Assessing the invasiveness of *Acacia stricta* and *Acacia implexa*: is eradication an option?

by

Haylee Kaplan

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Supervisor: Dr John R. Wilson

Co-supervisors: Prof. David M. Richardson

Dr Jaco J. Le Roux

Faculty of Science

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DECLARATION

i. I hereby declare that the work presented in this thesis is my own original work, unless otherwise stated, and has not previously been submitted in its entirety or in part at any university for any other degree.

ii. Chapter 2 was initiated as an Honours thesis in 2009, and ~ 50 % of the data used in Chapter 2 were collected as part of the project by Herbert van Zyl. All other data collection and analyses are my own original work unless otherwise acknowledged.

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ABSTRACT

This thesis investigates the invasiveness and current status of two *Acacia* species recently identified as invaders in South Africa in order to determine the feasibility of their eradication. Australian acacias are among South Africa's worst invasive species and many have had widespread damaging impacts on native ecosystems. In addition, several *Acacia* species still exist as small isolated populations in the country and have been targeted for eradication in order to prevent potential widespread impacts. This work assesses *Acacia implexa* (Chapter 2) and *Acacia stricta* (Chapter 3) as potential eradication targets by quantifying the extent of their invasion in South Africa, assessing the risk they pose to the country and evaluating the feasibility of their eradication based on estimated costs of clearing. Results of formal risk assessments show that both *A. implexa* and *A. stricta* should be considered high risk species, and bioclimatic model predictions indicate that both species have large potential ranges in South Africa. Detailed population surveys found that *A. implexa* and *A. stricta* each occur at several distinct localities all in the Western Cape Province. *Acacia implexa* populations were found at three sites (Tokai, Wolseley and Stellenbosch) where they have densified by means of vegetative suckering allowing *A. implexa* to outcompete native vegetation. No evidence of large seed banks of *A. implexa* were found, however vigorous resprouting following damage makes the control of *A. implexa* difficult. *Acacia stricta* was found at nine localities all in the Knysna area of the Garden Route, where populations are spreading along disturbed roadsides in plantations. *Acacia stricta* produces large amounts of seeds and can accumulate large seed banks. Seed spread is most likely due to large-scale soil movement by road maintenance vehicles which can easily lead to the establishment of new populations. We therefore used a predictive risk mapping approach based on the association of *A. stricta* to roadsides and disturbed plantations to enable effective searching to detect all infestations of *A. stricta*. Based on the high risk of both species and the limited range sizes of the currently known populations, we recommend that *A. implexa* and *A. stricta* remain targets for eradication. Management strategies proposed for these species (Chapter 4) include clearing on an annual (in the case of *A. stricta*) or biannual (for *A. implexa*) basis to prevent seed production, and targeted awareness campaigns at a national scale to determine whether our current knowledge of the extents of *A. implexa* and *A. stricta* are accurate. This work has shown that detailed assessments of species at intermediate stages of invasion is an important initial step in an eradication attempt, and better understanding of species specific invasion characteristics can help to improve management and potentially increase the probability of success of eradication.

OPSOMMING

Hierdie tesis ondersoek die invasieve en die huidige status van twee *Acacia* spesies onlangs geïdentifiseer as indringers in Suid-Afrika ten einde die lewensvatbaarheid van hul uitwissing om te bepaal. Australiese akasias is onder Suid-Afrika se ergste indringerspesies en baie het wydverspreide skadelike impak op die inheemse ekosisteme. Verder het verskeie *Acacia* spesies bestaan nog steeds as 'n klein geïsoleerde bevolkings in die land en wat geteiken is vir uitwissing in om moontlike grootskaalse impakte te voorkom. Hierdie werk beoordeel *Acacia implexa* (Hoofstuk 2) en *Acacia stricta* (Hoofstuk 3) as 'n moontlike uitwissing teikens deur die kwantifisering van die omvang van hul inval in Suid-Afrika, die beoordeling van die risiko wat hulle inhou vir die land en die evaluering van die haalbaarheid van hul uitwissing op grond van beraamde koste van die wiele ry. Resultate van formele risikobepalings toon dat beide die *A. implexa* en *A. stricta* moet oorweeg word om 'n hoë risiko spesies, en bioclimatic model voorspellings dui daarop dat beide spesies het 'n groot potensiaal bereik in Suid-Afrika. Uitgebreide bevolkings opname gevind dat *A. implexa* en *A. stricta* elk by verskeie afsonderlike plekke in die Wes-Kaap voorkom. *Acacia implexa* is op drie plekke (Tokai, Wolseley en Stellenbosch) gevind, waar hulle deur middel van vegetatiewe suier densified en inheemse plantegroei oorwin het. Geen bewyse van groot nageslag banke van *A. implexa* is gevind, maar in kragtige resprouting volgende skade maak die beheer *A. implexa* moeilik is. Die *Acacia stricta* is op nege plekke in die Knysna-omgewing van die Tuinroete, waar die bevolkings verspreiding langs die versteurde paaie in plantasies. *Acacia stricta* produseer groot hoeveelhede saad en kan versamel groot saadbanke. Saad versprei is waarskynlik te danke aan grootskaalse grond beweging deur die instandhouding van paaie voertuie wat kan lei tot die vestiging van nuwe bevolkings. Ons het dus 'n voorspellende risiko kartering benadering wat gebaseer is op die vereniging van *A. stricta* aan paaie en versteurde plantasies in staat te stel om doeltreffend te soek alle besmettings van *A. stricta* op te spoor. Gegrond op die hoë risiko van beide spesies en die beperkte reeks groottes van die bevolking wat tans bekend is, beveel ons aan dat *A. implexa* en *A. stricta* bly teikens vir uitwissing. Bestuurstrategieë vir hierdie spesies (Hoofstuk 4) voorgestel word, sluit in die skoonmaak op 'n jaarlikse (in die geval van *A. stricta*) of die halfjaarlikse (vir *A. implexa*) basis van die saad produksie, en geteikende bewusmakingsveldtogte om te voorkom dat 'n nasionale skaal om te bepaal of ons huidige kennis van die omvang van *A. implexa* en *A. stricta* akkuraat is. Hierdie werk het getoon dat uitgebreide aanslae van spesies op intermediêre fases van die inval is 'n belangrike eerste stap in 'n poging van die uitwissing, en 'n beter begrip van spesies spesifieke inval eienskappe kan jou help om te verbeter en potensieel verhoog die waarskynlikheid van sukses van die uitroeiing nie.

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CHAPTER 1

General introduction

1. Introductions of invasive species

Invasions by non-native species have become an important driver of global change, in that they pose a significant threat to biodiversity, and alter ecological processes and ecosystem functioning (Chornesky and Randall, 2003; Pimentel et al., 2000). Human-mediated introductions of plants occur on a global scale largely due to trade in ornamental and commercially utilised species (e.g. for horticulture or forestry; Richardson and Rejmanek, 2011), as well as accidental seed transfer associated with increased human and cargo movements around the world (Nathan et al., 2008; Westbrooks, 2004; Wilson et al., 2009). Many species that are introduced to a new area have the potential to become invasive. Species traits such as fast growth, vigorous vegetative reproduction and high seed production often promote invasiveness. High propagule pressure (i.e. number of seeds and frequency of introduction) is also considered an important factor in determining whether a species will become invasive (Lockwood et al., 2005). Given sufficient time for adaptation and naturalisation (Caley et al., 2008) and large climatically suitable area (Rouget et al., 2004), many invasive species can become widespread and potentially dominate landscapes, at which point they are very difficult to control and manage effectively.

South Africa, like many other countries, is host to many invasive species, some of which have spread across large areas and caused significant ecological and economic damage (Le Maitre et al., 2011a; Le Maitre et al., 2002). Many invasive species are already widespread, but there is a large group of non-native species that have not yet spread far from their original points of introduction and occur at a few localities. Most resources allocated to the management of invasive species have in the past been directed at controlling widespread invaders. There is, however, a strong trend towards allocating more resources to reduce the invasion debt by preventing currently limited invaders from spreading, with the aim of reducing the overall cost of weed management in the long term (Wilson et al., 2011).

2. *Eradication feasibility*

Given the uncertainty of the invasive potential and risks posed by introduced species, the precautionary response is to eradicate non-native species that pose a high risk of invasiveness before they spread widely. Eradication involves the removal of all individuals and propagules of a species in an area to which reinvasion is unlikely, while simultaneously preventing any further spread (Myers et al., 2000). There are various treatments that can be applied when attempting eradication, of which the most widely used in South Africa is manual clearing approaches and applying prescribed herbicides. Persistent seed banks (Richardson & Kluge, 2008), vigorous vegetative reproduction and low detectability frequently complicate eradication efforts. Many attempts require several years of repeated clearing and follow up actions before populations are successfully eliminated (Regan et al., 2006).

Eradication is usually only considered feasible for species whose populations are still relatively small, as the costs of eradicating a species increase exponentially with the size of infestations until it is no longer a feasible management strategy (Fig. 1.1; Moore et al., 2011; Rejmánek and Pitcairn, 2002). Consequently, detecting an invasive species early and responding immediately with appropriate control measures significantly increases the likelihood of successful eradication. An "early detection and rapid response" approach to invasive species management is thus a highly cost effective and risk-averse management strategy (Byers et al., 2002). Whether or not to attempt eradication is a key decision that needs to be made soon after an invasive (or potentially invasive) species has been detected in a given area. Many eradication attempts (particularly for plants) are unsuccessful, and the reasons for this can be linked to insufficient resources and effort invested, large number and size of infestations, low detectability, and longevity (e.g. long-lived seed banks), which can often result in a failure to contain a species (Moore et al., 2011; Cacho et al., 2006; Panetta et al., 2011). There needs to be sufficient population-level information available in order to decide whether eradication of a species is feasible and prioritise species for eradication (Cunningham et al., 2004) and others for containment or biological control.

South Africa has a legal framework that aims to limit the introduction of harmful non-native species, and reduce the impacts caused by the many existing invasions within its borders. This has largely been done through the listing of invasive (and potentially invasive) species, placing restrictions on how such species can be used, and requiring landowners to control invasive populations. For example, the *Conservation of*

Agricultural Resources Act (1983) (CARA) and the *National Environmental Management: Biodiversity Act (2004)* (NEM:BA) categorise non-native plant species according to the ecological and economic threats they pose, and placement in these categories determines how a particular species is to be managed. Species with small invasive population sizes are categorised as 1a under the proposed NEM:BA regulations and require compulsory control. These species comprise a list of targets for eradication by the national Early Detection and Rapid Response programme (EDRR; Box 1). The role of the EDRR programme is to identify and assess new invasive species and respond promptly with an appropriate management plan.

Box. 1: Early Detection and Rapid Response in South Africa

The National Early Detection and Rapid Response (EDRR) Programme for Invasive Alien Plants was initiated in 2008 by the South African National Biodiversity Institute (SANBI) in accordance with draft regulations in the National Environmental Management: Biodiversity Act (NEM:BA) to monitor and report on invasive species in South Africa. The role of the EDRR programme is to co-ordinate surveillance, assessment, rapid response and monitoring of emerging invasive plant species in order to prevent negative impacts on ecosystems and biodiversity (Ivey et al., 2010). The key objectives of EDRR are:

- Early detection of emerging weeds
- Species identification and verification
- Risk assessment and response planning
- Rapid response

EDRR currently targets 18 plant species for eradication and additional species are under surveillance as potential eradication targets.

3. *Assessing species for eradication*

Effective assessments of the potential options for control of invasive species rely on accurate delimitation of the invasion (Panetta and Lawes, 2005) and an evaluation of the threat posed by a species to native ecosystems. Invasive species may exist as one isolated population or as several infestations that have arisen from separate introduction events or spread from neighbouring infestations. The full extent of the invasion needs to be known prior to attempting eradication so that the amount of resources required, the optimum control strategies and the feasibility of eradication can be determined (Lawes and Panetta, 2004). With limited funds and resources available for invasive species management, species can be prioritised based on the level of risk they pose (Wainger and King, 2001). This can be evaluated using a weed risk assessment tool. An example of this is the Australian Weed Risk Assessment protocol. This has been used routinely by the biosecurity community in Australia and New Zealand (Pheloung et al., 1999) and has also been shown to be applicable to other parts of the world (Andreu and Vilà, 2010; Dawson et al., 2009; Gordon et al., 2008).

Targeted species-specific management is critical for any attempted eradication programme. Management strategies need to be based on sufficient prior knowledge of a species' life history traits (Simberloff, 2003), such as the age at reproductive maturity (in order to eliminate further seed production), seed longevity (to determine the duration of the control phase), and potential spread pathways (to ensure proper monitoring and containment).

Management plans can be further evaluated and improved upon using observations and data collected during control activities, such as the effectiveness of control treatments and search protocols (Cacho, et al., 2006). Detectability of the species, particularly in the juvenile phase, is another important factor influencing the amount of effort required to search and locate all individuals. Low detectability will require greater search effort, i.e. greater number and level of expertise of surveyors and a longer time taken to complete the search (Garrard et al., 2008; Tyre et al., 2003). Many of these factors are quantifiable given sufficient data, and should be incorporated into the eradication plan to ensure that species are managed optimally and cost-effectively.

4. *Australian Acacia species in South Africa*

Australian *Acacia* species (hereafter "acacias"; see Richardson et al., 2011 for details of the group) have a long history of introductions globally for various horticultural and commercial purposes. Acacias are well known for weedy traits such as fast growth rates (Witkowski, 1991), prolific seed production and long-lived seed banks (Gibson et al., 2011; Richardson and Kluge, 2008a), which have led to numerous *Acacia* species becoming important invaders in many countries, including South Africa (Richardson & Rejmanek, 2011). For this reason, acacias have recently been the topic of a wide range of studies, highlighting important aspects of their introduction histories (Carruthers et al., 2011; Griffin et al., 2011; Le Roux et al., 2011), biogeography (Richardson et al., 2011b), invasiveness (Castro-Díez et al., 2011; Gallagher et al., 2011; Morris et al., 2011), and management (Moore et al., 2011; Wilson et al., 2011; van Wilgen et al. 2011).

Invasions by acacias can have many negative impacts on biodiversity and ecosystem functioning, such as causing radical changes to nutrient regimes (Stock et al., 1995), outcompeting native species (Holmes and Cowling, 1997), disrupting water flow and altering fire regimes, which have significant ecological and economic costs (Le Maitre et al., 2011b). Therefore there is an urgent need to find efficient management strategies for this group of species. Like other invasive species, preventing further acacia introductions by means of pre-border risk assessments, border security and regulation of international trade in species is important to prevent any further invasions and impacts (Wilson et al., 2011). Approaches for managing existing widespread *Acacia* invasions involve a range of control and containment strategies that take into account long-lived seed banks and the ability of many acacias to sucker and resprout. These include repeated mechanical and chemical control, biological control, and preventing seed spread by monitoring and managing dispersal pathways (van Wilgen et al., 2011a; Wilson et al., 2011). Eradication of acacias is only considered for species with small populations, especially considering the number of years required to eradicate a species with large, long-lived seed banks.

Large-scale introductions of acacias to South Africa for forestry, horticulture and dune stabilization began in the 19th century (Poynton, 2009; Le Roux et al., 2011) and have resulted in Australian acacias becoming some of the country's worst invaders. Fifteen *Acacia* species are listed as invasive in South Africa, several of which are very widespread and have damaging impacts on native ecosystems over large areas (e.g. *Acacia cyclops, A. mearnsii* and *A. saligna*; van Wilgen et al., 2001). Because of the large extent of these invasions, eradication of these species is not practical and they are managed as part of general alien plant clearing initiatives such as the Working for Water (van Wilgen et al., 2011a; 2011b). Biological control is considered for species with large range sizes, and to date biological control agents have been introduced to control ten of these *Acacia* species while agents are being considered for a further two species (*A. podalyriifolia* and *A. elata*; Impson et al., 2011; Wilson et al., 2011).

There is a large gap in distribution between these widespread species (to the left on the graph in Fig. 1.2) and the others. This is most likely because many of the widespread species were widely planted and have been highly utilized in the past (and several species, e.g. *A. melanoxylon* and particularly *A. mearnsii*, are still utilized for forestry purposes). Some *Acacia* species were introduced but not widely planted and at least eight of these species are known to still occur as small isolated populations around the country (Fig. 1.2).

There are currently four *Acacia* species in South Africa that are listed as “1a” (i.e. requiring compulsory control) under draft regulations of NEM:BA which have been targeted for eradication by the EDRR programme (see Addendum A for details). The first of these species to be evaluated for eradication was *A. paradoxa* which occurs as a single population in Table Mountain National Park. An initial detailed assessment of the population (Zenni et al., 2009) and subsequent decision analysis based on infestation size and effectiveness of control (Moore et al., 2011) indicated that eradication of *A. paradoxa* was a feasible option, and optimum management strategies were proposed for achieving this outcome at an estimated cost of 5.4 million ZAR over 20 years (ZAR = South African Rand; I US $ = approx. 7.98 ZAR). This approach highlights the importance of obtaining sufficient data on a species’ invasion in order to effectively plan for its management. As part of the objectives of the EDRR programme, similar research is needed for all potential eradication targets.

One of the other species listed as 1a, *A. adunca*, is only known from one small population in the Western Cape. There are also some species which are known to have naturalised, e.g. *A. fimbriata*, *A. ulicifolia*, *A. viscidula* and *A. retinoides*, as well as several species that have been planted in small numbers but are not known to have become naturalized (e.g. *A. floribunda*, *A. pendula*). Given their small population and range sizes, eradication is a desirable management approach for these species which would probably not take much effort. The remaining two species, *A. implexa* and *A. stricta*, which are intermediate in range size (Fig. 2), are the focus of this work. The key question that I address is: at what point (expressed as overall range size and distribution of populations) is eradication of an introduced Australian *Acacia* species still feasible?

5. *Study species*

Acacia implexa and *Acacia stricta* are two plant species that are invasive in South Africa which require evaluations of their current invasive status and potential risks to determine whether their eradication is feasible and should be attempted. Both species are currently proposed as category 1a under NEM:BA. Unlike most other species in the “1a” category, *A. stricta* and *A. implexa* do not occur at only a single site in the country, but as several isolated populations. It is therefore uncertain whether either species should indeed be classified as 1a (i.e. localised and eradicable) or reclassified to 1b (i.e. to be managed as part of a national management plan). Assigning the species to 1a without a proper assessment of their population sizes, spread rates and reproductive traits could result in failure to eradicate them and a waste of resources. On the other hand, not attempting eradication could allow a potentially high-risk, yet eradicable, species to invade further. The recommendations for management and legislation therefore rely on thorough surveys and investigation.

Acacia implexa (screw-pod wattle; Fig. 1.3) is a fast-growing tree up to 15 m tall that is native to eastern Australia (Maslin, 2001). It was introduced to South Africa in 1886 for forestry purposes (Shaunessy, 1980), but was not widely planted and is currently known to exist only in three small populations in the Western Cape: Tokai, Stellenbosch and Wolseley. *Acacia implexa* is highly prone to suckering and vigorous resprouting following damage (Maslin, 2001), allowing for the formation of dense stands which could make it a difficult plant to control.

Acacia stricta (hop wattle; Fig. 1.4) is a smaller, erect shrub or tree growing up to 10 m tall that is native to south-eastern Australia and Tasmania. It is distinguished by its narrow erect phyllodes with a prominent mid-vein and yellow inflorescences occurring at the axil of each phyllode (Maslin, 2001). It is unclear when this species was first introduced to South Africa, but since 2005 it has been recognised as a problem species in the Knysna area in the eastern part of the Western Cape Province. Small populations of *A. stricta* currently occur at several localities in the area. From initial observations it appears that *A. stricta* occurs mainly along roadsides on plantation land and can form dense stands. Given the ability to form persistent soil seed banks in soil and the proximity of the populations to roadsides, there is a possible high risk of further spread of this species into adjacent conservation areas if it is not properly controlled.

There are no records of either *A. implexa* or *A. stricta* becoming invasive in other parts of the world (Richardson & Rejmanek, 2011), and there have been no previous attempts to eradicate these species from South Africa (Wilson et al., 2011). Both species are under the management of multiple landowners and their control to date has mostly been as a part of general alien plant clearing. If eradication is to be attempted, both species will require targeted and co-ordinated management plans based on the detailed population assessments to ensure continuity in management operations among respective landowners.

6. *Research questions and aims*

The aims of this thesis are firstly to assess the current status of the invasions by *Acacia implexa* and *A. stricta* in South Africa, and secondly to determine whether their eradication is feasible and to provide suitable guidelines for the optimal management of these species. Additionally, in the case of *A. stricta*, we also aim to identify areas at risk of spread that should be prioritised for monitoring.

The following chapters are intended to answer the following questions regarding the status and eradication of *A. implexa* and *A. stricta* populations in South Africa:

6.1. What is the current extent of the populations of <u>A. implexa</u> and <u>A. stricta</u> in South Africa?

In order to estimate how the populations are spreading and the requirements for eradicating these species, I first need to know the extent of the populations. This involves mapping all plants to determine the size and distribution of populations, as well as measurements of population structure and seed banks to determine how rapidly and by what means the populations are increasing. At a broader scale, I need to be certain that I have identified all sites where each of the species has invaded. For this I will rely on creating public awareness, largely by means of information flyers, and local knowledge of field managers and workers. Any reports of new sightings will be followed up on to determine whether either of the species is likely to be more widespread than is currently known.

