterns. Secondly, the seeds, leaves and bark of the tree are toxic to livestock. Therefore, a reduction in carrying capacity of livestock disrupts agricultural production, output and profitability.

1.5 AIMS AND OBJECTIVES

This study involves estimating the extent of a potential infestation of *R. pseudoacacia* on livestock grazing land, estimating the effects of the infestation on agricultural production, with specific emphasis on the livestock sector, and estimating the potential direct economic implications.

1.5.1 AIM

Considering the economic importance of agricultural production in South Africa, the potential impact which *R. pseudoacacia* has on agricultural production needs to be determined. Therefore, the aim of this study is to estimate the potential economic implications of the uncontrolled spread of *R. pseudoacacia* on agricultural production in South Africa, using the Clarens region as a case study.

1.5.2 OBJECTIVES

In order to achieve the aim of this study, certain objectives need to be fulfilled:

- ✓ Determine the potential distribution of *R. pseudoacacia* in South Africa.
    - o Construct an ecological niche model to predict the suitability of various regions in South Africa for the potential establishment of *R. pseudoacacia*.
    - o Determine the establishment of *R. pseudoacacia* in the different biomes in South Africa.
- ✓ Determine the grazing capacity of livestock in South Africa.
    - o Determine the distribution of livestock, based on grazing capacity (ha/LSU), in South Africa and in the grassland biome.
- ✓ Determine the potential economic implications of the potential distribution of *R. pseudoacacia.*
    - o Estimate the impact of *R. pseudoacacia* on the grazing capacity of livestock.
- ✓ Conduct a case study in the Clarens region.
    - o Determine the perceptions, impacts and the distribution of *R. pseudoacacia* within the study site.
    - o Determine the measures of control and the financial implications.

1.6 METHOD TO BE FOLLOWED

This study attempts to estimate the potential economic implications of *R. pseudoacacia* on agricultural production in South Africa, looking specifically at the livestock sector. For this, it is necessary to determine the potential spread of *R. pseudoacacia* in South Africa. A maximum-entropy predictive habitat model (MAXENT) will be used in combination with suitable bioclimatic predictor variables (Hijmans *et al.*, 2005). Various environmental layers will be overlaid onto the model to determine the potential areas of invasion in South Africa. The distribution of livestock, based on grazing capacity (ha/LSU), in South Africa and in the grassland biome will be determined. Thus, the potential direct economic impacts will be estimated by assessing the impact of the potential distribution of *R. pseudoacacia* on the carrying capacity of livestock.

1.7 STRUCTURE OF THE STUDY

This thesis is structured with the following chapters. Chapter one of this study briefly introduces IAPs, discusses them within a South African context as well as discusses their environmental and economic impacts. The chapter further briefly introduces *R. pseudoacacia* and discusses its invasive characteristics. The problem statement, aims and objectives as well as the method to be followed are addressed.

Chapters two and three are both literature review chapters. Chapter two provides a general literature review of IAPs, specifically addressing the effects of IAPs, IAPs in South Africa, the economics of IAPs and various methods of control for IAPs. Chapter three provides an in-depth discussion on *R. pseudoacacia.*

The methodological approaches, study design and research methods are recorded in chapter four. The data collection process and the analytical tools to be used are also described and explained, with respect to their application and appropriateness.

Chapter five analyses and discusses the results of the primary data collected for the case study in the Clarens region. The chapter addresses the study site and the current distribution of *R. pseudoacacia* followed by the survey results, concerning respondents' perceptions and various control attempts.

Chapter six addresses the results of the desktop study by analysing the secondary data. The economic implications of the potential establishment of *R. pseudoacacia* in grazing lands in

South Africa are then determined, as is an analysis of the economic implications in the grassland biome.

Finally, chapter seven concludes the study by discussing the results and putting them into context. Recommendations and areas for further research are identified, followed by concluding remarks.

CHAPTER 2
LITERATURE REVIEW

2.1 INTRODUCTION

Invasive alien species (IAS) are species whose introduction and/or spread outside their natural past or present distribution, pose as a threat to biological diversity (CBD, 2008). Brunel *et al.* (2013) suggested that IAS are a result of human assistance, either accidentally or deliberately, to an area that they could have not reached on their own.

Many studies have attempted to determine what makes a species invasive, as well as documenting the effects of IAS (Parker *et al.*, 1999). IAS are considered to be a significant and continuous growing threat to biodiversity and ecosystem stability worldwide, which impacts on important industries such as agriculture, forestry, fisheries, power production, and international trade (Wilcove *et al.*, 1998; Mack *et al.*, 2000; Lovell & Stone, 2005; García-Llorente *et al.*, 2008).

In South Africa, IAS are a direct threat to biological diversity, water security, the ecological functioning of natural systems and the productive use of land (Republic of South Africa, 2015b). IAS have had global economic effects. Pimentel *et al.* (2002) suggested that invasive alien plants (IAP) alone have caused global economic losses estimated at around US$1.5 trillion, representing 5% of global GDP. Costs of inaction have been estimated to be approximately US$138 billion per annum for the United States (U.S.), US$14.45 billion for China and over €12 billion in Europe (Kettunen *et al.*, 2009). In New Zealand, the costs of invasive species' impacts are estimated to amount to approximately 1% of GDP (Bertram, 1999).

The following chapter discusses relevant IAP literature, specifically addressing trade and transportation of IAPs (2.2), the effects of IAPs (2.3), IAPs in South Africa (2.4), the economics of IAPs (2.5), the impact of IAPs on agriculture (2.6), the value of livestock in South Africa (2.7), and various methods of control for IAPs (2.8), followed by a conclusion in section (2.9).

2.2 TRADE AND TRANSPORTATION

Simberloff (2001) stated that trade, travel and tourism have resulted in an increased spread of organisms worldwide. Plant invasions are closely related to global human travel and trade of goods and services (McNeely, 2001), which can be tied to historical and current human activities. Branco *et al.* (2015) believe that globalisation trends such as population growth and the liberalization of regulatory trade regimes (Perrings *et al.*, 2005; Kettunen *et al.*, 2009), have led to an increase in biological invasions. Perrings *et al.* (2005) proposed that the introduction of IAPs has increased worldwide due to the development of the transportation systems, which in turn has increased trade and tourist activities. Shipping is one of the greatest distributors of IAPs worldwide as shipping carries more than 80% of the world trade (Bax *et al.*, 2003). Thus, Bax *et al.* (2003) believe that these vectors are responsible for moving species, leading to unintentional species introductions. Humans have transported and traded plant species for millennia, dating as far back as 1500 AD, when radical changes in patterns of human demography, agriculture, trade and industry began (Preston *et al.*, 2004). Hulme (2009) stated that the understanding of the scale, mechanisms and historical trends of trade is essential for managing the risks of invasion. According to Van Wilgen *et al.* (2004), when IAPs are transported to a new continent without attendant enemies, they tend to exhibit 'ecological release.' This phenomenon describes how species spread rapidly and 'out-compete' native species.

2.3 EFFECTS OF INVASIVE ALIEN PLANTS

IAPs can have significant effects on ecosystem services[1] (Richardson & Rejma´nek, 2011; Branco *et al.*, 2015). IAPs effect ecosystem processes,[2] which are the key mechanisms of ecosystem services (Charles & Dukes, 2007). According to Sieg *et al.* (2010), the increasing effects of IAPs on ecosystem services are now being used as criteria for prioritising efforts to remove or manage invasive plants. IAPs have different ecophysiological traits, which sets them apart from native species (Dassonville *et al.*, 2008). Their invasive characteristics pose as a threat to ecosystem processes which are fundamental to flora, fauna and human well-being on a global scale (Mooney, 2005; Millennium Ecosystem Assessment, 2005). The intentional or unintentional introduction of an IAP species is, however, handicapped by uncertainty due to the unknown potential invasive behaviour (Brunel *et al.*, 2013). Various studies (Zavaleta, 2000; Charles & Dukes, 2007; Pejchar & Mooney, 2009) have demonstrated how IAPs can

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[1] Ecosystem services are the benefits humans derive from ecosystems (e.g. water supply, air and water purification, pollination, provision of food and fibre) (Wallace, 2007).
[2] Ecosystem processes are the conversion or movement of matter or energy resulting from interactions between organisms and their environments (e.g. evapotranspiration, decomposition, nutrient cycling, water infiltration and storage) (Wallace, 2007).

result in unintended decreases in ecosystem services. Gorgens & Van Wilgen (2004) suggested that many of the invasive woody plants in South Africa have had an effect on ecosystem services. Effects include decreases in surface water and the magnitude of stream flow, as well as increased evapotranspiration rates. Invasion of shrubs and trees into South African fynbos has greatly decreased water provisioning in a system already highly limited by water (Mark & Dickinson, 2008). Salt cedar (*Tamarix spp.*) have invaded areas throughout the south-western United States (U.S.), where they have altered stream channel morphology, resulting in increased flooding frequency and severity (Zavaleta, 2000). Moreover, Salt cedar consumes an additional 1.4-3.0 billion cubic meters of water each year more than the native riparian species. This was calculated to a forgone loss of US$26.3-US$67.8 million in water per annum. This water would otherwise be available for other ecosystem services.

Measuring the effects of IAPs on ecosystem services tends to be difficult. This is due to various challenges, which prevent researchers from confidently extrapolating those processes as proxies for services (Eviner *et al.*, 2012). Pejchar & Mooney (2009) suggested that one of the main reasons is because little is currently known about the mechanisms by which invasive plants affect services. Although IAPs tend to have negative impacts on native plant communities, these may not always be detrimental. However, it is important to consider that a shift in environmental conditions may no longer support native species and that invasive species may be critical contributors to the resilience of ecosystem services (Vilà *et al.*, 2009).

Quantifying the impact of IAPs on ecosystem services is an important aspect when considering effective practices and policies for invasive species management (Eviner *et al.*, 2012). Cook *et al.* (2007) used a stochastic bio-economic model, which estimated the economic impact of an IAP before its arrival within a new habitat. A hypothetical invasion of the Varroa bee mite (*Varroa destructor)* into Australia was used to test the model. This invasive species reduced bee honey production. The study found that if the invasive species were to be prohibited from the country for 30 years, economic costs avoided would be US$16.4-38.8 million per annum, illustrating the substantial benefits of maintaining an important ecosystem service.

Van Wilgen *et al.* (2008) conducted a study in South Africa to determine the current and potential impacts of IAPs on selected ecosystem services. The study found that if the selected IAPs in the study were to occupy the full extent of their potential range, the potential reduction in surface run-off would be eight times greater. Ground water recharge would also be affected, potentially amounting to approximately 1.5% of the estimated maximum reductions in surface water runoff. The reduction in grazing capacity could increase by 71%. These results suggest that the future impact of IAPs on ecosystem services is potentially high. Van Wilgen *et al.* (2008: 347) stated that "reductions in the provision of ecosystem services of the magnitude estimated in this study would generate significant, negative economic consequences."

Research has therefore shown that IAPs can - and do - have undesirable consequences for ecosystem services. Furthermore, IAPs pose significant impacts at the species, community and ecosystem levels (Vilà *et al.,* 2011).

2.3.1 COSTS OF INVASIVE ALIEN PLANTS

The introduction of IAPs into a non-native region has certain implications. Due to their invasive characteristics, such as faster growth rates, higher biomass and higher net primary production (Ehernfeld, 2003), IAPs often displace native vegetation. In addition, the displacement of indigenous plant communities can potentially lead to further implications, such as a decrease in agricultural productivity and disruption in ecosystem processes (Enright, 2000).

The costs of IAPs on ecosystem functioning will be discussed, namely; climate, soil stabilisation, floods and fires, cultural series, nutrient cycling, land, evapotranspiration and water.

a CLIMATE

The introduction and replacement of non-native IAPs on native plant species has the potential to effect the level of carbon dioxide released by the plants. This is because of the difference in the carbon storage capacity of the IAP and the native species. Prater *et al.* (2006) conducted a study to determine the net loss of carbon between the invasive grass species and the native sagebrush ecosystem in the U.S. Great Basin region. It was found that the net loss of carbon sequestration of (-0.5 μmol m^{-2} s^{-1}) over roughly 12.7 million ha, had the potential to contribute to climate warming. A similar study was conducted by Kaufman *et al.* (1998), who determined the carbon storage capacity that had been lost. In the Brazilian Amazon, fire-prone non-native pasture grasses have steadily replaced rainforest. Kaufman *et al.* (1998) found that carbon pools in post-fire pasture were only 3% of adjacent primary forest. However, in contrast to the previous studies conducted, Hughes *et al.* (2006) found that more carbon can be sequestered when woody species replace native grasslands. This phenomenon was found with the encroachment of Honey mesquite (*Prosopis glandulosa*) into the southern Great Plains (U.S.).

b SOIL STABILISATION

The introduction of IAPs exacerbate the rate of erosion, which in turn can affect the quality of water. Pejchar & Mooney (2009) suggested that increased levels of erosion could potentially result in (i) turbid water (ii) limit agricultural production and (iii) compromise the stability of land under homes and other infrastructures. There have been many cases recorded where IAPs have been introduced deliberately to limit erosion (Jensen Augustine *et al.*, 2006; Rédei *et al.*, 2011; Phillips *et al.*, 2013); however, these introductions often come with unintended consequences for other ecosystems services (Pejchar & Mooney, 2009). Forseth & Innis (2004) conducted a study on Kudza (*Pueraria lobata*). The invasive plant was introduced into the south-eastern U.S. in 1876 to assist with erosion control. The plant now covers roughly 3 million ha and is spreading at an alarming rate of 50 000 ha per annum. Moreover, Kudzu poses as a major economic liability, affecting air quality and smothering other native trees. Similarly, Black Locust (*R. pseudoacacia*) was introduced into Japan and has been planted extensively to help with erosion control and forestation in upper river basins (Ecological Society of Japan 2002; Rédei *et al.*, 2011). The invasive tree has spread significantly and is now considered to be in abundance. The Port Jackson willow (*Acacia saligna*) has been used extensively for soil stabilisation in South Africa, particularly to help stabilise sand dunes (Midgley & Turnbull, 2003), but in some areas, Port Jackson willow has developed into an invasive species with a wide range of negative impacts. This is especially apparent in the unique South African fynbos systems, where Port Jackson willow has displaced native species mainly through altering the fire regime (Musil, 1993; Holmes, 2002).

c FLOOD AND FIRE

By increasing the intensity or frequency of fires or floods, IAPs can exclude native species and increase the risk to nearby human communities. IAPs have the ability to increase the level of flood risk as they tend to narrow stream channels and decrease holding capacity, forcing water up and over the river banks (Zavaleta, 2000). Various studies have been conducted to determine the costs that IAPs have borne from an increase in flooding. The introduction of Salt cedar (*Tamarix spp.*), which resulted in increased flooding, caused an estimated loss of US$52 million annually. Thunberg *et al.* (1992) conducted a study in Florida (U.S.) to determine the effect of the removal of aquatic IAPs from dams, rivers and waterways. It was estimated that US$10 million in flood damages to residential structures were avoided annually (Thunberg *et al.*, 1992).

D'Antonio (2000) conducted various studies concerning the alteration of fire regimes by IAPs. IAPs have a significant impact on the fuel properties as well as the frequency, intensity, extent, type and seasonality of fires. Chambers *et al.* (2007) found that Cheat grass (*Bromus tectorum*) is fire-adapted and has permanently altered the native plant community. The native plants are unable to regenerate in the face of heightened fire frequency. D'Antonio & Vitousek (1992) recorded similar results in their study in Hawaii. The invasion of exotic grasses altered fire regimes, which could potentially result in substantial social and economic costs. The Broad-leaved paperbark (*Melaleuca quinquenervia*) caused US$250 million in fire damages by the year 2010, due to increasing fuel loads in Florida (Serbesoff-King, 2003).

d CULTURAL SERVICES

Pejchar & Mooney (2009: 502) defined the impacts of IAS on cultural services as "those attributes of an ecosystem that are non-consumptive and are difficult to assess because they are based on personal and local value systems". Cultural services threatened by aquatic and terrestrial IAPs include recreation and tourism. Lake Tahoe, in the U.S., is suggested to be worth US$30-45 million a year. The potential introduction of Eurasian water milfoil (*Myriophyllum spicatum*) can have significant impacts on the recreation revenue of the lake. The Eurasian water milfoil, which forms dense mats of vegetation on the surface of the water, takes over lakes which complicates fishing and boating. It was suggested that even a 1% loss in recreation revenue would cost up to US$500 000 per annum (Eiswerth *et al.*, 2005). The Yellow star-thistle (*Centaurea solstitialis*) poses as a significant threat to hikers in the western U.S. Its sharp thorns are able to lacerate hikers, which has decreased the recreation value of the area (Dudley, 2000). Moreover, the plant has cost millions per year in the loss of livestock forage value (Eagle *et al.*, 2007).

e NUTRIENT CYCLING PROCESS

The introduction of IAPs has the potential to change many components of the carbon (C), nitrogen (N), water, and other cycles of an ecosystem. Ehernfel (2003: 504) stated that when "the species composition of a community changes, due to an invasion, there follows consequent changes in the nutrient cycling process." Ehernfel (2003) conducted a study to determine a change in components of the carbon (C), nitrogen (N) and water for the invaded and non-invaded systems for invasions of 56 species. The results suggested that invasive plant species frequently increase biomass and net primary production (NPP), increase N availability, alter N fixation rates, and produce litter with higher decomposition rates than co-occurring natives. Contrary to these results, the study also suggested that opposite patterns occur, patterns of difference between exotics and native species show no trends in some other components of nutrient cycles. Ehernfel (2003), however, concluded that invasions of plants could extensively affect the storage and release of C, N, and other substances. On the contrary, Scott *et al.* (2001) found that changes in ecosystem processes triggered by plant invasions may be viewed as beneficial rather than adverse. The invasion of Mouse-ear hawkweed (*Hieracium pilosella*) in New Zealand was found to increase soil organic matter in overgrazed pastures, thus helping to improve productivity.

f EVAPOTRANSPIRATION

IAPs result in significant losses of water caused by extensive evapotranspiration (Chamier *et al.,* 2012). Water hyacinth (*Eichhornia crassipes*) has exacerbated the problem of evapotranspiration, created by the dense mats it forms over water surfaces. Water loss through evapotranspiration by water hyacinth ranges between 1.02 and 9.8 times greater than evaporation from open water surfaces (Singh & Gill, 1996). However, it is to be noted that high evapotranspiration rates cause sufficient drying of the soil to allow crop growth, which was seen in the case with Blady grass (*Imperata cylindrical*) (Hartemink & O'Sullivan, 2001).

g WATER

Mooney (2005) proposed that IAPs effect the availability of drinking water as well as irrigation water if the plant possesses at least one of the following characteristics: (i) deeper roots (ii) higher evapotranspiration rates or (iii) greater biomass. Gerlach (2004) found that the Yellow star-thistle (*Centaurea solstitialis*) depleted the soil moisture causing a significant loss of water to the Sacramento watershed, valued at between US$16 million and US$75 million per annum. The Broad-leaved paperbark (*Melaleuca quinquinervia*) is an alien invasive tree found in Florida and Australia. The tree has taproots which run deep into the soil, consuming large amounts of water. The host native plants are therefore starved of water, as they cannot compete with the invasive tree, which potentially depletes the groundwater (Schmitz *et al.,* 1997). It has been found that tall alien trees generally reduce total annual and low-season streamflow and increase evapotranspiration (Scott *et al.*, 2000; Bosch & Hewlett, 1982; Dye & Jarmain; 2004). In a study by Le Maitre *et al.* (2002), four catchment areas were assessed in terms of their impacts and costs of invasions. Between 2% and 54% of the catchments had been invaded to some degree and this effected a reduction in river flows of between 6% and 22.1% per catchment. The estimated cost of the control programmes to prevent these losses cost between US$4.1 and US$13.2 million per catchment.

2.3.2 CONFLICTING INTERESTS

There are mixed views that exist when dealing with IAPs. Predicting the potential future impact is deemed to be practically difficult as many IAPs, although viewed as having major (negative) impacts, also have a variety of benefits (De Wit *et al.*, 2001). Brunel *et al.* (2013) suggested that there seems to exist certain divergences in the perception of IAPs and whether the alien origin of a species is a reliable heuristic for predicting problematic spread. Davis *et al.* (2011) highlighted the importance of the opportunity costs of a strict prevention of IAPs, including the opportunity cost of losing the benefits that some alien species might provide. Although, IAPs are associated with having significant costs, some species are still viewed as being beneficial. For example, despite Black wattle (*Acacia mearnsii*) displaying many benefits in the rural communities of South Africa, the species also has various costs (De Wit *et al.*, 2001). This has often lead to conflicts over how IAPs should be managed (Van Wilgen & Richardson, 2014).

a BENEFITS OF INVASIVE ALIEN PLANTS

Numerous IAPs introduced into a new country or region provide key services, whether this be for timber or erosion control (Pejchar & Mooney, 2009). Several species of the Mesquite (*Prosopis*) were introduced into South Africa in the 1880s and the 1960s for their ability to produce fodder and shade (Henderson & Harding, 1992; Wise *et al.*, 2012). Subsequently, mesquite is now propagated for other benefits such as timber, medicinal purposes and honey (Wise *et al.*, 2012)[3].

[3] However, it is to be noted that mesquite is a prominent invader, due to its environmental consequences (Dube, 2010; Wise *et al.*, 2012).

In South Africa, IAPs have beneficial impacts on rural low-income communities. Certain invasive *Acacia* and *Pinus* species are used for thatching, medicine as well as firewood which has an estimated economic value of US$2.8milion (De Wit *et al.,* 2001; Turpie *et al.,* 2003).

Plantations of IAPs as a source of timber have been established globally (Le Maitre, 1998). In South Africa specifically, over 1.52 million ha of primarily pines and eucalypts have been planted. This has provided South Africa with numerous benefits as plantation forestry contributes US$300 million to GDP and employs over 100,000 people (FOA, 1998; Le Maitre *et al.,* 2002).

De Wit *et al.* (2001) proposed that the Black wattle (*Acacia mearnsii*) has many benefits in South Africa. One of the largest benefits - and the reason for the plants introduction - was because of its tannins, which are extracted from the bark. Tanning agents are used in the production of soft leather. De Wit *et al.* (2001) assigned the tannins a Net Present Value (NPV) of US$363 million[4]. Black wattle also provides various other benefits, such as (i) firewood, which had a NPV of US$143 million and is used as an important fuel source for rural communities; and (ii) building materials used for branding, laths and poles which had a total NPV of US$22 million. De Wit *et al.* (2001) also mentions timber, pulp, wood chips, charcoal, nitrogen fixation and medical products as well as combating of erosion as other benefits of the black wattle, but these benefits were not assigned a NPV. De Neergaard *et al.* (2005) also suggested that the Black wattle provides for a valuable ‘natural’ resource, widely used for construction purposes. In addition, carbon sequestration was identified as a benefit, as standing plantations and invasions store carbon as a counter to carbon build-ups. The carbon sequestration had an NPV of US$24 million.

Many IAP species have deep root systems and have adapted for growth in a wide variety of degraded soils (González-García *et al.*, 2011). Such IAPs are used as a remedy for erosion control (Forseth & Innis, 2004). The Kudzu (*Pueraria lobate*) and Black Locust (*Robinia pseudoacacia*) have been used in the U.S. and Japan respectively as remedies for erosion control (Rédei *et al.*, 2011).

[4] Net present value (1998, 1US$=ZAR6).

2.4 INVASIVE ALIEN PLANTS IN SOUTH AFRICA

IAPs were first introduced to South Africa during the 1600s (Shaughnessy, 1980 & 1986). The Cape of Good Hope was a refurbishing point for European ships sailing to and from the "Spice Islands" of Indonesia. Zimmermann *et al.* (2004) suggested that the southern tip of Africa became a focal point for the establishment of IAPs as a result of ships arriving at this point from Europe, Australia and America. Therefore, there was an intentional or unintentional influx of IAS into South Africa[5]. South Africa is particularly vulnerable to IAP species, and Richardson & Van Wilgen (2004) believe that South Africa has some of the biggest problems with plant invaders in the world. IAPs have been reported to impact on human well-being, biodiversity, river courses, catchments, agricultural lands, and in wilderness and conservation areas (Zimmermann *et al.,* 2004). Much of the concern about IAPs in South Africa has been centred on the consequences for the conservation of biodiversity (Van Wilgen *et al.,* 2001a).

Large numbers of IAPs, including many trees and shrubs (Henderson, 2001), have invaded South African ecosystems (Henderson, 2007; Kotze *et al.*, 2010), many of which are already well established and have negative ecological and economic impacts (Van Wilgen *et al.*, 2004). More than ten million hectares of land alone in South Africa has been invaded by over 180 IAPs (Van Wilgen *et al.*, 2001a). Henderson (2001) noted that 155 invasive tree and shrubs species have been identified, as have a further 73 species of terrestrial weeds and 10 aquatic IAPs. According to Van Wilgen *et al.* (2012), 98 more species have been added to the list since then. It is estimated that these plants cover about 10% of the country and the problem is growing at an exponential rate. A large proportion of these species (approximately 50%) originated from central and tropical America, with about 25% originating from Europe, Asia and the Mediterranean, and roughly 13% originating from Australia (Henderson, 2001). Figure 2.1 illustrates the distribution of IAPs species in South Africa.

[5] IAPs were bought into South Africa for a range of purposes: timber for firewood, garden ornamentals, stabilising of sand dunes and/or hedge plants (Van Wilgen *et al.,* 2001a).

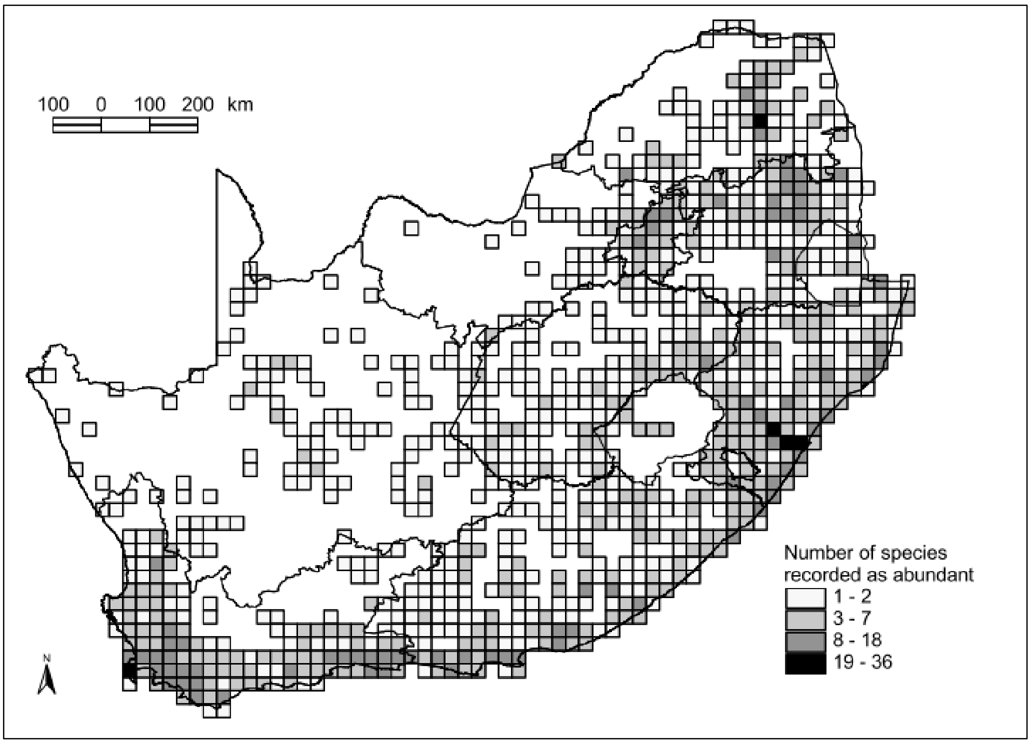

Figure 2.1. The distribution of invasive alien plant species in South Africa.
Source: South African Plant Invaders Atlas (Henderson, 1998; Henderson, 2001).

In recent years, studies have shown that IAPs use excessive volumes of South African water supplies (Le Maitre *et al.*, 2002; Brunel *et al.*, 2013; Le Maitre *et al.*, 2014). South Africa is a relatively water scarce country, thus one of the most detrimental effect of IAPs is their threat to the country's fresh water resources (Görgens & Van Wilgen, 2004; De Lange & Van Wilgen, 2010). McConnachie *et al.* (2012) believe that IAPs affect the quality and quantity of available water, which has further environmental and social implications. Furthermore, economic implications are also threatened as water is seen as an "economic good" (Rogers *et al.*, 2002). Thus, the effect of IAPs on water is seen to be increasingly limiting economic growth in South Africa (Görgens & Van Wilgen, 2004). Terrestrial invasive trees are the biggest suckers of water as they extract water from rivers, riparian areas and other surrounding areas (Van Wilgen *et al.*, 2001a; Brunel *et al.*, 2013). These trees often choke watercourses and therefore intercept water in the catchments *en route* to the watercourses (Moran *et al.*, 2005). Le Maitre *et al.* (2002) estimated that 7% South Africa's water supply is consumed by IAPs. They also documented that alien tree species cause river flow reduction in four river catchments of approximately 6-22%. Moreover, it was estimated that if control measures were not put into place and the IAPs were allowed to spread to full potential (26-30 years), the reduction in river flow could potentially range from 22% to 95% (Le Maitre *et al.*, 2002).

Existing literature indicated that the total costs of IAPs in South Africa are substantial (Higgins *et al.,* 1997; Pimentel *et al.,* 2001; Van Wilgen *et al.,* 2001a). Invasions of various plants have reduced the value of fynbos ecosystems by over US$11.75 billion (Higgins *et al.,* 1997). Moreover, the Agulhas Plain alone has an estimated cost (value of water loss) of invasion of US$3.2 billion (Turpie & Heydenrych, 2000) and the net present cost of invasion by Black wattle amounts to US$1.4 billion (De Wit *et al.,* 2001). A nationwide cost to clear IAPs in South Africa was conducted by Versfeld *et al.* (1998), who estimated a clearing cost of US$1.2 billion (roughly US$60 million per annum for 20 years). The clearing of the red water fern was part of this study, requiring potential total clearing costs of US$58 million (Van Wilgen *et al.,* 2001a).

The Southern African Plant Invaders Atlas (SAPIA) is a mapping project that collects data on IAPs in Southern Africa. The SAPIA database, which contains the distribution, abundance and habitat types of IAPs, lists over 70 000 locality records of more than 600 naturalized IAPs. At least 200 species have been flagged as being invaders or potential invaders. The SAPIA database is aligned with the South African National Biodiversity Institute's (SANBI) Early Detection and Rapid Response, both of which seek to provide for the management of IAPs in South Africa (Henderson, 2011).

SAPIA provides assistance for control programmes as it (i) determines the geographic extent and ecological requirements of IAPs (Wilson *et al.* 2007); (ii) provides early warning of new invaders or new foci of spread; (iii) predicts the potential spread of a species (Rouget *et al.*, 2004); (iv) provides a historical account of the introduction and expansion or contraction of invasive species (Richardson *et al.*, 2004), with and without biological control; and (v) can be used to prioritise IAS for management (Nel *et al.*, 2004; Henderson, 2011).

2.5 ECONOMICS OF INVASIVE ALIEN SPECIES

Traditionally, the issue of biological invasions is the responsibility of biologists. However, more recently, as IAPs have become more widely spread, their impacts on economic systems are more significant (Emerton & Howard, 2008). Evans (2003: 9) stated that "the economic discipline possesses the capability of valuing various market and nonmarket impacts and provides a means for assessing important trade-offs among various management alternatives, which can improve greatly the decision-making process for managing such risks." Thus, there is need for the use of economic approaches and tools to better understand and address IAPs

(Emerton & Howard, 2008). The valuation of invasive species and their potential effects is an important aspect. Invasions do not only occur from an environmental point of view, but can be economically motivated. Evans (2003) stated that nearly all of the invasions of alien species can be linked directly or indirectly to economic activities and Perrings *et al.* (2002) argued that the primary driver of the entry and establishment of IAPs is economic. Economic conditions or forces may expedite or encourage the transitions of a species to a situation where - once introduced - the species becomes invasive (Emerton & Howard, 2008). Examples include the liberalisation and deregulation of markets, or an expansion of trade and goods (Perrings *et al.*, 2000).

2.5.1 ECONOMIC CAUSES OF INVASIONS

The economic causes of invasions are either direct or indirect. Direct economic causes are related to production and consumption activities (Emerton & Howard, 2008). While the direct economic causes can either be intentional or unintentional, the economic conditions or forces often involve the transport or transfer of a species into a new environment (Vilà & Pujadas, 2001). Examples include the use of potential IAS for farming, food, biological control or biofuel productions (Emerton & Howard, 2008). Indirect economic causes, on the other hand, comprise of the underlying economic conditions or forces. Emerton & Howard (2008) believed that access to a particular market and for individuals to fill their needs, tastes and aspirations, are the broad indirect economic causes. More specific examples include subsidies, taxes, import quotas, multilateral agreements and access to food, cash and employment (Emerton & Howard, 2008).

2.5.2 CLASSIFYING INVASIONS IN ECONOMIC TERMS

Invasions into new environments interfere with indigenous species, hence, interfere with economic activities. Consequently, invasions are of economic concern because of their impact(s) on human wellbeing (Emerton & Howard, 2008). The incremental changes that occur during an invasion is of primary concern in economic terms. Invasions interfere with the functioning of an ecosystem, which yields a flow of economically valuable goods and services (Emerton & Howard, 2008).

Although market policy and institutional failures underlie invasions, these economic activities do not internalise the full costs of invasions to the wider economy (Emerton & Howard, 2008). According to Perrings (2002), biological invasions are seen as the external effects of market transactions. Such that, the market prices do not accurately reflect the full social and economic costs associated with invasions (Perrings *et al.,* 2000; Jensen, 2002). These costs are felt by the wider economy and not by the individual or company who introduced a particular species (Parks & Gowdy, 2013). This means the problem of invasion embodies many characteristics of an economic externality, whether it be a benefit or a cost (Perrings *et al.,* 2000; Farber *et al.,* 2002; Jensen, 2002). An externality cost or benefit may result from direct or indirect activity associated with an IAP (Colautti *et al.,* 2006) and may impact on market or non-market goods and services (Colautti *et al.,* 2006). However, invasions do differ from externalities as conventionally understood. Invasions, once set in motion, are largely self-perpetuating and their impacts often increase over time (Perrings *et al.,* 2000), whereas traditional externalities usually continue only if the source activity is perpetuated (Emerton & Howard, 2008). There are many advantages when understanding invasions as an externality, but it must be noted that many of the financial and economic instruments designed to deal with conventional externalities are not as well-suited to dealing with invasions (Emerton & Howard, 2008).

