The rate of recolonization by native plant species after the eradication of invasive alien plant species in Limpopo Province, South Africa

By

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Declaration

I, Melford Mbedzi, declare that this thesis is my original work and has not been submitted for any degree at any other university or institution. The thesis does not contain other persons' writing unless specifically acknowledged and referenced accordingly.

Signed (Student): Date: 25/11/2020

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Abstract

Invasive alien plant species found in riparian zones are known to compete with native plant species for water, space, sunlight, and other natural resources by lowering the structural diversity of native vegetation, thereby altering the functioning of plant communities. This may impact on the number and variety of organisms that a certain vegetation type may support.

The aim of the study was to investigate the rate of recolonization by native plant species after the eradication of the *in situ* alien invasive plant species. This was done by measuring the rate of recolonization by native species after the removal of alien invasive species (Chapters 3 and 4), and by identifying factors to consider for recolonization to occur (Chapter 5). By measuring and identifying the above, it was possible to develop an adaptive management plan (Chapter 6) for the study area, which would assist local and provincial conservation agencies in conserving native species that should improve ecosystem dinamics.

The study was carried out in the Waterberg and Makhado district municipalities, Limpopo Province, on farms that are highly infested with alien invasive plant species. Seventy-two permanent plots of 10 m^2 each were constructed along three transects. The alien tree species in the plots were eradicated during March 2016 via mechanical clearing, and the area periodically monitored over a period of 34 months. The choice for the dimensions of quadrats depended on the size and distribution of the alien invasive trees, which grow in an aggregated form (such as *Acacia decurrens* and *Populus alba*) and have small canopies, except for *Lantana camara* in some cases. The species present (native and invasive alien plant in the quadrats were identified and recorded during March 2016 before removal of the invasive alien plant species.

This was done to establish the rate at which the species (native or alien) were germinating.

Results from this study indicate that in the case of *Acacia decurrens* and *Lantana camara*, recolonization of native species would be possible in the near future with continued monitoring and management, as some native grass and herb species were emerging on the study site. *Populus alba* proved troublesome due to its ability to reproduce vegetatively through root suckering, which enabled it to subsequently re-occupy the study area in larger numbers than before due less competition from other native and alien species. Therefore, it would require more time to eradicate this species than the other two. It is conclyuded that, the recolonization by native plants is possible, though it needs more monitoring. It is recommended that there needs to be an adaptive management plan that would assist in providing more affective results in ecosystem recovery and conserve the native plant species.

Table of Contents

List of Figures

List of Tables

List of Abbreviations and Acronyms

AMP	Adaptive Management Plan
CARA	Conservation of Agricultural Resources Act
CBD	Convention on Biological Diversity
GDP	Gross Domestic Product
CFR	Cape Floristic Region
CSIR	Centre for Scientific and Industrial Research
CUIS	Cumberland Island National Seashore
IAP	Invasive Alien Plants
IAS	Invasive Alien Species
NEMBA	National Environmental Management: Biodiversity Act
NPS	National Park Service
OECD	Organization for Economic Co-operation and Development
PCI	Potential for Conflict Index
SAPIA	Southern African Plant Invaders Atlas
SCOPE	Scientific Committee on Problems of the Environment
US-NPS	United States National Park Services

List of conference contributions

i. The rate of regeneration of native plant species after the eradication of invasive alien plant species (*Acacia decurrens* WILLD.) in the Limpopo Province, South Africa, was presented at the 2018 South African Association of Botanists national conference held at the University of Pretoria.

ii. The rate of recolonization by native plant species after the eradication of the invasive alien plant species (*Lantana camara* L.) in the Limpopo Province, South Africa was presented at the 2019 South African Association of Botanists national conference held at the University of Johannesburg.

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ii. The rate of regeneration of native plant species after the eradication of invasive alien plant species (*Acacia decurrens* WILLD.) in the Waterberg District Municipality, Limpopo Province, South Africa

CHAPTER ONE

INTRODUCTION

1.1 Background of the study

1.1.1 The invasion process

The colonization of native vegetation by invasive alien plant species represents a global threat, which has significant effects on the functioning of the ecosystem and biodiversity (Pandey et al., 2019). Invasion successes are principally a consequence of ecosystem interruption, normally driven by human activities (Tomasetto et al., 2018). With the rise in globalization, transport modes have experienced a speedy advancement that has heightened the introduction of biological materials (for example, plants) into new and remote zones. A considerable number of these plant species become established and spread past their indigenous region to suitable localities (Mack et al., 2000). The vector of these materials past their native region can be deliberate or accidental. Deliberately, plant materials are introduced outside their indigenous range to serve as crops, timber and firewood, ornamentals and garden plants, biological agents, or to stabilize sand dunes subsequently preventing soil erosion (Van Wilgen et al., 2001; Witt et al., 2018). Accidental introduction, on the other hand, happens through different means like vessels, air or over land. Although coincidental introduction also has detrimental impacts, most of the negative effects result from deliberate introductions, following escapes from gardens, farming areas or forests (Lambdon and Hulme, 2006). Nevertheless, not all these alien organisms worldwide prevail to establish in new conditions, or progress towards becoming pests, even though they may change the native composition in one-way or the other (Mack et al., 2000). The globalization process is accelerating and will probably result in reshaping the native vegetation (Boivin et al., 2016).

Riparian ecosystems are exceedingly susceptible to colonization by invasive alien plant species because of the ease and dynamic hydrological nature of rivers and streams through which propagules are transported. Hence, invasive alien populations effortlessly end up being established in riverine areas (Hood and Naiman, 2000). Natural disturbances and anthropogenic activities add to the potential for invader species to become established (Tomasetto et al., 2018).

Different definitions and criteria have been used to classify invasive alien plant species; from introduced, aliens, weeds, naturalized, problem plants to biodiversity transformation (Ortega and Pearson, 2005). Unfortunately, various criteria are employed by different authors to classify alien species (Richardson and Rejmánek, 2011). Richardson and Rejmánek (2011) defined some plants as invasives that "have sustained self-replacing populations for no less than 10 years, without direct intercession by people (or regardless of human mediation) by recruitment from seed or ramets fit for autonomous development", and "recruit reproductive offspring at considerable distances from the parent plants". Species that transform vegetation were thought of as being a subset of invasive alien plants that "change the character, condition, form or nature of community over a significant zone in respect to the degree of that community" (Richardson et al., 2000).

At a national scale, there have been at least three attempts to map the extent of the invasion problem. In 1993, the Council for Scientific and Industrial Research mapped invasive alien plants in South Africa, with the goal of estimating their impact at a national scale (Le Maitre et al., 2000). The mapping techniques used were coarse due to the paucity of reliable data, but a map at a 1:250,000 scale was produced, based primarily on the local knowledge of natural resource experts from across South Africa. The project estimated that invasive plants occupied 10.1 million ha (6.82% of South Africa and Lesotho) (van Wilgen et al., 2020). The longest-running project aimed at recording information on the national extent of alien plants is the Southern African Plant Invaders Atlas (SAPIA), which was initiated in 1994 (Henderson, 2006). As of May 2016, SAPIA had over 87,000 geo-referenced records for 773 alien plant taxa that are present outside of cultivation in southern Africa, making it the most extensive source of information on the distribution of invasive plants in the region (Robinson et al., 2020).

1.1.2 Invasion of alien species of riparian zones

The Australian species of *Acacia* are thought to be a portion of the most South African exceedingly destructive invaders (Van Wilgen et al., 2011), particularly in riparian ecosystems. The greater part of these *Acacia* species was introduced for commercial use amidst the nineteenth century, as well as for social and agricultural reasons (Richardson and Kluge, 2008). Various key attributes give the Australian *Acacia* an upper hand over indigenous species (Morris et al., 2011); these incorporate a high reproductive yield, fast rates of development and the capacity to outcompete native plant species, the capacity to gather high biomass (Witkowski 1991) and large, tenacious seed pools, an advantage to fix nitrogen (Marchante et al., 2009), absence of natural predators (Gibson et al., 2011), and phenotypic plasticity.

The invasion of riverine zones by woody invasive alien plant species is most worrying, as these invasive alien plant species change ecosystems by replacing the indigenous vegetation out of their territories, changing fire mechanisms and expanding fire peril, lowering biodiversity and water yield, expanding soil water repellence, modifying soil structure, erosion and siltation, destabilizing and degrading river banks and disrupting numerous ecosystem services (Gaertner et al., 2011). A significant number of the above-mentioned effects interact with each other, and abiotic changes and biotic reactions occur concurrently (Le Maitre et al., 2011).

Riverine areas fulfil essential biological, physical, socio-economic, and chemical capacities (Richardson et al., 2007), and assume an interface role that is useful amongst aquatic systems (Naiman and Decamps, 1997). They act as critical transition zones and channels for the exchange of energy and material between systems (Ewel et al., 2001), despite comprising a relatively small surface area of the landscape (Holmes et al., 2005).

The decrease in water resources because of invasive alien plants is of great concern to South Africa (Blignaut et al., 2007). Stream flow reduction is the most vital concern about the invasion of riparian zones and mountain catchment regions (Versfeld et al., 1998). Estimates differ between catchments (Cullis et al., 2007; Crous et al., 2012), but it has been estimated that more than 3 000 million m^3 of surface water runoff (roughly 7%) is lost per year to invasive alien plants in South Africa or southern Africa?, and the clear majority of this is from the fynbos and grassland biomes (Le Maitre et al., 2016). If invasive alien plants are given a chance to spread unchecked and were to reach their maximum capacity, reductions could get to 25 000 million m^3 runoff of surface water (roughly 58% runoff of surface water of the South African (Van Wilgen et al., 2008).

South African riparian ecosystems are especially susceptible to invasion of woody alien plant species (Le Maitre et al., 2011). If invasion results in dense or close-stands

(100% cover), it will contain 33% (roughly 51 266 km) of the aggregate length of rivers in South Africa (Marais and Wannenburgh, 2008).

Riverine areas are firmly affected by other ecosystems they connect (Casco et al., 2010). They contrast from adjoining upland regions in geomorphology, hydrology, vegetation, microclimate and fuel characteristics (Smith-Adao and Scheepers, 2007). These systems are generally characterized by high species diversity and ecological processes that can be ascribed to the changing, non-equilibrium nature of riverine areas (Richardson et al., 2007). Since riverine areas create passages through the landscape, they also provide connectivity, and make fundamental contributions to re-establishing and maintaining regional diversity (Casco et al., 2010).

Riverine areas show both lateral and longitudinal gradients (Casco et al., 2010). The magnitude of the stream is what makes them differ in width, the stream location inside the catchment, the hydrological processes and the local geomorphology (du Preez and Rowntree, 2006). Riverine zones are normally narrowest in the headwaters and enlarge downstream. Riparian areas of bigger rivers frequently comprise floodplains that might be liable to significant lots of yearly flooding, with complex geomorphological features, for example, braiding and lateral channel relocation (Ciszewski and Grygar, 2016).

Riverine areas are especially hard to manage because they tend to form part of, and get inputs from massive watersheds, over which managers usually have no control (Ewel et al., 2001). These areas have been the focal point of escalated human habitation and activity for centuries and are one of the world's most affected ecosystems (Strayer and Dudgeon, 2010).

Weeds may unfavourably influence the diversity of plant species by replacing climax vegetation or restricting juvenile establishment of indigenous species (Gooden et al., 2009). For instance, Grant et al. (2003) noted that the seed germination and initial survival of a native perennial grass species was reduced, upon invasion by *Acroptilon repens,* a perennial forb in the semi-arid prairies of North America. The mechanisms restricting indigenous species establishment in invaded communities include competition for resources, for example, light, nutrients and space, and additionally non-resource mediated interference such as allelopathy (Gooden et al., 2009).

Quantitative field-based investigations of differential impacts of invader plants measured on native species development stages (e.g. juvenile against adults) may indicate long-term mechanism by which weeds influence diversity of species (Gooden et al., 2009). Comparison of the multi-site methodology looks at diversity of indigenous species diversity value in sites infested with invaders to potential sites that do not have invaders; nevertheless, direct effects on species by the weeds are hard to quantify because of the nature of the technique which is correlative (Gooden et al., 2009). Weed eradication studies can give more grounded proof to impacts of weeds on indigenous species since diversity species change and abundance after removal of weed can be measured directly (Gooden et al., 2009). Weed removal can likewise show mechanisms of weed impacts, for example, recruitment constraint by observing changes of demography in species diversity populations after removal of weeds (Gooden et al., 2009). Furthermore, effects of the residual invader, for example, changed seed pool nutrient compositions and exhausted local soil-stored seed pools, and in addition differential impacts of weed eradication regime can affect the indigenous species response to weed eradication (Gooden et al., 2009), which restricts the efficacy of this approach.

1.1.3 Historical and current status of the pathways of introduction

After South Africa's democratisation in 1994, commodity imports increased further (Figure 1.1), the country's trading partners expanded (Ahwireng-Obeng and McGowan 1998), and new infrastructure was developed [e.g. the harbour at Ngqura near Port Elizabeth was built in the 2000s (Mitchell, 2014)]. Today, people, goods and transport systems can access South Africa through 72 official ports of entry, including eight maritime ports, ten airports and 54 land border posts (Faulkner and Wilson 2018). The number of people entering South Africa has since been increasing over time, and surpasses 21 million individuals, including more than 10 million tourists (World Tourism and Travel Council 2017), entered the country in 2016 (Faulkner and Wilson 2018). The travel and tourism industry contribution to South Africa's Gross Domestic Product has also increased with time (Faulkner and Wilson 2018). As alien taxa are often transported within the luggage of tourists, this pathway is an example of many socioeconomically important pathways that are increasing in their importance and that, as a result, could be playing an increasing role in enhancing introductions (Faulkner and Wilson 2018). Therefore, it is not surprising that many pathways are promoting the dispersal of alien taxa to South Africa, and that for many pathways the rate of introduction has recently increased or remained constant (Faulkner and Wilson 2018). For example, hunting generates a total estimated revenue of ZAR 2.61 billion, and the hunting market in South Africa has been increasing through time (Taylor et al. 2015).

Alien species are introduced to game farms to increase the attractiveness of the property to both tourists and hunters (Taylor et al., 2015; Faulkner and Wilson 2018), and eleven new alien species were introduced to South Africa for hunting between 2000 and 2011 (van Wilgen and Wilson 2018). The introduction of organisms through some pathways is not increasing, but just being done repeatedly and in some instance, some taxa are from multiple sources. For example, many of medicinally important plants, most of which are exotic to South Africa, are introduced into the country by people travelling from China, India, Nigeria, Ghana, Somalia, Ethiopia, Eritrea and the Democratic Republic of Congo (Byrne et al. 2017; Burness, 2019). A lot of these alien plants are imported in the form of viable propagules, with multiple immigrant groups importing the same medicinal plants, but from different parts of the globe (Mack, and Lonsdale, 2001).

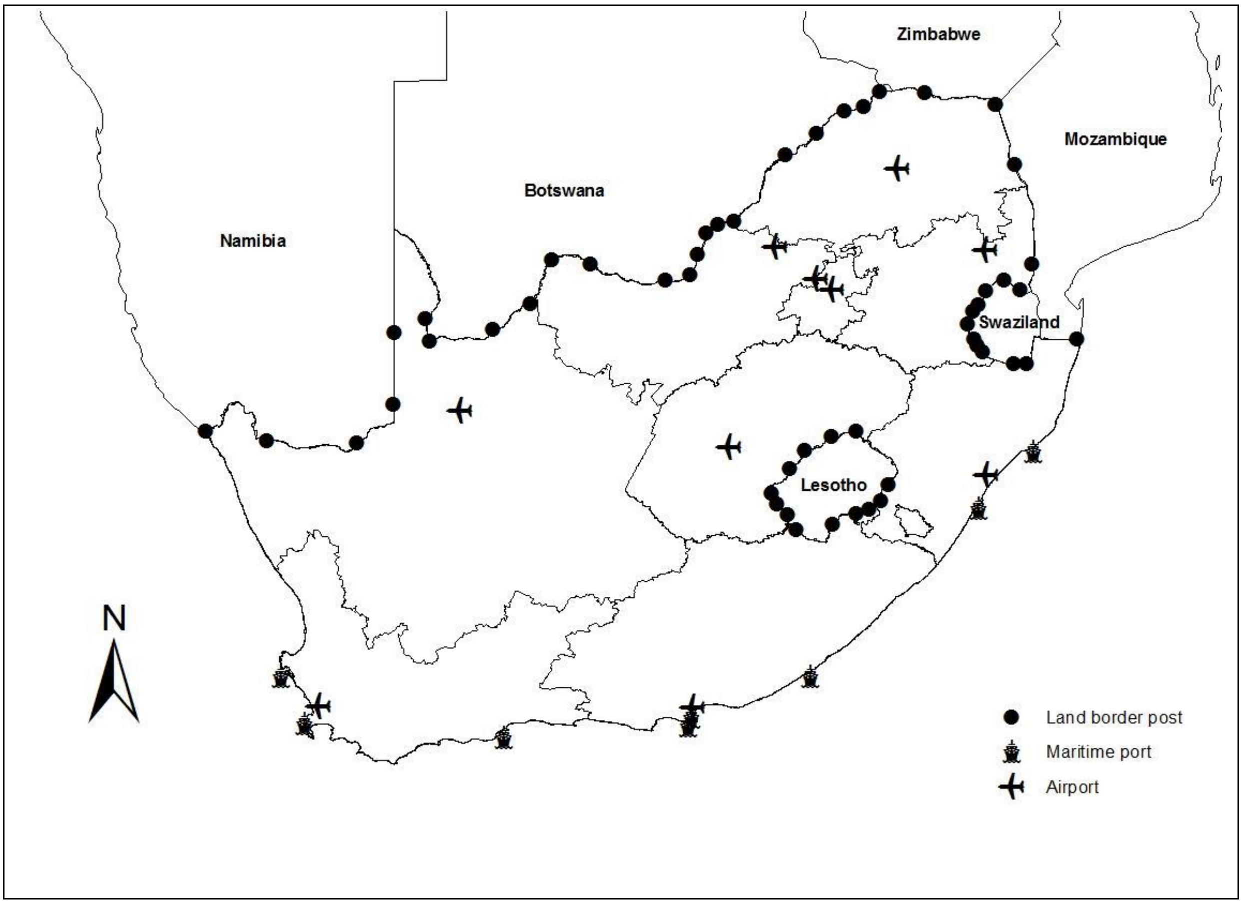

Figure 1.1: South African ports of entry (Courtesy of the South African Department of Home Affairs 2017).

When alien taxa enter South Africa through the ports of passage (Figure 1.1), further dispersal or spread is likely to occur. For instance, species that are traded at pet stores are regularly exchanged (for example, through private deals on sites) and moved across the nation by individuals of the public (Measey et al., 2017). Numerous invasive alien species have turned to be generally scattered through natural spread. After being introduced to the Western Cape, the common *Sturnus vulgaris* spread north (Picker and Griffiths, 2011). Alien organisms are also transported as pollutants of materials or as passengers along the nation's broad transport systems (Faulkner et al., 2017). For example, the Sirex wood wasp, *Sirex noctilio*, was probably imported and transported around the nation in pervaded timber (Picker and Griffiths, 2011; Hurley et al., 2012). Although no invasive alien species are known to have entered the nation through human-mediated transport infrastructure, organisms that have already been introduced might spread within the country through these corridors. For example, fish species are distributed along water channels or pipelines that are used to distribute water between river valleys (van Rensburg et al., 2011). Alien organisms can easily be transported countrywide through a variety of pathways, therefore, preventing the dispersal or spread of these species once they have been introduced would be nearly impossible (Faulkner and Wilson 2018).

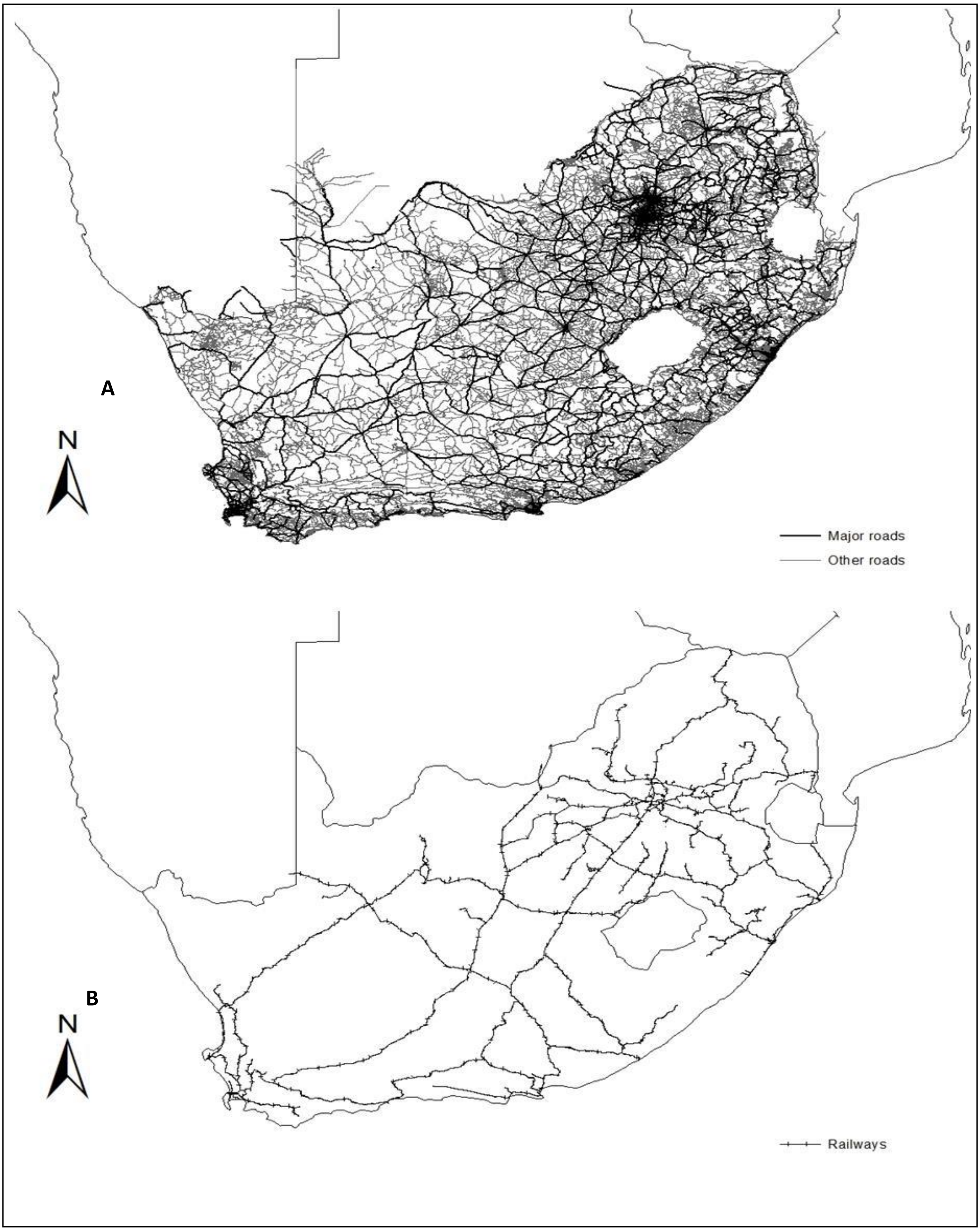

Figure 1.2: The South African (A) road and (B) rail networks. Major roads are motorways, primary and secondary roads (Faulkner and Wilson, 2018).

Data was obtained from Open Street Map contributors (2017). There is no expectation that significant expansion would occur in these networks in future, although traffic volumes have increased, and increase continuously. Evaluation of procedures of establishment are hindered by various doubts (Essl et al., 2015; Ojaveer et al., 2017; Tsiamis et al., 2017). The procedures of establishment are generally not known, especially for species that have been introduced accidentally, and the information that is accessible is frequently not reliable or detailed to assign procedures of establishment with confidence (Essl et al., 2015; Saul et al., 2017). In order to deal with this, assurance in procedure specification can be estimated (Essl et al., 2015; Ojaveer et al., 2017). However, as private and public gardens are dominated by alien species (Richardson et al., 2003), and as appropriate for new diversity of plants is shown by the South African consumer in the ornamental sector (Middleton 2015), this is improbable to be the occasion. The importance of the pathways may be underestimated due to the problem with the quality of data availability, while differing interpretations and errors may develop from other uncertainties (Tsiamis et al., 2017). Decision making and management may be in jeopardy due to these factors and make it difficult to determine the efficacy of management evaluations with confidence (Ojaveer et al., 2017).

The standing of the procedures of invasive alien plant species pathway establishment in South Africa and how they have varied through time has been lately evaluated utilizing historical establishment findings (Faulkner et al., 2016). However, this assessment did not use the pathway categorisation plan adopted by CBD, pathways were not ranked, and the viability of pathway-related control measures was not assessed. The more extensive classification plan of Hulme et al. (2008) was utilized. Here, historical introductions and socio-economic information were used to assess current pathway status and historical changes to the pathways, and where conceivable, socio-economic forecasts were utilized to get an indication of how these pathways might change in future (Faulkner et al., 2016).

Many invasive alien taxa have been unintentionally introduced to South Africa. The import of goods, for example, live plants and food has increased through time and even though control measures are set up to avoid the unintentional introduction of commodity contaminants, the rate at which invasive alien taxa are being presented through these pathways has not declined. In South Africa, the unintentional introduction of invasive alien taxa as stowaways on transport mechanisms is also playing a significant role that is probably going to increase in future; unfortunately, control measures are not set up for a considerable number of these pathways. The natural dispersal of invasive alien taxa into South Africa from our neighbouring countries are likely increase too; yet avoiding these introductions will be incredibly troublesome and to do as such would require regional co-operation (Faulkner et al., 2016).

1.1.4 The predictions of future pathways of introduction

While it is difficult to show how the pathways of introduction and dispersal will change in the future, some predictions can be made based on current or forecasted dynamics to the socio-economic importance of these introduction pathways (van Wilgen and Wilson 2018). Intentional introductions for some purposes are likely to increase in the future. For example, there has been an increase recently, in research for biological control and implementation (Zachariades et al. 2017; Faulkner and Wilson 2018), there is considerable interest in new agricultural opportunities [e.g. the introduction of grasses for biofuels (Visser et al. 2017)], and there is an increasing demand from consumers for new varieties of ornamental plants (Middleton 2015; Faulkner and

Wilson 2018). The release of biological control agents could continue to increase in the future, and the introduction of new taxa for agriculture and horticulture could also increase (Faulkner and Wilson 2018). Socio-cultural resistance could, however, affect introductions through some pathways. The hunting industry may benefit from the decrease in hunting opportunities available in other countries but could be affected negatively by increasing global anti-hunting sentiment and publicity (Taylor et al. 2015). It is, however, not certain whether introductions for hunting will continue at an increasing rate. Under the recently promulgated Alien and Invasive Species Regulations of the National Environmental Management: Biodiversity Act (NEM:BA, Act No. 10 of 2004), a permit is required to intentionally import a new alien taxon into South Africa. Such a permit is only approved by the Department of Environment, Forestry and Fisheries if a risk assessment, performed by a professional scientist, shows the risk of invasion to be low. Therefore, while new alien taxa will continue to be intentionally introduced through some pathways, these organisms should not pose a threat. However, it is important to note that compliance with and enforcement of the regulations could be problematic [for example, for aquarium plants (Martin and Coetzee 2011) and ornamental plants (Cronin et al. 2017).

1.1.5 The successful establishment of invasive alien plant species

When an invader plant species establishes, the struggle for limited resources is presumably what the recipient community has with the species interaction. Interspecific competition is viewed as the most important amongst the most critical procedures deciding the likelihood of plant invasion (Crawley, 1990), as proposed by the imperative part of disturbance, which decreases competition and increases the likelihood of invasion (Sher et al., 2000).

One of South Africa's provinces, Limpopo, is the most infested province by invasive alien plants, as a result of the high temperature and little precipitation that it gets. Districts including the study area have ideal conditions for invasive plants considering their warm climatic conditions and generally high annual precipitation. Invasive alien plants utilize certain strategies in order to survive and out-compete indigenous plants and become more successful. A significant part of the success of invasive alien plants is believed to be related with their superior ability to capture and maintain space. These invasive alien plants have the potential for rapid development especially in environments that are not resource limited. Alien plants invasion are becoming wide spread and serious worldwide (Richardson and van Wilgan, 2004).

1.1.6 Factors that influence the introduction of invasive alien plant species into new regions

The introduction and dispersal of alien species are promoted by various cooperating factors (including the environment and species characteristics), and patterns in financial elements (for example, management interventions and financial support) while assuming a critical task in building up the procedures of establishment and ascertaining how they vary with time (Zieritz et al., 2017). For instance, forecast changes to worldwide vitality markets may lead to an expansion in the quantity of aquatic species established in the USA via the removal of equilibrium water (Holzer et al., 2017). Acclimatization societies promote the removal of numerous invasive alien species in Australia, New Zealand, and the USA, and a decline in the public and technical assistance for these societies during the 20th century resulted in a decline in the activities of such societies (Seebens et al., 2017).

1.1.7 Invasive alien plant species management

The major part of effective plant eradication has been investigated just after a population had begun to mushroom and managed a spatial small degree, for example, less than 100 ha (Mack and Lonsdale, 2002). On the other hand, numerous endeavours to destroy infestations by invasive plant in California on areas bigger than 1000 ha have been unsuccessful (Rejmanek and Pitcairn, 2002).

Nonetheless, the spatial degree of newfound invasive species is typically ineffectively known, and strategies to define the spatial degree are proficiently few (Leung et al., 2010). Regardless of whether the removal is in fact attainable, it is yet important to evaluate the financial feasibility of feasible removal; specifically, whether expenses of control could be legitimized with regards to anticipated advantages related with the control (Panetta, 2009). Effective control relies upon containment that is effective. If control is disposed of as a choice at earlier stages, control resources could be re-established to lessen effects of the invasive alien or moderate its expansion (Panetta, 2009). For instance, initiatives may be assigned to constraining subordinate groups of species rather than trying to control built up sites (Moody and Mack, 1988), or attempts could be focussed outside of the group of species already in the vicinity to moderate expansion, for example, through making boundaries of inadmissible habitats.

Of the few strategies, the only one that resulted in extended improvements of the biological diversity was the removal of invasive alien species (McGeoch et al., 2010) wherein the removal was characterized by the eradication of all the individuals of a species from the re-colonization point which was probably not going to happen. Be that as it may, species removal needs coordinated managed exertion, and this is probably going to be successful on condition that there is a champion with the authority to guarantee exertion advancement (Kraus and Duffy, 2010), and when adequate administrative support, scope of environmental and biological characteristics have been distinguished, which then will influence the viability of removal. One of the lead bases utilized as a guideline of whether an attempt on eradication is probably going to be successful is the spatial degree of the invader population, having the removal of species being more likely when the site is smaller and when identification occurs not long after establishment, before seed pools or satellite population are introduced.

1.2 Problem statement

The South African government spends millions of rands to eradicate invasive alien plant species, however, the monitoring follow-ups after the removal of these alien species are very limited, which results in alien plant species re-establishment. This leads to the government being unsuccessful in restoring the replaced/displaced native plant species. It is therefore important to establish mechanisms that can render the eradication of invasive alien plant species effective. It has already been reported by Yurkonis et al. (2005), that invasive alien plants antagonistically impact on the diversity of plant species by replacing mature flora or restricting seedling establishment. It was demonstrated that invasion by the permanent forb *Acroptilon repens* in semi-arid fields of North America decreased the seed sprouting and primary survival of inhabitant perennial grass species (Grant et al., 2003). The factors hindering native species establishment in infested populations incorporate rivalry for natural resources, for example, nutrients and habitat (Gorchov and Trisel, 2003).

Fewer native juvenile species result from recruitment limitation in alien invaded sites to vegetation that are not invaded, yet this has not been demonstrated widely. Field-based quantitative investigations that quantify the differential impacts of invader plants on native populations' growth stages may demonstrate extended techniques by which invasive alien impacts on the diversity of species (Ens and French, 2008).

Monitoring of the eradicated sites is of outmost importance since failure to do so leads to the re-establishment of invasive alien plant species, and therefore calls for re-eradication of the very same site. This is the first study to look at this issue in the region, it is therefore a very important study which will give insights to the institutions whose main role is to control the invasive alien plant species.

1.3 Aim and objectives

The aim of this study was to investigate the rate of recolonization by native plant species after the eradication of the invasive alien plant species. To achieve the aim of the study, the following objectives were investigated:

i. Measuring the rate of recolonization by native species after the removal of alien invasive species.

ii. Identification of factors to consider for recolonization to occur.

1.4 Hypothesis

i. It takes a very long time the native plant species to recolonize an area previously infested with alien invasive plant species.

ii. There are factors to consider for recolonization success by native plant species to occur.

1.5 Rationale and justification

This study is one of the few studies in the Limpopo Province that look into the recovery of the native species community after eradication of invasive alien plant species in the Waterberg and the Vhembe district municipalities. It aims to provide comprehensive information on the rate of recolonization of native plant species after the removal of three invasive alien plant species, namely; *Acacia decurrens*, *Lantana camara* and *Populus alba*. It also assesses how factors such as human footprints affect the dispersal and distribution of these plant species as well as the economic implications posed by these invasive alien plants. These species were chosen because they of their invasibility and how they impact on the biodiversity.

Most studies focused on the Cape fynbos region and the Kruger National Park, therefore this study is of outmost importance. Inadequate recovery of riparian vegetation can result in soil erosion, loss of soil-stored propagules of native species, poor water quality, and a high risk of re-invasion by alien plant species (Holmes, 2001). In their study of riparian scrub recovery after alien-plant clearing in the Fynbos Biome, Galatowitsch and Richardson (2005) highlighted the need for research into the recruitment dynamics of disturbed riparian zones. It is important to ascertain whether the main supply of new propagules for recolonization comes from external sources, via water, wind or animal dispersal, or from in situ seed banks stored in the soil. The seed bank is defined as a reserve of viable seeds, fruits, propagules and other reproductive plant structures in the soil (Goodson et al., 2001).

This study gave insights on how the invasive alien plant species have affected the ecosystem, and how they can be managed. It provided an idea on how the areas which were previously infested by invasive alien plant species can be restored to their previous status. The study areas were chosen because they were highly infested with these three invasive alien species (*Acacia decurrens*, *Lantana camara* and *Populus alba*). They were seen having a negative impact on the indigenous species growing in the same areas as them (through an impromptu observation), and also, there is limited information on this kind of study in the Limpopo Province.

1.6 Significance of the study

This study will contribute significantly in cutting down the costs of invasive alien plant species eradication programme once the mechanisms that can prevent re-establishment of such species are determined. The restoration of native plant species through eradication of invasive alien plants may go a long way in the maintenance of biodiversity. Restoration is a process of re-establishing species, assemblages, structure, and/or ecological functions of the riparian habitat after alien clearing (Van Diggelen et al., 2001). If the hydrology and geomorphology of the invaded ecosystem is still functional and able to support an indigenous community, the re-establishment of species assemblages should be the target for ecosystem repair (Richardson et al., 2007). Many riparian specialist species are relatively widespread and, in this system, are predominantly resprouters. Ideal situations for riparian vegetation restoration are when the site can rapidly recover through natural re-colonization from undisturbed surrounding sites. This saves costs on restoration and ensures that the local gene pool is maintained (Prins et al., 2004). This study will therefore contribute immensely to the body of knowledge on management of invasive alien plants species, restoration of the sites previously infested with the invasive species to their natural state before the invaders came into the area and eradication of areas infested with invasive alien plants.

1.7 Structure of thesis

This thesis is structured in publication format in which each objective of the research is reported in a form suitable for journal submission. Putting this structure into consideration, it is necessary to state that certain repetitions are unavoidable in this thesis, as there might be certain components of the research which are useful for the publication of different sections of this research output.

The outline of the chapters is as follows: Chapter One outlines the background of the study, the aim and objectives, the overall idea of the effects, distribution, and the economic impacts of the invasive alien plants species. Chapter Two reviews the literature related to this study. Chapter Three focuses on *Acacia decurrens*, how it hinders native species re-establishment on the site they infested, how the native plant species are coping in the area previously infested by *Acacia decurrens* and how long it takes for them to occupy the area. Chapter Four gives an account on *Populus alba*, how it has affected the ecosystem and the indigenous plant species, and how the native plant species which previously occupied the area before being infested can recolonize the area after the removal of the invader plant species. Chapter Five looks at *Lantana camara* and its impacts on the native species vegetation on the same vicinity. It also assesses the rate at which the native plant species take to occupy the area previously occupied by the invasive alien plant species. Chapter Six provides the Adaptive Management plan of Acacia decurrens. How the conservation managers can tackle the problem caused by this species and be able to restore ecosystems to their former states. Chapter 7 gives a general summary, conclusions and recommendations. This chapter also provides an overall summary of this research project from Chapter One to Five.

References

Ahwireng-Obeng, F. and McGowan, P.J, 1998. Partner or hegemon? South Africa in Africa: part 1. *Journal of Contemporary African Studies*. 16, 5–38.

Blignaut, J.N., Marais, C. and Turpie, J.K., 2007. Determining a charge for the clearing of invasive alien plant species (IAPs) to augment water supply in South Africa. *Water SA* 33(1), 27–34.

Boivin, N.L., Zeder, M.A., Fuller, D.Q., Crowther, A., Larson, G., Erlandson, J.M., Denham, T. and Petraglia, M.D., 2016. Ecological consequences of human niche construction: Examining long-term anthropogenic shaping of global species distributions. *Proceedings of the National Academy of Sciences*, 113(23), 6388–6396.

Burness, A., 2019. An investigation of the international traditional medicine trade as an introduction pathway for alien plants into South Africa. Dissertation. University of Witwatersrand, Johannesburg

Byrne, M.J., Williams, V.L. and Wojtasik, E.M., 2017. The viability of propagules of alien plant species sold for traditional medicine in South Africa. *South African Journal of Botany*. 109, 281–287.

Casco, S.L., Neiff, J.J. and Poi de Neiff, A., 2010. Ecological responses of two pioneer species to a hydrological connectivity gradient in riparian forests of the lower Paraná River. *Plant Ecology* 209, 167–177.

Ciszewski, D. and Grygar, T.M., 2016. A review of flood-related storage and remobilization of heavy metal pollutants in river systems. *Water, Air, and Soil Pollution* 227(7), 239.

Crawley, M.J., 1990. The population dynamics of plants. *Philosophical Transactions of the Royal Society of London*. *Series B, Biological Sciences* 330, 125–140.

Cronin, K., Kaplan, H., Gaertner, M., Irlich, U.M. and Hoffman, M.T., 2017. Aliens in the nursery: assessing the attitudes of nursery managers to invasive species regulations. *Biological invasions*, 19(3), 925–937.

Crous, C.J., Jacobs, S.M. and Esler, K.J., 2012. Drought-tolerance of an invasive alien tree, *Acacia mearnsii* and two native competitors in fynbos riparian ecotones. *Biological Invasions* 14(3), 619–631.

Cullis, J.D.S., Görgens, A.H.M. and Marais, C., 2007. A strategic study of the impact of invasive alien plants in the high rainfall catchments and riparian zones of South Africa on total surface water yield. *Water SA* 33, 35–42.

Department of Home Affairs, 2017. South African ports of entry. Available at: http://www.home-affairs.gov.za/index.php/immigration-services/south-african-ports-of-entry [Accessed March 22, 2017].

Du Preez, L. and Rowntree, K.M., 2006. Assessment of the Geomorphological Reference Condition: An Application for Resource-directed Measures and the River Health Programme, Water Research Commission Report No. 1306/1/06. Pretoria, South Africa.

Ens, E. and French, K., 2008. Exotic woody invader limits the recruitment of three indigenous plant species. *Biological Conservation* 141, 590–595.