6.2. Do <u>A. implexa</u> and <u>A. stricta</u> pose significant threats to South Africa?

An important aspect of an EDRR approach to invasive species management is assessing the risk they pose to native ecosystems so that priorities can be assigned to the species that pose the greatest risk. Given the major impacts of other *Acacia* species in South Africa (Le Maitre et al., 2011), it is likely that *A. stricta* and *A. implexa* would have similar impacts if they become widespread. At a local scale, we have observed that both species have the ability to densify rapidly which poses a threat to the surrounding natural vegetation. The most commonly used approach for estimating the risk a species poses is to use a formal weed risk assessment that estimates a species' risk based on pre-determined criteria that have been found to be indicative of invasiveness in known invasive species. To predict the risk of *A. stricta* and *A. implexa* becoming problem weeds in South Africa I will use the Australian Weed Risk Assessment protocol. In addition, I will predict the potential for each species to spread across the country by modelling the climatic suitability of areas in South Africa for the survival and growth of the species using a bioclimatic niche model.

6.3. What are the best strategies for the long term management of these species?

Both species will require a management plan that is specific to the conditions of the invasion, reproductive abilities and spread of the populations. Using results obtained for both species, I will be able to recommend optimum strategies for clearing and monitoring populations based on seed production and viability, age at reproductive maturity, effectiveness of herbicide treatments, detectability and potential spread pathways. *Acacia implexa* and *A. stricta* are unlike most other emerging invasive species listed as category 1a under proposed NEM:BA legislation, in that they exist at more than one site in the country. Until now they have been categorised as 1a pending further assessment. Based on the work and conclusions of the following chapters, I will be able to determine whether it is feasible to attempt eradication of these species given their current extent and estimated costs of eradication, and thus recommend their listing as either 1a or 1b under NEM:BA.

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CHAPTER 2

Distribution and management of *Acacia implexa* (Fabaceae) in South Africa: is eradication an option?

H. Kaplan[a] , H. W. F. Van Zyl[a], J. J. Le Roux[a], D. M. Richardson[a], J. R. U. Wilson[a,b] *

[a] Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Matieland 7602, South Africa.

[b] South African National Biodiversity Institute, Kirstenbosch Research Centre, Claremont 7735, South Africa.

* Corresponding author: Tel. +27 21 808 3408, Fax. +27 21 808 2995, Email john.wilson2@gmail.com

Abstract

This study is the first detailed assessment of an invasion anywhere in the world by *Acacia implexa* (Benth.) (screw-pod wattle). It evaluates the threat of this species to native ecosystems in South Africa. The current survey found approximately 30 000 *A. implexa* individuals spread over about 600 ha in three geographically distinct populations all in the Western Cape, South Africa. Population structures indicate rapidly increasing populations at all sites, predominantly by vegetative suckering. Populations appear capable of rapidly densifying if given the opportunity, creating monocultures and even out-competing other invasive acacias. Although seed viability is high, there is relatively low recruitment from seed, likely as a result of high seed predation. While this appears to have limited population spread rates, seeds have begun to spread via roads and watercourses, in particular along the Eerste River near Stellenbosch and so there is a possibility of a dramatic increase in the extent of all three populations. Formal risk assessment and bioclimatic niche modelling indicate that this species has the potential to spread over large parts of South Africa, particularly in the Western Cape and along the eastern coast. We consider eradication a feasible management goal, but given the strong ability of *A. implexa* to resprout, proper control and follow-up would be essential to prevent re-establishment of dense stands and further spread. We support the proposed listing of the species as category 1a under the National Environmental Management: Biodiversity Act.

Keywords: *Early detection and rapid response; emerging invader; eradication; invasive alien plant*

1. Introduction

Australian *Acacia* species have a long history of introductions to South Africa for commercial, horticultural and cultural purposes (Kull and Rangan, 2008; Kull et al., 2011; Richardson et al., 2003; Stirton, 1978). Characteristics such as rapid growth rate (Witkowski, 1991) and prolific seed production (Milton, 1980) have led to some Australian *Acacia* species becoming the country's worst invaders. In addition, several introduced species of Australian *Acacia* remain as small populations within limited ranges, yet have the potential to become widespread invasive species (van Wilgen, et al., 2011; Zenni, et al., 2009).

Detection of an invasive species while the population is relatively small can significantly reduce the cost of its eradication, i.e. early detection and rapid response (Rejmánek and Pitcairn, 2002). With limited funds and resources available for invasive plant management, it becomes important to prioritise efforts based on the environmental and economic risks a species poses. An assessment of invasiveness and feasibility of eradication, based on a species' biology and population dynamics, will provide a good indication of the risk posed by a species and inform plans for the species' management (Zenni et al., 2009). While there have been no confirmed eradications of Australian *Acacia* populations to date, chiefly due to their persistent seed banks, there are several ongoing efforts and it is increasingly recognised as a desirable and achievable goal (Wilson et al., 2011).

Study species

Acacia implexa (Benth.) is a fast growing, erect leguminous tree, reaching about 15 m. The seed pods are linear, coiled and twisted (Henderson, 2001; Maslin, 2001), hence its common name screw-pod wattle. Phyllodes are curved and narrowly elliptic, with 3 - 7 prominent lateral veins (Fig. 1a). Inflorescences (flowering between December and March) are pale yellow to white and finely hairy (Fig. 1b; Maslin, 2001). *Acacia implexa* is highly prone to suckering and vigorous resprouting when damaged (Fig. 1c).

It is native to most of the eastern seaboard of Australia where it occurs in well-drained soil in woodlands and open forests (Maslin and McDonald, 2004). There are currently no records of *Acacia implexa* becoming naturalised or invading other parts of the world (Richardson and Rejmanek, 2011), although it has been introduced to many regions, particularly in Asia, where it is used commercially for fuel, pulp and tannins (Boxshall and Jenkyn, 2001; Griffin, et al., 2011). It was introduced to South Africa in 1886 to be used for tanning bark, but was found to be unsuited to the conditions of the plantation and thus did not become a highly utilized species in the country (Shaughnessy, 1980). There is however no mention of *A. implexa* in the section on acacias in Poynton's (2009) historical review of forestry plantings in South Africa.

Acacia implexa is often easily confused with *A. melanoxylon*, particularly at juvenile stages when differences in flower and seed pod morphologies cannot be detected. The similarity between these two closely related species led to the initial misidentification of *A. implexa* as *A. melanoxylon* in a recent survey of the riparian flora of the Eerste River (Meek et al., 2009). Following confirmation that putative populations of *A. implexa* in South Africa were the same as specimens of *A. implexa* from Australia based on ITS DNA sequence data (Le Roux unpublished data), *A. implexa* individuals assessed in this study were identified by visual inspection of distinguishing morphological features. *Acacia implexa* can be distinguished from *A. melanoxylon* by having longitudinally anastomosing minor nerves between main lateral veins on the phyllodes, while *A. melanoxylon* phyllodes have prominently rectangular nerve islands (Maslin, 2001). Seed morphology of the two species also differs; the aril on *A. implexa* seeds is white and tucked beneath the seed (Fig. 1d), whereas that of *A. melanoxylon* is pink to dark red and encircles the seed (Maslin, 2001).

In this study we assess the invasiveness and feasibility of eradication of *Acacia implexa*, an emerging invader listed as category 1a under the proposed draft regulations of the *National Environmental Management: Biodiversity Act (2004)* (NEM:BA). Under NEM:BA, category 1a species require compulsory control, effectively meaning that the goal is to eradicate the species from the country. Given the known invasiveness and impacts of various Australian wattles in South Africa (Richardson, et al., 2011), the aim of this initial assessment is to determine whether *A. implexa* has the potential of becoming a significant problem in the country and whether it should be targeted for eradication. To achieve this we 1) mapped the distribution of all currently known populations of *A. implexa* in South Africa, 2) assessed the risk of *A. implexa* becoming a widespread and problematic species in the country, and 3) evaluated the feasibility of eradicating this species. We also assess the current designation of *A. implexa* as a category 1a invader under NEM:BA and make recommendations for its legislation and long-term management.

2. Methods

2.1. Study sites

We used the Southern African Plant Invaders Atlas (SAPIA; Henderson, 2007) as well as knowledge of researchers working on *Acacia* introductions and invasions in South Africa to search for and locate all known populations of *A. implexa* in the country. We identified three populations of *A. implexa* in South Africa (all in the Western Cape) with population extents of 100–200 ha (Fig. 2.2; Table 2.1) — (1) at Tokai on the eastern slopes of Table Mountain, (2) at Wolseley near the Kluitjieskraal forestry station ~ 4 km west of the town and (3) in Stellenbosch where the main infestation is on Papegaaiberg though several plants are spread along the Eerste River, and there are a few individuals presumably planted for ornamental purposes in the J.S. Marais Park. The sites are currently under management but eradication of these populations has not been set as a management goal.

2.2. Population survey

As a basis for this study, we surveyed the three *A. implexa* populations between May 2009 and December 2010 to determine the current distribution of all known infestations in the country. Each site was systematically searched for plants by means of parallel walked transects, up to 20 m apart (depending on the density of vegetation; Fig. 2.3c). The location of each plant was recorded using a handheld Global Positioning System (Garmin GPSmap 60CSx). *Acacia implexa* is highly prone to suckering, which made it difficult to distinguish between individual plants and those with multiple stems emerging from a single root due to suckering. In such cases, each ramet (*sensu* Harper and White, 1974) was recorded as a separate individual. The search was abandoned when no plants were found for ~ 100 m in all directions, except while searching along riverbanks where the searching distance between plants was increased up to ~ 1 km.

Plant measurements and reproductive features were recorded for a haphazard subset of individuals from each population. We measured stem diameters at the base of each plant, and determined whether each individual was a resprout (above-ground resprouting stem), sucker (stem connected to below-ground suckering root), single plant (single stem emerging from root) or a seedling (newly germinated plant with stem diameter < 1 cm) by pulling it up to see whether it was growing from a single root or connected to a neighbouring stem. The presence of seed pods was also recorded and any biological control agents were collected and taken to Ms F. Impson of the ARC-Plant Protection Research Institute in Stellenbosch for identification.

2.3. Bioclimatic modelling

To estimate the potential distribution range of *A. implexa* in South Africa based on climate, we modelled the realised climatic niche of *A. implexa* using MaxEnt 3.3.2 (Phillips et al., 2006) and projected it onto the current South African climate. The bioclimatic variables used to create the model were obtained from the WORLDCLIM dataset (www.worldclim.org, Hijmans et al., 2005) at 5-minute resolution. We selected the eight least inter-correlated bioclimatic variables: mean annual temperature, mean diurnal range in temperature, isothermality, temperature seasonality, mean annual precipitation, precipitation of the driest month, precipitation seasonality, and precipitation of the warmest quarter (Loiselle et al., 2008). Presence data for *A. implexa* were obtained from the Australian Virtual Herbarium (chah.gov.au/avh/; accessed 14 July 2010) for records from its native range, and from our own distribution data for the invasive range in South Africa. We used Köppen-Geiger climate zones (Peel et al., 2007) from eastern Australia that contained occurrence records of *A. implexa* to draw the background for the model (see discussion in Thompson et al., 2011 and Webber et al., 2011). We fitted the model using all data, with duplicate records automatically removed from the analysis if more than one record existed per 5-min grid cell. We used a 10-fold cross-validation to estimate error around the average model fit and the average test area under curve (AUC) for model verification.

2.4. Risk assessment of potential invasiveness

The potential invasiveness of *A. implexa* in South Africa was assessed using Pheloung et al.'s (1999) Australian Weed Risk Assessment protocol. This assessment protocol was developed to assess the risk of species introduced into Australia and New Zealand, but has also been shown to produce highly accurate results across a broad geographic range (Andreu and Vilà, 2010; Gordon, et al., 2008; 2010). A species' invasiveness is evaluated based on 49 questions pertaining to the biogeographical, biological and ecological characteristics and invasive traits of a species (Pheloung, et al., 1999). We applied the guidelines for answering the questions for areas of the world outside Australia (Gordon et al., 2010).

2.5. Seed viability and soil seed bank measurements

We tested the viability of *A. implexa* seeds collected from seed pods on mature plants on Papegaaiberg using a standard tetrazolium test (Peters, 2005). Only seeds with no visible damage were tested. We evaluated 100 seeds (50 seeds x 2 replicates) which were scarified using boiling water and stained using a 1% 2, 3, 5-triphenyl tetrazolium chloride solution (pH 6.7) for 72 hr. Seed coats were removed and seeds with evenly stained embryos were identified as viable (Peters, 2005). To estimate the size of the seed bank, five 0.5 x 0.5 m soil samples dug to a depth of 15 cm were taken from directly under adult *A. implexa* plants on Papegaaiberg that had evidence of recent seeding. Soil was dried and sieved through a graduated sieve stack.

2.6. Post-fire survey of Acacia implexa recruitment

The southern and eastern slopes of Papegaaiberg burnt in wildfires between January and March 2011. Vegetation cover for most of the burnt area was greatly reduced, if not entirely removed. *Acacia implexa* was one of the first woody species to show signs of recovery after approximately 8 weeks. To determine the dominant method of regeneration by *A. implexa* following fire damage (resprouting vs. seedling recruitment) we surveyed an area of approximately 2 ha where *A. implexa* was known to have occurred prior to the fire. Surveys of the area were done in March 2011 (1 month after fire) and November 2011 (9 months after fire). We identified and counted all new *A. implexa* recruits within the survey area, and determined whether they were resprouters or seedlings by uprooting them to see whether they were connected to an underground sucker root (resprouting) or had germinated from seed. Where seedlings were found, three soil samples (0.25 m^2) were taken to estimate the size of the remaining seed bank.

2.7. *Management history*

The *A. implexa* populations assessed in this study are each the responsibility of a separate authority (Table 1) and thus have been managed differently at each site. To determine what control methods have been used and which is most effective for killing *A. implexa*, we gathered reports from land managers at each of the three sites on the types of treatments that had previously been applied to *A. implexa*, as well as estimates of costs and success of these treatments. We also estimated the amount of effort that would be required to search and clear the *A. implexa* populations, using standard Working for Water person day estimations per hectare for adult sprouting trees in a riparian zone.

3. **Results**

3.1. *Current distribution and population dynamics*

We recorded a total of 28 172 *Acacia implexa* plants across all three sites. In Wolseley, the total number of plants recorded was 3556 spread over an area of ~ 200 ha (Fig. 2.3a). Tokai had a total of 1639 individuals over ~ 192 ha (Fig. 2.3b). In Stellenbosch we recorded 22 978 plants, 99 % of which occurred within an area of ~ 208 ha on Papegaaiberg, with the densest stands found along the Plankenbrug River. There were also isolated plants found along the Eerste River, >5 km from the main infestation, and in the nearby J. S. Marais Park (Fig. 2.3c).

The largest *A. implexa* plants were found in Tokai, with stem diameters up to 86 cm (mean = 4.88 ± 7.49 SD). All three populations have a high proportion of smaller plants (Fig. 2.4), suggesting a high rate of population expansion at all three sites. Seedlings, which represented ~ 33 % of the sampled plants at Wolseley, were found in one area of <1 ha which had been cleared and treated five times prior to our survey. Only ~ 3 % of the 760 plants sampled in Stellenbosch were seedlings, and were found alongside the largest individuals in the municipal graveyard on the southern slopes of Papegaaiberg. Seedlings at Tokai were distributed throughout the infestation. Vegetative growth by means of suckering and resprouting was the dominant method of reproduction among plants at all three sites (Table 2.2).

The presence of a gall forming midge, *Dasineura dielsi* Rübsaamen (Diptera: Cecidomyiidae), a biological control agent released against *A. cyclops* was noted on several *A. implexa* plants in Tokai and Stellenbosch. We also observed seed damage caused by another biological control agent, the seed-feeding weevil *Melanterius acaciae* Lea (Coleoptera: Curculionidae), which were present on green seed pods of several plants in Tokai (Fig. 2.1f&g*). Melanterius acaciae* was introduced to control *A. melanoxylon* (Impson and Moran, 2003).

3.2. Detectability

During this study we received several reported locations of previously unrecorded *A. implexa* individuals or populations, three in the City of Cape Town and two in Stellenbosch. Only one of the five localities had *A. implexa* present (several individuals in the J. S. Marais Park in Stellenbosch) which was reported by members of the municipality already involved in *A. implexa* management. No other populations have been confirmed, despite the extensive research work and surveys conducted on the group in South Africa, particularly by the biological control research team for Australian invaders which has been active for over thirty years.

3.3. Bioclimatic suitability

The model provided a good fit of *A. implexa* when projected back onto its native distribution in Australia (AUC = 0.940 (SD ± 0.004); Fig. 2.6a). The projection onto the South African climate (Fig. 2.6b) predicts areas of suitable climatic conditions for *A. implexa* across a considerable proportion of the country, particularly along the southern and eastern coasts. The bioclimatic variables that contributed most to the model results were precipitation of the driest month and mean annual precipitation which had relative contributions of 81.4% and 11.9% respectively.

3.4. Risk of invasiveness in South Africa

Thirty-nine out of the 49 questions in the weed risk assessment were answered based on the available literature and data collected during our study (Table 2.3), meeting the requirements of the assessment (Pheloung et al., 1999). *Acacia implexa* scored a total of 16 points (14 points for biogeography, 1 point for undesirable traits and 1 point for biology/ecology). *Acacia implexa* would therefore fail a pre-border assessment as it scores higher than the threshold value of 6 that indicates species as being potentially invasive (Pheloung et al., 1999). In addition, a risk assessment of Australian *Acacia* species based on life history traits, human usage and native climate associations (Castro-Diez et al, 2011) predicted *A. implexa* to have a 26.5 % (9.4 – 55.5, 95 % CI) probability of being invasive, and *A. implexa* was grouped together with the most invasive *Acacia* species in a risk assessment based on native distribution patterns (Hui et al., 2011).

3.5. Seed banks and seed viability

Seeds were absent in all soil samples taken from Papegaaiberg despite the presence of empty seed pods in the leaf litter. Of the seeds collected from seed pods on mature plants, 59% (52 – 66, 95% CI) were viable.

3.6. Post-fire regeneration

In our initial survey of part of the burnt area on Papegaaiberg in March 2011, we observed 519 resprouting *A. implexa* individuals and found no evidence of seed germination within the 2 ha survey area. Where dead *A. implexa* stumps could be identified (by the presence of resprouts on the stem), we observed suckering resprouts extending up to ~4 m from the adult plant. The re-survey in November 2011 found 995 seedlings and 164 suckering stems. Seedlings occurred in several dense patches, presumably beneath former adult trees. No additional seeds were found in soil sampled from beneath seedling patches, suggesting a depletion of seed banks following fire.

3.7. Management costs

Cost estimates for initial clearing of *A. implexa* populations are shown in Table 2.4. Person-day estimations were made based on the Working for Water guidelines for calculating the effort required for clearing infestations. Herbicide treatments differed among sites, and chemical clearing costs were estimated based on previous clearing costs by the relevant land managers at each site. Tokai was reported to have the least resprouting following herbicide treatment. The estimated cost of eradicating *A. implexa* at its currently known extent is ~ 795 000 ZAR.

4. Discussion

4.1. Population dynamics and spread

Despite the fact that seed production by *A. implexa* is high (Pieterse, 1998), vegetative reproduction by means of suckering appears to be the dominant form of population expansion. While *A. implexa* is known to form persistent seed banks that require a heat stimulus for germination to occur (Richardson and Kluge, 2008), the size and distribution of the seed banks are suspected to be relatively small and limited to beneath the canopies of large trees.

The small proportion and limited distribution of seedlings within populations suggests low seed germination or high seed/seedling mortality. Typical viability of *A. implexa* seed in its native range is between 65% and 75% (Melbourne Indigenous Seedbank, 1996) yet seeds appear to have slightly lower viability on Papegaaiberg. However, seed mortality is likely to be the largest source of seed depletion in the invasive populations. Pieterse (1998) reported significant *A. implexa* seed mortality (14-100 % in March and 92-100 % in April) on Papegaaiberg, as a result of seed predation by a native alydid *Nariscus cinctiventris* (Heteroptera: Alydidae) and similar observations were apparent in this study. This high seed mortality is attributed to the long periods in which seeds remain on the tree after dehiscence of pods, allowing more opportunity for predation by insects (Pieterse, 1998). Seed damage by the seed-feeding weevil *M. acacia,* which has caused significant seed damage to populations of genetically similar *A. melanoxylon* (Impson et al., 2011), may also account for the lack of *A. implexa* seed banks, however this has not been quantified.

Predation of seeds by ants and rodents (Holmes, 1990) is a likely cause of seed depletion on the ground, further preventing the addition of seeds to the soil seed bank. Therefore, the observed densities of *A. implexa* populations are likely the result of suckering.

According to the earliest available records of *A. implexa* it appears that the initial introduction of the species was at the arboretum at Tokai. Maps of the Paddock arboretum (which is no longer in use) indicate plantings and nursery transplants of *A. implexa* as early as 1886 (Anonymous, 1886). There are no records of introductions to Papegaaiberg or Wolseley, however both areas have historically been used for forestry so it is likely that plants were intentionally grown as experimental plantings.