2.5.3 ECONOMIC VALUATION OF INVASIONS

According to Richardson & Van Wilgen (2004), IAPs not only have adverse environmental impacts, but also threaten the economic productivity of the country. IAPs have significant economic consequences, including damage costs such as biodiversity loss or habitat change and the costs of prevention, control or eradication (Pimentel *et al.,* 2000; Turpie & Heydendrych, 2000; Oreska & Aldridge, 2011; Eviner *et al.*, 2012). It was estimated by Pimentel *et al.* (2000), that in the U.S., the total cost from IAPs species was US$24 billion. Global losses are believed to be on the order of US$1.5 trillion per annum (Pimentel *et al.,* 2002).

According to Hoagland & Jin (2006), conducting an economic valuation of the actual or potential economic damages associated with an IAP consists of estimating in monetary terms the costs associated with alternative management responses. The alternative management responses include preventing the invasion, controlling the spread, eradicating the invader, or doing nothing at all (Hoagland & Jin, 2006). From the results of the economic valuation, an appropriate policy is then suggested. Economic valuations have the potential to be used to support decisions on whether or not to devote scarce financial resources (Born *et al.,* 2005). Hoagland & Jin (2006) highlight the importance of the economic value before and after an invasion, indicating the correct measure of damages associated with the invasion. Although the economic value of the potential invasion is hypothetical and based on certain assumptions, it must realistic and based on sound ecological understanding (Conrad & Clark, 1987).

Various techniques have been used in attempt to estimate the economic effects of IAPs. The techniques ranged from estimating the effect of a single-species at a national scale, to multi-species within a particular region or biome (Turpie, 2004). Many of the studies used benefit: cost ratios (Van Wilgen *et al.*, 2004; Wise *et al.*, 2012), net present values with or without clearing programmes (De Wit *et al.*, 2001; De Lange & Van Wilgen, 2010), rates of return or gross margin analysis (Mugasi *et al.*, 2000).

Economic valuations allow for the breakdown of multi-dimensionality IAP species' impacts by transferring these impacts into monetary units (Tisdell *et al.*, 1990). Several studies have shown that IAPs are strongly correlated with economic factors (Pyšek *et al.*, 2010; Essl *et al.*, 2011; Jeschke & Genovesi, 2011). There is an associated opportunity cost to economies from the foregone benefits of financial resources and labour diverted to the management of IAPs. Impacts on economic activities may be measured by the change (usually a decrease) in net social benefits caused by the introduction of IAPs (Bax *et al.*, 2003). Although arriving at a comprehensive figure for the total costs of IAPs is difficult, there are many studies which indicate that the total costs are substantial (Van Wilgen *et al.*, 2001a; McConnachie *et al.*, 2012; Wise *et al.*, 2012).

Mesquite (*Prosopis*) was introduced into South Africa to provide shade for livestock but, is now having significant impacts. The cost of the invasion was estimated for different scenarios, productive floodplains and upland areas. Wise *et al.* (2012) estimated that the net economic value of mesquite in 2009, covering 1.47 million ha, was US$3.5 - US$15.3 million. The value will become negative within 4-22 years, assuming annual rates of spread of 30% and 15%, respectively. The study found that control efforts should focus on floodplains, as the benefits of control in the floodplains exceeded that of costs, with the opposite was true for the uplands. In terms of control of Mesquite, it was found by Wise *et al.* (2012) to be economically beneficial to contain the spread to avoid the loss of substantial water and pasture benefit. The benefits of doing this, with the cost of clearing taken into account, are between US$56.4 million and US$137.3 million over 30 years for the relatively slow spread-rate scenario, and between US$122 million and US$376 million over 30 years for the relatively rapid spread-rate scenario.

McConnachie *et al.* (2012) conducted a study to evaluate the cost-effectiveness of IAP clearing in the Krom and Kouga river catchments in South Africa. The invasion of IAPs in catchments is influencing the total water resources system. Between 2002 and 2008 'Working for Water' had spent a total of R9.38 million on operational, management and implanting agent levy costs on the Kouga catchment, and R9.89 million on the Krom catchment. Detailed breakdown of the costs can be seen in table 2.1. The Kouga project was far less cost-effective at R70 517 per ha compared to R11 987 for the Krom catchment. And at the clearing rate of the Kouga project, the 'Working for Water' programme would only be able to contain the invasions if they spread at a rate of 0.14% or less annually, compared to 1.84% in the Krom project. Both estimates are well below the realistic annual spread rate of 8.5% (Le Maitre *et al.*, 2002).

Table 2.1. Costs of the Kouga and Krom Working for Water projects between 2002 and 2008 (ZAR millions).

	Kouga	Krom
Operational costs	7.35	7.75
Management costs	1.18	1.24
Implementing agent levy costs	0.85	0.90
Total cost	9.38	9.89

Source: Adapted from McConnachie *et al.* (2012).

The study concluded that, at the current rates of clearing, it would take 54 and 695 years to clear the Krom and Kouga catchments respectively (assuming no further spread). Moreover, if future spread was to be considered in this study, current control efforts would be inadequate, and invasions are likely to continue to spread in the catchments.

Another key issue when dealing with environmental economics is that of valuing the impacts or consequences which an IAP has on ecosystem functions or services (Eviner *et al.*, 2012). Various challenges exist in this regard, preventing researchers from confidently extrapolating those processes as proxies for services (Eviner *et al.*, 2012). Pejchar & Mooney (2009) suggested that one of the main reasons is because little is currently known about the mechanisms by which IAPs affect services. Recent attempts have been made to assess the impact of IAPs on ecosystem functions or services (Farber *et al.*, 2002; Limburg *et al.*, 2002; Howarth & Farber, 2002; Cook *et al.*, 2007; Van Wilgen *et al.*, 2008; Schägner *et al.*, 2013). The challenge seen by researchers is to place an accurate set of values on ecosystem goods and services, comprising of both use and non-use values (Farber *et al.*, 2002). De Lange & Van Wilgen (2010) conducted a study to determine the economic impact of four major functional groupings of invading alien plants, and assessed their impact on water, resources, grazing and biodiversity. They estimated the value of the potential ecosystem services to be R152 billion annually. The study found that R6.5 billion of potential ecosystem services was lost every year due to IAP invasions. When estimating the potential impact if no control had been carried out, they found that it would have amounted to be an additional R41.7 billion loss. The benefit: cost ratios of control ranged from 50:1 for invasive sub-tropical shrubs to 3726:1 for invasive Australian trees.

Invasions have significant economic costs as a result of the impacts on biodiversity. In the southwestern part of South Africa, Turpie & Heydendrych (2000) estimated that the values of harvesting wildflowers, and flowers for recreational use, was reduced from US$9.7 to US$2.3 per ha and from US$8.3 to US$1 per ha, respectively, when areas became densely invaded by IAPs.

IAPs are often associated with significant negative economic consequences (Pimentel, 2002; Perrings *et al.*, 2000). Private and government funds – although limited - are available to help minimise these negative economic consequences. As a result, economic valuations of such projects provide the necessary motivation for the allocation of these scarce funds (Turpie, 2004), with an economic evaluation essential in verifying the viability of a project and thereby attracting investment.

2.6 IMPACT ON AGRICULTURE

While the significant ecosystem damage caused by IAPs has been well documented, the economic consequences of specific IAPs are poorly understood (Van Wilgen *et al.*, 2001a). There is an increasing recognition of the importance of the effect that IAPs have on the agricultural sector (Cullen & Whitten, 1995; Leitch *et al.*, 1996; Ditomaso, 2000; Pimentel *et al.*, 2001; Acquaye *et al.*, 2005; De Neergaard *et al.*, 2005; Eagle *et al.*, 2007; Dube, 2010). IAPs have caused significant economic losses to agriculture production (Mack & D'Antonio, 1998; Busso *et al.*, 2013). Although the impacts of IAPs on agricultural production are significant, relatively few studies have attempted to estimate the economic effects of a specific IAP species on the agricultural sector. Notable exceptions include studies conducted in the U.S. on the Leafy spurge (*Euphorbia esula* L.) (Leistritz *et al.*, 1992; Leitch *et al.*, 1996), various species of Knapweed (*Centaurea diffusa Lam., C. maculosa Lam.,* and *Acroptilon repens* L.) (Hirsch & Leitch, 1996), as well as a study conducted by Eagle *et al.* (2007), who estimated the costs and losses imposed on California ranchers by Yellow star-thistle (*Centaurea solstitialis L*)[6].

IAPs could have substantial impacts on forage quantity and quality, increasing management costs, imposing land use changes, and thereby reducing agricultural production output and profitability (Eagle *et al.,* 2007). In addition, infestations have the potential to reduce recreational land values and cause human health problems (Ditomaso, 2000).

Invasive weeds have had significant impacts on agriculture around the world. In the U.S., approximately 50 000 non-indigenous species have caused environmental damages estimated at US$136 billion per year, with US$6 billion due to weeds in pastures (Pimentel *et al.,* 2000). In the U.S., it was estimated that invasive weeds cause a reduction of 12% in potential crop yields. In economic terms, this equates to approximately a US$33 billion loss in crop production annually (USBC, 1998). Pimentel (1993) estimated that about 73% of the weeds are non-indigenous, and found that the likelihood of crop losses due to invasive weeds would amount to roughly US$27.9 billion. In the United Kingdom (U.K.), weeds have caused a 10% to 32% reduction in crop yields (Spedding, 1985; Oerke *et al.,* 1994). Pimentel *et al.* (2001) suggested that roughly US$3.2 billion in total potential crop production is lost annually due to weed infestations. The Blackberry (*Rubus procerus*) introduced to Australia, caused US$77 million per year of damage to crop production alone (Davis *et al.*, 1995). Furthermore, Davis *et al.* (1995) estimated that weeds caused approximately US$4 billion per year in total damages in cropland and pastures combined (Davis *et al.*, 1995). Since 60% of these weeds are alien, they account for about US$2.4 billion per year in losses to agriculture (Groves, 1991).

The impacts on human activities are associated primarily with rangelands and livestock production (Ditomaso, 2000). This includes interfering with grazing practices and patterns, increasing management costs, lowering yield and quality of forage, poisoning livestock, slowing animal weight gain, reducing the quality of meat, milk, wool, hides etc., imposing land use changes and reducing land value (Frandsen & Boe 1991; Leitch *et al.* 1996; Ditomaso, 2000; Eagle *et al.,* 2007). In Montana, North Dakota, South Dakota, and Wyoming, Leitch *et*

[6] See Duncan *et al.* (2004) for a detailed discussion of the existing literature with regards to invasive plants on rangelands and wildlands in the United States.

al. (1996) estimated that the total direct and secondary annual economic impact of leafy spurge (*Euphorbia esula L.*) on the livestock industry in 1993 exceeded US$129 million. Similarly, Hirsch & Leitch (1996) estimated a US$42 million annual loss to Montana's economy, due to the infestation of three Centaurea species (Knapweed (*Centaurea diffusa, C.*), Spotted Knapweed (*Centaurea maculosa*), and Russian Knapweed (*Acroptilon repens*). Eagle *et al.* (2007) conducted a study to determine the costs and losses imposed on California cattle ranchers by Yellow star-thistle (*Centaurea solstitialis L.*). They investigated the infestation rates, loss of forage quantity and value, and control or eradication efforts. The study found that the total losses caused by Yellow star-thistle cost the State US$7.65 million annually in lost livestock forage with ranchers' out-of-pocket expenditures on control amounting to US$9.45 million per year. Together, these numbers amount to 7% of all revenue from active rangeland in California.

2.7 VALUE OF LIVESTOCK IN SOUTH AFRICA

The meat production chain in South Africa can be divided into the formal and informal sector (Musemwa *et al.*, 2007). The formal sector comprises of commercial farmers and the informal sector includes emerging farmers and small-scale subsistence farmers (Scholtz *et al.*, 2008; Spies & Cloete, 2013). In 1992, the South Africa meat industry was deregulated and new legislation including the Marketing of Agricultural Products Act (Act No. 47 of 1996) was introduced. This legislation allows producers to sell their products to their customers at a mutually agreed price, thus a free marketing system (Van Zyl *et al.*, 2006). This resulted in a surge in the informal sector leading to the sale of livestock through informal channels in South Africa (Soji *et al.*, 2015).

The monetary value of livestock (sales) in the informal sector is less compared to that of the formal sector. This is due to prices being determined on mutually based visual appraisal or live weight (Soji *et al.*, 2015). According to Groenewald & Jooste (2012), the major shortcomings of the informal markets are the seasonality of the markets, poor market information and the quality of the livestock. As a result of these short shortcomings in the informal market, farmers often sell their livestock below market value due to bad timing and a weak bargaining position (Groenewald & Jooste, 2012). Smallholder farmers sell their livestock to speculators and at auctions while subsistence farmers sell their livestock to livestock through private sales to neighbours and relatives (Musemwa *et al.*, 2007). Livestock producers in the informal sector have low transaction costs however, local buyers have low purchasing power and, therefore, offer prices below the actual value. Farmers are often price takers and sell off their animals from a position of low bargaining position as they may be pressed for cash (Soji *et al.,* 2015). However, the intrinsic value of livestock to the informal sector is greater than that of the monetary value. Livestock in the informal sector is considered to have multipurpose roles. Musemwa *et al.* (2007) and Tada *et al.* (2012) reported that income generation is the main reason for livestock keeping in the smallholder farming sector. Musemwa *et al.* (2008) found that 59% of farmers reported income generation as the main reason for keeping cattle while 26% of farmers reported that they keep cattle mainly for family consumption. Consumption, draught power provision, use of livestock in traditional ceremonies and "live banks" for immediate cash needs are some of the other reasons for livestock keeping (Mngomezulu, 2010). The multipurpose use of livestock further encourages the small-scale and emerging farmers to sell animals through the informal sector and thus, avoiding the formal sector (Musemwa *et al*., 2010).

2.8 CONTROL OF INVASIVE ALIEN PLANTS

The control of IAPs encompasses a broad spectrum of fields. The need for sustainable and effective control programmes has been recognised globally in order to counter the impacts of IAPs (Van Wilgen *et al.,* 2001b; Richardson & Kluge, 2008). Control procedures are put in place with the aim of reducing alien plant densities to acceptable levels (Van Wilgen *et al.,* 2001b). Because IAPs have a higher probability of unrestrained growth, which could ultimately lead to environmental and economic damages, control programmes are important to help manage their control (Brunel *et al.*, 2013). One such opinion - that of Henderson (2011) - noted that preventing or tackling the spread of IAPs is most effective and cost-effective in their early stages of establishment.

The selection of a control method is often determined according to the attributes of the specific plant. Other factors need to be considered, such as environmental factors (fire and flooding) and human factors, (financial budgets and levels of skill) (Van Wilgen *et al.,* 2001b; Brunel *et al.,* 2013). Economic impact assessments are used to make informed decisions about the control and management of invasive species (Rempel, 2010). Generally, benefit: cost analyses are used to determine the feasibility of control programmes (Van Wilgen *et al.,* 2004; Wander *et al.,* 2004). Determining the best control practice through the understanding of the effects of intervention is important, as interventions interact with the stable and stochastic factors, which can potentially affect the final outcome (Van Wilgen *et al.,* 2001b). Three different primary methods of control exist in order to combat the spread of IAPs, namely; mechanical, herbicidal and biological (Ditomaso, 2000; Van Wyk & Van Wilgen, 2002). Integrated control is another method, which encompasses a variety (at least two) of the methods mentioned above.

2.8.1 MECHANICAL CONTROL

This method of control comprises of the physical felling or uprooting of plants (Van Wilgen *et al.,* 2001b), with fire often used in combination. Richardson & Kluge (2008) stated that fire was the best available control option for killing seeds in the leaf litter of invasive Australian *Acacia* species in South Africa. However, clearing also stimulates the germination *en masse* of seeds from a large and persistent soil-stored seed bank (Holmes *et al.*, 2008).

Silky hakea (*Hakea sericea*), a shrub introduced to South Africa from Australia, has invaded the fynbos biome. Mechanical control methods have been introduced in an attempt to control the invasion. The shrubs are felled using chainsaws or slashers, and left for 12 to 18 months before burning (Van Wilgen *et al.,* 2001b), with this method of control proving successful in the clearing of large areas of invasions. This mechanical control strategy has also been successfully applied to other invasive Hakea species (*H. drupacea* and *H. gibbosa*), Pines (*Pinus* species) and Guava (*Psidium guajava*) (Van Wilgen *et al.,* 2001b). In saying this, there are certain drawbacks to this control method as physical damage is done to the soil (Van Wilgen *et al.,* 2001b) and this method has proved unsustainable. Mechanical control attempts have been made on the Prickly pear (*Opuntia ficus-indica*), which have been unsuccessful. The IAP is spread widely by the animals that consume the fruits and spread seeds over large distances (Moran *et al.,* 2011), making mechanical control attempts difficult.

Between 1996 and 2008 the South African government spent R435 million on mechanical control of Mesquite (*Prosopis*). Despite this substantial investment, mechanical control efforts were only able to treat about 0.6% of the estimated invaded area each year (Van Wilgen *et al.*, 2012), which is below the spread rate of the species (Shackleton *et al.*, 2015). Mesquite will continue to spread unless a damaging biological control agent such as *Evippe spp* is brought in to curtail the spread (Van Klinken, 2012).

Mechanical control is often labour-intensive and therefore can be expensive to use over extensive, remote or rugged areas. Where trees cannot be utilised, they are not felled but rather controlled *in situ* (Van Wilgen *et al.,* 2001b). The various *in situ* methods include basal bark, ring barking and hand pulling (Republic of South Africa, 2015a).

2.8.2 CHEMICAL CONTROL

Chemical or herbicide control involves the use of registered herbicides to kill a target plant. This method often involves the application of herbicides to cut stumps or to kill seedlings after felling or burning, which ultimately prevents sprouting from occurring (Van Wilgen *et al.,* 2001b). Another chemical control method is foliar spraying, where liquid fertiliser is sprayed directly to their leaves. Unfortunately, the use of herbicides has a potential negative impact on the environment as a result of their toxicity, their residence times (i.e. old vs. recent invasions) and their use on a specific plant. Chemical control is often governed by legislation, which encourages the safe use of the toxic chemicals (Edgin, 2007) and the use of herbicides requires a high level of training (Van Wilgen *et al.,* 2001b).

Black wattle (*Acacia mearnsii*), originally from Australia, has hard-coated seeds and spreads rapidly down rivers. This IAP requires mechanically felling of trees, followed by the application of herbicides to the stumps to prevent sprouting, killing the plant and thus prevents coppicing (McConnachie *et al.,* 2012). A follow-up is needed in the form of spraying a herbicide, to ensure that sprouting is prevented (Van Wilgen *et al.,* 2001b). Cheat grass (*Bromus tectorum)* is an annual grass, which dominates the regions of western North America. During the summer months the plant dries out, fuelling wild fires. Native shrubs and perennial grasses cannot recover, and after a few wildfire cycles a Cheat grass monoculture develops. The most common control method of Cheat grass is herbicidal, with an occasional burning application. Hand-pulling (mechanical control) of Cheat grass is labour-intensive and is only feasible for small infestations (Van Wilgen *et al.,* 2001b).

The success of both mechanical and chemical control is the timely follow-up treatments required to treat both seedlings and coppice re-growth. Moreover, the re-growth is compounded when previous treatments were poorly executed (Holmes *et al.,* 2008).

2.8.3 BIOLOGICAL CONTROL

Biological control is considered to be the most environmentally friendly, cost-effective and self-sustaining control method used to suppress IAPs (Zimmermann *et al.,* 2004). For over 150 years, many countries worldwide have used biological control for the suppression of IAPs, with more than 400 agents of biological control being deployed to attempt to eradicate approximately 280 IAPs worldwide (Julien & Griffiths, 1998). This method of control has been used as a powerful tool for reducing the costs of management of IAPs in South Africa (Van

Wilgen *et al.,* 2004). Marais *et al.* (2004) stated that biological control of IAPs is the only sustainable, effective and inexpensive solution to the most intractable of the IAP problems. The chances of successful biological control are greatly increased if an IAP species is targeted at an early stage of invasion (Olckers, 2004; Henderson, 1999). The method involves the deliberate screening and introduction of host-specific insects or plant pathogens that reduce the invasive plant's ability to invade (Threthowan *et al.,* 2011). Robertson *et al.* (2003) sought to prioritise IAPs based on their (i) potential invasiveness, (ii) spatial characteristics, (iii) potential impacts, and (iv) conflict of interest. The release of biological control agents, which is dominated by the manipulation and deployment of plant feeding insects (Moran *et al.,* 2011), reduces the seed production and/or vigour of IAPs (Higgins *et al.,* 1997). The aim of introducing these agents is to decline population densities, distribution and/or rates of spread of the problem plants, and therefore reduce the costs of other management practices (Zimmermann *et al.,* 2004). Van Wilgen *et al.* (2004) suggested that biological control is used to reduce the effects of ecological release. In other words, to reduce the invasive species population and spread to a situation where the plant or invaded area(s) are returned to a non-invasive status.

Van Wilgen *et al.* (2004) conducted a study to determine the costs and the benefits of biological control of IAPs in South Africa. The study estimated total historical cost (in 2000 ZAR values) for six research programmes. The costs for the implementation of biological control included expenditure on human resources, overheads and running costs. Costs of the various research programmes varied from R17.3 million for Lantana (*Lantana camara*) to R700 000 for Golden wattle (*Acacia pycnantha*). One of the most successful biological control programmes in South Africa was the release of three biological control agents namely, *Neodiplogrammus quadrivittatus* (1984) *Rhyssomatus marginatus* (1984) *Trichapion lativentre* (1970s), on Red sesbania (*Sesbania punicia*) (Hoffman & Moran, 1991; Hoffman & Moran, 1998; Van Wilgen *et al.,* 2004). The release of the control agents had extensive damages on the IAP, reducing the extent of the invasion to a point where no other control methods were needed to reduce the weed to acceptable levels. The research was conducted between 1978 and 1997 at an estimated cost of R3 million (2000 ZAR values). The benefit: cost ratio for Red sesbania was 8:1. Furthermore it was estimated that the benefit: cost of biocontrol between the initiation of research until the estimated date at which weed populations would cover all available habitat was 45:1.

Biological control, however, does have potential drawbacks. The release of biological control agents may not prove specific enough, resulting in the agents attacking non-targeted native species (Van Wilgen *et al.,* 2004). Moreover, the addition of a control agent is considered risky when introducing it into a complex ecosystem (Brunel *et al.,* 2013). Higgins *et al.* (1997), therefore, suggested that agents are species-specific but are not site-specific. The Lepidopteran (*Cactoblastis cactorum*) (Pyralidae) was successfully introduced in Australia, South Africa, Hawaii and the Caribbean Islands to manage the invasive *Opuntia* species, but it was also accidently introduced into Florida where it threatened a native *Opuntia* species. Although successful, the release of the *Cibdela janthina* used to combat Giant bramble (*Rubus alceifolius*), was subjected to susceptible criticism (Le Bourgeois & Della Mussia, 2009). Local farmers feared that the control agent would outcompete bees, jeopardising fruit production. Fortunately, after further studies, it was later deemed to have no impacts on bees and was an efficient control agent. This demonstrates the importance of the selection of a biological control agents, which are carefully studied through formal risk assessment protocols (Brunel *et al.,* 2013). Several years of safety tests are conducted under strict quarantine conditions to ensure that an agent feeds and maintains its populations on only one - or a limited number - of closely-related species of host plants (Zimmermann *et al.,* 2004).

2.8.4 INTEGRATED CONTROL

Integrated weed control usually involves a combination of at least two of the primary control methods, while effective control of IAPs requires the integration of different control options. Various IAPs have been managed by the use of integrated control, such as the Black wattle (*Acacia mearnsii*), Triffid weed (*Chromolaena odorata)* and the Yellow star-thistle (*Centaurea solstitialis*) (Van Wilgen *et al.,* 2001b). The control of the Prickly pear (*Opuntia stricta*) in South Africa, using biological control agents (*Cactoblastis* caterpillar and a cochineal insect) (Hoffmann *et al.,* 1999), has improved the situation substantially. However, an approach of combining herbicide control on scattered populations with the release of biological control agents on to larger infestations shows a more promising method for bringing the weed under control (Lotter & Hoffmann, 1998). The control of the Broad-leaved paperbark (*Melaleuca quinquinervia*) in the U.S. used a combination of all three methods of control. Mechanical control is predominately used in more accessible sites, while in less accessible sites a foliar herbicidal spray may be used. Prescribed burning is also applied to kill seedlings in wetlands (Hammer, 1996). The use of biological control agents, such as the snout weevil (*Oxyops* *vitiosa*), has been applied since 1997, and several other potential insects have been identified (Van Wilgen *et al.,* 2001b).

Van Wilgen *et al.* (2001b) conducted a study using various scenarios of control management options for the invasive alien shrub Port Jacksons Willow (*Acacia saligna*) in South African fynbos. The scenarios included no management control, two approaches of mechanical control (prioritising dense and scattered stands) with two budget levels, and the combination of one mechanical option with biological control that reduces seed production. In the scenario of no management control, the site would be fully invaded after 100 years. The scenario with mechanical control and an annual budget of 20 units still suggests that after 70 and 100 years the site would still be invaded. With an annual budget of 40 units, after 100 years the site would be almost free from the invasive tree. The effect of combining biological control with mechanical clearing with an annual budget of 20 units shows a complete removal. Therefore, annual budget of 20 units would achieve the same result as a budget of 40 units without biological control. This study demonstrated the effectiveness of biological control.

2.8.5 WORKING FOR WATER

The management of IAPs is imperative in South Africa, especially seeing that water is a critically scare resource. ‘Working for Water’ is a government programme that was established in 1995, which co-ordinates the integrated management of IAPs (Van Wilgen *et al.,* 1998), while providing social services and rural employment (Working for Water, 2007). Unlike other national control programmes that focus on prevention and early detection, ‘Working for Water’ spends most of its funds on labour-intensive clearing (Koenig, 2009). It is one of the largest conservation projects in Africa and the world’s most ambitious IAP control programme (Van Wilgen *et al.,* 2004; Koenig, 2009). The objective of the programme is to reduce the density of established, terrestrial IAPs, through various control methods by 22% per annum. ‘Working for Water’ is administrated by the Department of Water Affairs and Forestry and the Department of Environmental Affairs. Furthermore, the programme is aimed at removing IAPs from conservation areas, catchments and river courses. Since the programme has been up and running, they have spent approximately US$3.2 million on alien plant control across the country (Van Wilgen *et al.,* 2012)[7]. However, the extent of invaded areas in South Africa had grown since the inception of 'Working for Water' in 1995.

The programme has been relatively successful, as more than one million hectares of IAPs have been cleared. Currently, the organisation is running over 300 projects across all nine provinces in South Africa, using a variety of control methods to combat the spread of IAPs. Marais *et al.* (2004) assessed the programme's success by comparing the rate of clearing to the rough approximations of invaded area in 1996. The study found that at the prevailing rates of clearing, and depending on the species, it would take between 2 and 83 years to clear the most important species[8].

The 'Working for Water' programme is behind schedule on a national scale in terms of the expectations of clearing IAPs within a reasonable timeframe (Van Wilgen *et al.,* 2012). Furthermore, little is known about the cost-effectiveness of its clearing treatments at a project scale, due to ineffective monitoring and evaluations of their future goals (Van Wilgen *et al.*, 2012; Levendal *et al.*, 2008). Thus, it tends to be difficult when assessing any effectiveness in terms of progress towards the goal of restoring ecosystem health.

Despite 'Working for Water' being one of the largest IAP control programmes in the world (Rouget *et al.,* 2004), there are insufficient resources to reach all invaded areas so decisions need to be made as to where and when to implement control methods (Forsyth *et al.,* 2012). Rouget *et al.* (2004) reported that the initiatives lack objective protocols for prioritising IAPs. Forsyth *et al.* (2012)[9] conducted a study to determine the prioritisation of IAP control projects using a multi-criteria decision model which ranks criteria for prioritising alien plant control operations. Seventeen IAPs were considered and certain criteria were weighted according to the IAPs. The study found that there are many high priority catchments which are not receiving any funding, and low priority catchments which are receiving substantial allocation. Forsyth *et al.* (2012) concluded that priorities need to be realigned and that funds should be re-allocated in proportion to the agreed priorities.

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[7] The budget for research on emerging biological control of IAPS has been increased considerably. Another budget has also been allocated to develop human capacity in research into biological control of IAPs (Moran *et al.,* 2005).
[8] These results were under the (albeit unrealistic) assumption that no further spread would take place during this time.
[9] A similar study was conducted by Kumschick *et al.* (2012), which sought to develop a conceptual framework for prioritisation of IAPs for management according to their impact.

Hosking & du Preez (2003) conducted a benefit: cost analysis of the feasibility of the 'Working for Water' programme in the eastern and southern regions of the Cape, in South Africa. The study found that catchment management on all the sites[10] carried out by the programme was inefficient, but it did conclude that at lower discount rates - for instance 5% - the Kouga project is efficient. Furthermore, if 30% cost savings could be achieved and a discount rate of 5% be employed, both the projects on the Kouga and Tsitsikamma sites will become efficient.

2.9 CONCLUSION

Ultimately, a large number of IAPs have invaded ecosystems across the globe. Many of these IAPs are well-established and have negative ecological and economic effects. However, a conflict of interest does arise as many of the IAPs also provide various benefits. South Africa is particularly vulnerable to IAP species and it is believed that South Africa has some of the biggest problems with plant invaders in the world. One such problem is that of the effect which IAPs have on the economy, but more specifically, the threat that IAPs have on the agricultural sector. However, there are various control methods to combat this, which attempt to hinder the process of invasion.

[10] Catchment sites included: Tsitsikamma, Kouga, Port Elizabeth Driftsands, Albany, Kat River and Pott River.

CHAPTER 3

BLACK LOCUST (*Robinia pseudoacacia* L.)

3.1 INTRODUCTION

Black locust (*Robinia pseudoacacia* L.) is a deciduous tree that belongs to the Fabaceae (Legume) family, subfamily Faboideae (Gleason & Cronquist, 1991; Sabo, 2000; Cierjacks *et al.,* 2013). The tree is native to south-eastern North America and was introduced into Europe in the early 17th century as an ornamental (Wojciechowicz-Żytko & Jankowska, 2005; Boring & Swank, 1984). *Robinia* (Fabaceae) contains four species from North and Central America, all of which are considered as weeds worldwide (Sheppard *et al.,* 2006). *Robinia pseudoacacia* is a pioneer tree, which has been widely planted and has become naturalised in 35 countries, and is abundant in temperate North America, Europe, Australia and Southern Africa (Sheppard *et al.,* 2006; Kleinbauer *et al.,* 2010; Li *et al.,* 2014). In the last 300 years, the tree has spread widely all over the world through transplantation and cultivation (Li *et al.,* 2014). *Robinia pseudoacacia* is known to colonise a wide array of different habitats in its secondary range (Cierjacks *et al.,* 2013). However, throughout central and western Europe, the IAP invades nutrient-poor dry and semi-dry environments. These environments comprise of some of the most species-rich and most endangered habitat types of these regions (Holzner, 1986; Fischer & Stöcklin, 1997).

Robinia pseudoacacia, a deciduous legume, has been recognised as a problematic global invader (Kurokochi *et al.,* 2010). According to Sabo (2000), it possesses most of the characteristics associated with "weediness". It is able to colonise itself over a wide range of environmental conditions as well as produce a large quantity of seeds that are easily dispersed (Bazzaz, 1986). In Europe, it has been considered as one of the most problematic IAPs (Kowarik, 2003; Daisie, 2008). On the other hand, it is a species of high ecological and economic value (Li *et al.,* 2014).

The invasive tree is a threat to existing ecosystems as it spreads rapidly from suckering roots and seeds, creating monocultures that displace native species. It reduces the forest canopy composition diversity, as well as preventing shade-tolerant native species the ability to regain dominance (Pacyniak, 1981; Kowarik, 1990). The invasive tree stands along watercourses, restricting access to water for domestic and wild animals (Henderson, 2001). It also causes the narrowing of the river width (Tanaka & Yagisawa, 2009), resulting in river management difficulties. The seeds, leaves, and bark of *R. pseudoacacia* are toxic to both humans and animals (Jepson & Hickman, 1993; Sabo, 2000; Henderson 2001; Vanschandevijl *et al.*, 2010). *R. pseudoacacia's* chemical composition includes not only primary plant metabolites such as biological nitrogen, but also secondary plant metabolites such as phenolics (Cheeke, 1998). The IAP contains substances that are potentially toxic to livestock (Cheeke, 1998).

In addition, the large, fragrant blossoms of the tree compete with native plants for pollinating bees (Farrar, 1995; PCA, 2005). Henderson (pers comm, 2015), the IAP specialist at the South African National Biodiversity Institute (SANBI), stated that *R. pseudoacacia* is ranked as one of the most prominent invaders, especially in the grassland and savannah biomes. This is due to its ability to spread from suckers, its high seed production and its ability to invade pristine habitats.

This invasive tree species is well adapted for growth in a wide variety of ecological conditions (Rédei *et al.*, 2014). Furthermore, due to its fast growing, excellent coppicing and drought-tolerant characteristics, *R. pseudoacacia* has been planted throughout the world from temperate to subtropical areas (Mantovani *et al.*, 2014; Rédei *et al.*, 2014).

The following chapter addresses the life history of *R. pseudoacacia* (3.2), the impacts of *R. pseudoacacia* (3.3), methods of control (3.4), conflict of interest with regard to the IAP (3.5), *R. pseudoacacia* in South Africa (3.6) and a conclusion which follows in section (3.7).

3.2 LIFE HISTORY

Robinia pseudoacacia is a deciduous tree that belongs to the Fabaceae (legume) family and the *Caesalpinioideae* subfamily (Sabo, 2000). Mature trees have the ability to reach heights of approximately 30m with canopy diameters of up to 9m (Huntley, 1990; Sabo, 2000). The tree produces large pods with roughly four to eight seeds per pod. Generally, the best seed crops occur when the tree is between 15 and 40 years old, but seeding may occur as young as 6 years old. It typically produces abundant amounts of wind-carried seeds, but a thick seed coat lowers successful germination (Converse, 1984). Despite this, the invasive tree has the ability to reproduce vegetatively, particularly through root suckering and stump sprouting, thus enabling rapid proliferation (Zimmerman, 1984; Sheppard *et al.*, 2006).