Essl, F., Bacher, S., Blackburn, T.M., Booy, O., Brundu, G., Brunel, S., Cardoso, A.C., Eschen, R., Gallardo, B., Galil, B. and García-Berthou, E., 2015. Crossing frontiers in tackling pathways of biological invasions. *Bioscience* 65, 769–782.

Ewel, K.C., Cressa, C., Kneib, R.T., Lakes, P.S., Levin, L.A., Palmer, M.A., Snelgrove, P. and Wall, D.H., 2001. Managing critical transition zones. *Ecosystems* 4, 452–460.

Faulkner, K.T., Robertson, M.P., Rouget, M. and Wilson, J.R., 2016. Understanding and managing the introduction pathways of alien taxa: South Africa as a case study. *Biological Invasions* 18, 73–87.

Faulkner, K.T., Hurley, B.P., Robertson, M.P., Rouget, M. and Wilson, J.R., 2017. The balance of trade in alien species between South Africa and the rest of Africa. *Bothalia-African Biodiversity and Conservation 47*(2), 1–16.

Faulkner, K.T. and Wilson, J.R.U., 2018. Pathways of introduction. In: van Wilgen BW, Wilson JRU (Eds.). The status of biological invasions and their management in South Africa in 2017.South African National Biodiversity Institute, Kirstenbosch and DST-NRF Centre of Excellence for Invasion Biology, Stellenbosch, pp 40–71

Gaertner, M., Richardson, D.M. and Privett, S.D.J., 2011. Alteration of ecosystem properties by introduced plants: Implications for restoration of fynbos in the Agulhas Plain, South Africa. *Environmental Management* 48, 57–69.

Galatowitsch, S.M. and Richardson, D.M, 2005. Riparian scrub recovery after clearing of invasive alien trees in headwater streams of the Western Cape. *Biological Conservation* 122, 509–521.

Gibson, M.R., Richardson, D.M., Marchante, E., Marchante, H., Rodger, J.G., Stone, G.N.N., Byrne, M., Fuentes-Ramírez, A., George, N., Harris, C., Johnson, S.D., le Roux, J.J., Murphy, D.J., Pauw, A., Prescott, M.N. and Wandrag, E.M., 2011.

Reproductive ecology of Australian acacias: Fundamental mediator of invasive success? *Diversity and Distributions* 17, 911–933.

Gooden, B., French, K. and Turner, P.J., 2009. Invasion and management of woody plant, *Lantana camara* L., alters vegetation diversity within wet sclerophyll forest in south eastern Australia. *Forest Ecology and Management* 257: 960–967.

Goodson, J.M., Gurnell, A.M., Angold, P.G. and Morrissey, I.P., 2001. Riparian seed banks: Structure, process and implications for riparian management. *Progress in Physical Geography*: Earth and Environment 25(3), 301–325.

Gorchov, D.L. and Trisel, D.E., 2003. Competitive effects of the invasive shrub, *Lonicera maackii* (Rupr.) Herder (Caprifoliaceae), on the growth and survival of native tree seedlings. *Plant Ecology* 166, 13–24.

Grant, D.W., Peters, D.P.C., Beck, G.K. and Fraleigh, H.D., 2003. Influence of an exotic species, *Acroptilon repens* (L.) DC. on seedling emergence and growth of native grasses. *Plant Ecology* 166, 157–166.

Henderson, L., 2006. Comparisons of invasive plants in southern Africa originating from southern temperate, northern temperate and tropical regions. *Bothalia* 36, 201–222

Holmes, P.M., 2001. A comparison of the impact of winter versus summer burning of slash fuel in alien-invaded fynbos areas in the Western Cape. *Southern African Forestry Journal* 192, 41–49.

Holmes, P.M., Richardson, D.M., Esler, K.J., Witkowski, E.T.F. and Fourie, S., 2005. A decision-making framework for restoring riparian zones degraded by invasive alien plants in South Africa. *South Africa Journal of Science* 101, 553–564.

Holzer, K.K., Muirhead, J.R., Minton, M.S., Carney, K.J., Miller, A.W. and Ruiz, G.M., 2017. Potential effects of LNG trade shift on transfer of ballast water and biota by ships. *Science of the Total Environment* 580, 1470–1474.

Hood, W.G. and Naiman, R.J., 2000. Vulnerability of riparian zones to invasion by exotic vascular plants. *Plant ecology*, *148*(1), 105–114.

Hulme, P.E., Bacher, S., Kenis, M., Klotz, S., Kühn, I., Minchin, D., Nentwig, W., Olenin, S., Panov, V., Pergl, J. and Pyšek, P., 2008. Grasping at the routes of biological invasions: A framework for integrating pathways into policy. *Journal of Applied Ecology* 45, 403–414.

Hurley, B.P., Croft, P., Verleur, M., Wingfield, M.J. and Slippers, B., 2012. The control of the Sirex woodwasp in diverse environments: The South African experience. In: *The Sirex Wood wasp and its Fungal Symbiont*: Research and Management of a Worldwide Invasive Pest. Slippers, B., de Groot, P., Wingfield, M.J. (Eds). Dordrecht. Springer Sci Bus. Media, 247–264.

Kraus, F. and Duffy, D.C., 2010. A successful model from Hawaii for rapid response to invasive species. *Journal for Nature Conservation* 18, 135–141.

Lambdon, P.W. and Hulme, P.E., 2006. How strongly do interactions with closely related native species influence plant invasions? Darwin's naturalization hypothesis assessed on Mediterranean islands. *Journal of Biogeography* 33, 1116–1125.

Le Maitre, D.C., Versfeld, D.B. and Chapman, R.A., 2000. The impact of invading alien plants on surface water resources in South Africa: A preliminary assessment. *Water SA* 26, 397–408.

Le Maitre, D.C., Gaertner, M., Marchante, E., Ens, E., Holmes, P.M., Pauchard, A., O'Farrell, P.J., Rogers, A.M., Blanchard, R., Blignaut, J. and Richardson, D.M., 2011. Impacts of invasive Australian acacias: Implications for management and restoration. *Diversity and Distributions* 17, 1015–1029.

Le Maitre, D.C., Forsyth, G.G., Dzikiti, S. and Gush, M.B., 2016. Estimates of the impacts of invasive alien plants on water flows in South Africa. *Water Sa*, 42(4), 659–672.

Leung, B., Cacho, O.J. and Spring, D., 2010. Searching for non-indigenous species: Rapidly delimiting the invasion boundary. *Diversity and Distributions* 16, 451–460.

Mack, R.N. and Lonsdale, W.M., 2002 Eradicating invasive plants: Hard-won lessons for islands. Turning the tide: The eradication of invasive species (ed. by C.R. Veitch and M.N. Clout), pp. 164–172, IUCN SSC Invasive Species Specialist Group, IUCN, Gland, Switzerland and Cambridge, United Kingdom.

Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M. and Bazzaz, F.A., 2000. Biotic invasion: Causes, epidemiology, global consequences, and control. *Ecological Applications* 10, 689–710.

Mack, R.N. and Lonsdale, W.M., 2001. Humans as global plant dispersers: getting more than we bargained for: current introductions of species for aesthetic purposes present the largest single challenge for predicting which plant immigrants will become future pests. *BioScience* 51(2), 95–102.

Marais, C. and Wannenburgh, A.M., 2008. Restoration of water resources (natural capital) through the clearing of invasive alien plants from riparian areas in South Africa - costs and water benefits. *South African Journal of Botany* 74, 526–537.

Marchante, E., Kjøller, A., Struwe, S. and Freitas, H., 2009. Soil recovery after removal of the N2- fixing invasive *Acacia longifolia*: Consequences for ecosystem restoration. *Biological Invasions* 11, 813–823.

Martin, G.D. and Coetzee, J.A., 2011. Pet stores, aquarists and the internet trade as modes of introduction and spread of invasive macrophytes in South Africa. Water SA 37, 371–380.

McGeoch, M.A., Butchart, S.H.M., Spear, D., Marais, E., Kleynhans, E.J., Symes, A., Chanson, J. and Hoffmann, M., 2010. Global indicators of biological invasion: Species numbers, biodiversity impact and policy responses. *Diversity and Distributions* 16, 95–108.

Measey, J., Davies, S.J., Vimercati, G., Rebelo, A., Schmidt, W. and Turner, A., 2017. Invasive amphibians in southern Africa: A review of invasion pathways. *Bothalia – African Biodiversity and Conservation* 47(2), 1–12.

Mitchell, M., 2014. A brief history of transport infrastructure in South Africa up to the end of the 20th century Chapter 8: the development of a national road system during the second half of the 20th century: infrastructure. *Civil Engineering= Siviele Ingenieurswese*, 22(8), 55–59.

Middleton, L., 2015. South African consumers' selection criteria for ornamental plants: A market perspective. *South African Journal of Plant and Soil* 32(4), 253–255.

Moody, M.E. and Mack, R.N., 1988. Controlling the spread of plant invasions: The importance of nascent foci. *Journal of Applied Ecology,* 25(3), 1009–1021.

Morris, T.L., Esler, K.J., Barger, N.N., Jacobs, S.M. and Cramer, M.D., 2011. Ecophysiological traits associated with the competitive ability of invasive Australian acacias. *Diversity and Distributions* 17, 898–910.

Naiman, R.J. and Décamps, H., 1997. The ecology of interfaces: Riparian zones. *Annual Review of Ecology and Systematics* 28, 621–658.

Ojaveer, H., Olenin, S., Narščius, A., Florin, A.B., Ezhova, E., Gollasch, S., Jensen, K.R., Lehtiniemi, M., Minchin, D., Normant-Saremba, M. and Strāke, S., 2017. Dynamics of biological invasions and pathways over time: A case study of a temperate coastal sea. *Biological Invasions* 19(3), 799–813.

Ortega, Y.K. and Pearson, D.E., 2005. Weak vs. strong invaders of natural plant communities: Assessing invasibility and impact. *Ecological Applications* 15, 651–661.

Pandey, C.B., Singh, A.K., Saha, D., Mathur, B.K., Tewari, J.C., Kumar, M., Goyal, R.K., Mathur, M. and Gaur, M.K., 2019. *Prosopis juliflora* (Swartz) DC.: An invasive alien in community grazing lands and its control through utilization in the Indian Thar Desert. *Arid Land Research and Management,* 1–22.

Panetta, F.D., 2009. Weed eradication — an economic perspective. *Invasive Plant Science and Management* 2, 360–368.

Picker, M. and Griffiths, C.L., 2011. Alien and invasive animals: A South African perspective. Struik Nature. Cape Town, South Africa.

Prins, N., Holmes, P.M. and Richardson, D.M., 2004. A reference framework for the restoration of riparian vegetation in the Western Cape, South Africa, degraded by invasive Australian Acacias. *South African Journal of Botany* 70, 767–776.

Rejmánek, M. and Pitcairn, M.J., 2002. When eradication of exotic pest is plants a realistic goal? In: Veitch CR, Clout MN. (Eds.). Turning the Tide: The Eradication of Invasive Species. IUCN Invasive Species Specialist Group, Gland, Switzerland and Cambridge, United Kingdom.

Richardson, D.M., Cambray, J.A., Chapman, R.A., Dean, W.R.J., Griffiths, C.L., Le Maitre, D.C., Newton, D.J. and Winstanley, T.J., 2003. Vectors and pathways of biological invasions in South Africa: Past, present and future. Invasive Species. Vectors and Management Strategies, 292–349.

Richardson, D.M. and Kluge, R.L., 2008. Seed banks of invasive Australian *Acacia* species in South Africa: Role in invasiveness and options for management. *Perspective in Plant Ecological Evolutionary Systematics* 10, 161–177.

Richardson, D.M., Pysek, P., Rejmánek, M., Barbour, M.G., Panetta, F.D., West, C.J., 2000. Naturalization and invasion of alien plants: Concepts and definitions. *Diversity and Distributions* 6, 93–107.

Richardson, D.M. and Rejmánek, M., 2011. Trees and shrubs as invasive alien species – a global review. *Diversity and Distributions* 17, 788–809.

Richardson, D.M. and van Wilgen, B.W., 2004. Invasive alien plants in South Africa: How well do we understand the ecological impacts? *South African Journal of Science* 100, 45–52.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P. and Hobbs, R.J., 2007. Riparian vegetation: Degradation, alien plant invasions, and restoration prospects. *Diversity and Distribution* 13, 126–139.

Robinson, T.B., Peters, K. and Brooker, B., 2020. Coastal invasions: the South African context. In: van Wilgen BW, Measey J, Richardson DM, Wilson JR, Zengeya TA (Eds.). *Biological invasions in South Africa*. Springer, Berlin, pp 227–246.

Saul, W.C., Roy, H.E., Booy, O., Carnevali, L., Chen, H.J., Genovesi, P., Harrower, C.A., Hulme, P.E., Pagad, S., Pergl, J. and Jeschke, J.M., 2017. Assessing patterns in introduction pathways of alien species by linking major invasion data bases. *Journal of Applied Ecology* 54(2), 657–669.

Seebens, H., Blackburn, T.M., Dyer, E.E., Genovesi, P., Hulme, P.E., Jeschke, J.M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M. and Bacher, S., 2017. No saturation in the accumulation of alien species worldwide. *Nature Communications* 8(14435), 1–9.

Sher, A.A., Marshall, D.L. and Gilbert, S.A., 2000. Competition between native Populus deltoides and invasive Tamarix ramosissima and the implications for reestablishing flooding disturbance. *Conservation Biology*, 14(6), 1744–1754.

Smith-Adao, L.B. and Scheepers, A.C.T., 2007. An assessment of the channel morphological changes in the Lourens River, Western Cape. *Water SA* 33(4), 559–570.

Strayer, D.L. and Dudgeon, D., 2010. Freshwater biodiversity conservation: recent progress and future challenges. *Journal of the North American Benthological Society*, *29*(1), pp.344-358.

Taylor, A., Lindsey, P. and Davies-Mostert, H., 2015. An assessment of the economic, social and conservation value of the wildlife ranching industry and its potential to support the green economy in South Africa. Endangered Wildlife Trust, Johannesburg.

Tomasetto, F., Duncan, R.P., Hulme, P.E. and Wiser, S.K., 2018. Segregation, nestedness and homogenisation in plant communities dominated by native and alien species. *Plant Ecology and Diversity* 4, 479–488.

Tsiamis, K., Cardoso, A.C. and Gervasini, E., 2017. The European Alien Species Information Network on the Convention on Biological Diversity pathways categorization. *NeoBiota* 32, 21–29.

Van Diggelen, R., Grootjans, A.P. and Harris, J.A., 2001. Ecological restoration: State of the art or state of the science? *Restoration Ecology* 9, 115–118.

Van Rensburg, B.J., Weyl, O.L., Davies, S.J., Van Wilgen, N.J., Spear, D., Chimimba, C.T. and Peacock, F., 2011. Invasive vertebrates of South Africa. *Biological invasions: Economic and environmental costs of alien plant, animal, and microbe species*, 326–378.

Van Wilgen, B.W., Dyer, C., Hoffmann, J.H., Ivey, P., Le Maitre, D.C., Moore, J.L., Richardson, D.M., Rouget, M., Wannenburgh, A. and Wilson, J.R.U., 2011. National-scale strategic approaches for managing introduced plants: Insights from Australian Acacias in South Africa. *Diversity and Distributions* 17, 1060–1075.

Van Wilgen, B.W., Reyers, B., Le Maitre, D.C., Richardson, D.M, and Schonegevel, L., 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. *Journal of Environmental Management* 89, 336–349.

Van Wilgen, B.W., Richardson, D.M., Le Maitre, D.C., Marais, C. and Magadlela, D., 2001. The economic consequences of alien plant invasions: Examples of impacts and approaches to sustainable management in South Africa. *Environment, Development and Sustainability* 3, 145–168.

Van Wilgen, B.W. and Wilson, J.R.U., 2018.The status of biological invasions and their management in South Africa in 2017. South African National Biodiversity Institute, Kirstenbosch and DST-NRF Centre of Excellence for Invasion Biology, Stellenbosch.

van Wilgen, B.W., Wilson, J.R., Wannenburgh, A. and Foxcroft, L.C., 2020. The extent and effectiveness of alien plant control projects in South Africa. In *Biological Invasions in South Africa* (pp. 597–628). Springer, Cham.

Versfeld, D.B., Le Maitre, D.C. and Chapman, R.A., 1998. Alien Invading Plants and Water Resources in South Africa: A Preliminary Assessment. CSIR. (WRC Report No. TT 99/98).

Visser, V., Wilson, J.R., Canavan, K., Canavan, S., Fish, L., Maitre, D.L., Nänni, I., Mashau, C., O'Connor, T.G., Ivey, P. and Kumschick, S., 2017. Grasses as invasive plants in South Africa revisited: Patterns, pathways and management. *Bothalia-African Biodiversity & Conservation*, 47(2), 1–29.

Witkowski, E.T.F., 1991. Growth and competition between seedlings of *Protea repens* (L.) L. and the alien invasive, *Acacia saligna* (Labill.) Wendl. in relation to nutrient availability. *Functional Ecology* 5, 101–110.

Witt, A., Beale, T. and van Wilgen, B.W., 2018. An assessment of the distribution and potential ecological impacts of invasive alien plant species in eastern Africa. *Transactions of the Royal Society of South Africa* 73(3), 217–236.

World Tourism and Travel Council (2017) Data gateway. https://www.wttc.org/datagateway/. Accessed 12 June 2016.

Yurkonis, K.A., Meiners, S.J. and Wachholder, B.E., 2005. Invasion impacts diversity through altered community dynamics. *Journal of Ecology* 93, 1053–1061.

Zachariades, C., Paterson, I.D., Strathie, L.W., Hill, M.P. and Van Wilgen, B.W., 2017. Assessing the status of biological control as a management tool for suppression of invasive alien plants in South Africa. *Bothalia-African Biodiversity & Conservation*, 47(2), 1–19.

Zieritz, A., Gallardo, B., Baker, S.J., Britton, J.R., van Valkenburg, J.L., Verreycken, H. and Aldridge, D.C., 2017. Changes in pathways and vectors of biological invasions in Northwest Europe. *Biological Invasions* 19(1), 296–282.

CHAPTER TWO

LITERATURE REVIEW

2.1 Global economic impact of invasive alien plant species

The 2014 Gathering of the Gatherings for the Convection on Biological Diversity (CBD) was an opportunity which was essential for the global network to contribute to progressively strong activity on the management and biodiversity conservation in all nations, particularly developing ones. While the management and conservation of biodiversity in nations which are yet developing has since been a priority since the assumption of the Convection at the United Nations Conference on Environment and Development (UNCED) (informally the Rio Summit) in 1992, another point of view remains a pressing need globally (World Bank, 2012).

While developing nations contain a large portion of the world's biodiversity (Giam et al., 2010; Lenzen et al., 2012), consumption on biodiversity in those nations represents not exactly 50% of worldwide spent on biodiversity. Parker et al. (2012) gave the most noteworthy estimate of biological diversity spending in nations that are yet developing at 42% of worldwide spending. New expenditure is required fundamentally to mitigate biodiversity loss and advance positive biodiversity change in developing regions (McCarthy et al., 2012). For example, to positively affect biodiversity, notwithstanding existing protected territory, more financing is required (World Bank, 2012).

There are two clear ways to deal with managing this worldwide financing hole. To start with, numerous non-governmental organizations (NGOs) and different specialists have created frameworks for focusing on the subsidizing from the created world towards those environments in the developing world in the direst conditions (Waldron et al., 2013). These endeavours are huge and NGOs, intergovernmental organizations and governments have created successful strategies for recognizing biodiversity under threat. In any case, there is no clear proof that these techniques direct worldwide biodiversity funding in the best way (Holmes et al., 2012). Another methodology is to utilize new financing sources offered by creating markets and frameworks which compensate conservation instead of misuse of biodiversity. Payment for biological diversity management and other esteem-based frameworks for conservation and the board of biodiversity have grown essentially over the previous decade (TEEB, 2009). Be that as it may, these mechanisms are at present a little commitment to biodiversity funding (Parker et al., 2012) and they may work in on fundamentally diverse courses in various nations (Schomers and Matzdorf, 2013). It therefore, very important to note that these invasive alien plant species have a great impact financially on the global level, and the lack of financial support or, rather the shortage of finances has a large impact on the success of the programs that manage invasive alien plant species.

2.2 Economic impact of invasive alien plant species in Africa

Agriculture is essential for smallholders, yet the section is especially severely influenced by IAS. For example, tomatoes can be germinated all year, and beside use in the home, they can be sold on neighbourhood household markets at a high price; tomatoes are viewed as an extremely encouraging zone for agricultural growth, however, the division across Africa is as of now encountering noteworthy effects from *T. absoluta* which puts the eventual fate of this yield in danger (Tefera and Tefera, 2013; Tonnang et al., 2015).

Parthenium pervasions can lead to vocation influence past smallholder crop yield depletions, with school age kids, for instance, investing days eradicating and missing key times of education and constraining prospects. Furthermore, extended introduction to this allergenic weed can cause skin and respiratory issues. These kinds of effect might be hard to evaluate financially; however, they cannot be disregarded if the genuine expenses of IAS are to be comprehended. Evaluation of the present estimation of time put by smallholders in weeding *Parthenium* at a basic labour rate determined for each impacted investigation nation gives yearly figures of R2.9 million in Uganda; R4.3 million in Tanzania; R21 million in Kenya; and R244 million in Ethiopia (CABI, unpublished).

2.3 Economic impact of invasive alien plant species in South Africa

Approximately 500 million USD has been spent by the South African government via the 'Working for Water' Programme to control alien invasive species, particularly trees, to improve water supplies in catchment areas, protect agriculture and biodiversity and provide skills training and employment (Van Wilgen and Wannenburgh, 2016). Some of the worst invasions have been reduced by this significant investment, but broader societal involvement along with new tools for management is required in managing invasions and to reconsider how scientists and the state interact with society. In South Africa, there is a developing history on the literature of perceptions of control provided, naturalized and invasive alien flora (Bennett, 2015; Pooley, 2018).

The history of invasive plants in South Africa is dominated by one fact: national planning has been consistently led by the Cape and action invasive alien species and weeds, especially relating to invasive trees and agricultural weeds (Clarke, 1994). The belief that invasion is a nationwide issue that needed a national policy to control was initiated in the late twentieth century and implemented only after in 1994. The first weed concerns emerged in the Cape Colony. Throughout the twentieth century, interest in conserving alien and indigenous vegetation was also firm than anywhere else in the country (Clarke, 1994). There are no actions being taken with the species scuh as *Acacia decurrens*, *Lantana camara* and *Populus alba* with regard to management and restoration of impacted areas in the Limpopo Province.

2.4 Environmental impact of invasive alien plant species

Invasive alien plant species have demonstrated the ability to cause both economic and environmental influence in South Africa. Invasive alien plant species have an adverse influence on all components of biological diversity, from the whole ecosystem to the genes. Adverse effects of the invasive alien trees and shrubs are as follows:

- Reduction in water availability and stream flow;

- Potential depletion of prolific land;

- Potential depletion of pasture;

- Livestock and human subversion (for example, *Lantana camara*);

- Damage in wildfires and fire protection increase;

- Expansion of soil erosion after fires in densely infested regions;

- Dam residue;

- Soil nutrient status change;

- Threat to indigenous species and biodiversity loss;

- Ecosystems biomass change;

- Compatibility of habitat for indigenous animal species change; and

- They hybridize with related native genera (e.g. *Rubus* sp.), therefore, gene exchange.

In various ecological regions, South Africa has large important steps in explaining and measuring the environmental and economic impacts of invasive alien plant species (Van Wilgen et al., 2001). Table 1.1 shows different invasive alien plant species' documented influence on the environment.

Richardson et al. (1989) approximated that invasions in the fynbos biome could lessen the species richness by somewhere in the range of 45 and 67 percent. Different investigations have indicated proof of changed nutrient regimes because of either amplification of nitrogen fixation or expansion of the biomass decomposition (Yelenik, 2000) and influences on seed pools of indigenous fynbos species associated with attacks of *Acacia saligna* (Holmes and Cowling, 1997a, b). This study is also in agreement with this fact, since after the eradication of invasive species, the native species have difficulties recolonizing the area. They therefore, need some assistance in terms of active restoration of the eradicated sites.

The effect of invasive alien plant species on water sources in South Africa has been contemplated. Le Maitre et al. (2000) found that a reduction in stream flow was in the range of 4.7 and 13.0 percent. A nationwide study uncovered that invasive alien plants were utilizing 6.7 percent of the aggregate mean yearly surface overflow, or 9.95 percent of the useable surface runoff, based on designed estimates. These numbers are rough estimates since there are extensive changes even between fundamentally same adjoining watersheds, however, they are thought to be the most ideal estimates nationwide (Le Maitre et al., 2000).

The invasive alien plant species have also been ranked according to their water use (Le Maitre *et al*., 2000). In descending order of water use, they are *Acacia mearnsii*, *Acacia cyclops*, *Acacia dealbata*, *Pinus* sp., *Eucalyptus* sp., *Prosopis* sp., *Acacia saligna*, *Melia azedarach*, *Solanum mauritianum*, *Lantana camara*, *Chromoleana odorata*, *Hakea* sp., *Populus* sp., *Jacaranda mimosfolia*, *Sesbania punicea*, *Rubus* sp., *Acacia longifolia*, *Psidium guajava*, *Ceasalpinea decapetala*, *Salix* sp., *Acacia melanoxylon*, *Acacia decurrens*, and *Quercus robur*. Crous et al. (2012) showed that stream-flow reduction by forest trees is not constant for any tree species, it changes in both time and space considerably.

Table 1.1: Some of the notable environmental influences of invasive alien plants in South Africa and affected biomes (Van Wilgen et al., 2001).

Species	**Biome**	**Effects**
Acacia cyclops A.Cunn. ex G.Don.	Fynbos; Forest; and Savannah	Dynamic changes in coastal sediment Altering the dispersal of seed Indigenous species get out-competed Biomass increases Rare African penguins nesting habitat is provided here
Acacia longifolia (Andr.) Willd.	Fynbos Savannah	Increases litter fall Ground living invertebrate diversity decreases

		Stream flow reduction
Acacia mearnsii De Wild.	Grassland Fynbos Forest Savannah	Reduction of ground dwelling diversity invertebrates Reduction in stream flow Stream banks destabilization Can increase erosion, but also used for land stabilization
Acacia saligna (Labill.) H.L.Wendl.	Fynbos Forest Savannah	Litter increases Biomass increases Changes in lowland fynbos nutrient chemistry Fluctuations in seed dispersal Moisture content reduction leading to change in fire regime
Eucalyptus obliqua L'Hér.	Fynbos Grassland Savannah Forest	Water repellence increases Impacts soil erosion to a variable degree
Hakea sericea	Fynbos	Biomass increases

Schrad. & J.C.Wendl.	Forest	Distribution and size of fuel change Decreases moisture content resulting in change of fire regime Increased biomass results in very intense fires when felled plants burnt Dense stands limit options for fire management Alters flora structure leading in decrease in abundance and diversity of native birds Alters arthropod species makeup with some species expanding while others decrease Decreases leaf retention and seed percentage set in native Proteaceae
Melia azedarach L.	Forest Savannah	Indigenous species get out-competed Frugivorous birds' feeding dynamics change
Pinus pinaster Aiton.	Forest Savannah Fynbos	Indigenous species gets out-competed Fire management becomes limited by dense stands Stream reduction

Pinus patula Schiede ex Schltdl. & Cham.	Forest Savannah	Indigenous species out-competed Fire management options gets limited due to dense stands Stream reduction
Pinus radiate D.Don.	Fynbos	Stream flow reduction
Prosopis juliflora (Sw.) DC.	Savannah	Biomass increases Vegetation structure changes Accessibility decreases Pasture productivity decreases Indigenous species out-competed Dung beetle assemblage diversity decreases
Psidium guajava L.	Forest Savannah Grassland	Indigenous species out-competed

Rubus sp. L.	Savannah Grassland Forest	Hybridizes with native *Rubus* sp.
Salix babylonica L.	Karoo	Destabilizes river banks and excludes native plants
Sesbania punicea (Cav.) Benth.	Savannah Forest Fynbos Grassland	Accessibility decreases Bank erosion increases Stream flow decreases Poisons stock
Solanum *mauritianum* Scop.	Savannah Forest Grassland	The ground dwelling invertebrate diversity decreases Feeding ecology of Rameron Pigeon gets changed (and other native birds) Indigenous species out-competed
Lantana camara L.	Fynbos Grassland	The ground dwelling invertebrate diversity decreases Allelopathy suppresses regeneration Poisons livestock

2.5 Management of invasive alien plants

Early investigations on the influence of invasive alien plants in South Africa focused on ecosystem effects such as effects on native plant species, depletion of ecological utilities and biodiversity. The legislation that control the promotion of seed and germination of some species proclaimed as invaders were enacted by the government in South Africa (Government Gazette, 2001).

In South Africa, there are four principal instant procedures employed to manage invasive alien plants species:

- Mechanical management (hand pulling, slashing, rotor tilling, clipping, ring-barking, and felling);
- Chemical management (young saplings spraying, painting and frilling, cut-and-squirt treatment, and injecting chemicals into large trees);
- Controlled burning of fire; and
- The use of an integrative manner of mechanical and chemical control methods in combination, where suitable, coupled with biological control.

A long-term programme such as the Biological control has been developed using insects, other biological management factors or fungi, inter alia to annihilate seeds of invasive plants. This programme has substantially released an insect that infest seeds of *Acacia mearnsii*, a seed-feeding *Melanterius maculatus* in many of the regions of the country (Pieterse and Boucher, 1997).

There are propositions to enact the screening of new species for their prospective invasiveness just before establishment (Van Wilgen et al., 2000). The issue becomes a brutal cycle in which a set of tissues is replaced by another as new species come into the country assuming control from those brought under management because there is no convention for evaluating prospective invasive species before their establishment (Van Wilgen et al., 2000). Specialists are also investigating the likelihood of delivering seedless clones of financially germinated pines (Richardson and Higgins, 1998).

South Africa has a portion of the vital accomplishments and improvement in the management of invasive alien plant. The cost of managing invasive plant attacks in South Africa has been approximated to be almost R17.4 trillion, or R871 million every year for the set time of 20 years that will take to manage the issue (Chapman and Versfeld, 1998), however, few experts doubt the range of the 20-year time span and the resulting financial assessment, and others challenge the qualities utilized as part of costing the task.

The probable decrease in estimation of the fynbos biome (1 million ha) because of invasion (in view of six segments: water generation, harvesting of wild flower, visits by hikers, endemic, ecotourism and genetic storage) was approximated at R170.5 trillion per year (Higgins et al., 1997). Black wattle's (*Acacia mearnsii*) net present cost of invasions in South Africa has been approximated to reach about R20 trillion, even though some models used to draw some assumptions are debated (Le Maitre et al., 2011).

The plantation forestry, regularly contemplated as the significant source of invasion attacks, donates R4.35 trillion, or 2% of Gross Domestic Product (GDP), and hires around 100 000 individuals. The downstream forest-based businesses donate a further R23 trillion, in which much of it is in export profit. Nonetheless, it must be recalled that the forest business is frequently seconded by government in its establishment exercises, and furthermore, that presumably 50% of the troublesome species touch base as decorative (Van Wilgen et al., 2001).

More than 38% of the sites attacked by woody alien species in South Africa is inhabited by species utilized in profit-oriented forestry, however not necessarily planted for profit-oriented logging, as the business species were likewise utilized by formal organizations and land users as a source of firewood, shade, windbreaks, woodlots, and so on. The South African government committed R1.45 trillion somewhere in the range of 1995 and 2000 to the 'Working for Water' Programme for the management of these invasive alien plant species (Van Wilgen, 2001).

The eradication scheme, which is for the most part mechanical and chemical, has provided a lot of employment for poor communities. In most cases, it may seem as if though the creation of such programmes is only for job creation rather than fixing the problem, as they mostly lack follow-up monitoring of the areas after eradication. In 1998, around 40 000 individuals were hired by the 'Working for Water' Programme to remove invasive alien plant species. In South Africa, the biological management is visualized as a long-term alternative for controlling further attacks and re-invasions. The aggregate cost of biological control research activity between 1997 and 2000 was US$ 3 million (Van Wilgen, 2001). If there could be improvement in the management of the invasive alien species, it would certainly cut down the cost of such management.

Utilizing biological management to eradicate invasive alien plant species and shrubs might cost the government R5.8 trillion in more than 20 years, or 2.9 trillion per annum, a cost which is viewed as sensible for developing countries like South Africa. However, bio-control is currently only appropriate to a very narrow range of tree species and acute care must be taken to guarantee that there are no undesirable side-effects associated with the utilization of the invasive alien bio-control agents (Van Wilgen et al., 2001). This necessitate a lot of technical extent which is in constrained supply. The presentation of CBD must also be observed in this context. There must therefore, be specific biological control agents being introduced, such that the other species are safe from such agents of control.

Around 66% of the finances were spent on the main 10 taxa, with the rest divided among 95 less noticeable taxa. The biggest part of financing (R561.9 million) was spent on the management of *Acacia mearnsii*. When this is added to the expenses associated with the closely-related wattle species *Acacia dealbata* (R79.3 million), the cost of control of the two-species amounted to 19.4% of the expenses of all alien plant control. An aggregate of R435.5 million was spent on the following most targeted species, while R237.0 and R183.5 million were spent on Eucalyptus and Pinus species respectively. Remaining species in the top 10 were *Lantana camara* (R180.6 million), *Chromoleana odorata* (R171.8 million), *Solanum mauritianum* (R121.5 million), Hakea species (R69.0 million) and *Acacia cyclops* (R58.0 million) (Van Wilgen et al., 2012).

2.6 Post management species recovery

The rate at which a riverine vegetation regenerates inherently to a state where it was, after invasive alien species eradication, is influenced by the flora and soil structure, the invasive alien plant species involved, the invasion age and number of individuals, the control mechanisms used; and in fire susceptible ecosystems, the frequency of fire cycles applied to the invasive species before and after eradication (Macdonald, 2004). Numerous Invasive Alien Plants (IAP) form seed pools which make the extended recovery of an area more challenging and complicate control objectives (D'Antonio and Meyerson, 2002). Nevertheless, some researchers have been successful in using alien species to rehabilitate certain functions inside a destructed ecosystem in cases where the remaining indigenous species cannot perform the required functions (D'Antonio and Meyerson, 2002). The South African government initiated the 'Working for Water' programme in 1995 (Van Wilgen et al., 1998) to eradicate the watersheds of invasive plant species to improve stream flow (Dye and Jarmain, 2004).

The logs were removed from the riverine zones after cutting (to hinder log-jam damage along the stream), some left to decompose in cases where the log is not too heavy or burnt in slash piles. Since the invasive alien species can rapidly re-vegetate the barren land via ample production of seeds, swift seed pool collection and vigorous seed sprouting, native plant populations should be recovered as fast as feasible (Richardson and Cowling, 1992). In some of the areas, the logs are left, hindering the growth of other species except, alien ones that are adapted to different kinds of conditions, such as lack of sunlight.

Riverine ecosystem recovery is a procedure of re-colonizing species, accumulations, system, and/or environmental functions of the riverine habitat after alien eradication (Van Diggelen et al., 2001). The re-establishment of species accumulation should be selected for site restoration, on condition that the geomorphology and hydrology of the invaded site is still able to sustain a native population (Richardson et al., 2007). Most riverine specialist species are generally spread at a wider range and, in this situation, are mostly resprouters (Ndou and Ruwanza, 2016). Best possible conditions for riverine ecosystems recovery are when the area can quickly recolonize through natural re-vegetation from surroundings sites which are not disturbed (Ndou and Ruwanza, 2016). In this way, the local gene pool maintained and costs on restoration can be saved (Ndou and Ruwanza, 2016).

However, species re-introduction should be practiced using seeds or materials propagated if the landscape in the surrounding is highly degraded (Ndou and Ruwanza, 2016; Holmes et al., 2005). Ndou and Ruwanza (2016) suggest that for an appropriate vegetation structure and resilient plant cover to re-establish in the managed riparian sites, the common riparian generalist shrubs species should be re-introduced. Even though the 'Working for Water' programme has been functional since 1995, it is still not clear as to whether the eradicated areas require post-management restoration activities to help improve native riparian vegetation recovery since little or no monitoring was done (Holmes et al., 2005). This is a concern still, in many areas of the Limpopo province, since invasive species are just cut and native species are expected to recolonize the areas eradicated of the invasive ones passively without any monitoring taking place.

2.7 Restoration

The aim of restoration is to strive to recover a plant population back to a condition like the original one (Holmes and Richardson, 1999) and these are only done in regions with conservation importance, from biome to biome feasible procedures vary. The Centre for Scientific and Industrial Research (CSIR, 2000) provided instructions for native flora restoration following invasion by invasive alien plants. In restoration of fynbos sites previously invaded by alien invasive plant species, fire was found to be effective (Holmes and Foden, 2001). The composition of a community of the soil seed pool beneath impenetrable invasions of Australian acacias in currently infested fynbos was observed to be like that of adjacent uninvaded fynbos at all sites. Recovery to species abundant fynbos flora after the control of alien acacia species was indicated as a good potential (Cilliers et al., 2004). In areas where the indigenous vegetation has been eradicated, the re-establishing capabilities using contributor seed pools or antique seed pools must be investigated (Galatowitsch and Richardson, 2005). Restoration actions are still lacking in the Limpopo Province after the removal of the invasive species. There is, also, lack of monitoring of areas previously infested by invasive plant species, this, therefore, keeps the conservation managers and the other governments responsible for the management of invasive species moving in circles. The invasive species keeps on returning in the eradicated areas, and in large numbers than before eradication took place.

2.8 Passive restoration from seed

Past recommendations have presumed that forest managers can do little to enhance natural aspen seedling establishment, but we suggest that the current prevalence of human activity throughout many forest lands provides exactly this opportunity. Successful passive restoration of aspen from seed does require a naturally occurring seed rain and exacting climatic conditions for dispersal, germination, and early establishment, as well as the appropriate microsites that will provide the necessary resources and conditions for successful establishment. A range of different microsites can be provided through surface soil disturbances created naturally or artificially (Landhausser et al., 2019).

For example, natural surface soil disturbances can result from fire, windthrow, landslides, and floodplains while artificial disturbances can result from forest harvesting and subsequent site preparation, agriculture, mining and energy development, and road construction. Natural seedling establishment has been observed following these events and our observations can be instructive for the development of protocols that may enhance passive seedling recruitment (Landhausser et al., 2019).

The reconstruction of past ecological patterns and dynamics (Rhemtulla and Mladenoff 2007) can provide relevant important information to ecosystem conservation, restoration, and management. Historical ecology research has for quite some time been utilized to establish baseline conditions and set restoration targets (Alagona et al., 2012) and to characterize ecosystem degradation (Swetnam et al., 1999). Furthermore, past studies can serve as a “natural experiment” to research ecosystem response and resilience to historical disturbances and climatic changes (Nogues-Bravo et al., 2018), elucidate the natural range of variability of an ecosystem (Safford et al., 2012), identify persistent and novel sites or features in the contemporary landscape (Copes-Gerbitz et al., 2017), and provide information on lost or forgotten species or ecosystems that might serve as inspiration for current and future management, either in the same place or a location with an analog future climate (Grossinger et al., 2007).

In many cases, surprising results and management recommendations emerging from historical ecology analyses have altered management priorities and strategies (McClenachan et al., 2015).

2.9 Active restoration using planted seedlings

The production of aspen planting stock can be accomplished using either vegetative or seedling-based processes. Vegetative propagation of aspen from root cuttings can be successful, but the production of sufficient planting material has proven to be a significant limitation (Schier et al., 1985), and vegetative propagation does not increase genetic diversity via sexual recombination.

