Urban plant invasions: how to classify, prioritize and manage invasive alien plants; Cape Town as a case study
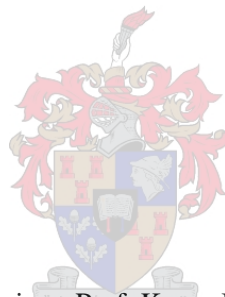
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Declaration

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March 2021

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Abstract

Urban areas are considered hotspots for invasions, with human activity being a major source of introduction and causing the dispersal of invasive alien plant species (IAPs). This makes natural and semi natural areas within and surrounding cities particularly susceptible to invasion. Cape Town is located within the Cape Floristic Region (CFR) which is considered a biodiversity hotspot; furthermore, this region has been identified as a UNESCO world heritage site. This biologically diverse area – which has some of the most recognizable and unique plant communities in the world – is therefore susceptible to invasion and is considered one of the most threatened biodiversity hotspots. IAPs pose a significant ecological and economic threat, impacting biodiversity and ecosystem services. Cape Town's natural areas are also fragmented due to urbanization, where low lying areas are often developed and areas at higher altitudes are set aside for conservation purposes (such at the Table Mountain National Park).

I reviewed the nature of urban invasions and the management strategies currently being employed to mitigate the impacts of IAPs within urban areas. I aimed to determine whether IAP richness and abundance is highest within natural areas within (or adjacent to) urban areas compared to peri-urban areas and rural areas and present management strategies to maintain ecological diversity throughout the city of Cape Town's biological network of natural and semi-natural areas. I present a classification protocol developed from a literature review based on the feasibility of IAP eradication and grouped widespread IAPs, emerging IAPs and potential eradication targets into different management categories. In addition to this I adopted and tested a model for prioritization of emerging IAPs (known as early detection and rapid response [EDRR]) in the City of Cape Town (CCT) Metropolitan area. Finally, I focused on the emerging invader *Hypericum canariense*, and use lessons from earlier chapters to discuss feasibility of eradication of this species.

I found that IAP species abundance was highest at the urban wildland interface (the peri-urban zone) between urban areas and rural areas. I discovered that disturbance in the natural and semi natural areas at the urban wildland interface plays a significant role in the invasion of IAPs into natural areas. In my classification protocol I identified that the area of infestation and the IAP species rate of spread are key indicators in classifying IAPs as widespread or emerging IAPs and for identifying potential eradication targets, noting that the three main management strategies for IAPS were mitigation, containment and eradication. Furthermore, the main criteria used in prioritizing emerging species in the Cape Town metropolitan area are "spread, negative impacts, ease of control and invasive potential". I discovered that the emerging invader, *H. canariense* is feasible for eradication and that it would cost approximately ZAR57000 over a 10 year period to eradicate this species.

I conclude that urban areas are important for effective IAP management and must not be disregarded or underestimate the role they play in the dissemination of IAPs into surrounding natural areas. The Peri-urban zone should be the focus of future management to ensure healthy biological networks.

Conservationists and land managers should classify and prioritize species for management efforts instead of tackling species arbitrarily when funding is limited. Emerging species should be investigated in terms of their feasibility of eradication before undertaking potentially lengthy and costly eradication program.

Opsomming

Stedelike gebiede word beskou as brandpunte vir plant indringing met menslike aktiwiteite as oorsaak en bron van bekendstelling en verspreiding van uitheemse indringer plant spesies. (UIPS) Dit maak natuurlike en semi-natuurlike areas binne stede en hulle omliggende areas besonder vatbaar vir indringing. Kaapstad is gelee in die Kaapse Floristiese Ryk, wat beskou word as n biodiverse brandpunt; verder, dit is aangewys as n Unesco wereld erfenis gebied. Hierdie biologiese diverse gebied, met van die beduidendste en unieke plantgemeenskappe in die wereld, is vatbaar vir indringing en word beskou as een van die mees bedreigde biodiverse brandpunte. UIPS hou n beduidende ekonomiese en ekologiese bedreiging in met n gevolglike impak op biodiversiteit en ekosisteemdienste.. Kaapstad se natuurlike gebiede ook gefragmenteer as gevolg van verstedeliking, met laagliggende gebiede dikwels meer ontwikkel en hoerliggende areas tersyde gestel vir bewaringsdoeleindes (soos in die geval van die Tafelberg Nasionale Park).