Figure 3.1. Line drawing of *Robinia pseudoacacia* leaves, seed pods, flowers and the tree.
Source: Roux (1995).

Once imbedded in an area, *R. pseudoacacia* seedlings begin competing for light, water and space (Hanover, 1990). Initial seedling densities have been visually observed at up to 20/m² or by extrapolation, 200 000/ha. These densities are significantly lower in stands of around 6 years, in which densities are estimated at around 4 - 5/m² or 40 - 50 000/ha (Coulson, 2015).

3.3 IMPACTS OF *ROBINIA PSEUDOACACIA*

Robinia pseudoacacia has shown a high level of silvicultural performance and is well adapted for growth in a wide variety of soils (degraded)[11] and environmental conditions (Gilman & Watson, 1994; González-García *et al.*, 2011). Globally, negative impacts of *R. pseudoacacia* have been recorded. Wieseler (2005) stated that outside its indigenous range - the U.S. - the tree poses a significant threat to native vegetation. Native North American prairie and savannah ecosystems have been greatly reduced in size and are now represented by small fragile fragments as a result of the spread of *R. pseudoacacia* (Sabo, 2000). Additionally, in Europe, the invasive tree has been recorded to be displacing indigenous vegetation, thereby hindering the return of the system to the desired state (Sabo, 2000). It has one of the largest distributions in Europe of any introduced plant (Cierjacks *et al.*, 2013). In France and Italy, the invasive tree grows rapidly along river banks displacing natural vegetation and impacting on

[11] Mainly due to the fact that it has nitrogen-fixing bacteria on its root system, which allows it to grow on poor soils (González-García *et al.*, 2011).

water flow. In Australia, it is regarded as an environmental weed in Victoria, the Australian Capital Territory, New South Wales, South Australia and Western Australia (DEEDI, 2014). Because of the invasive trees' life history characteristics (e.g., fast growth rate, clonal spread), it has become a successful colonising species in its native habitat and is able to spread rapidly from agricultural to natural systems where it has become a pest (Rice *et al.,* 2004). According to Henderson (pers comm, 2015), "*R. pseudoacacia* is ranked alongside some of our invasive acacia species".

The invasive tree is an aggressive invader, negatively impacting on indigenous flora and fauna on several fronts including dispersal, submission, pollination, soil chemical changes, livestock as well as having various socio-ecological implications (Coulson, 2015).

3.3.1 DISPERSAL

The tree utilises two methods of dispersal: i) seed and ii) suckering (Kowarik, 1996; Coulsen, 2015). Therefore, *R. pseudoacacia* is able to regenerate both sexually through seeds and asexually through adventitious buds on stumps and roots (Boring & Swank 1984; Iwai, 1986; Gyokusen *et al.* 1991). Transportation into completely new environments (primary succession) takes place via seed dispersal, primarily via wind dispersal, as well as by birds and other wildlife (Gilman & Watson, 1994; Coulson, 2015). Once seedlings have established, they begin growing rapidly and build huge stores of organic sugars in their root systems, which is later used for pioneering new ground. Subsequent range increases or secondary succession occurs as a combination of seed dispersal, basal regrowth (in the event of prior cutting) and suckering. The result at all sites is a mono-specific stand of non-native vegetation (Coulson, 2015).

3.3.2 SUBMISSION

Rapid seedling maturity results in a dense stand of *R. pseudoacacia* in which growth media (light, nutrients and soil moisture) become limited (Coulson, 2015). All indigenous vegetation under this canopy gradually dies back and is replaced by a dense leaf-litter, thus preventing any further establishment of indigenous seedlings (Sabo, 2000). Embedded within this litter is the seed of the tree. This acts as a security measure, should any light or space be made available from within the stand (Coulson, 2015).

3.3.3 POLLINATION

In spring, copious quantities of seed and flowers are produced. The tree's large fragrant blossoms compete with indigenous plants, as well as cash crops, such as fruit trees for essential pollinating bees (PCA, 2005). The nectar within these flowers is so abundant that pollinators preferentially pollinate *R. pseudoacacia* over indigenous wildflowers (Farrar, 1995; PCA, 2005; Başnou, 2006), thus, resulting in a gradual decline in other indigenous plant populations (Coulson, 2015).

3.3.4 SOIL CHEMICAL PROPERTIES

As a legume, excess nitrates are fixed into the inherently nutrient-deficient soils, thereby altering the soil chemical composition. Indigenous plants that have adapted to these nutrient-poor soils simply cannot adapt to the higher-than-average nutrient content of the soil (Coulson, 2015). Due to its nitrogen fixing ability, it is capable of colonising low-nutrient substrates where few other tree species can thrive (Rice *et al.,* 2004). In Germany, Veste & Kriebitzsch (2013) estimated that *R. pseudoacacia* had an annual nitrogen fixation of 47.9 - 84.9 kg N ha^{-1} yr^{-1} on reclaimed post-mining land. Optimum conditions include sandy/loamy, well-drained, aerated soils in humid climates and open, sunny locations (DNR, 2015; Sabo, 2000).

3.3.5 LIVESTOCK

Robinia pseudoacacia has been reported as being toxic to livestock. Toxic plant parts include the roots, young shoots, seeds, twigs, leaves and bark (Stephens, 1973; Cooper & Johnson, 1984; Cheeke, 1998). The bark and root of *R. pseudoacacia* contain secondary metabolite robin, (16,000 parts per million (ppm)) (Cheeke, 1998), which has been reported to be the most toxic to livestock. Robin, an extremely potent phytotoxin, is considered a glycoprotein (a lectin) that agglutinates red blood cells (*phytohemagglutinins*) (Cooper & Johnson, 1984). The inner bark is reported to contain amygdalin and urease (Duke, 2000). Furthermore, the bark of the tree also contains a glucoside robinitin (30,000 ppm), and tannins (33,000 to 70,000 ppm). The presence of high tannin levels in forage has been shown to negatively affect livestock, leading to a decrease in animal productivity. Symptoms include digestibility problems, voluntary intake, and biological nitrogen retention (Van Soest, 1982; Kumar & Vaithiyanathan, 1990; Silanikove *et al.*, 1996).

Cooper & Johnson (1984) reported that experimental feeding of the bark to horses (*Equus caballus*) produced toxicity when ingested as an aqueous extract at 0.1% of body weight and as powdered bark at 0.04% of body weight (Kingsbury, 1964). Horses that ingested the leaves, sprouts and bark showed clinical signs of toxicity as soon as one hour after consumption and required medical attention. The ingestion of the toxic plant components can be fatal to livestock (Jepson & Hickman, 1993) as toxicity symptoms include anorexia, weakness, lassitude, colic, depression, posterior paralysis, abdominal pain, nausea, diarrhoea and abnormalities in heart rate and/or rhythm (Kingsbury, 1964; Cooper & Johnson, 1984; Cheeke & Shull, 1985).

According to Hansen (1924) and Kingsbury (1964), experiments indicate that cattle (*Bos taurus*) are somewhat less sensitive to the toxins when compared that of horses. However, death to cattle due to indigestion of the leaves, sprouts and/or bark has been reported (Hansen, 1924; Kingsbury, 1964). Toxicity symptoms for cattle include anorexia, weakness, posterior paralysis, nausea, coldness of the extremities, and dilation of the pupils (Hansen, 1924; Kingsbury, 1964).

It was reported by Kingsbury (1964) that the toxic plant components of *R. pseudoacacia* do not necessarily affect sheep (*Ovis aries*) and goats (*Capra hircus hircus*) in the same way they affect horses and cattle. However, ingested seedpods, which have a trace of toxic agents, have caused minor illness in sheep (Kingsbury, 1964). These agents included the carbohydrates sucrose and raffinose, and the non-nutrient amino acid canavanine (Duke, 2000; Brown, 1998). Researchers reported that leaves, sprouts and/or bark are used as an effective goat feed (Papachristou, 1999; Papachristou *et al.*, 1999). Furthermore, researchers reported high preference of *R. pseudoacacia* plant components as browse for goats (Addlestone *et al.*, 1999; Lambert *et al.*, 1989; Papachristou & Papanastasis, 1994).

3.3.6 SOCIO-ECOLOGICAL IMPLICATIONS

According to Coulsen (2015), there are possible adverse effects related to *R. pseudoacacia* growth. These include (i) Native fauna and flora displacement - loss of biodiversity and subsequent loss of grassveld ecological stability, in particular, resource provisioning; (ii) Loss of arable land for grazing, planting, etc., and of parkland for recreational, cultural, spiritual and sporting event purposes; (iii) Uptake of moisture from streams, resulting in water shortages; (iv) Dense woody biomass causing fire-risks in the dry season, especially in grassland biomes;

(v) Eradication schemes are hindered by *R. pseudoacacia's* stand density (as contracted teams must navigate a safe path through long-sharp thorns on uneven terrain).

3.4 CONTROL METHODS OF *ROBINIA PSEUDOACACIA*

Due to the negative impacts of *R. pseudoacacia*, the management of this invasive tree has become an important task (Sabo, 2000). Sabo (2000:2) stated "*R. pseudoacacia* invasion needs to be controlled because it has the ability to vary its growth patterns, thrive in many regions, and grow at aggressive rates". However, controlling the invasive tree has proven difficult due to its rapid growth and clonal spread (Brown *et al.,* 2001; Akamatsu *et al.,* 2014; Hildegard, pers comm, 2015). Currently, no techniques are available that provide effective control of *R. pseudoacacia* invasions (Cierjacks *et al.,* 2013).

The most common mechanical control method is cutting. However, cutting alone is effective only if repeated several times per year for many years (Edgin, 2007). This is due to root suckering and stump sprouting (Zimmerman, 1984), which makes mechanical control almost impossible. In the U.S., the Wisconsin Department of Natural Resources (DNR) (2015) used serval mechanical control methods to eradicate *R. pseudoacacia.* Bulldozing the surface was deemed the most effective option, but suckers and sprouts still remained and high rates of soil erosion were present (Converse, 1984; DeLoach, 1997). Other mechanical control attempts were made, such as girdling and cutting near the base of the tree only to find that killing the main stem is often followed by the formation of suckers from the tree base and clonal spread (Solecki 1997; Akamatsu *et al.,* 2014). Mowing and burning can temporarily reduce above ground biomass, but will not inhibit the tree's ability to spread vegetatively (Edgin, 2007). Moreover, it was found that mowing of small seedlings would promote seed germination (Sabo, 2000). In dense stands in South Africa, especially in the highland areas, the tree has proven difficult to control mechanically due to access to these stands.

Due to the unsuccessfulness of using mechanical control methods alone, a common and relatively successful method of *R. pseudoacacia* control involves both mechanical and chemical treatment (Edgin, 2007). This method of girdling and the application of herbicides on the stump is feasible since it can be target selective. Sabo (2000) suggested basal bark application of herbicides has proven to be the most successful, but this method, much like the mechanical control method, is costly and timely.

The controlling of *R. pseudoacacia* using only chemical techniques is more common as it is less labour intensiveness. Chemical application is most feasible when the invasive tree has an extensive root system over a broad area. This technique is not recommended for high quality natural areas, as certain herbicides could cause potential damage to non-targeted neighbouring vegetation and wildlife (DNR, 2015; Edgin, 2007). A commonly used herbicide, triclopyr, sometimes used for controlling *R. pseudoacacia* in the U.S., releases volatile organic compounds (Sabo, 2000). However, the run-off from this herbicide can harm non-targeted species. In Japanese river management programmes, the application of glyphosate based herbicides to stumps completely inhibited stump sprouting but not root suckering or seedling germination. Additionally, they showed that there was leaching of the herbicide into the soils over the short term (Akamatsu *et al.,* 2014). Moreover, they suspected toxic effects from additives such as surfactants (e.g. *Polyethoxylated tallowamine*) as several studies have reported that such herbicide formulations were lethal to amphibians and aquatic invertebrates (Akamatsu *et al.*, 2014). In addition to harming non-target organisms, inefficient spraying can damage water quality. Furthermore, chemical treatment is often expensive for areas with low economic return (DeLoach, 1997).

In California, a herbicidal solution (made up of 2% solution of Ortho Brush-B-Gon (triclopyr), 7.5% Dexol Vitamin B-1 fertilizer and 0.5% dish soap) was used as an attempt to control the spread of *R. pseudoacacia* (Weitzenberg *et al.,* 1997). The method proved effective only in controlling small saplings less than 3-4 m in height and was ineffective in killing the large mature trees and clonal root systems (Weitzenberg *et al.,* 1997).

Currently in South Africa, the only moderately successful control has been achieved through a combination of tailored chemicals combined with mechanical control. Contractors regard clearing methods as slow, owing to the spiny nature of the plant. Infestation density, availability of suitable equipment and steep slopes are also factors which 'Working on Fire' and 'Working for Water' teams face (Coulson, pers comm, 2015). Methods of controlling the impacts of *R. pseudoacacia* in other countries could be considered in South Africa, but custom-made methods specifically designed for South African environments and conditions would be preferable.

In Europe, *R. pseudoacacia* is regarded as one of the top twenty potential weed targets for classical biological control from an ecological and socioeconomic perspective (Sheppard *et al.,* 2006). According to Sheppard *et al.* (2006), three North American biological control agents have unintentionally been established on *R. pseudoacacia* in Europe, namely; *Phyllonorycter robiniella* (Clem.), *Obolodiplosis robiniae* (Haldeman) and the *Megacyllene robiniae* (Forster locust borer) (Cerambycidae). *P. robiniella* causes premature leaf dropping that negatively influences the tree's appearance and as such, has itself been the target of a biological control programme in Italy (Wojciechowicz-Żytko & Jankowska, 2005). Relatively high infestations of *O. robiniae* also cause leaf fall but with rapid regrowth (Duso *et al.,* 2005). *M. robiniae* potentially is the most damaging and promising agent as a control option for *R. pseudoacacia*. It tunnels in the trunk and limbs of the tree, resulting in broken and dead limbs, weakened trees, excessive sprout production, and even death of the tree (Galford, 1984). *M. robiniae* also serves as entry points for the fungus *Phellinus rimosus* (Berk.) Pila´t (syn. Fomes rimosus (Berk.) Cooke), which causes extensive wood decay and root rot (Hoffard, 1992).

Based on biological information and various studies (Galford, 1984; Hoffard, 1992), the use of biological control agents offers a promising solution. However, the use of biological control for eradication purposes has not intentionally been implemented (Bossard *et al.,* 2000). The potential biological control agents discussed above are abundant and specific in their native range. Furthermore, the potential biological control agents are highly damaging to *R. pseudoacacia*, particularly *M. robiniae* (Keresztesi, 1980; Sheppard *et al.,* 2006).

3.5 CONFLICT OF INTEREST

Despite *R. pseudoacacia*'s invasive characteristics, it is still considered by many to be a useful species (Kurokochi *et al.,* 2010). Globally, there are numerous reasons for the introduction of the tree. In Japan it has been planted extensively to help with erosion control and forestation in upper river basins (Ecological Society of Japan, 2002; Rédei *et al.,* 2011) and is used as a source of timber for furniture, due to its fast growing strong rot resistant wood (Kurokochi *et al.,* 2010). This invasive tree is also an excellent nectar source[12] for honey bees (Keresztes, 1980; Nakamura, 2009), is used for reforestation, is deemed as a crop nurse species (Dzwonko & Loster, 1997; Torbert *et al.,* 1995), is used for mine soil reclamation (Zeleznik & Skousen, 1996), and serves as a shade-giving and greening tree in urban areas (Boring & Swank, 1984). As a horticultural species, *R. pseudoacacia* is valued for its showy fragrant flowers and have

[12] *Robinia pseudoacacia* is utilised as a honey plant and provides half of the national production of honey in Japan (Morimoto *et al.,* 2010).

been planted as roadside and ornamental trees (OPLIN, 2001; Ecological Society of Japan, 2002).

According to Gasol *et al.* (2010), when cultivated, the tree is considered as a potential energy crop in a low-input production regime. This regime leads to higher biomass yields and lower environmental impacts. Furthermore, the use of *R. pseudoacacia* provides indirect benefits as it does not compete with food and feed crops and its water requirements are relatively lower than other crops (Gasol *et al.,* 2010). In Hungary, it has been considered as one of the most suitable tree species for establishing energy plantations and for transforming existing traditional forests into energy forests (Halupa & Rédei, 1992; Rédei, 2003).

Due to the tree's fast growth and ability to grow in a variety of soils, *R. pseudoacacia* is used for revegetating landfills (Kim & Lee, 2005). Kim & Lee (2005) conducted a study to determine the potential of different tree species for restoring unsanitary landfills in South Korea. The study found that *R. pseudoacacia* was proven to be the dominant tree species, which formed canopy layers in the waste landfills. The basal area of *R. pseudoacacia* was 1.51 m^2/ha, and this species had the highest number of saplings among all tree species. Its diameter ranged from 3.71m to 11.29m. Kim & Lee (2005) found that as the patch diameter increased, so did the number of regenerated saplings. *Robinia pseudoacacia* invaded the dry habitat at a high growth rate via bud banks and spread clonally in a concentric pattern across the landfills. Moreover, the invasive tree was found to fix nitrogen symbiotically. Kim & Lee (2005) concluded that a mix of ubiquitous adaptable species - including *R. pseudoacacia* - has the ability to enhance the landscape through synergistic effects.

3.6 *ROBINIA PSEUDOACACIA* IN SOUTH AFRICA

In South Africa, *R. pseudoacacia* is a declared invader (Category 1b-Invasive species that requires control by means of an invasive species management programme, National Environmental Management Biodiversity Act (NEMBA) regulations published 1 August 2014) and has invaded all nine of South African provinces. The greatest damage is in the Eastern Cape, Kwa Zulu-Natal, Free State and Gauteng (Henderson, pers comm, 2015). It was reported by Price (pers comm, 2015) that "*R. pseudoacacia* is a serious invader and is possibly a crisis waiting to happen".

At the end of 2000, the invasive tree was recorded in 110 quarter degree squares (QDS) and was abundant in 14 QDS (Henderson, 2007). As of 2015, the invasive tree is recorded in 159

QDS and is abundant in 38 QDS (Henderson, pers comm, 2015). Figure 3.2 illustrates the distribution map of the tree in South Africa as of March 2015, showing that *R. pseudoacacia* has invaded all nine provinces in South Africa, being particularly abundant in Gauteng, eastern Free State and northern Eastern Cape provinces.

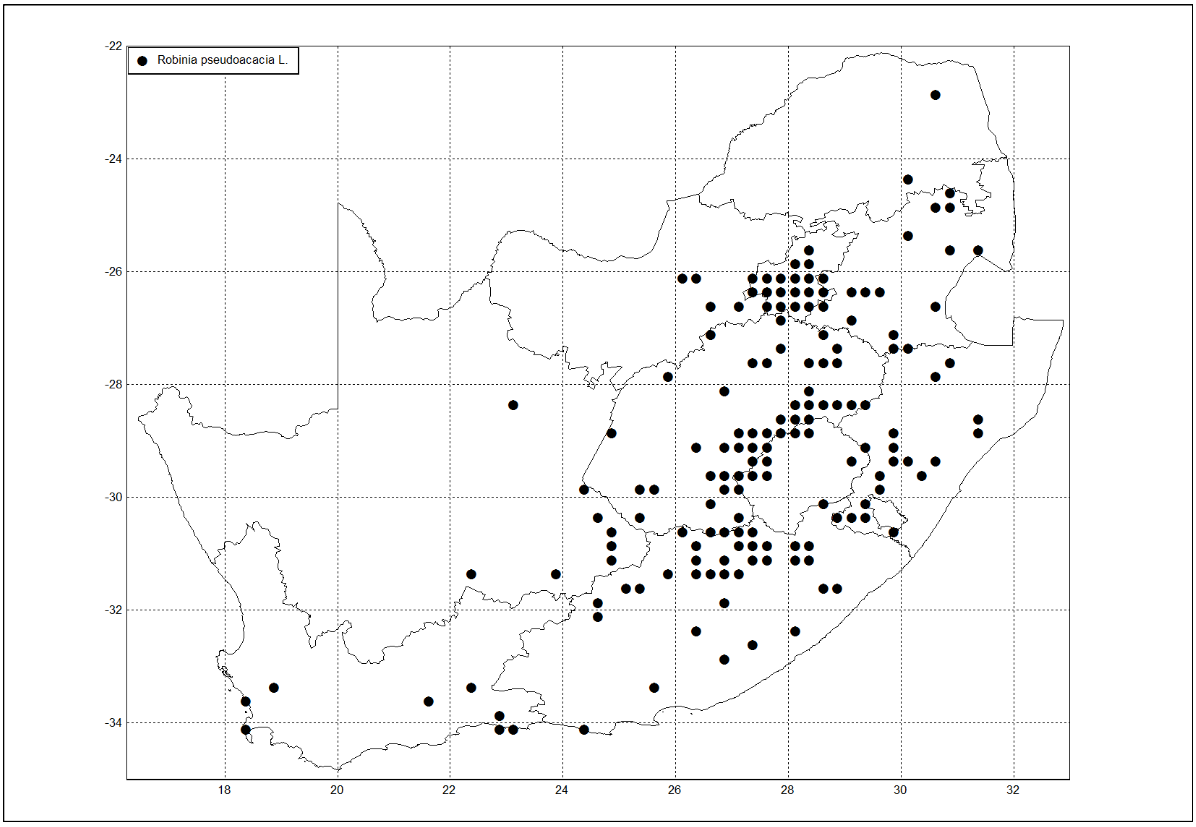

Figure 3.2. Distribution map of *Robinia pseudoacacia* in South Africa as of March 2015.
Source: SAPIA (2015).

3.7 CONCLUSION

Robinia pseudoacacia is ranked as one of the most prominent invaders (Henderson, pers comm, 2015). Although native to eastern United States, it has been widely planted and become naturalised globally. The invasive tree is a threat to existing ecosystems as it spreads rapidly from suckering roots and seeds creating monocultures that displace native species, reduces the forest canopy composition diversity, as well as preventing shade tolerant native species from regaining dominance (Pacyniak, 1981; Kowarik, 1990; Henderson, pers comm, 2015). It is also a prolific water user, capable of invading pristine environments, and its seeds, leaves, and bark are toxic to both humans and animals (Sabo, 2000). The bark and root of the tree contain secondary metabolite robin, which have been reported to be the most toxic to livestock, especially to horses and cattle (Kingsbury, 1964; Cooper & Johnson, 1984; Cheeke, 1998). Conversely, it is still considered by many to be a useful species as it is has been planted extensively to help with erosion control (Rédei *et al.*, 2011), is a useful source of timber for furniture (Kurokochi *et al.*, 2010), and is used for mine soil reclamation (Zeleznik & Skousen, 1996). The invasive tree is regarded as one of the toughest invaders to control as it suckers profusely and readily produces copious numbers of seeds. To date, there are no confirmed effective control measures in place to combat the spread of *R. pseudoacacia* (Cierjacks *et al.*, 2013), although potential biological control agents have been identified (Keresztesi, 1980; Hoffard, 1992; Sheppard *et al.*, 2006). In South Africa, *R. pseudoacacia* is a declared invader (Category 1b) and has invaded all nine of South African provinces. The distribution of *R. pseudoacacia* in South Africa has increased exponentially over the past 30 years. According to Van Wilgen (pers comm, 2015), "*R. pseudoacacia* is a potential big threat in South Africa".

CHAPTER 4
METHODOLOGICAL APPROACH, STUDY DESIGN AND RESEARCH METHODS

4.1 INTRODUCTION

This chapter describes the aim of the study (4.2), the study design (4.3), the research tools and the collection and analysis of data with respect to their application and appropriateness (4.4-4.8). The chapter further gives a discussion on the methodological and analytical procedures (4.9) and the MAXENT results (4.10). Lastly, this chapter address the environmental layers applied to the model (4.11), the process of estimating the economic implications (4.12) and the ethical considerations of the study (4.13).

4.2 AIM OF STUDY

While the significant ecosystem damage caused by IAPs has been well documented, the economic consequences of specific IAPs are poorly understood (Van Wilgen *et al.*, 2001a). There is an increasing recognition of the effect that IAPs have on the agricultural sector (Cullen & Whitten, 1995; Leitch *et al.*, 1996; Pimentel *et al.*, 2001; Acquaye *et al.*, 2005; De Neergaard *et al.*, 2005; Eagle *et al.*, 2007; Dube, 2010). Various studies have attempted to determine the effects of a variety of IAPs on agriculture around the world (Cullen & Whitten, 1995; Leitch *et al.*, 1996; Eagle *et al.*, 2007; Dube, 2010). In South Africa, invasive terrestrial trees are of a particular concern to the agricultural sector, and therefore to the economic development of southern Africa as a whole (Gorgens & Van Wilgen, 2004). The agricultural sector's significance in South Africa is largely because of its potential to create jobs and its substantial contribution to food security, in addition to being a key focus of the New Growth Path (Republic of South Africa, 2013). Due to its invasive characteristics, *R. pseudoacacia* is one such invasive terrestrial tree species. The potential economic impacts of *R. pseudoacacia* on agricultural production stem from the tree's ability to reduce the carrying capacity of livestock (Bangsund *et al.*, 1999). It was assumed that the spread of the tree into grazing lands would decrease grazing output. *Robinia pseudoacacia* reduces the carrying capacity of livestock in two ways. Firstly, infestations restrict access to grazing lands through encroachment and so decreases the amount of available forage for livestock and interferes with grazing patterns. Secondly, the seeds, leaves and bark of the tree are toxic to livestock (Cooper & Johnson, 1984;

Jepson & Hickman, 1993; Sabo, 2000). The reduction in carrying capacity of livestock disrupts agricultural production, output and profitability, resulting in a decrease in income. The impact of *R. pseudoacacia* has not been investigated in South Africa.

The primary aim of this study therefore is to determine the potential direct economic implications of the uncontrolled spread of *R. pseudoacacia* on livestock grazing in South Africa. For this, it is necessary to determine the potential spread of *R. pseudoacacia* in South Africa and the potential economic implications of this spread into grazing lands.

4.3 STUDY DESIGN

The research method used a mixed methods approach, which included the combined collection and analysis of both quantitative and qualitative data techniques (Creswell & Clark, 2007). Quantitative data was primarily used in order to support and inform the research objectives, defining the research within a positivist research paradigm. The mixed method approach allows for the integration or combination of methods through triangulation (Steckler *et al.*, 1992).

Quantitative methods included determining monetary figures for a simplistic approach. Data was presented in the form of tables and graphic representations, which provided measures of interpreting and comparing data. Qualitative methods included structured and open-ended questions from interviews and questionnaires in addition to other relevant literature.

4.4 DATA COLLECTION AND SOURCES

This research project used a complementary approach of both primary and secondary data in order to satisfy data requirements.

4.4.1 SECONDARY DATA COLLECTION

Secondary data was obtained from a number of sources. Online databases were accessed and data was downloaded. Four online databases were used in this study; 1) Global Biodiversity Information Facility (GBIF) (GBIF, 2015); 2) WorldClim database (Hijmans *et al.,* 2005); 3) Southern African Plant Invaders Atlas (SAPIA) (SAPIA, 2015) and; 4) Biodiversity GIS (BGIS) (SANBI, 2015). In order to determine the potential distribution of *R. pseudoacacia,* GBIF, SAPIA and the WorldClim databases were used in a maximum-entropy predictive habitat model (MAXENT) (Phillips *et al.,* 2010). The data provided by SAPIA, in quarter degree squares (QDS), presented a current yet incomplete distribution of *R. pseudoacacia* in

South Africa. BGIS data was used to overlay various agricultural zones and vegetation types over the distributions of *R. pseudoacacia* in South Africa.

According to Stewart & Kamins (1993), secondary data is said to enhance the efficiency of primary research efforts through access to well-researched records containing higher quality data more representative of areas or populations. All secondary data collected was checked for inconsistencies and redundancy and then corrected to enhance the accuracy of the data.

4.4.2 PRIMARY DATA COLLECTION

Primary data was used as supplementary research to acquire detailed information about *R. pseudoacacia*, within a specific area. Primary data collection took place in the form of interviews and questionnaires (See appendix 1) (approved by the Rhodes University ethics committee) with interested and affected parties within the Clarens region, Free State Province, South Africa.

4.5 QUESTIONNAIRE DESIGN

The questionnaire was divided into five sections. The first section included questions regarding a respondent's personal particulars. This was done for administrative purposes in case follow-up for clarification was needed. The second section pertained to plant identification. The questionnaire had various pictures of *R. pseudoacacia,* namely pictures of the tree, leaves, bark, flowers and seedpods. This section was necessary in order to confirm whether the respondents could identify *R. pseudoacacia*. The third section pertained to the interested and affected parties' perceptions of the invasive tree, providing detail pertaining to the distribution and impacts of *R. pseudoacacia* in the area and allowing for the establishment of the level of knowledge of the respondents. In the fourth section, interested and affected parties were asked questions relating to the control measures and financial implications of *R. pseudoacacia*. The final section allowed for further comments.

4.6 SAMPLE AND REPRESENTATION

This data collection was undertaken between 30 September and 6 October 2015 in the Clarens region. Only interested and affected parties took part in this study. Although this sample may be considered bias, parties who have not come across *R. pseudoacacia* before would not be aware of the affects and thus, would potentially skew the results. The interested and affected parties who took part in the study were selected based on their knowledge and/or experience of dealing with *R. pseudoacacia*. Ten farmers, 2 land owners, the Clarens Village Conservancy and the Clarens Golf Course were able to complete the questionnaires and/or provide some information relevant to the study. The majority of the farmers were livestock farmers, with the rest farming citrus or crops. The landowners were interested parties who had attempted to eradicate *R. pseudoacacia* from their smallholdings. The Clarens Village Conservancy is a group of individuals, homeowners and businesses concerned about the state of their environment and who wish to enhance and monitor it. The Clarens Village Conservancy over the years has attempted to eradicate *R. pseudoacacia* within the Clarens region. Lastly, the Clarens Golf Course is a local business which has found dealing with *R. pseudoacacia* problematic.

4.7 BIO-ECONOMIC MODEL

A bio-economic model (Figure 4.1) was first developed to guide research efforts from the biological aspects through to the economic impacts (Leitch *et al.*, 1996). This aspect of the study was based on the study by Leitch *et al.* (1996), who had developed a bio-economic model to estimate the economic impacts of Leafy spurge L. (*Euphorbia esula*) infestations. The model identifies key relationships between the changes in the level of leafy spurge infestation and changes in land output (e.g. carrying capacity for grazing livestock). Bio-economic modelling, according to Knowler (2002), is typically used by economists to describe models that have both economic and biophysical components. The model was used to address the relationship(s), which exist between them. More specifically, bio-economic models are capable of simultaneously addressing the various dimensions of an agricultural system (Flichman, 2011).

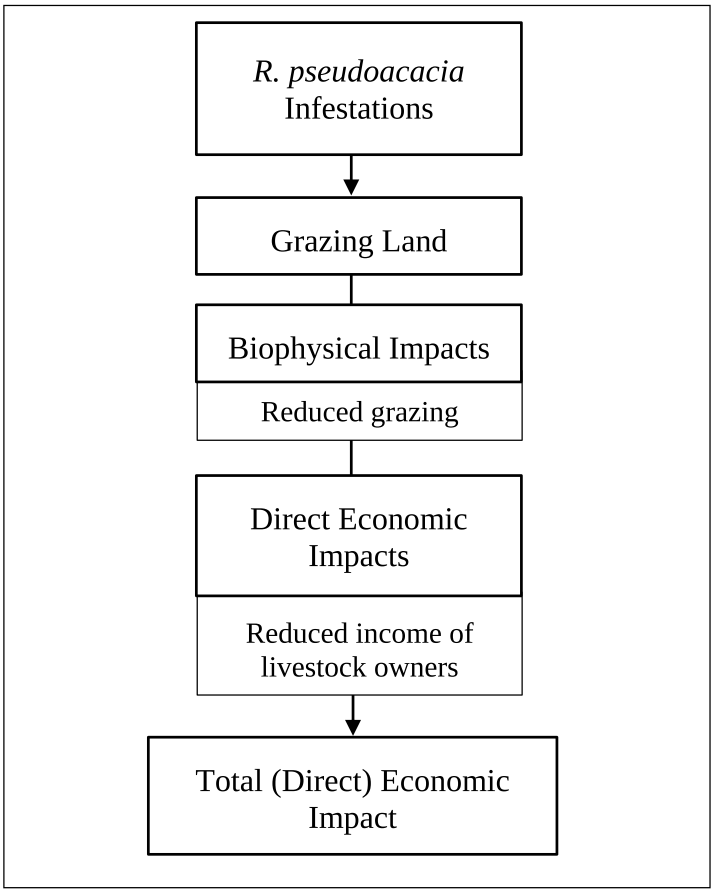

Figure 4.1. A conceptual bio-economic model of the economic impacts of *Robinia pseudoacacia* infestations.
Source: Adapted from Leitch *et al.* (1996).

Figure 4.1 describes the relationships between the biological aspects and the economic impacts: if an infestation of *R. pseudoacacia* occurred, this would impact the available grazing capacity in a number of ways. Firstly, *R. pseudoacacia's* toxic components would deplete livestock. Secondly, due to the clonal spread, specific areas of the grazing land would become restricted. The biophysical impacts would be seen as a reduction in grazing capacity, ultimately reducing the carrying capacity of a particular area. The economic impacts of this would result in a reduction in income for livestock landowners. In the last stage of the model, the total (direct) economic impact will be determined.

Due to the nature and the characteristics of this study, it is categorised as that of a bio-economic model. More specifically, according to Kragt (2012), this study is classified as agro-economic model, taking an accounting approach. Accounting models are simple descriptive bookkeeping systems of agricultural production system (Bouman *et al.*, 1999; Firth, 2002). The accounting models allow for an assessment of the impacts of land use on environmental indicators, but are limited in their ability to represent the dynamics of environmental processes, and feedback loops between environmental changes and land use decisions (Kragt, 2012).