Therefore, the use of aspen seedlings is currently the preferred method for producing aspen planting stock (Macdonald et al., 2015). The collection, processing and storage of seeds are the initial steps in the production of seedling stock, followed by nursery production of the seedlings and their outplanting on site. Following the target seedling concept, the most limiting site factors are used to determine ideal seedling stock types in regard to species- and site-specific traits (Landhausser et al., 2019). However, while seedling traits can influence afforestation success, identifying and potentially mitigating some of the most limiting site conditions, such as competing vegetation, is also crucial for the success of forest restoration using seedlings (Landhausser et al., 2019).

Until recently, the knowledge of seedling production techniques for aspen was based on information available for other commercially important tree species (Landhäusser et al., 2019). Some studies on aspen seedlings and their quality have shown promise in the ability to manipulate physiological and morphological characteristics during nursery culture, which has shown to impact early seedling performance on stressful restoration sites (Landhäusser et al., 2012). Considering that the development of quality seedling stock of conifers took decades to achieve, the development of quality growing stock for less commercially important species such as aspen should be considered to be in its infancy (Landhäusser et al., 2019).

Invasive plants are a major challenge for the maintenance of productive rangelands. The presence of invasive plants can reduce land value by inhibiting biodiversity, depressing forage productivity, and depleting soil and water resources (Silva et al., 2019). However, typical invasive species management approaches such as prescribed fire, herbicide application, mowing and targeted grazing might not be compatible with current rangeland practices or they might demonstrate limited utility in significantly reducing invasive plant cover and reproduction (Silva et al., 2019).

Feasible, cost-effective weed management strategies must be identified through science-based methods to facilitate widespread, successful control of invasive plants by land owners and land managers in rangeland systems (Gornish and James, 2016). Incorporating approaches from the field of ecological restoration into weed management might provide utility for controlling nonnative species. Although uncommon (Kettenring and Adams, 2011), seeding or planting desired plants in invaded areas after an initial weed control treatment holds promise for managing invasive plants (Silva et al., 2019).

This is because reseeding can reduce existing weed establishment while also increasing plant community resistance to future invasion (Funk et al., 2008). This can occur for two reasons. First, seeding species that are functionally similar in resource use to target weeds increases the magnitude of competition experienced by invasive species (Connell, 1983). Second, seeded or planted species can take up space in bare patches created by conventional weed control methods. These patches are often disproportionately invaded due to an absence of competition (Shumway, 1995).

The rapid modification of rangeland health in response to plant invasions highlights a critical need for the development of novel management strategies that enhance successful weed control. Seeding or planting desired species to address plant invasion can provide utility for management by facilitating competitive interactions (Berger, 1993), one of the major factors driving plant invasion (Gioria and Osborne, 2014).

2.10 The public attitudes towards invasive alien species management

The management decisions about the mechanisms for evaluating and responding to public attitudes will become even more crucial as public land management agencies such as the United States National Park Services (US NPS) strive to enforce stakeholder decisions made through research-oriented approach (White and Ward, 2010). The study defines an approach for assessing public attitudes toward invasive alien species management and provides mechanisms for estimating control alternatives based on stakeholder attributes (Kumschick et al., 2012).

Davis et al. (2005) showed that most Cumberland Island National Seashore (CUIS) visitors had little knowledge about the invasive alien species and their impacts on public areas. Even though there was exposure to media, it was very surprising that it did not seem to impact acknowledgement or preference for control of invasive species. In fact, findings of the cluster examination propose that the media's description of invasive alien species matters may in fact increase the assessment of current and future conflicts and impacts of invasive alien plant species on native biodiversity and ecosystems has become a matter of debate in the scientific community (Davis et al., 2005), as well as a priority for managers. The way in which the general public and visitors learn about environmental menaces such as invasive species are insufficient and most of the times, potentially inaccurate (Wittenberg and Cock, 2001).

To fight this problem, enhancement of on-site clarification and off-site advertisement by agencies such as the NPS to the public to help them comprehend effects of IAS and their threats should be done (Teillac-Deschamps et al., 2009). An applied study has already been initiated by the CUIS to examine the impacts of different educational programmes on visitors' assistance for IAS control (Sharp, 2010). However, their study demonstrated that the increasing amount of information and knowledge is not a perfect solution, but improved knowledge may help. If the threats posed and the invasive species responsible can be recognized by a certain individual, such a person may be able to advocate for effective control mechanisms (Somaweera et al., 2010). The likelihood of support for management is increased by the amount of knowledge of an invasive alien species (Garcia-Llorente et al., 2011). For more management support to be generated on CUIS, focus by managers should be on increasing IAS public knowledge with priority on the human function in managing invasive alien species and restoring natural ecosystems (Sharp et al., 2011).

The two categories of eco-centric that surfaced in their investigation may have provided a fragment of informative angles that are found on the conventional partially–biocentric continuation. Although all visitors seem to care about nature conservation, their passion is shown in various ways. For instance, a belief by eco-centric visitors is that each species has a right to persist and they do not really promote any kind of IAS management, they rather promote a hands-off approach in management where flora and fauna is left alone. Flexible eco-centric visitors are liable to promote a hands-on management which includes human involvement to conserve the ecosystem's integrity. A recipe for conflict arises whenever these angles are widely presented in an ecosystem (Sharp et al., 2011).

The Potential for Conflict Index (PCI) evaluated and revealed that this debate needed a course of reaction at CUIS, both from consensus and public acceptance. The regulated directions of interpreting PCI numbers have not yet been implemented (Vaske et al., 2010) and the relative disagreement amongst respondents is currently being provided by PCI metric. From a visitor point of view, the most appropriate and less conclusive control choice was the flexible on-site control of invasive species. The invasive species circumstances at CUIS are different. Control of invasive alien species (IAS) within geographically isolate ecosystems such as islands is much easier (Donlan, 2010), and in less insular environments, the findings and equivalent suggestions may not be appropriate. Furthermore, the exceptionally charismatic invasive species such as drought horses are featured in CUIS (Jarić et al., 2020).

Regardless of their coastal ecosystem effects which are severe, the sustenance for the animals that are still on the Islands has galvanized the allure of horses as a tourism asset (Beever and Brussard, 2000). In 1996, for instance, the proposal which recognized the drought horses as detrimental to CUIS's native areas was dismissed by the Congress (Rikoon and Albee, 1998). In West America, strong public oppositions have also met with the similar drought horse management efforts (Jeffress and Roush, 2010). Inevitably, even flexible eco-centric visitors who are keener to promote on-site control or total removal of IAS may be somehow hesitant to use the same technique to horse eradication. It would be almost impossible to remove an established, more conspicuous invasive alien species such as feral horses, but may not provide such emotional resistance if less publicized (Gilbert, 2012).

For instance, Garcia-Llorente et al. (2011) implied that if a different kind of species is considered in a Spanish protected area, humans were more prepared to pay for management than hindrance of IAS. Though at CUIS, a favourable option appeared not to be eradication, their study indicated that some level of social assistance for IAS removal may, even under these various conditions exist. In the ecosystems such as CUIS that are controlled by the US NPS, prioritized public contribution is very important and is a prerequisite for control proposition (Nimmo and Miller, 2007). However, it was discovered that the benefits from a community-based management approach are not only in protected areas such as national parks (Bajracharya et al., 2006).

Tanentzap et al. (2009) suggested that for a wider range of voices and ideas regarding invasive species control to be captured, the effects and social acceptance of IAS in native populations surrounding parks should be the focus of future researches. Preferences of managers and factors shaping the attitudes should be understood better. Park visitors and the public could generate combined assistance for invasive alien species management. To investigate a wide variety of invasive species and mechanisms, additional studies should be employed which will focus on complete and robust eco-centric perspectives amongst groups of collaborators (Vann-Sander et al., 2016).

2.11 Economic implications of invasive alien plants clearing programmes

The economic assessment of invasive alien species effect considers the adverse outcomes, not encompassing any ability for constructive outcome as a trade-off that may have aggregate economic effect of invasive alien species balance (Bananno, 2016). Human-induced infestation processes are typically more unique, quick, and dramatic than accidentally induced forms (Lockwood et al., 2007). The degree of their environmental and economic effects makes them a focal issue in environment and conservation biology (Bergmans and Blom, 2001). The financial cost of plant invasions at recent levels of invasion in South Africa is estimated to be R6.5 billion per year (De Lange and Van Wilgen, 2010). Prevention of such misfortunes related with the loss of water were fundamentally the purpose behind the commencement of 'Working for Water' programme (Van Wilgen et al., 2011). The effects raise genuine concern among environmentalists, conservationists and land supervisors; essentially because of the modification of agricultural systems, waterways, land cover change, and native biodiversity. Invasive plant species have significantly threatened biodiversity worldwide with tremendous ecological and economic effects particularly at small scale, for example, islands (Pyšek, 1995; Williamson, 1996; Pimentel et al., 2000). The vulnerability of islands to plant invasion has been ascribed to the presence of unstructured communities and poor competitive capacity of native species when faced with invasive alien species (Carlquist, 1965; Lloret et al., 2004).

The principle target of South Africa's 'Working for Water' programme is to decrease the population of invasive species in South Africa by 22% p.a. The programme spent R171.8 million battling *Chromolaena odorata* alone from the programme's establishment in 1995 to 2008 (Van Wilgen et al., 2012). Regardless of this attempt, of all the nation's invasive alien plant species, *C. odorata* moved from ranking 14th in 2000 to 4th in 2010 in terms of infested region (Van Wilgen et al., 2012), emphasizing the troubles associated with managing this species.

From a community or ecosystem point of view, the largest ecological issue caused by invasive alien plants is the destruction of whole ecosystem services (Schmitz et al., 1997). Nonetheless, governments and the public are hesitant to help for the prevention and management of invasive plants, possibly due to an absence of comprehension of the connection amongst nature and economy (Mack et al., 2000).

From an economic point of view, ecosystem disturbance because of invasion may prompt a loss in potential economic yield, that is, lessened harvest generation, fisheries, forestry and animal farming (U.S. Congress, 1993). Destruction of invasive plant species with the final target of recovering local flora is money-consuming (Pimentel et al., 2000). Although it is hard to measure the loss of biodiversity in terms of money value, Pimentel et al. (2000) tried to compile the yearly cost of invasive species in the United States (US). The investigations have uncovered that the US lost over R1 trillion every year because of invasive plant species. In Africa, limited studies have been conducted to evaluate the cost of invasive plant species (Bezeng et al., 2013). The impact, however, goes further than harming just the environment, as an estimate of R600 million is spend annually to clear over 10 million hectares of land in South Africa (Department of Environmental Affairs, 2018). In many countries, agricultural lands are encroached by *L. camara;* the pastures' carrying capacities get reduced and in many agricultural crops, it is regarded as a weed. In Australia, it has invaded almost 4 million ha of agricultural areas (Parsons and Cuthbertson, 1992). This resulted in economic losses of R111 739 320.00 in the early 1980s (Swarbrick et al., 1995).

These disturbing insights have driven the South African government to establish the 'Working for Water' programme, with the single goal of controlling and monitoring invasive alien plants, to preserve water resources and guarantee the security of water supply (Le Maitre et al., 1996; Van Wilgen et al., 1998). The South African government has exhausted over R1 450 530 000.00 on this program between 1995 and 2000. Such monetary consumptions are not well managed on the African continent where basic health, education, and agricultural services are still ineffectively delivered (Bezeng et al., 2013).

Most money is being spent on the Australian Acacia genus, this is because in most of the studies which have been done, it was found that the extent of this genus was higher than any other genus (Marais et al., 2004). The influence of several individuals on costs of clearing was confirmed by the results: It becomes more expensive if one takes too long before clearing the invasive plants. To get more enhanced evaluations of this genus, more work still needs to be done on the costs and *Eucalyptus*' workload. When denser stands are cleared, one could deduce that the return of investment is higher if it is evaluated in terms of expansion in yield. The prospective future effect of impenetrable sites of invasive alien plants on water security were not addressed or the chance cost of waiting until a site is densely invaded. The cost of clearing increases together with the impacts. It is between 3 and 20 times cheaper in costs when clearing invasions which are very dispersed in collation with impenetrable sites (Turpie and Heydenrych, 2000).

The reference values for a unit clearing are very high for the Water Management Areas of the Fynbos Biome where on average, the primary densities are 5% compared with a national average of 14%. The cost of comparative control per hectare for dense stands is lower than that of low levels of infestation. For example, it is almost R9000 in cost for clearing just a hectare of Acacia at 1–5% while it is less R4000 for a 75–100% stand. It is anticipated that removing unit worth for denser invasions in other biomes would be lower than the Fynbos Biomes (Pooley, 2014). The low normal control sizes of populations for the Fynbos Biome can most likely be credited to how a critical level of the clearing happens inside the announced mountain catchment regions (Mountain Catchment Territories Areas Act of 1970) and state forests. Besides tree infestations in the grassland, savanna and Karroo biomes will in general be more aggressive in riparian zones than in uplands, while in the Fynbos Pinus, they aggressively attack upland zones too. Amid the 1970s and mid-1980s, an extremely fruitful control programme was administered by the then Department of Forestry in the last regions. This has prompted invasive alien plants being under control in a significant part of the mountain catchment zones by the late 1980s (Pooley, 2014).

To lessen the cost of reassessment control, active restoration of indigenous vegetation cover needs to be implemented. If the overall value of clearing is more than R8000 per hectare and more than R4000 for the follow-up; as a preliminary assessment, then restoration interventions is warranted in this situation. Preliminary estimates of recolonization value are between R3000 and R6000 p/ha (Pooley, 2014). There will be some instances where it is more valuable to invest in re-colonization if one is to compare the above costs of clearing of R1200 for *Pinus* spp. and R2200 for *Acacia* spp. to lessen the requirement for follow-up. It means that for Acacia, 106 ha out 3519 or 3% of the site treated with an initial population size of 75 – 100% restoration should be considered. Only 27 out of 4536 ha or less than 1% should be considered for restoration for the class of 50 – 75% (Marais and Wannenburgh, 2008).

2.12 Cost and benefits of invasive alien plants

The abundance of cultural and ecological elements can be a very subject set to determine the benefits and costs resulting from invasive species (Shackleton et al., 2007; García-Llorente et al., 2008; Kull et al., 2011). There can be several conflicts of interest between different stakeholders because of both the benefits and costs that invasive alien species have. Therefore, there is a need to carefully consider the management of these species to address the needs of all stakeholders and seek solutions (Zengeya et al., 2017).

All invasive alien species effects should not be treated in the same manner by the policy and management as a requirement, but different types of the invasive species should be differentiated following their financials and advantages as well as the differing contributors who came across these impacts (Van Wilgen and Richardson, 2014). Besides the disservices and ecosystem services provided by invasive species (Vaz et al., 2017), various elements also affect how invasive alien species will impact the well-being, livelihood assets quantity and type, community's initial vulnerability, traits of invasive species, resource availability, land occupancy and the political and ecological frame and other elements (Kull et al., 2011). It is therefore essential to comprehend the various functions of invasive species for native livelihoods and society as this can make policy implementation and management complex (Shackleton et al., 2019).

Disservices supplied or the alteration in the nature and quantity of ecosystem services is brought about by the introduction of invasive alien species, which may then influence the well-being of humans (Vaz et al., 2017). Elements of well-being of humans encompass admission to basic products that can help provide proper life, security, health, freedom of choice and social relations, which are essential factors in the sustainable livelihood framework (Hanines-Young and Potschin, 2010; Smith et al., 2013). For instance, human health can be affected adversely by invasive species through expanding the pervasiveness of illnesses or increase in natural disaster intensities like fires, restrain peoples' options for income generation and can result in the depletion or uncertainty in the provision of natural resources or finances essential for maintaining a living and maintenance results. New resources that may enhance the well-being of some species primarily through providing novel livelihood outcomes can be provided for by some invasive alien species (Palmer et al., 2014; Rodgers et al., 2017).

2.13 Effect of invasive plants on water quality

South Africa’s terrene and freshwater environments have a serious environmental problem which is caused by invasive alien plant species (Bombino et al., 2014). Both surface water overflow and underground water revitalization is lowered because the alien trees and shrubs enhance above ground biomass and evapotranspiration (Görgens and Van Wilgen, 2004). The exceptional height, root profundity and centenarian are the reason of the biomass expansion and evapotranspiration percentage analogous with invasive alien trees, contrasted to the indigenous species that they displace (Calder and Dye, 2001). More acute fires that destroy the ecosystem, soil and cause to extravagant abrasion are caused by the biomass expansion that partners with plant invasions; furthermore, rangelands that support livestock are generally reduced by the invasive alien plant species and wildlife, and significant biodiversity reduction (Richardson and Van Wilgen, 2004).

It was estimated that 3 300 mm^3 of above ground water overflow was reduced because of recent expropriation making up to 7% of the nationwide total (Le Maitre et al., 2000), much of these are from the fynbos (shrubland) and grassland biomes (Van Wilgen et al., 2008). The Drakensberg Mountain grasslands is where much of South Africa's water emerges, and it makes most of these areas predominantly susceptible to invasions, and from a vegetation dominated by fynbos in the Cape Mountains (Turpie et al., 2008). Plants which escaped form profit-oriented plantations and forests on farmsteads are the ones that invaded the Cape fynbos that is, *Acacia, Eucalyptus* and *Pinus* species, and compromising the grasslands' environmental decency by needy realm using control methods, which involve fire regimes and overgrazing, and reclamation and damming swamps (Turpie et al., 2008).

If invasive alien plants were to inhabit the full range of their possible range, the possible reductions in water would have to be more than 8 times (Van Wilgen et al., 2008). The cost of these invasions is very crucial economically, about R6.5 billion annually is estimated, which is about 0.3% of South Africa's GDP of around R2 000 billion and may potentially rise to > 5% of GDP if all the suitable habitats are infested by invasive plants (Le Maitre, 2016).

The significance of these effects has been recognised long ago, which caused the formation of a national programme in 1995 known as the 'Working for Water' to manage invasive alien flora and minimize their detrimental effects (Van Wilgen et al., 2011). In order to prioritize and manage effectiveness to guarantee that these capitals are distributed to regions where they will be more cost-effective, various specifications have been initiated and this programme has a R500 million per annum budget (Roura-Pascual et al., 2009). Specifications encompass effects on water resources and conditions of rangelands, and the need for biodiversity conservation to minimize fire risk and mitigate neediness. The effects of invasive alien plants on groundwater recharge and surface water overflow are understood very much and priorities should be set using them. Even though scientists recognize that water quality as well as quantity is impacted negatively upon by invasive plants, systems like the 'Working for Water' have scarcity of knowledge which prohibited them from prioritizing exercises regarding these species (Lidströmet al., 2016).

2.14 Stream flow and groundwater storage of invasive alien plants infested water courses

Indigenous plant species have lower evaporation rates than invasive alien plant species, therefore these invasive plants, especially trees and shrubs, use most water than the flora they supersede (Malan and Day, 2002). Reduced groundwater reserves and reductions in river flows result from this increased evaporation (Malan and Day, 2002). The dilution capacity is reduced by the depletion of the quantity of water in the river, pollutants and increased concentrations of nutrients, and modified absorption amplitude of the environment and increased salinity (Nagler et al., 2008).

The influence of thinning magnitude during flash floods and high runoff seasons has been the focus of most studies instead of the impacts of the quantity of water reduction. The dilution capacity which is reduced can have impacts on water quality which are severe. Unpleasant taste comes from water that has high salinity and constitutes a well-being threat to humans (DWAF, 1996), causes corrosion of industrial pipes and reduces yield by crops. Point-source contaminants such as heavy metals of high concentrations could be poisonous to the well-being of humans and have tendency to build-up in the riverine ecosystem (Mishra and Tripathia, 2008). Eutrophication processes are brought about by increased nutrient concentrations, which result in algal blooms which are potentially toxic (Rossouw, 2004). Almost quadrupled groundwater salinity was found on the invasion of *Tamarix chinesis* and increasing evapotranspiration in the Colorado River in the United States (Nagler et al., 2008).

The influence of invasive alien plants on the flow of the stream, overflow or groundwater restore have been quantified by only a few studies. The earliest investigations in South Africa (Le Maitre et al., 2000) stream flow reductions because of commercial plantations were used (Dye, 1996) to approximate the invasion influence of virgin lands by the same species. Since invasions also flourish in impenetrable sites with the same make up and biomass approach is justifiable, invasions are not managed to main good wood production while the plantations do. Furthermore, the river floodplain and riparian zones often contain invasions, and high transpiration rate are found in trees (Dye and Jarmain, 2004).

Dye and Jarmain (2004) measured the eradication of invasive trees and shrubs which leads up to an increase in surface runoff that provides support for the earlier studies assumptions. The yearly rainfall and the invasive species distress define the magnitude of reduction. The typical examples of results include the stream flow reduction of 82% in the KwaZulu-Natal Drakensberg 20 years after replacing grasslands with pines (Bosch, 1979); stream flow depletion of 55% in the Western Cape 23 years after sowing pines in fynbos (Van Wyk, 1987); and streams drying up which happened 6 and 12 years after the grassland was completely replaced respectively with eucalyptus and pines in the Mpumalanga Province (Van Lill et al., 1980).

Cullis et al. (2007) assessed that the overall disappearance of feasible water resulting from the invasion of alien flora in catchments obtaining more than 800 mm of precipitation annually is 695 mm^3, 75% of which is caused by invasions of riverine regions by alien floras. Incorporated with invasive plant species away from riparian zones, this only accounts for 4% of the overall recorded water use in South Africa. Less investigations have determined the influence of invasions on water attribute following the expansion in evapotranspiration and, thus, weakened magnitude and river flow decreases. Many influences are presumed and are not endorsed by evaluated results. The site-specific processes are intimately linked with the influence of invasive alien plants on water quality, causal factors are therefore difficult to isolate (Chamier et al., 2012).

2.15 Plant biomass, eutrophication and nutrient cycling

The amount and structure of biomass is changed by the invasive alien plants, and the nutrient dynamics and carbon also change. The growth of invasive alien plants is very rapid, often increasing the proportion of flowers, seed and twigs that after abscission become ‘terrestrial litter’ (Guendehou et al., 2014). This litter is kept in water bodies after it enters, where its measure of examination by the feeding invertebrates as well as bacterial and fungal activity can differ from that indigenous plant processes (Stewart and Davies, 1990). Indigenous species litter input is often lower than that of invasive alien plants which can drastically change the nutrient cycle in ecosystems (Stock et al., 1995).

The amount of metabolized nutrients is often brought about by the expansion in the biomass supplied by invasive flora, leading to a rapid increase of natural processes of eutrophication (Kalff, 2002) as well as rooted aquatic macrophytes and free-floating invasions (Lee, 1973). The systematic alterations in the plant and animal communities is brought about by eutrophication, a slow reduction in water and natural territory condition as well as the development of potentially toxic blooms (Kalff, 2002).

South Africa is generally a water-scarce country and nearly all South African attainable freshwater has therefore, been assigned for use, and it is a water-scarce country (of which the largest users are the mining, agriculture, and industry) (DWA, 2010). The water quality complications at a nationwide scale exacerbated the shortage of available water. These include 71% of highly unacceptably nutrient attentiveness at the country's examination sites, as well as high levels of salinization at 30% in the monitoring sites (DWA, 2011). Inadequately functioning wastewater reception works cause such situations as unserviced areas run-off, runoff from agricultural areas, discharges from industrial wastewater and impacts from mines (DWA, 2011).

The growing scarcity of water and the backdrop of a looming water quality crisis and the influences of invasive alien flora on water quality are likely to be minor. However, an already dire situation is worsened by them; furthermore, aquatic weeds invasion is promoted by eutrophication, decreasing water quality and starting a downward spiral of invasion. In South African water resources, some aquatic weeds have flourished which have been enriched with nutrient by municipal's ineffective treatment plants and effluent sewage (Van Ginkel, 2011). Further deterioration may result in these areas due to these invasive alien plants and exceedance of fitness-for-use thresholds. Lakes Kariba and Victoria aquatic weed invasions (Hill, 1999) are examples of sewage pollution which is enriching the water, which made favourable situations for the outspread of water hyacinth and Kariba weed.

The water quality is further reduced by these invasions, water movement inhibition and becomes a reproduction site for mobile micro-organisms. The invasion of the Western Cape fynbos by *Acacia saligna* is a perfect example of the influence of invading flora biomass-connected on water quality (Jovanovic, 2009). Furthermore, the allelopathic influences documented in species such as *Lantana camara*, *Prosopis* and *Eucalyptus* species have not yet been adequately investigated regarding water quality. It is understandable that the phenol having terrestrial waste is conceivable, especially that of *Eucalyptus* species, its acidity could affect water quality, nitrification processes and organic matter accumulation even though quantitative data has not yet been produced to substantiate these claims (Subbarao et al., 2009).

The implications of invasive alien species such as *Eucalyptus* and *Quercus'* leaf litter decomposition on nutrient dynamics and water quality impacted are yet to be evaluated. The influence of invasive alien flora on water quality and water quantity which are not considered to be as important should also be communicated, mainly in regions where the influences may aggravate already vital water quality problems (Shin-ichiro et al., 2009)

References

Alagona, P.S., Sandlos, J. and Wiersma, Y.F., 2012. Past imperfect: using historical ecology and baseline data for conservation and restoration projects in North America. *Environmental Philosophy* 9(1), 49–70.

Bajracharya, S.B., Furley, P.A. and Newton, A.C., 2006. Impacts of community-based conservation on local communities in the Annapurna Conservation Area, Nepal. *Biodiversity & Conservation* 15(8), 2765–2786.

Beever, E.A. and Brussard, P.F., 2000. Charismatic megafauna or exotic pest? Interactions between Popular Perceptions of Feral Horses (*Equus caballus*) and their Management and Research. In: Salmon, T.P., Crabb, A.C. (Eds.), Proceedings of the 19th International Vertebrate Pest Conference. University of California, Davis, CA, pp. 413–418.

Bennett, B.M., 2014. Model invasions and the development of national concerns over invasive introduced trees: Insights from South African history. *Biological Invasions* 16, 499–512.

Bennett, B.M., 2015. Plantations and Protected Areas: A Global History of Forest Management. The MIT Press, Cambridge, United States of America.

Bergmans, W. and Blom, E., 2001. Invasive plants and animals. Is there a way out? The Netherlands Committee for IUCN, Amsterdam.

Bezeng, B.S., Savolainen, V., Yessoufou, K., Papadopulos, A.S., Maurin, O. and Van der Bank, M., 2013. A phylogenetic approach towards understanding the drivers of plant invasiveness on Robben Island, South Africa. *Botanical Journal of the Linnean Society* 172(2), 142–152.

Bombino, G., Boix-Fayos, C., Gurnell, A.M., Tamburino, V., Zema, D.A. and Zimbone, S.M., 2014. Check dam influence on vegetation species diversity in mountain torrents of the Mediterranean environment. *Ecohydrology 7*(2), 678–691.

Bonanno, G., 2016. Alien species: to remove or not to remove? That is the question. *Environmental Science and Policy 59*, 67–73.

Bosch, J.M., 1979. Treatment effects of annual and dry period streamflow at Cathedral Peak. *South African Forestry Journal* 108, 29–38.

Centre for Agriculture and Bioscience International, 2014. *Populus alba* (silver-leaf poplar). Invasive Species Compendium. http://www.cabi.org.

Calder, I. and Dye, P., 2001. Hydrological impacts of invasive alien plants*. Land Use Water Resource Restoration* 1(7), 1–8.

Carlquist, S., 1965. Island life: A natural history of the islands of the world. Natural History Press, New York, United States of America.

Copes-Gerbitz, K., Arabas, K., Larson, E. and Gildehaus, S., 2017. A multi-proxy environmental narrative of Oregon white oak (*Quercus garryana*) habitat in the Willamette Valley, Oregon. *Northwest Science*, 91(2), 160–185.

Chamier, J., Schachtschneider, K., Le Maitre, D.C., Ashton, P.J. and Van Wilgen, B.W., 2012. Impacts of invasive alien plants on water quality, with particular emphasis on South Africa. *Water SA* 38(2), 345–356.

Chapman, R.A. and Versfeld, D.B., 1998. Invasive Vegetation and Water Resources: The Consequences of a lack of Management to the Water Resources of the Western Cape. Unpublished report. Also available at: http://www.ru.ac.za/departments/iwr/index/byauthor.html.

Cilliers, C.D., Eesler, K.J. and Boucher, C., 2004. Effects of alien plant management and fire on soil seed banks and regeneration in the Cape Peninsula National Park, South Africa. *South African Journal of Botany* 70, 705–712.

Clarke, R., 1994. Human identification in information systems: Management challenges and public policy issues. *Information Technology and People* 7(4), 6–37.

Crous, C.J., Jacobs, S.M. and Esler, K.J., 2012. Drought-tolerance of an invasive alien tree, *Acacia mearnsii* and two native competitors in fynbos riparian ecotones. *Biological Invasions* 14, 619–631.

Cullis, J.D.S., Görgens, A.H.M. and Marais, C., 2007. A strategic study of the impact of invasive alien plants in the high rainfall catchments and riparian zones of South Africa on total surface water yield. *Water SA* 33(1), 35–42.

D'Antonio, C. and Meyerson, L.A., 2002. Exotic plant species as problems and solutions in ecological restoration: A synthesis. *Restoration Ecology* 10, 703–713.

Davis, M.A., Thompson, K. and Grime, J.P., 2005. Invasibility: The local mechanism driving community assembly and species diversity. *Ecography* 28, 696–704.

De Lange, W.J. and Van Wilgen, B.W., 2010. An economic assessment of the contribution of weed biological control to the management of invasive alien plants and to the protection of ecosystem services in South Africa. *Biological Invasions* 12, 4113–4124.

Department of Water Affairs (DWA), 2010. Integrated Water Resource Planning for South Africa. DWA Report No. P RSA 000/00/12910. Department of Water Affairs, Pretoria, South Africa.

Department of Water Affairs (DWA), 2011. Directorate Water Resource Planning Systems: Water Quality Planning. Resource Directed Management of Water Quality. Planning Level Review of Water Quality in South Africa. Sub-Series No. WQP 2.0. Department of Water Affairs, Pretoria, South Africa.

Department of Water Affairs and Forestry (DWAF), 1996. South African Water Quality Guidelines. Volume 1 – Domestic use (2nd edition). Department of Water Affairs and Forestry, Pretoria, South Africa.

Department of Environmental Affairs. (2018). [online] Available at: https://www.environment.gov.za/projectsprogrammes/wfw/research#stats [Accessed 3 May 2018].

Donlan, C.J., 2010. Aliens, globalization and biodiversity. *Trends in Ecology and Evolution* 25, 624–625.

Dye, P. and Jarmain, C., 2004. Water use by black wattle (*Acacia mearnsii*): Implications for the link between removal of invading trees and catchment stream flow. *South African Journal of Science* 100, 40–44.

Dye, P.J., 1996. Climate, forest and streamflow relationships in South African afforested catchments. *Commonwealth Forestry Review* 75, 31–38.

European and Mediterranean Plant Protection Organization, 2015. EPPO report on notifications of non-compliance [WWW Document]. EPPO Report. Serv. no. 07-

2015 Num. Artic. 2015/138. EPPO Secr. (2015-07). URL ⟨https://gd.eppo.int/reporting/article-4936⟩.

Funk, J.L., Cleland, E.E., Suding, K.N. and Zavaleta, E.S., 2008. Restoration through reassembly: plant traits and invasion resistance. *Trends in Ecological Evolution* 23, 695–703.

Galatowitsch, S.M. and Richardson, D.M., 2005. Riparian scrub recovery after clearing of invasive alien trees in headwater streams of the Western Cape, South Africa. *Biological Conservation* 122, 509–521.

García-Llorente, M., Martín-López, B., González, J.A., Alcorlo, P. and Montes, C., 2008. Social perceptions of the impacts and benefits of invasive alien species: Implications for management. *Biological Conservation* 141, 2969–2983.

Garcia-Llorente, M., Martin-Lopez, B., Nunes, P.A.L.D., Gonzalez, J.A., Alcorlo, P. and Montes, C., 2011. Analyzing the social factors that influence willingness to pay for invasive alien species management under two different strategies: Eradication and prevention. *Environmental Management* 48(3), 418–435.

Giam, X., Bradshaw, C.J.A., Tan, H.T. and Sodhi, N.S., 2010. Future habitat loss and the conservation of plant biodiversity. *Biological Conservation* 143, 1594–1602.

Gilbert, O., 2012. *The ecology of urban habitats*. Springer Science & Business Media.

Gioria, M. and Osborne, B.A., 2014. Resource competition in plant invasions: emerging patterns and research needs. *Frontiers of Plant Sciences* 5, 501

Gioe, M., 2006. The horticultural sector in Sub-Saharan Africa: A solution to the decline of commodity prices? *Crossroads* 6, 16–65.

Gitonga, Z.M., Chabi-Olaye, A., Mithöfer, D., Okello, J.J. and Ritho, C.N., 2010. Control of invasive *Liriomyza leafminer* species and compliance with food safety standards by small scale snow pea farmers in Kenya. *Crop Protection.* 29, 1472–1477.

Gornish, E.S. and James, J.J., 2016. Interactions among habitat, management, and demography for an invasive annual grass. *Plant Ecology*, 217(10), 1247–1258.

Gorgens, A.H.M. and van Wilgen, B.W., 2004. Invasive alien plants and water resources in South Africa: Current understanding predictive ability and research challenges. *South African Journal of Science* 100, 27–33.

Government Gazette [of the Republic of South Africa]. 2001. Amendments to the Conservation of Agricultural Resources Act, 1983 (Act No. 43 of 1983). Government Gazette, 429 (22166) of 30 March 2001. Department of Agriculture, Republic of South Africa.

Grossinger, R.M., Striplen, C.J., Askevold, R.A., Brewster, E. and Beller, E.E., 2007. Historical landscape ecology of an urbanized California valley: wetlands and woodlands in the Santa Clara Valley. *Landscape Ecology*, 22(1), 103.

Guendehou, G.S., Liski, J., Tuomi, M., Moudachirou, M., Sinsin, B. and Makipaa, R., 2014. Decomposition and changes in chemical composition of leaf litter of five dominant tree species in a West African tropical forest. *Tropical Ecology* 55(2), 207–220.

Hanines-Young, R. and Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D., Frid, C. (Eds.), Ecosystem

Ecology: A New Synthesis. BES Ecological Reviews Series. Cambridge, United Kingdom. pp.110–139.

Higgins, S.I., Turpie, J.K., Costanza, R., Cowing, R.M., Le Maitre, D.C., Marais, C. and Midgley, G.F., 1997. An ecological economic simulation model of mountain fynbos ecosystems: Dynamics, valuation and management. *Ecological Economics* 22, 155–169.

Hill, M.P., 1999. The world's worst aquatic weed. *Pesticide Outlook* 10(2), 58–62.

Holmes, P.M., Richardson, D.M., Esler, K.J., Witkowski, E.T.F. and Fourie, S., 2005. A decision-making framework for restoring riparian zones degraded by invasive alien plants in South Africa. *South Africa Journal of Science* 101, 553–564.

Holmes, G., Scholfield, K. and Brockington, D., 2012. A comparison of global conservation prioritization models with spatial spending patterns of conservation non-governmental organizations. *Conservation Biology* 26, 602–609.

Holmes, P.M. and Cowling, R.M., 1997a. Effects of invasion by *Acacia saligna* on the guild structure and regeneration of capabilities of fynbos shrublands. *Journal of Applied Ecology* 34, 317–332.

Holmes, P.M. and Cowling, R.M., 1997b. Diversity, composition and guild structure relationships between soil-stored seed banks and mature vegetation in alien plant-invaded South African Fynbos shrublands. *Plant Ecology* 133, 107–122.

Holmes, P.M. and Foden, W., 2001. The effectiveness of post-fire soil disturbance in restoring fynbos after alien clearance. *South African Journal of Botany* 67, 533–539.

Holmes, P.M., and Richardson, D.M., 1999. Protocols for restoration based on recruitment dynamic, community structure and ecosystem function: Perspectives for South African Fynbos. *Restoration Ecology* 7, 215–230.

Jarić, I., Courchamp, F., Correia, R.A., Crowley, S.L., Essl, F., Fischer, A., González-Moreno, P., Kalinkat, G., Lambin, X., Lenzner, B. and Meinard, Y., 2020. The role of species charisma in biological invasions. *Frontiers in Ecology and the Environment* 18(6), 345–353.

Jeffress, J. and Roush, P., 2010. Lethal hoof beats: The rising toll of feral horses and burros. *Wildlife Professional* 4, 50–55.

Jovanovic, N.Z., Israel, S., Tredoux, G., Soltau, L., Le maitre, D., Rusinga, F., Rozanov, A. and Van Der Merwe, N., 2009. Nitrogen dynamics in land cleared of alien vegetation (*Acacia saligna*) and impacts on groundwater at Riverlands Nature Reserve (Western Cape, South Africa). *Water SA* 35(1), 37–44.

Kalff, J., 2002. Limnology: Inland Water Systems. Prentice Hall, Upper Saddle River, New Jersey, United States.

Kettenring, K.M. and Adams, C.R., 2011. Lessons learned from invasive plant control experiments: a systematic review and meta-analysis. *Journal of applied ecology*, 48(4), 970–979.

Kull, C.A., Shackleton, C.M., Cunningham, P.J., Ducatillon, C., Dufour-Dror, J.M., Esler, K.J., Friday, J.B., Gouveia, A.C., Griffin, A.R., Marchante, E., Midgley, S.J., Pachard, A., Rangan, H., Richardson, D.M., Rinaudo, T., Tassin, J., Urgenson, L.S., van Maltitz, G.P., Zenni, R.D. and Zylstra, M.J., 2011. Adoption, use and perception of Australian acacias around the world. *Drivers of Distribution* 175, 822–836.

Kumschick, S., Bacher, S., Dawson, W., Heikkilä, J., Sendek, A., Pluess, T., Robinson, T. and Kühn, I., 2012. A conceptual framework for prioritization of invasive alien species for management according to their impact. *NeoBiota* 15, 69–100.

Landhäusser, S.M. and Lieffers, V.J., 2012. Defoliation increases risk of carbon starvation in root systems of mature aspen. *Trees*, 26(2), 653–661.

Landhäusser, S.M., Pinno, B.D. and Mock, K.E., 2019. Tamm Review: Seedling-based ecology, management, and restoration in aspen (*Populus tremuloides*). *Forest Ecology and Management*, 432, 231–245.

Le Maitre, D.C., De Lange, W.J., Richardson, D.M., Wise, R.M. and Van Wilgen, B.W., 2011. The economic consequences of the environmental impacts of alien plant invasions in South Africa. *Biological invasions: Economic and environmental costs of alien plant, animal, and microbe species*, pp.295-323.

Le Maitre, D.C., Forsyth, G.G., Dzikiti, S. and Gush, M.B., 2016. Estimates of the impacts of invasive alien plants on water flows in South Africa. *Water SA* 42(4), 659–672.

Le Maitre, D.C., Van Wilgen, B.W., Chapman, R.A. and McKelly, D.H., 1996. Invasive plants and water resources in the Western Cape Province, South Africa: Modelling the consequences of a lack of management. *Journal of Applied Ecology* 33, 161–172.

Le Maitre, D.C., Versfeld, D.B. and Chapman, R.A., 2000. The impact of invading alien plants on surface water resources in South Africa: A preliminary assessment. *Water SA* 26(3), 397–408.

Lee, G.F., 1973. Role of phosphorus in eutrophication and diffuse source control. *Water Resource* 7(1-2), 111–128.

Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L. and Geschke, A., 2012. International trade drives biodiversity threats in developing nations. *Nature* 486, 109–111.

Lidström, S., West, S., Katzschner, T., Pérez-Ramos, M.I. and Twidle, H., 2016. Invasive narratives and the inverse of slow violence: alien species in science and society. *Environmental Humanities* 7(1), 1–40.

Lloret, F., Médail, F., Brundu, G. and Hulme, P.E., 2004. Local and regional abundance of exotic plant species on Mediterranean islands: Are species traits important? *Global Ecology and Biogeography* 13(1), 37–45.