Ek het die aard van stedelike indringing nagegaan en die bestuurstrategieë t.o.v. die impak van UIPS tans in gebruik geëvalueer. My doel was om te bepaal of UIPS werklik meer oorvloedig voorkom in natuurlike gebiede wat grens aan stedelike gebiede en om bestuurstrategieë te ontwikkel om ekologiese diversiviteit in stand te hou in die biologiese netwerk van natuurlike en semi-natuurlike gebiede in die stasd Kaapstad. Hiermee bied ek n klassifikasie protokol aan wat ontwikkel is uit n literatuuroorsig en gebaseer op die moontlikheid om UIPS uit te wis en die wydversreide UIPS te groepeer asook om nuwe (opkomende) UIPS en potensiële uitwissingsteikens in verskillende bestuurstrategieë te kategoriseer. Daarbenewens het ek n model vir die prioritisering van opkomende UIPS (vroe opsporing en spoedige reaksie strategie) vir die Metropolitaanse gebiede van Kaapstad aangeneem en getoets. Vir hierdie navorsing het ek gefokus op die opkomende indringer *Hypericum canariense* en gebruik gemaak van lesse geleer uit die voorafgaande hoofstukke om die moontlikheid van uitwissing van hierdie spesie te bespreek.

Ek het bevind dat UIPS se voorkoms (volopheid) die hoogste was by die stedelike wildland-koppelvlak (peri-stedelike sone). Ek het vasgestel dat versteuring in die natuurlike en semi-natuurlike gebiede by die peri-stedelike sone n beduidende rol speel in die indringing van UIPS na natuurlike areas. (In my klassifikasie protokol demonstreer ek dat die gebied van besmetting en die tempovan die UIPS se indringing kritiese aanwysers is t.o.v. die klassifikasie van UIPS as opkomend of wydversreid, en vir die identifisering van potensiële uitwissingsdoelwitte (daaroplettend dat die drie hoof bestuurstrategieë vir UIPS versagting, uitwissing en beperking is). Verder, die hoof kriteriums gebruik in die prioritisering van opkomende spesies in die Metropool van Kaapstad is verspreiding, negatiewe impak,hoe maklik beperking geïmplimenteer sal kan word en indringingspotensiaal. Ek het gevind dat die opkomende indringer, *H. canariense*, geskik is vir uitwissing en dat dit ongeveer ZAR 57000 oor n tien jaar periode sal kos om die spesie uit te wis.

Ek het tot die gevolgtrekking gekom dat stedelike gebiede 'n noodsaaklike rol speel in effektiewe UIPS beheer en nie onderskat moet word t.o.v. hulle aandeel in die verspreiding van UIPS na omliggende natuurlike gebiede nie. Die peri-stedelike sone behoort die fokus te wees vir effektiewe bestuurspraktyke om toekomstige gesonde biologiese netwerke te verseker. Natuurbewaarders en land bestuurders moet spesies klassifiseer en prioritiseer vir effektiewe bestuurstrategieë, eerder as om n spesies na willekeur aan te pak, terwyl fondse beperk is. Nuwe, opkomende spesies moet ondersoek en geëvalueer word t.o.v. die moontlikheid van hulle effektiewe uitwissing voordat potensiële duur en uitgerekte uitwissing programme geïmplimenteer word.

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Chapter one: Introducing urban invasions and their management.

This chapter aims to provide a broad and concise overview of invasions in urban areas, considering cities as hotspots for invasions. In it, I review the different management approaches (i.e. mitigation, containment and eradication) for invasive species (i.e. widespread, emerging and Early Detection and Rapid Response eradication targets) in urban areas; the current and historic proposed frameworks for the feasibility of eradication of invasive alien plants (IAPs); urbanisation and the role that it has on native biodiversity; IAPs in urban areas and the effects that they have on native plant species in urban areas; and the potential pathways of introduction and movement of IAPs into and around urban areas.