4.8 SPECIES DISTRIBUTION MODELLING

Studying invasive species at large spatial scales traditionally has proven to be difficult. However, the recent rise in species distribution modelling (SDM), also known as niche ecological modelling, has made the process more feasible (Hoffman *et al.*, 2008). SDM is a popular method which is used to predict the potential geographic distribution of an organism (Robertson *et al.*, 2001, 2003; Mau-Crimmins *et al.*, 2006; Steiner *et al.*, 2008) and to predict the environmental suitability of regions that have not yet been invaded by invasive species (Mgidi *et al.*, 2007; De Meyer *et al.*, 2008). This is done by quantifying the species-environment relationship, so that the correlation between the occurrence of the species and the environmental parameters within a specific region are determined (Guisan & Thuiller, 2005). SDM plays a leading role in biogeography and regional ecology in estimating the niche and distribution area of a species (Booth *et al.*, 2014). SDM is often used in situations where distribution data of a species is limited (Elith *et al.*, 2006). With the advancement of computer technology, the current availability of species and climate information has greatly enhanced the field of SDM (Hijmans *et al.*, 2005; Jetz *et al.*, 2012; Beck *et al.*, 2013). Various models and programmes exist in order to determine this, such as climatic envelope models (CEMs) (Rougt *et al.*, 2004), MAXENT[13] (Phillips *et al.*, 2010) and the Bayesian hierarchical regression (Latimer *et al.*, 2004). The performance of various SDM algorithms has been evaluated through numerous comparative studies, taking into account multiple factors (Elith *et al.*, 2006; Tsoar *et al.*, 2007; Wisz *et al.*, 2008). Phillips *et al.* (2006) found that MAXENT outperformed the genetic algorithm for rule set prediction (GARP). Elith *et al.* (2006) conducted a study on 226 species in 6 different regions to determine the best SDM methods. They proved that out of 16 different methods for modelling the distributions, MAXENT was one of the best. Similarly, Wisz *et al.* (2008) tested 12 different prediction models which also showed that MAXENT was one of the best.

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[13] This is a ‘‘general-purpose’’ presence-only modelling method that estimates the probability distribution by predicting the maximum entropy based on a set of constraints (Hoffman *et al.*, 2008).

4.8.1 MAXENT

Because *R. pseudoacacia* is a wide-ranging species with a global distribution, a maximum-entropy predictive habitat model (MAXENT-Version 3.3.0) was used to simulate its potential distribution. MAXENT is a software tool with a simple and precise mathematical formulation (Phillips *et al.,* 2006). It is a relatively new statistical modelling technique applied to model the potential distribution of species and to estimate niche occupations (Peterson *et al.,* 2008). When modelling invasive species distribution, models may include a variety of data including native range data, invaded range data or a combination of both (Philips *et al.,* 2006; Wisz *et al.,* 2008). A better indication of the potential spread of a species into a non-indigenous range can be achieved by using a combination of data[14] (Mau-Crimmins *et al.,* 2006). The results can be used to optimise invasive species management programmes by determining areas that are susceptible to the establishment of an invasive species, thus allowing land managers to develop proactive management approaches (Hoffman *et al.,* 2008).

MAXENT is used for making predictions or inferences from incomplete information (Phillips *et al.,* 2006) and is regarded as one of the premier distribution-modelling software packages available (Thompson *et al.,* 2011). MAXENT allows one to estimate (approximate) the probability distribution of a species (Phillips *et al.,* 2006). The advantage of using MAXENT is that it can make use of presence-only data, as opposed to presence-absence data (Trethowan *et al.,* 2011). MAXENT therefore, is more valuable in regions where collecting absence points has proven problematic (Phillips *et al.,* 2009). MAXENT has been used in a wide variety of research studies as a highly effective tool for predicting potential species distributions (Hoffman *et al.,* 2008; Yost *et al.,* 2008; Wolmarans *et al.,* 2010; Trethowan *et al.,* 2011; Li *et al.,* 2014).

MAXENT applies five different feature constraints (linear, quadratic, product, threshold and hinge) to the environmental variables in order to estimate species distribution. The estimated MAXENT probability distribution of location (χ) is exponential in a weighted sum of environmental features (f) divided by a scaling constant ($Z\lambda$, Equation 1) to ensure that the

[14] It is to be noted that the models used can only be as good as the data which has been used to calibrate them. This can pose as a potential problem when predicting the potential range of invasive species (Wolmarans *et al.,* 2010).

probability values range from 0 to 1 and sum to 1 (Yost *et al.,* 2008; Li *et al.,* 2014). The MAXENT probability distribution takes the form:

$$q_{\lambda}(\chi) = \frac{\exp(\sum_{j=1}^{n} \lambda_j f_j(\chi))}{Z_{\lambda}}$$

Equation 4.1. MAXENT probability distribution formula.

where n is the number of environmental features, λ is the vector of the feature weights, with real values, and Zλ is the normalising constant that guarantees that the probability distribution sums to one over the area of interest.

The software utilises a set of input layers, or environmental variables, as well as a set of georeferenced occurrence locations or training data (Phillips *et al.,* 2006). The model then expresses the suitability of each grid cell as a function of the environmental variables at that grid cell (Kalle *et al.,* 2013). A high value of the function at a particular grid cell indicates that the grid cell is predicted to have suitable conditions for that species (Phillips *et al.,* 2006). The distribution chosen is the one that has maximum entropy, subject to some constraints: it must have the same expectation for each feature (derived from the environmental layers) as the average over sample locations (Phillips *et al.,* 2006). If a pixel in the study has a similar distribution to the training data, higher values are assigned and pixels with a different distribution are assigned lower values (Negga, 2007).

4.9 ANALYTICAL PROCEDURE

Predicting the potential distribution of *R. pseudoacacia* followed a similar method to that of Trethowan *et al.* (2011), who sought to determine the potential distribution of Pompom weed (*Campuloclinium macrocephalum*) in South Africa. The current distribution data of *R. pseudoacacia*, which was obtained from GBIF and SAPIA databases, was used to model the potential distribution of *R. pseudoacacia* using MAXENT. Input layers, which act as environmental variables, are used in the software in order to generate a probability distribution, starting from the uniform distribution and repeatedly improving the fit to the data (Phillips *et al.,* 2006).

4.9.1 DISTRIBUTION RECORDS

In order to determine the known distribution of *R. pseudoacacia*, a number occurrence records were sourced from online databases (GBIF and SAPIA). A total of 40 026 records were collected from the GBIF database and 347 from the SAPIA database. The data which was collected was scrutinised and ‘cleaned’ as follows: only records that were sufficiently accurate were retained, records that were older than 10 years were discarded, records which possessed no date of occurrence were discarded, records with no co-ordinates or inexplicit co-ordinates (less than 3 decimal points) were disregarded, records which were misidentified were discarded and duplicates were removed. Furthermore, maps of occurrence data were produced in Arc Map 10.3 to check for obvious errors. To avoid pseudo-replication, only one occurrence record per 10 min grid cell was used. Ultimately, 4514 (4295 from GBIF and 219 from SAPIA) records were used.

4.9.2 CLIMATIC VARIABLES

One of the most important environmental factors influencing the distribution of species is climate (Pearson & Dawson, 2003; Guisan *et al.,* 2013). Suitable bioclimatic predictor variables were selected and downloaded from the WorldClim database (Hijmans *et al.*, 2005). These climatic variables were chosen as they have been found useful in defining the eco-physiological tolerances of species (Hijmans *et al.*, 2005). It is also the most commonly used interpolated global climate data resource. Slater *et al.* (2012), however, stated that the WorldClim database does have a major drawback as the climate surfaces represent average temperature or precipitation over a period, with no indication of the annual variability.

The 19 BioClim variables (See appendix 2) have been widely used in SDM studies (Steiner *et al.,* 2008; Trethowan *et al.,* 2011; Li *et al.,* 2014), as the data can be easily downloaded from the WorldClim database with no further calculations required (Acosta, 2008; Li *et al.,* 2012). MAXENT was then used to model the potential distribution of *R. pseudoacacia.* A trial run was conducted, using all 19 bioclimatic variables (BIO 1-19) (see appendix 3). However, several of bioclimatic variables did not provide any relevant contribution to the MAXENT model. Those bioclimatic variables were removed and the model was run again using only the most significant predictor variables (See appendix 4) (Thompson *et al.,* 2011). The significant predictor variables included Annual Mean Temperature (BIO 1), Mean Temperature of Warmest Quarter (BIO 10), Mean Temperature of Coldest Quarter (BIO 11), Annual Precipitation (BIO 12), Precipitation Seasonality (BIO 15), Precipitation of Coldest Quarter

(BIO 19), Temperature Seasonality (BIO 4), Temperature Annual Range (BIO 7) and Min Temperature of Coldest Month (BIO 6).

4.9.3 MODEL CALIBRATION AND EVALUATION

The MAXENT model was calibrated using all the records for *R. pseudoacacia* to predict the potential invaded ranges in South Africa. Default MAXENT parameters were applied: 'logistic output', 'create response curves', 'jack-knife measures of variable importance', 'do clamping' and a regularisation value of 1 (Thompson *et al.,* 2011). The feature type was restricted to 'hinge features' to create smoother response curves to focus models on the 'strongest trends' in the data (Elith *et al.*, 2010). Elith *et al.* (2010) and Thompson *et al.* (2011) recommended this approach for introduced species, as it produces models that are likely to be more ecologically friendly. Where absence data was unavailable, MAXENT created pseudo-absence data drawn randomly from a geographically defined background (Engler *et al.,* 2004). It has been noted that the size of background from which to obtain pseudo-absence data can significantly influence model results (Phillips *et al.*, 2009; Thompson *et al.,* 2011).

The model was calibrated to avoid sample bias, by randomly reducing the introduced range data to 70% and keeping 100% of the native range data, a spilt recommended by Trethowan *et al.* (2011). The independent 'test data set' was used to test the accuracy of the model and comprised 30% of the introduced range data (Trethowan *et al.*, 2011).

4.9.4 MODEL ASSESSMENT

Model assessment allows the user to assess objectively the quality of the model's predictions. Without an objective assessment, the accuracy of the model is unknown (Martin, 2013). The best means of objectively assessing model performance is to use an independent set of locality records and a quantitative accuracy measure.

The model was evaluated firstly by a jack-knife analysis and with the use of various response curves, used to identify the most influential predictor variable(s) and to assess the predictive success of the model. Secondly, using the Area Under Curve (AUC) statistic[15]. There are other measures of predicted success available (Fielding & Bell, 1997), but - according to McPherson *et al.* (2004) - AUC has been found to be the most robust method. The AUC values range from

[15] Detailed descriptions of the AUC curve and jack-knife analysis can be found in Pearson *et al.* (2007).

0-1 where a value greater than 0.5 indicates a better-than-random performance event (Fielding & Bell, 1997). It is generally accepted that an AUC of less than 0.8 is regarded as a poor model, between 0.8 and 0.9 is a fair model and between 0.9 and 0.95 a good model and >0.95 an excellent model (Beaumont *et al.,* 2009; Trethowan *et al.*, 2011).

4.10 MAXENT RESULTS

The MAXENT predictions of *R. pseudoacacia* distribution were highly accurate (training AUC = 0.993; testing AUC = 0.993; standard deviation = 0.001), suggesting an excellent model (Beaumont *et al.,* 2009; Trethowan *et al.*, 2011). The relative importance of the climatic factors is shown in table 4.1. The environmental variables identified as most suitable for the SDM were subjected to jack-knife analysis to determine the influence of each variable.

Table 4.1. Relative contributions of the environmental variables to the MAXENT model.

Variable	Percent contribution
Annual Mean Temperature (BIO 1)	32.3
Precipitation Seasonality (BIO 15)	17.0
Precipitation of Coldest Quarter (BIO 19)	13.0
Mean Temperature of Warmest Quarter (BIO 10)	11.1
Mean Temperature of Coldest Quarter (BIO 11)	7.9
Temperature Annual Range (BIO 7)	6.6
Min Temperature of Coldest Month (BIO 6)	5.9
Annual Precipitation (BIO 12)	4.0
Temperature Seasonality (BIO 4)	2.2

The jack-knife analysis indicated that annual mean temperature (BIO 1), precipitation seasonality (BIO 15) and precipitation of coldest quarter (BIO 19) had the most influence on the potential distribution in comparison to the other variables. These three factors explain 52.3% of the variance (13%-32.2% for each factor). Mean temperature of warmest quarter (BIO 10), mean temperature of coldest quarter (BIO 11) and temperature annual range (BIO 7) explain 25.6% of the variance (6.6%-11.1% for each factor). The jack-knife analysis indicated that that the variables of least overall influence of *R. pseudoacacia* into South Africa were temperature seasonality (BIO 4), annual precipitation (BIO 12) and min temperature of coldest month (BIO 6). These climatic factors were less important in determining the geographical distribution of *R. pseudoacacia* (collectively, they explained 12.1% of the variance, 2.2%–5.9% for each factor).

MAXENT was used to generate potential distribution maps of *R. pseudoacacia* in South Africa (Figure 4.2.). The figure illustrates that *R. pseudoacacia* has the potential to be distributed across a large portion of the country. The distribution results are similar to that of Li *et al.* (2014), who sought to determine the global potential geographical distribution of *R. pseudoacacia*. Li *et al.* (2014) used MAXENT to simulate the potential distribution of *R. pseudoacacia* on a global scale and to determine the dominant climatic factors affecting its distribution. The study was conducted as no techniques are available that provide effective control of *R. pseudoacacia* invasions. Therefore, in order to make practical and controlled use of this multipurpose species, Li *et al.* (2014: 2774) stated that it was "essential to determine its global potential distribution area, significant environmental factors and species response curves, as prerequisites for top-level design in the introduction, cultivation, afforestation and invasion control of this species".

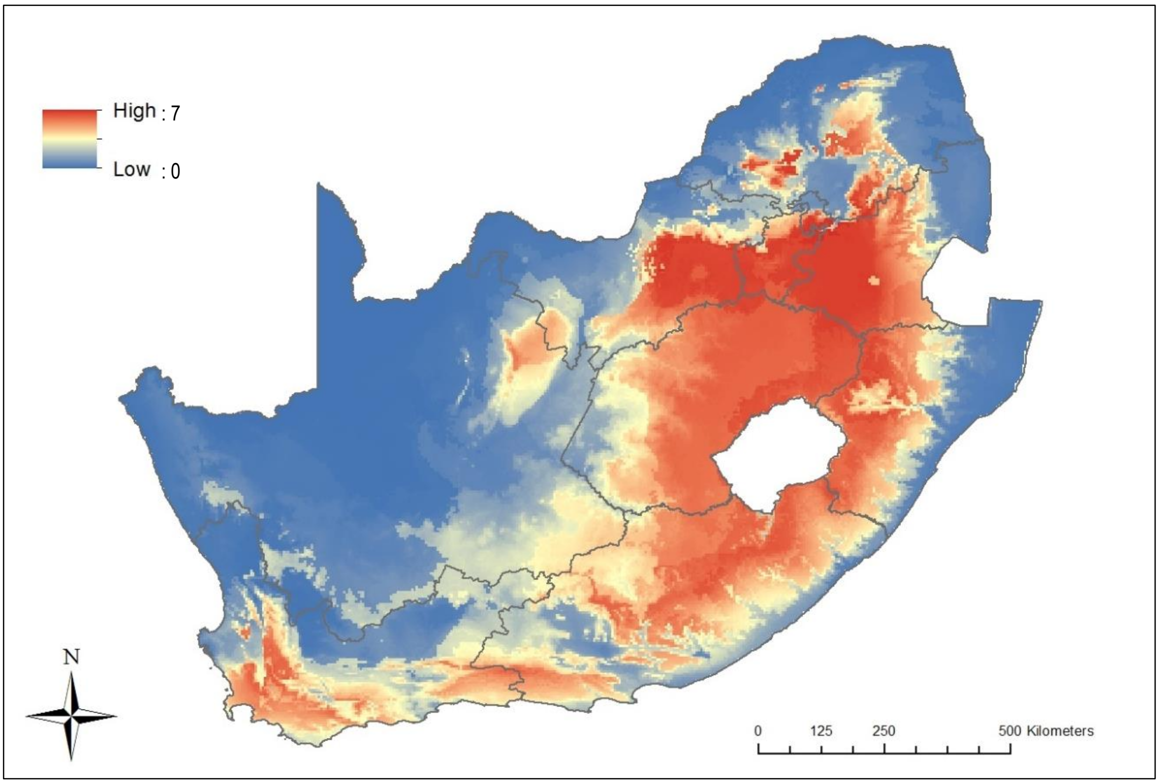

Figure 4.2. The potential distribution area of *Robinia pseudoacacia* predicted by the MAXENT model in South Africa.

The potential distribution of *R. pseudoacacia* was ranked by MAXENT on a scale from 1-7, where 7 represents a high probability of *R. pseudoacacia* establishment and 0 indicating areas less suitable (Figure 4.2). Higher probability (values closer to 7) represents areas most suitable for *R. pseudoacacia*, while zero or lower probability indicates areas less suitable for *R.*

pseudoacacia. The MAXENT model indicates that the regions of highest suitability for *R. pseudoacacia* are generally distributed towards the eastern portion of South Africa. This includes the Free State and Gauteng provinces, while there is a low probability of establishment of *R. pseudoacacia* into the western portion of the country (largely in the Northern Cape Province). This is due to the climatic suitability of eastern portion of the country for *R. pseudoacacia*. The regions of highest probability coincided with those regions where the IAP had been recorded in the SAPIA database[16] (see figure 4.3). Areas of moderate probability exist in the north and north-western portions of the Eastern Cape Province, as well as along the south-western coastline. Furthermore, there is a low probability in the northern tip of the country, in the Limpopo province, as well as along the eastern coastline.

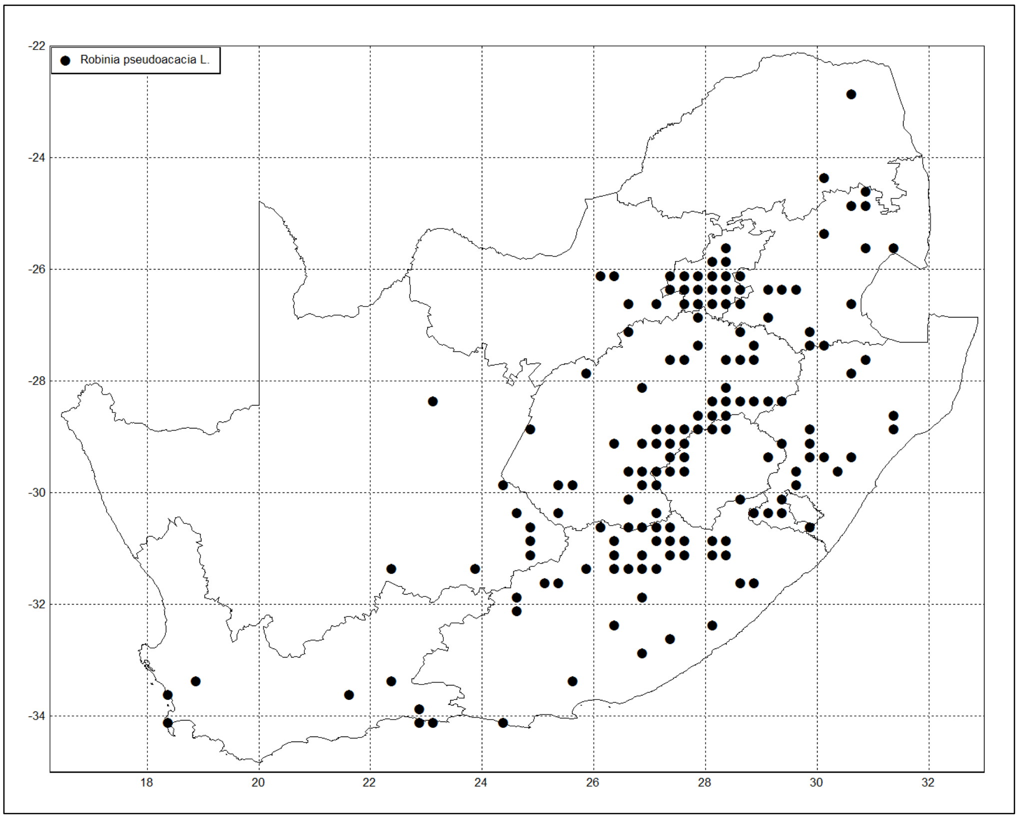

Figure 4.3. Distribution map of *Robinia pseudoacacia* in South Africa in quarter degree squares as of March 2015.
Source: SAPIA (2015).

Table 4.2 tabulates the calculated potential distribution of *R. pseudoacacia* in South Africa in hectares and percentage coverage based on probability. The probability of invasion column in

[16] It is to be noted that the SAPIA databases are by no means a conclusive record of the current distribution of *R. pseudoacacia* in South Africa.

the table depicts the potential probability of the likelihood of an invasion occurring derived from the MAXENT model.

Table 4.2. The potential distribution of *Robinia pseudoacacia* in South Africa.

Probability of Invasion	Area (ha)	Land Cover (%)
0	45,816,900	37.52
1	10,494,600	8.60
2	11,749,000	9.62
3	8,341,620	6.83
4	11,893,400	9.74
5	10,287,300	8.43
6	15,992,000	13.10
7	7,268,170	5.95
Total	121,842,990	

Approximately 37.52% or 45 million ha of the total land coverage in South Africa has zero probability for *R. pseudoacacia* establishment. Approximately 25% or 30 million ha of land are vulnerable to low levels of establishment (1-3), with approximately 18% or 22 million ha of land vulnerable to moderate levels of invasion (4 and 5). Although the highest probability of invasion (7) is only 5.95% of total land in South Africa, this represents over 7 million ha of land (mainly within the Gauteng and Free State provinces) which are suitable for establishment. Furthermore, the second highest probability of invasion (6) suggests that approximately 13.10% or almost 16 million ha of land is highly suitable for establishment by *R. pseudoacacia*. Overall, based on the MAXENT results, *R. pseudoacacia* has the potential for establishment in 62.48% of all land in South Africa.

4.11 ENVIRONMENTAL LAYERS

Once the MAXENT model had predicted the potential distribution of *R. pseudoacacia,* ARC-MAP 10.2 (ESRI, 2011) was used to overlay different environmental layers onto the model in order to further refine the data and to make it more accurate and suitable. These layers included the MAXENT layer, the National Land Cover (NLC) 2009 (Bhengu *et al.,* 2009), Mucina & Rutherford (2006) biome layer and the Agricultural Research Council (ARC) (2009) grazing capacity layer. The environmental layers improved the accuracy of the model and were crucial for determining the overall economic implications.

4.11.1 MAXENT LAYER

Once the potential distribution of *R. pseudoacacia* had been modelled using MAXENT, it was then brought into ARC-MAP 10.2 (ESRI, 2011) as a floating point raster image. The MAXENT data encompassed a global potential distribution of *R. pseudoacacia* and was refined to include only South African data. Extra MAXENT data was removed and only South African data was kept in order to make the file smaller and easier to handle. The file was then converted from a floating point raster image (MAXENT data) into an integer data set. This was done to group the data into a manageable and usable dataset in ARC-MAP. It also allowed an attributes table to be constructed for the dataset. The attributes table was constructed so that the number of pixels within the data set could be counted. The data was then projected from universal WGS84 data used in MAXENT, into an Albers equal area projection specifically set for South Africa (central meridian = 24º, standard parallel 1 = -24º, standard parallel 2 = -33º). Albers equal area is a conic, equal area map projection that uses two standard parallels. Albers equal area uses two standard parallels to reduce some of the distortion produced when only one standard parallel is used (GeoComm, 2014). This projection allowed accurate area calculation to be conducted on the maps. An area query was run using "tabulate area" in ARC-MAP, calculating the potential area of *R. pseudoacacia* in South Africa. This data was exported into Excel, converted into hectares and further tabulated. There were 64 603 cells within this data set, with each cell being 4509m^2 in size.

4.11.2 LAND-COVER CLASSES LAYER

The data set was the further modified to exclude specific land-cover classes in South Africa, where *R. pseudoacacia* does not pose as a threat to agricultural production - for example, water bodies and urban built-up areas. The South African National Biodiversity Institute (SANBI) National Land Cover (NLC) 2009 data layer was downloaded from the SANBI website and used (Bhengu *et al.*, 2009). NLC divides land cover (LC) and land use (LU) into various classes: natural, cultivated, degraded, urban built-up, water bodies, plantations, mines and other. For this study, it was decided that *R. pseudoacacia* has the potential to invade only natural land and degraded land classes[17]. The NLC layer was brought into ARC-MAP as a raster data set, which was then converted into an Albers equal area projection, in order to be compatible and aligned with the South African map projection of *R. pseudoacacia* distribution.

[17] However, it is to be noted that in the case study the tree was still found in cultivated fields and in urban areas.

By using the “extract by mask” tool in ARC-MAP, the unwanted land-cover classes were cut out of the layer, leaving only natural and degraded land suitable for *R. pseudoacacia* infestation. The NLC was overlaid onto the MAXENT layer and the “tabulate area tool” in ARC-MAP used to determine a new refined potential distribution of *R. pseudoacacia* in South Africa. This data was exported into Excel, converted into hectares and further tabulated.

4.11.3 BIOME LAYER

The model was further modified by using the Mucina & Rutherford (2006) biome data layer downloaded from the SANBI website and which is based on dominant forms of plant life and prevailing climatic factors. Mucina & Rutherford (2006) map nine biomes in South Africa: Fynbos biome, Succulent Karoo, biome, Desert biome, Nama Karoo biome, Grassland biome, Savanna biome, Albany Thicket biome, Indian Ocean Coastal Belt and Forests. The biome layer was brought into ARC-MAP as a raster data set, which was then converted into an Albers equal area projection. Due to the Mucina & Rutherford (2006) biome layer being mapped for Southern Africa, the biome layer had to be cut to fit the South African data set. This was done in ARC-MAP using “cut tool.” The potential distribution results of *R. pseudoacacia* were then overlaid onto the vegetation biomes in South Africa. The “tabulate area tool” in ARC-MAP was used to determine the overlapping areas between the MAXENT *R. pseudoacacia* potential distribution data layer, the NLC layer and the Mucina & Rutherford (2006) biome layer. This data provided the potential distribution of *R. pseudoacacia* within each biome in South Africa. It was exported into Excel, converted into hectares and further tabulated.

4.11.4 GRAZING CAPACITY LAYER

ARC (2009) grazing capacity data was used, which was sourced from Dr Anthony Palmer, who is a specialist scientist at ARC. Grazing capacity data for recent years was available, but Dr Palmer suggested the use of the 2009 data was accepted as an average year in terms of climate conditions and available vegetation. Furthermore, the paper by Messiner *et al.* (2013) also used the ARC 2009 data set.

ARC grazing capacity data was brought into ARC-MAP. The grazing capacity data is based on satellite imagery and net primary production (NPP), which represents the number of hectares needed per large stock unit (LSU). Messiner *et al.* (2013: 291) reports how this data set was created, “using the vegetation annual NPP for 2009 from the MODIS satellite programme (MOD 17), the g C/m^2 was converted to kg of dry matter/ha using a factor of 1.5.

This dry matter production was then partitioned into what is presumed available for consumption by livestock and the remainder which is generally not consumed. As the MODIS NPP product includes forests and woody components, this model assumes all plant functional types are available for consumption by herbivores." Messiner *et al.* (2013: 290) furthermore stated that "grazing capacity was predicted, according to the standard LSU definition[18], which in dry matter intake terms equates to about 9 kg/day. For the calculation, it was assumed that provision should be made for vegetation material that is available but not consumed because of preference and other reasons and therefore the dry matter intake estimate was escalated to 11.25 kg/day (Messiner *et al.,* 2013)".

The number of ha/LSU was determined within each biome in South Africa. The ARC grazing capacity data layer was overlaid with the Mucina & Rutherford (2006) biome layer and MAXENT layer. The "tabulate area tool" in ARC-MAP was used to determine the overlapping areas between the ARC grazing capacity data layer and the Mucina & Rutherford (2006) biome layer. This data provided the grazing capacity (ha/LSU) within each biome in South Africa. This data was exported into Excel, converted into hectares and further tabulated. The data was categorised further into the total number of hectares supporting high (<10), moderate (10 to 20) and low (>20) grazing capacity levels of LSUs (i.e. high grazing capacity category means that less than 10ha are needed per LSU, thus holding the most animals per area of grazing).

4.12 ESTIMATING THE IMPACT OF *ROBINIA PSEUDOACACIA* ON LIVESTOCK

Estimating the economic impact of infestations requires consideration of both biological and economic parameters (Leistritz *et al.,* 1993). A change in an area's agricultural production practices can affect agribusiness firms, local trade and service sectors (Leistritz & Murdock, 1981; Leistritz & Ekstrom, 1986). The potential economic impacts of *R. pseudoacacia* on agricultural production stems from the tree's ability to reduce livestock grazing capacity (Bangsund *et al.,* 1999). The establishment of the invasive tree restricts access to grazing lands and the seeds, leaves and bark of the tree are toxic to livestock[19] (Stephens, 1973; Cooper & Johnson, 1984; Cheeke, 1998). A critical step in estimating the economic impact of an invasion

[18] LSU is defined as the equivalent of one head of cattle with a body weight of 450 kg and gaining 500 g per day (Meissner *et al.*, 1983).

[19] Other economic impacts of invasions, which were not included in this study are lowering yield and quality of forage, increasing costs of managing and producing livestock, foregone livestock sales and potential decreases in land values (Ditomaso, 2000).

into grazing lands, was to estimate the potential reduction in the number of LSUs. In order to determine this, the ARC (2009) grazing capacity data was converted from the number of ha/LSU within each biome, to the number of LSU/ha within each biome (see appendix 5). The average number of ha/LSU within each category was used. For example, 2.5 was used in the 2-3 ha/LSU category. The grazing capacity of livestock in South Africa is expressed as the number of ha/LSU, since the capacity of the natural pastures of South Africa to carry stock is so low on account of the low rainfall and the arid nature of the larger part of South Africa (Du Toit, 2002). However, the data set was converted to LSU/ha, in order to calculate the total number of LSUs. It is to be noted that numbers in the horizontal axis title of appendix table 5.2 in appendix 5 are small, as one hectare of land is not able to carry one LSU. Only in the case of cultivated pastures in South Africa would one hectare of cultivated pasture is able to carry more than one LSU (Du Toit, 2002).

The impact of the potential distribution of *R. pseudoacacia* at different invasion probabilities was determined on LSUs in South Africa. In ARC-MAP, the ARC grazing capacity data layer was overlaid with the MAXENT *R. pseudoacacia* potential distribution data layer. The "tabulate area tool" in ARC-MAP was then used to determine the overlapping areas between the ARC grazing capacity data layer and the MAXENT *R. pseudoacacia* potential distribution data layer. This data provided the impact of the potential distribution of *R. pseudoacacia*, at different invasion probabilities (high, moderate and low), on LSUs in South Africa. It was exported into Excel and converted into hectares.

However, MAXENT only predicts the potential distribution of *R. pseudoacacia*, and does not predict the canopy cover of the growth - areas where *R. pseudoacacia* was predicted for growth (at any probability level), according to the MAXENT model, could range from a single tree to a large infestation. Thus, one could not assume that intermittent patches of *R. pseudoacacia* or a *R. pseudoacacia* monoculture would have the same impact on LSUs (Hirsch & Leitch, 1996). A large infestation of *R. pseudoacacia* would have a greater effect on LSUs, compared to that of a single tree.

In order to combat the problem of the unknown canopy cover of the potential invasion, three canopy cover invasion scenarios were constructed. The scenarios were based on guidelines developed by Le Maitre & Versfeld (1994), which provide for a range of density classes from rare (<0.01%) to closed (100% canopy cover). The case study provided insight into the size of infestations within the area, ranging from sparse (mainly in the town), to medium size areas (approximately 1 ha), to large infestations (approximately 10 ha).

In the first scenario, a high probability of invasion (5-7) was assumed. A dense canopy cover assumption of >20% was selected as some of the largest infestations discovered in the case study made up approximately 20% of a farms grazing land. It was further assumed in scenario one that *R. pseudoacacia* had the potential reduce the carrying capacity of LSUs by 50%, as an invasion would affect farming practices and some of the livestock would be exposed to *R. pseudoacacia*. Fifty percent reduction of LSUs was assumed, as not all LSUs within an area would necessarily be affected by an invasion. For example, if an infestations broke out on a farm, farmers would be able to move some of their livestock into areas on the farm where *R. pseudoacacia* has not invaded. The impacts of *R. pseudoacacia* on LSUs are based on the IAP being toxic to livestock and infestations restricting access to grazing lands through encroachment. It was assumed that the higher the probability of invasion - and the denser the canopy cover - the greater the impact of *R. pseudoacacia* on the carrying capacity of LSUs.

In the second scenario, a moderate probability of invasion (3-4) and medium canopy cover (5-20%) was assumed. It was also assumed that *R. pseudoacacia* had the potential to reduce the carrying capacity of LSUs by 25%. Due to only a moderate probability of invasion (3-4) and only medium canopy cover, the impact of *R. pseudoacacia* on the carrying capacity of LSUs was assumed to be less than in scenario one.

In the third scenario, low probability of invasion (1-2) and light canopy cover (<5%) was assumed. In addition, it was assumed that *R. pseudoacacia* had the potential to reduce the carrying capacity of LSUs by 12.5%. The impact of *R. pseudoacacia* on the carrying capacity of LSUs was assumed to be only 12.5%, as it was assumed low probability of invasion (1-2) and light canopy cover (<5%). Based on these assumptions, the impact of *R. pseudoacacia* on the carrying capacity of LSUs was assumed to be less than compared to scenarios one and two. These scenarios are summarised in table 4.3.

Table 4.3. Summary of the canopy cover invasion scenarios.

Scenario	Probability of Invasion	Canopy cover assumption	Impact on LSU
1	High (5-7)	>20%	50%
2	Moderate (3-4)	5-20%	25%
3	Low (1-2)	<5%	12.5%

Based on the relevant assumptions for each scenario, the total number of LSUs potentially effected by an invasion of *R. pseudoacacia* were determined. The total number of LSUs within each probability of invasion scenario, which was previously calculated, was multiplied by the canopy cover assumption and the impact on LSUs assumption. This was done for the whole of South Africa as well as for the grassland biome.

The potential impact on the gross margin in the livestock sector was estimated, within each probability of invasion scenario. The total number of LSUs within each probability of invasion scenario, was multiplied by the gross margin. Gross margins for livestock were obtained from the livestock enterprise budget, compiled by VKB in the eastern Free State. An average gross margin per LSU of R1000 was assumed[20]. Gross margins per LSU vacillated quite significantly, depending on the size and weight of the animals. Therefore, a gross margin of R1000 per LSU was chosen as it represents an average gross margin per LSU. Gross margin was defined as the difference between the annual gross income for that enterprise and the variable costs directly associated with the enterprise (Rural Solutions SA, 2012). Gross margins were used in this study, as in recent years, with the increasing economic pressure on agriculture, there has been a greater use of cost accounting techniques (Firth, 2002). Furthermore, gross margins provide a useful indication of the production and economic efficiency of an enterprise. Gross margins which are published are seen as being the "best possible estimates" (Firth, 2002), and represent "average case scenarios" (Rural Solutions SA, 2012). If the gross margin should change in the model, the economic impact results would also change accordingly. It is important to note that the gross margin of R1000 represents the formal sector livestock value of a LSU. See chapter 2, section 2.7 for a discussion on the value of livestock in South Africa.