Lockwood, J.L., Hoopes, M.F. and Marchetti, M.P., 2007. Invasion Ecology. Blackwell Publishers. Oxford, England.

Macdonald, I.W., 2004. Recent research on alien invasions and their management in South Africa: A review of the inaugural research symposium of the Working for Water Programme. *South African Journal of Science* 100, 21–26.

Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M. and Bazzaz, F.A., 2000. Biotic invasion: Causes, epidemiology, global consequences, and control. *Ecological Applications* 10, 689–710.

Malan, H.L. and Day, J.A., 2002. Linking Discharge, Water Quality and Biotic Response in Rivers: a Literature Review. WRC Report No. 956/2/02. Water Research Commission, Pretoria, South Africa.

Marais, C., Van Wilgen, B.W. and Stevens, D., 2004. The clearing of invasive alien plants in South Africa: A preliminary assessment of costs and progress: Working for Water. *South African Journal of Science* 100(1-2), 97–103.

Marais, C. and Wannenburgh, A.M., 2008. Restoration of water resources (natural capital) through the clearing of invasive alien plants from riparian areas in South Africa—costs and water benefits. *South African Journal of Botany* 74(3), 526–537.

McCarthy, D.P., Donald, P.F., Scharlemann, J.P., Buchanan, G.M., Balmford, A., Green, J.M., Bennun, L.A., Burgess, N.D., Fishpool, L.D. and Garnett, S.T., 2012. Financial costs of meeting global biodiversity conservation targets: Current spending and unmet needs. *Science* 338, 946–949.

McClenachan, L., Lovell, S. and Keaveney, C., 2015. Social benefits of restoring historical ecosystems and fisheries: alewives in Maine. *Ecology and Society*, 20(2).

Mishra, V.K. and Tripathia, B.D., 2008. Concurrent removal and accumulation of heavy metals by the three aquatic macrophytes. *Bioresource Technology* 99(15), 7091–7097.

Nagler, P.L., Glenn, E.P., Didan, K., Osterberg, J., Jordon, F. and Cunningham, J., 2008. Wide-area estimates of stand structure and water use of *Tamarix* spp. on the lower Colorado River: Implications for restoration and water management projects. *Restoration Ecology* 16(1), 136–145.

Ndou, E. and Ruwanza, S., 2016. Soil and vegetation recovery following alien tree clearing in the Eastern Cape Province of South Africa. *African Journal of Ecology* 54(4), 460–470.

Nimmo, D.G. and Miller, K.K., 2007. Ecological and human dimensions of management of feral horses in Australia: A review. *Wildlife Research* 34(5), 408–417.

Nogués-Bravo, D., Rodríguez-Sánchez, F., Orsini, L., De Boer, E., Jansson, R., Morlon, H., Fordham, D.A. and Jackson, S.T., 2018. Cracking the code of biodiversity responses to past climate change. *Trends in ecology & evolution*, 33(10), 765–776.

Palmer, S., Martin, D., De Lauer, V. and Rogan, J., 2014. Vulnerability and adaptive capacity in response to the Asian long horned beetle infestation in Worcester, Massachusetts. *Human Ecology* 42, 265–977.

Parker, C., Cranford, M., Oakes, N. and Leggett, M., 2012. The Little Biodiversity Finance Book. Global Canopy Programme, Oxford.

Parsons, W.T. and Cuthbertson, E.G., 1992. Noxious Weeds of Australia. Inkarta Press Melbourne, Australia.

Pieterse, P.J. and Boucher, C., 1997. A.C.A.C.I.A (A Case Against Controlling Introduced Acacias) - 19 years later. *Southern African Forestry Journal*, 180, 37–44.

Pimentel, D., Lach, L., Zuniga, R. and Morrison, D., 2000. Environmental and economic costs of nonindigenous species in the United States. *Bioscience* 50, 53–65.

Pooley, S., 2014. Burning Table Mountain: an environmental history of fire on the Cape Peninsula. Springer.

Pyšek, P., 1995. Recent trends in studies on plant invasion (1973 - 1993). In: Pyšek, P., Prach, K., Rejmanek, M., Wade, M. (Eds). Plant invasions-general aspects and special problems. SPB Academic Publishing. Amsterdam, Netherland. pp. 223–236.

Rhemtulla, J.M., Mladenoff, D.J. and Clayton, M.K., 2007. Regional land-cover conversion in the US upper Midwest: magnitude of change and limited recovery (1850–1935–1993). *Landscape Ecology*, 22(1), 57–75.

Richardson, D.M. and Higgins, S.I., 1998. Pines as invaders in the southern hemisphere In Ecology and Biogeography of *Pinus*, (Eds.). D.M. Richardson, pp. 450–473. Cambridge University Press. Cambridge, United Kingdom.

Richardson, D.M. and Van Wilgen, B.W., 2004. Invasive alien plants in South Africa: How well do we understand the ecological impacts? *South African Journal of Science* 100, 45–52.

Rikoon, S. and Albee, R., 1998. Wild and free, leave 'em be: Wild Horses and the struggle over nature in the Missouri Ozarks. *Journal of Folklore Research* 35(3), 203–222.

Rogers, P., Nunan, F. and Fentie, A.A., 2017. Reimagining invasions: The social and cultural impacts of Prosopis on pastoralists in southern Afar, Ethiopia. *Pastoralism*, *7*(1), pp.1–13.

Rossouw, J.N., 2004, May. Water quality in the ecological reserve for river ecosystems. In Process, Water Institute of Southern Africa (WISA) Biennial Conference. pp. 2–6.

Roura-Pascual, N., Richardson, D.M., Krug, R., Brown, A., Chapman, R.A., Forsyth, G.G., Le Maitre, D.C., Robertson, M.P., Stafford, L., Van Wilgen, B.W. and Wessels, N., 2009. Ecology and management of alien plant invasions in South African fynbos: Accommodating key complexities in objective decision making. *Biological Conservation* 142(8), 1595–1604.

Subbarao, G.V., Kishii, M., Nakahara, K., Ishikawa, T., Ban, T., Tsujimoto, H., George, T.S., Berry, W.L., Hash, C.T. and Ito, O., 2009. Biological nitrification inhibition (BNI)—Is there potential for genetic interventions in the Triticeae?. *Breeding science* 59(5), 529–545.

Schier, G.A., Jones, J.R. and Winokur, R.P., 1985. Vegetative regeneration. In: DeByle, Norbert V.; Winokur, Robert P., editors. Aspen: Ecology and management in the western United States. USDA Forest Service General Technical Report RM-119. Rocky Mountain Forest and Range Experiment Station*, Fort Collins, Colo.* 119, 29–33.

Schmitz, D.C., Simberloff, D., Hofstetter, R.H., Haller, W. and Sutton, D., 1997. The ecological impact of non-indigenous plants. In: Simberloff, D., Schmitz, D.C.

and Brown, T.C. (Eds). Strangers in paradise. Island Press. Washington DC, United States of America.

Schomers, S. and Matzdorf, B., 2013. Payments for ecosystem services: A review and comparison of developing and industrialized countries. *Ecosystem Services* 6, 16–30.

Silva, A.D., Roche, L.M. and Gornish, E.S., 2019. The use of strip-seeding for management of two late-season invasive plants. *Heliyon*, 5(5), p.e01772.

Shackleton, C.M., McGarry, D., Fourie, S., Gambiza, J., Shackleton, S.E. and Fabricius, C., 2007. Assessing the effects of invasive alien species on rural livelihoods: Case examples and a framework from South Africa. *Human Ecology* 35, 113–127.

Shackleton, R.T., Shackleton, C.M. and Kull, C.A., 2019. The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of environmental management* 229, 145–157.

Sharp, R.L., 2010. The Effects of Two Educational Programs on Visitors' Knowledge of and Attitudes Towards, and Support for Control of Invasive Species on Cumberland Island National Seashore. Unpublished doctoral dissertation. University of Georgia, Georgia

Sharp, R.L., Larson, L.R. and Green, G.T., 2011. Factors influencing public preferences for invasive alien species management. *Biological Conservation* 144(8), 2097–2104.

Shin-ichiro, S.M., Usio, N., Takamura, N. and Washitani, I., 2009. Contrasting impacts of invasive engineers on freshwater ecosystems: an experiment and meta-analysis. *Oecologia*, 158(4), 673–686.

Shumway, S.W., 1995. Physiological integration among clonal ramets during invasion of disturbance patches in a New England salt marsh. *Annals of Botany*, 76(3), 225–233.

Smith, L.M., Case, J.L., Smiht, H.M., Harwell, L.C. and Summers, L.C., 2013. Relating ecosystem services to domains of human well-being: Foundations for a U.S. index. *Ecological Indication* 28, 79–90.

Somaweera, R., Somaweera, N. and Shine, R., 2010. Frogs under friendly fire: how accurately can the general public recognize invasive species? *Biological Conservation* 143, 1477–1484.

Stafford, R., Davies, M.S. and Williams, G.A., 2012. Misinterpreting the potential benefits of aggregation for reducing desiccation in the intertidal: a simple analogy. *Marine Ecology*, 33(4), 512–515.

Stewart, B.A. and Davies, B.R., 1990. Allochthonous input and retention in a small mountain stream, South Africa. *Hydrobiologia* 202, 135–146.

Stock, W.D., Wienand, K.T. and Baker, A.C., 1995. Impacts of invading N2-fixing *Acacia* species on patterns of nutrient cycling in two Cape ecosystems: Evidence from soil incubation studies and 15N natural abundance values. *Oecologia* 101(3), 375–382.

Swarbrick, J.T., Wison, B.W. and Hannan-Jones, M.A., 1995. The biology of Australian weeds. 25. *Lantana camara* L. *Plant Protection Quarterly* 10, 82–95.

Swetnam, T.W., Allen, C.D. and Betancourt, J.L., 1999. Applied historical ecology: using the past to manage for the future. *Ecological applications*, 9(4), 1189–1206.

Tanentzap, A.J., Burrows, L.E., Lee, W.G., Nugent, G., Maxwell, J.M. and Coomes, D.A., 2009. Landscape-level vegetation recovery from herbivory: Progress after four decades of invasive red deer control. *Journal of Applied Ecology* 46(5), 1064–1072.

Tefera, A. and Tefera, T., 2013. Tomato Production in Ethiopia Challenged by Pest. USDA Foreign Agricultural Service. Global Agricultural Information Network (GAIN), Ethiopia.

Teillac-Deschamps, P., Lorrilliere, R., Servais, V., Delmas, V., Cadi, A. and Prevot-Julliard, A.C., 2009. Management strategies in urban green spaces: Models based on an introduced exotic pet turtle. *Biological Conservation* 142, 2258–2269.

Tonnang, H.E.Z., Mohamed, S.F., Khamis, F. and Ekesi, S., 2015. Identification and risk assessment for worldwide invasion and spread of *Tuta absoluta* with a focus on Sub-Saharan Africa: Implications for phytosanitary measures and management. *PLoSOne* 10,19 http://dx.doi.org/10.1371/journal.pone.0135283.

Turpie, J. and Heydenrych, B., 2000. Economic consequences of alien infestation of the Cape Floral Kingdom's Fynbos vegetation. In C. Perrins, M. Williamson and S. Dalmazzone (Eds.). *The Economics of Biological Invasions*, Cheltenham, Edward Elgar, pp. 152–182.

Turpie, J.K., Marais, C. and Blignaut, J.N., 2008. The Working for Water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. *Ecological Economics* 65(4), 788–798.

U.S. Congress, Office of Technology Assessment, 1993. Harmful non-indigenous species in the United States. OTA-F-565. Washington DC, U.S. Congress Government Printing Office, United States of America.

Van Diggelen, R., Grootjans, A.P. and Harris, J.A., 2001. Ecological restoration: State of the art or state of the science? *Restoration Ecology* 9, 115–118.

Van Ginkel, C.E., 2011. Eutrophication: Present reality and future challenges for South Africa. *Water SA* 37(5), 693–701.

Van Lill, W.S., Kruger, F.J. and Van Wyk, D.B., 1980. The effect of afforestation with *Eucalyptus grandis* Hill ex Maiden and *Pinus patula* Schlecht. Et. Cham. on streamflow from experimental catchments at Mokubalaan, Transvaal. *Journal of Hydrology* 48(1-2), 107–118.

Van Wilgen, B.W. and Richardson, D.M., 2014. Challenges and trade-offs in the management of invasive alien trees. *Biological Invasions* 16, 721–734.

Van Wilgen, B.W. and Wannenburgh, A., 2016. Co-facilitating invasive species control, water conservation and poverty relief: Achievements and challenges in South Africa's Working for Water programme. *Current Opinions on Environmental Sustainability* 19, 7–17.

Van Wilgen, B.W., Biggs, H.C., O'Regan, S.P. and Marè, N., 2000. A fire history of the savanna ecosystems in the Kruger National Park, South Africa, between 1941 and 1996. *South African Journal of Science*, 96(4), pp.167–178.

Van Wilgen, B.W., Cowling, R.M. and Le Maitre, D.C., 1998. Ecosystem services, efficiency, sustainability and equity: South Africa's Working for Water programme. *Trends in Ecology and Evolution* 13, 378.

Van Wilgen, B.W., Forsyth, G.G., Le Maitre, D.C., Wannenburgh, A., Kotzé, J.D., van den Berg, E. and Henderson, L., 2012. An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. *Biological Conservation* 148(1), 28–38.

Van Wilgen, B.W., Khan, A. and Marais, C., 2011. Changing Perspectives on Managing Biological Invasions: Insights from South Africa and the Working for Water programme. In: Richardson, D.M. (Ed.), Fifty Years of Invasion Ecology: The Legacy of Charles Elton. Wiley-Blackwell. Oxford, England.

Van Wilgen, B.W., Reyers, B., Le Maitre, D.C., Richardson, D.M. and Schonegevel, L., 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. *Journal of Environmental Management* 89(4), 336–349.

Van Wilgen, B.W., Richardson, D.M., Le Maitre, D.C., Marais, C. and Magadlela, D., 2001. The economic consequences of alien plant invasions: Examples of impacts and approaches to sustainable management in South Africa. *Environment, Development and Sustainability* 3, 145–168.

Van Wyk, D.B., 1987. Some effects of afforestation on stream flow in the Western Cape Province, South Africa. *Water South Africa* 14(1), 31–36.

Vann-Sander, S., Clifton, J. and Harvey, E., 2016. Can citizen science work? Perceptions of the role and utility of citizen science in a marine policy and management context. *Marine Policy* 72, 82–93.

Vaske, J.J., Beaman, J., Barreto, H. and Shelby, L.B., 2010. An extension and further validation of the Potential for Conflict Index. *Leisure Sciences* 32, 240–254.

Vaz, A.S., Kuffer, C., Kull, C.A., Richardson, D.M., Vicente, J.R., Kühn, I., Schröter, M., Hauck, J., Bonn, A. and Honrado, J.P., 2017. Integrating ecosystem services and disservices: Insights from plant invasions. *Ecosystem Services* 23, 94–107.

Versfeld, D.B., Le Maitre, D.C. and Chapman, R.A., 1998. Alien Invading Plants and Water Resources in South Africa: *A Preliminary Assessment.* Report No. TT99/98, Water Research Commission. Pretoria, South Africa.

Waldron, A., Mooers, A.O., Miller, D.C., Nibbelink, N., Redding, D., Kuhn, T.S., Roberts, J.T. and Gittleman, J.L., 2013. Targeting global conservation funding to limit immediate biodiversity declines. Proceedings of the National Academy of Science U.S.A. 110, 12144–12148.

White, P.C.L. and Ward, A.I., 2010. Interdisciplinary approaches for management of existing and emerging human–wildlife conflicts. *Wildlife Research* 37, 623–629.

Williamson, M., 1996. Biological invasion. Chapman and Hall. London, United Kingdom.

Wittenberg, R. and Cock, M.J. eds., 2001. Invasive alien species: a toolkit of best prevention and management practices. CABI.

World Bank, 2012. Expanding Financing for Biodiversity Conservation: Experiences from Latin America and the Caribbean. Washington DC, United States of America.

Yelenik, S., 2000. Ecosystem-level feedback of N-fixing invasive aliens in lowland fynbos. MSc dissertation, University of Cape Town. Cape Town, South Africa.

Zengeya, T., Ivey, P., Woordford, D.J., Weyl, O., Novoa, A., Shackleton, R., Richardson, D. and Van Wilgen, B., 2017. Managing conflict-generating invasive species in South Africa: Challenges and trade-offs. *Bothalia* 47(2), 1–11.

CHAPTER THREE

THE RATE OF REGENERATION OF NATIVE PLANT SPECIES AFTER ERADICATION OF INVASIVE ALIEN PLANT SPECIES (*Acacia decurrens* WILLD.) IN THE WATERBERG DISTRICT MUNICIPALITY, LIMPOPO PROVINCE, SOUTH AFRICA

Abstract

Riparian invasive alien plants are known to compete with native plant species for water, space, daylight, and different other resources by decreasing structural diversity of native vegetation and subsequently changing the functioning of the ecosystem. The aim of this study was to investigate the rate of native plant species recolonization after

the eradication of *A. decurrens*. The investigation was done in the Waterberg District Municipality, Limpopo Province in a farm, which is highly infested with A. decurrens. Twenty-four permanent plots of 10 m^2 were constructed and the A. decurrens individuals in the plots were removed and the area was monitored for a period of 2 years. The size of quadrats was based on the size and distribution of the invasive alien plants which develop in an aggregated form and have exceptionally small canopies.

3.1 Introduction

About half a million acres of wattle trees are grown as an economic crop in the Union of South Africa, primarily for the wattle bark, or its extract, which have valuable tanning properties. The bark reaches maturity when the trees are from eight to ten years of age and the timber is used in the gold mining industry and as fuel. Two species of wattle, originally introduced from Australia where they are indigenous, are grown commercially from natural seed; green wattle, *Acacia decurrens* Willd., and black wattle. *A. mollissima* Willd. They are members of a series of species or varieties according to some taxonomists, known as the *decurrens* group. The chromosome number of *A. decurrens* is 2n=26 (Ghimpu, 1936), and Newman (1934) has pointed out the possibility of this being a secondary polyploid number. The growth of green wattle is generally regarded as being superior to black wattle under adverse conditions in South Africa but, because its bark contains too much red colouring matter, the small percentage of green wattle in cultivation is gradually disappearing in favour of black wattle.

The ecological and economic impact caused by invasion of native plants in Australia is documented by few case studies, despite increasing awareness of the potential of native plants as weeds. The impacts of invasive native plants can include disruption to ecological processes by accelerated biomass accumulation, reduced light penetration, increased nitrification, changed fire intensity and frequency, altered geo-morphological processes, hybridisation with congeners, which can lead to declines in species richness and abundance. Many of these impacts are similar to the invasion of plants originating from other countries (exotic plants). However, quantified impact data on biodiversity values are only published for *Pittosporum undulatum* Vent. (Mullett and Simmons, 1995; Rose and Fairweather, 1997), *Leptospermum laevigatum* (Sol. ex Gaertn.) F. Muell. (Lam and van Etten, 2002) and *Acacia longifolia* (Andrews) Willd.

The ecological impacts of invasive Australian plants, particularly acacias, are best studied in South Africa, where 13 Acacia species are naturalised, and eight species cause widespread transformation of biological communities and ecological processes (Richardson and van Wilgen, 2004). While the same scale of invasion and impacts are yet to be realised from native acacias within Australia, circumstantial evidence indicates that the potential is there. Rapidly expanding populations of *A. longifolia*, *A. dealbata* Link, *A. pycnantha* Benth. and *A. decurrens* Willd. in Western Australia; *A. cyclops* Cunn. ex Don in South Australia; and *A. saligna* (Labill.) W.L. Wendl., *A. baileyana* F. Muell. and *A. longifolia* in eastern Australia indicate broad-scale impacts may be inevitable without the implementation of appropriate control measures.

As invasive native plants are increasingly recognised as problematic in natural vegetation (Adair, 2008) many are subject to suppression programs to protect biodiversity values. Control options vary according to life-form, susceptibility, risk of non-target damage, ease of implementation, size of infestation and outcome targets. Control options for native plants include the use of herbicides, planned fire, grazing, manual removal, biological control, and integrated methods, including the highly effective method (for some woody plants) of ‘rolling’ infestations with heavy equipment, then burning after a period of drying (Muyt, 2001).

3.1.1 Occurrence of species under study

3.1.1.1 Acacia decurrens

Acacia decurrens is indigenous to plateaus of Victoria and New South Wales and temperate littoral to inland zones that are cool but not to xeric or hot regions of inland New South Wales. It occurs in high precipitation zones with 600 – 1,400 mm per year and is generally tolerant to a wide range of environments. In forests and xeric sclerophyll forests in New South Wales, it establishes with plants such as *Eucalyptus punctata* and small leaved *Eucalyptus crebra* (Benson and McDougall, 2001). In territories where it has become naturalised, *Acacia decurrens* is commonly found along the roads, creek lines and in wasteland areas. It also develops in disturbed locations adjacent to bushlands and open forests (Benson and McDougall, 2001). It was broadly propagated in New South Wale, and it is hard to tell whether it is indigenous or naturalised in zones close to its native rangelands.

3.2 Methodology

The study was conducted in the Waterberg district municipality which is found in the north west of Mookgophong town and highly infested with *Acacia decurrens*. It lies at 24.22’S and 28.46’E. As shown in figure 3.1, the Waterberg district municipality is one of the five district municipalities of Limpopo Province in South Africa. The area is mainly dominated by Northern Sotho and Afrikaans speaking people. The population of *Acacia decurrens* was distributed along the stream which passes through the farm. It receives summer rainfall, and the vegetation is that of a savanna biome and includes species such as *Combretum molle, Themeda triandra, Setaria sphacelata, Terminalia sericea* and *Burkea africanum*.

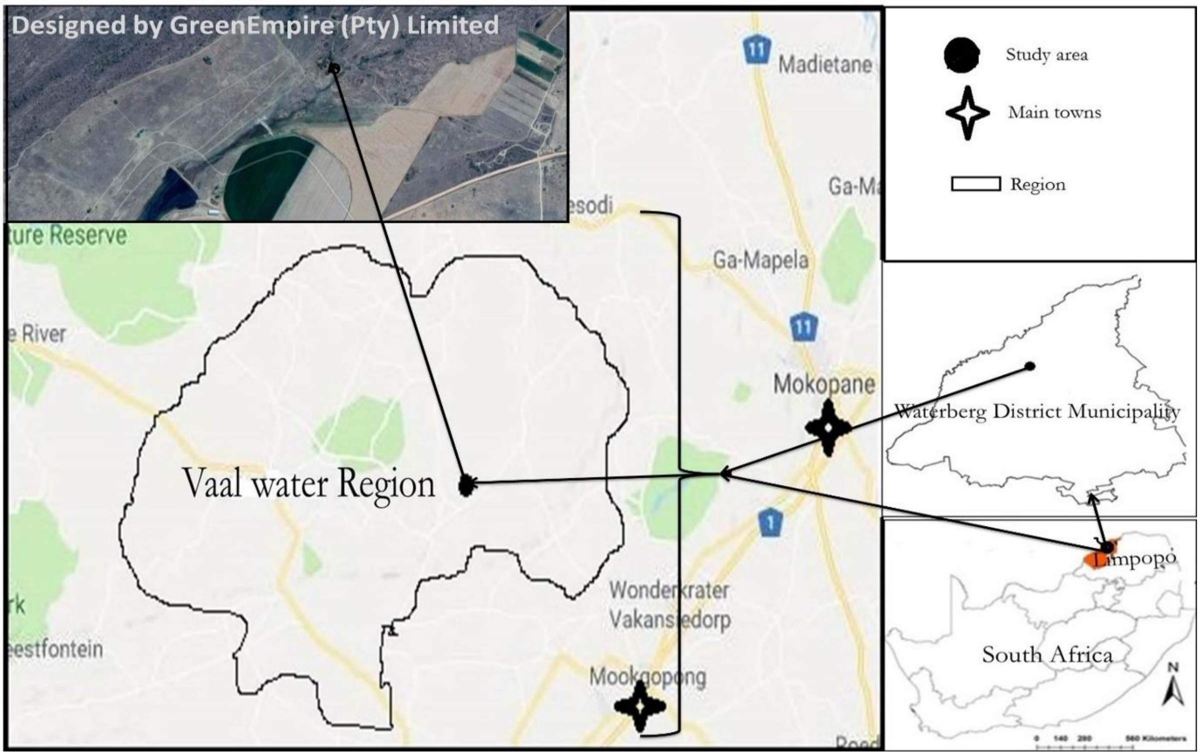

Figure 3. 1: Map of the Waterberg District Municipality showing the study area in the Limpopo Province (Courtesy of the Green Empire (Pty) Limited)

Twenty-four permanent plots of 10 m x 10 m each were constructed along the riparian zone using iron bars as corner posts. The choice of the size of the quadrat was made after assessing the morphology and distribution of invasive alien species under study. The number of alien invasive species and native species present in quadrats were recorded before they were removed. Invasive alien plant species were removed mechanically. No chemicals were used since there are concerns about their adverse environmental implications. The areas were monitored for the re-establishment of alien invasive species so that they could be removed when they reoccur. Only native plant species were left alone once they establish themselves in the study site. All data collected were entered and stored in Microsoft Excel 2010, which were also used in the descriptive statistical analysis of the results. They were then analized using Primer V6 and PERMANOVA.

Figure 3.2 shows a population of the wattle which re-sprouted a few months after the fire which occurred on the site which had dry trees felled by the personnel from the 'Working for Water' program. The trees were cut in June 2015 and by December the same year, a fire of high intensity occurred due to the litter from fallen trees.

Figure 3.2: Picture showing the constructed quadrat in the *Acacia decurrens* population.

3.2.1 Description of the species under study

3.2.1.1 Acacia decurrens

Acacia decurrens is indigenous to the New South Wales where it is found primarily on the coast and plateau from the Hunter Valley south to the Australian Capital Domain. This distribution is outside the target region, however, not a long way from its eastern border. *Acacia decurrens* has been widely cultivated as an ornamental species and has turned out to be naturalized in numerous regions, both within Australia and abroad. In Australia, acclimatized individuals are found in southwest Western Australia, southeast South Australia, and southeast Queensland, parts of New South Wales, the Australian Capital Domain, Victoria and Tasmania. Kodela and Tindale (2001) provided maps of both the natural and naturalized distributions. Presently, conventional wattle orchards cover 130 000 ha in the KwaZulu-Natal and Mpumalanga areas. The species is very invasive by its nature (Figure 3.3). It reproduces large number of hard-covered seeds which are generally long-lived and are spread rapidly down water bodies and across the activity of soil. Invasions are in all zones in South Africa where the annual precipitation surpasses 500 mm. The province normally influenced are the Western Cape, Eastern Cape, KwaZulu-Natal and Mpumalanga, however parts of provinces like Free State, Gauteng and Limpopo are likewise influenced (De wit et al., 2001).

Figure 3.3: A dense population of *Acacia decurrens* trees at the study site before initial clearing.

3.2.1.2 Establishment

Acacia decurrens can be effortlessly grown from seeds but, similarly as with many acacias, the seeds must undergo some pre-treatment to promote germination. The best strategy is to heat the seeds at 100˚C for 1 minute in a relatively vast volume of water (Richardson et al., 2015). The seeds must then be allowed to cool and consume water for 24 hours. Seeds that float to the surface are mostly not potent and ought to be disposed of while viable seeds ought to swell and sink. The seeds can likewise be shaken (Wrigley and Fagg, 1996) and mechanical scratching of the seed coat is an alternative. The average viability of *A. decurrens* seeds, considering laboratory tests of 12 provenances, is 57 000 seeds for every kilogram and optimum temperature for ideal germination is 25˚C. Germination starts after five days and every viable seed generally develops within 25 days (Richardson et al., 2015).

3.2.1.3 Impacts

There are numerous effects related with black wattles in South Africa. A considerable lot of these emerge from formal plantations, yet a few (such as fuel wood, charcoal and materials for building) may be derived from sites of invasion. There are different kinds of negative effects that can be ascribed to both wattle estates and invasion, for instance; it affects both lower surface overflow and water accessibility, and effect on biological diversity (De wit et al., 2001).

3.3 Results

3.3.1 Species Richness

Significant difference in species richness was observed between the sampling periods. The observations of the eradicated sampling plots showed that species richness in the dug area of *Acacia decurens* was found to be 09 (Sep-2016) before eradication and 15 (Jun-2017), 11 (Sep-2017), 21 (Dec-2017) and 13 (Mar-2018) post eradication **(Table 3.1).**

Table 3.2: Floral species richness during the sampling periods following eradication of Acacia decurens

Response variable	Treatment	Sep-2016	June-2017	Sep-2017	Dec-2017	Mar-2018
Species richness	Digging	09	15	11	21	15

Significant decrease in number of *A. decurens* was accompanied by a significant increase in number of other species (Table 3.1, Figure 3.4). Highest number of the other species were observed during the same period as the lowest number of *A. decurens* (2017-Dec).

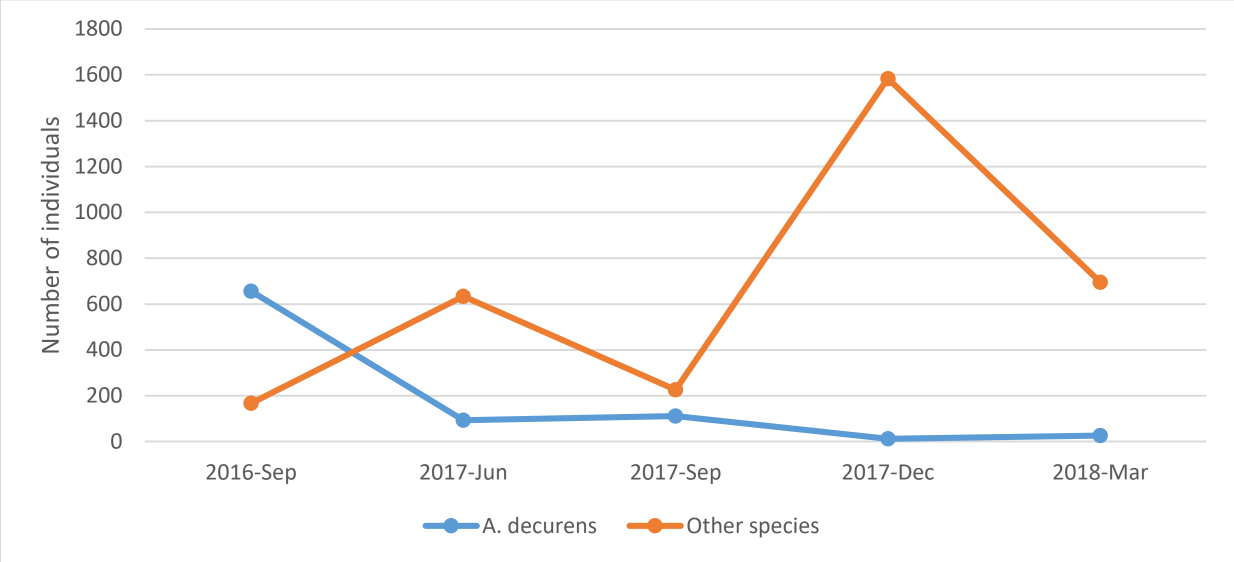

Figure 3.4: Line graph showing the number of *Acacia decurens* and other plant species pre (Sep-2016) and post eradication.

The MDS plot of species composition clearly indicated the different in species richness between the sampling periods (Figure 3.5). More species are observed in 2017-Dec, which is like what is presented on species richness table (Table 3.1). The similarity between the sampling periods was primarily attributed to species such as *A. decurens, Cleome, B. pilosa* and *S. pinnata* which were the dominant plant species among the quadrants during the sampling periods.

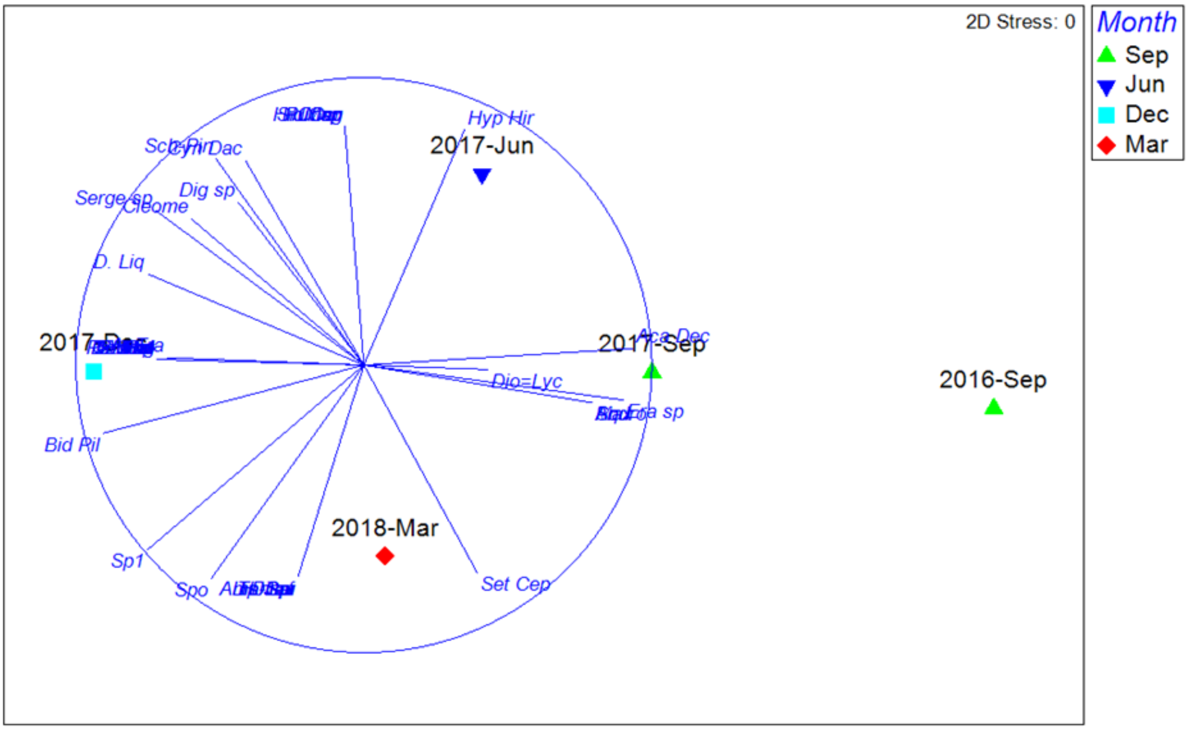

Figure 3.5: Multidimensional scaling ordination plots of species composition in different sampling periods between Sep-2016 and Dec-2017 following the eradication of *Acacia decurens*.

3.3.2 Frequency

The five sampling periods hosted different species before and after digging. Frequency was high after the eradication of *A. decurens* (Table 3.2). There was an increase of six species (*Cleome* sp, *Mupatsapatsane, Digitaria sp, Schkurhia pinnata, Purple sp, Hibiscus cannabis, Bidens pilosa, Sida cardifolia, D. liquidios and Congesta*). This was because when *A. decurens* was dug, new species were observed with only *A. decurens* observed during all the sampling periods, followed by *S. cephaselata, Cleome, S. pinnata* and *B. pilosa* observed during four of the sampling periods. These species are the ones that seem to be able to resist or compete with *A. decurens*.

Table 3.2: Frequency of native plant species following eradication of *Acacia decurens*

Species name	2016-Sep	2017-Jun	2017-Sep	2017-Dec	2018-Mar
Acacia decurens Willd.	100	100	100	66.666	100
Eragrostis spp.	100	0	83.333	0	0
Setaria sphacelata L.	33.333	16.666	33.333	0	100
Andropogon distachyos L.	16.666	0	0	0	0
Diospyros lycioides Desf.	16.666	0	16.666	0	0
Cynadon dactylon (L.) Pers	16.666	33.333	0	33.333	0
Cleome gynandra L.	0	100	50	100	100
Bankisa squarossa R.Br	33.333	0	0	0	0
Hyparrhenia hirta (L.) Stapf	16.666	50	16.666	0	0
Mupatsapatsane	0	16.666	0	0	0
Lippia javanica (Burm.f.) Spreng	0	0	0	0	16.666
Maximiliana martiana H.Karst.	33.333	0	0	0	0
Digitaria eriantha Steud.	0	83.333	0	16.666	33.333
Scholtzia capitata F.Muell. ex Benth.	0	66.666	50	33.333	33.333
Hibiscus cannabis L	0	100	0	16.666	0
Bidens pilosa L.	0	66.666	33.333	100	100
Amaranthus spinosus L.	0	0	0	0	33.333
Corchorus tridens L.	0	0	0	16.666	0
Hypoxis hermirocallidae Fisch.Mey. & Avé-Lall.	0	0	0	16.666	0
Mutomane (herb spp.)	0	0	0	16.666	0
Eragrostis panicoides (J.Presl) Steud.	0	0	0	16.666	0
Panicum coloratum L.	0	0	0	33.333	0
Sida cordifolia L.	0	16.666	0	0	33.333
Digitaria leucitis (Trin.) Henrard	0	66.666	33.333	66.666	0
Aristida congesta Roem. & Schult.	0	16.666	0	0	0
Cyperus spp.	0	16.666	0	16.666	0
Herb Spp1	0	0	50	33.333	66.666
Sambucus canadensis L.	0	0	0	16.666	0
Eschscholzia califonica Cham.	0	0	0	16.666	0
Asparagus asparagoides (L.) Druce.	0	0	0	33.333	0
Sporobolus africanus (Poir.) Robyns & Tournay	0	0	0	33.333	100
Digitaria singuinalis (L.) Scop.	0	0	0	16.666	0
Physcomitrella patens (Hedw.) Bruch & Schimp.	0	0	0	16.666	0
Aristida transvaalensis Henrard	0	0	0	0	16.666
Dispyros spp.	0	0	0	0	16.666

Figure 3.6: Study site with felled *Acacia decurrens.*

The two figures above (Figure 3.6 (a) and (b)) are showing the study site after two different events had occurred. In Figure 3.6 (a) *A. decurrens* litter, after the people from the ‘Working for Water’ Programme had cut some and ring-barked some of the trees and applied the chemicals. This was before the fire occurred and burnt all the litter and (b) the study site just after the fire which was said to have occurred accidentally in December 2015.

Figure 3.7: Some events at the study area.

Figure 3.7(a) shows part of the study site few weeks after the fire occurred, (b) shows numerous seedlings germinating on the study site just few weeks after the fire which removed the litter left by the people from the WfW programme after removing the invader, (c) shows the study site colonized by grasses and small herbaceous plants after 12 months monitoring the area and (d) shows the trees found adjacent to the study site showing *A. decurrens* monopolizing the area which was not monitored.

No grasses or any other plant species were seen growing in the same vicinity as *A. decurrens* except for other invader plant species such as the *Eucalyptus*. After the eradication, grasses started to colonize the area (Figure 3.8).