Cities are considered hotspots for invasions (Holmes et al., 2018; Palma et al., 2016; Kowarik, 2011; Kowarik et al., 2013; van Wilgen et al., 2020) with most of the IAPS being introduced through tourism and the trade of horticultural plants (Dehnen-Schmutz et al., 2007). Complex dispersal pathways and vectors within urban areas promotes the dissemination of propagules within urban areas and into the surrounding natural and semi natural areas (Alston and Richardson 2006; McLean et al., 2017) As such there is an increased likelihood of newly introduced species (and potential eradication targets) in urban areas due to human-dominated and highly modified habitat (van Wilgen et al., 2020). The differences between urban areas and rural areas means that IAPs may need to be managed differently in urban areas in comparison to rural areas.

Mankind often brings about land transformation through the utilization of resources in the landscape which can lead to changes in abundance or the local extinction of native plant species (Vitousek et al., 1997). Urbanization is a case in point. Urbanization as a major global trend profoundly changes biodiversity patterns and leads to the homogenization of urban biota due to expanding ranges of exotic species and declining native species (Trentanovi et al., 2013). Urban environments (or ecosystems) are areas in which mankind lives at high densities and where infrastructure for housing, business and transport dominates the land surface (Pickett et al., 2011). Urbanization is a significant aspect of global land use transformation (Pickett et al., 2001). With increasing urbanisation, the importance of cities for the conservation of biodiversity is growing (Kowarik, 2011). Urbanisation endangers more species than any other human activity, resulting in environments with significantly altered compositional heterogeneity, land use complexity and patterns of ecological fragmentation (Anderson, 2006). More than 50% (increased from 30% half a century ago) of the global human population now lives in urban areas and this number is currently increasing by 76 million people a year (Pickett et al., 2011). As transformed landscapes become more wide-ranging, and the remaining fragments of native vegetation become more isolated, the importance of the remaining fragments becomes increasingly important for conservation (Dures and Cumming, 2010).

Urban expansion may cause the destruction of natural habitat fragments with high conservation value adjacent to urban areas (Hansen et al., 2005). Within cities, species assemblages are drastically altered because of intensive urban land use associated with habitat fragmentation and changes in ecosystem functioning which leads to a decline in native habitat species (Knapp and Khun, 2012; Duncan et al., 2011). Landscape level transformation induced by sprawling urban growth is regarded as a major threat to biodiversity (Hansen et al., 2005) where the increase in population density in urban areas results in development of infrastructure and the subsequent transformation of natural areas (Lubbe et al., 2011).

Natural and semi-natural areas are threatened by a variety of edge effects and are particularly susceptible to invasion (Alston and Richardson, 2006). Plant invasions can further exacerbate negative ecological impacts by promoting disturbance (Gulezian and Nyberg, 2010) and homogenising vegetation in urban areas (Quin and Ricklefs, 2006; Kuhn and Klotz, 2006). Management of these invasion foci may be the best method for the control of IAPs (Hulme, 2009).

Reduced fragment or patch quality is intrinsic to urban ecology as anthropogenic disturbance is likely to cause erosion, trampling, invasion and pollution and other human induced disturbances (McKinney, 2002). Urbanization has important effects in regional landscapes as cities are no longer compact spherical aggregations but rather they proliferate in fractured configurations (Picket et al., 2001) and are often located within biodiversity hotspots (Kowarik, 2011). Thus, natural areas are becoming increasingly fragmented and embedded within the urban matrix (Alston and Richardson, 2006). This extraordinary growth and concentration of human populations in urban areas brings about increased conflicts with conservation aims (Kuhn et al., 2004).