4.13 ETHICAL CONSIDERATIONS

While carrying out and conducting this research, it was ensured that all requirements of the Rhodes University Research Ethical Standards Policy were observed. The survey instrument (questionnaire) was given ethical clearance by the Department of Economics Ethical Standards Sub-committee. This ensured that the rights, privacy and anonymity of any participants were acknowledged and respected, with no subject being offended or harmed in any manner. All participants were provided with consent forms giving precise details of the study's aim and objectives and how data would be used. Participants reserved the right to exit participation at

[20] For more information on the classification of livestock, see Sjoi *et al.* (2015).

any point in the study. Feedback will also be provided to all participants in the form of a summary report. The higher degrees ethical board of Rhodes University approved this research project.

CHAPTER 5

CASE STUDY - EASTERN FREE STATE PROVINCE

5.1 INTRODUCTION

A case study was conducted in the eastern Free State Province (Clarens, Fouriesburg and Ficksburg) of South Africa, as there is a growing abundance of *R. pseudoacacia* within the region. The rapid spread of this IAP species throughout Clarens and surrounding areas is cause for great concern (Coulsen, 2015). The case study allowed for a hands-on approach and was the source of our primary data collection. Primary data collection took place in the form of interviews and questionnaires (See appendix 1) (approved by the Rhodes University ethics committee). Numerous interested and affected parties were approached (local farmers, land owners etc.) and the majority of them reported that *R. pseudoacacia* is a problematic species. A large majority of the respondents had attempted to eradicate the tree, without much success when using mechanical and/or herbicidal control methods. These eradication attempts seemed to have only exacerbate the problem as the spread and regrowth of the IAP is more aggressive. Of particular concern is the investment of significant funds by the majority of the interested and affected parties into their eradication attempts, as each attempt has only aggravated the already considerable growth in the area (Coulsen, 2015).

This chapter analyses and discusses the results of the data collected for the case study. The chapter addresses the study site (5.2), the current distribution of *R. pseudoacacia* (5.3), the survey results, with emphasis on the respondent's perceptions and various control attempts (5.4), followed by a conclusion in section (5.5).

5.2 STUDY SITE

In economics, case studies are often used to investigate the structure of a given industry or region (Yin, 1994). This case study was conducted in the eastern Free State (predominantly in the Clarens region but also in Fouriesburg and Ficksburg) of South Africa (Figure 5.1). The location was selected as the study site as there is a growing abundance of *R. pseudoacacia* within the region. The climatic conditions of the Clarens region make highly suitable for the growth and spread of *R. pseudoacacia.* The invasive tree was believed to have been introduced to Clarens and immediate surroundings approximately 80 years ago (Coulson, 2015).

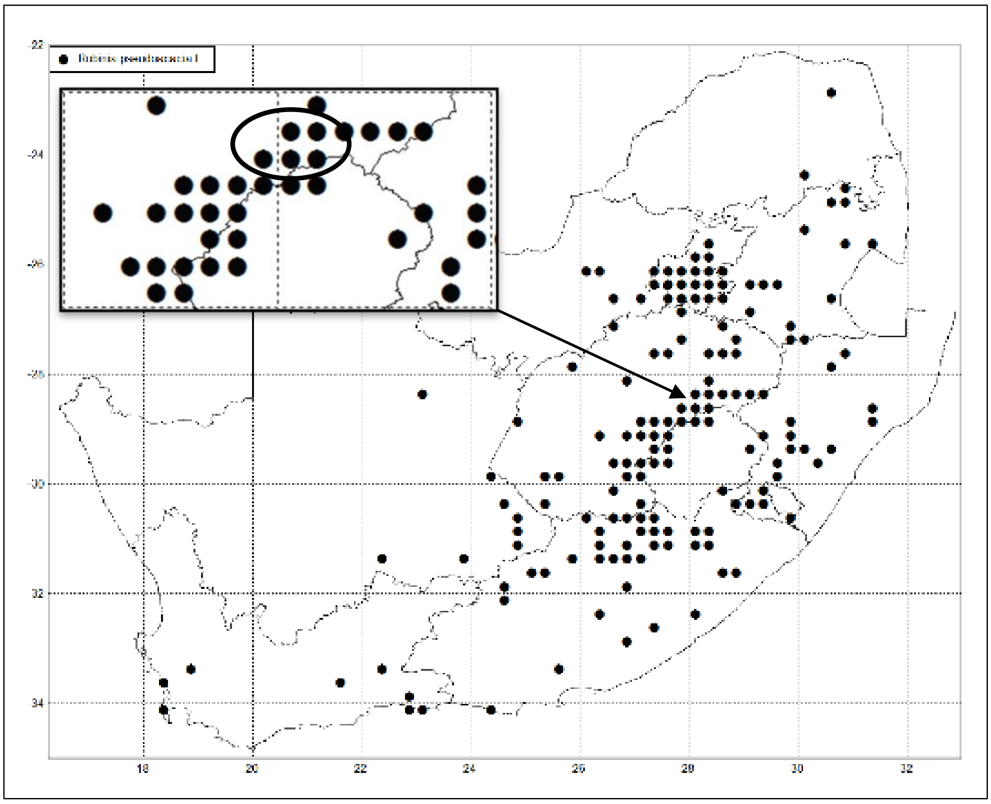

Figure 5.1. Distribution map of *Robinia pseudoacacia* in South Africa up until March 2015, illustrating the large concentration in the Clarens region.
Source: SAPIA (2015).

Clarens is situated in the Thabo Mofutsanyana District Municipality, more specifically in the Dihlabeng Local Municipality. Clarens is a small town, with a population of about 6500 people (Census, 2011). It is situated in foothills of the Maluti Mountains, in close proximity to the Golden Gate National Park and Lesotho, in the Free State province of South Africa. The town functions as a small service centre and is increasingly being supported by the tourism industry as it is within driving distance to Johannesburg and Durban, making it a popular weekend getaway. This has become a major driving force behind the small town's economy. Another major driving force is the various farmlands, as economic activities within the municipality are dominated by farming (29.4%) (StatsSA, 2011).

The region falls under the grassland biome, characterised more specifically as being an Eastern Free State Sandy (Gm4) Grassland, with the Basotho Montane Shrubland (Gm5) (Mucina & Rutherford, 2006) embedded in the hills. The region experiences a continental climate (warm and temperate), with rainfall throughout the year. Clarens receives on average 700mm of rain per year, with most rainfall occurring mainly during mid-summer. During the summer months, the temperature averages 25°C. The region is extremely cold during the winter months, averaging 14°C, with below zero degrees on average during the night. The region is flat to slightly undulating terrain with streams and rivers. The closed grassland is dominated by *Eragrostis curvula, Tristachya leucothrix* and *Themeda triandra* (Mucina & Rutherford, 2006).

5.3 CURRENT DISTRIBUTION

Interested and affected parties with an infestation of *R. pseudoacacia* on their land were approached within the eastern Free State area. The size of the infestations in the region ranged from sparse (mainly in the town), to some medium size areas (approximately 1 ha), as well as to large infestations (approximately 10 ha). As some of the interested and affected parties were not available for comment, the infestations on their land was documented from the roadside. Many of these infestations have been around for decades, where the trees stand tall and flower during spring. Other infestations are relatively new with a mixture of both tall and medium standing trees as well as new sprouting shoots.

Robinia pseudoacacia was been recorded in all the main waterways in Clarens, namely in the Caledon, Ash and Axle river[21]. It was also recorded in pockets adjacent to waterways in the Clarens Nature Reserve, in dongas, on road verges in the town, open parklands, and in untended and even maintained properties in Clarens (Coulson, 2015). One specific area where *R. pseudoacacia* has established itself in an abundance is in the "Naauwpoort Nek[22]". Figure 5.2 illustrates the magnitude of the spread and the density of *R. pseudoacacia* in the "Naauwpoort Nek" between 2009 and 2013.

[21] Furthermore, outside Fouriesburg, a large infestation was spotted following a river course. The size of the infestation following the river course was approximately 20km.
[22] A strip of land that falls within the north-eastern boundaries of the Clarens Nature Reserve and leads out of Clarens on the Bethlehem road.

Figure 5.2. Invasion of *Robinia pseudoacacia* into the “Naauwpoort Nek” between 2009 and 2014. Source: Google Earth (2009; 2013).

Figure 5.3 illustrates the current distribution of *R. pseudoacacia* in the “Naauwpoort Nek” as of October 2015. The spread of the IAP throughout the “Naauwpoort Nek” between 2013 and 2015 has been significant. The images below demonstrate the clonal spread characteristics which *R. pseudoacacia* possesses.

Figure 5.3. Invasion of *Robinia pseudoacacia* into the “Naauwpoort Nek” as of October 2015. Source: Authors; Google Earth (2013).

5.4 SURVEY RESULTS

The following section analyses and discusses the survey results which were undertaken in the study area during the period 30 September and 6 October 2015. The data under analysis includes data collected from 14 interested and affected parties who found *R. pseudoacacia* problematic or had attempted to eradicate the species within the Clarens region. The survey results are categorised into two main sections; perceptions of *R. pseudoacacia* (section 5.4.1) and the various control attempts (section 5.4.2).

5.4.1 PERCEPTION

Ninety-three percent of the respondents reported that this species is problematic. Forty-three percent of the respondents noticed the presence of *R. pseudoacacia* in the 1980s, however the infestations were significantly smaller. Fifty-seven percent of the respondents only noticed the IAP 2-5 years ago. Respondents reported that they only noticed the IAP due to the recent clonal spread of the species, believed it has been in the area for some time. Ninety-three percent of the respondents stated that, in the last few years, the spread of *R. pseudoacacia* has been significant. Many of them had attempted to control the recent spread, but had only exacerbated the problem, 50% of the respondents stated that the natural rate of spread had increased.

Forty-three percent of the respondents believed that *R. pseudoacacia* came from the grafted stems of mop head trees. Twenty-one percent believed that the IAP was introduced into South Africa for ornamental purposes, 7% believed that *R. pseudoacacia* was grown for use as axels for ox wagons many years ago and 29% of the respondents had no comment.

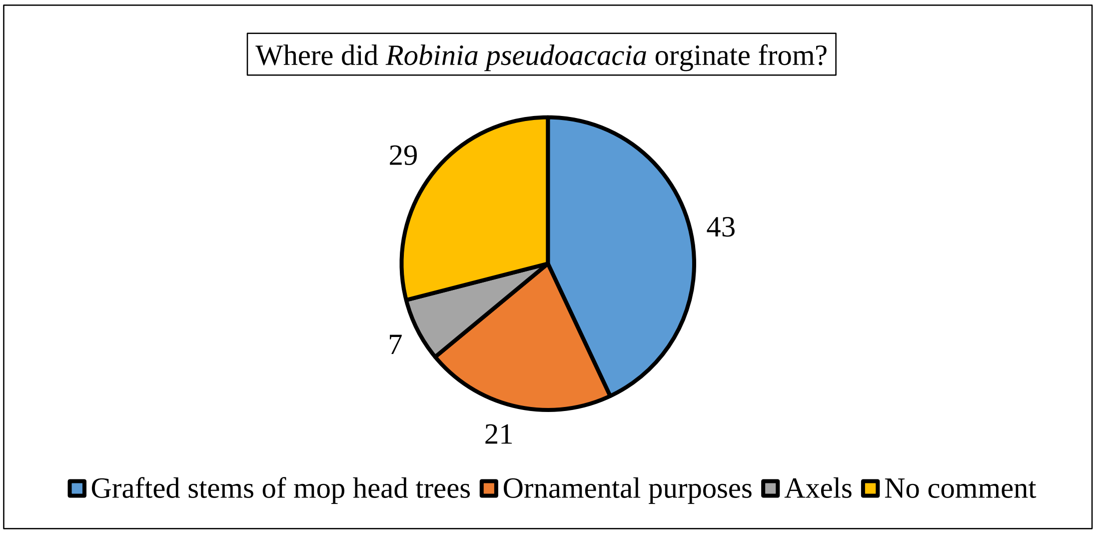

Figure 5.4. Diagram illustrating the respondents' answers with regards to where *Robinia pseudoacacia* originated from.

Fifty percent of the respondents regarded *R. pseudoacacia* as being the most problematic IAP within the area. This was a particular response from parties who had attempted to control the spread. They deemed it problematic as most of them were unsuccessful in controlling the spread and found that the regrowth to be more aggressive.

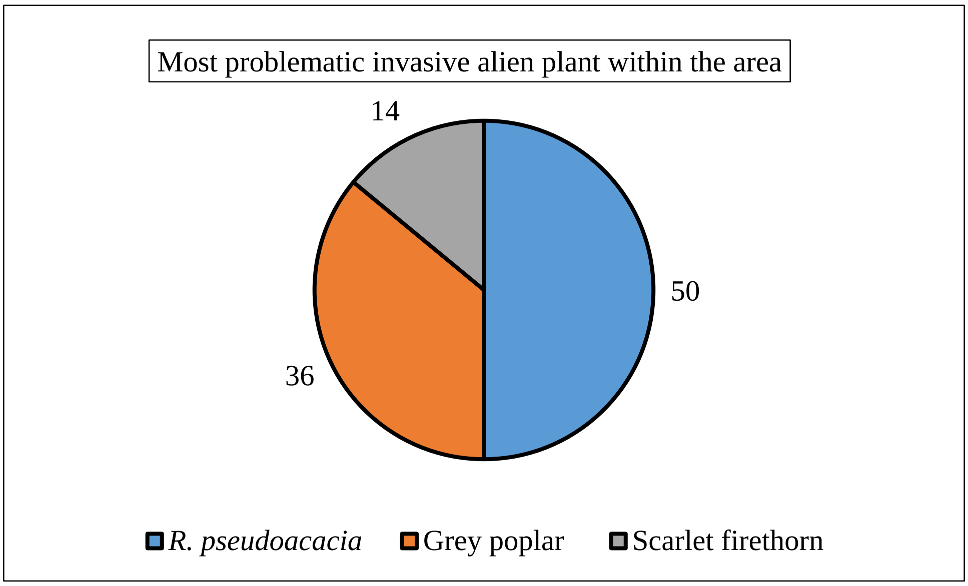

Figure 5.5. Diagram illustrating the respondents' answers with regards to most problematic invasive alien plant within the area.

Fifty percent of the respondents reported that other IAPs within the area are more problematic. Thirty-six percent of the respondents reported that Grey poplar (*Populus canescens*) was more problematic than *R. pseudoacacia* and only 14% stated that Scarlet firethorn (*Pyracantha coccinea*) was more problematic. However, these respondents had not attempted to control the spread of *R. pseudoacacia.*

Various reasons exist as to why *R. pseudoacacia* was regarded as being problematic. The first problem identified was associated with the spread of the IAP. The invasive tree spreads rapidly from suckering roots and suckers[23] profusely, resulting in new shoots popping up around the original stem of the tree. The invasive tree spreads into indigenous growth, ultimately replacing native vegetation. *R. pseudoacacia* is a prolific water user with a large underground biomass. Furthermore, it was found that indigenous undergrowth is virtually eliminated. The rapid spread of *R. pseudoacacia* results in the IAP restricting access to grazing lands and to water sources, as well as choking dongas. The thickness of the regrowth poses as a problem if there were to be a fire within the area as the infestation is often not penetrable, which could potentially result in parties not being able to tend to a fire (Figure 5.6).

[23] This describes vigorous stem growth coming from a trees root system, but some distance away from the crown. In other words, when a root sends up a new stem away from the main stem.

Figure 5.6. Illustration of the thickness and density of the regrowth of *Robinia pseudoacacia*.
Source: Authors.

The Clarens Village Conservancy said that *R. pseudoacacia* restricts access to some of the hiking and bicycle trails, encroaching on the trails and with the thorns posing a risk to humans. This could be cause for concern for the Conservancy as this could potential affect tourism within Clarens. Furthermore, *R. pseudoacacia* grows next to the side of the road, which makes it difficult to see oncoming traffic, especially on sharp bends.

The second problem identified was associated with the control of the IAP. Interested parties had attempted a variety of mechanical and/or herbicidal methods in attempt to control the spread, but all found that any attempt to control the spread resulted in a more rapid and aggressive regrowth and spread. Farmers who attempted to use mechanical and/or herbicidal control methods said that this needed to be consistently repeated. Many of the interested and affected parties also found that the thorns on the tree made it difficult to cut it down (Bongarten *et al.*, 1992; Dini, 1993). The thorns on the regrowth of the tree would also be harder, longer and often split.

Thirdly, many of farmers stated that the seeds and the leaves of the tree are toxic to their sheep and cattle. Livestock would experience weight loss and could eventually become anaemic. However, this was not the case with goats who are able to digest the seeds and leaves (Luginbuhl *et al.*, 2000). Grazing land is therefore restricted, as livestock can only graze in certain areas where *R. pseudoacacia* is not present. Grazing areas where *R. pseudoacacia* had encroached were no longer suitable for livestock.

Fourthly, due to minimal plant life growing beneath *R. pseudoacacia*, the land is prone to experiencing mechanical erosion (Figure 5.7) with the minimal plant life beneath *R. pseudoacacia* in turn increasing the level of surface runoff, leading to erosion. This is in contrast with the Ecological Society of Japan (2002) and Kowarik (2010), who suggested that *R. pseudoacacia* was planted for erosion control in Japan. Allelopathy is suspected by the Clarens Village Conservancy, but this cannot be confirmed (Czarapata, 2005). Another possible explanation is that of nitrogen fixation to the soil (Rice *et al.*, 2004).

Figure 5.7. Illustration of minimal plant life growing beneath *Robinia pseudoacacia.*
Source: Authors.

Despite *R. pseudoacacia* being perceived as a problematic species, 57% of the respondents suggested that the invasive tree does have some advantages. Firstly, all of the farmers stated that *R. pseudoacacia* provides for very good fence pole, due to the extremely hard wood of the tree and because the tree takes years to decay once cut down. Secondly, *R. pseudoacacia* can be used for fire wood as it is long-burning. However, many respondents stated that they do not use it as fire wood because the tree is extremely hard to fell, the thorns make it difficult to cut down and the burning wood gives off a bad odour. Thirdly, *R. pseudoacacia* provides as a source of grazing for goats. One farmer reported that he specifically grows and cuts the trees for his goats who graze on the seeds, leaves and flowers. Other farmers also stated that they use the leaves for early season fodder for some of their livestock.

5.4.2 CONTROL ATTEMPTS AND FINANCIAL IMPLICATIONS

The rapid spread of *R. pseudoacacia* throughout Clarens and proximate farmlands was cause for great concern (both environmentally and economically) with various attempts made at clearing *R. pseudoacacia* in and immediately surrounding Clarens. Four main attempts were conducted by government programmes on two of the larger infestations surrounding the town ("Naauwpoort Nek" and the Clarens Nature Reserve). These attempts were unsuccessful, due to unconsolidated management plans and ineffective removal methods. Funding of several million rand from government sources was utilised, but each attempt only aggravated the already considerable growth within the area. The government funded programmes are addressed in section (5.4.2.a-d). Various other smaller control attempts have also been conducted by the Clarens Golf Course, home owners and farmers which are discussed in section (5.4.2.e).

a FIRST CONTROL ATTEMPT

During the mid-1990s, the first attempt of clearing *R. pseudoacacia* was initiated, with a clearing of the infestations within the Clarens area by the municipality. However, the clearing of the invasive tree was not necessarily a control method, but was rather politically motivated with the intention of harvesting *R. pseudoacacia* as firewood and for building supplies for the local community. No herbicide was applied as the municipality was not necessarily aware - or concerned - with the potential spread of *R. pseudoacacia.* Furthermore, labourers were not necessarily contracted for the removal of the invasive tree, but rather were allowed to keep the timber as payment. It is believed that this management strategy of *R. pseudoacacia* triggered the initial spread.

b SECOND CONTROL ATTEMPT

In 2007, an Expanded Public Works Programme (EPWP) project was initiated, entitled: AP1 - River Rehabilitation for 5 Free State Rivers. This project involved the rehabilitation of the upper reaches of various rivers in the Free State (Axle River, Caledon River, Klip River, Lower Little Caledon River, Wilge River, Vaal River and Orange River catchment areas), by cutting down and removing IAPs within the floodplains. In the budget, funds were allocated for the various rivers. River Ranger Management was the organisation responsible for the Clarens region. The project was continuously funded and ran for approximately 4 years. The budget of the project also continuously increased over the years, totalling R14 million. It was reported that, of this R14 million, approximately half was spent on attempting to control *R. pseudoacacia* within Clarens. This took place at several localities, including the Clarens Nature Reserve (Figure 5.8) and the “Naauwpoort Nek”.

Figure 5.8. Mechanical and herbicidal clearing of *Robinia pseudoacacia* in the Clarens Nature Reserve by the EPWP.
Source: Wainright (2007).

A combination of tailored chemicals combined with mechanical control was used. Selective follow-up clearing attempts were initiated by River Ranger Management in order to combat regrowth, where they used foliar spraying as primary chemical application method, but these control measures were ineffective as within a few weeks new shoots had started growing. Within two years, the trees had grown back more aggressively and spread significantly (Figure 5.9).

Figure 5.9. The regrowth of *Robinia pseudoacacia* in the Clarens Nature Reserve two years after the clearing attempt by EPWP.
Source: Wainright (2007).

c THIRD CONTROL ATTEMPT

In 2014, Rand Water Foundation (RWF) made an attempt to control *R. pseudoacacia*, as well as other IAP, utilising a combination of cut-stump and foliar herbicide application in the Clarens Nature Reserve, in the townships of Phahameng and Kgubetswana, and in the area of the reserve adjacent the “Naauwpoort Nek”. Various sources reported that the work done by the RWF was inconsistent and that there were unconsolidated management plans. For this reason, there was a lack of follow-up, and the attempts for the eradication of *R. pseudoacacia* were completely ineffective (Figure 5.10). Although there is no evidence indicating the total funding of this project, it was estimated by various interested and affected parties that total funding was approximately R2 million.

Figure 5.10. The regrowth of *Robinia pseudoacacia* in the Clarens Nature Reserve, one year after the clearing attempt by the RWF.
Source: Authors.

Table 5.1. Summary table of the control attempts at the Clarens Nature Reserve.

Control Attempt	Year	Party Responsible	Control Method	Result	Cost
1	2007	Expanded Public Works Programme (EPWP)	Mechanical and herbicidal	Unsuccessful-thicker regrowth	R7 million between Clarens Nature Reserve and "Naauwpoort Nek"
2	2014	Rand Water Foundation (RWF	Cut-stump and foliar herbicide application	Unsuccessful-increased spread	R2 million between Clarens Nature Reserve and "Naauwpoort Nek"
3	2015	Clarens Village Conservancy	Mechanical and herbicidal	Unsuccessful-regrowth and increased spread	R20 000 per annum

d FOURTH CONTROL ATTEMPT

The Clarens Village Conservancy is currently attempting to eradicate *R. pseudoacacia* within Clarens. They have realised that past approaches to the eradication of *R. pseudoacacia* were completely ineffective and have begun to research and develop appropriate removal methods. They reported that the majority of the eradication attempts were done by using mechanical control methods, but were ineffective. Within a few weeks, new shoots would regrow from the original stem and the tree would also sucker, usually within a 1-3m radius of the original stem. The Clarens Village Conservancy made use of bulldozers to eradicate *R. pseudoacacia.* This did prove effective next to the road side; however, it was costly, damaging and would prove difficult when accessing some of the dense stands and dongas.

The Clarens Village Conservancy used a combination of tailored chemicals combined with mechanic al control (Figure 5.11). In 2013, the Clarens Village Conservancy began researching possible herbicides. An environmentally friendly herbicide, Plenum, which has tested to be effective on a wide range of IAPs Cotoneaster (*Cotoneaster franchetti*), Yellow Firethorn (*Pyrocantha angustifolia*) and Privet (*Ligustrum vulgare*) was selected for trials at test sites.

Figure 5.11. Mechanical and herbicidal clearing of *Robinia pseudoacacia* in the Clarens Nature by the Clarens Village Conservancy.
Source: Authors.

Their initial treatment consisted of combination cut-stump (at 150mm above ground level) followed immediately by basal application of Plenum using slashers and foliar applications. The use of slashers resulted in uneven cuts and absorption of herbicide and posed a tripping hazard and potential equipment damage for municipal vehicles trying to reclaim the land as parkland. Subsequently, their methodology was refined to cut-stump application (as level as possible) at 50mm below ground level utilising chainsaws (for stumps > 100mm) and hand saws. Herbicide was then applied only to the junction ring between cambium and bark for the larger stumps, and the entire stump surface for anything less than 100mm in diameter. Foliar applications were limited to IAPs of height <1000mm. After cut-stump applications, a window of 15 minutes is available before the plant begins to exude a seal of sap (Coulsen, 2015).

Despite the Clarens Village Conservancy's eradication efforts, it was evident that their attempts to control the spread of *R. pseudoacacia* were unsuccessful. Once mechanical and herbicidal application was applied, sprouting stems up to 100mm would appear after 5 days and up 300mm within 3 weeks (Figure 5.12).

 

Figure 5.12. Sprouting stems of *Robinia pseudoacacia* after mechanical and herbicidal application by the Clarens Village Conservancy.
Source: Authors.

In early 2014, the Clarens Village Conservancy decided to focus solely on its removal from within the Clarens Nature Reserve and all primary waterways within the Village. The decision was based on the knowledge that Cotoneaster, Yellow Firethorn and Privet were already established in all riparian areas and were not actively encroaching into new areas, whereas the reverse was apparent for *R. pseudoacacia.* The Clarens Village Conservancy, with aid from the Clarens 'Working on Fire' team, began clearing a large infestation in the Clarens Nature Reserve as well as at a secondary site behind a local school, Clarens Primary.

They soon realised that the team of 4 rangers and 27 'Working on Fire' personnel were insufficient to combat the spread of *R. pseudoacacia.* This was despite dedicating 6-8 hours a day and 3-4 working days/week on its removal (with a workforce of approximately 30 personnel, for 6 hrs a day, for 3 days a week, for 3 months - approximately 6480 man-hours). Since the mechanical and herbicidal control methods required monthly follow-ups for 3-4 months, from later spring to the end of summer, their continuous efforts to control the spread have been somewhat costly (Coulsen, 2015). It was estimated that the total control costs average between R15 000 and R20 000 per annum. The Clarens Village Conservancy reported that they have been attempting to eradicate *R. pseudoacacia* for approximately 3 and half years, at a total estimated financial cost of R60 000.

e OTHER CONTROL ATTEMPTS

The Clarens Golf Course was another interested and affected party experiencing problems with the spread of *R. pseudoacacia.* The golf course is situated just below the infestation at

“Naauwpoort Nek”. The infestation has now spread onto the golf course and is now spreading on the tee boxes closest to the “Naauwpoort Nek” infestation (Figure 5.13). Other relatively new patches of *R. pseudoacacia* have also been spotted on the course (Figure 5.14). The managers of the course have been using mechanical and herbicidal methods of control.

Figure 5.15. Invasion of *Robinia pseudoacacia* onto the tee boxes of the golf course.
Source: Authors.

Figure 5.16. Invasion of *Robinia pseudoacacia* on the golf course.
Source: Authors.

Due to the infestations being relatively small and mainly consisting of new sprouts, they have been able to cut the encroaching tree back. However, this has only aggravated the spread as every time they have attempt to control the tree, the regrowth is more aggressive. Furthermore, patches of *R. pseudoacacia* have also been spotted on the golf estate, primarily on plots of land which are for sale, but these patches were only recently identified and had not attempted to be controlled yet. The Clarens Golf Course reported that although the costs of control so far have been relatively small, if the IAP continues to spread further onto the golf course, the potential damages and control costs could be exponential.

The first home owner approached was relatively successful in eradicating *R. pseudoacacia* having first cut the tree down at the base of the stump, chain sawed an “X” onto the cut stump and then immediately pouring a strong salt water solution. This was the most successful control method recorded, as the home owner reported that there were relatively few new sprouting’s around the original stem although a follow-up was needed a year later.

The second home owner approached attempted to eradicate *R. pseudoacacia* from the banks surrounding the dam as well as in a nearby donga using a mechanical and herbicidal approach. The eradication attempt consisted of a cut-stump followed by the home owner drilling a hole into side of the stump and filling it with herbicide (herbicide-unknown). This method was highly unsuccessful as this only aggravated the regrowth of the tree (Figure 5.15).

Figure 5.17. Regrowth of *Robinia pseudoacacia* after mechanical and herbicidal control attempts on home owner two's property.
Source: Authors.

Seventy-five percent of farmers within the area reported that *R. pseudoacacia* was causing damages to their farming practices, mainly in terms of restricting access to grazing, thus decreasing the amount of grazing by livestock. Figure 5.16 illustrates the spread of *R. pseudoacacia* into the grazing lands. Many of the farmers had to relocate their livestock to different grazing areas when *R. pseudoacacia* had encroached onto their grazing lands. The farmers, however, did not necessarily realise the potential longer term effects (i.e. the potential spread of *R. pseudoacacia* into all available grazing areas).

Figure 5.18. The spread of *Robinia pseudoacacia* into grazing lands.
Source: Authors.

Eighty percent of the farmers had attempted to control the spread into their grazing lands. Twenty percent of them used mechanical control methods, while 60% of the farmers used a combination of mechanical and herbicidal control methods. They all reported that this only resulted in the regrowth of tree to be more dense and vigorous. One farmer, who has particularly large infestations on his farm, stated that "all methods used so far are expensive and have only resulted in increased regrowth." Trying to determine the financial costs of these controls for the farmers proved difficult, as they did not see this as a "cost." Their view was that their labourers receive a monthly salary to work on the farm, but after explaining to them to the opportunity cost of their labourers working elsewhere on the farm, many of them realised the cost of control. Costs of control were estimated ranged from R4 000 to R10 000 per annum. The majority of the farmers said that they only attempted control the spread of *R. pseudoacacia* once. They reported that once they saw increased rate of spread and the dense regrowth, it was not worth their time and their money to attempt control it again. Furthermore, the majority of them stated that they do not want to attempt to control the spread again until there is a successful method of control. Lastly, the majority of farmers could not put a monetary value on the costs to agricultural production per annum. This was extremely difficult to try and estimate as the infestations of *R. pseudoacacia* were mostly in one place on the farm. Farmers would just move their cattle to other grazing areas and could not provide values (in term of production) for the forgone grazing land, but one farmer did estimate that it was approximately R10 000 per annum.

5.5 CONCLUSION

Ultimately, in this case study it was found that *R. pseudoacacia* is a problematic species in the Clarens region. While *R. pseudoacacia* does seem to provide a few benefits, the problems reported were clearly larger. It is evident from the results that the control of *R. pseudoacacia* is a difficult and a long-term battle, given the way in which the tree spreads easily across the landscape. For farmers and other interested and affected parties, attempting to control *R. pseudoacacia* with their own resources can be financially draining, especially since mechanical and herbicidal control efforts require repetition. Although it was difficult to provide accurate monetary values of the impacts and costs of control, this case study demonstrates that all interested and affected parties had invested funds into various control measures (approximately R 9 million of governments funds and R100 000 of private funds), resulting only in the increased spread of *R. pseudoacacia*. Furthermore, it was reported, from an interested party who was involved in all the main clearing attempts, that "the clearing attempts have only aggravated the already considerable growth in the area. The spread of *R. pseudoacacia* is now approximately 5 times worse than what it was 2 decades ago". While *R. pseudoacacia* has not yet had any significant impacts of farmer's productions, the continuous spread into the grazing lands is a cause for great concern. Lastly, a key conclusion from this case study is that the spread of *R. pseudoacacia* is only aggravated if the tree is attempted to be controlled.

CHAPTER 6

POTENTIAL ECONOMIC IMPLICATIONS OF *ROBINIA PSEUDOACACIA*

6.1 INTRODUCTION

This chapter addresses the results of the desk top study by analysing the secondary data, and further examines the MAXENT models by overlaying available environmental layers (6.2), followed by the importance of agriculture in South Africa and the Grassland biome being addressed. The economic implications of the potential establishment of *R. pseudoacacia* in grazing lands in South Africa are then determined, as is an analysis of the economic implications in the Grassland biome (6.3), followed by a conclusion (6.4).

6.2 ENVIRONMENTAL LAYERS

Various environmental layers were overlaid over the *R. pseudoacacia* potential distribution. These layers included the National Land Cover (NLC) 2009 (Bhengu *et al.,* 2009), the Mucina & Rutherford (2006) biomes and the ARC (2009) grazing capacity[24] layer (see chapter 4, section 4.11 for more details on environmental layers). The environmental layers improved the accuracy of the model and were crucial for determining the overall economic implications. In order to complete the area calculations the potential distribution of *R. pseudoacacia* was cut using data from the environmental layers.

6.2.1 NATIONAL LAND COVER (NLC) LAYER

South African National Biodiversity Institute (SANBI) National Land Cover (NLC) 2009 data layer was used to refine the data (Bhengu *et al.,* 2009). Only natural land and degraded land classess in South Africa were selected and cut out using an editing tool in ARC-MAP (see chapter 4, section 4.11.2 for more detail). Figure 6.1 depicts the MAXENT potential distribution results of *R. pseudoacacia,* overlaid with the NLC (2009) layer. The areas in black on the map represent the land classes which are not suitable for the growth of *R. pseudoacacia.*

[24] Grazing capacity-the carrying capacity of an area expressed as the number of ha/LSUs.