The only survey of *A. implexa* we found on Papegaaiberg was from 1989. It showed a small population on the southern slopes with individuals up to 18 m tall (Fig. 2.5; Landman and Nel, 1989). We found no records of *A. implexa* at Wolseley before 2005 when clearing efforts began. Populations at Tokai and Papegaaiberg, where we had access to previous distribution records, have shown significant expansion from the original point of introduction. In particular, the population on Papegaaiberg has increased at least 5-fold since it was first surveyed in 1989. At Tokai plants have spread up to 1 km in all directions from the site of the original plantings in 1886. The presence of elaiosomes on *A. implexa* seeds (Fig. 2.1d) indicates an adaptation to dispersal by ants (Gibson et al., 2011; Gorb and Gorb, 2003; Hughes and Westoby, 1990). This may account for some uphill dispersal of seeds (Hughes et al., 1994), particularly on Papegaaiberg, although this remains to be confirmed. All three populations occur along or close to rivers and roads, which will facilitate long-distance dispersal (Gelbard and Belnap, 2003; Johansson et al., 1996). Indeed, many plants were found up to 5 km from Papegaaiberg along the Eerste River (Meek et al., 2009) where the invasion on Papegaaiberg is the likely source.

Population expansion is however currently partially limited by surrounding developed areas in Tokai and Stellenbosch, restricting spread in most directions. In Wolseley there is no significant limitation to the spread of the population, and there is considerable area available for potential population expansion at this site.

Dispersal over long distances is unlikely as *A. implexa* is not currently cultivated or traded in South Africa, nor does it have any natural adaptations for long-distance dispersal (other than along rivers and roads).

However, Tokai and, to a lesser extent, Papegaaiberg are both public areas which imposes a risk of accidental movement of seeds by humans. With many areas in the country predicted to be suitable for *A. implexa* growth, seed transfer could lead to the establishment of further populations and increase the risk of this species becoming a widespread invader.

Despite its long residence time in the country and its potential to become widespread and invasive, the distribution of *A. implexa* still remains limited compared to similar invasive Australian acacias in South Africa. This can in part be attributed to the low initial propagule pressure (compared with other invasive acacias e.g. *A. cyclops* and *A. mearnsii* which were introduced in large numbers across the country); and the fact that *A. implexa* was not used for commercial forestry or agroforestry in South Africa, and so was not widely distributed and planted across the country. The advantage of this for management is that the populations of *A. implexa* remain small enough to be controlled and potentially eradicated before the opportunity for large scale range expansion arises.

4.2. Management

Clearing of *A. implexa* has been done at Wolseley by Working for Water teams from the Department of Water Affairs since June 2005 by means of cut-stump treatment of all plants annually for the first three years followed by an additional two treatments involving foliar herbicide spraying. Populations at Tokai and Papegaaiberg have been cleared along with other invasive species as part of routine clearing operations by SANParks and the Stellenbosch municipality respectively. Herbicide usage varies among sites (Table 2.4) as *A. implexa* does not have official herbicide treatments assigned to it by the Department of Agriculture, and herbicide selection is therefore based on those used for similar acacia invasions in the area.

Observations suggest that herbicide treatments at Tokai are the most effective for killing plants as the incidence of resprouting is low following herbicide application compared to the other sites. Based on this anecdotal evidence, we recommend that juvenile plants be treated with Garlon 3% (480 g/L) as a foliar spray, and adult plants with Lumberjack 3% (360 g/L) on frilled stems or cut stumps. Resprouting can be reduced by cutting stems close to the ground (Witkowski and Garner, 2008) and applying herbicide immediately after cutting.

The ability of *A. implexa* to produce suckers and to resprout vigorously makes mechanical clearing of plants difficult and it will likely require several years of follow up treatments to remove all existing plants. While we did not find any literature indicating the minimum age at reproduction for *A. implexa*, for the purpose of management we can assume a conservative estimate of 2 years, the same as that of closely related and morphologically similar *A. melanoxylon* (Maslin and McDonald, 2004). We therefore recommend biannual follow-up clearing after the initial removal of all plants to ensure that no new recruits are able to reach reproductive age and set seed. Considering the lack of evidence for large persistent seed banks, follow up clearing at all sites should require less intensive search and removal effort and should therefore have significantly reduced costs each year.

At a global level, eradication of small invasive populations has become a well established management goal, with several Australian acacias being targets for eradication in various parts of the world (Wilson et al., 2011). In South Africa, attempts to eradicate localised invaders with high invasive potential (e.g. *Acacia paradoxa*) are ongoing, and species targeted for eradication are categorised under proposed national legislation and strategic management practices (van Wilgen et al., 2011). The invasiveness and limited distribution of *A. implexa* place it in the category for eradication, the success of which will largely be determined by the effectiveness of management (Moore et al., 2011).

There is still some uncertainty as to whether any further populations of *A. implexa* exist in the country (particularly given the taxonomic confusion with *A. melanoxylon*). The three known populations of *A. implexa* are included in SAPIA and there are ongoing efforts to locate any further populations, mainly through the distribution of information leaflets around the Western Cape to raise public awareness and collect reports of additional unknown populations. However, the discovery of several further populations in the country is unlikely to affect the feasibility of eradication. *Acacia implexa* seems to have a relatively low rate of spread, and, if properly treated, plants can be controlled. However the feasibility of eradication will become less likely if available resources are overwhelmed by the discovery of many more populations in the next few years.

Deciding the point at which eradication should be abandoned in favour of other control methods is a matter of on-going research (Panetta, 2009). However a recent study on *Acacia paradoxa*, another Australian *Acacia* species with a currently restricted distribution in the Cape Floristic Region, suggested that for this species eradication was the most economical management choice for infestations up to ~ 800 ha if the population is correctly delimited (Moore et al., 2011), although this was highly dependent on management efficacy. Given *A. implexa* is predicted to have a smaller seed-bank, and arguably lower spread rate, the major limitation remaining is to provide a control method that will prevent resprouting and suckering, and to ensure continuity in management. This continuity and co-ordination will be the responsibility of the Early Detection and Rapid Response Programme for Invasive Alien Plants of the Working for Water programme based at the South African National Biodiversity Institute (van Wilgen et al., 2011).

4.3. Conclusions

Although *Acacia implexa* does not yet occupy a large geographic range in South Africa, its biological attributes and the climatic suitability of many areas in the country, particularly the Western Cape, make it a high risk species with the potential to replicate the major impacts of other Australian *Acacia* species in South Africa (Le Maitre et al., 2011b). Given the current size of infestations and absence of large seed banks, costs of eradicating *A. implexa* are still within feasible limits. Eradication of species with limited ranges, such as *A. implexa*, conforms to the strategy proposed by van Wilgen et al. (2011) for managing Australian acacias in South Africa, and so *A. implexa* should remain a target for eradication in South Africa (i.e. a category 1a species under proposed NEMBA regulations).

Successful eradication of *A. implexa* will depend largely on sufficient resources to manually clear all plants, effectiveness of herbicide treatment to prevent regrowth, and co-ordination and continuity of management operations between the three areas in the Western Cape where populations currently occur. As such it is a suitable target for the national Early Detection and Rapid Response Programme for Invasive Alien Plants.

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CHAPTER 3

Integrating risk mapping and stratified surveys for assessing the potential of eradicating an invasive species

H. Kaplan[a] , A. van Niekerk[b], J. J. Le Roux[a], D. M. Richardson[a], J. R. U. Wilson[a,c] *

[a] Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Matieland 7602, South Africa.

[b] Centre for Geographical Analysis, Department of Geography and Environmental Studies, Stellenbosch University, Matieland 7602, South Africa.

[c] South African National Biodiversity Institute, Kirstenbosch Research Centre, Claremont 7735, South Africa.

*Corresponding author: Tel. +27 21 808 3408, Fax. +27 21 808 2995, Email john.wilson2@gmail.com

Abstract

Successful eradication of invasive plants depends on being able to detect all populations, and this detection can be achieved cost-effectively through targeted surveillance in high risk areas. We outline a predictive risk mapping approach to delimiting invasions to determine eradication feasibility and inform management using the only known invasion of *Acacia stricta* in South Africa. Based on field surveys and observations, *A. stricta* is a high-risk invader likely to cause similar problems to other Australian acacias in South Africa, but is considered to be a suitable target for eradication. The species has invaded approximately 110 ha at nine sites where it is currently spreading along disturbed roadsides in plantations. Local dispersal of *A. stricta* is largely via seed transport by road maintenance vehicles and its establishment appears to be restricted to disturbed areas. It has not yet invaded natural undisturbed vegetation. We use a correlative approach based on associations of the currently known distribution of *A. stricta* to predict areas with several land and road usage correlates to determine spread pathways and predict areas with high risk of invasion in order to improve and prioritise surveillance methods for this species. Our predictions indicated that, in climatically suitable areas in the country, surveillance should be targeted particularly in forestry plantations where road-side searching should be conducted in highly disturbed areas. This approach enables targeted searches and awareness at multiple scales, and reduces the overall costs of searching and monitoring. We further use estimates of seed banks, reproductive output and regrowth to recommend suitable management strategies for potential eradication of *A. stricta*. To date, the detection of populations has been a result of good local knowledge of invasions and active involvement of local land managers is an important component of *A. stricta* management and eradication success. Based on its current extent and high risk we consider *A. stricta* a suitable target for eradication. Initial surveying and clearing of *A. stricta* sites required a total of 38 field days at a total cost of ~ 45 000 ZAR. Eradication of the current *A. stricta* invasion will rely on co-ordinated management and active involvement of all stakeholders, and is estimated to cost ~ 1 million ZAR over the next 25 years. While risk maps may help in improving surveillance and reducing monitoring costs of invasive species, training locals in detecting *A. stricta* could prove more effective than outsider surveys.

Keywords: *Early detection, eradication, invasive plant, risk mapping, surveillance.*

1. Introduction

Invasive species management can be highly effective if invasions are detected early and comprehensive control measures are implemented rapidly enough to prevent widespread impacts accruing (Simberloff, 2003). In this context, eradication is often a desirable goal (Myers, et al., 2000). However considering the often high costs of eradication programmes, it is important to determine the feasibility of eradicating a species based on the extent of invasion, detectability and risk of further spread.

Eradication of plant species is usually only considered feasible for species with small range sizes (Rejmánek and Pitcairn, 2002). However, the success of eradication is not only a function of range size, but is also dependent on the ability to find all propagules. As a species' invasive range increases, there is a greater need to accurately delimit the extent of the invasion (i.e. number and size of populations) to determine eradication feasibility (Moore, et al., 2011; Panetta and Lawes, 2005). The costs and effort required to search for a species over a large potential range are often prohibitively high. Furthermore, there is a high probability of missing infestations using random searching (Cacho, et al., 2006). Invasion delimitation therefore requires systematic search protocols that enable rapid delimitation of all infestations and sufficient surveillance to detect new infestations that result from spread. Consequently, identifying areas where search efforts should be focussed, based on probability of invasion success in those areas, can reduce the cost of delimiting the extent of an invasion.

Habitat suitability predictions have been used to identify vulnerable areas for invasions and predict spread pathways of invasive species in order to improve search and management strategies (Butcher and Kelly, 2011; Giorgis et al., 2011; Peltzer et al., 2008). If eradication is to be attempted, early detection of infestations is essential to limit seed set and densification. However, the probability of detecting new infestations before reproductive maturity is often low (Kery and Gregg, 2003). In addition, species with long-lived seed banks may be present at a site but remain undetected until germination occurs (Cacho et al., 2007). Having a better idea of where to conduct intensive searches for a species could reduce the overall search area and minimise the risk of missing infestations. Highlighting areas with high suitability or risk of invasion by a species will thus improve the efficiency of searching and enable early detection of infestations before they are able to densify and spread.

Study system

Australian *Acacia* species have a long history of introductions and widespread plantings around the world, and as such several species in this group have become important invaders in many countries (Richardson, et al., 2011; Richardson and Rejmanek, 2011). *Acacia* species are well known for invasive traits such as high seed production, long-lived seed banks and short juvenile phases (Gibson, et al., 2011; Richardson and Kluge, 2008), and can also have profound effects on ecosystem functioning (Le Maitre, et al., 2011). These features have allowed many species to become competitively dominant and to transform natural habitats. Management strategies for these species are largely dependent on the extent and stage of the invasion, which determine the optimum method of control, i.e. complete eradication for species with small invasive ranges, or biological control and maintenance of current infestations for widespread species (van Wilgen et al., 2011; Wilson et al., 2011). In South Africa there are several widespread invasive *Acacia* species that have had large-scale damaging impacts on local ecosystem services and biodiversity (Le Maitre et al., 2011). More than a third of Working for Water's budget was spent on controlling *Acacia* invasions between 1995 – 2008 (van Wilgen et al., in press). Besides these widespread invaders, there are a few *Acacia* species that have not become widespread yet (most likely because they were not highly utilized and thus not planted widely, or because they have a short residence time in the country) and still exist as isolated populations. Of these species, four are currently targeted for eradication by South Africa's Early Detection and Rapid Response Programme for Invasive Species (EDRR). Three of these species (*A. paradoxa*, *A. adunca*, and *A. implexa*) are known to occur at only a few sites and are thus considered feasible candidates for eradication (e.g. Zenni, et al., 2009). The fourth species, *A. stricta*, requires further investigation to determine whether eradication should be attempted, in particular given it is found across a wider distribution than the other three species.

Acacia stricta (Andrews) Willd. is a small tree native to south-eastern Australia, growing to approximately 8 m tall with narrow, erect phyllodes with prominent mid-veins and small, yellow inflorescences growing close to the stem (Maslin, 2001). *Acacia stricta* is not known to be invasive elsewhere in the world (Richardson and Rejmánek, 2011), although it is recorded as naturalised in New Zealand. Unlike most *Acacia* species that have been introduced to South Africa, there are no records of introduction or planting of *A. stricta* to indicate how long it has been present in the country. Recent records of *A. stricta* in the Southern African Plant Invaders Atlas (SAPIA; Henderson, 1998) indicate several infestations in the Knysna area of South Africa, and one record of the species from 1981 reports several plants in Stellenbosch. Since 2004 it has been reported as a problem invader in the Knysna and Wilderness sections of the Garden Route National Park.

Acacia stricta is currently listed as a category 1a species under the proposed regulations of the *National Environmental Management: Biodiversity Act (2004)* (NEM:BA) which specifies that eradication should be attempted for this species. However, given that it occurs at several sites, it is unclear whether eradication is feasible. In order to assess eradication feasibility, there is a need for a search method to capture all populations and determine the total extent of the invasion by *A. stricta*. Here we propose a framework for delimiting the extent of invasive species in order to assess the feasibility of eradication (Fig 3.1). Using *A. stricta* as a case study, we identify the necessary steps that enable rapid and cost-effective detection of all infestations.

Understanding the dispersal pathways and vectors of a species is an essential step in the development of a surveillance protocol (Pysek and Richardson, 2010). Similar to the initial stages of many other invaders, *A. stricta* is currently found mostly along highly disturbed roadsides. Roads have been shown to be major conduits for the spread of invasive species due to high levels of disturbance that promotes colonisation (Gelbard and Belnap, 2003; Harrison et al., 2002; Spooner et al., 2004) and greater dispersal opportunities for seeds when road maintenance vehicles move soil (Ferguson et al., 2003). In this study we incorporate proximity to roads and similar habitat associations in a “seek-and destroy” approach (Fox et al., 2009) to locate *A. stricta* populations. We also consider the value of passive surveillance (i.e. reports from local land owners; Cacho et al., 2010) in locating and monitoring *A. stricta* infestations.

The disjunct distribution of *A. stricta* and consequent uncertainty of the extent of its distribution, together with good knowledge of *Acacia* invasions in general, make this a suitable system to explore processes in determining whether eradication should be attempted. Using a risk mapping approach, that includes bioclimatic and habitat suitability modelling, we apply predictions of potential range and spread pathways to target searches and awareness to areas of high risk of invasion at national, regional and local scales in order to locate all invasive populations. In addition, we identify reproductive traits (e.g. seed production, seed bank size) and dispersal mechanisms (i.e. vectors and pathways of seed spread) that could inform management planning and resource allocation.

2. Methods

2.1. Assessment of potential risk

To assess the potential invasiveness of *A. stricta* to South Africa, we used the Australian Weed Risk Assessment (AWRA) protocol developed by Pheloung et al. (1999) along with the guidelines for applying the assessment in areas outside Australia (Gordon, et al., 2010). This assessment was designed to be used pre-border; however it can also be applied to species already present within a country's borders and it has been shown to be applicable to many areas around the world (Andreu and Vilà, 2010; Gordon, et al., 2008), including South Africa (Zenni et al., 2009).

We used the distribution of *A. stricta* in eastern Australia to develop a climatic model, using MAXENT 3.3.2 (Phillips, et al., 2006), that could predict its potential distribution in South Africa. Presence data was compiled from records of *A. stricta* from the Australian Virtual Herbarium (chah.gov.au/avh/; accessed 14 July 2010). We selected the eight least inter-correlated bioclimatic variables from the WORLDCLIM dataset (www.worldclim.org, Hijmans, et al., 2005): mean annual temperature, mean diurnal range in temperature, isothermality, temperature seasonality, mean annual precipitation, precipitation of the driest month, precipitation seasonality, and precipitation of the warmest quarter (Loiselle, et al., 2008). The background for the model was drawn from eastern Australia where *A. stricta* naturally occurs. The model was trained using all presence data. Duplicate records within each 5-minute grid cell were deleted. Model error based on the predicted suitability, was estimated using a 10-fold cross-validation. The ability of the model to correctly predict actual occurrences was assessed using the average test area under curve (AUC).

2.2. Population survey

In an attempt to determine the total extent of the invasion by *A. stricta*, we began a thorough survey in 2010 as a basis for future surveying of *A. stricta* as part of its ongoing management. This included a preliminary survey in March and a larger scale survey in August and September. The preliminary survey consisted of interviews with local plantation and conservation managers with good local knowledge of plant invasions to gather reports of known populations, vehicular surveying to follow up on these reports and a survey of an invaded site to gather baseline data. A 1 km stretch of road at the invaded site was searched by means of walked transects ~ 10 m apart parallel to the roadside to locate and record all *A. stricta* individuals.

Following the reports from local plantation and conservation managers in the Knysna area, nine localities where *A. stricta* was thought to be invasive were identified and used as a starting point for determining the survey area. The localities occurred within a total area of approximately 1900 km^2, in the Knysna and Wilderness areas of the Garden Route National Park (Fig. 3.2). The survey area was made up predominantly of plantations (mainly *Pinus* and *Eucalyptus* species), natural forest, and farmland. Several other *Acacia* species, e.g. *A. melanoxylon*, *A. cyclops* and *A. mearnsii*, are common invaders in this area, and have caused considerable habitat transformation and degradation (Vromans et al., 2010). *Acacia stricta* represents another potential threat to native vegetation in the area.

The preliminary search confirmed reports that *A. stricta* predominantly invaded along disturbed roadsides in plantations (no plants were found further than 8 m from the roadside at the surveyed site). The main survey in August-September 2010 was based on observations from the preliminary survey and involved three levels of searching: 1) distribution of information flyers to and interviews with local land-managers to gather reports of known populations of *A. stricta*, 2) vehicle surveys of all roads in the affected areas, and 3) walked searches at all sites where plants were reported or discovered.

2.2.1. Information flyers

In order to get further reports of *A. stricta* infestations, information flyers were distributed by hand to local land managers. Managers were then asked for the locations of infestations of this species if they had encountered any. The flyers were targeted at groups who routinely work in the field and were therefore likely to recognise *A. stricta* infestations. These included conservation organisations (SANParks and CapeNature) and plantation managers at forestry companies (MTO and PG Bison), as well as a local tree nursery. The flyers contain a detailed description of *A. stricta* including its distinguishing features (see Appendix A). Reports are directed to EDRR to be followed up on to determine if they are accurate and whether they are indeed new infestations or have been previously recorded.

2.2.2. Vehicle surveys

Roads in the affected area were searched during the flowering period of *A. stricta* (August – September) when plants were most visible. A total of ~ 523 unique km of road were searched within the study area (approximately 20 % of the total road matrix) during vehicle surveying. Searching was done at an average speed of ~ 20 km/h with one person driving and one observer. The aims were to follow up on and locate reported populations of *A. stricta*, as well as to search as many roads as were accessible throughout the survey area for other infestations. The study area has an extensive road network, and with no prior knowledge of where to search and insufficient time to survey every kilometre of road, vehicle surveys were directed based on field observations and accessibility. Plantation areas and disturbed roadsides, particularly in areas surrounding known populations were thus the focus of the search (approximately 60 % of searching was done in plantations). However, roads in natural habitats, farmland and urban areas were also searched, making up approximately 25 %, 10 % and 5 % of the total searched area respectively.