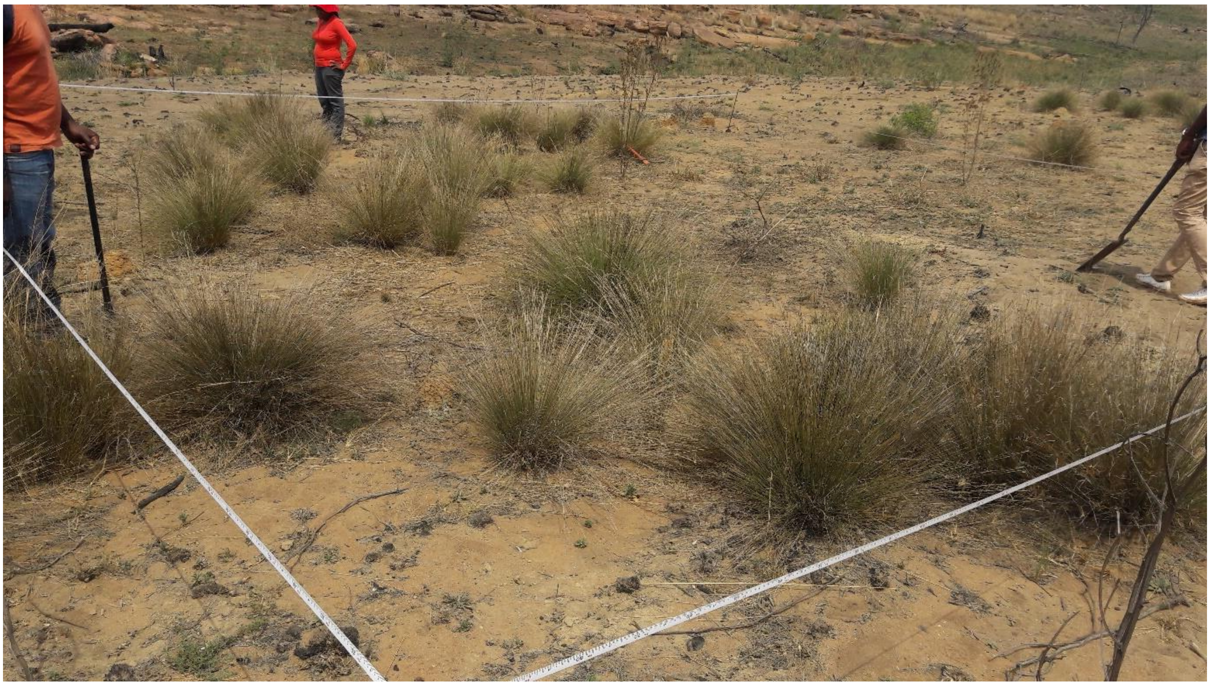

Figure 3.8: Grasses which colonized the area previously infested with *Acacia decurrens* trees

Some of the herbaceous species that occupied the area disappeared during the study, but these grasses remained persistent throughout. This area was covered with only an impenetrable layer of *A. decurrens* plant species before the experiment was conducted.

3.4 Discussion

The fundamentally lower native species spread for controlled sites is the same with other different investigations where invasive alien vegetation was eradicated from riparian zones (Harms and Hiebert, 2006; Galatowitsch and Richardson, 2005). As anticipated, the reference zones had less invasive alien species spread than infested and controlled regions. The lower native species spread inside controlled sites was influenced by the presence of alien species. Since the investigation assessed the early recuperation of species (1 to 4 years), it is foreseen that the moderate establishment of riparian species that have endured invasion and control will increase their spread as time advances. Much of the time, restoration of invasive alien invaded riparian vegetation in South Africa is genuinely crude and its objective is basically to return key components of ecosystem functioning. This investigation indicated that there is potential for self-repair, even in regions that have been intensely infested. Not all components normally return unaided. These regions need re-introduction of species to complete the restoration procedure. The benefit of maintaining a seed source for the woody, animal-dispersed riparian species cannot be neglected (Wassie and Teketay, 2006). It is proposed that in localities where such seed sources are inadequate with regards to re-introductions through planting, propagated specimens ought to be a part of the post-disturbance restoration process. Without this fundamental step, restoration is probably going to be fruitless, and may result in the re-establishment of invasive alien plants (Vosse et al., 2008).

Herbaceous and grass species were seen colonizing the study site after the clearing of alien invasive plant species. The brief timeframe of this investigation makes it hard to completely investigate the seral (changes in species composition through time) changes in the native vegetation. The significant fact with respect to the restoration of post-clearance is that recolonization of vegetation is generally by native species instead of the unpleasant condition of complete secondary invasion by the same or other alien species. Where clearing of invasive alien plant species was not performed, there was no establishment by any other plant species except for *Acacia decurrens*.

This demonstrates that *A. decurrens* affects the indigenous plant populations recovery, as it hinders their existence in its vicinity. However, the large amounts of discovered total native re-establishment shows that the mechanism demonstrates a very important level of flexibility to disturbance by invasive alien plants. Flexibility is explained as the capacity of a vegetation to recover to its previous condition after a disturbance or pressure (Wali, 1999). Numerous riverine species are naturally flexible because of regular and severe disturbance (Richardson et al., 2007) and have distribution and formation mechanisms, for example, the capacity to colonize virgin landscape and hostile clonal development that allows for quick recuperation following a disturbance (Naiman and Decamps, 1997). Continued monitoring and eradication of invasive alien plants by ‘Working for Water’ programme is a vital component that equips this flexibility in the mechanism, as the repetitive clearing exhausts the invasive alien species seed pools and keeps invasion at levels that are generally easier to control provided it is done before they reach reproductive stage.

More than 1000 native plant species are pressurized by invasive alien species (Raimondo et al., 2009), and if invaders somehow managed to achieve total degree of their possible dissemination, the total biological diversity in the Cape floristic region (CFR) could be lowered by an estimated 40% (Van Wilgen et al., 2008). The greater part of the regions’ watersheds exists in protected zones, where progressing invasion by trees and bushes is compromised, thereby decreasing surface water overflow by as much as 36% (whenever permitted to achieve the full degree of their possible dissemination), with significant financial effects (Van Wilgen et al., 2008). Considering worries about the loss of water reservoirs and biological diversity, the South African Department of Water Affairs initiated an expansive programme to cut weeds in 1995.

This programme 'Working for Water' which works nationwide and inside the Cape Floristic Region (CFR), gives funding to the control of invasive alien plants both within and outside the protected regions. In sites where the programme has been active in the CFR, there are signs that the site infested by invasive plants has been lowered by practically half (McConnachie et al., 2016). However, the program has just achieved a little amount (4– 13%) of the total infested region (Van Wilgen et al., 2012). At the CFR's scale of protected regions, there has been no endeavour up to so far to precisely measure the size of the issue, or the expense of management, nor has it been conceivable to evaluate advancement regarding lowering the infestation because of the shortage of a follow-up observation scheme (Van Wilgen and Wannenburgh, 2016).

The fire that occurred played a huge role on the re-establishment of A. decurrens in the same area they previously infested, as it evident that the success of the seeds of this plant speciesdepends on them being pre-treated with fire. That is why there were a lot of seedlings just after the study site experienced fire.

3.5 Conclusion and Recommandations

This study indicates that much can be learned about the success and direction of the recolonization path at an early-stage of restoration. In order to correct ecosystems that are either recovering slowly or are going in a different direction off the desired recolonization path, early-stage evaluations must be conducted to gain insight into the status and path that the recolonization project is currently on.

At early-stages of recolonization, success is mostly dependent upon the survival and establishment of native vegetation, either through (active restoration) plantings or (passive restoration) natural regeneration and is therefore largely dependent on the management techniques used to promote ecosystem recovery. Longer-term success is dependent on the status of earlier-stage recolonization results, the application of ongoing management techniques, and the presence (or absence) of further ecosystem disturbance. Although complete success is yet to be determined, evaluating early-stage results provide a chance for midcourse correction, where unsuccessful recolonization can be sped up or steered back on track by altering management techniques and on-site ecosystem conditions.

In conclusion, from the results obtained during the investigation, it is possible for recolonization by indigenous plant species to occur in the regions previously infested with invasive alien plant species such as *Acacia decurrens*, though follow-up monitoring of the controlled areas needs to be practiced after the eradication of invasive alien plant species.

References

Adair, R.J., 2008. Biological control of Australian native plants, in Australia, with an emphasis on acacias. *Muelleria*, 26(1), pp.67–78.

Benson, D. and McDougall, L., 2001. Ecology of Sydney plant species. Part 8. Dicotyledon families Rutaceae to Zygophyllaceae. *Cunninghamia*, 7(2), 241–462.

De Wit, M.P., Crookes, D.J. and Van Wilgen, B.W., 2001. Conflicts of interest in environmental management: estimating the costs and benefits of a tree invasion. *Biological invasions*, 3(2), 167–178.

Galatowitsch, S.M. and Richardson, D.M., 2005. Riparian scrub recovery after clearing of invasive alien trees in headwater streams of the Western Cape. *Biological Conservation* 122, 509–521.

Cooper, D.C., 1936. Chromosome numbers in the Leguminosae. *American journal of botany*, pp.231–233.

Harms, R.S., and Hiebert, R.D., 2006. Vegetation response following invasive Tamarisk (*Tamarix* spp.) removal and implications for riparian restoration. *Restoration Ecology* 14, 461–472.

Kodela, P.G. and Tindale, M.D., 2001. *Acacia dealbata* subsp. *subalpina* (Fabaceae: Mimosiodeae), a new species from south-eastern Australia. *Telopea* 9:319–32.

Koenig, R., 2009. Unleashing an army to repair alien-ravaged ecosystems. *Science* 325, 562–563.

Lam, A. and van Etten, E., 2002. Invasion of indigenous vegetation in south-western Australia by *Leptospermum laevigatum* (Myrtaceae). In Proceeding of the 13th Australian Weeds Conference pp. 545–548.

McConnachie, M., Van Wilgen, B.W., Ferraro, P.J., Forsyth, A.T., Richardson, D.M., Gaertner, M. and Cowling, R.M., 2016. Using counterfactuals to evaluate the cost-effectiveness of controlling biological invasions. *Ecological Applications* 26, 475–483.

Mullett, T. and Simmons, D., 1995. Ecological impacts of the environmental weed sweet pittosporum (Pittosporum undulatum Vent.) in dry sclerophyll forest communities, Victoria. *Plant Protection Quarterly*, 10(4), 131–138.

Muyt, A., 2001. Bush invaders of south-east Australia: a guide to the identification and control of environmental weed found in south-east Australia. R and FJ Richardson: Melbourne.

Naiman, R.J. and Décamps, H., 1997. The ecology of interfaces: Riparian zones. *Annual Review of Ecology and Systematics* 28, 621–658.

Newman, H.H., 1934. Dermatoglyphics and the problem of handedness. *American Journal of Anatomy*, 55(2), 277–322.

Raimondo, D., Van Staden, L., Foden, W., Victor, J.E., Helme, N.A., Turner, R.C., Kamundi, D.A. and Manyama, P.A., 2009. Red list of South African plants. Strelitzia 25. South African National Biodiversity Institute. Pretoria, South Africa.

Richardson, D.M. and Van Wilgen, B.W., 2004. Invasive alien plants in South Africa: How well do we understand the ecological impacts? *South African Journal of Science* 100, 45–52.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P. and Hobbs, R.J., 2007. Riparian zones degradation, alien plant invasions and restoration prospects. *Diversity and Distributions* 13, 126–139.

Richardson, D.M., Le Roux, J.J. and Wilson, J.R., 2015. Australian acacias as invasive species: Lessons to be learnt from regions with long planting histories. *Southern Forests: A Journal of Forest Science* 77(1), 31–39.

Rose, S. and Fairweather, P.G., 1997. Changes in floristic composition of urban bushland invaded by *Pittosporum undulatum* in northern Sydney, Australia. *Australian journal of botany*, 45(1), 123–149.

Van Wilgen, B.W. and Wannenburgh, A., 2016. Co-facilitating invasive species control, water conservation and poverty relief: Achievements and challenges in South Africa's Working for Water programme. *Current Opinion on Environmental Sustainability* 19, 7–17.

Van Wilgen, B.W., Forsyth, G.G. and Le Maitre, D.C., 2008. The prioritization of species and primary catchments for the purposes of guiding invasive alien plant control operations in the terrestrial biomes of South Africa. Unpublished report. CSIR Natural Resources and the Environment. Stellenbosch, South Africa.

Van Wilgen, B.W., Forsyth, G.G., Le Maitre, D.C., Wannenburgh, A., Kotzé, J.D.F., Van den Berg, E. and Henderson, L., 2012. An assessment of the effectiveness of a large, national scale invasive alien plant control strategy in South Africa. *Biological Conservation* 148, 28–38.

Vosse, S., Esler, K.J., Richardson, D.M. and Holmes, P.M., 2008. Can riparian seed banks initiate restoration after alien plant invasion? Evidence from the Western Cape, South Africa. *South African Journal of Botany* 74, 432–444.

Wali, M.K., 1999. Ecological succession and the rehabilitation of disturbed terrestrial ecosystems. *Plant Soil* 213, 195–200.

Wassie, A. and Teketay, D., 2006. Soil seed banks in church forests of northern Ethiopia: Implications for the conservation of woody plants. *Flora-Morphology, Distribution, Functional Ecology of Plants*, 201(1), 32–43.

Wrigley, J.W. and Fagg, M., 1996. Australian Native Plants: Propagation. Cultivation and Use in Landscaping, 4th edition, Reed, Australia.

CHAPTER FOUR

THE RATE OF RECOLONIZATION BY NATIVE PLANT SPECIES AFTER THE ERADICATION OF THE INVASIVE ALIEN PLANT SPECIES (*Populus alba* L.) IN THE NYLSVLEY NATURE RESERVE, LIMPOPO PROVINCE, SOUTH AFRICA.

Abstract

Biological invasion is one type of ecosystem modification which can, to some degree, be effectively controlled. This incorporates the identification, number and invasion standings of invasive alien species, and additionally the drivers and procedures of invasive alien species introduction. Twenty-four permanent plots of 10 x 10 square meters each were constructed. The number of invasive alien species and indigenous species present in the quadrats were recorded before the plots were cleared. The invasive alien plant species were removed mechanically. There was no significant difference between the numbers of *Populus alba* (White poplar) recorded before the removal in 2016 and after its removal in 2018. The invasive alien plant species still came back in large numbers.

4.1 Introduction

Invasion by invasive alien species is one type of modifying the environment such that, their effective control is almost impossible (Tu, 2009). Such control is basically reliant on data which is adequate and a comprehension of the origin, nature and size of invasion (McGeoch et al., 2010). This incorporates the identification, number and invasion position of invasive alien species, and additionally the drivers and procedures of invasive alien species introduction (Kolar and Lodge, 2001). For the occurrence of distinct stage of development in invasion, further consideration must be made whereby a species can be native in a novel territory for a significant measure of territories for invasion, biotic interaction and surroundings state. Preventing invasion is by a wide margin the most cost-effective and efficient mitigation suggestion, especially in protected zones (Tu, 2009). Discovering what drives formation, the quantities of species that turn out invasive, and how this differs throughout the taxonomic groups, gives a fundamental point for the expression of management approaches and suitable strategy.

Populus alba (white poplar), in several other cases, was not considered a control priority or as a problematic species. It receives lower priority for control and has a less impact potential in localities where it has not been planted extensively and does not have the possibility to hybridize with indigenous aspens. Thirty-five Canadian botanists answered a survey, where most of the interviewees stipulated that white poplar was not a "problem species" and was only invasive on a local scale. The investigation was sent to botanists around Canada, but the regional dissemination of interviewees was not described. In Farmington, western Maine, white poplar was relatively rare, and spread of duplicate was easily pursued back to sites where white poplar was sowed. It was revealed by investigations of vegetation in New London County and Connecticut that white poplar populations were generally restricted to disturbed sites, with stable community magnitudes (Owfi, 2017).

Underground growth of white poplar was reported to have adverse effects on buildings. The problems near houses have been caused by the extensive white poplar root system or other urban developments. It has been reported by several sources that the roots of white poplar can blog drains, sewages, and water passages (Alarcon and Sassenrath, 2016). In his arboreous plants manual, Dirr (1998) stipulated that white poplar "becomes a nuisance and liability after a time". Dirr (1998) further proposed that home owners "avoid this pest". A brochure assembled by England's Forestry Commission produced a pamphlet reporting that soil moisture can be removed by white poplar rapidly during dry, hot days. In London and Essex which are low rainfall areas, shrinkage and rapid drying of clay soils were caused by white poplar, which can upset dwelling foundations (Herbert, 1968).

As white poplar stands expand, the impacts on associated vegetation and age may change. Through productive root germination, dense stands of white poplar may be formed, which can reduce species diversity by crowding and shading the native vegetation (Weber, 2003). Vegetation can be damaged by the breakage of brittle white poplar, as the stands age (Mehrhoff et al., 2003). In the central Transvaal area of South Africa, where white poplar is non-native, there is "marked correlation between the occurrence of naturalized and planted white poplar", but white poplar does not exist in solitary territories; alternatively, it exists on the rest of the river location and has outspread from the water boundary too far outside the riparian ecosystem. In its formation of "absolutely pure stands", white poplar out-competed and suppressed the existing vegetation (Wells et al., 1980). The aim of this study was to investigate the rate at which the native plant species would recolonize the area which was previously infested with invasive alien plant species (*Populus alba*) and to assess the factors to consider for successful recolonization or restoration of the invaded site.

The invasive species are those species that influence adversely on biological diversity which may also vary. It is said that relatively a smaller number of invasive alien species are thought to be invasive (Richardson and Pysek, 2006). Richardson et al. (2005) discovered to some degree, distinctive indicators of quantities of invasive versus alien plant species richness throughout Quarter Degree Grid Cells (QDGCs) in South Africa. Human factors were connected to invasive species richness and it was better estimated by environmental elements.

4.1.1 Occurrence of species under study

Populus alba is a species indigenous in riparian grasslands and coastal forests of central and southern Europe. Its wide range covers from North Africa to Poland and from the Iberian Landmass to western Siberia and Focal Asia (Praciak, 2013). It was propagated in the United States in the eighteenth century as a shade and ornamental plant and more recently in every other continent, getting to be naturalized in numerous territories and invasive in certain nations (Bossard et al., 2000).

It grows very quickly and is a light demanding plant colonizing forest edges and open sunny territories including meadows, wetlands and riparian zones. It establishes in many areas, enduring from waterlogged to dry territories and from acidic to strong alkaline soils, but developing in shrub form in extreme conditions. It performs with high developing rates on ideal destinations characterised by great water availability and much textured soils that have neutral-alkaline pH and rich in nutrients (Flocca et al., 2004).

4.2 Methodology

4.2.1 Description of the study area

The study was conducted in the Waterberg District Municipality, which lies at 29.39'S and 28.45'E. It is situated 10 km south of south-east of the Mookgophong town in the Limpopo Province, South Africa (Figure 4.1). Nylsvley Nature Reserve is a savanna ecosystem situated in the upper part of the Nyl River and includes part of the country's largest floodplain. The avifaunal diversity which is found in this nature reserve is exceptional, with over 370 species recorded (Coetzee et al., 1977).

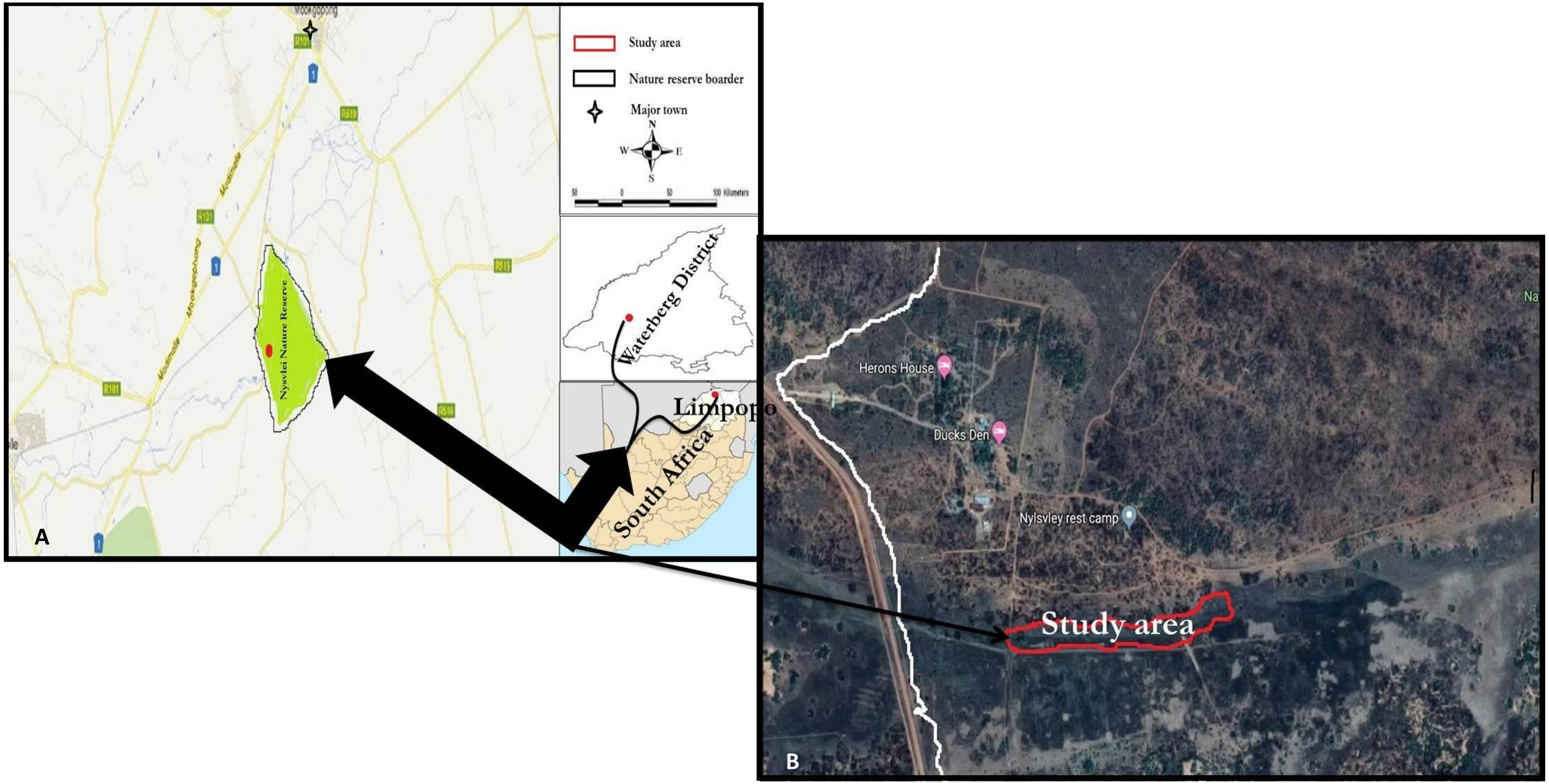

Figure 4.1: The Nylsvley Nature Reserve (The Green Empire (Pty) Ltd)

4.2.2 Description of species under study

4.2.1.1 Populus alba

Populus alba (white poplar) outcompetes numerous indigenous tree and shrub species in generally warmer regions, for example, fields and forest boundaries, and interacts with the natural progression of regular virgin ecosystem. It is a particularly solid contender since it can establish in a range of soil types; produce substantial seed crops, and resprouts effortlessly responding to destruction. Other plant species are prevented from coexisting by lowering the size of nutrients, sunlight, space and water availability (Floate et al., 2016).

Figure 4.2: A population of *Populus alba* in juvenile stage

4.2.1.2 Establishment

Populus alba can be propagated by bare root as it is very good in root suckering. In the soil it is just a network of roots connecting a number of individuals. This plant species normally grows in an aggregated form. It can also be propagated by its cuttings and seeds (Rushforth, 1999).

4.2.1.3 Benefits

Populus alba has high quality wood known as “poplar” with a greenish colour. It produces a lighter and more porous material. It is suitable for several applications because of its close grain and flexibility, like those of willow. It was used by the Greeks to make shields. Its durability is like that of an oak, but with a reduced substantial weight. Paper, snowboards, as well as the bodies of drums and electric guitars are manufactured widely form the white poplar (Verlinden et al., 2015).

4.2.3 Data sampling and analysis

Permanent plots of 10 m x 10 m each were constructed using iron rods. The number of invasive alien plant species and indigenous plant species present in the quadrats were recorded before they could be removed. The invasive alien plant species were removed mechanically. No chemicals were used since there are concerns about their adverse environmental implications. The areas were monitored for the reestablishment of invasive alien species; if they reappeared, they were removed. Only the native plant species were left alone if they happened to be seen establishing in the study site.

All data collected were entered and stored in Microsoft Excel 2010, which were also used in the descriptive statistical analysis of the results. They were then analized using Primer V6 and PERMANOVA.

4.3 Results

Species Richness

Species richness was found to be 08 (Sep-2016) before eradication and 08 (Jun-2017), 10 (Sep-2017), 13 (Dec-2017) and 11 (Mar-2018) after eradication **(Table 4.1).** Although Dec-2017 was the most species rich with 13 species, the different was not significant with the least rich (Sep-2016 and June-2017) having 08 species each.

Table 4.1: Floral species richness in quadrants eradicated of *Populus alba*

Response variable	Treatment	Sep-2016	June-2017	Sep-2017	Dec-2017	Mar-2018
Species richness	Digging	08	08	10	13	11

Individuals of *P. alba* and the other species had a similar trend of increasing from 2016-Sep to 2017-Jun, followed by a decrease and increasing again (2017-Jun to 2017-Dec). The increase of other species (2017-Sep to 2017-Dec) was more significant than the slight increase of *P. alba* species which decreased thereafter (Table 4.1, Figure 4.3).

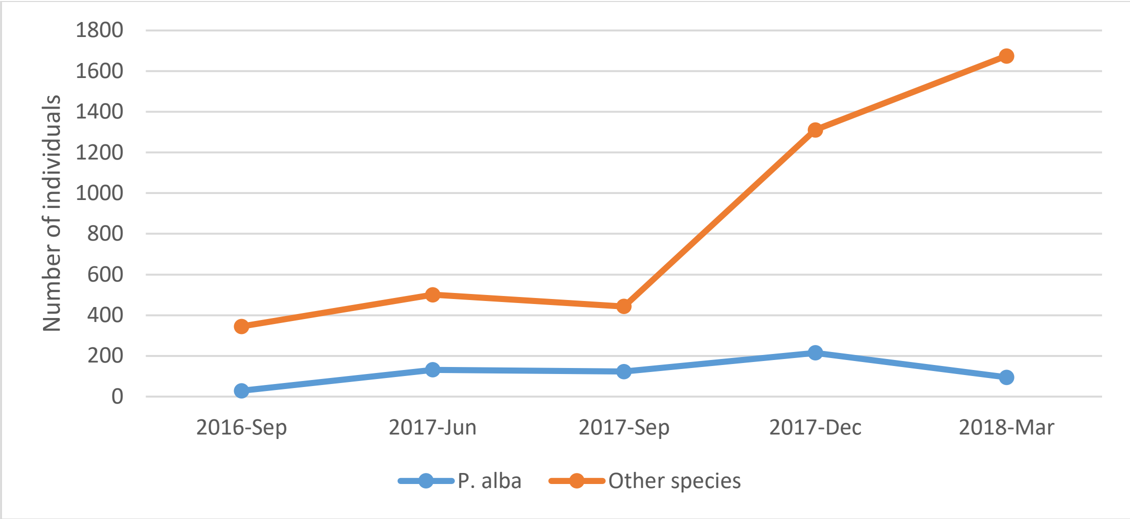

Figure 4.3: Line graph showing the number of *Populus alba* and other plant species pre (Sep-2016) and post eradication.

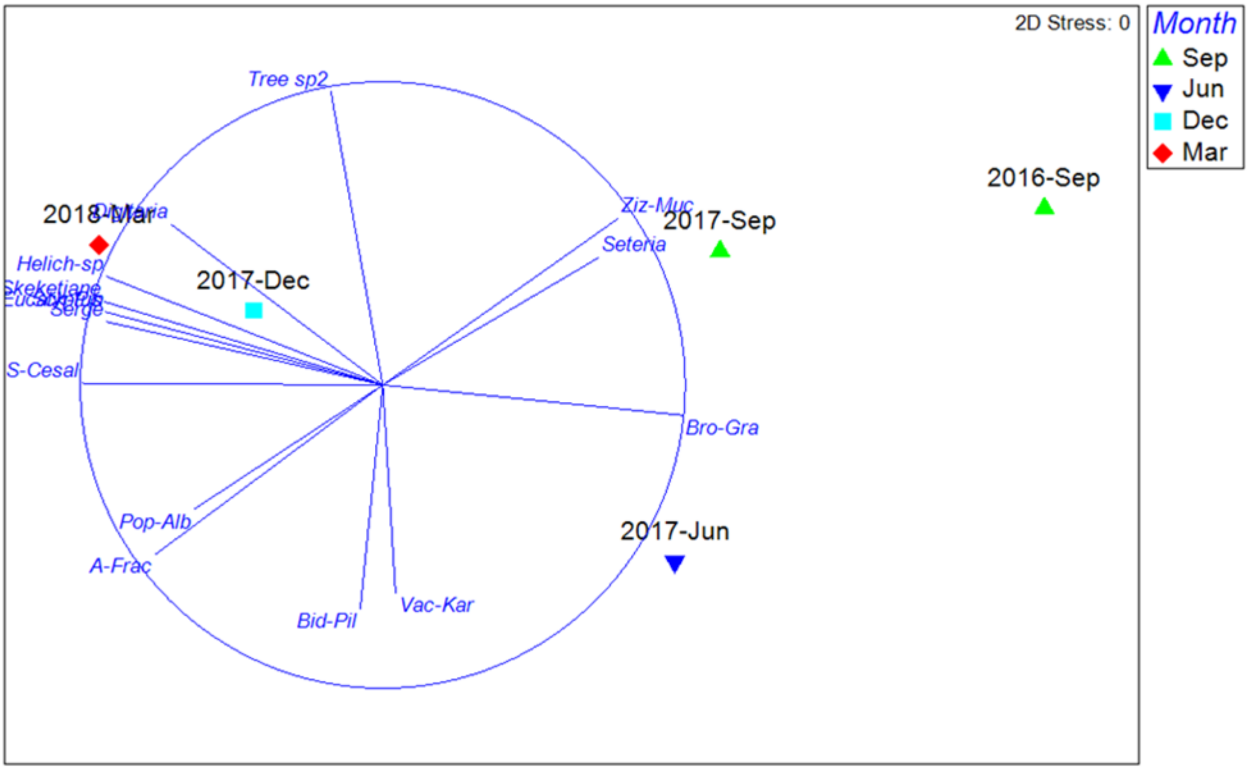

Figure 4.4: Multidimensional scaling ordination plots of species composition in different sampling periods between Sep-2016 and Dec-2017 following the eradication of *Populus alba*

The MDS plot of species composition indicate the abundance of species across the different sampling periods (Figure 4.4). More species were observed in 2017-Dec followed by 2018-Mar, which is similar to the data presented in table 4.2. Similarity across the five sampling periods is attributed to the five species (*P. alba, Digitaria* spp., *V. karoo, Brown grass, Serge, B. pilosa, S. cephaselata*) which are the main species that occurred in either all or four of the sampling periods (Figure 4.4).

Frequency of *P. alba* was observed to be high post its eradication (33.333 in 2016-Sep) and 100 thereafter, while that of *Digitaria* spp. fluctuated. Brown grass remain constant (50) from 2016-Sep to 2017-Sep before decreasing (16.666) in 2017-Dec and completely disappearing in 2018-Mar (Table 4.1). Of the eight species that occurred pre eradication (2016-Sep), four (*P. alba, Digitaria, V. karoo* and Serge) increased in frequency in 2017-June with some fluctuating i.e. *Digitaria* and *V. karoo,* others increasing i.e. Serge and others remaining constant i.e. *P. alba*. Species such as *S. cephaselata* and *Helichrysum* were observed to be among species that occurred in high frequency post eradication. These species were not observed during pre-eradication sampling periods (Table 4.1).

Table 4.2: Frequency of native plant species following eradication of *Populus alba*

Species Name	2016-Sep	2017-June	2017-Sep	2017-Dec	2018-Mar
Populus alba L.	33.333	100	100	100	100
Digitaria spp.	50	83.333	100	33.333	66.666
Brown grass	50	50	50	16.666	0

Vachellia Karro (Hayne) *Banfi & Galasso.*	50	83.333	83.333	66.666	83.333
Ziziphus Mucronata Willd.	16.666	0	16.666	0	0
Surge grass	16.666	33.333	33.333	66.666	83.333
Setaria sphacelata (Schumach.) Stapf & C.E.Hubb. ex M.B.Moss.	0	50	16.666	50	100
Bidens pilosa L.	0	50	16.666	16.666	0
Helichrysum sanguineum Mill.	0	0	16.666	100	100
Skeketjane (herb spp.)	0	0	0	16.666	33.333
Schkurhia pinnata (Lam.) Kuntze ex Thell.	0	0	0	33.333	33.333
Eucalyptus	0	0	0	16.666	16.666
Tree sp 2	16.666	0	16.666	16.666	16.666

4.4 Discussion

Vegetative ground spread of native species increased after the control of invasive alien plant species for the most part because of an increase in grass spread (Morris, 2008). The large amounts of overall observed native plant recolonization demonstrate that the system shows an exceptionally high level of versatility to disturbance by invasive alien species. Versatility is the capacity of an ecosystem to recover to its previous state following a disturbance (Wali, 1999). Herbaceous development forms increased considering clearing of IAPs in quadrats that were already infested densely. This is congruent with the thought that recuperation from plant infestation is observed rapidly in changes to herbaceous diversity and abundance, followed by bushes and trees (Mentis and Ellery, 1994).

There was no significant difference between the numbers of *P. alba* recorded before and after its eradication in 2016. The plant species still came back in large numbers than before uprooting them. This was probably because there were numerous pieces of roots which were left behind during the uprooting of the plant species, resulting in a lot of suckers emerging a few months later and growing vigorously in the study site.

Fragmentation and layering occur through clonal growth and spread, and so does root sprouting. The procedure by which new ramets can develop describes fragmentation from branches or pieces of roots that become slightly submerged in the soil or silt (Floate et al., 2016). Researchers in South Africa concluded that *P. alba* and *Populus tremula* were dispersed across the migration of vegetative parts in water (Henderson, 2007).

The root sprouts of white poplar have been described as productive, "vigorous", and "objectionable" (Alarcon and Sassenrath, 2016). Large areas can be covered by dense colonies or thickets from root sprouts. Root suckers were usually announced far away from the parent tree when white poplar seedlings were germinated and investigated as windbreak trees in the northern Great Plains (Symstad and Leis, 2017). A review by Mabizela et al. (2017) suggested that vegetative seedlings can exist up to 160 feet (50 m) from the parent plant.

Figure 4.5: Site infested with *P. alba,* (a) shows a stand of *P. alba* population before they were removed, (b) shows uprooting of *P. alba* in process, (c) shows a stand with uprooted trees, and (d) shows seedlings of *P. alba* germinating three months after being eradicated.

Figure 4.6: A root sucker of *Populus alba* which emerged from a piece of root left behind while uprooting the parent plant

Populus alba can regenerate from a very small piece of root left in the soil (Figure 4.6) and produces shoots that grow very fast.

4.5 Conclusions and Recommendations

Working with *Populus alba* is very challenging since it is capable of root suckering. When you leave a little piece of *P. alba* root in the ground while eradicating it, it can produce a sucker and grows vigorously. The sucker reached 2 m in a period of just 3 months. However, restoration is still possible with continued monitoring of the areas controlled for plant invaders. *P. alba*, just like other invader species such as *Acacia decurrens*, grows in aggregated form and pushes other plant species out of their territories, therefore monopolizing the area. With this being said, it is advisable to detect this plant at an early stage and introduce a biological control agent for this specific plant if it shows signs of re-establishment. More studies still need to be done on this species' control method in order to find one suited for it.

References

Alarcon, V.J. and Sassenrath, G., 2016. Hydrologic and nutrient modeling within an agricultural watershed in Southeast Kansas. *Kansas Agricultural Experiment Station Research Reports* 2(3), 18.

Bossard, C.C., Randall, J.M. and Hoshovsky, M.C. eds., 2000. Invasive plants of California's wildlands. University of California Press.

Dirr, M.A., 1998. Manual of Woody Landscape Plants: Their Identification, Ornamental Characteristics, Culture, Propagation and Uses. 5th ed. Champaign, IL: Stipes Publishing. Illinois, United States of America.

Floate, K.D., Godbout, J., Lau, M.K., Isabel, N. and Whitham, T.G., 2016. Plant–herbivore interactions in a trispecific hybrid swarm of *Populus*: Assessing support for hypotheses of hybrid bridges, evolutionary novelty and genetic similarity. *New Phytologist* 209(2), 832–844.

Coetzee, B.J., Van der Meulen, F., Zwanziger, S., Gonsalves, P. and Weisser, P.J., 1977. A phytosociological classification of the Nylsvley Nature Reserve. *Bothalia* 12, 137–160.

Herbert, L., 1968. Know your Broadleaves. Forestry Commission Booklet No. 20. London: Her Majesty's Stationery Office. pp 142.

Mabizela, G.S., Slabbert, M.M. and Bester, C., 2017. The effect of rooting media, plant growth regulators and clone on rooting potential of honeybush (*Cyclopia subternata*) stem cuttings at different planting dates. *South African Journal of Botany* 110, 75–79.

McGeoch, M.A., Butchart, S.H.M., Spear, D., Marais, E., Kleynhans, E.J., Symes, A., Chanson, J. and Hoffmann, M., 2010. Global indicators of biological invasion: Species numbers, biodiversity impact and policy responses. *Diversity and Distributions* 16, 95–108.

Mehrhoff, L.J., Silander Jr, J.A., Leicht, S.A., Mosher, E.S. and Tabak, N.M., 2003. IPANE: invasive plant atlas of New England. University of Connecticut, Storrs, Connecticut., USA.

Mentis, M.T. and Ellery, W.N., 1994. Post-mining rehabilitation of dunes on the north-east coast of South Africa. *South African Journal of Science* 90(2), 69–74.

Morris, T.L., Witkowski, E.T.F. and Coetzee, J.A., 2008. Initial response of riparian plant community structure to clearing of invasive alien plants in Kruger National Park, South Africa. *South African Journal of Botany* 74, 485–494.

Owfi, R.E., 2017. Ecophysiological study of *Paulownia tomentosa. International Journal of Current Research* 9(12), 63582–63591.

Praciak, A., 2013. The CABI encyclopedia of forest trees (CABI, Oxfordshire, UK).

Richardson, D.M. and Pyšek, P., 2006. Plant invasions: Merging the concepts of species invasiveness and community invasibility. *Progress in Physical Geography* 30(3), 409–431.

Rushforth, K.D., 1999. *Collins Wildlife Trust Guide: Trees: A Photographic Guide to the Trees of Britain and Europe*. HarperCollins.

Symstad, A.J. and Leis, S.A., 2017. Woody encroachment in northern Great Plains grasslands: Perceptions, actions, and needs. *Natural Areas Journal* 37(1), 118–128.

Tu, M., 2009. Assessing and managing invasive species within protected areas. In: Ervin, J. (Eds.), Protected Area Quick Guide Series. The Nature Conservancy. Arlington, South Africa.

Verlinden, M.S., Broeckx, L.S. and Ceulemans, R., 2015. First vs. second rotation of a poplar short rotation coppice: Above-ground biomass productivity and shoot dynamics. *Biomass and Bioenergy* 73, 174-185.

Wali, M.K., 1999. Ecological succession and the rehabilitation of disturbed terrestrial ecosystems. *Plant Soil* 213, 195–200.

Weber, E., 2003. Invasive plant species of the world: A reference guide to environmental weeds. CABI Publishing. Cambridge, United Kingdom.

Wells, M.J., Duggan, K. and Henderson, L., 1980. Woody plant invaders of the central Transvaal. In: Neser, S., Cairns, A.L.P., (Eds). Proceedings, 3rd national weeds conference of South Africa; 1979 August; Pretoria, South Africa. Cape Town, South Africa: A. A. Balkema: pp.11–23.