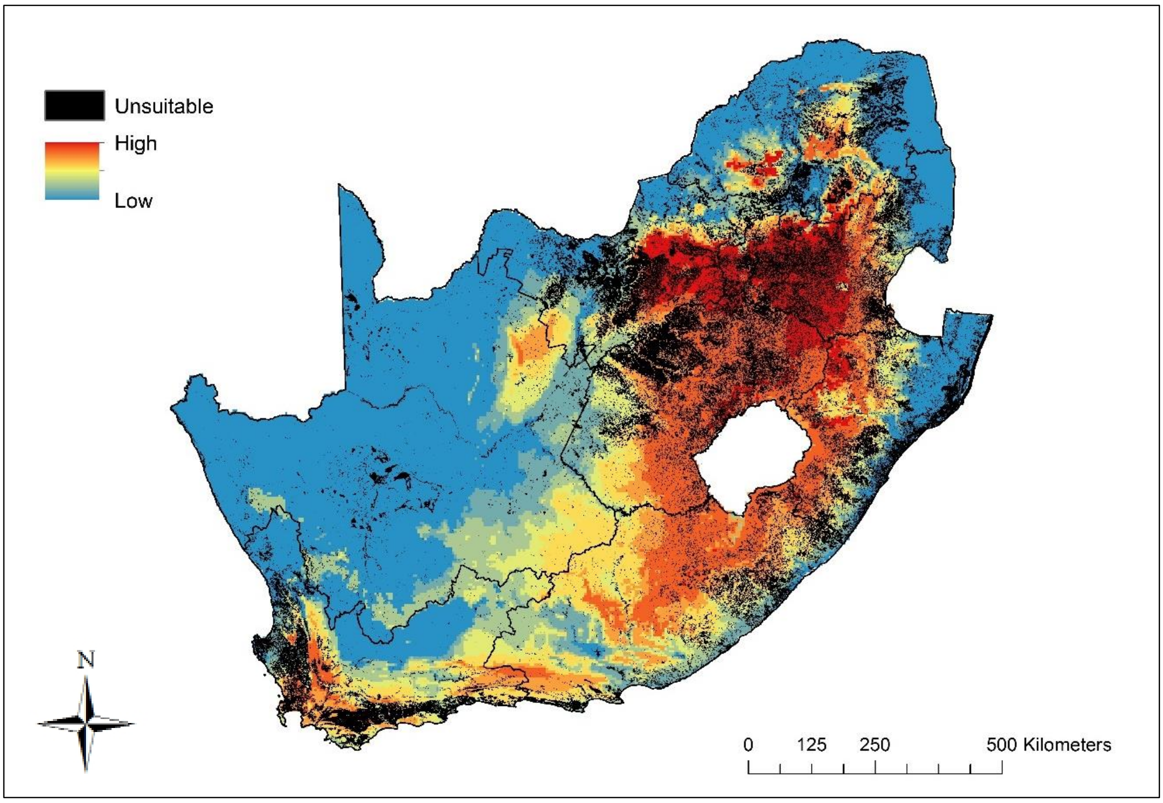

Figure 6.1. Map of the potential distribution of *Robinia pseudoacacia* in South Africa, with the NLC (2009) layer.

The potential distribution of *R. pseudoacacia*, less the areas not suitable for *R. pseudoacacia* establishment, was then calculated and tabulated (Table 6.1). The amount of land susceptible for establishment is less than previously calculated in chapter 4, section (4.10) from the MAXENT results (approximately 98 million ha versus approximately 121 million ha).

Table 6.1. The potential distribution of *Robinia pseudoacacia* in South Africa, with the NLC (2009) layer.

Probability of Invasion	Area (ha)	Land Cover (%)
0	41,256,800	33.79
1	8,453,440	6.92
2	9,935,540	8.14
3	6,650,120	5.45
4	9,329,690	7.64
5	7,158,390	5.86
6	11,029,300	9.03
7	3,974,620	3.26
	97,787,900	80.09

The NLC (2009) layer improved the accuracy of the distribution model. The highest probability of invasion (7) suggests that approximately 3.26% - or almost 4 million ha - of land is suitable for the establishment of *R. pseudoacacia.* Approximately 33.79% - or 41 million ha - of the total land coverage in South Africa has zero probability of potential distribution. Approximately 20.5% or 25 million ha of land is vulnerable to low levels of establishment (1-3), while approximately 13.5% - or 16.5 million ha - of land is vulnerable to moderate levels of establishment (4-5). Overall, based on the MAXENT results and the NLC (2009) layer, *R. pseudoacacia* has the potential to invade 46.3% of all land in South Africa (based on probability values of 1-7).

6.2.2 BIOME LAYER

The refined potential distribution of *R. pseudoacacia* was overlaid over the Mucina & Rutherford (2006) vegetation biome layer (see chapter 4, section 4.11.3 for more detail). This was done in order to determine which biome is the most susceptible for the establishment of *R. pseudoacacia*. Figure 6.2 illustrates the biome types in South Africa.

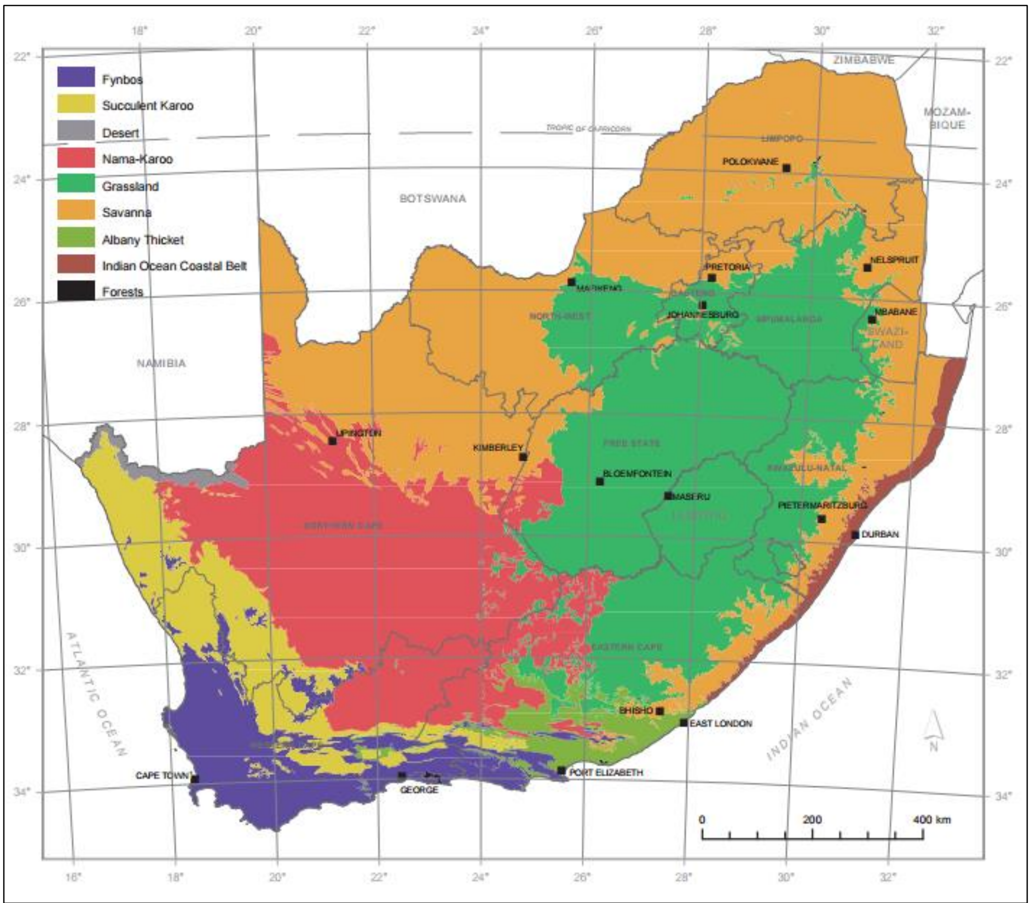

Figure 6.2. Biome types in South Africa.
Source: Mucina & Rutherford (2006).

Figure 6.3 depicts the refined MAXENT potential distribution results (i.e. potential distribution less the areas not suitable for *R. pseudoacacia* establishment) within the vegetation biome types in South Africa. The results illustrate that regions of highest suitability for *R. pseudoacacia* are distributed predominantly in the Grassland biome.

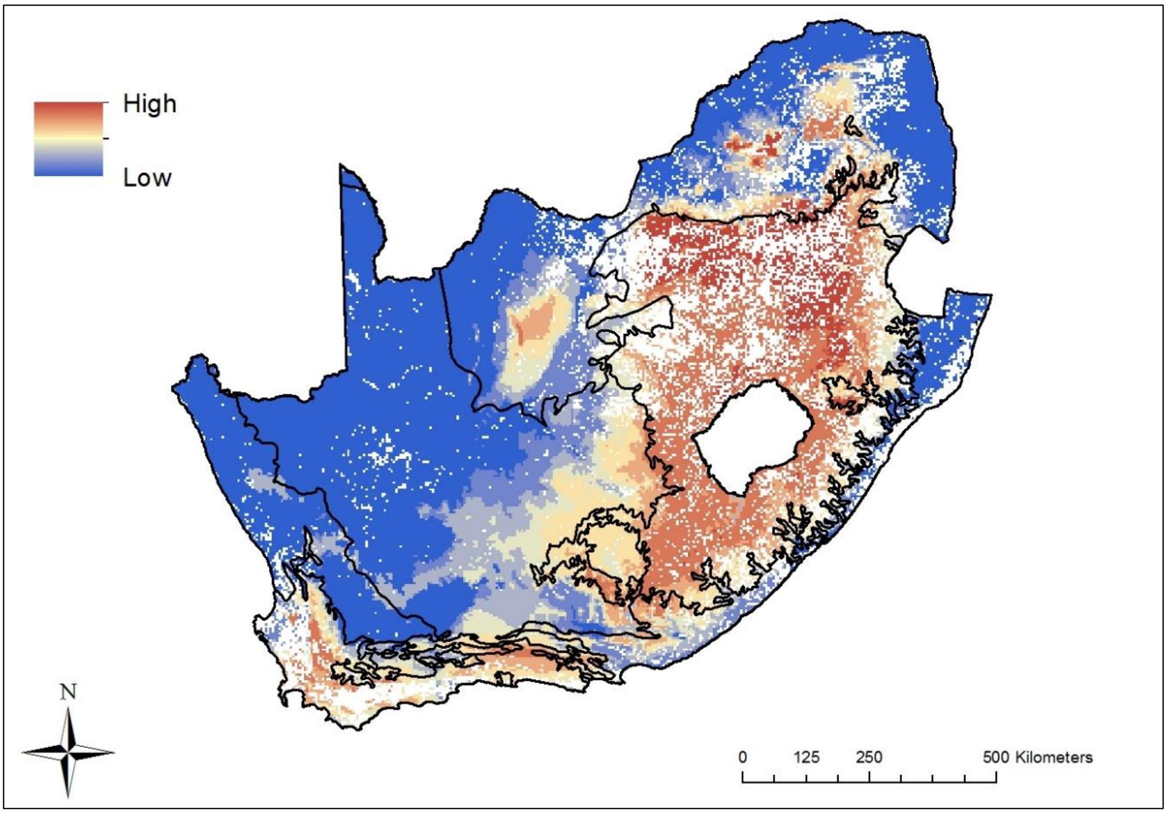

Figure 6.3. Potential distribution of *Robinia pseudoacacia* in different biome types in South Africa.

The Fynbos and Nama Karoo vegetation biomes are moderately suitable for the potential distribution of *R. pseudoacacia* with the potential suitability of *R. pseudoacacia* into the Desert, Succulent Karoo and Forest vegetation biomes at relatively low levels.

The potential distribution of *R. pseudoacacia,* at different probability distributions within the various biome types[25] in South Africa, was then calculated and tabulated (Table 6.2). The results illustrate that the Savanna biome is the second most suitable biome for invasion, with approximately 420 000 ha of land suitable for the potential distribution of *R. pseudoacacia* at the highest probability of invasion. The results suggest that no other biome is suitable for an invasion at the highest probability level (7). The Savanna and the Nama Karoo biomes are both extremely susceptible to an invasion, but only at low (1-3) and moderate (4-5) probability

[25] Albany Thicket and Indian Ocean Costal Belt biomes were not included.

levels. The Grassland, Savanna, Fynbos and Nama Karoo vegetation biomes are suitable for *R. pseudoacacia* at moderate probability levels of invasion (4-5).

Table 6.2. The potential distribution of *Robinia pseudoacacia* in different biome types in South Africa (ha).

Probability of Invasion	Savanna	Nama Karoo	Desert	Grassland	Succulent Karoo	Fynbos	Forest
0	15,217,413.4	19,289,621.7	50,826.4	209,404.6	6,200,816.4	189,074.1	0.0
1	4,456,455.6	3,216,292.3	0.0	294,792.9	187,041.0	296,826.0	0.0
2	3,490,754.7	4,596,736.4	0.0	481,833.9	819,321.0	544,858.6	0.0
3	2,252,624.5	2,630,772.6	0.0	890,477.9	235,834.3	636,346.1	4,066.1
4	2,647,037.0	3,185,796.5	0.0	2,047,285.9	180,941.9	1,240,163.3	26,429.7
5	1,805,352.5	1,002,295.9	0.0	3,175,631.2	24,396.7	1,146,642.8	0.0
6	827,453.2	311,057.3	0.0	**9,370,348.5**	8,132.2	496,065.3	0.0
7	418,809.2	0.0	0.0	**3,555,812.4**	0.0	0.0	0.0
	31,115,900.1	34,232,572.8	50,826.4	20,025,587.5	7,656,483.5	4,549,976.1	30,495.8

The results suggest that the Grassland biome is the most suitable for the potential distribution of *R. pseudoacacia* at the highest probability of invasion (7) and the second highest probability (6), which is approximately 3.6 and 9.4 million ha, respectively (Figure 6.4).

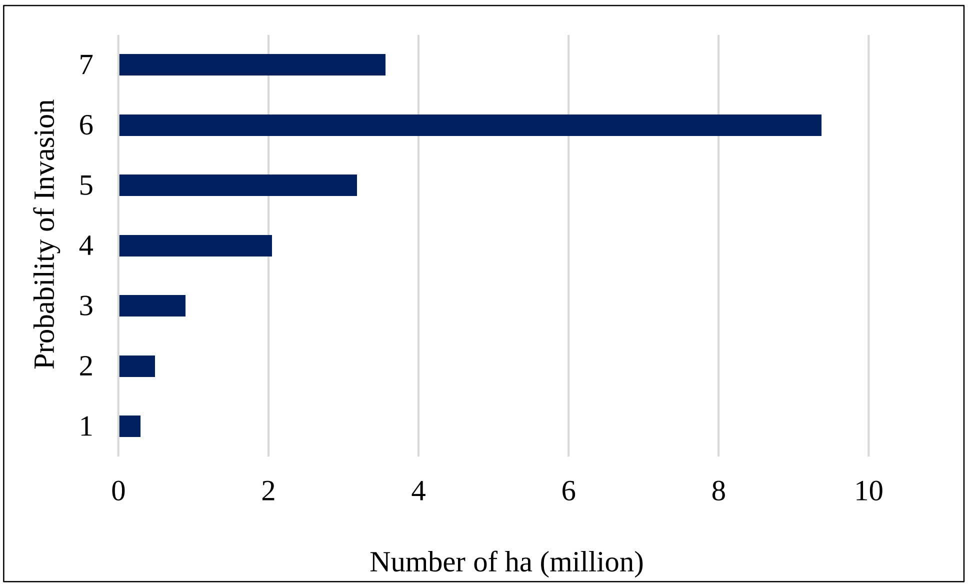

Figure 6.4. The potential distribution of *Robinia pseudoacacia* into the Grassland biome (ha).

Furthermore, *R. pseudoacacia* has the potential to invade approximately 19.8 million ha of the Grassland biome in South Africa (based on probability values of 1-7). Figure 6.4 illustrates that *R. pseudoacacia* has the potential to invade the Grassland biome at higher probabilities (4-7) rather than at lower probabilities (1-3). This suggests that the Grassland biome is the most suitable biome for *R. pseudoacacia.*

6.2.3 GRAZING CAPACITY LAYER

ARC (2009) grazing capacity data was used to determine the livestock grazing capacity in South Africa. The data is based on satellite imagery and net primary production (Meissner *et al*., 2013) (see chapter 4, section 4.11.4 for more detail). Figure 6.5 depicts the number of hectares needed per LSU. Fewer number of hectares (<7) needed to support a LSU are mainly distributed in the eastern portion of South Africa (KwaZulu-Natal Province, Mpumalanga Province and the eastern portions of both the Free State and Eastern Cape Provinces). A moderate number of hectares (8-20) needed to support a LSU are mainly distributed in the central regions South Africa, as well as along the south-eastern coast line, while a large number of hectares (>20) needed to support a LSU are mainly distributed in the western portion of South Africa (predominantly in the Northern Cape).

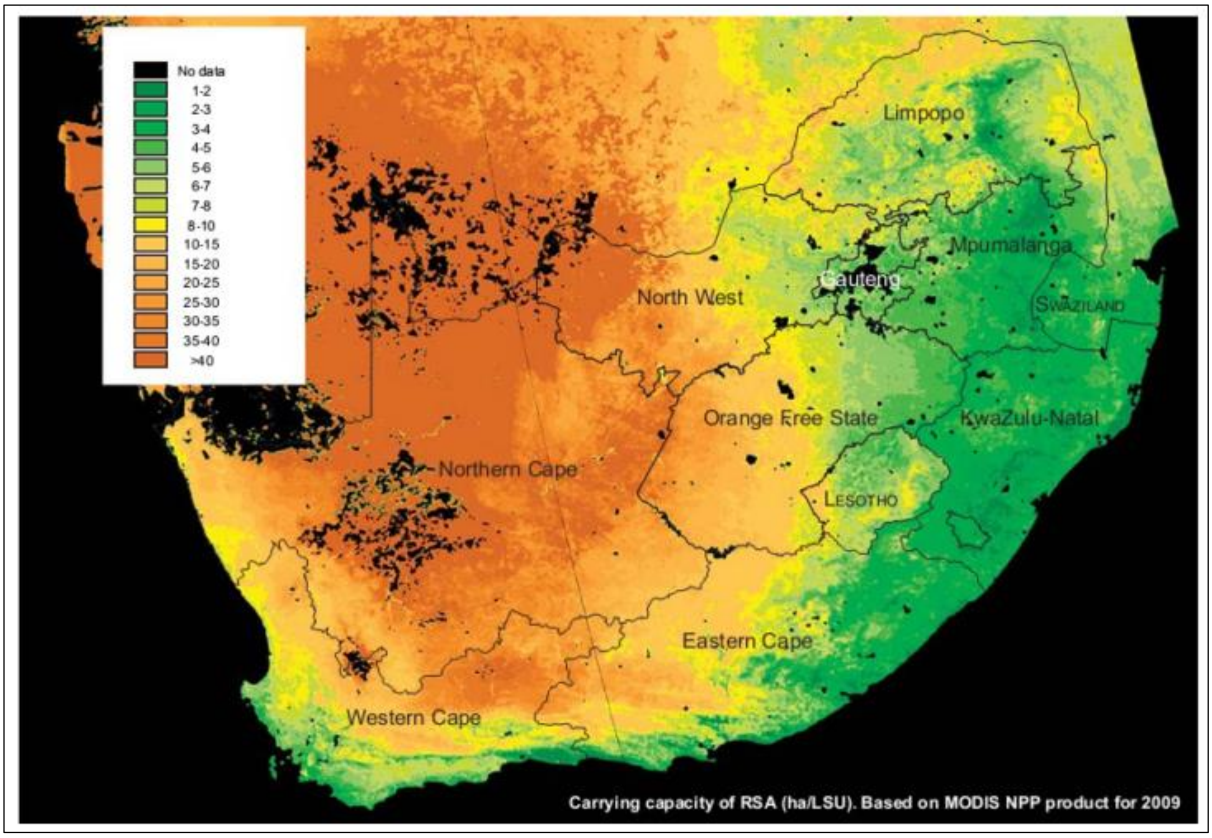

Figure 6.5. Livestock grazing capacity map of South Africa (ha/LSU).
Source: Meissner *et al.* (2013).

According to Meissner *et al*. (2013), the eastern portion of South Africa has the largest concentration of beef cattle. Eastern Cape (1.5 million), KwaZulu-Natal (1.4 million), and the Free State (1.2 million) province are the leading provinces in terms of beef cattle concentration.

The number of hectares supporting the relevant grazing capacity (ha/LSU) classes within various biomes in South Africa was determined (Table 6.3). For example, the Nama Karoo biome supports 24 493 476 hectares in the (35-40) ha/LSU class, suggesting that the Nama Karoo is considered the worst biome for grazing. Conversely, the Nama Karoo biome also supports the greatest number of hectares in the (2-3) ha/LSU class. A plausible explanation for this is that there is a large infestation of Mesquite (*Prosopis spp.*) within the Northern Cape Province. Because the ARC (2009) data is based on available vegetation, the Mesquite infestations are potentially influencing the Nama Karoo biome results. The results indicate that the Grassland biome supports 92 208 hectares in the (2-3) ha/LSU class. This suggests that the Grassland biome is the second best biome for grazing at the (2-3) ha/LSU class.

Table 6.3. Number of hectares supporting the relevant grazing capacity (ha/LSU) class, within the South African biomes.

Grazing Capacity	Savanna	Nama Karoo	Desert	Grassland	Succulent Karoo	Fynbos	Forest
2-3	87630	**319294**	3760	**92208**	60082	37848	490
3-4	434635	279321	2616	397768	45450	72507	22153
4-5	722539	183107	1635	480739	37194	146404	22234
5-6	1770665	136595	736	1178100	30000	123434	3924
6-7	1919358	120982	899	2551732	24360	196105	1553
7-8	767334	59755	409	1854616	14469	171581	1063
8-10	2723723	231010	82	6941732	83461	1384668	4414
10-15	8955914	403327	409	9474909	313653	2584267	163
15-20	8358036	749678	245	5863769	1931211	946436	0
20-25	3485253	2490588	0	1798703	1750883	684691	0
25-30	3137594	2651461	0	356487	1517257	419921	0
30-35	3475444	2492223	82	188993	1091451	162017	0
35-40	2484784	**24493476**	981	73979	1100770	59265	0

The data was then further grouped into the total number of hectares supporting high (<10), moderate (10 to 20) and low (>20) grazing capacity (ha/LSU) classes (i.e. high grazing capacity class means that less than 10ha are needed per LSU, thus holding the most animals per area of grazing). Simplifying the data allowed for pattern and trend identification as represented in table (6.4) and figure (6.6).

Table 6.4. Total number of hectares supporting the relevant high (<10), moderate (10 to 20) and low (>20) grazing capacity (ha/LSU) class, within the South African biomes.

	Grazing Capacity		
	<10	10 to 20	>20
Savanna	8425883	17313949	12583075
Nama Karoo	1330063	1153004	32127747
Desert	10136	654	1063
Grassland	**13496896**	15338678	2418162
Succulent Karoo	295015	2244864	5460361
Fynbos	2132547	3530703	1325894
Forest	55831	163	0

The results indicate that the Grassland biome has the largest number of ha/LSU, within the high grazing capacity class (<10). Further suggesting that the Grassland biome is the best biome for grazing, as fewer hectares of land are needed per LSU. The Savanna biome has the second largest number of ha/LSU within the high grazing capacity class (<10), with the Fynbos and Nama Karoo biomes having relatively low ha/LSU within this class. The results suggest that the Desert and Forest biomes have the least number of ha/LSU across all the grazing capacity classes, with the Fynbos and the Succulent-Karoo biomes having a relatively low number of ha/LSU across all the grazing capacity classes. This suggests that, within these biomes, there is not a lot of land susceptible for grazing at any grazing capacity class.

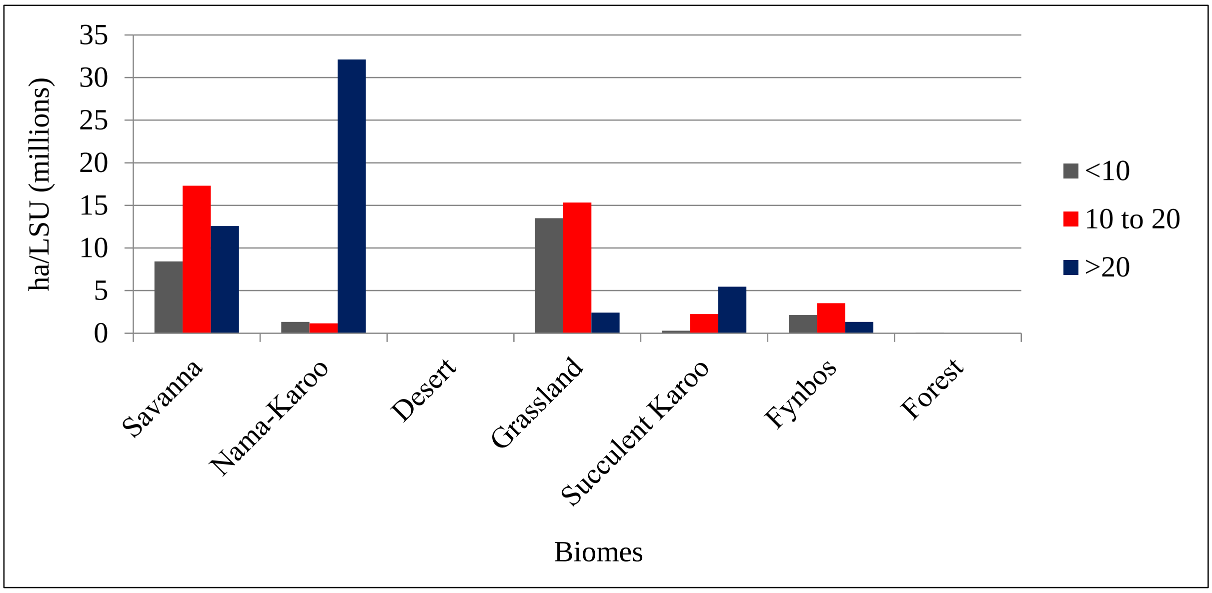

Figure 6.6. Total number of hectares supporting the relevant high (<10), moderate (10 to 20) and low (>20) grazing capacity (ha/LSU) class, within the South African biomes.

The Savanna biome has the greatest number of ha/LSU within the moderate grazing capacity class (10-20), closely followed by the Grassland biome. The other biomes within the moderate class have relatively small numbers of ha/LSU. Lastly, the Nama Karoo and Savanna biomes have the largest number of ha/LSU within the low grazing capacity class (>20). Although the Savanna biome has a reasonable number of ha/LSU in the high and moderate classes, the biome also has a large number of ha/LSU in the low grazing capacity class. The Nama Karoo on the other hand, has a relatively small number of ha/LSU in the high and moderate classes and a large number of ha/LSU in the low grazing capacity class.

6.3 ECONOMIC IMPACT ON LIVESTOCK

The potential economic impacts of *R. pseudoacacia* on agricultural production stem from the tree's ability to reduce livestock carrying capacity (Bangsund *et al.*, 1999). This is because an infestation of the tree, potentially restricts access to grazing lands and the seeds, leaves and bark of the tree are toxic to livestock (see chapter 4, section 4.12 for more detail). Meissner *et al.* (2013) stated that grazing capacity may deteriorate as a result of an invasion by alien vegetation. A critical step in estimating the economic impact of an invasion into grazing lands is to estimate the potential reduction in the number of LSUs. In order to determine this, the data was converted from the number of ha/LSU, to the number of LSU/ha (see appendix 5).

The total potential number of LSUs within each biome which may potentially be impacted by the tree was determined (Table 6.5 and Figure 6.7). It is to be noted that these predictions, which are based on satellite imagery and NPP (Meissner *et al*., 2013), represent the number of LSUs within the grazing lands of South Africa. This does not include LSUs grazing in cultivated lands, urban agriculture and feedlots. The results of the number of LSUs in South Africa are therefore lower when compared to existing literature (DAFF, 2013a; RMRD SA 2012; Meissner *et al*., 2013).

Table 6.5. Potential number of LSUs impacted, within each South African biome.

Biome	Number of LSU
Grassland	3 076 481
Savanna	2 978 237
Nama-Karoo	1 337 354
Fynbos	610 821
Succulent Karoo	397 063
Forest	13 064
Desert	3 026
	8 416 046

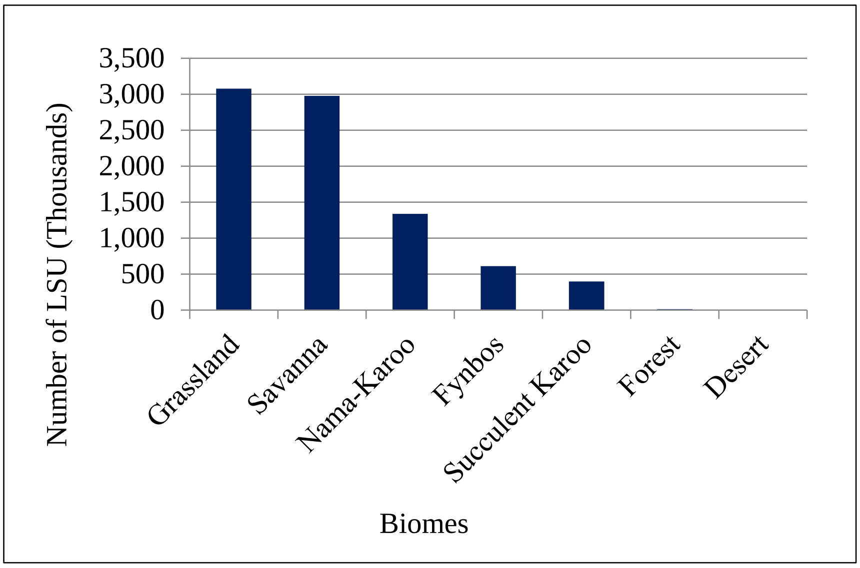

Figure 6.7. Potential number of LSU's impacted, within each South African biome.

The results indicate that the Grassland biome potentially contains the largest number of LSUs, at approximately just over 3 million. This is followed closely by the Savanna biome which potentially contains approximately 3 million LSUs. The Nama Karoo biome contains approximately half the number of potential LSUs to that of the Grassland and Savanna biomes. The Fynbos and the Succulent Karoo biomes contain relatively smaller numbers of potential LSUs of approximately 600 000 and 400 000, respectively. The Forest and the Desert biomes contain low numbers of potential LSUs, relative to the other biomes.

6.3.1 IMPACT OF *ROBINIA PSEUDOACACIA* ON LIVESTOCK IN SOUTH AFRICA

South Africa has a dual agricultural economy. The agricultural economy incorporates both well-developed commercial farming and subsistence-based production (Tibane & Vermeulen, 2014). South Africa has a diverse range of agricultural farming activities, including crop production, cattle and poultry ranching, cut flowers, citrus and deciduous fruits, game, animal products and livestock (Tibane & Vermeulen, 2014).

Despite only contributing 2.3% to South Africa's primary production in 2013[26], agriculture is an important employer of labour, is responsible for supplying local and global commodities, has a significant influence on alleviating food scarcity and provides foreign exchange earnings (Meissner *et al.*, 2013; Republic of South Africa, 2013; Soji *et al.*, 2015). It is a key focus of the New Growth Path as well as a significant contributing component to South Africa's socio-

[26] Note: Figure is for agriculture, forestry and fisheries contribution to GDP (DAFF, 2013b).

economic development (Nieuwoudt *et al.,* 2004; Eagle *et al.,* 2007; Republic of South Africa, 2013). According to the DWA (2012), approximately 8.5 million people directly or indirectly gain employment or income through the agricultural sector in South Africa. Moreover, the primary agricultural sector represents about 7% of formal employment in South Africa (Tibane & Vermeulen, 2014). Gross farming income from all agricultural products for 2013 was estimated at R182 966 million, which is 8.6% higher than the 2012 year (Republic of South Africa, 2014).

In South Africa, there are approximately 100 million hectares of agricultural land. Arable land makes up 14%, 72% is used for extensive grazing, 11% for nature conservation and 1% for forestry (Feynes & Meyer, 2003). Therefore, agricultural land in South Africa is primarily livestock-based (Meissner *et al.,* 2013). The grazing capacity of land increases eastwards, in accordance with increased rainfall (Goldblatt, 2010). Sheep farming is primarily conducted in the western and central areas of the country, while cattle farming generally in the eastern parts of the country. However, cattle is still farmed in the North West and in the Northern Cape provinces (Goldblatt, 2010). The Eastern Cape Province dominates in the livestock sector as the province has the highest concentration of cattle, sheep and goats in South Africa. The Free State and KwaZulu-Natal provinces are closely behind. Northern Cape is second in sheep production, with Free State closely behind (Meissner *et al.,* 2013). According to Gbetibouo & Ringler (2009), most of the grazing land in South Africa is stocked beyond its long-term grazing capacity. This is said to be especially prominent in the communal rangelands of Limpopo, KwaZulu-Natal and in the Eastern Cape provinces.

There has been a significant increase in the demand for livestock products since 2000/1 (Taljaard, 2006; Thornton *et al*., 2009; Spies, 2011). The consumption of beef in 2000/1 was 33.7g per capita per day, which rose to 46.8g per capita per day in 2010/11. Similarly, the consumption of pork rose from 7.12g to 12.6g per capita per day, from the year 2000/1 to 2010/11 (Meissner *et al.,* 2013). This demonstrates that livestock products in South Africa contribute substantially to food security. Furthermore, Meissner *et al.* (2013:282) stated "livestock farming is the backbone of the socio-economy and provides the sustenance of most non-metropolitan towns and rural communities."

Livestock production not only contributes substantially to food security in South Africa (Meissner *et al*., 2013), but forms a critical part of South Africa's socio-economic and socio-political stability (Tibane & Vermeulen, 2014). According to Meissner *et al.* (2013), it was estimated that 38 500 commercial farms and 2 million small-scale/communal farmers are involved with livestock. The livestock sector is a major employer in South Africa, employing an estimated 245 000 workers. Livestock foods on a weight basis contribute 27% of the consumer food basket, with red and white meat contributing 13% (DAFF, 2010a). Livestock products[27] accounted for approximately 47% of South African agricultural GDP in the period 2006/2010 (Tibane & Vermeulen, 2014; Meissner *et al.*, 2013), while future global trends predict an increase in demand for meat of almost double towards 2050 (FAO, 2009). With populations of some 13.9-million cattle, 28.8 million sheep and 6.4 million goats, livestock farming is the largest agricultural sector in South Africa (Palmer & Ainslie, 2006; DAFF, 2013a; Tibane & Vermeulen, 2014).

For developing countries, livestock is critical as it contributes to multiple livelihood objectives and offering ways out of poverty (Randolph *et al.*, 2007). Furthermore, livestock is the primary driver underpinning sustainable rural agriculture (Palmer *et al.*, 2010). Livestock owned by the poor in deep rural communal areas is a valuable asset as a store of wealth (Van Rooyen *et al.*, 1981; Boval & Dixon, 2012) and is a significant contributor to food security, clothing while providing a safety net (Meissner *et al.*, 2013). Moreover, livestock is utilised as collateral for credit in difficult times (DAFF, 2010b), contributes substantially to maintaining health and constitutes the main source of nutrition (Randolph *et al.*, 2007; FAO, 2009).