2.2.3. Site surveys

Every invaded site identified during the vehicle survey was searched on foot. Transects ~ 10 m apart parallel to the road enabled a search for plants up to 25 m from the roadside on both sides of the road. The location of each plant found was noted, and plant height, stem diameter, the presence of reproductive features (i.e. flowers or seedpods) and distance to the road edge (to nearest 0.5 m) were recorded. Plants were then either pulled up by the roots or cut at the base and sprayed with a glyphosate herbicide. The search was discontinued at a site when no plants were found for at least 250 m along the road in either direction. In total it took 38 field days for three people to survey and remove all discovered populations of *A. stricta*.

2.3. Reproductive output, seed bank size, and seed viability

Size at reproduction was estimated from the complete data set collected during the flowering period in 2010 using a generalised linear model with binomial errors (with presence of reproductive structures as the response variable). To estimate how reproductive output scales with plant size, we measured plant height and number of flower buds present on plants at an infestation of ~ 70 individuals. Flower bud counts were used as a proxy for the maximum seed production per plant.

To get a preliminary estimate of seed bank size, three 0.5 x 0.5 m soil samples dug to a depth of ~10 cm were taken from beneath single large plants (5-6 m tall) and seeds counted. As road grading (resurfacing and digging of drainage ditches on dirt roads) is thought to be the primary dispersal agent of *A. stricta* seeds, we also sampled soil that had accumulated on the blade of a road grader immediately after it had dug a drainage ditch into a roadside patch of *A. stricta*. This was to determine if seeds were able to be transported along roads during road grading. To provide an estimate of how far off the road seeds were deposited, soil-cores (8 cm diameter x 10 cm depth) were taken along transects that intersected a newly graded plantation road. The road had no large plants but a high number of seedlings on the road which indicates that seeds had probably been deposited during road maintenance. A total of 11 transects spaced 10 m apart were positioned perpendicular to the road and extending 6 m either side of the road. Core samples were dug at 2 m intervals along each transect. Samples were sieved through a graduated sieve stack and seeds counted.

Seeds collected from soil samples were tested for viability using a standard tetrazolium test (Peters, 2005). A sample of 200 seeds (50 x 4 replicates) was first scarified using sulphuric acid and then stained using a 1% 3, 5-triphenyl tetrazolium chloride solution (pH 6.7) for 72 hr. Seed coats were removed and viable seeds (indicated by even staining) counted.

2.4. Regrowth from seed bank

The study area was resurveyed in September 2011 both to remove seedlings that had germinated and to determine whether the survey and clearing of *A. stricta* in 2010 was effective in finding infestations and reducing population numbers. Local land-owners kindly provided workers to conduct the work, which had the advantage of both engaging with locals and increasing the number of people searching for plants. The same destructive sampling method was used to survey all previously recorded sites, and incidence of resprouting was recorded. Vehicle surveys of surrounding areas were also done to determine if any new infestations had emerged since or been missed in the previous survey. All roads surveyed in 2010 were resurveyed as well as an additional 42 km of road in other parts of the study area. As in the previous year, there was insufficient time to survey every road.

2.5. Risk mapping of Acacia stricta

In an attempt to direct search efforts for *A. stricta* surveillance, we created a habitat suitability map to highlight areas at risk to invasion by *A. stricta*. We based the model on presence data collected during surveys in 2010 and a randomly generated set of 1000 absence data from within the study area. As all populations occurred in plantations, a second set of 1000 absence points restricted to plantation areas was created in order to refine the predictions to plantations only. Presence and absence data were sampled using grid and random sampling at three spatial resolutions (Table 3.1) to account for different levels of spatial autocorrelation.'

To identify factors that could influence the spread of *A. stricta* we selected six abiotic and anthropogenic variables as possible predictors of *A. stricta* occurrence (Table 3.2). Variables were extracted at each sampled presence or absence point from various land cover and topographic map layers in a GIS. Anthropogenic variables included land use (as we observed *A. stricta* populations to occur on plantation land), post-fire veld age (germination of *Acacia* seeds is known to be stimulated by fire for many species; Richardson and Kluge, 2008), distance to roads [roads can act as conduits for invasions (Gelbard and Belnap, 2003; Mortensen et al., 2009), and Spooner et al. (2004) found that disturbance from road maintenance increased recruitment of *Acacia* species], and compartment age (as the disturbance and vehicle movement during plantation activity may aid in the dispersal and recruitment of *A. stricta*).

To test whether these variables influenced the presence or absence of *A. stricta*, classification trees were drawn using recursive partitioning (package *rpart*) in R 2.11.0 (R Development Core Team 2010) for each of the three subsets of data. Trees were pruned to minimise the cross-validated prediction error estimates.

Misclassification errors were calculated for each tree using a test dataset of 256 presence and absence points sampled at a 100 m resolution from the study area. The prediction accuracy was estimated based on the AUC value of each model. The best tree was projected back onto the study area in ArcGIS 10.0 to produce a probability map of *A. stricta* occurrence and highlight areas suitable for future spread.

3. Results

3.1. Potential risk of Acacia stricta in South Africa

Based on data and observations gathered during this study and available literature, *Acacia stricta* would fail a pre-border risk assessment (Appendix B; overall score was 18, where >6 indicates potentially invasive) and should be considered a high risk species in South Africa.

The bioclimatic model provided a suitable fit of *A. stricta* distribution in its native range (AUC = 0.971 ± 0.004 SD). Projection of the model onto the South African climate (Fig. 3.2a) predicted high climatic suitability for ~15 % of the country. The bioclimatic variables that contributed most to the model were precipitation of the driest month and annual mean temperature, which had relative contributions of 49.2 % and 18.7 % respectively. Using these bioclimatic suitability predictions and the observed association of *A. stricta* with plantation areas, we highlighted all plantations within climatically suitable areas in the country to which search efforts should be expanded (Fig. 3.2b).

3.2. Population distribution

The survey of the study area in 2010 found 19 843 *A. stricta* plants at eight localities, with a total invaded area of ~ 110 ha (estimated using minimum convex polygons; Fig. 3.2c). All eight localities had been reported to us by local plantation and conservation managers, i.e. we found no additional populations on driving surveys. All populations occurred on forestry plantations with no spread as yet into adjacent fynbos or native forest. The majority (99 %) of plants recorded at sites occurred within 20 m from the roadside (Fig. 3.3). Populations in the south east of the study area had the highest number of plants and had begun spreading along a national highway (N2) where high density patches of seedlings were found close to the road.

An exhaustive survey of the area in Stellenbosch where *A. stricta* was reportedly found in 1981 as part of another study of *A. implexa* invasions (Chapter 2) found no *A. stricta* plants. Ongoing control in this area will be able to detect plants in future.

3.4. Reproductive output, seed banks and seed viability

The minimum height at which plants were found to reproduce was ~ 30 cm, while 67 % of plants 1 – 2 m showed signs of reproductive maturity (Fig. 3.4a). Reproductive output (estimated from flower bud counts) increased exponentially with plant height (R^2 = 0.788, F = 125.1, p <0.0001; Fig. 3.4b). The largest tree measured in the infestation (3.8 m) had 12 146 flower buds.

The seed bank size was estimated at ~ 1000 seeds/ m^2 per 4 - 5 m tall adult plant (an average of 251 seeds ± 2.1 SD per 0.25 m^2 soil sample). The soil collected from the road grader also contained two seeds, showing that *A. stricta* seeds are, as expected, transported during road grading. Soil cores sampled from across-road transects showed that seeds had been deposited up to 6 m from the road, but that 79 % of seeds were accumulated along road edges (i.e. along the regularly maintained roadside drainage ditches and ridges). Only 6 % (2-11, 95% CI) of the seeds sampled from the seed bank were viable.

3.5. Effectiveness of management

The re-survey of the study area in 2011 found ~ 15 126 plants (i.e. a 24 % reduction from 2010) with a total invaded area of approximately 92 ha. However, one new locality (making a total of nine) was found after following up on a report from a plantation field worker. The site (~ 4500 plants) was the furthest east of the other sites and had not been part of the route covered in the previous vehicle surveys. This highlights the benefit of active involvement in the project for locating new populations. Most of the plants found at sites were new seedlings of < 50 cm (Fig. 3.5), with the incidence of resprouting low at all sites (56 plants in total). Field observations showed that resprouting only occurred if the initial cut was made above the lowest branch; proper cutting and herbicide stump application appears to be highly effective. At most sites plants had not spread more than ~ 100 m from previously recorded locations, except at the largest site at Kruisfontein where several new infestations had emerged up to 1 km from the nearest source infestation. These new infestations were small patches of juvenile plants, likely a result of recent seed spread from a nearby infestation. We did not observe any spread of infestations away from roadsides since the previous survey in 2010.

3.6. Risk map for Acacia stricta

Land use was the most significant predictor of *A. stricta* occurrence, with all populations occurring on plantation land. Models refined to plantations only were therefore used for this analysis. Classification trees for models 1 and 2 are shown in Fig. 3.6 (AUC = 0.792 & 0.784 respectively). Model 3 (which included only 9 presence points) did not identify any discriminating variables to predict *A. stricta* occurrence. The misclassification error of model 1 was 32.8 % and 25.4 % for model 2. Based on the lower misclassification error and the simpler rules defined by model 2, we selected this model as most suitable for predicting *A. stricta* occurrence. The model predicts *A. stricta* occurrence based on distance to roads and age of plantation compartments. The resulting risk map (Fig. 3.2d) predicts ~ 579 km of road within the study area are at higher risk of invasion by *A. stricta* and to which search efforts should be focussed. This amounts to 83 % lower search effort from the total of ~ 3425 km of roads in the whole study area, and only overlaps ~ 30 % with the actual surveyed roads. The model also correctly predicted the location of the new population found in the 2011 survey as a high risk area.

4. Discussion

4.1. Delimiting invasions for eradication assessment

Deciding whether to attempt eradication of a species or opt for containment is an important management question and a matter of ongoing research. Uncertainty of invasion extent when attempting eradication can lead to failure if the invasion is poorly delimited and resources are insufficient to remove additional populations (Moore et al., 2011). Accurate delimitation is therefore a key component of assessing a species for eradication. We have demonstrated a predictive approach to locate invasive populations for rapid delimitation of a species for which there is uncertainty of the total extent (Fig. 3.1). Using the assessment of *A. stricta* as an example, we identified spread pathways (i.e. disturbed roadsides and plantations) that enabled predictions of where to search at national, regional and local scales to locate all infestations and determine eradication feasibility.

This approach can be applied more generally to similar invasions (i.e. potential eradication target species whose extents are poorly defined). Incorporating knowledge of dispersal pathways for a species into a risk mapping approach to enable early detection of populations and rapid invasion delimitation is a highly cost-effective method of assessing species for eradication. Additionally, a predictive approach to management of invasive species can help prioritise the allocation of resources depending on the specific risks associated with an invasion and the predicted targets for species awareness campaigns. A predictive, cost-effective approach to invasive species eradication can thus improve the likelihood of success.

Several factors promote the success of eradication attempts, including early detection, habitat specificity, low reproductive rates, effective monitoring, and sufficient education and awareness of a species (Myers et al., 1998). Many previous attempts at eradication have failed due to lack of consistent control over time and premature discontinuation (Gardener et al., 2010), reinvasions (Abdelkrim et al., 2007), and lack of public and landowner co-operation and involvement (Genovesi and Bertolino, 2001; Kazmierczak, Jr and Smith, 1996; Oppel et al., 2010). Despite the high risk of spread of *A. stricta*, at present it occurs as small localised populations confined to roadsides. Given that the invasion by *A. stricta* has been detected at a relatively early stage, and we have so far provided strategies for effective surveillance and awareness, we consider *A. stricta* to be a viable candidate for eradication at its present extent if immediate action is taken to control infestations and reduce spread. Although there is still some uncertainty of the full extent of the *A. stricta* invasion at this early stage in its management, efficient searching and improved awareness of this species aims to resolve this.

The discovery of *A. stricta* populations to date has been a result of good local knowledge of invasive species by plantation managers and conservation organisations. By creating awareness of *A. stricta*, by distributing information flyers, we hope that if new infestations are detected they will be reported. However, given the low usage of some of the roads (i.e. driven only during certain months of the year), additional surveys are required to detect infestations early and prevent (or at least limit) further seed set. Risk mapping predictions enable early detection of infestations through targeted road-side surveys within the affected region. Interestingly, however, the only new infestation detected was the result of neither the flyers nor the road-side surveys, but from someone actively involved in the eradication work.

At a national scale, searching for *A. stricta* should be done within areas with high climatic suitability for the species (Fig. 3.2a). As the forestry industry is the most likely pathway of introduction of this species and given the current association of *A. stricta* with plantations in its known invasive range, targeted searches and awareness campaigns should be focussed particularly in these areas (Fig. 3.2b). If new invasions are found elsewhere in the country, a similar approach to searching at regional and local scales should be applied to detect all populations and plants within the affected area. The feasibility of eradication of *A. stricta* will need to be re-evaluated if additional populations are found elsewhere in South Africa. Although given the distinctive erect growth habit and flower position that easily distinguish adult plants from other acacias, it seems unlikely that this species has gone unnoticed in other parts of the country.

The current extent of *A. stricta* is still within feasible limits of eradication, given its relatively limited distribution and confinement to highly disturbed roadsides in plantations, making infestations relatively easy to detect and manage. Based on the current population size and seed viability of 6% we estimate a total seed bank size of ~ 500 000 viable seeds. Assuming a 4 % annual seedling germination rate as indicated by survey data we estimate that it will take up to 25 years to accomplish eradication if current seed production is successfully limited. However, resources will likely be insufficient to control the species across a much larger range, at which point containment would be considered as an alternative management option.

4.2. Management of Acacia stricta

Although the reason for introduction of *A. stricta* is unknown, we speculate that it was introduced in plantation nurseries in the early 20th century. The current distribution of *A. stricta* in the Knysna area is most likely the result of spread from one or two initial introductions. Its small seeds and large persistent seed banks make it easy for the seeds to be spread in contaminated soil. The close proximity of infestations to roadsides increases the likelihood of its spread by vehicles. However as most populations are found along low usage roads (i.e. low traffic and less frequent maintenance) and are associated with high disturbance areas, the current distribution of *A. stricta* currently remains relatively localised due to the restricted opportunities for seed spread. The spread of *A. stricta* seeds along the heavily utilized and frequently maintained N2 national highway is of most concern, especially given the large climatically suitable areas along the entire route of the highway (i.e. south and east coasts of South Africa) to which *A. stricta* could spread.

Invasive species management across the study area is not co-ordinated as most land managers operate essentially independently. Previous clearing of *A. stricta* has been as part of general alien clearing by the MTO Forestry company, usually every few years or after clear felling at a plantation compartment. As such the management of *A. stricta* has been sporadic and inconsistent across sites. If eradication of this species is to be attempted a focused co-ordinated management plan needs to be implemented that provides effective strategies for finding and removing all infestations of *A. stricta*. Following discussions with stakeholders, it was agreed that a collaborative effort involving all relevant land managers and co-ordinated by EDRR would be the best way to manage *A. stricta*. Ensuring stakeholder buy-in and collaborating with people on the ground that have good local knowledge of invasive species is a reliable way of finding new populations in the future. A long-term management plan that involves annual targeted vehicle searches and removal of plants at all sites was agreed upon by all parties (Addendum B).

Clearing of existing *A. stricta* populations is a relatively straightforward task. Most plants up to two years old are easily pulled up by roots, and glyphosate herbicide has been found to be effective in preventing resprouting of larger plants when applied correctly. Since plants reach reproductive maturity at less than a meter in height and *A. stricta* is able to grow up to a meter in one year, follow up clearing should be done on an annual basis to prevent plants adding to the seed bank. With the current distribution (in 2011) of approximately 92 ha and an estimated cost of clearing of 400 ZAR per ha (based on MTO alien clearing rates), the estimated cost of removing all plants at existing sites is ~ 36 800 ZAR per year. Total cost for clearing of *A. stricta* is therefore estimated to be ~ 920 000 ZAR over the 25 year duration of the eradication programme.

Although preventing new reproduction appears achievable, it will be more problematic to prevent the movement of material in the soil seed banks. While the land-owners are willing to provide human resources and access to sites, it is too expensive to establish quarantine sites for heavy-equipment working in invaded areas. Instead, increased monitoring should be incorporated into the management plans. In this way, new infestations that may have arisen due to seed spread by vehicles can be detected soon after and removed.

At a fine scale, the seeds of *A. stricta* appear to be mostly underneath the canopy. At a slightly broader scale, the movement of *A. stricta* is most likely a result of soil seed bank spread by road maintenance vehicles such as road graders and plantation harvesting vehicles or equipment. Seed spread along roadsides at a particular site is suspected to occur regularly, as road grading is done up to four times per year (depending on the usage of the road). Seeds are thus able to be transported several hundred meters each time allowing for the potential establishment and densification of new infestations at a site. We also suspect that spread of plants away from roadsides occurs following disturbance and soil movement off the roadside during planting, harvesting and clear-felling of plantation compartments. Long-distance dispersal of plants is probably fairly stochastic and may occur as a result of shared vehicles and harvesting equipment between plantations.

The effectiveness of vehicle surveying is largely influenced by the detectability of infestations*. Acacia stricta* is easily confused with *A. melanoxylon* at juvenile stages (due to secondary lateral vein noted on young *A. stricta* phyllodes), and with *A. cyclops* when mature. Given its distinctive placing of flowers at the stem apex, surveying during the flowering period of *A. stricta* improves the likelihood of detecting plants along a road and limits confusion with other species. However the probability of detecting juveniles from a vehicle is low considering the search effort is less intensive than surveying on foot. The predictive mapping of *A. stricta* provides a means of prioritizing both vehicular and potentially foot surveys by minimizing the total area that requires searching and increasing the probability of detecting an *A. stricta* infestation before it is able to densify. Therefore the overall costs and effort of surveying are significantly reduced (by 83 % or ~12 000 ZAR per year) and clearing costs would potentially be less if infestations are detected at low density and before any significant seed banks are able to form. The predictions of vulnerable areas also highlighted the influence of anthropogenic disturbance (road usage and maintenance, plantation activity) on *A. stricta* occurrence, which is more easily monitored and controlled than abiotic factors and can be incorporated into a management plan.

One major concern in the approach taken here is that these predictions of risk are largely a function of the stage of invasion (Peterson, 2005). *Acacia stricta* is in a relatively early stage of invasion and has thus not reached distributional equilibrium. As such, predicting habitat suitability for *A. stricta* based on its current distribution might under-predict the total area where surveillance is required (Jimenez-Valverde et al., 2011). Restricting surveillance to plantations and limiting monitoring in natural areas might not detect spread to areas we are most interested in conserving. The risk maps of *A. stricta* should therefore not be considered as a predictor of potential long-term population expansion, but rather as a tool to guide the immediate systematic surveillance to be done on a regular basis. It is also recommended that occasional surveillance be undertaken in areas where *A. stricta* is not predicted to occur as verification (Fox et al., 2009).

4.3. Conclusions

We have proposed a predictive risk mapping approach to delimiting invasions by species at an intermediate stage of invasion when eradication feasibility is uncertain. Applying suitable surveillance protocols at multiple spatial scales to accurately delimit invasions is an important aspect of assessing species for eradication. Risk mapping can serve as a useful tool for focussing search efforts in high risk areas, and thereby reducing the total cost and effort required for surveillance.

Local knowledge of *A. stricta* invasion has to date been the most successful method of finding populations, and this should be targeted in an effective way, by training local land managers and workers to detect and report sightings of *A. stricta*. There is a need to promote national scale awareness of *A. stricta* through targeted distribution of information flyers in order to determine if *A. stricta* has invaded other parts of the country. While at a regional scale, the focus of *A. stricta* management is monitoring and surveillance to ensure populations are detected rapidly as well as attempting to reduce seed spread. Locally, co-ordinated surveying and clearing of plants is necessary to ensure effective search and control of plants at all sites to prevent reproduction.

While *A. stricta* has only invaded transformed plantation roadsides to date, it poses a risk of spreading into adjacent natural areas and increasing in numbers if uncontrolled. Given risks posed by Australian acacias to South Africa and the assessment here, *A. stricta* should be considered a high risk species and should therefore remain a target for eradication (i.e. category 1a under proposed NEM:BA regulations) despite its large range size relative to other current eradication attempts (van Wilgen et al., 2011).

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Appendix A

Hop Wattle

Scientific name *Acacia stricta*

If you see this plant or know of areas where it occurs, please contact us so that we can record its location and remove it.

SOUTH AFRICAN national biodiversity institute SANBI

Water Affairs
Agriculture, Forestry and Fisheries
Environmental Affairs

C·I·B DST-NRF Centre of Excellence for Invasion Biology

South African NATIONAL PARKS

MTO FORESTRY

WORKING FOR WATER

EXPANDED PUBLIC WORKS PROGRAMME
CONTRIBUTING TO A NATION AT WORK

Acacia stricta can be easily identified by its distinctive erect leaves (phyllodes), yellow flower-heads that sit close to the stem and a single midrib on each leave (unlike rooikrans (*Acacia cyclops*) which has 3–5 veins). It grows to approximately 6–8 m tall and has an upright and scraggly appearance. *Acacia stricta* (Hop Wattle) is an alien plant native to Australia that has recently been identified as an invasive problem in the Knysna area, where it is spreading along disturbed roadsides into forestry plantations. It is currently listed as a Category 1 (a) invasive species which requires compulsory control and eradication. Given the widespread invasions of other similar Australian Acacia species in South Africa (particularly the Western Cape), *Acacia stricta* may potentially become a significant threat to biodiversity. SANBI, Early Detection and Rapid Response Programme (funded by Working for Water Programme) and the Centre for Invasion Biology at Stellenbosch University are working together to map the extent of the invasion by *Acacia stricta*. Efforts to eradicate it from Knysna will be made in collaboration with SANParks and MTO Forestry.