CHAPTER FIVE

THE RATE OF RECOLONIZATION BY NATIVE PLANT SPECIES AFTER THE ERADICATION OF THE INVASIVE ALIEN PLANT SPECIES (*Lantana camara* L.) AT VALDEZIA, LIMPOPO PROVINCE, SOUTH AFRICA

Abstract

Invasive alien flora can overwhelm riverine network and modify the hydrology. Little is known about riparian invasive alien trees impacts on general ecology although invasive alien trees are by far the biggest threat to universally red-listed dragonflies (Odonata) in South Africa. Twenty-four permanent plots of 10 m^2 each were constructed. The number of alien invasive species and native species present in the quadrats were recorded before they were removed. The invasive alien plant species were removed mechanically. No chemicals were used since there are concerns about their adverse environmental implications. There was no significant difference in the species diversity and richness before and after clearing of *Lantana camara* in the study site. *L. camara* still came back in large numbers even after being uprooted. The emergence of *L. camara* after the clearing is thought to be due to its ability to reproduce through the stem cuttings which might be lying on the ground, thereby reproducing via stem layering and from the seeds in the ground as some of the seedlings were seen being produced from the seeds.

5.1 Introduction

The introduction of an invasive alien species is often responsible for an increase in predation and competition, habitat reduction, an assortment of illnesses in animals and hereditary change in population (the indigenous species influenced by these invasive alien species may change their hereditary makeup). Once these species are well established, it is often very difficult to remove them. At the point when the removal is possible, it comes at a staggering expense financially and ecologically. In any case, ecosystems are forever in the process of trying to control invasive alien plants (PA Department of Conservation and Natural Resources, Invasive species Management Plan 2005). Invasive species cause major ecological and related financial misfortunes which can add up to billions of dollars each year.

Invasive alien species have been distinguished as the second main driver of biological diversity depletion following habitat annihilation (Alonso et al., 2001) and the primary driver of species depletion in Island landscapes (Clout and Veitch, 2002). Invasive alien trees can overwhelm riverine ecosystem and modify the hydrology (Castro-Diez and Fernandez, 2017). Little is known about riparian invasive alien trees' impact on general ecology (Richardson and Van Wilgen, 2004) although invasive alien trees are by far the biggest threat to universally red-listed dragonflies (Odonata) in South Africa (Samways and Taylor, 2004). South Africa is imperative for biological diversity since two of the world's hotspots are within it (Myers et al., 2000).

It has been demonstrated that invasive species (including plants, animals and microbes) can impact the distribution, abundance and recovery of indigenous species (Aravind et al., 2010). Exotic plant species can also divert pollinators and dispersers of indigenous plant species towards themselves, therefore obstructing the reproductive achievement of indigenous species. Non-indigenous species are imperative operators of worldwide ecological change (Aravind et al., 2010) and are perceived as the second most noteworthy risk on biodiversity after anthropogenic habitat loss and ecosystem destruction (Gooden et al., 2009). The aim of this study was to investigate the rate at which the native plant species would recolonize the area which was previously infested with invasive alien plant species (*Lantana camara*) and to assess the factors to consider for successful recolonization or restoration of the invaded site.

5.1.1 Occurrence of the species under study

5.1.1.1 Lantana camara

Lantana camara is found in various climatic and geographic ranges. It is found in a range of ecosystems including riparian zones, open grazing lands, forest regions, national parks, rainforest edges, and along roads, creeks and fence lines, and are additionally cultivated as ornamentals in parks and gardens. Pervasions go from single shrubberies to thick monocultures, displacing local vegetation or pasture species (Day et al., 2003).

5.2 Methodology

5.2.1 Description of the study area

The study was conducted in the Makhado District Municipality, in a farm at a village known as Valdezia (Figure 5.1). The study site is along the road (R524) towards Louis Trichardt and was highly infested with *L. camara*. It lies at 23.8’17’’S and 30.8’17’’E. The soil type in this region is mainly clay red to gravel type. The species composition found here is mainly *Vachellia karroo, Sclerocarya birrea* subsp *caffra,* and *L. camara*.

The area receives summer rainfall and is dominated by Tshivenda and Xitsonga speaking people.

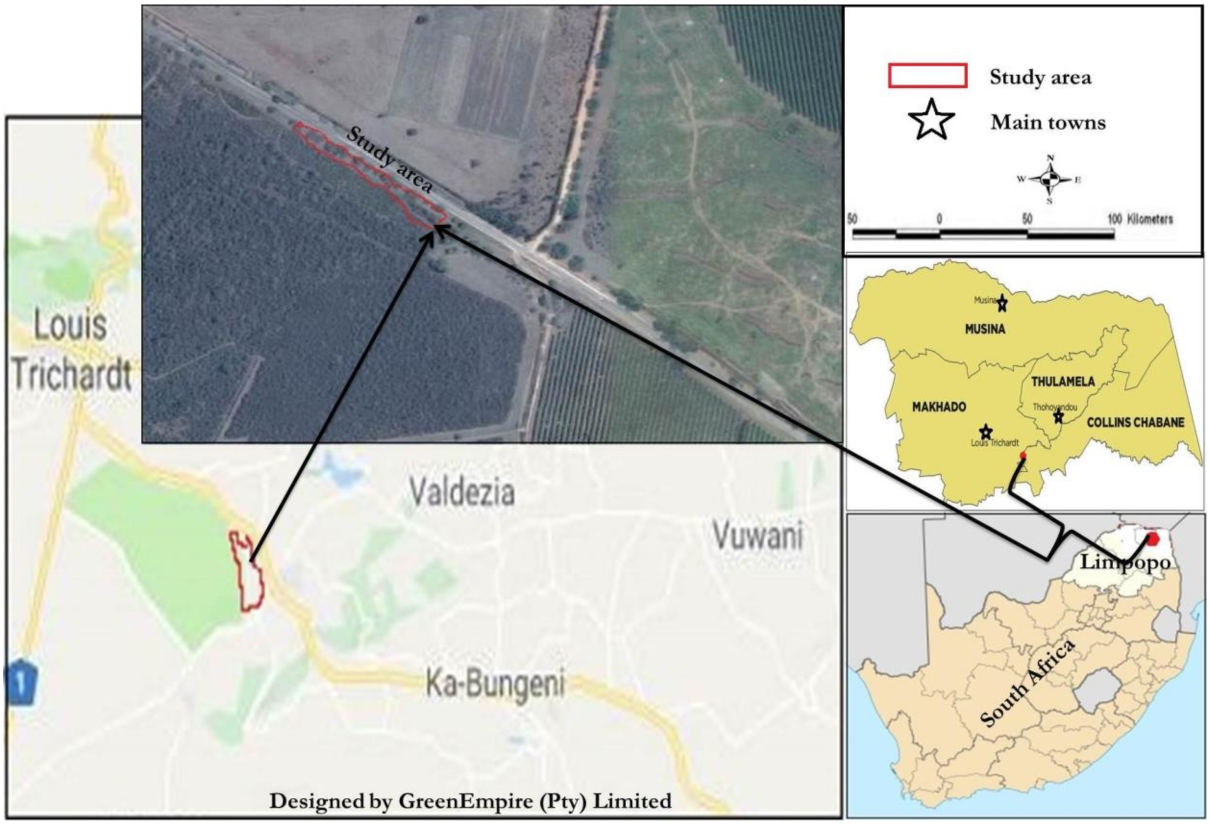

Figure 5.1: The Makhado District Municipality including the study area (The Green Empire (Pty) Ltd).

5.2.2 Description of species under study

5.2.2.1 Lantana camara

Lantana camara L. is an indigenous woody shrub of tropical America (Holm et al. 1997) which is found in 47 nations and has been portrayed as one of the world's ten most exceedingly awful weeds (Cronk and Fuller 1995). *L. camara* is a low upright, vigorous, climbing shrub with stout bent prickles and a black current repulsive smell; it reaches heights of 2.4 meters (or 7 meters in some cases). It has a very strong root system and produces a new flush of shoots even after repeated cutting (Rosacia et al., 2004).

Lantana camara is mostly indigenous to tropical and subtropical America, yet limited taxa are indigenous to Africa and tropical Asia. It currently exists in approximately 50 nations where under a few hundred species are vascular plants (Pyšek et al., 2020). The documented quantities of *Lantana* species vary from 50 to 270 specific and sub-specific units, but it seems that 150 species are a better approximation (Mushi, 2019). The genus is a troublesome one to classify systematically since species are not steady and hybridization is broad, with age, the shape of inflorescence changes, and the colour of the flowers fluctuate with maturity and age (Munir and Adelaide, 1996).

Dutch travellers brought *Lantana camara* from Brazil into the Netherlands in the late 1600s and later pioneers from different nations conveyed seeds to North America, Europe and Great Britain (Mohamed et al., 2016). It soon outspread to the islands of Pacific, Australia and Southern Asia, following its introduction into Hawaii for ornamental purpose. In comparative route from Natal (South Africa), it was quickly dispersed by birds to hotter regions of South Africa (Mohamed et al., 2016). In the eighteenth and nineteenth century, nurserymen marketed, and advanced numerous vivid forms and it is currently cultivated worldwide as an ornamental plant (Mohamed et al., 2016). Of the 650 vascular plant names in the genus, the numerous parts are related with the *L. camara* complex (Mishra, 2015). The plant is an aggressive and constrain out-breeder invader that has infested huge spans of fields, plantations and forest areas in numerous tropical and subtropical territories. It has been estimated that 4 million ha in Australia and 160 000 ha in Hawaii are infested with *L. camara* (Ross, 1999). It has been viewed as one of the 10 most noxious weeds on the planet and much has been written on its encroaching habit, techniques for eradication and control (Parson and Cuthberson, 1992; Munir and Adelaide, 1996). Besides its notoriety as an orchard plant, *L. camara* is said to form an important windbreak and gives a decent readiness to crops, making the progress with fine leaf with a protective cover. The plant is not voluntarily ingested by cattle except if fodder is lacking. In tropical nations, the mature blue-dark berries are ingested, but ingestion of the green berry has ended up to human fatalities (Morton, 1994; Ross, 1999).

The ecological limitations to distribution of *Lantana* are obscure. Inside the limits of its present occurrence, it is proceeding to invade new regions and increases in number. The way that *Lantana* is absent in a few regions does not mean it is unsuited to them; it has been portrayed as a 'sleeper weed', anticipating conditions to spread. *Lantana camara* was put at category one on the CARA legislation in the country. Records demonstrate that *Lantana* was brought into South Africa in 1858 in Cape Town, Western Cape Province (WC), where, under Mediterranean climatic conditions it spread very little, and in 1883 at Durban, KwaZulu-Natal Province (KZN) (Stirton, 1977), where it prospered under subtropical situations. It was pronounced a poisonous weed in 1946 in KZN, which incorporated South African invasion of 80%, every decade doubling in area (Marr, 1964; Kwembeya et al., 2013). Measured by ongoing requests from people in general for exhortation on the control of *Lantana*, the weed is currently spreading and expanding in population size mainly in the province of Mpumalanga (MP) and Limpopo (LP), and in addition in the North West (NW), Eastern Cape (EC), Gauteng (GP) and the southern part of Western Cape (WC), that is, essentially in the warmer and external parts of its present occurrence. *Lantana* was evaluated by experienced researchers to have invaded more than 2.2 million ha in South Africa, which, if condensed, would totally cover a zone in excess of 69 000 ha (Versfeld et al., 1998). An ongoing, statistically valid, national, invasive alien plant study found that *Lantana* covers 560 000 ha of the landscape, including riparian zones (Kotze et al., 2010).

Flowers are small, generally orange, occasionally changing from white to red in different shades and having a yellow throat in axillary heads. Fruits are small, greenish-blue black, blackish, drupaceous, and sparkling with two nullets, nearly all year round, dispersed by birds. Seeds sprout easily. *Lantana camara* berries draw in frugivorous birds and mammals that assist to scatter its seeds widely. The species forms dense, impenetrable thickets that smother indigenous vegetation and pasture (Zachariades et al., 2017). The outspread of *L. camara* in forests is an exceptional reason for worry as it might change forest structure and composition affecting indigenous species assemblages (Aravind et al., 2006). This species can out-compete indigenous plants since it can develop on nutritiously poor soils (Platt et al., 2016). It flowers all year-round and produces a lot of nectar which attracts potential pollinators (Aravind et al., 2010). It changes forest structure by replacing the native understorey species, and this, in turn, can influence the distribution and behaviour patterns of animal populations.

South African specimens of the weed *Lantana* that Sanders has investigated are generally crossbreeds between *L. nivea* subsp. *mutabilis* and *L. camara* var. *aculeata* (Zachariades et al., 2017). Others seem to have been contributed from both above parent varietal plus *L. scabrida, L. horrida* Kunth subsp. *Tilifolia* (cham.) R.W. Sanders from South Brazil, *L.* (X) *strigocamara, L. hirsute*, conceivably *L. depressa* var. *depressa* from Florida and the Callowiana Hybrid group (*Lantana* (X) *strigocamara* X *L. depressa var. depressa*), and there are also some unblended *L. nivea* subsp. *mutabilis* exhibit (Zachariades et al., 2017).

Noxious triterpines and lantadenes A and B that lead to fatalities to, cattle, horses, goats, sheep, and rabbits by the miss-performance of the liver and different organs are found in the leaves of Lantana (Munyua et al., 1990; Morton, 1994). Nevertheless, many animals would cautiously abstain from ingesting this plant when they have a choice. The green fruits also contain the toxic substances and have led to sickness and fatalities in kids (Morton, 1994). *Lantana camara* leaves and their leachates exert allelopathic impacts *in vitro* and to a lesser degree in soil on the germination of the seeds, root elongation, and plant development of numerous species (Casado, 1995).

5.2.2.2 Habitat Description

The various and wide geographic dissemination of *L. camara* reflects its broad environmental resistance. It is distributed in varying habitats and soil types. It normally establishes best in open sunny wastelands, for example, rainforest boundaries, coastal areas, and in forests recuperating from fire (Zachariades et al., 2017). *Lantana camara* does not invade solid rainforests, but rather, is located on their edges (Zachariades et al., 2017). Where wet sclerophyll forests and rainforests have been disturbed by logging, spaces are constructed; this enables *Lantana* to invade on the forests.

Lantana camara develops under extensive variety of climatic constraints. In Australia, the inland limits of its geographical range coincide with the 750 mm isohyet in southern Queensland and the 1250 mm isohyet in the north, with invasion being limited to creek lines in drier zones. It does not seem to have a higher temperature or precipitation limitation and is frequently found in tropical regions getting 3000 mm of precipitation annually, if soils are adequately well drained. *L. camara* rarely occurs where temperatures fall beneath 5˚C most of the times, and in South Africa, it occurs in zones with a mean yearly surface temperature greater than 12.5˚C (Zachariades et al., 2017). Extended chilling temperatures damage aerial woody parts and cause defoliation. *Lantana camara* dominates low laying vegetation in disturbed zones, regularly producing thick mono-specific bushes, and is compromised in widespread depletion of indigenous plant species (Gooden et al., 2009).

5.2.2.3 Reproduction

If adequate moisture and light are available, *L. camara* flowers all year round in most places, in wet summer months, the flowering is at its peak. Flowering takes place only in the wetter or warmer months in cooler or drier regions due to drought or frost damage (Mello et al., 2005). As early as the second growing season, plants can start to flower. The variations in the length of corolla, number of flowers per inflorescence and its diameter lead to some butterfly species visiting certain taxa of *Lantana* regularly than others. According to this view, different varieties of *Lantana* would have different pollinator species. Therefore, there may be little cross-pollination between species or varieties of *Lantana* both in naturalized and native ranges of the section Camara (Zachariades et al., 2017).

5.2.2.4 Benefits

In numerous agricultural diversities, *L. camara* is sown worldwide as an ornament. In temperate zones, it is planted as a yearly bedding species. *Lantana camara* oil, an aromatic blend which differs from native plant heterogeneity, is conveyed from Brazil (Weyerstahl et al., 1999). Steeps of the leaves and other parts of the plant are utilized as healing remedies because of their anti-inflammatory properties (Oyedapo et al., 1999), they also serve as an expectorant and tonic, and are also added to baths as antirheumatics. *Lantana camara* extracts have also been shown to be strong medicinal drugs (Liogier, 1990). Since the leaves and some different parts of *L. camara* are toxic, caution must be observed when they are utilized therapeutically. The mature fruit is tender and eaten in large quantities by birds and regularly eaten by people in some nations (Herzog et al., 1994). In some agricultural settings, invasions of *Lantana* are believed to prevent soil compaction, and are valued as source of organic matter for pasture field remodel or improvement. It is also valuable in steep zones and stream banks for stabilizing soil and preventing erosion. At times, *L. camara* smothers weeds perceived to be more threatening. The sale of ornamental *Lantana* in a few states provides income to the nursery business. Concentrates of *L. camara* leaves have demonstrated solid pesticide and antimicrobic action in various tests. Preserving potatoes with *Lantana* leaves almost eliminates harm by *Phthorimaea operculella* Zeller, the potato tuber moth (Lal, 1987). Stems and leaves are utilized as mulch. However, of lesser standard on account of magnitude and morphology, *Lantana* stems are broadly utilized as fuel wood in less developed nations.

5.2.3 Data sampling and analysis

Twenty-four permanent plots of 10 m x 10 m each were constructed using iron rods. The number of alien invasive species and native species present in the quadrats were recorded before they were removed. The invasive alien plant species were removed mechanically. No chemicals were used, since there are concerns about their adverse environmental implications. The areas were monitored for the reestablishment of alien invasive species; if they occurred, they were removed. Only the native plant species were left alone, if they happened to be seen establishing in the study site.

All data collected were entered and stored in Microsoft Excel 2010, and were also used in the descriptive statistical analysis of the results. They were then analized using Primer V6 and PERMANOVA.

5.3 Results

5.3.1 Species Richness

Species richness was not significantly different between the sampling periods. Observations of the eradicated sampling plots showed that species richness in the dug area of *L. camara* was found to be 08 (Sep-2016) before eradication and 08 (Jun-2017), 10 (Sep-2017), 09 (Dec-2017) and 11 (Mar-2018) after eradication **(Table 5.1).** Only Mar-2018 was richer with 11 species, with the least rich (Sep-2016 and Jun-2017) having 08 species respectively.

Table 5.1: Floral species richness during the sampling periods following the eradication of *Lantana camara*.

Response variable	Treatment	Sep-2016	June-2017	Sep-2017	Dec-2017	Mar-2018
Species richness	Digging	08	08	10	09	11

Individuals of *L. camara* slightly increased after eradication (2016-Sep to 2017-Dec), decreased thereafter (2017-Dec to 2018-Mar). On the other hand, the number of the rest of the species slightly decreased after eradication (2016-Sep to 2017-Jun), and the highest number of *L. camara* species were observed when the rest of the species were at their lowest (2017-Dec) (Table 5.1, Figure 5.2)

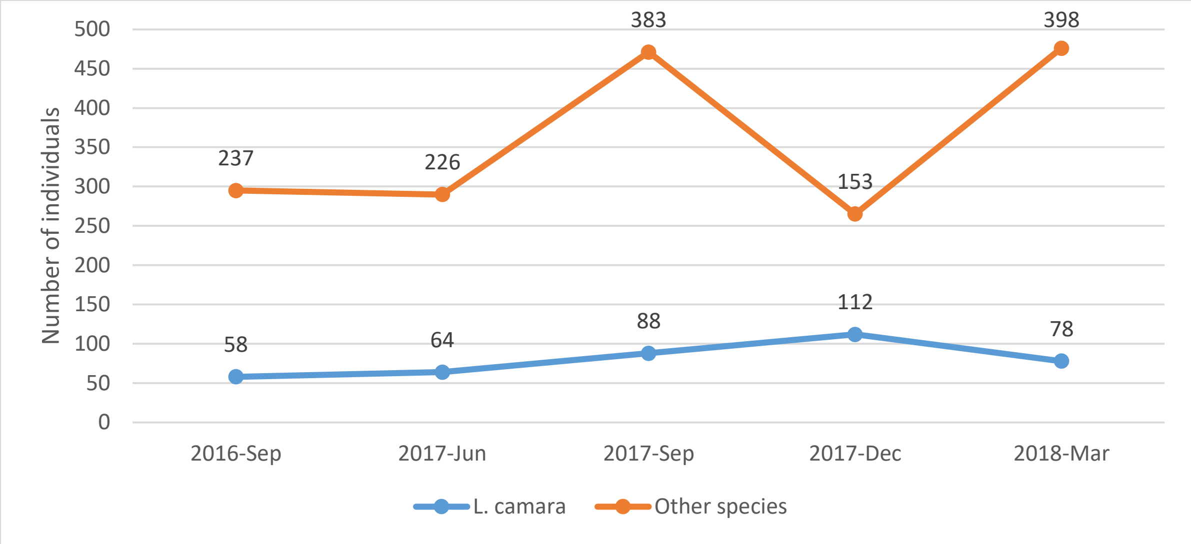

Figure 5.2: Line graph showing the number of *Lantana camara* and other plant species pre (Sep-2016) and post eradication.

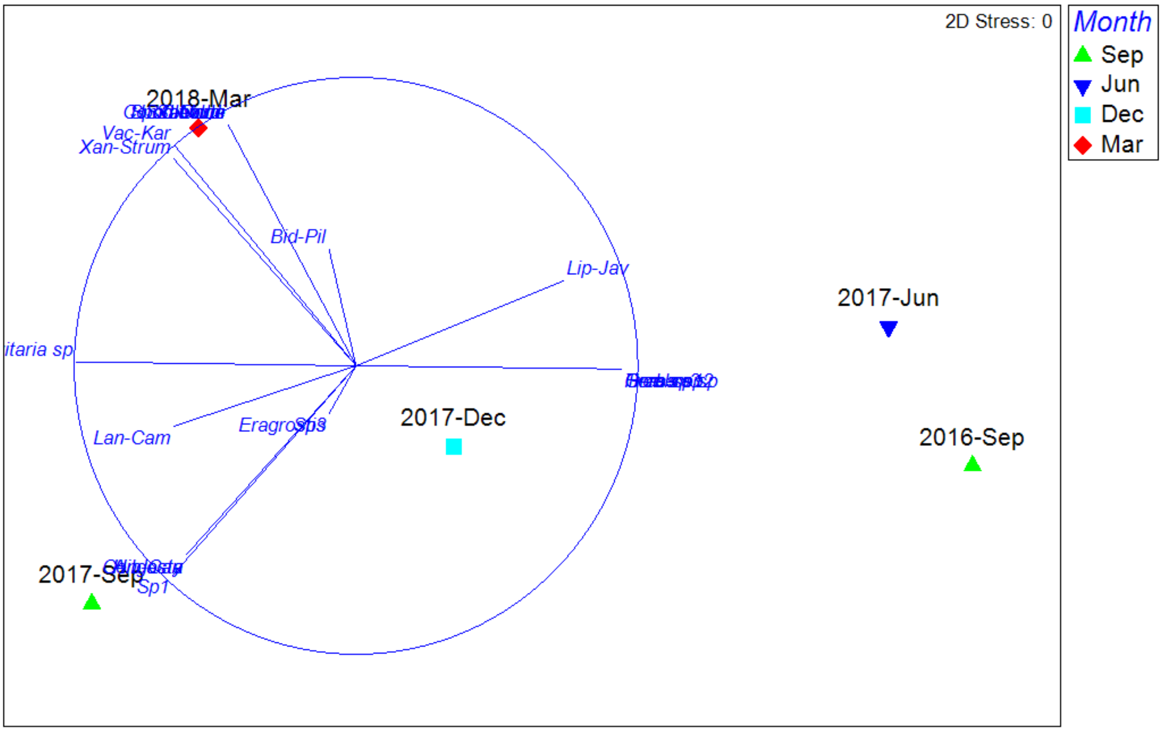

Figure 5.3: Multidimensional scaling ordination plots of species composition in different sampling periods between Sep-2016 and Dec-2017 following the eradication of *Lantana camara.*

The MDS plot of species composition clearly indicated that species were shared almost similarly between the sampling periods (Figure 5.2). More species are observed in 2018-Mar, which is similar to what is presented on species richness table (Table 5.1). The similarity in all the sampling periods was attributed to species such as *L. camara, V, karoo, B. pilosa* and *L.javanica*, which are the only species that were observed throught the sampling periods.

Different plant species were observed during the five sampling periods, however, 2016-Sep and 2017-Jun had completely similar plant species differing only in the frequencies (Table 5.2), with other species observed from 2017-Sep to 2018-Mar. Four species (*L. camara, V. karoo, B. pilosa* and *L. javanica*) were the only species observed throughout the sampling periods. These four species are attributed to have contributed the most to the similarity between 2016-Sep and 2017-Jun and have contributed also to the similarity across the sampling periods. Frequency was observed to be high during the 2018-Mar sampling period.

Table 5.2: Frequency of native plant species following eradication of *Lantana camara*

Species Name	**2016-Sep**	**2017-June**	**2017-Sep**	**2017-Dec**	**2018-Dec**
Lantana camara L.	83.333	100	100	100	83.333
Vachellia Karoo (Hayne) Banfi & Galasso.	66.666	100	66.666	66.666	66.666
Bidens Pilosa L.	83.333	66.666	100	83.333	100

Lippia Javanica L.	83.333	83.333	33.333	66.666	33.333
Herb spp1	33.333	33.333	0	0	0
Combretum spp.	16.666	16.666	0	0	0
Grass spp2	16.666	16.666	0	0	0
Tree spp3	16.666	16.666	0	0	0
Hibiscus cannabis L.	0	0	83.333	0	0
Spp1	0	0	83.333	50	0
Digitaria spp.	0	0	83.333	66.666	33.333
Cynodon dactylon (L.) Pers	0	0	33.333	0	0
Congesta	0	0	16.666	0	0
Xanthium Strumarium L.	0	0	16.666	16.666	66.666
Eragrostis spp.	0	0	0	100	0
Spp3	0	0	0	100	0
Sida cordifolia L.	0	0	0	0	16.666
Combretum molle R.Br. ex G.Don.	0	0	0	0	16.666
Cleome	0	0	0	0	16.666
Sporobolus spp.	0	0	0	0	16.666

5.4 Discussion

Preceding clearing by ‘Working for Water’ in 2016, transects had invasive alien species density of up to 97%. These high IAP densities seemed to negatively affect community diversity which is congruent to various other researcher's investigations (Morris et al., 2008; Holmes et al., 2000; Pyšek and Pyšek, 1995). After the lowering of invasive alien species density, there was an eminent corresponding increment in the native diversity of vegetation in the already densely-infested regions. The significant contribution to the high invasive alien species densities was from annuals or short-lived perennial species, for example, *X. strumarium* (L.).

Despite being uprooted several times during the observation and monitoring of the area since 2016, *L. camara* kept on emerging in the same area. This is probably because of the soil seed bank which had not yet been exhausted by then. In other words, *L. camara,* because of its presumably large seed bank continues to regenerate for several seasons until the seed bank is depleted. It is believed that with further monitoring, the soil seed bank might be exhausted, and the *Lantana* population might start disappearing for good in the area. In regions where it is prevailing, *L. camara* is likely to reduce indigenous species composition by hindering recolonization after some environmental pressures. The influence of *L. camara* invasion on indigenous plant biodiversity was prevalent in such a way that indigenous herbs, shrubs and trees were compromised seriously in such vegetation. Attempting to eradicate *L. camara,* led to its stimulation as several individuals re-established after trying to eliminate them.

There was an increase in the newly established individuals compared to what existed before disturbance. This may have been initiated by the increase in propagule supply during follow-up observation events by invader species control officers, as well as soil disturbance during manual eradication. Mason and French (2007) reported on a large increase in the number of *L. camara* following disturbances such as soil disturbance and others. Species richness in each growth form declined significantly with increasing *Lantana* cover, indicating that the threat of Lantana is pervasive across all life forms in the recipient community. Similar pervasive losses of plant species across multiple growth forms have been detected elsewhere for *Lantana* (Gooden et al., 2009), possibly representing a general effect of the invader on native communities.

Some researchers announced an increase in both abundance and richness of indigenous species recovery after *Lantana* eradication in north Queensland (Gooden et al., 2009), and this was not the situation with this investigation. Just other invasive herb species occupied the site in large numbers than indigenous plant species. Trees, relative to alternate life forms, largely drove the abundance in indigenous species recovery. This compares with others' discoveries of contrasts among plant species considering *Lantana* eradication, however they revealed a noteworthy increase in herbs and shrubs with respect to woody trees (Gooden et al., 2009).

There is exact proof that Lantana can allelopathically suppress indigenous species regeneration (Gentle and Duggin, 1998); however, there is no genuine quantifiable proof for this from field evaluations (Gooden et al., 2009). Expansion in species richness in controlled areas proposes that institute *L. camara* is an obstruction to recuperation of species variety after disturbance of the vegetation. Then again, expansion in species richness may correlate to control rather than the original *L. camara* invasion, for example, soil disturbance and amplification of nutrient availability (Davis et al., 2000).

Figure 5.2: (a) A dense population of *Lantana camara* before removal, (b) the cutting and digging of *L. camara* in progress, (c) cleared site just three months after clearance, and (d) *L. camara* recolonizing the site after a few months

The emergence of *L. camara* after clearing is thought to be due to its ability to reproduce through the stems which might be lying on the ground. The species can produce roots on the stem if it comes into contact with the ground, thereby reproducing via stem layering, and also from the seeds.

5.5 Conclusions and Recommandations

Lantana camara is a difficult species to deal with because it spreads in three ways. Firstly, through vegetative reproduction where a stem sends roots into the soil (layering) which allows for dense stands to form and spread to short distances very quickly. Secondly, birds and other animals such as foxes consume the fruits, and the droppings they pass would contain seeds, in this way they get dispersed over a large distance. Digestion of seeds generally increases the rate of their germination. Lastly, it is also capable of re-seeding and re-sprouting. The site previously infested with *L. camara* needs more monitoring after eradication and possibly active restoration in the form of planting the native plant species which previously occupied the area before the infestation by the invasive species. It is therefore more important to study the effect the seed bank might have on the recovery of this problematic species and find out better ways of control.

References

Alonso, A., Dal Lmeier, F., Granek, E. and Raven, P., 2001. Biodiversity connecting with the tapestry of life, Smithsonian Institution / Monitoring and assessment of biodiversity Program and President's committee of Advisors on Science and Technology. Washington DC, United States of America.

Aravind, N.A., Dinesh, R.A.O., Ganeshaiah, K.N., Uma Shaanker, R. and Poulsen, J.G., 2010. Impact of the invasive plant, *Lantana camara*, on bird assemblages at Male Mahadeshwara Reserve Forest, South India. *Tropical Ecology* 51(2S), 325–338.

Aravind, N.A., Rao, D., Vanaraj, G., Ganeshaiah, K.N., Shaanker, R.U. and Poulsen, J.G., 2006. Impact of *Lantana camara* on plant communities at Malé Mahadeshwara reserve forest, South India. Invasive Alien Species and Biodiversity in India. Banaras Hindu University, Varanasi, pp.1–12.

Casado, C.M., 1995. Allelopathic effects of *Lantana camara* (Verbenaceae) on morning glory (*Imomoea tricolor*). *Rhodora* 97, 264–274.

Castro-Díez, M.P. and Fernández, Á., 2017. Effects of non-native riparian plants in riparian and fluvial ecosystems: a review for the Iberian Peninsula. *Limnetica* 36, 525–541.

Clout, M.N. and Veitch, C.R., 2002. Turning the tide of biological invasion: the potential for eradicating invasive species. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group, Gland, Switzerland and Cambridge, United Kingdom, pp.1–3.

Cronk, Q.C.B. and Fuller, J.L., 1995. Plant Invaders. Chapman and Hall. London, United Kingdom.

Parsons, W.T., Parsons, W.T. and Cuthbertson, E.G., 2001. *Noxious weeds of Australia*. CSIRO publishing.

Davis, M.A., Grime, J.P. and Thompson, K., 2000. Fluctuating resources in plant communities: A general theory of invasibility. *Journal of Ecology* 88, 528–534.

Day, P.K., LeDuc, H.G., Mazin, B.A., Vayonakis, A. and Zmuidzinas, J., 2003. A broadband superconducting detector suitable for use in large arrays. *Nature 425*(6960), 817–821.

Gentle, C.B. and Duggin, J.A., 1998. Interference of shape *Choricarpia leptopetala* by shape *Lantana camara* with nutrient enrichment in mesic forests on the Central Coast of NSW. *Plant Ecology* 136(2), 205–211.

Gooden, B., French, K. and Turner, P.J., 2009. Invasion and management of woody plant, *Lantana camara* L., alters vegetation diversity within wet sclerophyll forest in south eastern Australia. *Forest Ecology and Management* 257, 960–967.

Herzog, F., Farah, Z. and Amado, R., 1994. Composition and consumption of gathered wild fruits in the V-Baoule, Cote d'Ivoire. *Ecology of Food and Nutrition* 32(3–4), 181–196.

Holm, L., Doll, J., Holm, E., Pancho, J. and Herberger, J., 1997. *Word weeds:* Natural Histories and Distribution. John Wiley & Sons, Inc. New York, United States of America.

Holmes, P.M., Richardson, D.M., Van Wilgen, B.W. and Gelderblom, C., 2000. Recovery of South African fynbos vegetation following alien woody plant clearing and fire: Implications for restoration. *Australian Ecology* 25, 631–939.

Kotze, I., Beukes, H., Van Den Berg, E. and Newby, T., 2010. National invasive alien plant survey. Report No. GW/A/2010/21 to the Department of Water and Environmental Affairs, Working for Water Programme. Agricultural Research Council-Institute for Soil, Climate and Water. Pretoria, South Africa.

Kwembeya, A., Rugare, J.T. and Mabasa, S., 2013. Allelopathic effects of lantana (lantana camara) on blackjack (bidens pilosa) and pearl millet (pennisetum glaucum). *Asian Journal of Agriculture and Rural Development* 3(8), pp.543–553.

Lal, L., 1987. Studies on natural repellents against potato tuber moth (*Phthorimaea operculella* Zeller) in country stores. *Potato Research* 30(2), 329–334.

Liogier, H.A., 1990. Plantas medicinales de Puerto Rico y del Caribe. Ibero americana de Ediciones, Inc., San Juan, PR 563p.

Marr, R.S., 1964. Get rid of *lantana*. *Farming in South Africa, February 1964*, pp.42–45.

Mason, T.J. and French, K., 2007. Management regimes for a plant invader differentially impact resident communities. Institute for Conservation Biology, University of Wollonong, NSW 2522, Australia.

Mello, F.B., Jacobus, D., Carvalho, K. and Mello, J.R., 2005. Effects of *Lantana camara* (Verbenaceae) on general reproductive performance and teratology in rats. *Toxicon* 45(4), 459–466.

Mishra, A., 2015. Allelopathic properties of Lantana camara. *International Research Journal of Basic and Clinical Studies*, 3(1), 13–28.

Mohamed, N.M., Makboul, M.A., Farag, S.F., Jain, S., Jacob, M.R., Tekwani, B.L. and Ross, S.A., 2016. Triterpenes from the roots of *Lantana montevidensis* with antiprotozoal activity. *Phytochemistry Letters* 15, 30–36.

Morris, T.L., Witkowski, E.T.F. and Coetzee, J.A., 2008. Initial response of riparian plant community structure to clearing of invasive alien plants in Kruger National Park, South Africa. *South African Journal of Botany* 74, 485–494.

Morton, J.F., 1994. *Lantana*, or red sage (*Lantana camara* L. Verbenaceae), notorious weed and popular garden flower; some cases of poisoning in Florida. *Economic Botany* 48(3), 259–270.

Munir, A.A. and Adelaide, J., 1996. A taxonomic review of *Lantana camara* L. and *Lantana montevidensis* (Spreng) Briq. (Verbenaceae) in Australia. Board of the Botanic Gardens and State Herbarium. Adelaide, South Australia. 1976–2015 (Vol. 1, 29).

Munyua, S.J.M., Nienga, M.J., Karitu, T.P., Kimoro, T.P., Kiptoon, J.E. and Buoro, I.B.J., 1990. A note on clinical-pathological findings and serum enzyme activity in sheep, goats and Friesian calves with acute *Lantana camara* poisoning. *Bulletin of Animal Health and Production in Africa* 38(3), 275–279.

Mushi, G.V., 2019. *Local communities' perceptions on lantana camara and management responses in East Usambara, Tanzania* (Doctoral dissertation, Sokoine University of Agriculture).

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A.B. and Kent, J., 2000. Biodiversity hotspots for conservation priorities. *Nature* 403, 853–858.

Oyedapo, O.O., Sab, F.C. and Olagunju, J.A., 1999. Bioactivity of fresh leaves of *Lantana camara*. *Biomedical Letters* 59, 179–183.

Pennsylvania Department of Conservation and Natural Resources, Invasive species Management Plan., 2005.

Platt, R.V., Ogra, M.V., Badola, R. and Hussain, S.A., 2016. Conservation-induced resettlement as a driver of land cover change in India: An object-based trend analysis. *Applied Geography* 69, 75–86.

Pyšek, P. and Pyšek, A., 1995. Invasion by *Heracleum mantegazzianum* in different habitats in the Czech Republic. *Journal of Vegetation Science* 6, 711–718.

Pyšek, P., Hulme, P.E., Simberloff, D., Bacher, S., Blackburn, T.M., Carlton, J.T., Dawson, W., Essl, F., Foxcroft, L.C., Genovesi, P. and Jeschke, J.M., 2020. Scientists' warning on invasive alien species. *Biological Reviews* 95(6), 1511–1534.

Richardson, D.M. and Van Wilgen, B.W., 2004. Invasive alien plants in South Africa: How well do we understand the ecological impacts? *South African Journal of Science* 100, 45–52.

Rosacia, W.Z., Achivar, A.N. and Avanzado, M.B., 2004. *Lantana* and Hagonoy: Poisonous weeds prominent in rangeland and grassland areas. *Research Information Series on Ecosystems (Department of the Environment and Natural Resources, Republic of the Philippines),* 16(2).

Ross, I.A., 1999. Medicinal plants of the world. Chemical constituents, traditional and modern medical uses. Human Press. New Jersey, United States of America.

Samways, M.J. and Taylor, S.T., 2004. Impacts of invasive plants on red-listed South African dragonflies (Odonata) *South African Journal of Science* 100, 78–80.

Stirton, C.H., 1977. Some thoughts on the polyploid complex *Lantana camara* L. (Verbenaceae). In *Proceedings of the Second National Weeds Conference of South Africa.* pp. 321–340.

Versfeld, D.B., Le Maitre, D.C. and Chapman, R.A., 1998. Alien invading plants and Water Resources in South Africa: A Preliminary Assessment. Water Research Commission Report No. TT99/98.

Weyerstahl, P., Marschall, H., Eckhardt, A. and Christiansen, C., 1999. Constituents of commercial Brazilian *Lantana* oil. *Flavour and Fragrance Journal* 14(1), 15–28.

Zachariades, C., Paterson, I.D., Strathie, L.W., Hill, M.P. and Van Wilgen, B.W., 2017. Assessing the status of biological control as a management tool for suppression of invasive alien plants in South Africa. *Bothalia – African Biodiversity and Conservation* 47(2), 1–19.

CHAPTER SIX

AN ADAPTIVE MANAGEMENT PLAN

Abstract

Given that there is not even a single method of invasive alien plant species (such as *Acacia decurrens*, *Lantana camara* and *Populus alba*) control that is entirely effective, there need to be a strategy on how to deal with invasive species on an individual basis. Even though not 100% effective, the chemical control method seems to work, but with some associated negative effects on the native plant species. There are also concerns with mechanical control of invasive species, since it constitutes a disturbance of the ecosystem, and therefore may have negative impacts on the regeneration of the native plant species, which may lead to secondary invasion of the cleared area. It is therefore advisable to use the different kinds of control simultaneously for them to be a success in combating the invasive alien plants. It is therefore, very important to have an adaptive management plan (AMP) for the species, due to the fact that none of the current control methods seem to work successfully in the study. Different kinds of control methods (Biological, Chemical and Mechanical) needs to be utilized for successful eradication results on the species such as *Acacia decurrens*, *Lantana camara* and *Populus alba* and recolonization by native plant species in the study area. All the previous chapters showed the need for AMP to be implemented. Regular monitoring of the study area is also very crucial for positive results. It was evident that AMP is needed because the invasive alien plant species kept on re-establishing in the study area regardless of the efforts to control the species. It is also suggested that the conservation managers involve the stakehoders (researchers, the public/citizens,

Non-Government Organizations, and other groups affected) in their management plan discussion as this would serve to disserminate information and create awareness about the damages and cost and benefits of invasive alien species control. It would get a lot of people to volunteer in controlling invasive alien species once they get informed and involved in the decision making.