Recognising that livestock is an important component in agriculture production in South Africa, the impact of the potential distribution of *R. pseudoacacia*, at different invasion probabilities, on LSUs in South Africa was determined. Table 6.6 represents the number of LSUs that would potentially be impacted by the tree, should it reach its full potential distribution. Table 6.6 illustrates high (5-7), moderate (3-4) and low (1-2) probabilities of *R. pseudoacacia* invasions and the corresponding potential reductions in the number of LSUs in South Africa.

[27] Game species' products (meat, skins and hides) are not included in livestock products as their contribution is insignificant.

Table 6.6. The impact of the potential distribution of *Robinia pseudoacacia* on LSUs in South Africa.

Probability of Invasion	Number of LSU
High (5-7)	1 922 717
Moderate (3-4)	1 047 292
Low (1-2)	1 037 499

The results suggest that at a high probability of invasion, there would be a reduction of approximately 1.9 million LSUs. At moderate and low probabilities of invasion, there would be reductions of approximately one million LSUs.

However, MAXENT only predicts the potential distribution of *R. pseudoacacia*, and does not predict the canopy cover of the invasion. In order to combat the problem of the unknown canopy cover of the potential invasion, canopy covers were estimated and three canopy cover scenarios were constructed. These canopy cover invasion scenarios are summarised in the table 6.7 (see chapter 4, section 4.11 for further explanation).

Table 6.7. Summary of the canopy cover invasion scenarios.

Scenario	Probability of Invasion	Canopy cover assumptions	Impact on LSU
1	High (5-7)	>20%	50%
2	Moderate (3-4)	5-20%	25%
3	Low (1-2)	<5%	12.5%

Based on our canopy cover and impacts on LSUs assumptions, the total number of LSUs that could potentially be impacted by an invasion of *R. pseudoacacia* were estimated at different invasion probabilities (Table 6.8). This allowed for a more accurate and realistic estimation.

Table 6.8. Number of LSUs potentially impacted by an invasion of *Robinia pseudoacacia* in South Africa, based on the relevant scenario assumptions.

Scenario	Probability of Invasion	Number of LSUs Impacted
1	High (5-7)	961 359
2	Moderate (3-4)	261 823
3	Low (1-2)	129 687

The results suggest that in the first scenario, at a high probability of invasion, *R. pseudoacacia* has the potential to impact 961 359 LSUs (assuming a dense canopy cover of >20% and a reduction of LSUs by 50%). This represents just less than one third of the total number of LSUs within the Grassland biome (see table 6.5). The second scenario, at a moderate probability of invasion, indicates that the invasive tree has the potential to impact 261 823 LSUs (assuming a medium canopy cover of 5-20% and a reduction of LSUs by 25%). Lastly, in the third scenario, at a low probability of invasion, the invasive tree has the potential to impact 129 687 LSUs (assuming a light canopy cover of <5% and a reduction of LSUs by 12.5%).

In monetary terms, the impact of the potential invasion of *R. pseudoacacia* at a high probability level on LSUs in South Africa (Scenario 1) could potentially risk causing a reduction in gross margin in the livestock sector of approximately R961 million[28]. It was estimated that an invasion at a moderate probability level (Scenario 2) could potentially cause a reduction in gross margin in the livestock sector of approximately R262 million and an invasion at a low probability level (Scenario 3) could potentially cause a reduction in gross margin in the livestock sector of approximately R130 million. This suggests that the uncontrolled spread of *R. pseudoacacia* has the potential to have significant economic implications on the South African agricultural industry.

6.3.2 IMPACT OF *ROBINIA PSEUDOACACIA* ON LIVESTOCK IN THE GRASSLAND BIOME

As discussed previously, the Grassland biome is the most suitable for the establishment of *R. pseudoacacia*, as it has the potential to invade almost 20 million ha of the Grassland biome (based on probability values of 1-7) (see table 6.2). Figure 6.8 illustrates the potential distribution of *R. pseudoacacia* in the Grassland biome. A high potential distribution of *R. pseudoacacia* is located within the central and northern regions of the Grassland biome. The potential distribution on the outskirts of the eastern side of the biome is relatively low. The areas in white in the Grassland biome represent the land classes which are not suitable for the growth of *R. pseudoacacia*, based on the NLC (2009) (Figure 6.8).

[28] Gross margin per LSU of R1000 was assumed. See chapter 4, section 4.11 for more details on the gross margin.

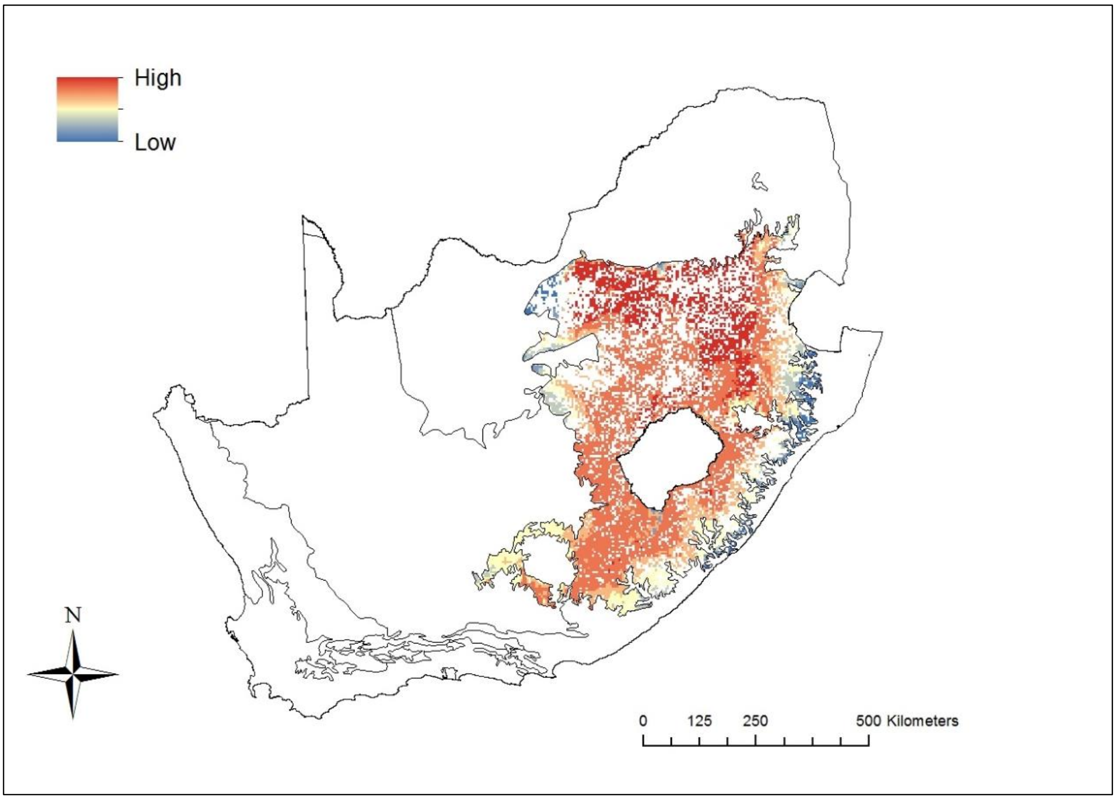

Figure 6.8. Potential distribution of *Robinia pseudoacacia* in the Grassland biome.

The Grassland biome is the second largest biome in South Africa, covering 29% of South Africa (Stephens & Tau, 2013) and is situated mainly in the central, high-lying regions of South Africa (O'Connor & Bredenkamp, 1997; Palmer & Anislie, 2006). In the grasslands of South Africa, mean annual rainfall is 600-1200 mm (Schulze & Lynch, 2007) and occurs predominantly in the spring and summer months. The biome harbours a rich species, community and ecosystem diversity (Reyers & Tosh, 2003), supplying essential ecosystem services and supporting crop and livestock agricultural activities (O'Connor & Kuyler, 2009). Thus, grasslands are central to the livelihoods and economies for both small-scale/communal farmers and commercial farmers (Boval & Dixon, 2012).

There are regional, national and global multifunctional uses of the Grassland biome (Boval & Dixon, 2012). These include water catchments, social and cultural needs for many rural societies, reducing greenhouse gas emissions (Soussana *et al.*, 2010), cultural and recreational needs (McDermott *et al.*, 2010; Thornton & Herrero, 2010), preservation of ecosystem biodiversity (O'Connor & Kuyler, 2009; DeFries & Rosenzweig, 2010) and as a feed base for grazing livestock (Boval & Dixon, 2012).

Livestock productivity per hectare is constrained by the amount of available forage matter (Boval & Dixon, 2012). The potential establishment of *R. pseudoacacia* into the Grassland biome poses a threat to grazing livestock. The Grassland biome is an important form of land use, as livestock farmers depend on grasslands for their livelihoods (Bouwman *et al.,* 2005; Suttie *et al.*, 2005).

Boval & Dixon (2012) stated “grasslands have been one of the foundations of human activities and civilizations by supporting production from grazing livestock.” Grasslands provide the feed base for grazing livestock as well as for other livestock related products, such as fertilizer and leather (Boval & Dixon, 2012). Furthermore, Carlier *et al.* (2009) believed that both wild and domesticated herbivores are dependent on pastoral rangelands. Much of the increasing demands for meat and dairy products are supplied from the Grassland ecosystems (Boval & Dixon, 2012). According to Boval & Dixon (2012), the Grassland biome is mainly used for grazing wild and/or domesticated herbivores, as this form of land use is the most appropriate in terms of economic utilisation of the land resource (Suttie *et al.*, 2005).

Figure 6.9 illustrates the potential grazing capacity of LSUs in the Grassland biome. The grazing capacity of LSUs in the eastern portion of the Grassland biome is greater than that of the central and western portions. Comparing the potential distribution of *R. pseudoacacia* (Figure 6.8) and the distribution of the grazing capacity of LSUs (Figure 6.5) in the Grassland biome, *R. pseudoacacia* invades predominantly throughout the central and northern regions of the Grassland biome, where the grazing capacities of LSUs is moderate to high.

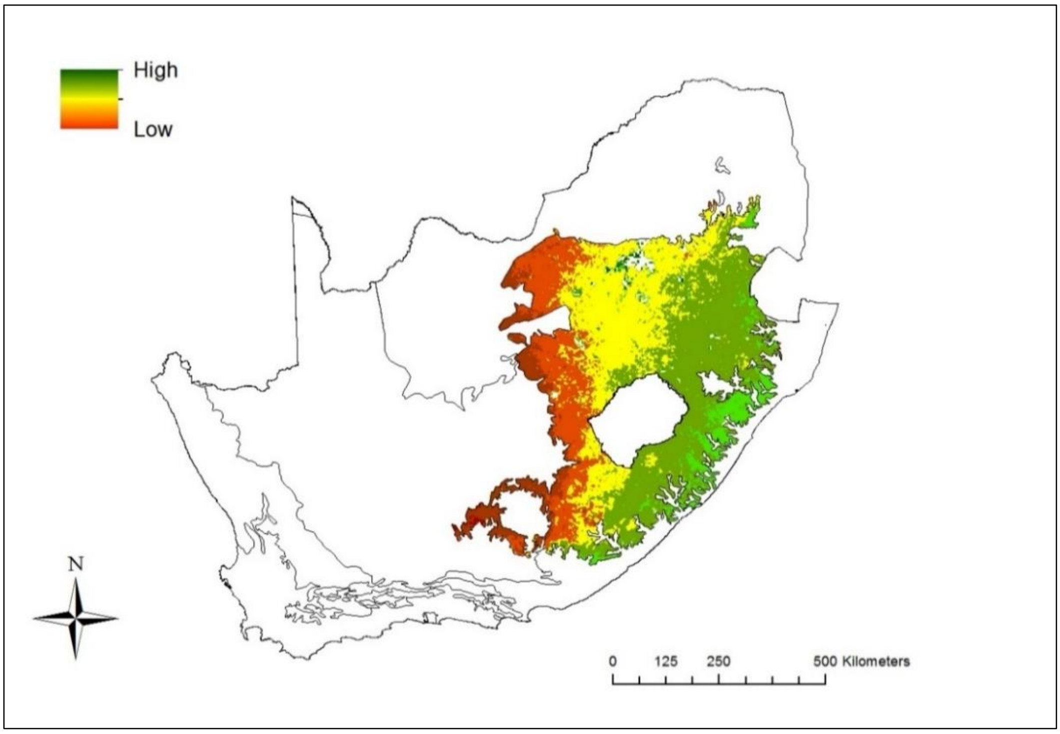

Figure 6.9. Potential grazing capacity of LSUs in the Grassland biome (ha/LSU).

As the Grassland biome has the largest number of LSUs and *R. pseudoacacia* has the highest probability of establishing itself in the Grassland biome, the impact of *R. pseudoacacia* in the Grassland biome was determined. To do this, the MAXENT potential distribution data in the Grassland biome was overlaid with the grazing capacity data (ARC, 2009) and was calculated and tabulated. Table 6.9 illustrates high (5-7) (Scenario 1), moderate (3-4) (Scenario 2) and low (1-2) (Scenario 3) probabilities of *R. pseudoacacia* invasions and the impact of these invasions on the number of LSUs in the Grassland biome. The same scenario canopy cover assumptions and percentage impacts on LSUs (Table 6.8) used in section (6.3.1) were assumed. The total number of LSUs that could potentially be impacted by an invasion of *R. pseudoacacia* in the Grassland biome at different invasion probabilities were estimated (Table 6.9).

Table 6.9. Number of LSUs potentially impacted by an invasion of *Robinia pseudoacacia* the Grassland biome, based on the relevant scenario assumptions.

Scenario	Probability of Invasion	Number of LSU Impacted
1	High (5-7)	741 462
2	Moderate (3-4)	67 592
3	Low (1-2)	11 005

Scenario 1 in Table 6.9 shows that - at a high probability of invasion - *R. pseudoacacia* has the potential to impact 741 462 LSUs in the Grassland biome. Moreover, the invasive tree has the potential to impact 67 592 and 11 005 LSUs at a moderate (Scenario 2) and at a low (Scenario 3) probability of invasion, respectively.

Figure 6.10 compares the number of LSUs potentially impacted by *R. pseudoacacia* in South Africa and in the Grassland biome. At a high probability of invasion, the Grassland biome contains approximately 77% of the total number of LSUs in South Africa. This further shows the importance of the Grassland biome for livestock. At moderate and low probabilities of invasion, the Grassland biome is not particularly vulnerable, housing only 26% and 8% of all LSUs in South Africa at moderate and low invasion probability levels, respectively.

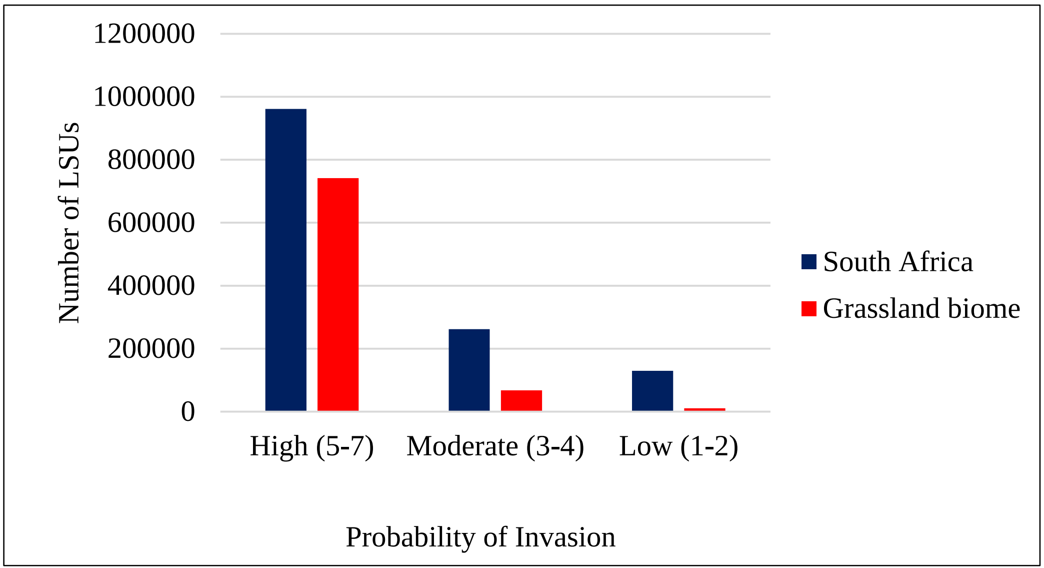

Figure 6.10. Number of LSUs potentially impacted by an invasion of *Robinia pseudoacacia* in South Africa and in the Grassland biome.

In monetary terms, the impact of the potential invasion of *R. pseudoacacia* at a high probability level on LSUs in the Grassland biome, could potentially cause a reduction in gross margin of in the livestock sector approximately R741 million[29]. It was estimated that an invasion at moderate and low probability levels could potentially cause a reduction in gross margin in the livestock sector of approximately R68 million and R11 million respectfully.

6.4 CONCLUSION

The environmental layers improved the accuracy of the model and were crucial for determining the overall economic implications. The results suggested that there are potentially 8.4 million LSUs in South Africa, of which, approximately 3 million are in the Grassland biome. Based on the MAXENT potential distribution results, the Grassland biome is the most vulnerable biome to invasion. In South Africa, the economic impacts range from a high probability of invasion, which has the potential to impact 961 359 LSUs, to a low probability of invasion, which has the potential to impact 129 687 LSUs. This equates to a reduction in gross margin in the livestock sector of between R961 million and R130 million, respectively. In the Grassland biome, 741 462 LSUs may potentially be impacted by the tree, at a high probability of invasion. In monetary terms, this could potentially cause a reduction in gross margin in the livestock sector of approximately R741 million. Overall, the results suggest that the potential

[29] Gross margin per LSU of R1000 was assumed. See chapter 4, section 4.11 for more details on gross margin.

distribution of *R. pseudoacacia* has the potential to have significant economic implications for the livestock sector in South Africa.

Chapter 7
DISCUSSION AND CONCLUSION

7.1 INTRODUCTION

The following chapter is divided into 4 main sections. The first section (7.2) provides a discussion of the results, putting them into context and identifying what they mean for IAP control policy in South Africa. Section (7.3) reviews the limitations and assumptions of the research methodology. Recommendations for future research are included in section (7.4), followed by a conclusion in section (7.5).

7.2 DISCUSSION

IAPs carry a host of environmental and socio-economic impacts (see chapter 2). Most of the existing literature on the impacts on IAPs focused on valuating their direct or indirect impacts on various ecosystem goods and services (Van Wilgen *et al.*, 2008; Pejchar & Mooney, 2009; De Wit *et al.,* 2012; Costanza *et al.,* 2014). A reasonable number of attempts have been conducted in order to determine economic consequences of the establishment of an IAP (Van Wilgen *et al.,* 2001a; Reinhardt *et al.,* 2003; Sinden *et al.,* 2004; Pimentel *et al.,* 2005; Colautti, 2006), but many of these studies were conducted on a broad scale. Only a few studies have looked at the economic impacts of an IAP on agricultural activities (Mullahey *et al*., 1994; Hirsch & Leitch, 1996; Leitch *et al.,* 1996; Bangsund *et al.,* 1999; Zavaleta, 2000; Eagle *et al.,* 2007). However, none of the studies pertaining to economic impacts of an IAP on agricultural activities were conducted in South Africa, nor did they specifically focus on the livestock sector. The depauperate literature on this topic is concerning as this study has shown the significant impact an IAP could potentially have on this important component of South African agriculture. Therefore, this study provides insight into estimating the potential direct impacts of *R. pseudoacacia* on agricultural in South Africa, specifically looking at the livestock sector. It also provides interesting information on the potential impact of invasive terrestrial tree species on agriculture in South Africa and the potential impact of poisonous species on agriculture.

In this study's analysis, the results from the first scenario (at a high probability of invasion) suggest that *R. pseudoacacia* has the potential to reduce the gross margin of agriculture by approximately R961 million (see chapter 6). In scenario 3, at a low probability of invasion, *R.*

pseudoacacia has the potential to reduce gross margin by approximately R130 million. The results indicate that potential distribution of *R. pseudoacacia* has the potential to have significant direct economic implications on the livestock sector in South Africa. The survey results from the case study further confirmed this (see chapter 5). According to DAFF (2014), the gross value[30] of beef production in 2012/13 was approximately R18 billion. Therefore, while the impacts are relatively small within the national agricultural production system, losses and costs due to *R. pseudoacacia* infestations do constrain South Africa's livestock grazing sector. It is to be noted that DAFF (2014) measure the value of beef production in terms of gross value, while this study looked at gross margin. While, a direct comparison of the impact of *R. pseudoacacia* on the total gross value of the livestock sector is difficult to determine, it is still expected to be significant.

Compared to related existing literature, this study is in line with other studies, albeit with differences in methods (Leitch *et al*., 1996; Hirsch & Leitch, 1996; Pimentel *et al.*, 2000; Eagle *et al.*, 2007) (see chapter 2, section 2.6 for more detail). They suggest that IAPs, including *R. pseudoacacia*, have the potential to have substantial impacts on the agricultural sector.

The economic theory of IAPs suggests that IAPs should be treated as economic externalities. The study found that the spread of *R. pseudoacacia* imposes potential costs to the economy, particularly in the agricultural sector. The costs of the invasion are felt by the wider economy, (producers, consumers etc.), and not by the individual or company who introduced *R. pseudoacacia* to South Africa (Parks & Gowdy, 2013).

Approximately 60% of cattle available in South Africa are owned by commercial farmers and 40% by emerging and communal farmers (DAFF, 2014). Therefore, there are a larger number of livestock units not being sold in the formal market. Thus, assigning a gross margin of R1000 per LSU in this study is an under-estimation of the value of livestock as this only represents the value of livestock in the formal sector and not in the informal sector, to which it is more valuable (see chapter 2, section 2.7 for a detailed discussion on the value of livestock in South Africa).

Furthermore, it is estimated that the grazing capacity data (ARC, 2009) is an under-estimation of the total number of LSU in South Africa as it is based on available vegetation (Palmer, 2015

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[30] Gross value of beef production is dependent on the number of cattle slaughtered and the prices received by producers from abattoirs (DAFF, 2014).

pers comms). It was further reported that the farmers in the informal sector are carrying a lot more LSUs per ha (overstocking)[31] than the grazing capacity data (ARC, 2009) suggests. In the communal and emerging farm lands, Palmer (2015 pers comms) reported that the actual number of LSU per ha may be double the amount predicted by the grazing capacity data (ARC, 2009)[32]. Thus, with more potential LSUs than previously estimated in the study, the results produced in this study may represent an under-estimation of the number of livestock in South Africa and hence, an under-estimation of the potential economic impact of the establishment of *R. pseudoacacia.*

7.2.1 A SOLUTION TO THE PROBLEM

The spread of *R. pseudoacacia* has the potential to cause extensive damage to the agricultural sector, specifically to livestock as seen in the study. Although this study only looked at one element - the impact on grazing capacity - the potential economic impacts are significant. There remains a role for public intervention to control *R. pseudoacacia,* as this will yield public benefits for a diverse array of other natural resource service flows negatively impacted by *R. pseudoacacia* (Eagle *et al.,* 2007). Thus, in order to prevent the potential negative impacts from occurring, a solution to the problem is needed.

Mechanical and herbicidal control methods have proven to be unsuccessful, as seen in the literature (Brown *et al.,* 2001; Edgin, 2007; Cierjacks *et al.,* 2013) (see chapter 3, section 3.4) and in the case study, where in total, approximately R9 million was spent attempting to control *R. pseudoacacia* in the eastern Free State (see chapter 5). The costs of control rise exponentially as each control attempt only aggravates the spread. Mechanical control methods result in prolific root suckering (Zimmerman, 1984) and clonal spread (Czarapata, 2005) and no complete or long-term herbicidal solution exists (DeLoach, 1997; Weitzenberg *et al.,* 1997; Sabo, 2000; Edgin, 2007; Cierjacks *et al.,* 2013). However, one control option which has not yet been attempted is biological control.

Biological control is considered to be the most environmentally friendly, cost effective and self-sustaining control method used to supress IAPs (Zimmermann *et al.,* 2004). This method

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[31] Furthermore, overstocking of LSUs inevitably leads to overgrazing and veld degradation. Todd *et al.* (2009) found that overgrazing was identified as the primary threat to biodiversity and ecosystem functioning in the Namakwa District, Northern Cape, South Africa.

[32] However, the potentially suitable area for the establishment of *R. pseudoacacia* which overlaps with small-scale farmers rangelands is unknown.

of control has been used as a powerful tool for reducing the costs of management of IAPs worldwide as well as in South Africa (Van Wilgen *et al.,* 2004). A variety of control agents have been released on a variety of IAPs in South Africa which have proven to be successful (Olckers & Hill, 1999; Cruttwell McFadyen, 2000; Van Wilgen *et al.,* 2004). Potential biological control agents exist for *R. pseudoacacia* (see chapter 3, section 3). The implementation of biological control has the potential to reduce the spread of *R. pseudoacacia.*

7.2.2 COSTS OF BIOLOGICAL CONTROL

Economic analyses of biological control programs are a valuable input into the decision-making process for biological control programmes[33] (Jetter, 2005). McFayden (2008) indicates the importance of performing economic impact assessments in order to identify and measure all research and research-related costs from all programmes undertaken, and to assess this against all benefits gained. This provides the true probability of a positive return on investment.

A successful biological control programme in South Africa was the release of three biological control agents on Red sesbania (*Sesbania punicea*) (see chapter 2, section 2.8.3). Red sesbania falls within the same subfamily of *R. pseudoacacia,* Faboideae. Furthermore, the biological control agents that were released on Red sesbania in South Africa are similar to the potential biological agents for *R. pseudoacacia.* Therefore, the costs of biological control for *R. pseudoacacia* can be expected to be similar to that of Red sesbania (R3 million), making the benefit (avoiding the potential impacts of *R. pseudoacacia* on the livestock sector) substantially larger than the estimated costs of control. Based on the biological control costs of Red sesbania and the results found in this study, the implementation of a biological control programme for *R. pseudoacacia* would have a benefit: cost of between 43:1 to 320:1 at a low probability of invasion (scenario 3)[34] and at a high probability of invasion (scenario 1)[35] respectively, although this has not been proven by means of a full benefit: cost analysis. However, it provides an indication of the potential significant benefits of biological control. The magnitude of the potential economic benefits suggests that the benefit: cost ratio could be quite favourable. Also, the cost of biological control is a lot cheaper when compared to mechanical and herbicidal control methods (Marais *et al*., 2004). In the case study, total mechanical and herbicidal control

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[33] For a more detailed analysis on using economic analyses for determining decision making for biological control programmes, see Jetter (2005).
[34] Benefit R130 million: Cost R3 million.
[35] Benefit R961 million: Cost R3 million.

costs were approximately R9 million for one site in the eastern Free State region (see chapter 5). Therefore, in addition to the success of biological control programmes, the implementation of biological control also reduces the costs of other control management practices (Zimmermann *et al.,* 2004).

7.2.3 SOCIO-ECONOMIC IMPACT

The agricultural sector in South Africa is responsible for supplying local and global commodities, is an important employer of labour, has a significant influence on alleviating food scarcity and provides foreign exchange earnings (Nieuwoudt *et al.,* 2004; Eagle *et al.,* 2007; Republic of South Africa, 2013). Therefore, the potential impact of *R. pseudoacacia* on the livestock sector is significant, especially if secondary or indirect impacts are considered.

DAFF (2014) estimated that there are approximately 50 000 commercial farmers, 240 000 emerging farmers and 3 million communal farmers in South Africa. Furthermore, there are approximately 70 feedlots in South Africa and 495 abattoirs. The beef industry is a major employer with 500 000 people employed and 2 125 000 dependent on the livestock sector for their livelihood (DAFF, 2014). Therefore, a reduction in the number of livestock would have significant impacts on all parties mentioned above. Many farmers, abattoirs and their employee's income would be reduced significantly and many of them may even be forced out of business.

LID (1999) estimated that two-thirds of resource-poor rural households keep some type of livestock. Furthermore, Meissner *et al.* (2013) estimated that approximately 10-12 million dependants of small-scale and communal farmers receive partial sustenance from livestock-based food, clothing and decorative materials. Livestock is therefore important to rural communities as a source of producing food, generating income, providing manure, traction and transport, serving as financial aids and enhancing social status (Meissner *et al.,* 2013). The potential reduction in livestock, as a result of a *R. pseudoacacia* invasion, would have the most significant impact on poor communities, as livestock forms an integral and indispensable part of social life and sustenance (Meissner *et al.,* 2013).

7.3 RESEARCH LIMITATIONS AND ASSUMPTIONS

The results reported in the previous chapter need to be understood within the context of the research limitations and assumptions. The research limitations and assumptions exist due to a number of research challenges encountered. The economic impact assessment required at least plausible approximations of some of the complex biophysical phenomena pertaining to *R. pseudoacacia* infestations (Leitch *et al.*, 1996). The relationships and models used in this study lack direct empirical foundations but they were necessary for the broad level assessment of the tree's economic impact.

The direct economic impact of *R. pseudoacacia* was only estimated for grazing livestock (LSU) in South Africa, based on ARC (2009) grazing capacity data. The potential impact on ruminant wildlife (e.g. antelope species) or other agricultural activities (Leitch *et al.*, 1996) was not considered for this study. Therefore the economic losses estimated in this study are strictly limited to reductions in grazing opportunities for domestic livestock. If this study was to be validated empirically, it would require extensive resources to complete (data collection, analysis, and documentation over several years) (Leitch *et al.*, 1996). However, due to limitations and the scope of this study, the goal was produce plausible estimates of the direct agricultural economic impacts of *R. pseudoacacia.*

Due to limitations (limited available data on the exact number of livestock in South Africa) and the scope of this study, only the direct economic impacts were taken into account. No estimates of secondary impacts on regional economies of reduced grazing activity due to *R. pseudoacacia* were considered (Eagle *et al.*, 2007). The estimates should therefore not be construed as representing the total economic impacts of *R. pseudoacacia* caused by its negative effects on grazing.

Grazing land was defined as all land used for grazing of domestic livestock, including all natural land and degraded land classes of the NLC (2009). It was also assumed that land which was invaded by *R. pseudoacacia* could not be utilised for the grazing of livestock.

It was assumed that the spread of the tree into grazing lands would decrease grazing capacity. *Robinia pseudoacacia* reduces the carrying capacity of livestock in two ways: firstly, infestations restrict access to grazing lands through encroachment, which decreases the amount of available forage for livestock and interferes with grazing patterns; secondly, the seeds, leaves and bark of the tree are toxic to livestock. Therefore, a reduction in the carrying capacity of livestock disrupts agricultural production, output and profitability.

Limited information exists with concern to carrying capacities in South Africa. ARC (2009) grazing capacity data, which is based on satellite imagery and NPP, was therefore assumed to be accurate and could produce plausible estimates. Various methods of estimating the grazing capacity in South Africa were discussed with Dr Palmer, who recommended that using the ARC (2009) grazing capacity data was the most accurate method.

The potential distribution results of *R. pseudoacacia* were based solely on MAXENT and climatic variables. Richardson & Van Wilgen (2004) believed that in order to understand invasion risk, additional factors need to be considered. This study, however, did not take into account other variables which could have possibly influenced the potential distribution (e.g. soil type, slope etc.). MAXENT has been ranked as the best SDM programme (Elith *et al.,* 2006; Tsoar *et al.,* 2007; Wisz *et al.,* 2008) and within the scope of this thesis, was deemed adequate in order to predict the potential distribution of *R. pseudoacacia* in South Africa. It was assumed the relevant MAXENT distribution probabilities would hold for future infestation levels, and it was assumed that the NLC (2009) data would also hold for future infestation levels.

Due to canopy cover data not being available for *R. pseudoacacia*, three canopy cover invasion scenarios were created. The scenarios were based on guidelines developed by Le Maitre & Versfeld (1994), which provide for a range of density classes from rare (<0.01%) to closed (100% canopy cover), and in the case study which provided insight into the size of *R. pseudoacacia* infestations. Experts within the field of IAPs were approached and with their input the canopy cover scenario assumptions were developed. The impact of *R. pseudoacacia* on LSUs for the three scenarios was estimated by an agricultural economist.

An average gross margin per LSU of R1000 was assumed. Gross margins for livestock were obtained from the livestock enterprise budget, compiled by VKB in the Eastern Free State.

7.4 RECOMMENDATIONS FOR FUTURE RESEARCH

Understanding the research limitations discussed in this chapter provides a platform for future research in this field. It is important to note that *R. pseudoacacia* tends to predominantly invade and occupy ecological niches, such as the grassland areas. These areas typically offer relatively low per-hectare values when compared to that of more productive agricultural lands. For this study, it was assumed that *R. pseudoacacia* has the potential to only have an impact on the carrying capacity of livestock. Therefore, the potential monetary losses incurred may be greater for more productive agricultural lands. It is important to remember that this study focused on the impact of *R. pseudoacacia* on the livestock sector. Therefore, one must keep in mind that the monetary losses of other negative impacts of *R. pseudoacacia* may be greater than those examined here. These impacts may include other ecological impacts (for example, increased soil erosion, runoff of nutrients, losses in biodiversity etc.), as well as depressed recreational activities (for example hiking or hunting) (Eagle *et al.*, 2007). These have not been documented to date for *R. pseudoacacia* and that represents a useful next step in research.

Determining a variety of economic effects, whether it be ecological impacts or recreational activity impacts, allows for further research into available future control methods. Such information may be useful for allocating resources to develop or refine new control technologies (Leistritz, 1993).

Refining the research methodology used within this particular study would improve the accuracy of the results. This includes the use of extensive resources (data collection, analysis, and documentation over several years) and more precise description of the physical relationships between infestations and agricultural land uses, as well as more detailed estimates of the cover/density on which an infestation occurs. A more in-depth analysis of this study would allow for a better understanding of the economic impact *R. pseudoacacia* on the livestock sector.

Furthermore, it is recommended that economic valuations of a similar nature should be conducted in South Africa for other IAPs, which could provide valuable information as to where the greatest externalities lie. Private and government funds are available to help minimise these negative economic consequences, although these funds are limited. The economic valuations of such projects provide the necessary motivation for the allocation of these scarce funds (Turpie, 2004). Jetter (2005) stated that policy decisions are based on economic criteria, which in turn depend on the probability of success. Thus, an economic evaluation is essential in verifying the viability of a project and thereby attracting investment. It is recommended that this thesis forms the bases for future research.