Contact:

alienplants@sanbi.org.za

Haylee Kaplan:	0725880231	email:	hkaplan@sun.ac.za
John Wilson:	021 808 3408	email:	jrwilson@sun.ac.za
Ernita van Wyk:	021 799 8743	email:	Er.vanWyk@sanbi.org.za

SANBI Graphics. August 2010.

Appendix B: Weed risk assessment of *Acacia stricta* following Pheloung et al (1999)

Question	Answer	Reference	Score	Range of possible scores
Is the species highly domesticated?	No		0	0 or -3
Species suited to South African climates	High	Fig. 8	1	2
Quality of climate match data (0-low; 1-intermediate; 2-high)	Intermediate		1	0 – 2
Broad climate suitability (environmental versatility)	Yes. Found in sub-tropical and temperate type climates.	1	1	0 – 2
Native or naturalised in regions with extended dry periods	Yes	1	1	0 or 1
Does the species have a history of repeated introductions outside its natural range?	No		0	0 or 1
Naturalised beyond native range	Yes. In South Africa and New Zealand.	1	2	-2 – 2
Garden/amenity/disturbance weed	Yes. Invades disturbed roadsides.	Pers. obs.	2	0 – 2
Weed of agriculture/horticulture/forestry	Yes. Invades forestry plantations.	Pers. obs.	3	0 – 4
Environmental weed	Not known		?	0 – 4
Congeneric weed	Yes	2	2	0 – 2
Produces spines, thorns or burrs	No		0	0 or 1
Allelopathic	No		0	0 or 1
Parasitic	No		0	0 or 1
Unpalatable to grazing animals	Not known		?	1 or -1
Toxic to animals	No		0	0 or 1
Host for recognised pests and pathogens	Not known		?	0 or 1
Causes allergies or is otherwise toxic to humans	Not known		?	0 or 1
Creates a fire hazard in natural ecosystems	Not known		?	0 or 1
Is a shade tolerant plant at some stage of its life cycle	No		0	0 or 1
Grows on infertile soils	Yes	3	?	0 or 1
Climbing or smothering growth habit	No		0	0 or 1
Forms dense thickets	Yes	Pers. obs.	1	0 or 1
Aquatic	No		0	0 or 5
Grass	No		0	0 or 1
Nitrogen fixing woody plant	Yes		1	0 or 1
Geophyte	No		0	0 or 1
Evidence of substantial reproductive failure in native habitat	No		0	0 or 1
Produces viable seed	Yes	Results	1	1 or -1
Hybridises naturally	Yes. Possibly with *A. paradoxa*	4	1	1 or -1
Self-fertilisation	Unknown		?	1 or -1
Requires specialist pollinators	No		0	0 or -1
Reproduction by vegetative propagation	Yes	5	1	1 or -1
Minimum generative time (years)	1 year		1	-1 – 1
Propagules likely to be dispersed unintentionally	Yes		1	1 or -1
Propagules dispersed intentionally by people	No		-1	1 or -1
Propagules likely to disperse as a produce contaminant	No		-1	1 or -1
Propagules adapted to wind dispersal	No		-1	1 or -1
Propagules buoyant	Not known		?	1 or -1
Propagules bird dispersed	Not known		?	1 or -1
Propagules dispersed by other animals (externally)	Not known		?	1 or -1
Propagules dispersed by other animals (internally)	No		-1	1 or -1
Prolific seed production	Yes	Pers. obs.	1	1 or -1

Evidence that a persistent propagule bank is formed (>1 yr)	Yes	Results	1	1 or -1
Well controlled by herbicides	Yes	Pers. obs.	-1	1 or -1
Tolerates or benefits from mutilation, cultivation or fire	Yes	Pers. obs	1	1 or -1
Effective natural enemies present in Australia	Not known		?	1 or -1

[1] Australian Virtual Herbarium; [2] ; [3] www.intertwine.com.au; [4] Maslin, 2001; [5] http://www.yarraranges.vic.gov.au

CHAPTER 4

General conclusions

Both *Acacia implexa* and *A. stricta* pose significant threats to South Africa. They possess invasive traits similar to many other invasive *Acacia* species: the ability to grow quickly; produce large numbers of seed, reproduce vegetatively and form persistent seed banks; and in the case of *A. implexa,* extensive ability to resprout on clearing (Table 4.1). The relatively limited distribution of *A. implexa* and *A. stricta* compared to other Australian acacias is likely a result of lower propagule pressure and utilisation since neither species was widely planted or used for commercial purposes. Despite this, both species have naturalised and we have found evidence of considerable population spread at some sites – the populations of *A. implexa* have spread up to 1 km at Tokai and ~ 5km along the Eerste River; and populations of *A. stricta* have dispersed several kilometres along roads.

Acacia implexa and *A. stricta* are currently targeted for eradication, and given their intermediate range sizes (i.e. neither widespread nor isolated to a single site) there is uncertainty regarding the feasibility of eradication. The aim of this work was to determine the current extent of the invasions by *A. implexa* and *A. stricta* to provide an objective assessment of whether eradication should be attempted and to provide suitable strategies for their effective management. The previous two chapters have provided detailed assessments of the current status and management of *A. implexa* and *A. stricta* from which we can draw the following conclusions:

1. *Introduction history and current extent of invasions by <u>Acacia implexa</u> and <u>A. stricta</u>*

Populations of *A. implexa and A. stricta* were found to occur at several distinct localities within the Western Cape Province. Both species are associated with forestry areas. Commercial forestry is one of the main reasons for the importation of many *Acacia* species to South Africa (Poynton, 2009) and although neither *A. implexa* nor *A. stricta* were commercially planted, the presence and current distribution of both species is linked with this introduction pathway. *Acacia implexa* was introduced for forestry as a potential timber species and now occurs at three isolated localities (Stellenbosch, Wolseley and Tokai) which have historically been used for forestry. The reason for introduction of *A. stricta* is unknown but it was most likely introduced at plantation nurseries. Populations of *A. stricta* were found at nine localities all within plantations in the Knysna and Wilderness sections of the Garden Route.

Population structures of both *A. implexa* and *A. stricta* are characteristic of rapidly expanding populations. *Acacia implexa* reproduces predominantly by suckering which has enabled high densification of stands. However, despite prolific seed production, predation of seeds, predominantly by insects has resulted in a general lack of substantial seed banks and relatively slow spread rates. In contrast, *A. stricta* accumulates large seed banks comprising up to 1000 small seeds/m^2 per reproducing plant which germinate easily following disturbance. *Acacia stricta* seeds are spread mainly in soil by road maintenance vehicles and the risk of seed spread is therefore much higher than that of *A. implexa*.

2. *Current impacts and risks posed by <u>Acacia implexa</u> and <u>A. stricta</u> to South Africa*

Acacia implexa and *A. stricta* both received high scores in the Australian Weed Risk Assessment (AWRA; Pheloung et al., 1999) which indicates that both these species should be considered highly invasive and potential threats to native habitats. Risk assessments by Castro-Díez et al. (2011) based on life history attributes, human usage and native climate associations and Hui et al. (2011) based on native distribution patterns did not rate *A. stricta* as a particularly high risk invader (17.9 % probability of being invasive) compared with other *Acacia* species. *Acacia implexa* was predicted more likely to be invasive (26.5 %; Castro-Diez et al., 2011) and based on native distribution patterns, *A. implexa* clustered with other known invasive *Acacia* species (Hui et al., 2011). Based on climatic suitability predictions, both species have considerably large potential ranges in South Africa.

Impacts of these species have not been directly measured here, but both species are able to form dense monospecific stands (*A. implexa* by means of suckering and *A. stricta* via seedling germination from a large seed bank) and thus have the potential to outcompete native species. *Acacia implexa* has transformed natural vegetation through the formation of large, dense stands, particularly native vegetation along the banks of the Plankenbrug River in Stellenbosch. In addition, promotion of high fire intensity and high water usage by these species could further transform natural habitats. As both species occur adjacent to conservation areas, these potential threats are of even greater importance. *Acacia stricta* still remains confined to plantations where it favours areas of high disturbance. Road maintenance acts both as a dispersal vector and promoter of colonisation for *A. stricta* by transporting seeds and creating disturbance along roadsides where seeds can germinate and grow. Several populations occur on forest margins and, like many other *Acacia* species which have invaded natural fynbos and forest in the Garden Route National Park, *A. stricta* could potentially invade native vegetation following disturbance.

3. Management strategies for ***Acacia implexa*** *and* ***A. stricta***

Based on the currently known extents of *A. implexa* and *A. stricta*, together with their high risk of invasiveness, eradication should be attempted for these species (i.e. both species should remain category 1a under NEM:BA). The slow spread rates of *A. implexa* populations and the restriction of *A. stricta* infestations to disturbed roadsides make eradication of these species feasible. In order to accomplish this, each species requires a set of management strategies that are specific to their reproductive and dispersal characteristics (for an example see the *A. stricta* management plan in Addendum B).

While both *A. implexa* and *A. stricta* have histories of management, these have generally been sporadic and at the discretion of various land managers on whose land the species occurred. As previous management practices have not been targeted at eradicating either species, none have been effective in permanently removing populations. A recommendation of this work is that management of *A. implexa* and *A. stricta* be co-ordinated by the national Early Detection and Rapid Response programme to ensure continuity of management among populations for the duration of the eradication programmes.

Manual clearing and herbicide application is prescribed for the removal of *A. implexa* and *A. stricta. Acacia implexa* is highly prone to vigorous resprouting following physical damage, and herbicide application must therefore follow correct procedures to prevent any vegetative regeneration. For *A. stricta*, seed production should be prevented as far as possible by ensuring the removal of plants before reproductive maturity (i.e. after one year’s growth). In addition, plantation activities at sites where *A. stricta* occurs should be monitored for seed spread and the emergence of new infestations.

At a broader scale, there is a need to determine whether our current knowledge of the extents of *A. implexa* and *A. stricta* is accurate by expanding the search for populations to the rest of the country. This has been done primarily by means of information flyer distribution. Expert knowledge of plant invasions has proved to be a valuable resource for locating populations of *A. implexa* and *A. stricta* thus far and relying on this for national scale surveillance is probably the most effective and viable method for locating new invasions. While we are relatively confident in our estimation of *A. implexa* extent, our confidence is less so for *A. stricta* where the higher risk of dispersal over longer distances creates a need for greater search effort.

Besides creating awareness of this species, the management of *A. stricta* also requires annual searches of the entire study area to locate any new populations. We therefore used a correlative habitat suitability approach to predict spread pathways and areas with high risk of invasion by *A. stricta* to direct search efforts. We found that *A. stricta* occurrence was associated with minor roads (non-tarred roads that are used and maintained less often) and recent planting or harvesting activity in a plantation compartment. Based on these predictions, vehicle searches should be focussed along gradable roads in recently disturbed plantations. By refining the search to high risk areas, the total area to be surveyed and costs of surveying are reduced, and detecting infestations is more likely.

Our risk mapping predictions are however based on the current distribution of *A. stricta* which is still in early stages of its invasion. Given sufficient opportunities for spread, *A. stricta* could potentially invade areas not predicted to be suitable. Although risk mapping can reduce the search area, the possibility of missing infestations in areas outside of high risk areas is possible. To account for this, there needs to be a secondary search method that can effectively detect new infestations without intensive searching. We found local knowledge of plant invasions and reports from local plantation managers and field workers proved to be the most valuable means of locating populations of *A. stricta*. Similarly, all populations of *A. implexa* were located based on reports from land managers and expert knowledge of plant invasions. It is therefore important to actively involve people who work with invasive species regularly in the management of *A. stricta* and *A. implexa* to effectively locate all infestations of these species within affected areas and the rest of the country.

4. Implications for Acacia management in South Africa

Australian acacias are among the most widespread and damaging plant invaders in South Africa. A national strategy for managing acacias at different levels of impact was recently proposed to effectively manage *Acacia* species at all stages of invasion (van Wilgen et al., 2011; Fig. 4.1). While impacts on natural landscapes can be reduced and mitigated, the eradication of species before they become widespread remains the most cost-effective way to prevent large-scale damage. Assessments of eradication feasibility are necessary for eradication targets, particularly those closer to the upper limits of eradication feasibility (i.e. where there is greater uncertainty of whether to contain or eradicate a species).

This work has shown that detailed assessments of species at intermediate stages of invasion is an important initial step in an eradication attempt, and better understanding of species specific invasion characteristics can help to improve management and potentially increase the probability of success of eradication.

Detection of *A. implexa* and *A. stricta* at relatively early stages of their invasions has allowed eradication programmes to be put in place before the species are too widespread to eradicate (Fig. 4.2). If allowed to persist for a further 10-15 years, it is expected that eradication would no longer be an option for these species. In this case, containment would need to be considered as an alternative. As the majority of *A. implexa* seeds are predated upon by native insects, biological control of *A. implexa* would not be a necessary management strategy. However this could be considered as an option for *A. stricta* management in the case of failed eradication. Evaluation of eradication programmes and adaptive management will be essential to ensure effective future management of both *A. implexa* and *A. stricta*.

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Addendum A

Assessment and attempted eradication of Australian acacias in South Africa as part of an Early Detection and Rapid Response programme

John R. Wilson[1,2*], Haylee Kaplan[1], Carlo de Kock[3], Dickson Mazibuko[1], Jason de Smidt[3], Rafael D. Zenni[1], Ernita van Wyk[2]

[1]Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Matieland 7602, South Africa

[2]South African National Biodiversity Institute, Kirstenbosch National Botanical Gardens, Claremont 7735, South Africa.

[3]Invasive Species Control Unit, South African National Parks, Westlake Square, Corner Steenberg Road & Westlake Drive, Cape Town, South Africa

* Corresponding author, Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Private Bag X1, Matieland 7602, South Africa, john.wilson2@gmail.com, Tel: +27 (0)21 808 3408, Fax: +27 (0)21 808 2995

Abstract

One of the targets of the initial phase of the South African National Programme for Early Detection and Rapid Response (EDRR) of Invasive Alien Plants has been introduced wattles (*Acacia* subgenus *Phyllodineae* (DC.) Ser.). While 15 Australian acacias are listed in South African regulations on invasive plants, only eleven are widespread. The remaining four species (*A. adunca*, *A. implexa*, *A. paradoxa*, *A. stricta*) have not been investigated in depth, nor has there been a concerted or sustained effort to manage these invasive populations. In this paper we describe EDRR's involvement in Australian *Acacia* species, in particular: current plans to eradicate *A. paradoxa* from Table Mountain; initial field and risk assessments for *A. implexa* and *A. stricta*; and surveys to determine the status of other introduced Australian *Acacia* species.

Introduction

Australian *Acacia* species (or wattles, i.e. *Acacia* subgenus *Phyllodineae*) are regarded as a model group in invasion biology (Richardson et al., 2011). Management practices around the world have focussed on the most widespread invaders, but given the difficulties of long-lived persistent seed-banks, preventing or eradicating new invasions before invasions become established will be the best strategy (Wilson et al., 2011).

South Africa is in the process of developing a strategic plan for managing biological invasions, and wattles have been used as a test case (van Wilgen et al., 2011). Draft South African regulations on invasive alien species (National Environmental Management: Biodiversity Act, 2009) lists fifteen species of wattles. Eleven of these have been subjected to substantial investigation and are found at several sites throughout the country (Figure 1). These species are indisputably invasive (category E* according to the scheme proposed by Blackburn et al. 2011); are the subject of on-going management; and are proposed under the draft regulations either as category 1b if they are not widely used, or as 2 or 3 if they still provide benefits in some instances.

The remaining four species (*A. adunca*, *A. implexa*, *A. paradoxa*, *A. stricta*) are proposed to be listed as category 1a (defined as "requiring compulsory control"). For management purposes this is taken to mean they are eradication targets. While *A. implexa*, *A. paradoxa*, and *A. stricta* are spread over several hundred hectares (and so are category E), given its restricted distribution, *A. adunca* is taken to be category D1**. One more species, not included in the legislation, *A. viscidula*¸ is also recorded as naturalised and spreading from a single site (D1).

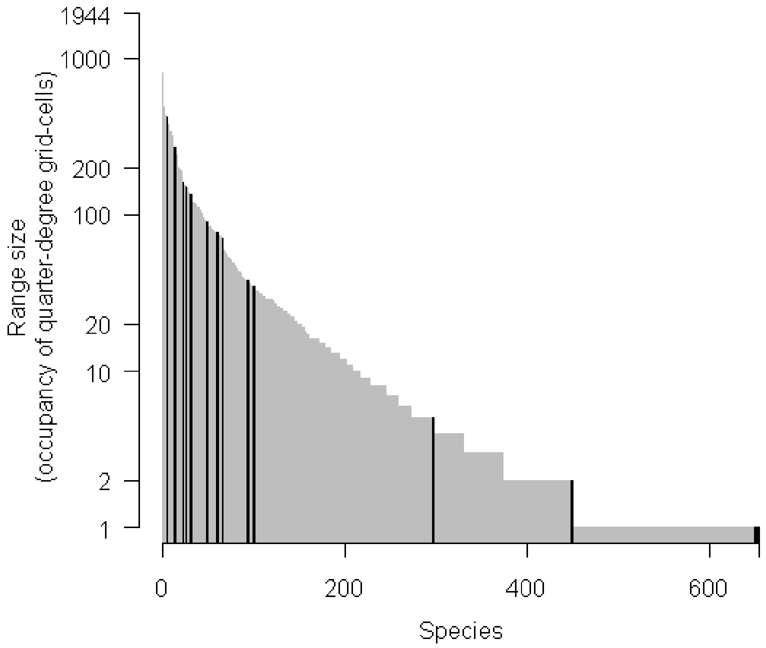

Figure 1: Frequency distribution of invasive alien plants range sizes in South Africa, with fifteen invasive *Acacia* subgenus *Phyllodineae* shown in black. Data are from the South African Plant Invaders Atlas (accessed 2009). Continental South Africa covers 1944 quarter-degree grid cells (QDGCs). The total number of species recorded in SAPIA changes through time with taxonomic revisions and new findings, in particular the numbers shown here do not reflect the revised results from the EDRR work.

* Category E species are "fully invasive species, with individuals dispersing, surviving and reproducing at multiple sites across a greater or lesser spectrum of habitats and extent of occurrence "defined by Blackburn et al. (2011).
** Category D1 species consist of a "self-sustaining population in the wild, with individuals surviving a significant distance from the original point of introduction" defined by Blackburn et al. (2011).

Another nineteen species of wattle have been introduced to Southern Africa for commercial reasons according to a recent review (Poynton, 2009). A further sixty or so species are recorded in South African Herbaria, suggesting they have been introduced at some time or other. See Richardson et al. (2011) for a compilation of all the species lists.

In this article we describe the work done by the South African National Programme for Early Detection and Rapid Response of Invasive Alien Plants (EDRR) (Ivey et al., this volume) to combat wattle invasions. The specific aims are to: a) implement the eradication of *A. paradoxa* from Table Mountain through adaptive management; b) provide both initial field and risk assessments for *A. implexa* and *A. stricta*; and c) determine the status of other introduced Australian *Acacia* species.

Eradication of Acacia paradoxa from Table Mountain (Cape Town, SA)

Acacia paradoxa D. C. (Kangaroo Thorn) is currently restricted in South Africa to around ~3.1 km^2 on Table Mountain (Devil's peak) (Moore et al., 2011). The current population is thought to be the result of a few plants initially introduced as a curiosity around the end of the nineteenth century, followed by a long history of neglect (Zenni et al., 2009). Alien plant clearing operations started targeting the plant in the mid-1990s. While the intention of the recent management efforts was to eradicate the population, the clearing until now can be categorised as sporadic and partial, focussing mostly on the largest and presumably oldest plants. The first detailed survey of the population was conducted in 2008 as part of a student project (Zenni et al., 2009), and since then targeted efforts have been co-ordinated by EDRR to ensure the population is eradicated.

General alien clearing operations in the affected area (based on figures from 2009/2010) cost around 400–600 ZAR ha^{-1}, with the return time in any one location approximately 3–5 years. However, this is insufficient to prevent plants producing seeds, particularly as one year old plants can possibly set seed and plants over 2m tall are missed during the clearing. In an area of 45 000 m^2 evaluated 3 years after general alien clearing operations, around 1 000 *A. paradoxa* plants were found, most showing signs of reproduction (Zenni et al., 2009).