The process of urbanization is widely recognized as to facilitate the introduction, spread and impacts of alien species (Pysek, 1998). However, surprising little studies have been undertaken to show how IAPs spread through South African urban ecosystems (Gaertner et al., 2017; van Wilgen et al., 2020). The increase in biological invasions strongly contribute to the alteration and homogenisation of biodiversity globally (Lososova et al., 2016). Cities act as considerable points of entry and are responsible for the secondary release of introduced IAPs (Kowarik, 2011) with traffic, trade and horticulture as the most prolific pathways of dispersal (Dehnen-Schmutz et al., 2007; McLean et al., 2017). Furthermore, urban centres are often associated with riparian and transport corridors. In addition to this, storm water infrastructure, rivers and transport infrastructure (von der Lippe and Kowarik, 2007) are known to facilitate the spread of IAPs over significant distances. Historically, IAPs introduced into urban areas during the colonial period of South Africa were to provide timber and fuel from forestry activities and for dune stabilisation (Richardson et al., 2003). In addition to this, early European settlers wanted to recreate European gardens which lead to the introduction and subsequent of IAPs into local culture (Davoren et al., 2016). These initial introductions of IAPS had a significant impacts on natural and semi-natural areas (van Wilgen et al., 2020)

It has been shown that IAP species make up about 40% of the total flora in central European cities (Lososova et al., 2012). Two major mechanisms for the establishment of IAPs in urban areas are disturbance associated with an increase in soil nutrients (Lonsdale, 1999) and alien propagule pressure from urban gardens and parks (Alston and Richardson, 2006; Dean and Milton, 2019; Dures and Cumming, 2010; Trimble and Aard, 2014). As a result, urban areas comprise a sizeable pool of alien plant species (Botham et al., 2009).

Several management options are employed by land managers and decision makers to deal with the threat of biological invasions. These management options include prevention (risk assessment), early detection and rapid response, eradication, containment and various forms of mitigation (Pysek and Richardson, 2010). A study by Blackburn et al. (2011) proposed a unified framework (adapted from Richardson et al., 2000) for human mediated biological invasions. This framework identified that the invasion process can be divided into a range of stages; in each stage certain barriers (geographical, cultivation, survival, reproductive, dispersal and environmental) need to be overcome for a species to pass onto the next stage of invasion. The framework proposes that at each stage or series of stages, a different management approach should be applied, namely prevention, eradication, containment and mitigation. However, management of IAPs in urban areas has shown to differ substantially in different parts of the world. In some urban areas, limited funding is directed towards the prioritization of urban greenspaces, whereas in other areas limited funds is funnelled into "environmental issues" or other priorities which have socio-political alignment (Ilrich et al., 2017; van Wilgen et al., 2020). The nature of urban fabric lends to wards complex land-tenure patterns which may impede or obscure the coordination of management activities (Gaston et al., 2013). A large number of land owners and land managers often means divergent policies and practices for managing IAPs with a strong possibility of conflicts of interest arising (Gaertner et al., 2016).

It is important to recognise a combination of management strategies may need to be implemented to solve the invasive species problem (Westbrooks, 2004). A number of control methods (mechanical control, chemical control, and biological control) are available to land managers and conservationists for eradicating, containing or mitigating the impacts of harmful IAPs (Rejmanek and Pitcairn, 2002). In this thesis I unpack three of these main management approaches, namely: eradication, containment and mitigation.

For IAPS with high potential threats (potential for high ecological and economic impact), eradication at an early stage of invasion is favoured because other management alternatives of species that have spread widely (i.e. containment and or mitigation) require ongoing and indefinite investment of resources (Panetta, 2009). Myers et al., (1998) stated that eradication requires the elimination of every individual from an area in which recolonization is unlikely to occur. According to Panetta et al. (2015), the eradication of a species occurs through two main processes. The first process requires the

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elimination of the target species in both space and time (extirpation) and the second is the prevention of further geographical expansion (containment). Eradication efforts consist of a series of control treatments to an infestation over several years (Rejmanek and Pitcairn, 2002, van Wilgen 2011). It is important that all reproductive individuals within an infestation must be susceptible to control measures and must be extirpated at a faster rate than their rate of increase (Panetta, 2009) and reinvasion must be avoided (Dodd et al., 2015). It is therefore important that infestations must be inspected regularly and effectively by search and control efforts until the soil seedbank of the species to be fully depleted. Seed production must be virtually eliminated from infestations if eradication is to succeed (Panetta, 2007). Several IAP eradication programmes (with no specific focus on urban areas) have been attempted in South Africa with limited success thus far (van Wilgen et al., 2020). This is mostly because species-based eradication efforts were focussed on widespread species across the country including urban areas. This perpetuated a plethora of elements (e.g. the delimitation of infestation, adequate funding to see eradication programmes to completion) which negate the basic requirements to achieve eradication (Wilson et al., 2017).