7.5 CONCLUDING REMARKS

The agricultural sector is of vital importance to South Africa. IAPs impose externalities on the agricultural sector and can ultimately threaten South Africa's economic growth prospects. The potential invasion of *R. pseudoacacia* can have detrimental effects on the grazing capacity of livestock in South Africa. The threat is especially concerning as *R. pseudoacacia* has the potential to predominately invade the grassland biome, which houses a large concentration of livestock. This poses as a serious problem for farmers and other parties that are dependent upon the livestock sector. Livestock farming plays an enormous role in providing sustenance to poor communities and stabilising the economies of towns in non-metropolitan areas.

The results suggest that the potential effects of the IAP on the productivity of grazing lands can be significant. The potential economic impact of *R. pseudoacacia* on the livestock sector in South Africa is substantial, with direct negative economic impacts, in terms of gross margin, ranging between approximately R130 million and R961 million. While the estimates are based on a variety of assumptions and limitations and are not definitive, they certainly demonstrate the potential magnitude of economic impacts.

Evans (2003: 9) stated that "the true value of economics should therefore not be seen solely in the precision of the numbers generated, albeit this is important, but the extent to which the discipline aids decision makers to formulate consistent and rational decisions." The potential high levels of foregone income and business activity found in this study reaffirm the need to devote resources to develop a viable, economical and effective control method (Bangsund, 1991). Invasions carry a large potential impact on the economy and controlling them will produce the greatest benefit to society. Both existing literature and the case study suggest that mechanical and herbicidal controls methods for *R. pseudoacacia* are unsuccessful. One method of control for *R. pseudoacacia* which has not yet been explored in South Africa is that of biological control. Biological control success in other countries, the absence of indigenous *Robinia* species in South Africa, availability of biological control agents and potential benefits from control suggest that a biological control project against *R. pseudoacacia* is potentially a viable control option. Furthermore, the chances of successful biological control are greatly increased if an IAP species is targeted at an early stage of invasion (Olckers, 2004; Henderson, 1999), indicating that the potential returns biological control could be substantial.

Economic valuations of this kind afford important insight into how scarce funds and resources can be allocated more effectively, providing another step towards improved alien plant control. Implications for both policy makers and researchers can be drawn from this effort to estimate the economic impacts of *R. pseudoacacia*. Insight and awareness for policy makers has been provided with regards to the economic implications of the current and potential situations. Implementing more effective control measures should be an issue of concern to policymakers generally, rather than just to those representing the livestock sector.

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APPENDIX

1. Questionnaire

RHODES UNIVERSITY
Where leaders learn

•LUKE HUMPHREY • DEPARTMENT OF ECONOMICS AND ECONOMIC HISTORY •

•Tel: +27 83 646 8878 • Email: lukeh192@hotmail.com •

Questionnaire

My name is Luke Humphrey and I am a Masters degree student at Rhodes University. This questionnaire is being conducted as part of a study in the Department of Economics and Economic History. The title of the thesis is **"the potential economic implications of Black Locust (*Robinia pseudoacacia L.*) on agricultural production in South Africa, using the Clarens region as a case study."** Black Locust is a prominent invader in South Africa, particularly in the Clarens region. Due to Black Locust's invasive characteristics, it has the potential to invade agricultural land and disrupt agricultural activities-which poses as a problem for managers of grazing land. The threat presented by Black Locust is the establishment and spread of the invasive alien plant on rangelands, potentially resulting in an unnecessary loss in agricultural production, output and profitability.

All the information acquired will be used in the case study section of the thesis. This questionnaire is designed to gather information pertaining to the infestation of Black Locust, management and control of Black Locust and the effects of Black Locust on agricultural activities, within the Clarens region.

All participants will remain anonymous and the data will not be used without consent for any research other than what is indicated. Feedback will be provided where indicated.

All information collected is purely for academic purposes. Due to participation being voluntary, please indicate below that you have read and understood the terms of the survey and are willing to participate in this survey. You can choose not to answer any of the questions and can end the interview at any stage.

The questionnaire is divided into 4 categories: (1) Farm details (2) Plant identification (3) Perceptions (4) Financial effects (5) Other.

After reading the conditions of this questionnaire, I consent to answering the questionnaire.

Signed: ____

☐ Please tick if you would like to receive feedback from this study and provide your email:

____

A. Farm Details (for administrative purposes and in case follow-up for clarification is needed)

FARM OWNER		
CONTACT DETAILS (optional)	Number: Email:	
FARM NAME		
AREA WITHIN CLARENS		
YEARS OF OWNERSHIP		
SIZE OF FARM (HA)		
TYPE OF FARMING (e.g. Livestock, citrus)		Grazing Capacity: __________ ha/LSU

B. Plant Identification

1. Can you identify Black Locust (*Robinia pseudoacacia* L.)?

____

C. Perception

1. Have you noticed this tree on your farm?

Yes ☐ No ☐

2. When did you first notice it?

____

3. Where do you think it came from?

____

____

4. What is the approximate area infested by Black Locust on your farm?

____

____

____

5. How would you compare the presence of Black Locust with other invasive species within the area?

____

____

____

6. How has the tree population increased?

-Increased and still increasing at a very high rate ☐

-Increased but of late the rate has slowed down ☐

-Not increasing ☐

-Decreased ☐

7. Do you regard Black Locust as problematic?

Yes ☐ No ☐

If yes, please list the perceived problems below:

- ____
- ____
- ____
- ____
- ____

8. Do you think the presence of Black Locust brings any advantages?

Yes ☐ No ☐

If yes, please list the advantages below:

- ____
- ____
- ____
- ____

9. Have you noticed Black Locust elsewhere within the Clarens region? If so, where?

____

____

____

10. Do you think the spread of Black Locust should be controlled?

____

____

____

D. Financial effects

1. What types of costs have you incurred as a result of the tree infestation? E.g. livestock/restricting farm access/water quantity.

- How much of an effect have these costs had on agricultural production per annum?

2. Have you ever attempted to eradicate/control Black Locust? If so, what types of control have you performed?

- What was the financial cost of these controls?

E. Other

Thank you very much for your time, any other comments are welcome?

2. Bioclimatic Predictor Variables

BIO1 = Annual Mean Temperature

BIO2 = Mean Diurnal Range (Mean of monthly (max temp - min temp))

BIO3 = Isothermality (BIO2/BIO7) (* 100)

BIO4 = Temperature Seasonality (standard deviation *100)

BIO5 = Max Temperature of Warmest Month

BIO6 = Min Temperature of Coldest Month

BIO7 = Temperature Annual Range (BIO5-BIO6)

BIO8 = Mean Temperature of Wettest Quarter

BIO9 = Mean Temperature of Driest Quarter

BIO10 = Mean Temperature of Warmest Quarter

BIO11 = Mean Temperature of Coldest Quarter

BIO12 = Annual Precipitation

BIO13 = Precipitation of Wettest Month

BIO14 = Precipitation of Driest Month

BIO15 = Precipitation Seasonality (Coefficient of Variation)

BIO16 = Precipitation of Wettest Quarter

BIO17 = Precipitation of Driest Quarter

BIO18 = Precipitation of Warmest Quarter

BIO19 = Precipitation of Coldest Quarter

3. MAXENT Trial Results: Analysis of variable contributions

The following table gives a heuristic estimate of relative contributions of the environmental variables to the Maxent model. To determine the estimate, in each iteration of the training algorithm, the increase in regularized gain is added to the contribution of the corresponding variable, or subtracted from it if the change to the absolute value of lambda is negative. As with the jackknife, variable contributions should be interpreted with caution when the predictor variables are correlated.

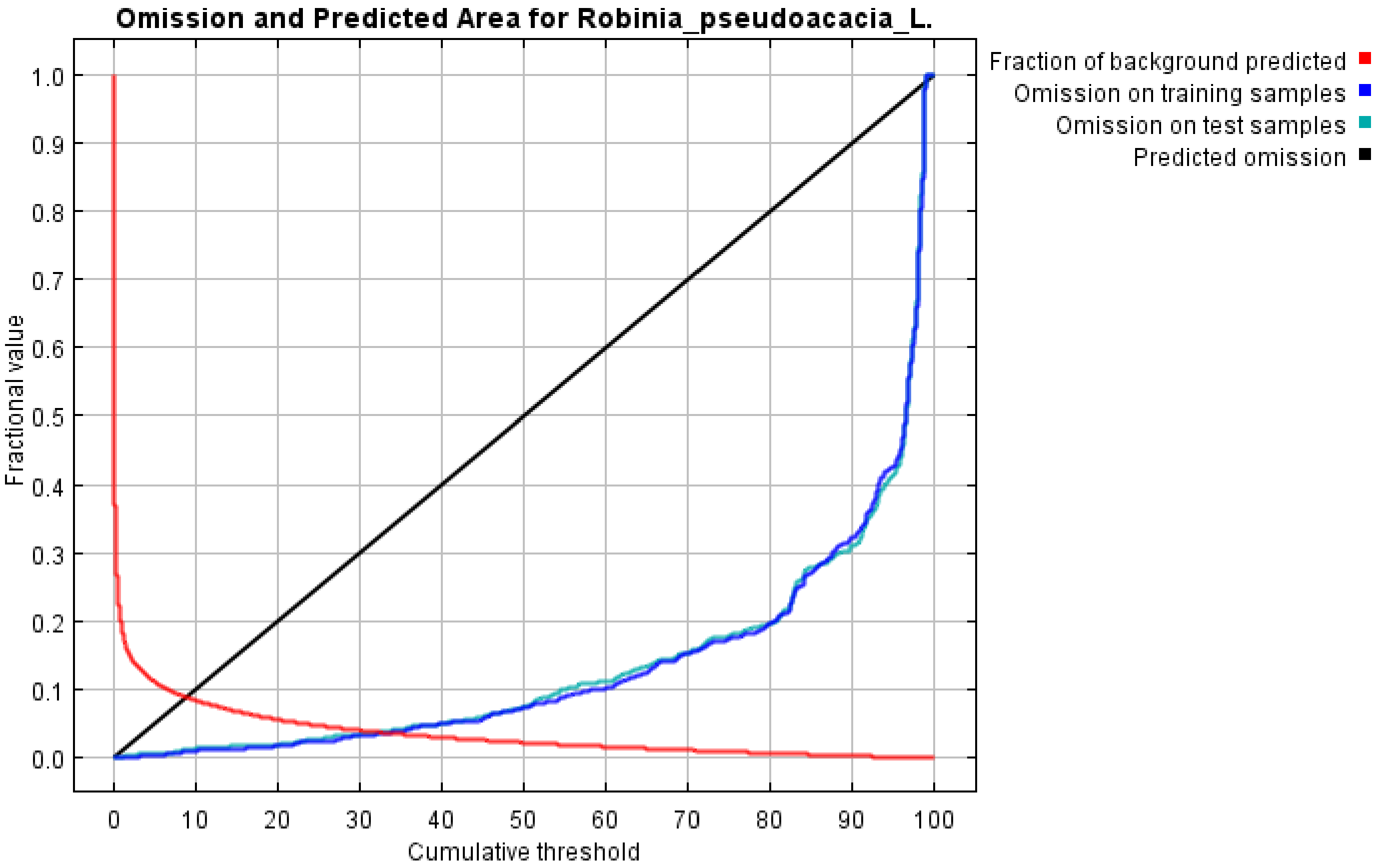
Variable	Percent contribution
bio_1_AMT	32
bio_15_PSEAS	17.4
bio_19_PRECOLDQUART	12.6
bio_10_MTWARMQ	10.1
bio_7_TAR	7.4
bio_11_MTCQ	6.7
bio_6.MTCM	5
bio_4_TS	2.9
bio_12._ANNPREC	2.2
bio_16_PWQAURT	1
bio_5.MTWM	0.8
bio_2_MDR	0.7
bio_8_MTWQ	0.4
bio_13.PWM	0.3
bio_14_PDM	0.3
bio_9_MTDQ	0.1
bio_18_PREWARMQUART	0.1
bio_17_PREDRYQUART	0
bio_3_ISM	0

Maxent model for Robinia_pseudoacacia_L.

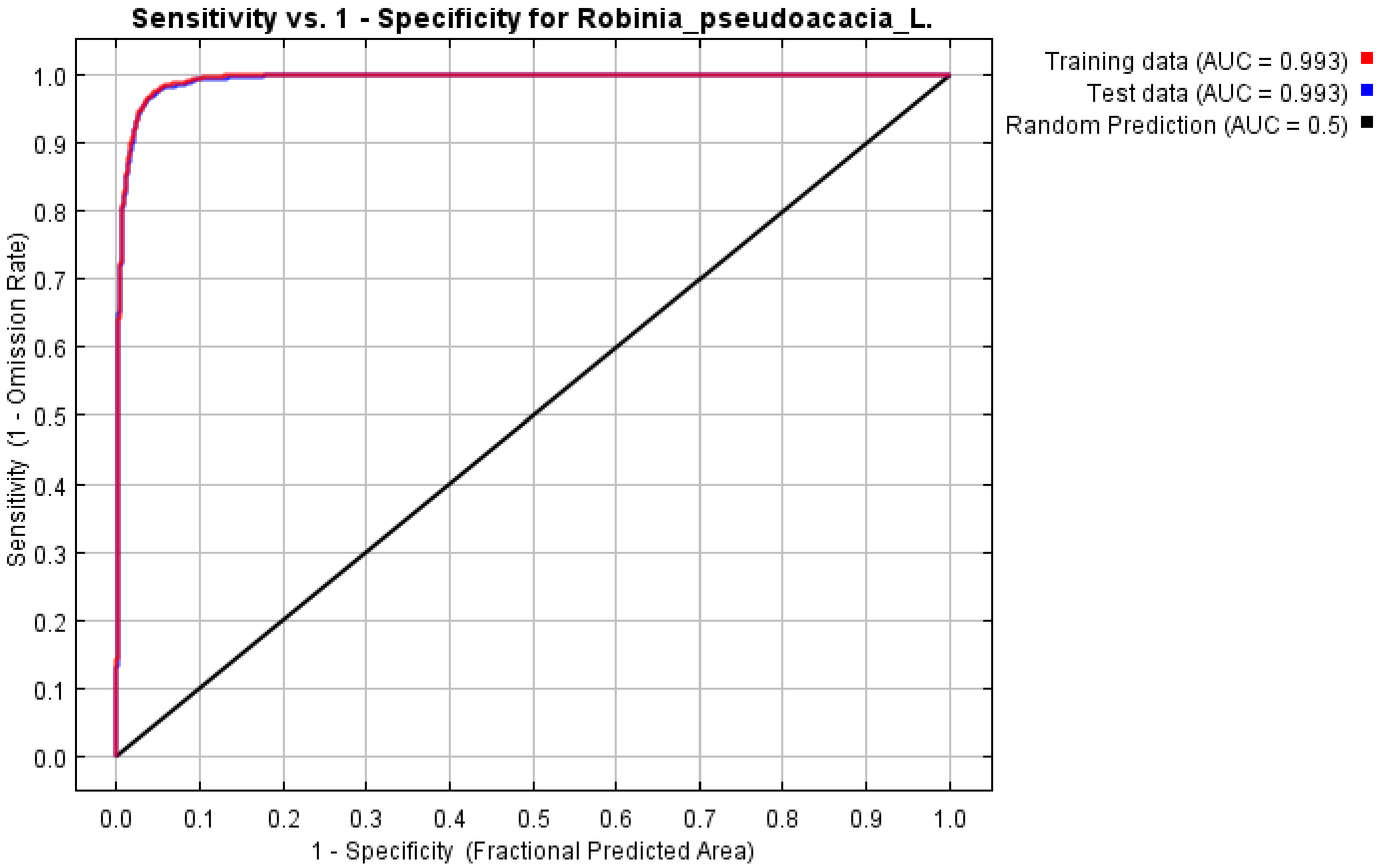
This page contains some analysis of the Maxent model for Robinia_pseudoacacia_L., created Wed Jun 10 14:48:59 CAT 2015 using Maxent version 3.3.0. If you would like to do further analyses, the raw data used here is linked to at the end of this page.

Analysis of omission/commission

The following picture shows the omission rate and predicted area as a function of the cumulative threshold. The omission rate is is calculated both on the training presence records, and (if test data are used) on the test records. The omission rate should be close to the predicted omission, because of the definition of the cumulative threshold.



The next picture is the receiver operating characteristic (ROC) curve for the same data. Note that the specificity is defined using predicted area, rather than true commission (see the paper by Phillips, Anderson and Schapire cited on the help page for discussion of what this means). This implies that the maximum achievable AUC is less than 1. If test data is drawn from the Maxent distribution itself, then the maximum possible test AUC would be 0.964 rather than 1; in practice the test AUC may exceed this bound.



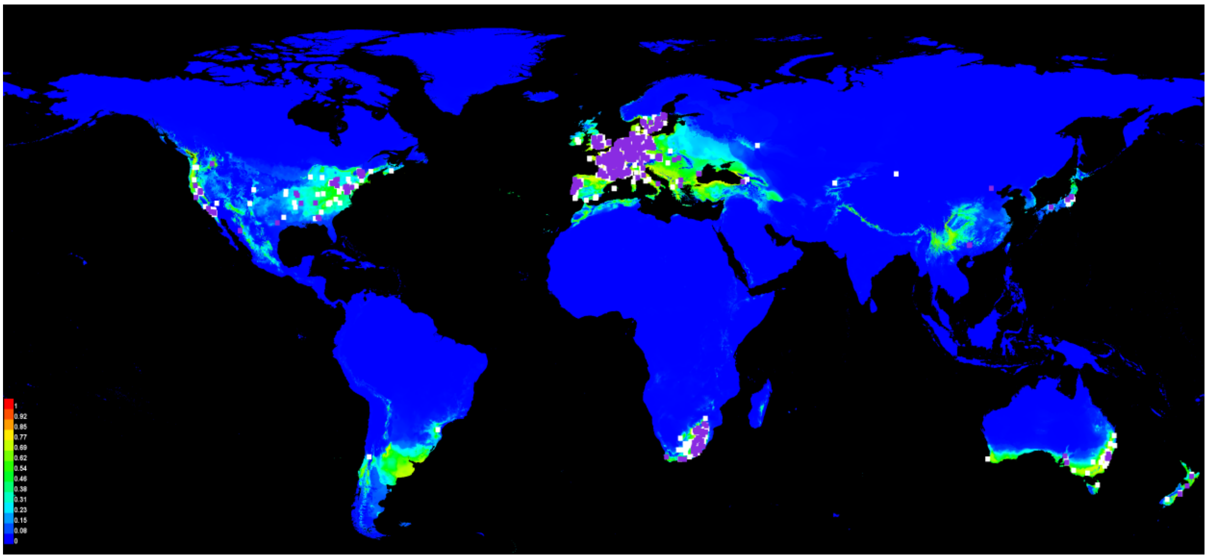
Some common thresholds and corresponding omission rates are as follows. If test data are available, binomial probabilities are calculated exactly if the number of test samples is at most 25, otherwise using a normal approximation to the binomial. These are 1-sided p-values for the null hypothesis that test points are predicted no better than by a random prediction with the same fractional predicted area. The "Balance" threshold minimizes 6 * training omission rate + .04 * cumulative threshold + 1.6 * fractional predicted area.

Cumulative threshold	Logistic threshold	Description	Fractional predicted area	Training omission rate	Test omission rate	P-value
1.000	0.014	Fixed cumulative value 1	0.188	0.001	0.001	0E0
5.000	0.105	Fixed cumulative value 5	0.112	0.003	0.006	0E0
10.000	0.205	Fixed cumulative value 10	0.085	0.010	0.014	0E0
0.541	0.006	Minimum training presence	0.233	0.000	0.001	0E0
59.457	0.636	10 percentile training presence	0.017	0.100	0.113	0E0

33.188	0.473	Equal training sensitivity and specificity	0.037	0.037	0.039	0E0
27.012	0.403	Maximum training sensitivity plus specificity	0.045	0.026	0.033	0E0
32.460	0.461	Equal test sensitivity and specificity	0.038	0.036	0.038	0E0
22.000	0.345	Maximum test sensitivity plus specificity	0.052	0.020	0.021	0E0
1.796	0.035	Balance training omission, predicted area and threshold value	0.154	0.002	0.003	0E0
7.226	0.153	Equate entropy of thresholded and original distributions	0.097	0.007	0.008	0E0

Pictures of the model

This is a representation of the Maxent model for Robinia_pseudoacacia_L.. Warmer colors show areas with better predicted conditions. White dots show the presence locations used for training, while violet dots show test locations. Click on the image for a full-size version.

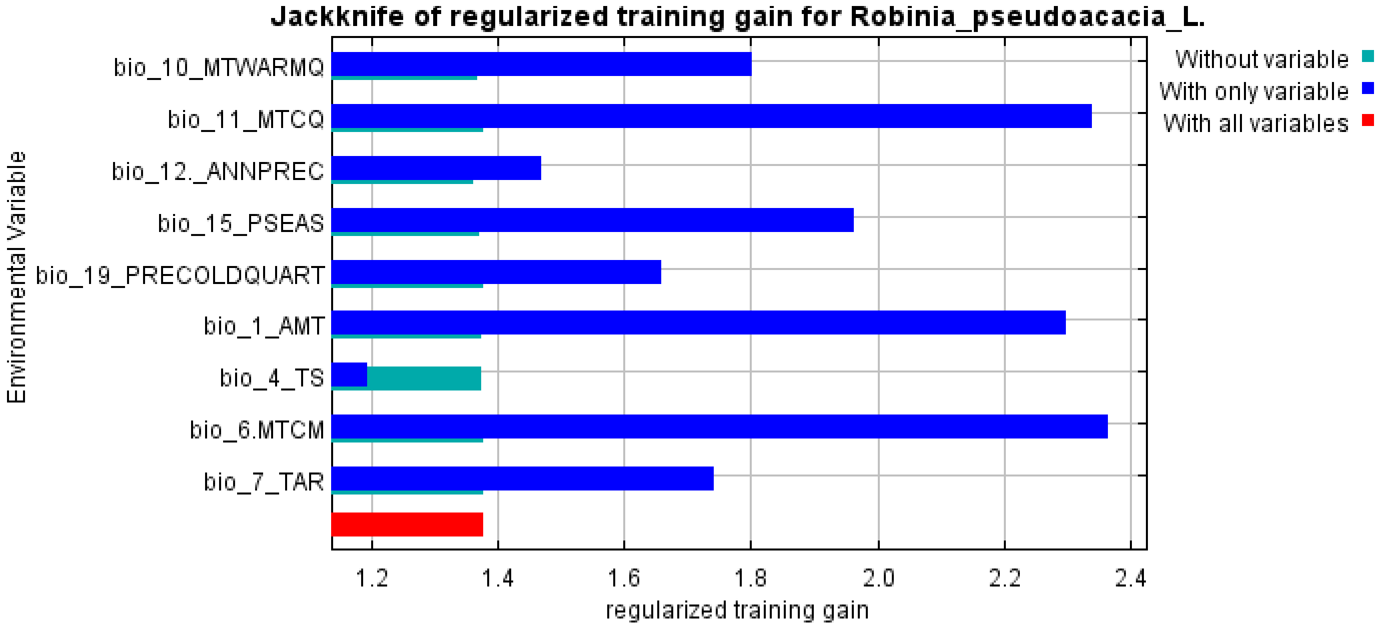


Analysis of variable contributions

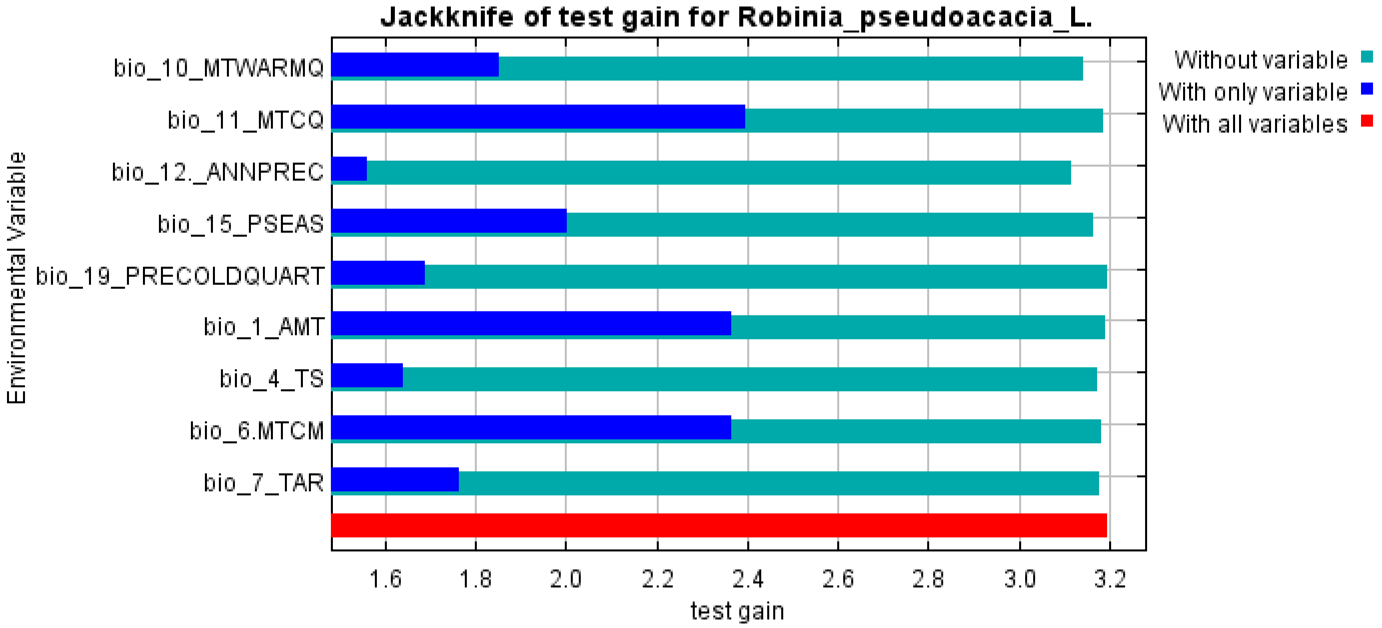
The following table gives a heuristic estimate of relative contributions of the environmental variables to the Maxent model. To determine the estimate, in each iteration of the training algorithm, the increase in regularized gain is added to the contribution of the corresponding variable, or subtracted from it if the change to the absolute value of lambda is negative. As with the jackknife, variable contributions should be interpreted with caution when the predictor variables are correlated.

Variable	Percent contribution
bio_1_AMT	32.3
bio_15_PSEAS	17
bio_19_PRECOLDQUART	13
bio_10_MTWARMQ	11.1
bio_11_MTCQ	7.9
bio_7_TAR	6.6
bio_6.MTCM	5.9
bio_12._ANNPREC	4
bio_4_TS	2.2

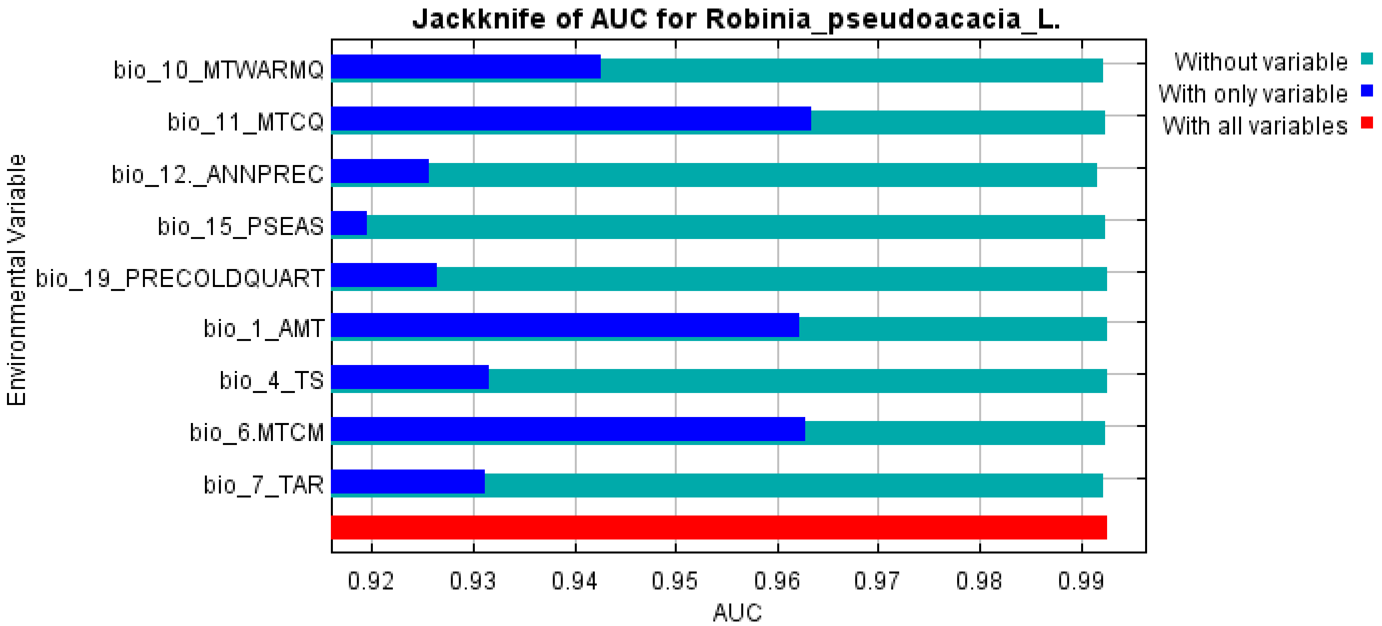
The following picture shows the results of the jackknife test of variable importance. The environmental variable with highest gain when used in isolation is bio_6.MTCM, which therefore appears to have the most useful information by itself. The environmental variable that decreases the gain the most when it is omitted is bio_12._ANNPREC, which therefore appears to have the most information that isn't present in the other variables.



The next picture shows the same jackknife test, using test gain instead of training gain. Note that conclusions about which variables are most important can change, now that we're looking at test data.



Lastly, we have the same jackknife test, using AUC on test data.



Raw data outputs and control parameters

The data used in the above analysis is contained in the next links. Please see the Help button for more information on these.
The model applied to the training environmental layers

The coefficients of the model
The omission and predicted area for varying cumulative and raw thresholds
The prediction strength at the training and (optionally) test presence sites
Results for all species modeled in the same Maxent run, with summary statistics and (optionally) jackknife results

Regularized training gain is 1.378, training AUC is 0.993, unregularized training gain is 1.405.
Unregularized test gain is 3.195.
Test AUC is 0.993, standard deviation is 0.001 (calculated as in DeLong, DeLong & Clarke-Pearson 1988, equation 2).
Algorithm terminated after 500 iterations (13 seconds).

The follow settings were used during the run:
2504 presence records used for training, 1073 for testing.
12502 points used to determine the Maxent distribution (background points and presence points).
Environmental layers used (all continuous): bio_10_MTWARMQ bio_11_MTCQ bio_12._ANNPREC bio_15_PSEAS bio_19_PRECOLDQUART bio_1_AMT bio_4_TS bio_6.MTCM bio_7_TAR
Regularization values: linear/quadratic/product: 0.050, categorical: 0.250, threshold: 1.000, hinge: 0.500
Feature types used: hinge product linear threshold quadratic
responsecurves: true
jackknife: true
outputformat: logistic
outputdirectory: C:\Users\Luke\Desktop\Luke\MAXENT\Maxent\Test 2
samplesfile: C:\Users\Luke\SkyDrive\Test 1.csv
environmentallayers: C:\Users\Luke\Desktop\Luke\MAXENT\env_layers(ASCII)
randomtestpoints: 30
replicatetype: crossvalidate
Command line used:

Command line to repeat this species model: java density.MaxEnt -r -a nowarnings noprefixes -E "" -E Robinia_pseudoacacia_L. responsecurves jackknife outputformat=logistic outputdirectory=C:\Users\Luke\Desktop\Luke\MAXENT\Maxent\Test 2 samplesfile=C:\Users\Luke\SkyDrive\Test 1.csv environmentallayers=C:\Users\Luke\Desktop\Luke\MAXENT\env_layers(ASCII) randomtestpoints=30 replicatetype=crossvalidate-N bio_13.PWM-N bio_14_PDM-N bio_16_PWQAURT-N bio_17_PREDRYQUART-N bio_18_PREWARMQUART-N bio_2_MDR-N bio_3_ISM-N bio_5.MTWM-N bio_8_MTWQ-N bio_9_M

5. Converting ha/LSU to LSU/ha

Appendix table 5.1. Number of ha supporting the relevant carrying capacity levels (ha/LSU) within each biome in South Africa.

	2-3	3-4	4-5	5-6	6-7	7-8	8-10	10-15	15-20	20-25	25-30	30-35	35-40
Savanna	87630	434635	722539	1770665	1919358	767334	2723723	8955914	8358036	3485253	3137594	3475444	2484784
Nama-Karoo	319294	279321	183107	136595	120982	59755	231010	403327	749678	2490588	2651461	2492223	24493476
Desert	3760	2616	1635	736	899	409	82	409	245	0	0	82	981
Grassland	92208	397768	480739	1178100	2551732	1854616	6941732	9474909	5863769	1798703	356487	188993	73979
Succulent Karoo	60082	45450	37194	30000	24360	14469	83461	313653	1931211	1750883	1517257	1091451	1100770
Fynbos	37848	72507	146404	123434	196105	171581	1384668	2584267	946436	684691	419921	162017	59265
Forest	490	22153	22234	3924	1553	1063	4414	163	0	0	0	0	0

Appendix table 5.2. Carrying capacity of LSU/ha, within each biome in South Africa.

	0.40	0.29	0.22	0.18	0.15	0.13	0.11	0.08	0.06	0.04	0.04	0.03	0.03
Savanna	35051.99	124181.39	160564.12	321939.08	295285.84	102311.26	302635.85	716473.11	477602.03	154900.14	114094.34	106936.74	66260.90
Nama-Karoo	127717.41	79805.87	40690.53	24835.43	18612.57	7967.35	25667.73	32266.14	42838.72	110692.79	96416.75	76683.77	653159.37
Desert	1504.10	747.38	363.31	133.76	138.34	54.50	9.08	32.70	14.01	0.00	0.00	2.52	26.16
Grassland	36883.06	113648.04	106830.82	214199.99	392574.23	247282.19	771303.61	757992.72	335072.54	79942.37	12963.17	5815.17	1972.76
Succulent Karoo	24032.85	12985.68	8265.26	5454.58	3747.67	1929.17	9273.45	25092.25	110354.91	77817.01	55173.00	33583.11	29353.86
Fynbos	15139.06	20716.36	32534.26	22442.55	30169.96	22877.53	153852.00	206741.34	54082.08	30430.71	15269.85	4985.15	1580.39
Forest	196.19	6329.35	4940.99	713.41	238.95	141.69	490.47	13.08	0.00	0.00	0.00	0.00	0.00