Fortunately, the population does not appear to have spread far from the initial point of introduction, and the seed-bank is confined almost exclusively to below the canopy. If annual search-and-destroy operations systematically survey the affected area during the flowering season (i.e. August–October) (Fig. 2), it is likely that seed-set can be prevented. Indeed, a recent decision analysis suggested that the optimal management goal for this population is eradication (Moore et al., 2011).

a) 

b) 

Figure 2: Example of surveying work on *Acacia paradoxa* on Table Mountain in December 2009. a) physical area surveyed; b) track-lines recorded and plotted on Google Earth. Each icon represents the location of an *Acacia paradoxa* that was found and treated. The survey consisted of teams of three or four, walking up,

then down with one of the surveyors carrying a GPS. So at least four people walking parallel to each other will have surveyed in the gap between tracks (see Zenni et al., 2009 for more details).

After the first year of clearing in response to the report of Zenni et al. (2009), a wild-fire in early 2009 went through much of the affected area. This allowed an assessment of the effect of fire on seed germination. In both field assessments and lab trials, fire stimulated up to 90% seed germination compared to an average of around 10% in normal conditions (D. Mazibuko, unpublished data). Despite the fact that in the dense areas there was 100% cover with *A. paradoxa* following the fire, there was a substantial regrowth of species other than *A. paradoxa*.

The project is now in a follow-up phase involving further search-and-destroy surveys and pulling of seedlings that have emerged following the Vredehoek fire in May 2009. In 2010 the unburnt areas (1.45 km^2), were resurveyed costing 64 000 ZAR, and about a hundred adult plants were found (not found on previous surveys). Later in the year and in the start of 2011, in the burnt section over 600,000 seedlings were hand-pulled on a contract costing 400 000 ZAR. As such the exercise is much more expensive than general clearing operations (which will still continue in the area separate to the *A. paradoxa* work). Initial estimates suggest that working in groups of 2–4 people, each person can cover 1–2 ha per day (so a total requirement of around 200 person field days per year). However, this approach is estimated to be much more cost-effective than if either no action is taken, or containment is attempted (Moore et al., 2011). The total cost estimate if control is successful is 5.4 million ZAR spread over 20 years.

In addition to the walked search-and-destroy surveys (Fig. 2), areas immediately adjacent to the park will be surveyed (EDRR provides a more flexible mandate than if the process was controlled solely by South African National Parks); and in steep areas within Table Mountain National Park, specifically trained and equipped "high angle teams" will be used, and plants treated as before.

As for most of the invasive Australian acacias in South Africa (Richardson and Kluge, 2008), *A. paradoxa* has a significant long-lived seed-bank of >1000 seeds m^{-2} in places, and our concern is that in dense areas, the seed-bank will persist for decades. Given the fact that Table Mountain National Park is a World Heritage Site, alien clearing operations are likely to be a part of land management for many years to come. Nonetheless, efforts to reduce the seed-bank would be advisable. Unfortunately, the infested site is very close to Cape Town (see Figure 1), and as such the requirements for allowing fires in this area are stringent.

The main future steps in the eradication will be to assess the success and control of current practices and assess the likely benefits of using different methods to reduce the seed-bank.

Initial field and risk assessments for A. implexa and A. stricta

The initial assessment of *A. implexa* was started in 2009, and was completed in early 2011 confirming the view that this species should also be an eradication target (Kaplan *et al*. in review South African Journal of Botany). Three populations of *A. implexa* have been identified, mapped, and studied as part of a student project funded by EDRR. The survey found approximately 30 000 *A. implexa* individuals within a total invaded area of 6 km^2 across the three sites. While *A. implexa* produces a prodigious amount of seed, it appears not to have spread widely yet (perhaps through poor dispersal and high seed mortality), although it is beginning to spread along one water-course. Control is problematic given its strong ability to sucker, but general clearing operations (co-ordinated but not managed by EDRR) are on-going.

The exact delimitation of the species in South Africa is not certain as the species is difficult visually to separate from *Acacia melanoxylon* R.Br. in W.T. Aiton. Indeed, several new reports of sightings have subsequently been confirmed as *A. melanoxylon*. We have distributed identification leaflets asking for new sightings, and if many new reports are confirmed, then the project will need to be reassessed. But given the relatively slow spread, but the threat it poses to biodiversity, it will remain an eradication target for the present.

Acacia stricta (Andrews) Willd. is only known from a few populations scattered through the Knysna and Wilderness areas of the Southern Cape. Much of the population appears to be situated close to road-sides in pine plantations, though it is reaching high densities in places, and is again a prolific seed producer. It is unclear how it reached the area, but has been there for at least 15 years, perhaps being spread by vehicles or road resurfacing work. Control of mature plants appears to be relatively straight-forward, although the seed-bank may represent a major challenge for eradication, and certainly the populations along the major highway (the N2) were of concern regarding its potential spread.

A thorough survey of the area in September 2010 found eight populations of *A. stricta* and a total of ~ 20 000 plants, all of which occurred on plantation land. The infestations straddle various land-owners and management areas (in particular the Mountain to Ocean Forestry Company). EDRR facilitated a meeting of all stakeholders involved in May 2011, and the group are developing a joint management plan. This is a good example where EDRR can act as an independent co-ordinator to ensure that appropriate control occurs wherever plants are found (see paper by Ivey et al., this volume).

Surveys of other introduced Australian Acacia *species*

Records of Australian acacias in the Southern African Plant Invaders Atlas as well as records in South African Herbaria are being collated and followed up. *Acacia adunca* Cunn. ex Don is currently known to have naturalised in only one site in South Africa, “Bien Donné” Experimental Farm in the Franschhoek Valley, and the population is being assessed. However, it has not spread widely and is not an immediate priority for eradication.

Several other species have also been found. *Acacia viscidula* Benth. is invading Newlands Forest on the slopes of Table Mountain in Cape Town, and a few plants of *Acacia ulicifolia* (Salisb.) Court and *Acacia retinoides* Schldl. have naturalised at Tokai Arboretum in Cape Town (i.e. category C3). Reports of *Acacia fimbriata* Cunn. ex Don in Grahamstown have been followed up, but no plants were found (potentially category A2). We still need to confirm if a reported naturalised population of *A. cultriformis* Cunn. ex Don from Ladybrand exists. There are also arguable two species that are planted in some numbers but are not recorded as invasive *Acacia pendula* Cunn. ex Don and *Acacia floribunda* (Vent.) Willd. (potentially category B2), but again more work is required to confirm their status as not naturalised.

Conclusions

Given the number and diversity of Australian acacias introduced to South Africa, they represent an excellent test case both for general theories of invasiveness and for our ability to conduct eradications. The long history of introduction and plantings means that there is a high possibility for many species that are currently at low density to become invasive in later years, and the long-lived seed bank represents a challenge for control.

We would, however, conclude that specific EDRR type projects are warranted on Australian *Acacia* species as: they allow the flexibility to look at infestations across administrative and management boundaries; they provide continuity of funding; and EDRR provides the focus required for eradication. The last point is particularly important given the number of invasive Australian *Acacia* species that require general management.

Acknowledgements

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Addendum B

Invasive alien plant eradication plan for *Acacia stricta* in South Africa (2011-2016)

Summary

This plan outlines the proposed management of *Acacia stricta* for 2011–2016, a species targeted for eradication from South Africa. Management operations will include annual search and destroy surveys of *A. stricta* during flowering season (September/October), additional surveys for new populations, and any new sightings will be followed up. Progress will be reported on annually, with the plan reviewed after five years (2016). This is a collaborative effort by Mountain to Ocean (MTO) Forestry, SANParks and SANBI, with co-operation from South African National Roads Agency Ltd. (SANRAL) and PG Bison. Funds will be made available to implement the management plan through the Early Detection and Rapid Response Programme, but relevant land-owners will assist where appropriate and possible (e.g. clearing activities will involve both local stake-holders and a co-ordinator who will visit all affected sites).

Background

Acacia stricta is an emerging invasive species currently found only in the Knysna and Wilderness areas of the Garden Route. Search-and-destroy surveys in 2010-11 found and removed approximately 20 000 *A. stricta* plants distributed across 9 sites. The total invaded area recorded was ~110 ha. Given the similarities to other Australian acacias, *Acacia stricta* is considered a high risk invader. Moreover, it has shown an ability to spread to new areas particularly along roadsides in plantation areas where disturbance is likely an important factor in promoting its establishment. Given its currently restricted distribution and the potential for invasion, its management is a matter of national priority.

Currently *A. stricta* occurs on land owned by MTO Forestry, SANParks, SANRAL and plantations owned and managed by PG Bison. All parties are committed to managing *A. stricta* and agree that eradication from South Africa should be attempted. The attempted eradication of *A. stricta* is to be a joint undertaking by MTO, SANParks, with co-operation from SANRAL and PJ van Reenen, under the auspices of the SANBI / Natural Resource Management Programme–Early Detection and Rapid Response Programme for Invasive Alien Species (EDRR).

Purpose of plan

The purpose of the management plan for *Acacia stricta* is to ensure that there are clearly defined objectives and activities to guide annual management operations over a five-year period. The plan is intended to ensure continuity in management in the medium-term in case of changes in land ownership or management. Emphasis is placed on assigning responsibilities, defining and scheduling management operations, and quantifying the amount of effort required.

Monitoring of the management plan is to be done on an annual basis through data collection and reporting co-ordinated by EDRR. A full evaluation of the plan by all stakeholders is to be done after five years, at which point the management aims and plan should be updated accordingly.

Management aims & objectives

Acacia stricta is a category 1a invader under the Conservation of Agricultural Resources Act (Act 43 of 1983; amended 2008), meaning all infestations of this species must be removed. The desired outcome is eradication of *A. stricta* from South Africa.

To achieve this, the following management actions are proposed:

1. Revisit and remove seedlings from all sites with known infestations

Seed banks of *A. stricta* are large and can remain in the soil for several years, therefore making them difficult to control. Preliminary surveys show seedlings are mostly emerging at previously infested sites, therefore this survey will focus on areas where *A. stricta* has been recorded.

All localities where *A. stricta* has been recorded are to be thoroughly surveyed by means of walked transects parallel to roadsides (one transect along road edge and one transect 10 m from roadside) to search for seedlings. Searching is completed when no plants are found for at least 250 m in either direction along the road at a site. If plants are found more than 20 m from roadside, the area will need to be searched using walked transects perpendicular to the road (transects will be up to 10 m apart depending on the density of the vegetation). The search area will depend on the nature of the site, for example, for a plantation this would involve the whole of a particular block, any paths or trails searched as per road-surveys, otherwise at least 100m radius from any plants seen should be searched.

All plants found are to be measured and their locations recorded with a GPS. Smaller plants are to be hand-pulled, and larger plants are to be cut at the base of the stem and sprayed with a glyphosate herbicide of at least 1.5% active ingredient applied as per standard application procedures. Note, plants will resprout unless they are cut at the base and sprayed immediately.

Surveying is to be done annually during flowering season (September/October) when *A. stricta* is most easily detected. Population data needs to be collected in the following format for annual monitoring and assessment purposes:

Survey data sheet

Date: ______________________ **Lat/long:** ______________________

Team: ______________________ **Site:** ______________________

Waypoint	Height (cm)	Stem diam (cm)	Flowers (y/n)	Pods (y/n)	Seeds (y/n)	Distance from road (to nearest 0.5 m)	Resprout (y/n)	Notes

Given the need for co-ordination, continuity and prior experience in identifying plants, EDRR will organise one or two people to visit all the sites. To ensure compliance with all legislation and so the work is co-ordinated with local efforts, each site will be surveyed by teams of 2-3 people (in addition to the EDRR people) provided by the local land manager responsible for the area (See table below). Teams are to include at least one Pest Control Officer. The local staff will also be encouraged to keep an eye out for new infestations where they work.

Land managers responsible for clearing *Acacia stricta* at sites

Site	Approx. area (ha)	Expected duration of task (days, based on 8 field hours)	Current land manager
Bergplaas	3	1	MTO Bergplaas (Marius Strydom)
Karatara	19	3	MTO Bergplaas (Marius Strydom)
Farleigh	2	1	SANParks Farleigh (Jonathon Britton)
Kraaibos	0.5	1	PG Bison (Johan Vermaak)
Buffelsnek	6	2	MTO Buffelsnek (Tom Eckley)
Gouna	8	3	MTO Buffelsnek (Tom Eckley)
Kruisfontein	48	4	MTO Kruisfontein (Bradley Joemat)
Swaneberg	14	2	MTO Kruisfontein (Bradley Joemat)
N2	10	1	MTO Kruisfontein (Bradley Joemat)
Fisantehoek	1	1	MTO Kruisfontein (Bradley Joemat)

2. Searching for new infestations (targeted vehicle surveys)

Acacia stricta may easily be spread to new areas and it is possible that sites were missed during previous surveys. Vehicle surveys need to be done annually during flowering season to search for new infestations of *A. stricta*. Surveys should be focussed on, but not limited to, areas surrounding existing infestations and areas highlighted as “high risk” on the risk map of vulnerable areas for *A. stricta* invasion. Areas recently burnt should also be searched, and incidence of fire at sites should be recorded by the survey team. Ideally vehicle surveys should be done by a single team of 2 people who have experience in *A. stricta* clearing and are therefore better able to spot *A. stricta* along roadsides. This should take up to 5 days to complete.

In addition, all reported new sightings of *A. stricta* should be followed up on and if confirmed, sites must be surveyed during the following annual survey.

3. Searching for new infestations (link with managers in areas and elsewhere)

Information leaflets on *Acacia stricta* will be distributed to conservation and forestry managers around South Africa to create awareness of this species and to gather any reports of sightings of *A. stricta* around the country. Locally, land managers and staff should be made aware of *A. stricta* and report any new sightings. All reports must be sent to alienplants@sanbi.org.za.

4. **Controlling spread of *Acacia stricta***

To reduce the risk of *A. stricta* spreading to new areas, the movement of contaminated soil needs to be monitored. Information on annual activities in affected areas and incidences of fires must be made available so that there can be increased monitoring and surveying effort in those areas. Roads leading out of invaded areas, especially into conservation areas, should be monitored for spread of *A. stricta* along roadsides.

Evaluation of management plan

The management of *A. stricta* will be monitored annually through data collected during each year's survey. If many large plants are found a year after clearing, this may indicate that plants were missed during the previous survey and efforts should be increased in order to find and remove all plants at a site. If many large resprouts are found, this may indicate that herbicide treatment is ineffective. Management operations should be altered accordingly.

The management aims will be reviewed after 5 years to assess whether eradication of *A. stricta* is feasible. This will be evaluated based on the status of *A. stricta* populations after 5 years of clearing and the costs and effort that were required for management operations. If necessary, management aims will be reconsidered.

Agreement in Principle

The undersigned organisations agree to the approach outlined in the management plan and to assist with its implementation, where possible.

____

Name: **Position:**

On behalf of MTO Forestry

____

Name: **Position:**

On behalf of SANParks

____

Name: **Position:**

On behalf of SANBI

____

Name: **Position:**

On behalf of SANRAL

____

Name: **Position:**

On behalf of PG Bison

Fig. 1.1.

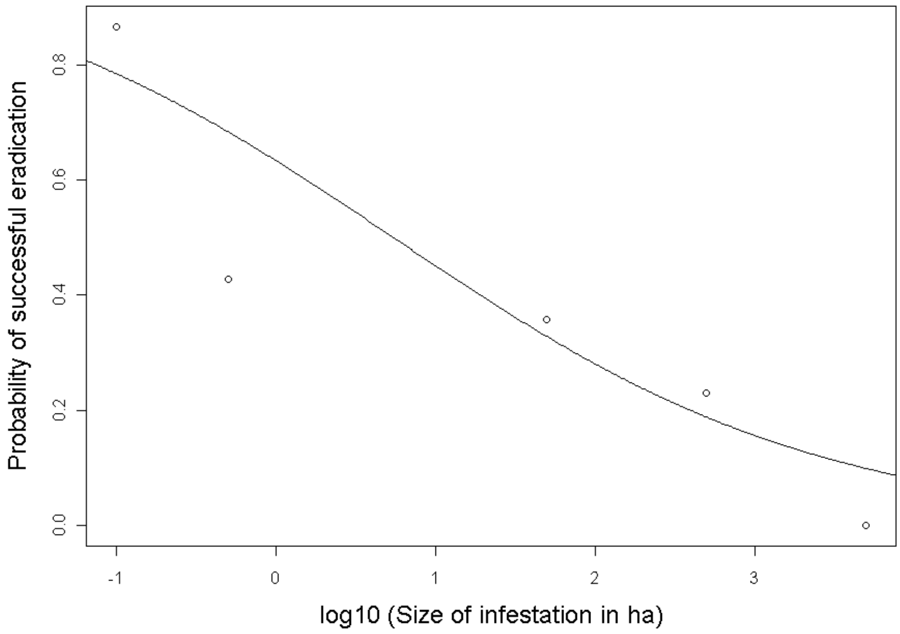

Probability of successful eradication
0.0
0.2
0.4
0.6
0.8
-1
0
1
2
3
log10 (Size of infestation in ha)

Fig. 1.2.

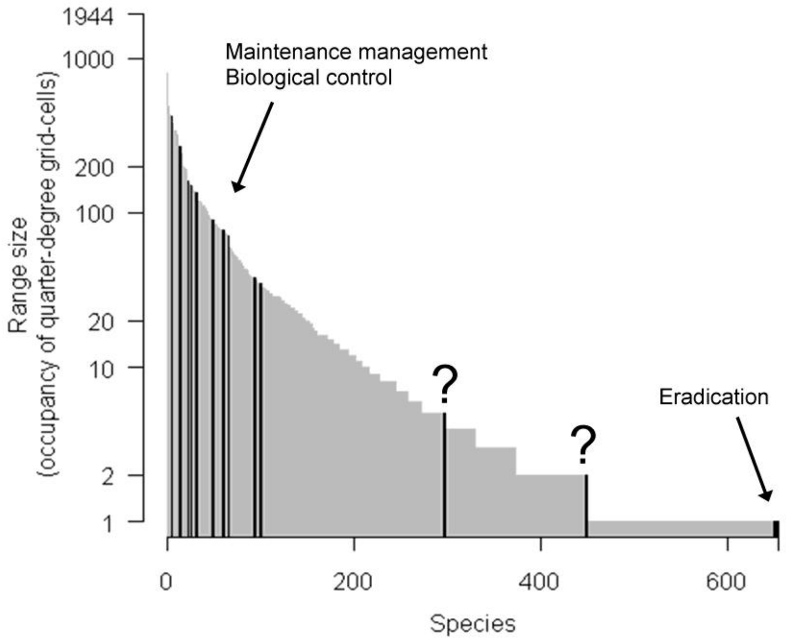

1944
1000
200
100
20
10
2
1
Range size
(occupancy of quarter-degree grid-cells)
Maintenance management
Biological control
?
?
Eradication
0
200
400
600
Species

Fig. 1.3.

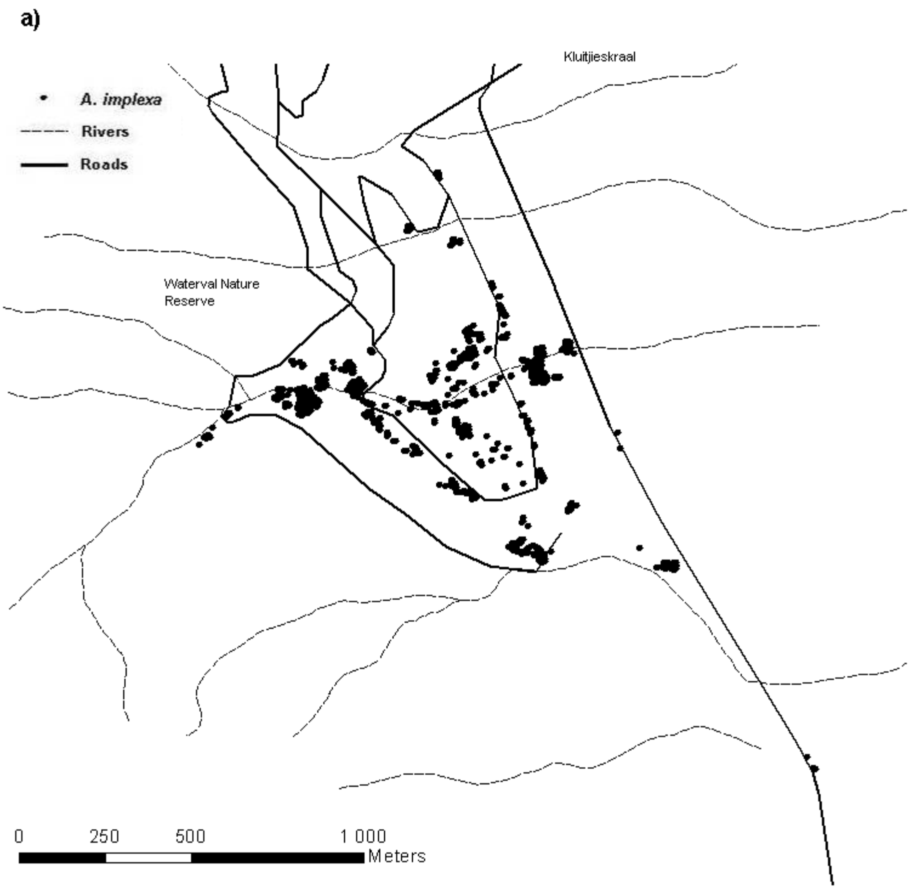
Fig. 1.4.

Fig. 2.1.