6.1 Introduction

Ecosystem and human well-being increasingly under threat from the inter-connected challenges of climate change, land degradation, pollution, invasive alien species and other drivers which are bringing us closer to exceeding the world's planetary boundaries (Steffen et al., 2015). Dynamic interactions between these drivers of change in socio-ecological systems make environmental management and conservation issues extremely complex, not least because they require equal consideration of both ecological and social processes (Ostrom, 2009). This is particularly important in invasion science, the study of causes and consequences of the introduction of organisms outside their native ranges by humans, some of which becomes invasive and cause impacts to humans and the environment (Jeschke et al., 2014). Invasion illustrate this complexity because tackling the challenges of invasive species depends on the perceptions, attitudes and behavior of stakeholders as it does on the ecology of how an invasive alien species spreads and causes impacts (Shakleton et al., 2018). For example, there are often conflicts of interest between stakeholders surrounding the management of invasive species due to trade-offs between costs and benefits around economic, social and environmental factors and intrinsic issues (Zengenya et al., 2017). Therefore, new frameworks and approaches for resolving these issues are urgently needed and are being developed (Novoa et al., 2018).

Other challenges have been identified that relate to stakeholders' lack of knowledge and awareness surrounding invasive species (Novoa et al., 2017), concerns regarding the ethics of some management approaches and poor cooperation between different stakeholders (Shackleton et al., 2016). For this reason, researchers are paying more attention to the role of stakeholder engagement in management of biological invasions to improve long-term effectiveness and efficiency of invasive species management. The role of stakeholder engagement is increasingly being recognized in environmental decision making, including national and international policy formulation (Sterling et al., 2017). For example, the Convention on Biological Diversity and the European Strategy on Invasive Alien Species, which underpins European Union regulations (Genovesi et al., 2015), both explicitly recognize the need for stakeholder engagement. Engagement is important for understanding perceptions and practices, promoting awareness, reaching consensus and agreements, solving conflicts, aiding prioritization and planning and formulating co-management programs (Gaertner et al., 2017; Bravo-Vargas et al., 2019). Engagement has led to successful co-management of invasive mink (*Nevison vison*) in Scotland (Bryce et al., 2011), has reduced conflicts of interest and improved consensus regarding the management of invasive cacti in South Africa (Novoa et al., 2016), has promoted collaborative research and awareness through citizen science projects and monitoring in Europe (Marchante et al., 2017), and has aided planning and prioritization of the management of European house borer (*Hylotrupes bajulus*) in Australia (Liu et al., 2010).

Stakeholders are defined as any individual, group or organization who is affected either positively or negatively by invasive species, or who has the capacity to promote or limit the spread of invasive species (Freeman, 1984). They include the public/citizens, researchers, government departments (who are responsible for the management of invaded areas or as policy makers), non-governmental organizations (NGOs), businesses and industry, and many other groups (Novoa et al., 2017). Stkeholder engagement is the process of involving stakeholders in decision making, management actions and knowledge creation surrounding invasive species. Depending on the environmental governance framework within which engagement takes place and the goals it seeks to reach, engagement may vary in terms of its agency, being initiated and facilitated from the top-down by external agencies, bottom-up by affected communities, or some combination of the two (Reed et al., 2017). It may also vary in terms of the mode of engagement, including unidirectional communication, consultation, deliberation and co-production.

While the number of eradication attempts has rapidly increased since the 2000s, cases of successful initiatives away from confined island settings are few (Robertson et al., 2016). Professionally-led management of established invasions over extensive areas is unlikely to be feasible without extensive resources (Simberloff, 2002). As current funding is considered to be insufficient to tackle even the most concerning invasions (Larson et al., 2011), conservation organisations increasingly seek to involve local stakeholders and the general public in theprevention, control and surveillance of invasive alien species (Bryce et al., 2011), and such campaigns can take many forms (Atchison et al., 2017).

Volunteers have become increasingly involved in the management and monitoring of the natural environment worldwide (Tulloch et al., 2013). They represent a significant workforce and source of information in nature conservation projects (Theobald et al., 2015), notably in invasive alien species management programmes (Marchante et al., 2017). Understanding what facilitates and how to encourage people's engagement in volunteering activities is therefore important to improve the success of invasive species management. Volunteering in invasive alien species management can be conceptualised as a form of “local environmental stewardship”, consisting of “actions taken by individuals, groups or networks of actors, with various motivations and levels of capacity, to protect, care for or responsibly use the environment in pursuit of environmental and/or social outcomes in diverse social-ecological contexts” (Bennett et al., 2018).

6.2 Management plan strategy

Conservation managers are required to make decisions and take action in complex systems (Game et al., 2014) that frequently require trade-offs, in the midst of limited resources and data deficiencies (Reed, 2008). Approaches proposed to improve the robustness of decision making, such as Structured Decision Making and Systematic Conservation Prioritisation (Schwartz et al., 2018), entail inclusive stakeholder engagement in the setting of management objectives and priority actions to determine a management strategy. In this way, multiple management actions can be prioritized and management actions determined (Regan et al., 2005). The wide range of inputs arising from the inclusion of scientific, political, social and economic stakeholder perspectives may, however, lead to the formulation of excess or conflicting management objectives (Roper et al., 2018). It is possible that convoluted problems may result in any agreeable solution being deemed better than no solution at all (Game et al., 2013). Acceptance of suboptimal or conflicting objectives can undermine the effective use of limited available conservation funding and resources (Bruner et al., 2004).

Therefore, the assessment of the conservation outcomes of a chosen strategy is needed to determine the potential impact of conflicting objectives. The overall management goal of alien plant control programmes is to reduce the occurrence of Invasive alien plants to densities that have no negative impact on native biodiversity. To achieve this goal, clear objectives, strategies, adaptive planning, adequate resources and funding for long-term implementation are required (van Wilgen et al., 2016).

The potential effectiveness of management can be assessed in terms of administrative and scientific feasibility. Administrative feasibility considers the logistics of management (for example, programme duration, funding, permitting and personnel). In contrast, the extent of the invasion, the biology of the target species and the availability of appropriate control methods determine scientific feasibility. Assessing scientific feasibility is often the initial step of management programmes, in particular, this involves determining the susceptibility of the target species to control techniques (Panetta, 2015). This can be determined using previous experience or through field trials undertaken in the local context. But prior to the initiation of a long-term or large-scale eradication programme, there should ideally also be some estimation of the administrative feasibility.

The goal of management in any discipline is to control the components of any system under consideration. The components of any system can be controlled only if their characteristics are understood (Harper, 1979). Conservators and management practitioners need to be more aware of research findings, and how they might improve management strategies (Negro et al., 2013).

Minimizing the impact of human activities on the structure and composition of ecosystems, and to seek more harmious forms of interaction between the biological diversity and human species is one of conservaton bioloogy's main concerns (Primack et al., 2001). The anthropogenic impact has been defined as a crisis discipline sicnce its origin, referring to the fact that it requires urgent action, and it includes, in most cases, high amount of uncertainty (Soulé, 1985).

Biodiversity conservation actions cannot wait until full knowledge of all factors involved in each situation is available, nor their necessary relationships. This fact goes against traditional scientific formation that drives natural science proffessionals to seek increasingly more information before they can feel comfortable in making decisions. Even though it may still be risky to adopt management measures without precise information, it is unrealistic to think that generic scientific studies can contribute decisively to an improvement in decision making, or that qualitative studies such as lists of species or soil maps are indispensable prior steps for making decision. Avoiding carrying out management action is a decision whose consequences can be more serious than doing something unintentionally (Zalba, 2007).

Biological invasions' management is a clear example of a situation wherein invasive alien species are major agents for environmental degradation (Shackleton et al., 2019). It has been proven that as the invasion process advances, the possibilities for limiting their impact on natural ecosystems is reduced significantly (Lockwood et al., 2013) Therefore, immediate action and early control are actions of maximum priority. The Convention on Biological Diversity (2007) recommends confronting the problem of invasive alien species based on the precaution principle that the lack of scientific certainity should not be used as justification to postpone or not to implement eradication, contention or control actions. Similarly, rapid action to prevent the introduction, establishment or spread of an invasive alien species is recommended, even if there is uncertainity about its impacts in the long term (IUCN, 2000).

6.3 Invasive alien species management framework

A framework that is well informed is necessary in the management of invasive alien plant species. Below is a proposed *Acacia decurrens* management framework for Dion's farm in the Waterberg District of Limpopo Province: *A*. *decurrens* was chosen based on the fact that, it is the species that showed vigorous response in terms of its regeneration in the study area after it was eradicated. There limited studies that show the management of *A*. *decurrens* in South Africa. Therefore, that made it a species of more interest in this study.

6.3.1 Information on management of invasive alien plants

A critical step to efficient management of an invasion process is to detect the key uncertainities or gaps in information, which if solved, would improve the capacity of dealing with the problem significantly. The lack of information can be made into questions, and in turn guide the formulation of hypotheses that are tested by management strategy. Waiting for all the answers before starting with the action is not always necessary especially, with a species such as *Acacia decurrens*. It is best to begin with the control and take advantage of the opportunity to get better understanding to the problem. This is the basis for adaptive management (Nyberg, 1999), a continuous cycle of actions, monitoring, learning and adjustment of new actions that permits increasing the efficiency of control practices for invasive alien species. In adaptive management, control actions are organized as experiments that clearly layout the expected consequences of interventions should the problem's initial diagnosis and premises about the ecosystem's functioning be correct. This study, for

Acacia decurrens, is in agreement with Nyberg (1999). Having the necessary information on filling the knowledge gaps will inform the stakeholders in development of proper management plans. In the case of *A. decurrens,* information gathered will enable conservation managers to recognize the best point to disrupt its invasion cycle.

With the results obtained from the previous chapters, it is therefore suggested that conservation managers to work in close relation with the stakeholders; that is, seek interventions from the public. Get the public involved as volunteers on eradicating the invasive alien species and monitoring the managed sites. This is because, despite the attempts to control these invasive alien plants, they still re-establish with other native or alien species.

6.3.2 Importance of availing information

Despite the obvious benefits of invasive alien plant species management, park programs designed to control invasive species are often viewed unfavourable by the public. Many people oppose invasive alien species management because it can involve activities (that is, lethal removal) that people oppose (Temple, 1990). Extreme opponents have cited nativism or xenophobic motives for invasive species control (Simberloff, 2003). Decisions to eradicate invasive alien species also generate controversy because the general public’s views and perceptions about nature frequently differ from those of ardent conservationists and natural resource managers (Lundberg, 2010).

A general lack of understanding of management goals and objectives can also lead to confusion (McNeely, 2001). Although reduction in damage caused by invasive alien species is the ultimate goal, many public stakeholders perceive the optimal outcome to be an overall reduction in the number of invasive species – a goal that is not necessarily attainable or realistic (Lodge and Shrader-Frechette, 2003). These conflicts show that invasive species management problems can be more political, economic, and cultural than biological (Tanentzap et al., 2009). Hence, the human element of the invasive species issue should not be overlooked or underestimated. In fact, many environmental managers cite a lack of public awareness and support as one of the major obstacles to successful invasive alien species management (Andreu et al., 2009). Popular publications that are easily accessible can play a major role in dissemination of information to the public regarding the management of invasive alien plant species.

It is of outmost importance to make information on *A. decurrens* available, since most people do not know much about this species around the Waterberg District Municipality and Limpopo Province as a whole. However, people utilize it for firewood, fencing poles and as house building materials. This utilization role might be important in minimizing its spread, but it may not that effective since it is not frequently used. Therefore, informing the public of its role in the utilization profile of other communities may make it a species of choice in the rest of the communities. The destructive harvesting pattern for utilization may assist in its eradication if properly monitored.
It is therefore, of outmost importance that the public/communities living near the affected areas be informed or educated about the impacts invasive alien plant species may have on their wellbeing and the socio-ecological effects thereof, so that they see it as their responsibility as well to get involved in saving the native diversity before it is lost.

6.3.3 Forums

A forum should be the nexus for intergrating all components of the invasive alien species management cycle. The forum should include the park managers, staff of the Invasive Species Control Unit (ISCU), project managers, stakeholders and regional ecologists as well as specialist researchers where needed. These forums should take place at least more than once in a year to reassess progress and revise plans, thereafter provide feedback to all. This group of individuals would be the ones to come up with plans, and the proposed planning process would contribute to achieving a common understanding of the overall invasive alien species control programme (Foxcroft and McGeoch, 2011). This study, of *Acacia decurrens*, is in agreement with the above. Therefore, keeping such forums alive will make sure that the issue of management of invasive alien plant speices should always remain foremost.

Disseminating information about *A. decurrens* in stakeholders' forums, would help open the eyes of the public about effects this species has on the indigenous species. Although it is utilized for its wood, it has some devastating impacts of displacing native species because of its aggressive distribution pattern (aggregated) in nature.

6.3.4 Adaptive management policy on invasive alien plants

It is suggested that the stakeholders be involved in all aspects of invasive alien species management as it has some benefits such as building collaboration and consensus among different stakeholders, promoting awareness and social learning, and for advancing and adapting management methods. This also help in coming up with kinds of management plans which would be apllied one after the other in combacting the invasive species. It would also see a lot of the public/citizens wanting to be involved or volunteering (even though it would not be free of charge), after seeing their leaders or fellow community members involved.

Although systems for implementing strategic adaptive management for invasive alien species management have been proposed (Bestbier et al., 2000; Freitag-Ronaldson and Foxcroft, 2003), emphasis has been largely on developing specific threshold of potential concerns. In some cases, the lack of clarity of departmental responsibility for responding to the breached threshold of potential concern led to confusion, delaying management action. This means that, although the thresholds were designed to trigger appropriate interventions (mainly rapid eradication or containment strategies as described by Foxcroft (2009), the lack of an overall policy and management framework within which they could operate impeded a coordinated response. Furthermore, even though the threshold of potential concern includes monitoring for new incursions of alien species in new areas, almost all attention has been on control, with little focus on prevention. Although this is a worldwide trend (Keller and Lodge 2010), a substantial increase in focus on prevention is essential if different areas, such as parks are to manage invasive alien plants successfully in the long run.

In a large natural system such as the Kruger National Park, where many alien species have been introduced (Foxcroft et al., 2003), not all species can or necessarily need to be managed. Other species will also experience fluctuations in their population density and spatial distribution over time, even though the most invasive and potentially damaging species will tend to become and remain highly abundant and widespread (Freitag-Ronaldson and Foxcroft, 2003). It is these 'transformer' species that alter the landscape over a wide area, often in both structure and function (Pysek et al., 2004), that are of most concern and therefore the focus of management attention. Without an overarching framework and coordinated effort to identify where and when thresholds of potential concern are required (Tu, 2009), and to respond when they are breached, efforts will remain largely opportunistic and potentially ineffective (Hall and Fleishman, 2009).

Adaptive management is another aspect that requires further consideration, which is the process by which social systems are organized or reorganize themselves (Folke et al., 2005). Management provide the structures and processes needed in which collective decisions are taken. Adaptive governance networks join teams that contain different experiences to develop a common understanding of management and polices. The governance networks function across multiple levels within organizations and between individuals (Folke et al., 2005). These networks provide for enhanced flexibility (a requisite for rapid response to invasions), but still operate within the organizational hierachy, holding specific agencies, management levels or indivuduals accountable for inaction or mismanagement (Foxcroft and McGeoch, 2011).

In discussing biosecurity risks of invasive alien species in Australia, Cook et al. (2010) discussed governance structures for interagency coorperation to improve information transfer for surveillance, response, diagnostics, risk analysis and border inspections. These functions are carried out by a number of organizations in different ways. It is therefore, suggested that greater focus on the governance structure of invasive alien species management is required for improved utility of the framework (Foxcroft and McGeoch, 2010). Stakeholders around conservation areas must also be looked upon in terms of governance for developed adaptive management policies to be embraced and be effective.

It is important to identify problems or threats posed by *Acacia decurrens* and decide on the management policy to be used in order to yield positive results. *A. decurrens* management policy should indicate how the species should be eradicated. The objective need to be clear, in this case the objective was to eradicate the area infested with *A. decurrens* and monitor the area for any recolonization or secondary invasion by the species. In Dion's farm, *A. decurrens* re-established in large numbers after its removal in 2013. It is therefore, very important for close monitoring to be practiced in the area that is infested with the invasive species. The methods need to be changed every time they prove unsuccessful, until those that work (meeting objectivest) are identified.

In line with the proposed objectives centred adaptive management plan (Figure 6.1), active restoration of native plant species should also be introduced in the farm, after assessing the species compatibility to establish in that area. This would assist in making sure that invasive alien plant species are suppressed while giving the opportunity for native species to fully establish with less competition from non-native species.

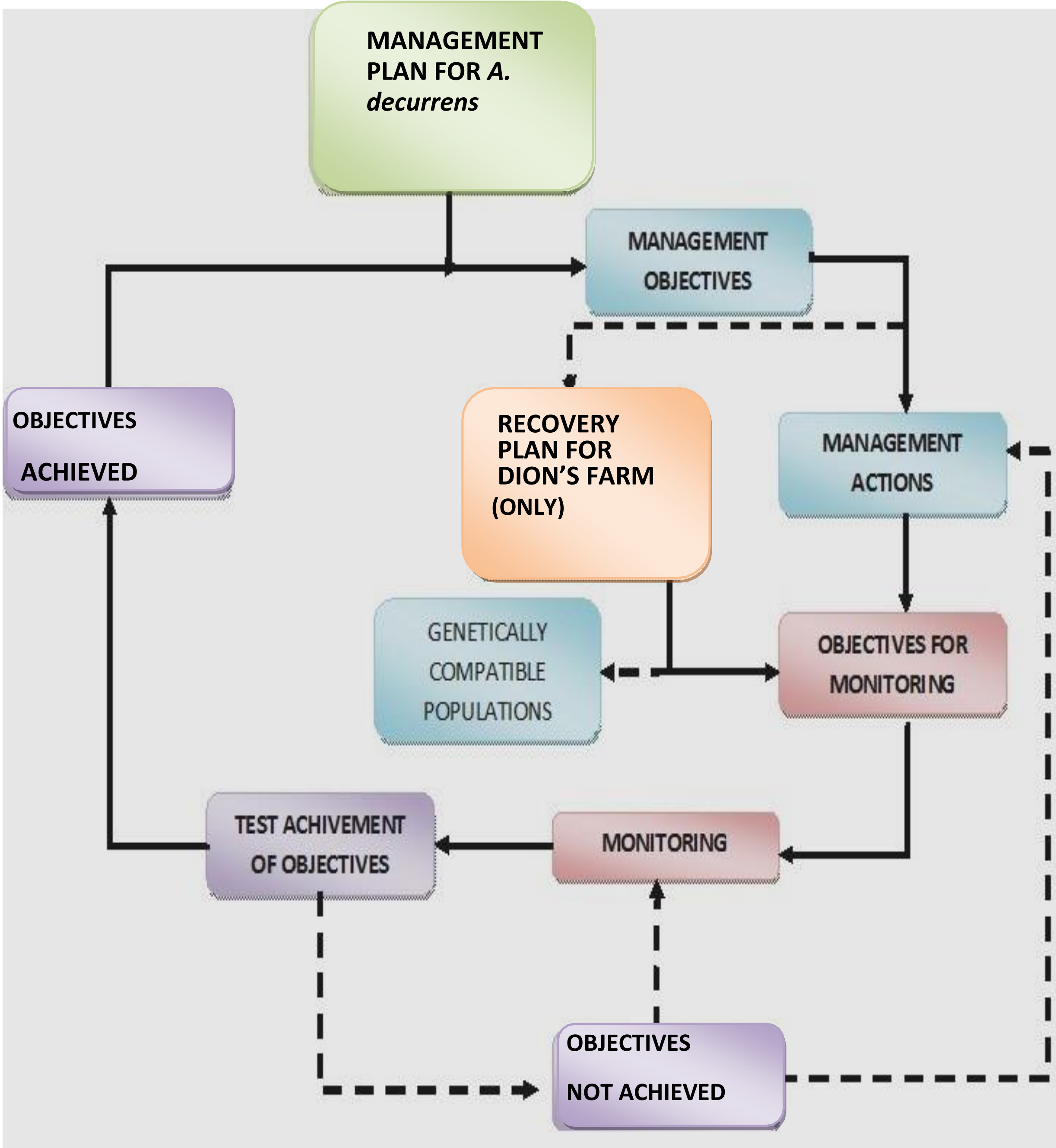

Figure 6.1: Proposed adaptive management plan for *Acacia decurrens.*

Restoration plan should promote more grass cover to suppress the invasive seeds that are in the seed pool. Planting and/or transplanting of native plant species upslope would also be helpful, in the sense that when it rains the native seeds would be washed down the slope towards the river and therefore, giving more chance of establishment to native plant species.

The management actions should include the removal of these invasive plants mechanically, by means of cutting and applying the herbicides on the stumps of the individuals. Followed introduction of a specific biological control agent that will attack the invasive individuals as they emerge as seedlings or sprouts from the stumps.

The objective for monitoring the controlled sites is to make sure that any plant species that germinates there, is removed immediately before is can fully establish and cause problems to native species. To monitor the progress of the native plants and see if there is any factor (soil structure) that seems to impose negative impacts on them.

The recovery plan includes plans that may be made when the native plant species cannot recolonize the sites on their own. Therefore, there should be active restoration of the area, in the form of transplanting seedlings of the native plant species found in the same vicinity. This would present the native plants with a significant chance to establish successfully. Checking the genetically compatiable plant species (by looking at the adjacent species of the area) is very important, since not all the species can withstand the conditions of the area.

6.3.5 Management and control of invasive alien plants

Acacia decurrens, *Lantana camara* and *Populus alba*, have all been exposed to different kinds of control mechanism, including mechanical and chemical methods of control, and still they emerged in the same areas. It would seem that the two methods of control are not effective in controlling these invasive species which are seen displacing the native species growing in the same vicinity. It is therefore suggested that the combination of these control methods including biological control, should be used in this case. The methods would need to be administered one after the other, for example; start with biological control, cutting of trees, and apply herbicides on the stumps.

6.3.5.1 Mechanical control

Mechanical control options include the physical felling or uprooting of plants, their removal from the site, often in combination with burning. When fire is used, it can be applied in conjunction with physical control (for example fell and burn or burn and follow-up with hand weeding). The equipment used in mechanical control ranges from hand-held instruments (such as saws, slashers and axes) to power-driven tools such as chainsaws and brushcutters, and even to bulldozers in some cases. Mechanical control is labour-intensive and thus expensive to use in extensive and dense infestations, or in remote or rugged areas (Lambert et al., 2010).

In most cases, mechanical control seems to be less effective when used alone while eradicating and the area left alone to recover without active restoration. This was the case with *A. decurrens* in the study area (Dion's Farm) back in the years 2012, 2013 and 2014, when the people from the Working for Water Programme eradicated the species. It is of much importance to use the combination of different kinds of control methods to get rid of the invasive alien species such as *A. decurrens*. Regular monitoring of the area which was previously infested with the species is important for positive results to be observed.

6.3.5.2 Chemical control

Herbicides can be applied to prevent sprouting of cut stumps, or to kill seedlings after felling or burning. Herbicides can target, for example, grasses or broad-leaved species, leaving other plants unharmed. However, there are legitimate concerns over the use of herbicides in terms of potential environmental impacts. Although newer herbicides tend to be less toxic, have shorter residence times, and are more specific, concerns over detrimental environmental impacts still remain. The use of chemical control is often governed by legislation, and the effective and safe use of herbicides requires a relatively high level of training; both of these factors can restrict the use of chemical control on a large scale (Netherland, 2014). Generally, chemical control works, but for *A*. *decurrens*, only to a limited extent at Dion's farm. The chemicals used to control invasive species may have a negative impact on the native species growing in the same vicinity as *A. decurrens*.

It is, therefore, suggested that for it to be more effective it would need to be used at the same time with mechanical and biological control. *Acacia decurrens* at Dion's farm, survived the chemical and mechanical control in 2013. Even when the trees were cut and the herbicide applied on the stumps, some of the stumps were still able to produce sprouts that established successfully. The lack of regular monitoring of the area might also have played a role. After eradication in 2016 using mechanical control (uprooting the whole tree), the only way the *A. decurrens* was seen establishing was through seedlings.

6.3.5.3 Biological control

Biological control (or biocontrol) involves using species specific insects or other invertebrates, and diseases, from the alien plants region of origin. Most invasive alien plants show no weedy behaviour in their natural ranges their ability to grow vigorously and produce huge amounts of seeds is kept in check by a host of co-evolved organisms. Some species, when transported to a new region without the attendant enemies, grow more vigorously and produce many more seeds than in their native ranges, and become aggressive invaders. Biocontrol aims to reduce the effects of this phenomenon, and to achieve a situation where the formerly invasive alien plant becomes a non-invasive naturalised alien. Biological control has many potential benefits, including its potential cost-effectiveness, and the fact that it is (usually) environmentally benign. Some interest groups have expressed concern about the potentially negative effects on non-target plants, or on weeds that may have important commercial value, such as the pines, which are important for timber production but invade large areas outside plantations (van Wilgen et al., 2001). This study is in agreement with the above information, because it talks about using species specific biological agents.

If a species-specific biological control agent is used in Dion's farm infested with *A. decurrens*, it would assist in eradicating it in the area, which in turn will enable other native species to establish. However, it would need the area to be monitored and other methods of control be used together with the biological control. No species specific agent for A. decurrens, this is therefore an area that needs to be investigated further in future.

6.3.6 Improving management efficiency

Monitoring not only permits adjusting the diagnosis but assist in selecting the suitable control methods for each species in each situation, as well as the most efficient procedure to apply these methods. The success of the control action in an area is not measured by the number of cut trees, but by the changes associated with vegetational community, since there is no avenue to eliminate invasive alien species if cutting causes erosion problems that aggravate degradation of natural systems (Zalba, 2005).

Creating awareness, involving the public, Non-Government Organizations, researchers and many other groups of people, would see the management of these invasive alien species being a success. Since we are saying that it is sometimes extremely difficult working in local communities around the management of invasive species, since there are concerns that such species have grown to be useful for the affected group of people/community.

Reproductive aspects should be studied so as to determine at what stage the invading trees begin to produce seeds, what happens to the seeds that are stuck in cut trees which were not removed, what impact fallen trees have on adjacent vegetaion and the effect herbivores and fire may have on the advancement of invasive alien trees, among others. The answers to these questions would then be used to make decisions on which areas are priorities for control, when is it most convenient to cut the trees or with what frequency is it necessary to return to each control area and repeat the actions until the seed bank or sprouting process is eliminated (Cuevas, 2005).

Closely monitoring an area after eradication is very important, in a sense that any unwanted change can be detected early and dealt with before it can fully establish. This was observed with *A. decurrens*, wherein the seedlings would be uprooted at an early stage. It gave other plant species (grasses and herbs) a chance to establish in Dion's farm. It then showed that there is a chance for other plants species to colonize the area previously infested with *A. decurrens* provided there is continued regular monitoring of the area.

6.3.7 Multiple management mechanisms

It is of outmost importance that if one method fails, the other method be tried out until a suitable method of control is obtain. In some cases, what is need is continued monitoring of the eradicated areas for reinvasion by the same species or another species. To determine a management conservation, managers are required to make decisions and act in complex systems (Game et al., 2014) that frequently require trade-offs, in the midst of limited resources and data deficiencies (Reed, 2008). In this way, multiple management actions should be prioritized and management actions determined. The wide range of inputs arising from the inclusion of scientific, political, social and economic stakeholder perspectives may, however, lead to the formulation of excess or conflicting management objectives (Roper et al., 2018). It is possible that convoluted problems may result in any agreeable solution being deemed better than no solution at all (Game et al., 2013). Acceptance of suboptimal or conflicting objectives can undermine the effective use of limited available conservation funding and resources (Emerton et al., 2006; Ferraro and Pattanayak, 2006). Therefore, the assessment of the conservation outcomes of a chosen strategy is needed to determine the potential impact of conflicting objectives.

Predictive models that consider ecological drivers such as fire, invasion rate, ecological impact and factors that increase uncertainty, such as clearing efficiency, can provide estimates of expected outcomes defined by particular sets of conservation objectives and resource allocations (Cheney et al., 2019).

Mechanical (cutting individual trees at 5 cm above the ground) and chemical (glyphosate) methods have been used before in 2013 to control *Acacia decurrens* in Dion’s farm. The methods proved to be a success, but only to lesser extent as the species recolonized the area in large numbers. Even after fire was used, the success was short lived, as the area was soon covered with large number of seedlings in a matter of just 5 months. It is therefore, suggested that, using the three different kinds of control methods (viz, Chemical, mechanical and biological method) would yield positive results. Nonetheless, it would need the area to be closely monitored for any undesired plant species.

6.4 Monitoring and evaluation of invasive alien plants

Monitoring of the eradicated areas is very crucial. With the three species under study, monitoring was done, even though they kept on re-emerging, it gave other species a chance to establish, which would have not been possible if the areas were not closely monitored after eradication. The eradicated areas need also to evaluated, if there are any factor (such as soil structure) that seem to hinder the establishment of native species, it should be attended to allow for previous species to recolonize the area.

Other studies also support this suggestion, for example, Zalba and Ziller (2007), Monitoring not only permits adjusting the diagnosis, but it also helps select the most appropriate control methods for each species in each situation, as well as the most efficient manner to apply these methods. Monitoring is inherent in defining adaptive management because management is designed to meet objectives, while monitoring is designed to determine if objectives are met (Elzinga et al., 1998). Monitoring is a cornerstone on which all forms of adaptive management (Salafsky et al., 2002), mainly because adaptive management requires implementing management actions as experiments. As a consequence, monitoring is crucial in determing how the system responds to management (Biggs and Rodgers, 2003).

There have been two eradication programmes in Australia that have met the criteria Myers et al. (2000) established for successful eradication. The six criteria are first: sufficient funding to undertake a large-scale project; second, that authority must be granted to an agency to allow for the development and maintenance of regulations, treatments and monitoring; third, for the biology of the invasive animal to be known and specifically for its susceptibility to control measures to be assessed; fourth, to prevent reinvasion; fifth, to monitor the pest at low densities; and finally, for restoration and continued management of the invaded environment, particularly if the invader has become a ‘keystone’ species in the habitat.

6.5 Implication to conservation

6.5.1 Biodiversity

The impacts and contributions of invasive alien species to global biodiversity loss have reached levels that attract the attention of all scientists concerned. The threat posed to biodiversity by invasive alien species is considered second only to that of habitat loss (CBD, 2005). Invasive alien species may threaten native species as direct predators or competitors, as vectors of disease, or by modifying the habitat or altering native species dynamics (MA, 2006). *Acacia decurrens* may out-compete native species, repressing or excluding them, and therefore fundamentally changing the ecosystem due to the way they are distributed (aggregated) in nature, thereby hindering indigenous species from getting some of the resources, such as, sunlight, water and nutrients. They may transform the structure and species composition of the ecosystem by changing the way by which nutrients are cycled through the ecosystem (McNeely et al., 2001). Furthermore, invasive alien species may cause changes in environmental services, such as flood control, water supply, water assimilation, nutrient recycling, conservation, and regeneration of soils (GISP, 2004). The colonization of fruit tree canopies, as observed in the field gene bank, can reduce the fruiting potentials of the trees, and this can adversely affect the population of frugivorous animals and birds that depend on the fruits for food. There is also risk in the gene bank of habitat degradation and changes in the species composition.

Given the increasingly high cost and economic burden of controlling invasive species (such as *Lantana camara*), in agricultural and natural ecosystems, there is a clear need to determine the spatial and temporal scale over which impacts occur, the identities of the invasive plants that drive the greatest impacts, and the ecosystems most vulnerable to change, so that the limited resources for control can be prioritised to areas most likely to be impacted. The very scarce resources available for invasive plant control in natural ecosystems means that the likelihood of eradicating widespread and well-established invaders is diminishingly small (Panetta and James, 1999). Prioritisation must be given in such circumstances to controlling widespread alien plants in sites of high conservation priority (such as the wetlands, which where infested by *Populus alba* in the study area) and containing their spread elsewhere (Cousens, 1987).

6.5.2 Water sources

Many species of invasive alien plants, especially trees (*Acacia decurrens* and *Populus alba*) and shrubs (*Lantana camara*), have higher evaporation rates than indigenous species do and, therefore, use more water than the vegetation they replace (Malan and Day, 2002). The increased evaporation results in reductions in river flows and reduced groundwater reserves (Malan and Day, 2002). The reduction in the amount of water in the river reduces its dilution capacity, resulting in increased concentrations of nutrients and pollutants, increased salinity and altered buffering capacity of the ecosystem (Nagler et al., 2008).

Le Maitre et al. (2000), estimated a reduction in surface water runoff as a result of invasions to be 3 300 mm^3 (about 7% of the national total), most of which is from the fynbos (shrubland) and grassland biomes (Van Wilgen et al., 2008). These areas are especially sensitive to invasions as most of South Africa’s surface water originates from the Drakensberg mountain grasslands, and from the Cape Mountains which are dominated by fynbos vegetation (Turpie et al., 2008).

The potential water reductions would be more than 8 times greater if invasive alien plants were to occupy the full extent of their potential range (Van Wilgen et al., 2008). These invasions come at a significant cost to the economy, estimated at about R6.5 billion per annum (about 0.3% of South Africa's GDP of around R2 000 billion), and with the potential to rise to > 5% of GDP if invasive plants were to be allowed to invade all of the suitable habitat (De Lange and Van Wilgen, 2010).

The impacts of invasive alien plants on surface water runoff and groundwater recharge are relatively well understood and can be used to set priorities. Although scientists recognise that invasive alien plants have negative impacts on water quality as well as quantity, a shortage of information regarding these impacts has prevented them from being considered in prioritisation exercises by institutions such as Working for Water.

6.6 Recommendations and Conclusions

It is recommended that the conservation managers engage a team of stakeholders with the ideas to conserve the natural resources or the indigenous species. The stakeholders should be part of the decision making team and be involved all the eradication activities. Comprehensive stakeholder engagement should be crucial facet of all management project proposals and most applied proposal and should be formally evaluated.

Conservation managers are required to make decisions and act in complex systems (Game et al., 2014) that frequently require trade-offs, in the midst of limited resources and data deficiencies (Reed, 2008). Approaches proposed to improve the validity and robustness of decision making, such as Structured Decision Making and Systematic

Conservation Prioritization (Bower et al., 2017; Schwartz et al., 2018), entail inclusive stakeholder engagement in the setting of management objectives and priority actions. Prioritization models for invasive alien plant management that attempt to account for multiple objectives and uncertainties have been developed for a number of applications and include water catchment areas (van Wilgen et al., 2007; Forsyth et al., 2012), protected area management (Forsyth and Le Maitre, 2011; van Wilgen et al., 2016) and maximization of economic cost-benefit ratios (de Wit et al., 2001). However, the prioritization of areas for management intervention is sensitive to the weighting of component factors, objectives and the availability of information (Roura-Pascual et al., 2010). In addition, the impact of strategy on priority area selection is generally only tested as a static, once-off assessment without consideration of iterative and changing reality over time (Roura-Pascual et al., 2010; Forsyth et al., 2012).

Ecosystems everywhere are subject to disturbance, fragmentation, and invasion by alien species that are driving them outside of their historical ranges of variability (Seastedt et al., 2008) and thus they become ‘novel ecosystems’ (Hobbs et al., 2006). The restoration of novel ecosystems to their original state is practically and almost always unachievable (Seastedt et al., 2008). Where conventional mechanical and chemical control or other non-biological methods for dealing with invasive alien plants prove to be ineffective, biological control becomes the only viable, sustainable solution. In areas that are already severely and irreversibly degraded (or will become so in the absence of biological control), it would be incorrect, or misleading, or arguably unethical to insist that WBC agents should not be used because they have the potential to effect some changes to existing ecosystem processes, their composition and structure.

References

Andreu, J., Vila, M. and Hulme, P.E., 2009. Anassessment ofstakeholder perceptions and management of noxious alien plants in Spain. *Environmental Management* 43, 1244–1255.

Atchison, J., Gibbs, L. and Taylor, E., 2017. Killing carp (Cyprinus carpio) as a volunteer practice: implications for invasive species management and policy. Aust. Geogr. 48, 333–348.

Bennett, N.J., Whitty, T.S., Finkbeiner, E., Pittman, J., Bassett, H., Gelcich, S. and Allison, E.H., 2018. Environmental stewardship: a conceptual review and analytical framework.Environ. Manag.61, 597–614

Bestbier, R.X., Jacoby, D.L. and Rogers, K.H., 2000. A goal maintenance system for the management of the Kruger National Parks riverine alien vegetation: developing a protocol and a prototype. *WRC Report* (813/2/00).

Biggs, H.C. and Rogers, K.H., 2003. An adaptive system to link science, monitoring, and management in practice. In J.T. du Toit, K.H. Rogers & H.C. Biggs (eds.), *The Kruger Experience: ecology and management of savanna heterogeneity*, pp. 59-80, Island Press, Washington DC.

Bower, S.D., Brownscombe, J.W., Birnie-Gauvin, K., Ford, M.I., Moraga, A.D., Pusiak, R. J.P., Turenne, E.D., Zolderdo, A.J., Cooke, S.J. and Bennett, J.R., 2017. Making tough choices: picking the appropriate conservation decision-making tool. *Conservation Letters* 11, e12418.

Bravo-Vargas, V., Garcia, R.A., Pizzarro, C., Pauchard, A., 2019. Do people care about pine invasion? Visitor perceptions and willingness to pay for pine control in a protected area. *Journal of environmental Management* 229, 57 – 66.

Bryce, R., Oliver, M.K., Davies, L., Gry, H., Urquhart, J., Lambin, X., 2011. Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management. *Biological Conservation* 144, 575 – 583.

Convention on Biological Diversity, 2005. Invasive Alien Species. Convention on Biological Diversity

Cook, D.C., Liu, S., Murphy, B., Lonsdale, W.M., 2010, 'Adaptive approaches to biosecurity governance', *Risk Analysis* 3(9).

Cousens, R., 1987. Theory and reality of weed control thresholds. *Plant Protection Quarterly* 2, 13-20.

De Lange, W, J. and van Wilgen, B.W., 2010. An economic assessment of the contribution of weed biological control to the management of invasive alien plants and to the protection of ecosystem services in South Africa. *Biological. Invasions* 12 (12) 4113-4124.

de Wit, M.P., Crookes, D.J. and van Wilgen, B.W., 2001. Conflicts of interest in environmental management: estimating the costs and benefits of a tree invasion. *Biological Invasions* 3, 167–178.

Elzinga, C.L. and Salzer, D.W., 1998. *Measuring & monitoring plant populations*. US Department of the Interior, Bureau of Land Management.

Emerton, L., Bishop, J. and Thomas, L., 2006. Sustainable Financing of Protected Areas: A Global Review of Challenges and Options. IUCN.

Ferraro, P.J. and Pattanayak, S.K., 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biology* 4, e105.

Folke, C., Hahn, T., Olsson, P. and Norberg, J., 2005. Adaptive governance of social-ecological systems. *Annual Review of Environment and Resources* 30, 441–473.

Forsyth, G.G. and Le Maitre, D.C., 2011. Prioritizing national parks for the management of invasive alien plants: report on the development of models to prioritize invasive alien plant control operations. In: CSIR Natural Resources and the Environment Report Number: CSIR/NRE/ECO/ER/2011/0036/B. CSIR, Stellenbosch.