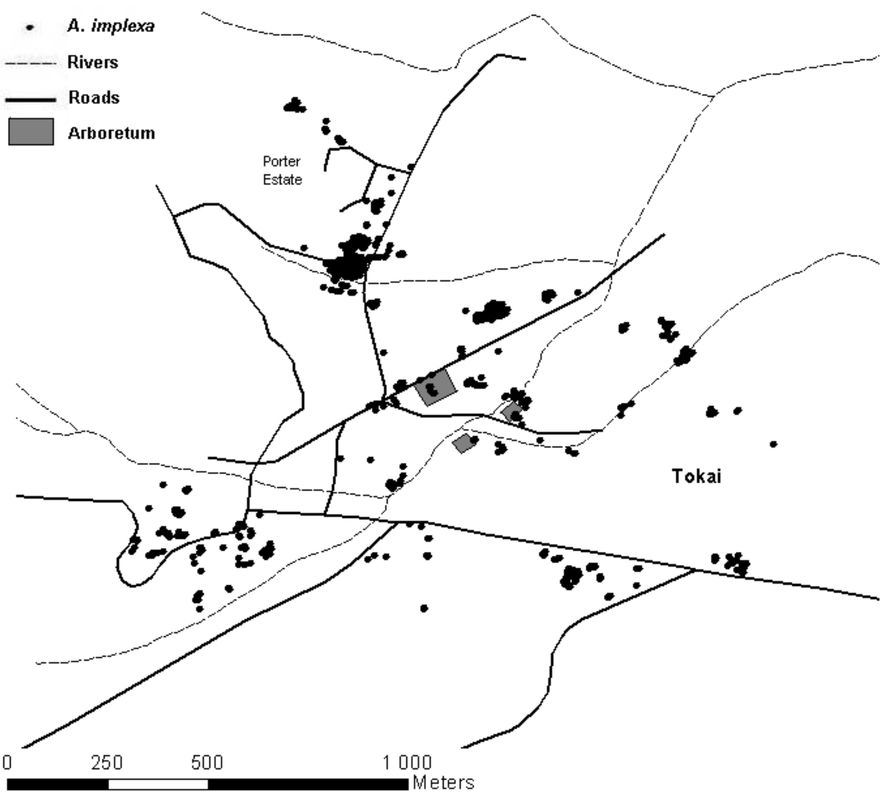
Fig. 2.2.


South Africa
Wolseley
Stellenbosch
Tokai

Fig. 2.3.


a)
Kluitjieskraal
A. implexa
Rivers
Roads
Waterval Nature Reserve
0
250
500
1 000
Meters

b)


A. implexa
Rivers
Roads
Arboretum
Porter Estate
Tokai
0
250
500
1 000
Meters

c)

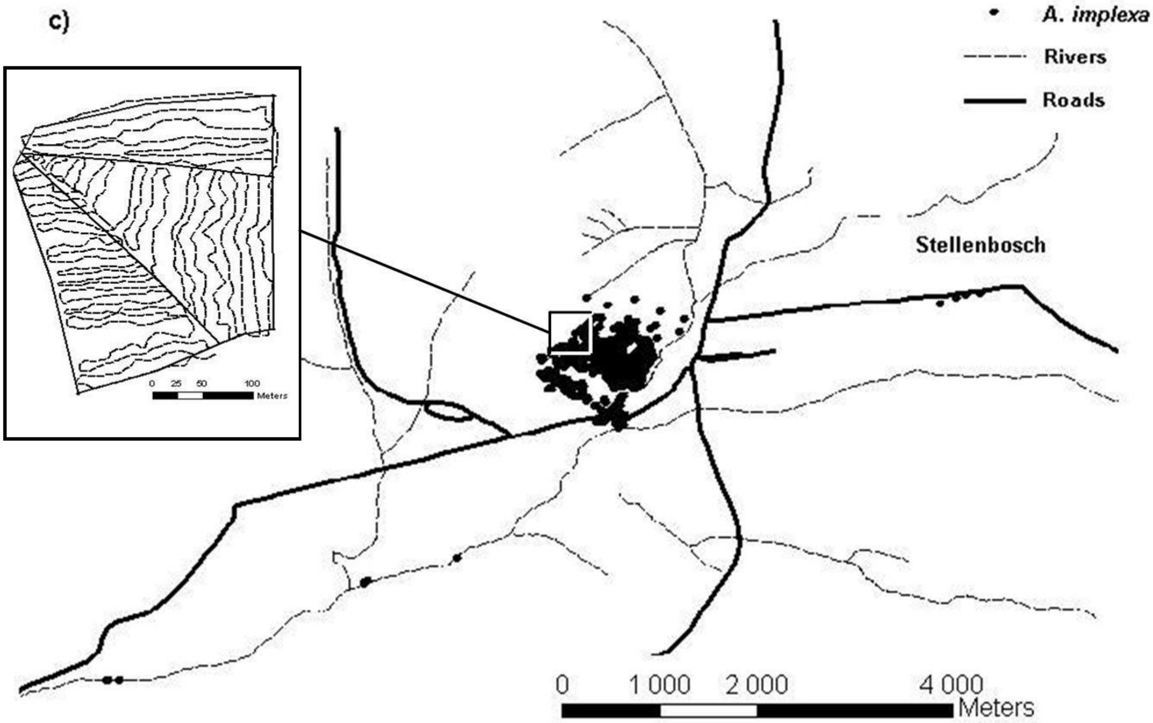

A. implexa
Rivers
Roads
Stellenbosch
0
1 000
2 000
4 000
Meters

Fig. 2.4.

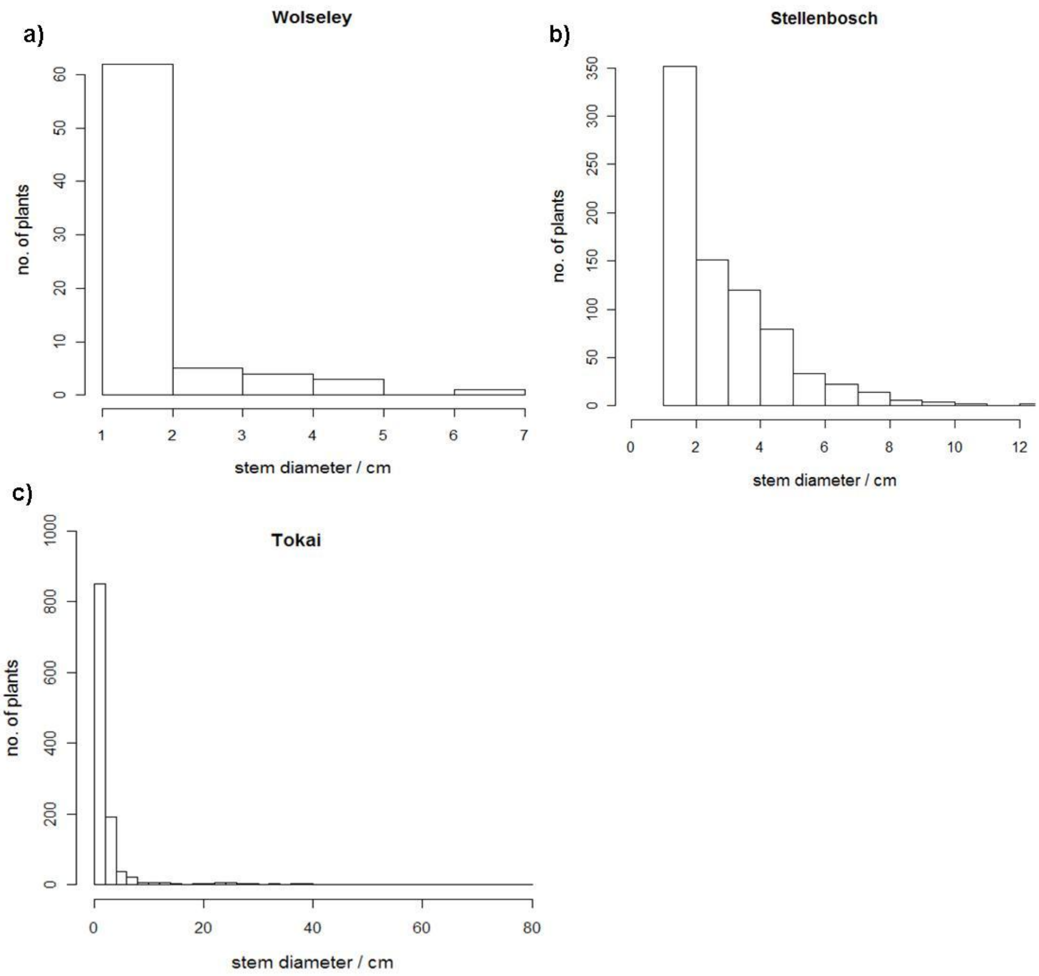

a)
Wolseley
no. of plants
stem diameter / cm
b)
Stellenbosch
no. of plants
stem diameter / cm
c)
Tokai
no. of plants
stem diameter / cm

Fig. 2.5.

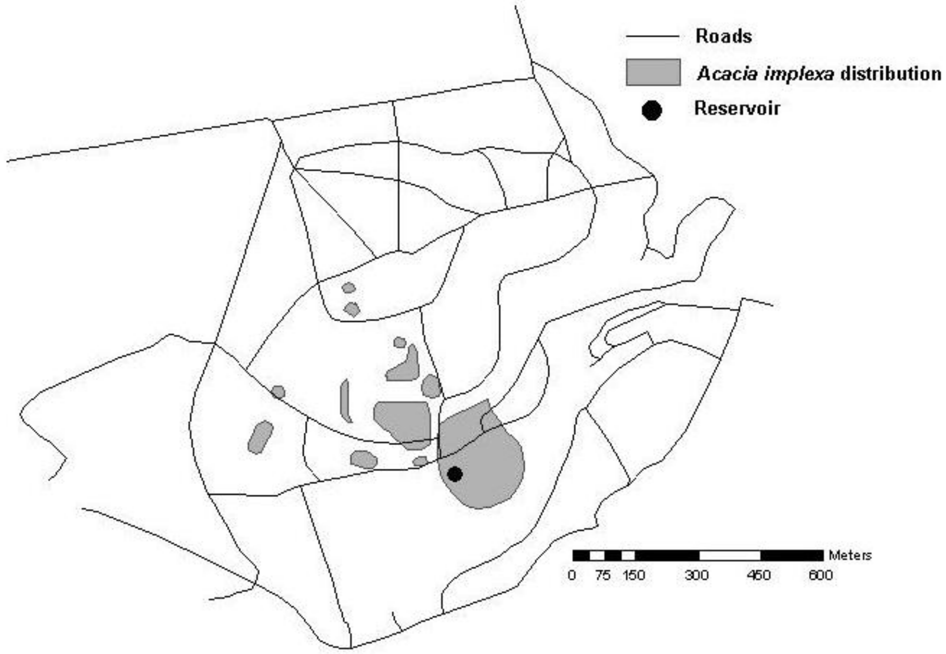

Roads
Acacia implexa distribution
Reservoir
Meters
0 75 150 300 450 600

Fig. 2.6.

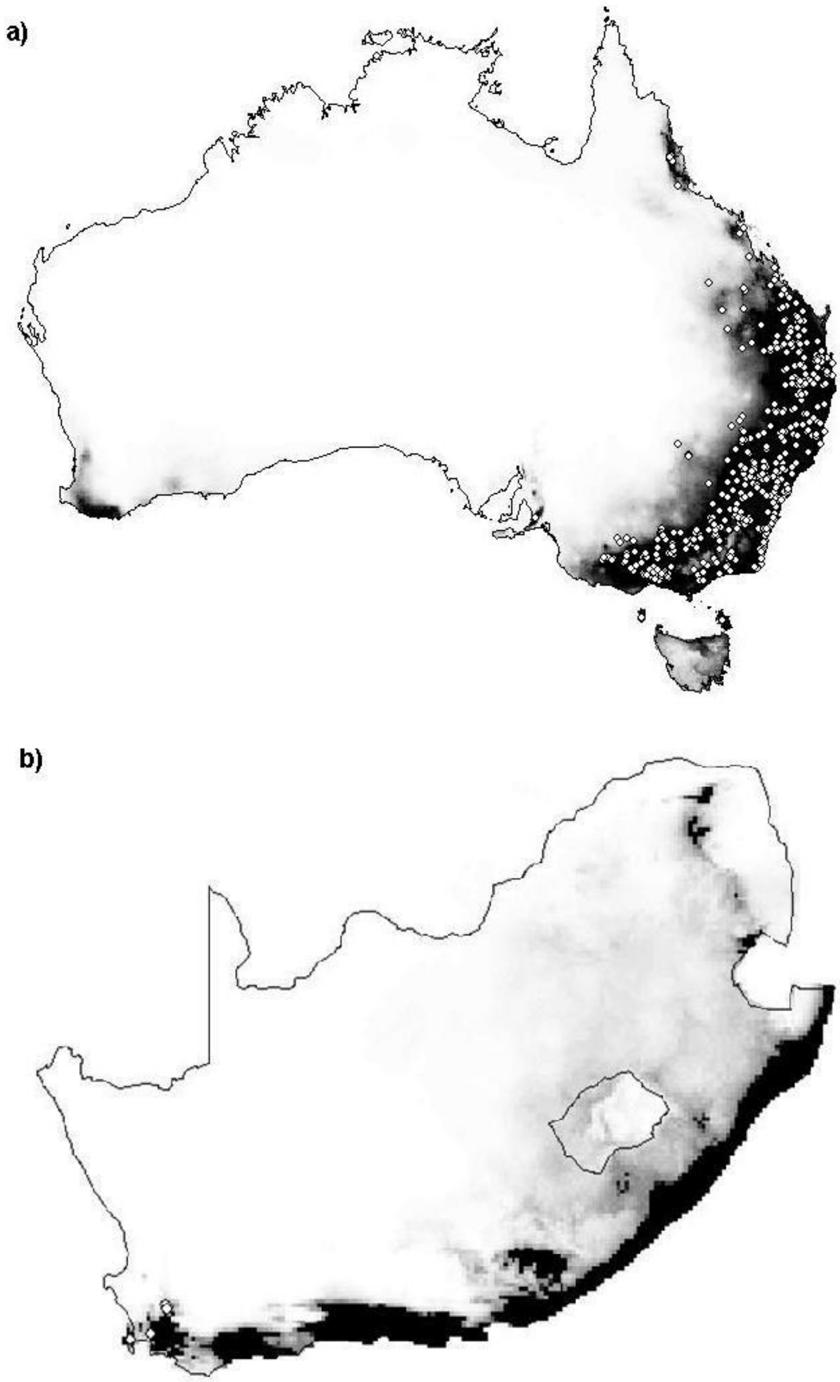


Fig. 3.1.

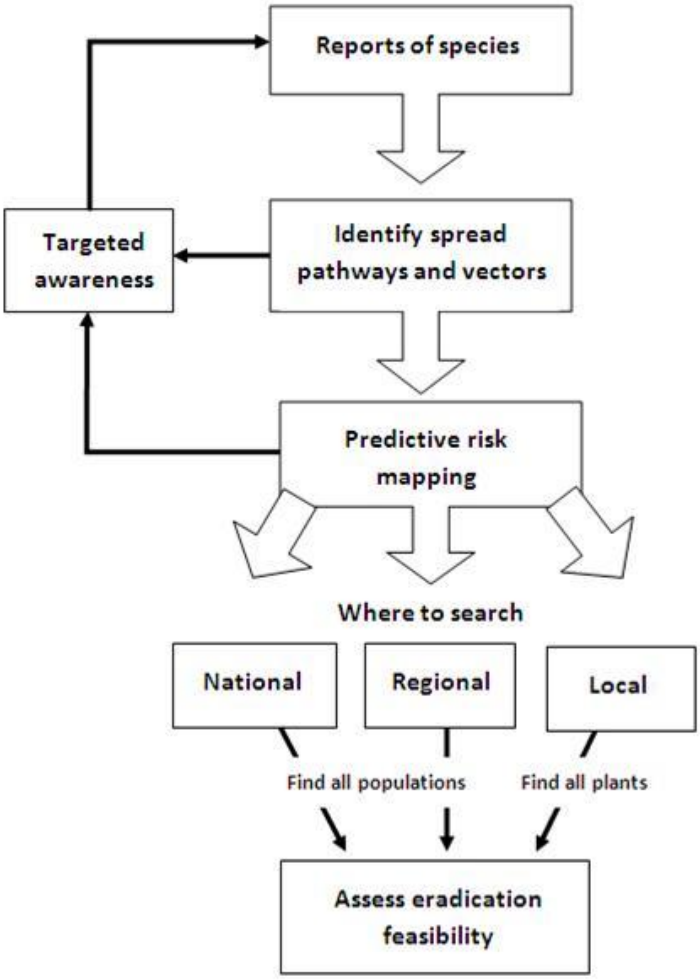

Reports of species
Identify spread pathways and vectors
Targeted awareness
Predictive risk mapping
Where to search
National
Regional
Local
Find all populations
Find all plants
Assess eradication feasibility

Fig. 3.2.

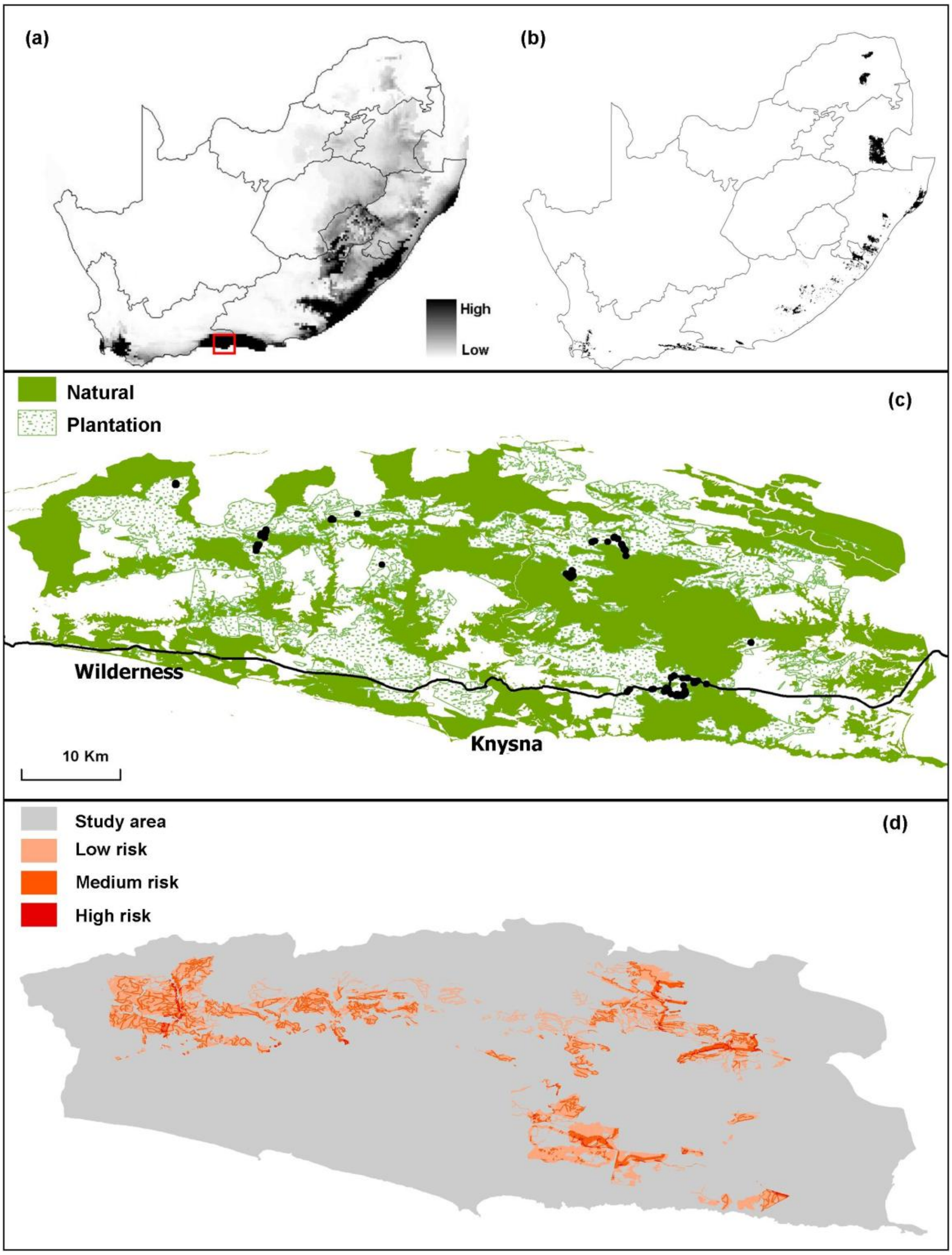

(a)
High
Low
(b)
Natural
Plantation
(c)
Wilderness
Knysna
10 Km
Study area
Low risk
Medium risk
High risk
(d)

Fig. 3.3.

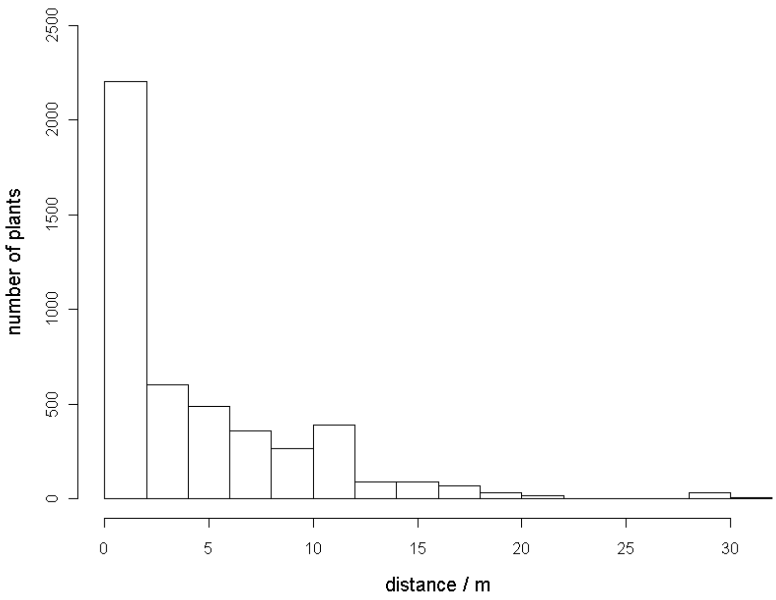

number of plants
0
500
1000
1500
2000
2500
0
5
10
15
20
25
30
distance / m

Fig. 3.4.

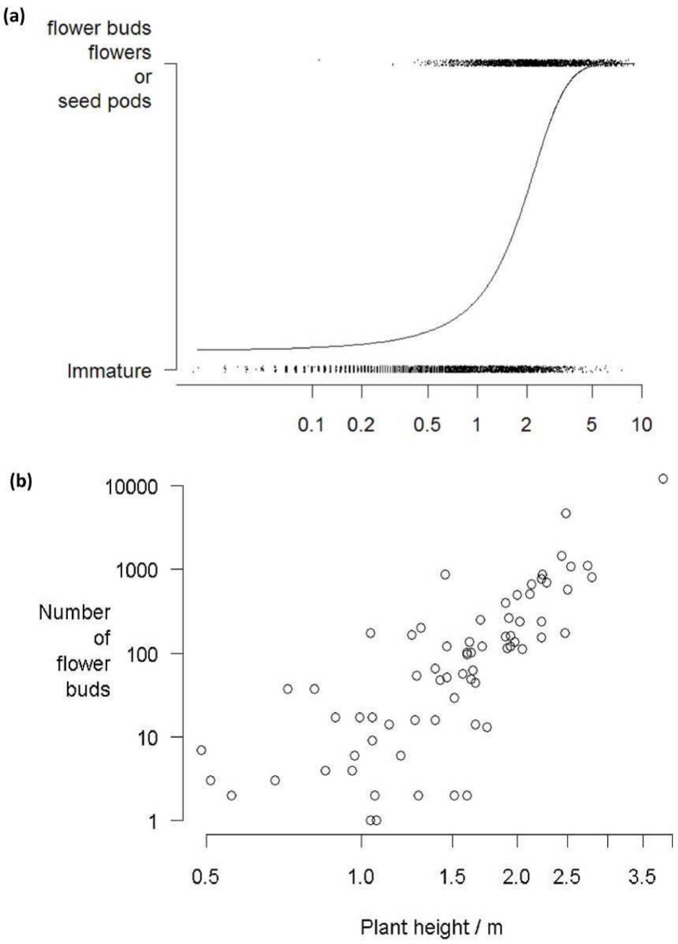
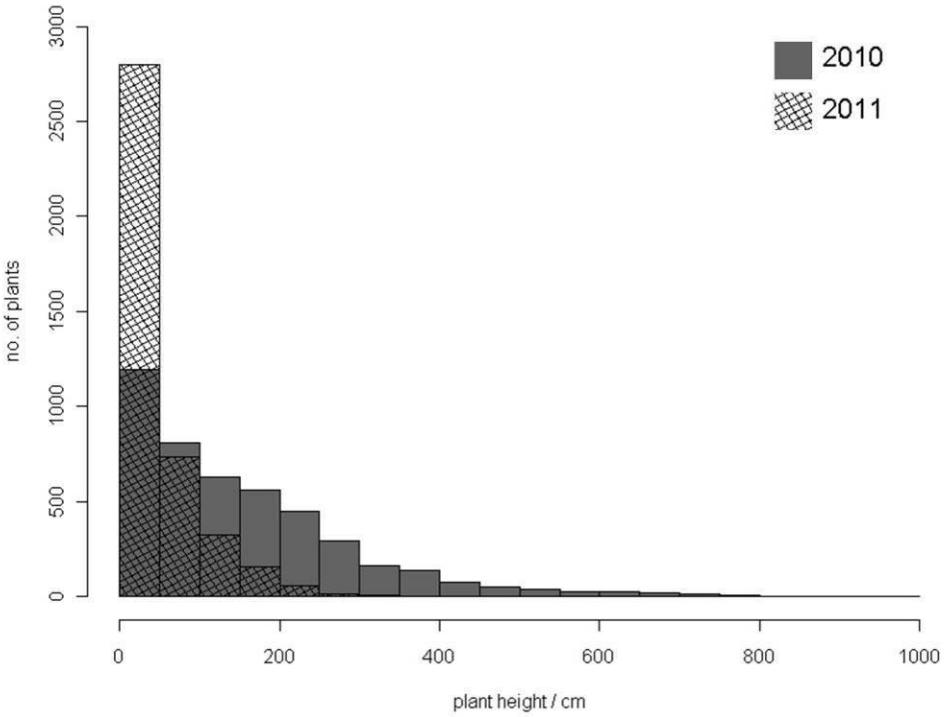

(a)
flower buds
flowers
or
seed pods
Immature
0.1
0.2
0.5
1
2
5
10
(b)
10000
1000
Number
of
flower
buds
100
10
1
0.5
1.0
1.5
2.0
2.5
3.5
Plant height / m

Fig. 3.5.



2010
2011
no. of plants
0
500
1000
1500
2000
2500
3000
0
200
400
600
800
1000
plant height / cm

Fig. 3.6.

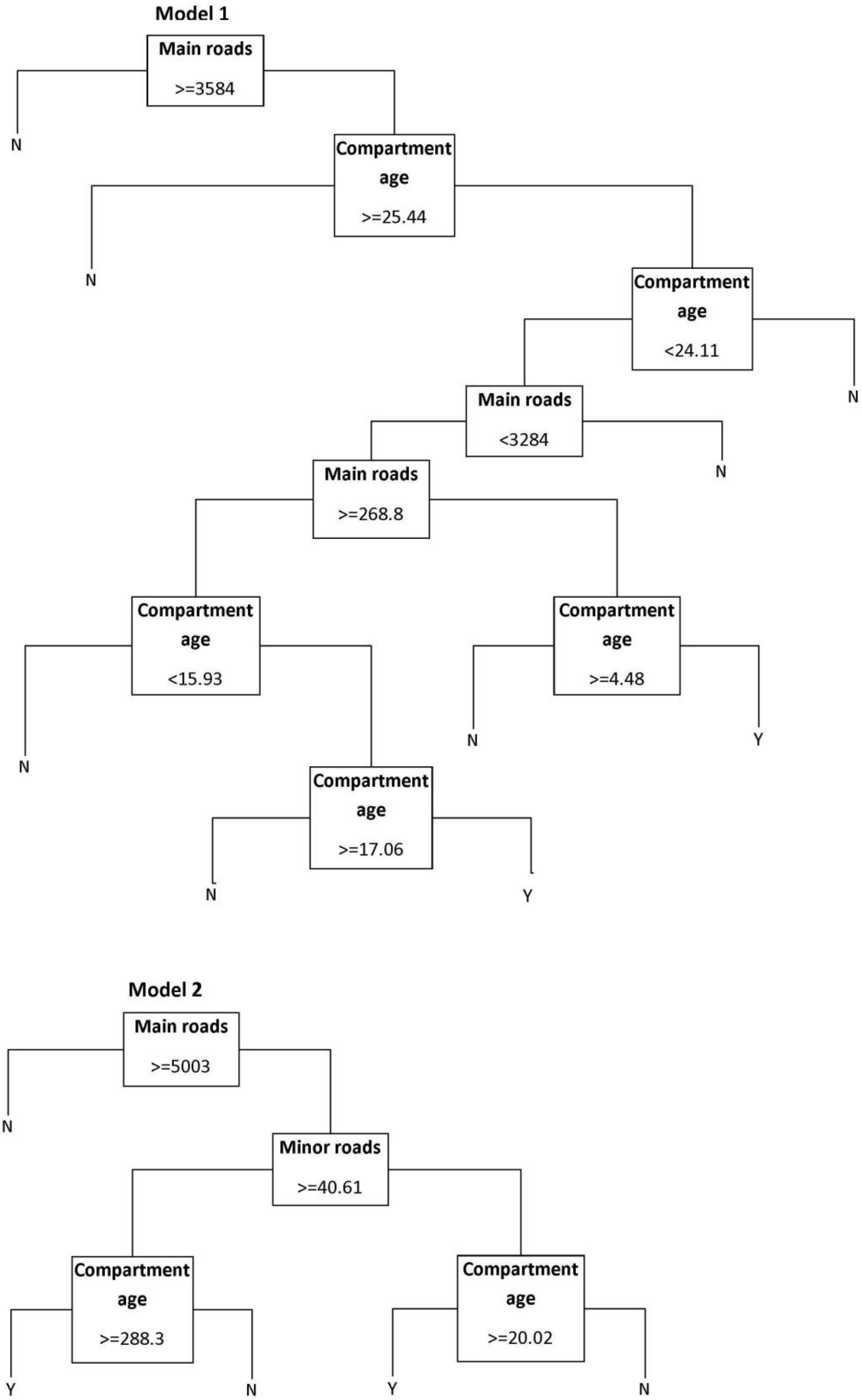

Model 1
Main roads
>=3584
N
Compartment
age
>=25.44
N
Compartment
age
<24.11
N
Main roads
<3284
N
Main roads
>=268.8
Compartment
age
<15.93
N
Compartment
age
>=4.48
N
Y
Compartment
age
>=17.06
N
Y
Model 2
Main roads
>=5003
N
Minor roads
>=40.61
Compartment
age
>=288.3
Y
N
Compartment
age
>=20.02
Y
N

Fig. 4.1.

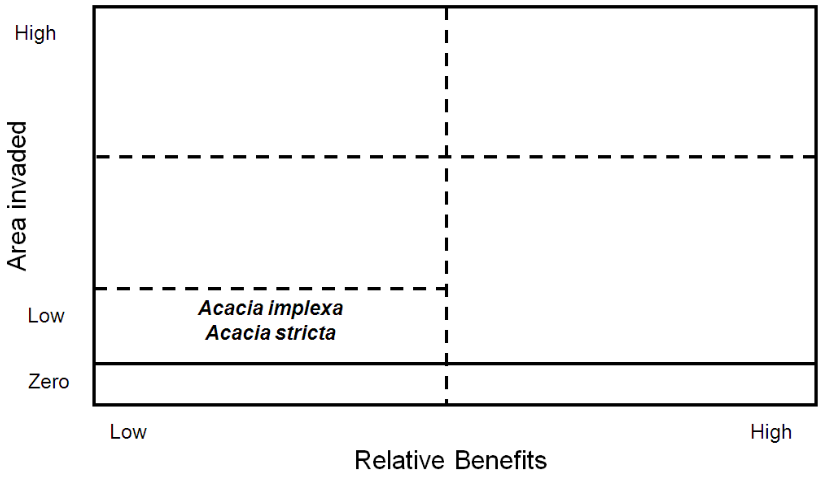

High
Area invaded
Low
Zero
Acacia implexa
Acacia stricta
Low
High
Relative Benefits

Fig. 4.2.

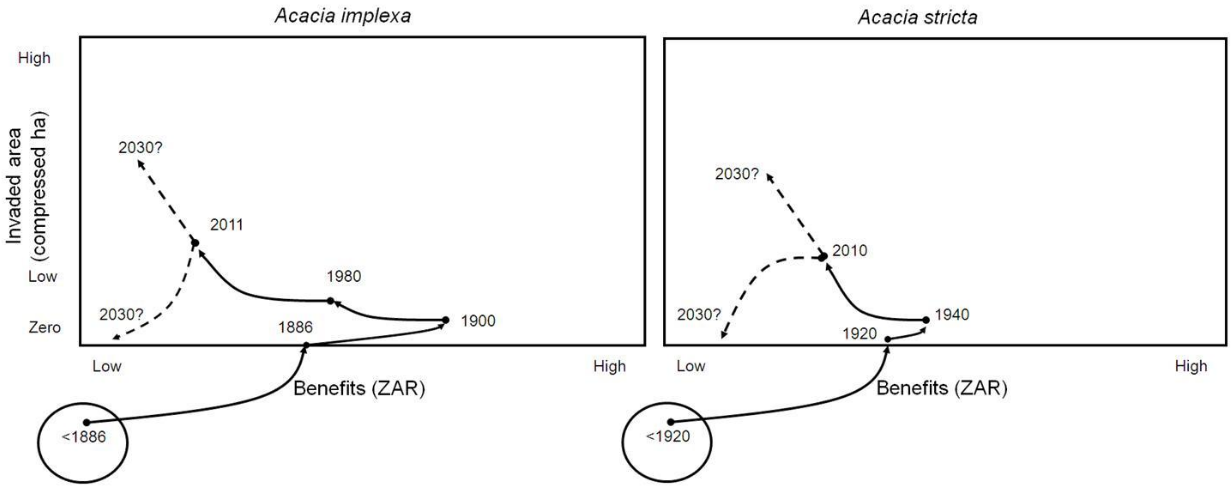

Acacia implexa
Acacia stricta
High
Invaded area (compressed ha)
Low
Zero
2030?
2011
1980
1900
1886
<1886
2030?
2010
1940
1920
<1920
Low
High
Benefits (ZAR)

Table 2.1. Summary of characteristics and management of *Acacia implexa* at three study sites in the Western Cape, South Africa

	Tokai	**Wolseley**	**Stellenbosch**
Location	S 34.057 E 18.422	S 33.446 E 19.152	S 33.938 E 18.846
Introduction date	1886	Unknown	Unknown
Reason for introduction	Forestry	Unknown	Unknown
History of invaded area	Forestry	Forestry	Forestry
Habitat invaded	Pine plantations	Fynbos/agricultural	Fynbos
Infestation area	200 ha	192 ha	208 ha
Number of plants	1639	3556	22 978
Present on riverbanks	Yes	Yes	Yes
Local authority	SANParks/Porter Estate	MTO Forestry[a]/Cape Nature/ private landowners	Stellenbosch municipality/ private landowners
Previous management	Yes	Yes	Yes
Present in SAPIA database[b] prior to project initiation	No	No	Yes

[a] MTO Forestry is a private forestry company which owns and manages pine plantations in the Western and Eastern Cape Provinces

[b] Southern African Plant Invaders Atlas (SAPIA) is a mapping project started in 1994 to gather information on invasive and naturalized plant species in South Africa, Lesotho and Swaziland (Henderson, 2001; Henderson, 2007)

Table 2.2. Growth strategies of *Acacia implexa* indicating the dominant reproductive methods of each population

	Seedling	Resprout	Sucker	Single plant	**Total**
Tokai	18	62	99	88	267
Wolseley	23	12	32	2	69
Stellenbosch	25	147	379	209	760

Table 2.3. Weed risk assessment of *Acacia implexa* following Pheloung et al (1999)

Question	Answer	Reference	Score
Is the species highly domesticated?	No. No uses besides ornamental in South Africa.		0
Species suited to South African climates	High	Fig. 4	2
Quality of climate match data (0-low; 1-intermediate; 2-high)	Intermediate		1
Broad climate suitability (environmental versatility)	Yes. Found in tropical, sub-tropical and temperate type climates.	1	1
Native or naturalised in regions with extended dry periods	Yes	1	1
Does the species have a history of repeated introductions outside its natural range?	No		0
Naturalised beyond native range	Yes. In South Africa and possibly China.	2	2
Garden/amenity/disturbance weed	Yes. Used for shade.	2	2
Weed of agriculture/horticulture/forestry	Yes	Pers. obs.	3
Environmental weed	Not known		?
Congeneric weed	Yes		2
Produces spines, thorns or burrs	No		0
Allelopathic	No		0
Parasitic	No		0
Unpalatable to grazing animals	No		-1
Toxic to animals	No		0
Host for recognised pests and pathogens	Not known		?
Causes allergies or is otherwise toxic to humans	Not known		?
Creates a fire hazard in natural ecosystems	Not known		?
Is a shade tolerant plant at some stage of its life cycle	No		0
Grows on infertile soils	Yes		1
Climbing or smothering growth habit	No		0
Forms dense thickets	Yes	Pers. obs.	1
Aquatic	No		0
Grass	No		0
Nitrogen fixing woody plant	Yes	3	1
Geophyte	No		0
Evidence of substantial reproductive failure in native habitat	No		0
Produces viable seed	Yes	4,5	1
Hybridises naturally	Yes. Possibly with *A. trinervata.*	6	1
Self-fertilisation	No	7	-1
Requires specialist pollinators	No	8	-1
Reproduction by vegetative propagation	Yes	8, Pers. obs	1
Minimum generative time (years)	1 year	4,9	1
Propagules likely to be dispersed unintentionally	No		-1
Propagules dispersed intentionally by people	No		-1
Propagules likely to disperse as a produce contaminant	No		-1
Propagules adapted to wind dispersal	No		-1
Propagules buoyant	No. Buoyant seeds are not viable.	4	-1
Propagules bird dispersed	Yes. In Australia.	10	1
Propagules dispersed by other animals (externally)	No		-1
Propagules dispersed by other animals (internally)	Yes. Ant dispersal (in Australia)	8	1

Prolific seed production	Yes	5, Pers. obs.	1
Evidence that a persistent propagule bank is formed (>1 yr)	Not known, but possible.		?
Well controlled by herbicides	No	Pers. obs.	1
Tolerates or benefits from mutilation, cultivation or fire	Yes	8, Pers. obs	1
Effective natural enemies present in Australia	Not known		?

[1] Maslin and McDonald, 2004; [2] Henderson, 2001; [3] Cole et al., 1996; [4] Schortemeyer et al., 2002; [5] Ralph, 2003; [6] Pieterse, 1998; [7] Maslin, 2001; [8] Kenrick, 2003; [9] Earl et al., 2001; [10] Stanley and Lill, 2002

Table 2.4: Management history and estimated costs of clearing *Acacia implexa* at study sites

Site	Treatment	Herbicide usage	Chemical control		Labour		Total cost (ZAR)
			Area (ha)	Cost (R/ha)	Time (person-days)	Cost (R/ person-day)	
Stellenbosch	Cut stump	Mamba (glyphosate 360 g/L) 5%	208	R 1 600	2617	R 140	R 699 180
	Foliar spray	Mamba (glyphosate 360 g/L) 2%					
Tokai	Cut stump Frilling	Lumberjack (triclopyr amine salt 360 g/L) 3%	192	R 100	102	R 140	R 33 480
	Foliar spray	Garlon (triclopyr butoxy ethyl ester 480 g/L) 3%					
Wolseley	Cut stump	Confront (clopyralid / triclopyr 90/270 g/L) 3%	200	R 210	146	R 140	R 62 440
	Foliar spray	Garlon (triclopyr butoxy ethyl ester 480 g/L) 3%					
							R 795 100

Table 3.1. Point selection and spatial resolution of models used to predict the occurrence of *Acacia stricta*.

	Model 1	**Model 2**	**Model 3**
Spatial resolution	Point data (resolution of ~10 m as determined by GPS)	Data converted to presences at resolution of 500 m	One data point per population converted to 9 presences (resolution of ~10 m as determined by GPS)
Sample size	2000 data points	90 data points	1009 data points
	(1000 presence/1000 absence)	(45 presence/ 45 absence)	(9 presence/ 1000 absence)

Table 3.2. Summary of variables used in generalised linear models predicting the occurrence of *Acacia stricta*

Variable	**Description**	**Values**
Land use	Current land use (factor)	Plantation, Urban, Conservation, Farm, Degraded, Alien transformed
Post-fire veld age	Number of years since last fire (bounded numeric)	years from 0 to a maximum of >100
Elevation	Metres above mean sea level	0 – 1184 m
Compartment age	No. of years since planting or clear felling of plantation compartments	0 - 89 yr
Distance from minor roads	Gradable roads, hiking trails, plantation access roads	0 – 1462 m
Distance from main roads	National routes, main roads, secondary roads, urban streets	0 – 9588 m

Table 4.1. Summary of invasions by *Acacia implexa* and *A. stricta* compared with *A. paradoxa* which is currently being targeted for eradication

	Acacia implexa	***Acacia stricta***	***Acacia paradoxa***
No. of populations	3	9	1
Total invaded area	600 ha	110 ha	295 ha
Age at reproduction	Probably 2 years	1 year	1 year
Seed production	High	High	High
Seed viability	59%	6%	97%
Seed banks	Unknown, but likely small	up to 1000 seeds m^{-2}	up to 1000 seeds m^{-2}
Resprouting	High	Low	Low