Forsyth, G.G., Le Maitre, D.C., O'Farrell, P.J. and van Wilgen, B.W., 2012. The prioritization of invasive alien plant control projects using a multi-criteria decision model informed by stakeholder input and spatial data. *Journal of Environmental Management* 103, 51–57.

Forsyth, T., 2004. Critical political ecology: the politics of environmental science. Routledge.

Foxcroft, L.C. and McGeoch, M., 2011. Implementing invasive species management in an adaptive management framework. *Koedoe*, 53(2), 105–115.

Foxcroft, L.C., 2009. Developing thresholds of potential concern for invasive alien species: hypotheses and concepts. *Koedoe* 50(1), 6.

Foxcroft, L.C., Rouget, M., Richardson, D.M. and Mac Fadyen, S., 2004. Reconstructing 50 years of Opuntia stricta invasion in the Kruger National Park, South Africa: environmental determinants and propagule pressure. *Diversity and Distributions* 10(5-6), 427–437.

Freeman, R.E., 1984. Strategic Management: A Stakeholder Approach. Basic Books, New York.

Freitag-Ronaldson, S. and Foxcroft, L.C., 2003. Anthropogenic influences at the ecosystem level. The Kruger experience: Ecology and management of savanna heterogeneity, pp.391–421.

Gaertner, M., Novoa, A., Fried, J. and Richardson, D.M., 2017. Managing invasive species in cities: a decision support framework applied to Cape Town. Biological Invasions 19, 3461–3469.

Game, E.T., Kareiva, P. and Possingham, H.P., 2013. Six common mistakes in conservation priority setting. *Conservation Biology* 27, 480–485.

Game, E.T., Meijaard, E., Sheil, D. and McDonald-Madden, E., 2014. Conservation in a wicked complex world; challenges and solutions. *Conservation Letters* 7, 271–277.

Genovesi, P., Carboneras, C., Vila, M. and Walton, P., 2015. EU adopts innovative legislation on invasive species: a step towards a global response to biological invasions? Biological invasions 17, 1307–1311.

GISP, 2004. Africa Invaded: The Growing Danger of Invasive Alien Species. Global invasive Species Programme, Cape Town.

Hall, J.A. and Fleishman, E., 2010. Demonstration as a means to translate conservation science into practice. *Conservation Biology*, 24(1), 120–127.

Harper, K.T., 1979. Some reproductive and life history characteristics of rare plants and implications of management. *Great Basin Naturalist Memoirs*, pp.129–137.

Hobbs, R.J,, Arico, S., Aronson, J., Baron, J.S., Bridgewater, P., Cramer, V.A., Epstein, P.R., Ewel, J.J., Klink, C.A., Lugo, A.E., Norton, D., Ojima, D., Richardson, D.M., Sanderson, E.W., Valladares, F., Vila, ` M., Zamora, R. and Zobel, M. 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15, 1–7.

International Union for Conservation of Nature, 2000. Red book of threatened mammals of Bangladesh. IUCN-The World Conservation Union, 294.

Jeschke, J.M., Bacher, S., Blackburn, T.M., Dick, J.T., Essl, F., Evans, T., Gaertner, M., Hulme, P.E., Kuhn, I., Mrugala, A. and Pergl, J., 2014. Defining the impact of non-native species. *Conservation Biology* 28, 1188–1194.

Keller, R.P. and Lodge, D.M., 2010. Prevention: designing and implementing national policy and management programs to reduce the risks from invasive species. *Bioinvasions and globalization: Ecology, economics, management, and policy*, p.220.

Lambert, E., Dutartre, A., Coudreuse, J. and Haury, J., 2010. Relationships between the biomass production of invasive Ludwigia species and physical properties of habitats in France. *Hydrobiologia*, 656(1), 173–186.

Larson, D.L., Phillips-Mao, L. and Quiram, G., 2011. A framework for sustainable invasive species management environmental, social, and economic objectives. J. Environ. Manag. 92, 14–22.

Le Maitre, D.C., Van Wilgen, B.W., Chapman, R., McKelly, D., 1996. Invasive plants and water resources in the Western Cape Province, South Africa: modelling the consequences of a lack of management. *Journal of Applied Ecology* 161–172.

Le Maitre, D.C., Versfeld, D.B. and Chapman, R.A., 2000. Impact of invading alien plants on surface water resources in South Africa: A preliminary assessment.

Liu, S., Proctor, w. and Cook, D., 2010. Using an intergrated fuzzy set and deliberative multi-criteria evaluation approach to facilitate decision-making in invasive species management. *Ecological Economics* 69, 2374–2382.

Lodge, D.M., Shrader-Frechette, K., 2003. Nonindigenous species: ecological explanation, environmental ethics, and public policy. *Conservation Biology* 17, 31–37.

Lockwood, J.L., Hoopes, M.F. and Marchetti, M.P., 2013. *Invasion ecology*. John Wiley & Sons.

Lundberg, A., 2010. Conflicts between perception and reality in the management of alien species in forest ecosystems: a Norwegian case study. Landscape Research 35, 319–338.

MA, 2006. Ecosystems and Human Well-being: Current State and Trends. Volume 1 Millennium Ecosystem Assessment. Island Press, Washington

Malan, H. and Day, J., 2002. Linking discharge, water quality and biotic response in rivers: a literature review p. 132.

Marchante, H., Morais, M.C., Gamela, A. and Marchante, E., 2017. Using a WebMapping platform to engage volunteers to collect data on invasive plants distribution. T. GIS 21, 238–252.

McNeeley, J.A., Mooney, H.A., Neville, L.E., Schei, P. and Waage, J.K. 2001. Global Strategy on Invasive Alien Species. IUCN – the World Conservation Union, Gland.

Myers, J.H., Savoie, A. and van Randen, E., 1998. Eradication and pest management. *Annual Review of Entomology* 43, 471–491.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A. and Kent, J., 2000. Biodiversity hotspots for conservation priorities. *Nature*, 403(6772), 853–858.

Nagler, P.L., Glenn, E.P., Didan, K., Osterberg, J., Jordon, F. and Cunningham, J., 2008. Wide-area estimates of stand structure and water use of Tamarix spp. on the lower Colorado River: Implications for restoration and water management projects. *Restoration Ecology*. 16 (1), 136–145.

Netherland, M.D., 2014. Chemical control of aquatic weeds. Biology and control of aquatic plants. A best management practices handbook. 3rd edition. Aquatic Ecosystem Restoration Foundation: Marietta (GA), pp.71–88.

Novoa, A., Kaplan, H., Wilson, J.RU. and Richardson, D.M., 2016. Resolving a prickly situation: involving stakeholders in invasive cactus management in South Africa. *Environmental Management* 57, 998–1008.

Novoa, A., Shackleton, R., Canavan, S., Cybele, C., Davies, S.J., Dehnen-Schmutz, K., Fried, J., Gaertner, M., Geerts, S., Griffiths, C.L. and Kaplan, H., 2018. A framework for engaging stakeholders on the management of alien species. *Journal of environmental management*, *205*, pp.286-297.

Nyberg, B., 1999. An Introductory Guide to Adaptive Management for Project Leaders and participants. Forest Practices branch, British Columbia Forest Service, Victoria, BC. pp 22.

Ostrom, E., 2009. A general framework for analyzing sustainability of social-ecological systems. *Science*, *325*(5939), 419–422.

Panetta, F.D. and James, R.F., 1999. Weed control thresholds: a useful concept in natural ecosystems? *Plant Protection Quarterly*, 14, 68–76.

Pyšek, P., Richardson, D.M., Rejmánek, M., Webster, G.L., Williamson, M. and Kirschner, J., 2004. Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. *Taxon*, 53(1), pp.131-143.

Reed, M.S., 2008. Stakeholder participation for environmental management: a literature review. *Biological Conservation* 141, 2417–2431.

Robertson, P.A., Adriaens, T., Lambin, X., Mill, A., Roy, S., Shuttleworth, C.M. and SuttonCroft, M., 2016. The large-scale removal of mammalian invasive alien species in Northern Europe. Pest Management Science.

Roper, B.B., Capurso, J.M., Paroz, Y., Young, M.K., 2018. Conservation of aquatic biodiversity in the context of multiple-use management on national forest system lands. *Fisheries* 43, 396–405.

Roura-Pascual, N., Krug, R.M., Richardson, D.M., Hui, C., 2010. Spatially-explicit sensitivity analysis for conservation management: exploring the influence of decisions in invasive alien plant management. *Divers Distribution* 16, 426–438.

Salafsky, N., Margoluis, R., Redford, K.H. and Robinson, J.G., 2002, 'Improving the practice of conservation: a conceptual framework and research agenda for conservation science', *Conservation Biology* 16(6), 1469–1479.

Schwartz, M.W., Cook, C.N., Pressey, R.L., Pullin, A.S., Runge, M.C., Salafsky, N., Sutherland, W.J., Williamson, M.A., 2018. Decision support frameworks and tools for conservation. *Conservation Letters* 11, e12385.

Seastedt, T.R., Hobbs, R.J. and Suding, K.N., 2008. Management of novel ecosystems: are novel approaches required? *Frontiers in Ecology and the Environment* 6, 547–553

Shackleton, C.M., Ruwanza, S., Sinasson Sanni, G.K., Bennet, S., De Lacy, P., Modipa, R., Mtati, N., Sachikonye, M. and Thondhlana, G., 2016. Unpacking Pandora's Box: understanding and categorising ecosystem disservices for environmental management and human wellbeing. Ecosystems 19, 587–600.

Shackleton, R.T., Witt, A.B.R., Nunda, W. and Richardson, D.M., 2017. Chromolaena odorata (Siam weed) in eastern Africa: distribution and socio-ecological impacts. Biol. Invasions 19, 1285–1298.

Shackleton, R.T., Richardson, D.M., Shackleton, C.M., Bennett, B., Crowley, S.L., Dehnen-Schmutz, K., Estévez, R.A., Fischer, A., Kueffer, C., Kull, C.A. and Marchante, E., 2019. Explaining people's perceptions of invasive alien species: a conceptual framework. *Journal of Environmental Management*, 229, 10-26.

Simberloff, D., 2002. Today Tiritiri Matangi, tomorrow the world! Are we aiming too low in invasives control? In: Veitch, C.R., Clout, M.N. (Eds.), Turning the Tide: the Eradication of Invasive Species. IUCN SSC Invasive Species Specialist Group, Gland, Switzerland and Cambridge, U.K, pp. 4–12.

Soulé, M.E., 1985. What is conservation biology? *BioScience* 35(11), 727–734.

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., De Vries, W., De Wit, C.A. and Folke, C., 2015. Planetary boundaries: Guiding human development on a changing planet. *Science*, 347(6223).

Sterling, E.J., Betley, E., Sigouin, A., Gomez, A., Toomey, A., Cullman, G., Malone, C., Pekor, A., Arengo, F., Blair, M. and Filardi, C., 2017. Assessing the evidence for stakeholder engagement in biodiversity conservation. *Biological conservation* 209, 159–171.

Tanentzap, A.J., Bazely, D.R., Williams, P.A. and Hoogensen, G., 2009. A human security framework for the management of invasive nonindigenous plants. *Invasive Plant Science and Management* 2, 99–109.

Temple, S.A., 1990. Editorial: the nasty necessity: eradicating exotics. *Conservation Biology* 4, 113–115.

Theobald, E.J., Ettinger, A.K., Burgess, H.K., DeBey, L.B., Schmidt, N.R., Froehlich, H.E., Wagner, C., HilleRisLambers, J., Tewksbury, J., Harsch, M.A. and Parrish, J.K., 2015. Global change and local solutions: Tapping the unrealized potential of citizen science for biodiversity research. *Biological Conservation*, 181, 236–244.

Tu, M., 2009, 'Assessing and managing invasive species within protected areas' in J. Ervin (ed.), Protected area quick guide series, pp. 1–40, The Nature Conservancy, Arlington, VA.

Tulloch, A.I.T., Mustin, K., Possingham, H.P., Szabo, J.K. and Wilson, K.A., 2013. To boldly go where no volunteer has gone before: predicting volunteer activity to prioritize surveys at the landscape scale. *Divers of Distribution* 19, 465–480.

Turpie, J.K., Marais, C. and Blignaut, J.N., 2008. The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. *Ecological Economics*. 65 (4), 788–798.

van Wilgen, B.W., Fill, J.M., Baard, J., Cheney, C., Forsyth, A.T., Kraaij, T., 2016. Historical costs and projected future scenarios for the management of invasive alien plants in protected areas in the Cape Floristic Region. *Biological Conservation* 200, 168–177.

van Wilgen, B.W., Nel, J.L., Rouget, M., 2007. Invasive alien plants and South African rivers: a proposed approach to the prioritization of control operations. *Freshwater Biology* 52, 711–723.

van Wilgen, B.W., Reyers, B., Le Maitre, D.C., Richardson, D.M. and Schonegevel, L., 2008. A biome-scale assessment of the impact of invasive alien plants on ecosystem services in South Africa. *Journal of Environmental Management* 89 (4), 336–349.

Van Wilgen, B.W., Richardson, D.M., Le Maitre, D.C., Marais, C. and Magadlela, D., 2001. The economic consequences of alien plant invasions: Examples of impacts and approaches to sustainable management in South Africa. *Environment, Development and Sustainability* 3, 145–168.

Zalba, S. and Ziller, S.R., 2007. Adaptive management of alien invasive species: putting the theory into practice. *Natureza & Conservação*, 5(2), pp.86-92.

Zengeya, T., Ivey, P., Woordford, D.J., Weyl, O., Novoa, A., Shackleton, R., Richardson, D. and van Wilgen, B., 2017. Managing conflict-generating invasive species in South Africa: challenges and trade-offs. Bothalia 47, a2160.

CHAPTER SEVEN

GENERAL DISCUSSION, CONCLUSION AND RECOMMENDATIONS

7.1 General discussion

Native plant species can recolonize areas that were previously taken over by invasive alien plant species, provided the site is monitored continuously after the removal of those invasive alien plants. It is crucial that a prudent application is assumed when controlling the introduction of invader plants in order to reduce their influence on biological diversity and the ecosystem services extracted from it. *Populus alba* (white poplar) outcompetes numerous indigenous tree and shrub species in mostly sunny areas, for example, forest edges and fields, and impedes with the normal progress of regular natural community. It is a particularly more competitive species since it can establish in a range of soil type; produce substantial seed crops, and resprouts effortlessly in reaction to harm. The ability of *P. alba* to reproduce asexually renders it a challenge when trying to get rid of it and restoring the area to its original state. Therefore, there is a need for more re-eradication activities during follow-up observation and monitoring of the sites. *Populus alba* (white poplar) can produce suckers from a small piece of root which may be left behind in the soil during eradication, and this makes it a very difficult species to deal with than the *A. decurrens* species.

The increase and decrease in the number of species with season was thought to have been due to the fact that during different seral stages of succession, species are seen disappearing and appearing during its cause. Lower species making way for higher species. The allelopathic effect caused by these invasive alien plants may also be the cause, as some of the native species can germinate but may not fully establish because of alteration of the soil structure.

Researchers’ observations in South Africa have steered to encompass that *Populus alba* and *Populus tremula* were distributed via the movement of vegetative parts in water (Henderson, 2007). Though the restoration is still possible with continued monitoring of the areas controlled for plant invaders, *Populus alba*, just like other invader species such as *Acacia decurrens*, grows in aggregated form and pushes other plant species out of their territories, therefore monopolizing the area.

Lantana camara can produce numerous seeds and can reproduce asexually by stem layering. It is often difficult to restore the lands previously infested by this species. It needs continued and close monitoring for successful recolonization to take place. The establishment of an invasive alien species is habitually responsible for an increase in predation and competition, habitat reduction, an assortment of illnesses in animals and hereditary changes in population (the indigenous species influenced by these invasive alien species may change their hereditary makeup in order for them to some way or another overcome the issues caused by the invasive alien plant species).

Lantana camara can grow together with the native plant species when it is still young, but it outcompetes the natives that grow in its vicinity when it reaches a certain stage. It is not that clear as yet if there could be any success with restoring the areas previously infested by *Lantana camara*, as it regenerated vigorously in the study site after its removal. Some other alien species were also seen in the site which were not there before clearing. In this instance, we can talk about secondary invasion or rather reinvasion by the same invasive species.

Invasive alien plants can rapidly replicate and spread over considerable areas in relatively short periods of time. Some of the invaders, especially Acacias, have the potential to alter environmental variables such as fire regimes, water use and soil nitrogen levels, and have persistent seed banks, and can out-compete native plants.

The site previously occupied by the *Acacia decurrens* population showed some signs that it could be restored to its original state before the infestation. This was evident because there were a lot of grass species observed in the area after clearing, even though there were a few invasive herb species such *Bidens pilosa.* There is a high chance that with continued monitoring, the whole area could be covered with native plant species after a few years. The rate of regeneration in this study was measured by looking at the number of individual plant species that were found occupying the site during different seasons of observations.

It is unquestionable that invasive alien species regulate an ecological influence on their host ecosystems, and that impacts suggest changes. Be that as it may, should such ecosystem changes be viewed as destructive? And if yes, to what degree? Shockingly, most investigations have neglected to give an answer.

Human beings play a large role in the dispersal of alien plant species as they travel the world. The advancement in the transportation industry has increased the rate of invasive alien species dispersal. This leads to intentional and accidental introduction of alien species into different parts of the world.

The establishment and dissemination of alien species is promoted by various cooperating factors (including the environment and species characteristics) and patterns in financial elements (for example, management interventions, fashions, financial conditions) assume a critical role in shaping the procedures of establishment and controlling how they change over time. An increase in invasive alien plants will probably be devastating to the biological diversity or exasperate the ecosystem services, and will also increase the impacts on natural environment and threats of vegetation disturbance in competition with indigenous plants.

From an economic point of view, ecosystem disturbances resulting from invasion may prompt losses in potential economic yield, that is, they may lead to lessened harvest generation, fisheries, forestry and animal farming. The benefits and costs stemming from invasive species are determined by a numerous social and natural factors.

Additionally, various invasive alien species have both advantages and costs which can prompt irreconcilable situations between various stakeholders.

Invasive alien species eradication is a disputable issue where economic, socio-cultural and biological factors regularly conflict in the field of environmental management. Likely, few would address endeavours by society to control, or even kill invasive alien species causing health concerns, financial loss, and harm to essential ecosystem services.

7.2 Conclusions

Invasive alien species control is a top preference when human wellbeing is in question, particularly if we reckon that invasive species' impact on human wellbeing is supposed to aggravate because of the expansion of chances of invasion favoured by introduction of new mechanisms, ecosystem destruction and atmospheric changes. Alien plant clearing operations are succeeding in reducing alien plant cover within riparian areas.

From Chapters 3, 4 and 5, it is evident that the recolonization by native plant species is possible. However, the rate of vegetation recovery may be negatively impacted upon by insufficient follow-up control and changes in invader-species composition. This investigation does not recommend conservation and land managers to forsake their endeavours to adapt to significant issues brought about by some established species, or government to quit attempting to keep threat species from being introduced into their nations.

The lack of economic evaluation of the costs and benefits of invasive alien species control presents a significant barrier to the uptake of effective invasive alien species prevention and management. Such data would give important support to investigate policy level actions. Stakeholders ought to prioritise whether invasive alien species cause advantages or mischief to human wellbeing, economies, social conventions, ecosystem services, and biological diversity. The battle on the species invasion is one that is very difficult to win. This is mainly because of traits that the invasive alien species possess such as rapid growth and reproduction, short maturity period and their allelopathic actions which change the soil structure.

The amount spent on the mitigation of the invasive alien plant species increases through the years, mainly because the spread of these invasive species keeps on increasing. It is unreasonable to endeavour to re-establish biological systems to some historical order. Researchers ought to acknowledge the reality of 'innovative environments' and fuse many invasive alien species into the mitigation plans, instead of embarking on endeavours to seek after the frequently unimaginable objective of removing them or diminishing their abundance (Davis et al., 2011).

Achieving the goal of restoration requires a lot of monitoring of the eradicated area as failure to do so may, and most often, result in secondary invasion. Recolonization by native plant species occurs at a slower rate due to the soil structure which is changed by invasive plant species after being introduced into a certain region.

Economic parameters as well as social standards and foundations are urgent for basic decision making. Societies and residents may choose to put an unending incentive on certain environmental goods for social or spiritual reasons, be it an uncommon animal variety, a sacred grove or an extraordinary landscape, and not expose it to a cost–effective investigation. Accomplishing reasonable conclusions needs an incorporated appraisal of ecological, social and financial system, and an open discussion among contributors (Ring et al., 2010). Comprehension of the social qualities and individual inspirations driving the public preferences for some invasive species should better get ready environmentalists and managers for their debates with different contributors (McNeely, 2005; Coates, 2006). If this information enables environmentalists and managers to understand that individuals can truly differ on when invasive alien species effect is viewed as valuable or destructive, at that point the discussion between the public, ecologists and managers will be unavoidably improved. It is therefore concluded that the native species recovery into the sites previously infested by the invasive alien plant species is attainable, provided factors (such as continued monitoring, re-eradication, exhaustion of the soil seed banks) that hinder the successful recolonization are attended to closely.

7.3 Recommendations

Recommendations of removal ought to be practical about the degree of societal investment required. Having requirements on time and finance, the management endeavours with objectives of co-existing are more possible than ones considering management (Davis, 2009).

There is a dire requirement for coordinated effort at national, provincial and global dimensions, whereby viable follow-up, early cautioning and the management is organized in demanding a strong political condition and the improvement of a long-term policy on IAS management in Africa (Mugisha, 2002). Enhanced awareness of prospective IAS dangers and likely methods of establishment may be an incentive for the generation of energetic management plans; the point of which would be to control an IAS before it can turn out to be widely distributed and costly or difficult to contain. The accomplishment of this motion would require improved monitoring and a reasonably clear chain of responsibility regarding new IAS attacks. The 'Working for

Water' programme shou+ld follow up on monitoring and management regarding the eradicated sites for successful recolonization by native plant species to occur, rather than expecting the ecosystem to recover on its own after removal of the invasive alien species.

Increasingly viable and continued follow-up clearings are expected to manage invasive species recovery which happens through both resprouting from cut stumps and roots and seedling establishment (for example *Solanum mauritianum*) (Witkowski and Garner, 2008). The absence of clearing achievement additionally indicates the requirement for more prominent adaptability in planning invasive alien plant management in the Savanna and Grassland riparian woodlands. Invasive alien plant species respond quickly to disturbances, for example, the major flood occasion in 2000 (Leroy, 2003), or a high precipitation year (Morris et al., 2008). Therefore, clearing endeavours ought to respond properly by being more frequent for few years after a noteworthy flood, or for just 1 or 2 years after a high precipitation occasion to keep invaders from re-establishing, producing and dispersing seeds downstream (Richardson et al., 2007;). It is additionally recommended that control activities should be first attempted upstream and dynamically move downstream to limit reinvasion of downstream regions. More prominent adaptability in dealing with the clearing tasks again indicates a versatile management approach.

References

Coates, P., 2006. American Perceptions of Immigrant and Invasive Species. University of California Press. Berkeley CA, United States of America.

Davis, M.A., 2009. Invasion Biology. Oxford University Press. New York, United States of America.

Davis, M.A., Chew, M.K., Hobbs, R.J., Lugo, A.E., Ewel, J.J., Vermeij, G.J., Brown J.H., Rosenzweig, M.L., Gardener M.R., Carroll, S.P., Thompson, k., Pickett, S.T., Stromberg, J.C., Del Tredici, P., Suding, K.N., Ehrenfeld, J.G., Grime, J.P., Mascaro, J. and Briggs, J.C., 2011. Don't judge species on their origins. *Nature* 474, 153–154.

Henderson, L., 2007. Invasive, naturalized and casual alien plants in southern Africa: a summary based on the Southern African Plant Invaders Atlas (SAPIA). *Bothalia* 37, 215–248.

Leroy, M., 2003. Changes in the native and alien plant species composition of the Sabie River, Kruger National Park, after the February 2000 flood. MSc Dissertation, University of the Witwatersrand. Johannesburg, South Africa.

McNeely, J.A., Mooney, H.A., Neville, L.E., Schei, P. and Waage, J.K., 2005. A global strategy on invasive alien species. Gland, Switzerland and Cambridge, UK, World Conservation Union (IUCN).

Morris, T.L., Witkowski, E.T.F. and Coetzee, J.A., 2008. Initial response of riparian plant community structure to clearing of invasive alien plants in Kruger National Park, South Africa. *South African Journal of Botany* 74, 485–494. doi:10.1016/j.sajb.2008.01.177.

Mugisha, A.R., 2002. *Evaluation of community-based conservation approaches: management of protected areas in Uganda.* Doctoral Thesis, University of Florida. Florida, United States of America.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P. and Hobbs, R.J., 2007. Riparian vegetation degradation, alien plant invasions and restoration prospects. *Diversity and Distributions* 13, 126–139.

Ring, I., Hansjürgens, B., Elmqvist, T., Wittmer, H. and Sukhdev, P., 2010. Challenges in framing the economics of ecosystems and biodiversity: the TEEB initiative. *Current Opinion in Environmental Sustainability 2*(1-2), 15–26.

Witkowski, E.T.F. and Garner, R.D., 2008. Seed production, seed bank dynamics, resprouting and long-term response to clearing of the alien invasive *Solanum mauritianum* in a temperate to subtropical riparian ecosystem. *South African Journal of Botany* 74, 476–484. doi:10.1016/j.sajb.2008.01.173.

Appendix

Published Article

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The rate of recolonization by native plant species after the Eradication of the invasive alien plant species (*Populus alba* L.) in the Limpopo Province, South Africa

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ABSTRACT

Biological invasion is one type of environmental change that can, at least to some degree, be effectively managed. This includes the identity, number and invasion status of invasive alien species, and additionally the drivers and pathways of invasive alien species introduction. Twenty four Permanent plots of 10 square meters each were constructed. The number of alien invasive species and native species present in the quadrats were recorded before the plots were cleared. The invasive alien plant species were removed mechanically. There was no significant difference between the numbers of *P. alba* recorded before the removal in 2016 and after its removal in 2018. The invasive alien plant species still came back in large numbers.

Key words : *Populusalba, Invasive alien plant species, Native plant species, Eradication, Recolonization*

Introduction

Biological invasion is one type of environmental change that can, at least to some degree, be effectively managed (Tu, 2009). Such management is basically reliant on adequate information and a comprehension of the source, size and nature of invasion (McGeoch *et al.*, 2010). This includes the identity, number and invasion status of invasive alien species, and additionally the drivers and pathways of invasive alien species introduction (Kolar and Lodge, 2001). Further consideration must also be made for the phenomenon of lag phases in invasion, whereby a species can be native in a novel territory for a significant measure of sites for invasion, biotic interaction and environmental conditions. Preventing invasion is by a wide margin the most efficient and cost effective management option, especially in protected zones (Tu, 2009). Discovering what drives introductions, the quantities of species that become invasive, and how this differs across taxonomic groups, gives a fundamental basis for the formulation of appropriate policy and management approaches.

White poplar was not considered a problematic species or a control priority in several other cases. It

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probably has less potential impact and receives lower priority for control in areas where it has not been widely planted and does not have the potential to hybridize with native aspens. In a survey answered by 35 Canadian botanists, most respondents indicated that white poplar was not a "problem species" and was invasive only locally. The survey was sent to botanists across Canada, but the regional distribution of respondents was not reported. White poplar was relatively rare in Farmington, western Maine, and spread of clones was easily tracked back to areas where white poplar was planted. Surveys of the flora in New London County, Connecticut, revealed that white poplar populations were uncommon and generally restricted to disturbed sites. Population sizes were stable (Hill, 1996).

Impacts from underground growth: The extensive white poplar root system has caused problems near houses or other urban developments. Several sources anecdotally report that white poplar roots can clog drains, sewers, and water channels (Elbert, 1961). In his manual of woody plants, Dirr (1998) indicates that white poplar "becomes a nuisance and liability after a time". Dirr (1998) suggests that homeowners "avoid this pest". A pamphlet produced by England's Forestry Commission reports that white poplar can remove soil moisture rapidly during dry, hot days. In low-rainfall areas such as London and Essex, white poplar has caused rapid drying and shrinkage of clay soils, which can upset dwelling foundations (Herbert, 1968).

Impacts to associated vegetation: Impacts on associated vegetation may change as white poplar stands expand and age. Through prolific root sprouting, white poplar can develop dense stands, which can crowd and shade native vegetation and reduce species diversity (Weber, 2003). As stands age, the breakage of brittle white poplar wood can damage nearby vegetation (Mehrhoff et al., 2003). In the central Transvaal area of South Africa, where white poplar is non-native, there is "marked correlation between the occurrence of naturalized and planted white poplar", but white poplar no longer occurs as isolated stands; instead, it occupies whole river reaches and has spread from the water's edge to far outside the riparian zone. White poplar has "out-competed" and suppressed existing vegetation in its formation of "absolutely pure stands" (Wells *et al.*, 1980).

The drivers of invasive species richness versus invasive species, that is, characterized here as those species that impact adversely on biodiversity may also vary. It is said that a relatively small percentage of invasive alien species are thought to be invasive (Richardson and Pysek, 2006), and Richardson et al. (2005) discovered to some degree distinctive indicators of quantities of invasive versus alien plant species richness across quarter degree grid cells (QDGCs) in South Africa (number of aliens was connected more to human factors and invasive species richness was better predicted by environmental factors).

Fig. 1. Map of the Limpopo Provincce showing the Nylvley Nature Reserve.

Methodology

The study was conducted in the Waterberg district municipality, it lies at 29.39′S and 28.45′E. It is situated 10 km SSE of the town Mookgopong, which is in the Limpopo Province, South Africa. Nylsvley Nature Reserve is a savanna ecosystem, situated in the upper part of the Nyl River, and include the part of the largest floodplain in the country. The nature reserve is known for its exceptional avifaunal diversity, with over 370 species recorded. Permanent plots of 10 square meters each were constructed using iron rods. The number of alien invasive species and native species present in the quadrats were recorded before they can be removed.The invasive alien plant species were removed mechanically. No chemicals were used, since they are concerns about their adverse environmental implications. The areas were monitored for the reestablishment of alien invasive species; if they occur, they were removed. Only the native plant species were left alone, if they happen to be seen establishing in the study site.

Description of species under study

Populusalba

Populus alba (white poplar) outcompetes numerous indigenous tree and shrub species in mostly sunny areas, for example, forest edges and fields, and interferes with the normal progress of regular natural community. It is a particularly solid contender since it can establish in a range of soil type; produce substantial seed crops, and resprouts effortlessly in response to damage. Dense stands of white poplar prevent other plant species from coexisting by reducing the amount of sunlight, nutrients, water and space availability. Natural cross breeds are accounted for between *P. alba*and *P.granndidentata* or *P. tremuloides* in USA (Dickman, 2001).

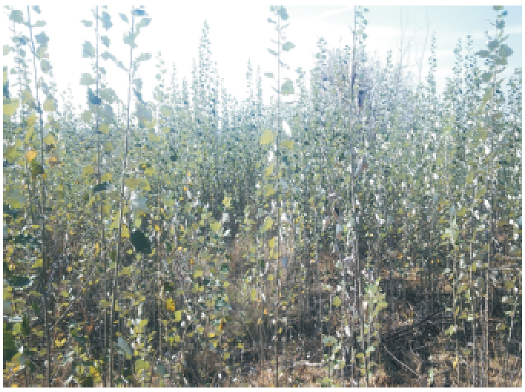

Fig. 2. A population of *Populus alba* in juvenile stage.

Establishment

It can be propagated by bare root, as it is very good in root suckering. In the soil is just a network of roots connecting each and every tree species of populous. This plant species normally grows in an aggregated form. It can also be propagated by their cutting and seeds.

Benefits

It has high quality wood known as "poplar" with a greenish colour. *Populus* is a lighter, more porous material. Its flexibility and close grain make it suitable for a number of applications, similar to those for willow. It was used by the Greeks to make shields. It is also renowned for durability similar to that of oak, but at a substantial reduction in weight. Poplar is widely used for the manufacture of paper, snowboards and in the bodies of electric guitars and drums (Aylott *et al*., 2008)

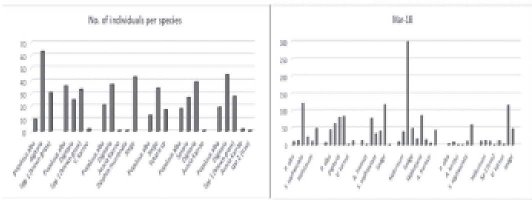

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Fig. 3. (a) Number of individuals per species encountered in the study area in September 2016 before uprooting the invasive alien plants (*Populus alba*). The grasses were seen dominating the study, and only a few tree species were encountered, such as, *Ziziphus mucronata and Vachelliakarroo* (b), shows the number of individuals on the last observation in March 2018, with herbs and grasses still dominating and *P. alba* still persistent.

Results

Discussion

There was no significant difference between the number of *Populus alba* recorded before the its removal in 2016 and after its removal. The plant species still came back in large numbers than before uprooting them. This was probably because there

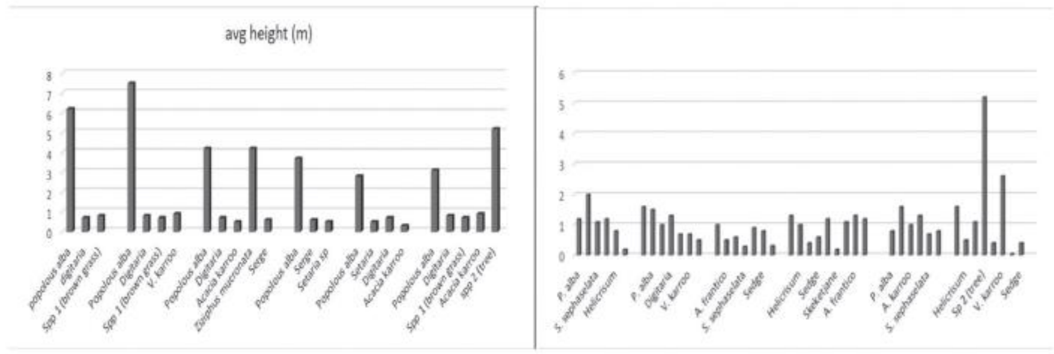

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Fig. 4. (a) Average height per species in the study site before removal of the invasive alien species (*P. alba*) in September 2016.
Populusalba was seen being dominant when it comes to height.(b),shows the average height on the last day of observation in March 2018. *P. alba* was still seen dominating some parts of the study site.

were numerous pieces of roots which were left behind during the uprooting of the plant species, therefore a lot of chances for suckers to emerge a few months later and grow vigorously in the study site.

Clonal growth and spread most commonly occur by root sprouting, but may also occur through fragmentation and layering. Fragmentation describes the process by which new clones can develop from twig or root pieces that become partially buried in sand or silt (Siebel, 1998 review by Dickmann, 2001). Observations in South Africa led researchers to conclude that white poplar and *Populus alba* × *Populus tremula* were dispersed through the movement of vegetative parts in water (Henderson, 2007).

White poplar root sprouts have been described as prolific, "vigorous", and "objectionable" (Elbert, 1961). Dense colonies or thickets from root sprouts can cover large areas. When white poplar seedlings were grown and evaluated as shelterbelt trees in the

Fig. 5. (a) A stand of *P. alba* population, (b) Uprooting of *P. alba* in process, (c) Uprooted trees, and (d) Seedlings of *P. alba* germinating three months after being removed.

Fig. 6. A root sucker of *Populus alba* which emerged from a piece of root left behind while uprooting the parent plant.

northern Great Plains, root suckers were frequently reported long distances from the parent tree (George, 1953). According to a review by Spies (Spies, 1978), vegetative sprouts can occur up to 160 feet (50 m) from the parent plant.

Despite continuous uprooting of the plant species, *P. alba* was persistent in the study site. This was possible because it is capable of vegetative reproduction through root suckering.

Conclusion

It is very challenging when working with *Populus alba*, since it is capable of root suckering. When you leave a little piece of root in the ground while eradicating, it can produce a sucker and grows vigorously, it can reach 2 m in a periods of just 3 months. Though the restoration is still possible with continued monitoring of the areas controlled for plant invaders. *Populus alba*, just like other invader species such as *Acacia decurrens*, grow in aggregated form and pushes other plant species out of their territories therefore monopolizing the area.

References

Dickmann, D.I. 2001. An overview of the genus Populus. In: Dickman, Donald I., Isebrands, J. G., Eckenwalder, James E., Richardson, Jim, eds. *Popular Culture in North America*. Ottawa, ON: National Research Council of Canada, Research Press. Ottawa, Canada.

Dirr, M.A. 1998. *Manual of woody landscape plants: Their identification, ornamental characteristics, culture, propagation and uses*. 5th ed. Champaign, IL: Stipes Publishing. Illinois, United States of America.

George, E.J. 1953. Tree and shrub species for the Northern Great Plains. Circular No. 912. Department of Agriculture. Washington DC, United States of America.

Henderson, L. 2007. Invasive, naturalized and casual alien plants in southern Africa: a summary based on the Southern African Plant Invaders Atlas (SAPIA). *Bothalia*. 37 (2) : 215-248.

Hill, S.R. 1996. The flora of Latimer Point and vicinity, New London County, Connecticut. *Rhodora*. 98 (894) : 180-216.

Elbert, L. 1961. Sixty trees from foreign lands. Agricultural Handbook No. 212. Washington, DC: U.S. Department of Agriculture. p 30.

Herbert, L. 1968. Know your broadleaves. Forestry Commission Booklet No. 20. London: Her Majesty's Stationery Office. p 142.

Kolar, C.S. and Lodge, D.M. 2001. Progress in invasion biology: predicting invaders. *Trends in Ecology and Evolution*. 16 : 199–204.

Mehrhoff, L.J., Silander, J.A. Jr., Leicht, S.A., Mosher, E.S. and Tabak, N.M. 2003. IPANE: Invasive Plant Atlas of New England, [Online]. Storrs, CT: University of Connecticut, Department of Ecology and Evolutionary Biology (Producer). Available: http://nbii-nin.ciesin.columbia.edu/ipane/ [2008, May 28].

McGeoch, M.A., Butchart, S.H.M., Spear, D., Marais, E., Kleynhans, E.J., Symes, A., Chanson, J. and Hoffmann, M. 2010. Global indicators of biological invasion: species numbers, biodiversity impact and policy responses. *Diversity and Distributions*. 16 : 95-108.

Richardson, D.M., Rouget, M., Ralston, S.J., Cowling, R.M., van Rensburg, B.J. and Thuiller, W. 2005. Species richness of alien plants in South Africa: environmental correlates and the relationship with indigenous plant species richness. *Ecoscience*. 12 (3) : 391–402.

Richardson, D.M. and Pyšek, P. 2006. Plant invasions: merging the concepts of species invasiveness and community invasibility. *Progress in Physical Geography*. 30 (3) : 409-431.

Siebel, H.N. and Bouwma, I.M. 1998. The occurrence of herbs and woody juveniles in a hardwood floodplain forest in relation to flooding and light. *Journal of Vegetation Science*. 9 (5) : 623-630.

Spies, T.A. 1978. The occurrence, morphology, and reproductive biology of natural hybrids of *Populus alba* in Southeastern Michigan. Ann Arbor, MI: University of Michigan, School of Natural Resources. 125 p.

Tu, M. 2009. Assessing and managing invasive species within protected areas. In: Ervin, J. (Ed.), *Protected Area Quick Guide Series*. The Nature Conservancy, Arlington.

Weber, E. 2003. *Invasive Plant Species of The World: A Reference Guide to Environmental Weeds*. Cambridge, MA: CABI Publishing. p 548.

Wells, M.J., Duggan, K. and Hendersen, L. 1980. Woody plant invaders of the central Transvaal. In: Neser, S.; Cairns, A. L. P., eds. *Proceedings, 3rd national weeds conference of South Africa; 1979 August; Pretoria, South Africa. Cape Town, South Africa*: A. A. Balkema: 11-23.