Containment (also referred to as maintenance management) involves controlling IAP species to the point where the damage it causes is bearable. This is the usual approach adopted, particularly for IAPs that are widely distributed (Panetta, 2015). According to van Wilgen et al. (2011) containment of a species is appropriate for species for which eradication is not feasible but where there is still a significant risk of expansion to new areas and associated impacts in those areas. The control of IAP species before they reproduce is crucial if eradication of an IAP species ins the management target for that species (Panetta, 2007).

Mitigation (impact reduction) is considered the only feasible management action for widespread invasive species (van Wilgen et al., 2011). For the mitigation management strategy to be successful, a basic understanding of the ecosystem and the related biological impacts of the IAP of the invaded ecosystems components and processes is required (Pearson and Ortega, 2009) as well as an improved understanding of the interacting factors which bring about the impacts (le Maitre et al., 2011). In some cases, mitigation can include eradication and containment of an IAP with the goal being to prevent the spread of an IAP (Perrings, 2002). Furthermore, mitigation can also refer to the reduction of the rate of introduction of IAPs into a bio-geographical area (Von der Lippe and Kowarik, 2007).

The eradication of invasive alien animals has been well documented over the latter decades of the 20th century (Panetta, 2009). However, the feasibility of plant eradication has received less attention in comparison to the feasibility of eradication of animals (Panetta and Timmons, 2004). Very few successful or unsuccessful eradication programs have been documented (Panetta, 2009; Panetta, 2007). According to Panetta, (2015) plants differ from animals as eradication targets because plants have a resting stage (seed stage) whereas animals do not. In addition to this, plants must be removed or

controlled from wherever they occur and cannot be trapped or baited in the same manner that animals can. This resting stage makes the eradication of plants particularly challenging.

Rejamnek and Pitcairn (2002) conducted one of the first comprehensive assessments on weed eradication projects, which investigated 18 species and 53 infestations which were targeted between 1972 and 2000 across the state of California (USA). The study showed that infestations of IAPs or weedy species which are larger than 1000ha in size should not be attempted to be eradicated but rather be contained to avoid further environmental and economic impact. The largest known documented success of IAP eradication is that described by Dodd, (2004). This program targeted Kochia (*Bassia scoparia*) which was an intentionally introduced species in Western Australia. The program successfully extirpated a net area (treated area) of 2480ha. The reasons behind the success of this eradication program were due to the fact that very little resources had to be invested into the delimitation of the infestation because it was an intentionally introduced species and as such, all sites of introduction were known (Panetta, 2015).

Generally, the success or practical feasibility of an eradication program is determined by operational, biological, economic and socio-political factors (Panetta, 2009). Theoretical models that calculate the total effort for eradication feasibility are useful in determining the feasibility of eradication of IAPs (Panetta, 2015). Panetta (2009) argued that eradication feasibility must be viewed in context of the amount of resources that can be invested in a eradication program and that this economic factor must be considered to determine the feasibility of eradication even though the impacts of IAPs are difficult to quantify (Panetta and Timmons, 2004). Cacho et al. (2008) proposed a decision model to determine if and when immediate eradication of an IAP should be attempted, or more generally whether control should be considered at all. The results of this study showed that in the absence of a budget constraint, it may be desirable to eradicate invasions from areas as large as 8000ha. However, when posed with budget constraints feasible eradicable areas are less than 1000ha.

The effort (investment) required to achieve weed eradication consists of the effort required to delimit IAP infestations and search and control effort required to prevent the reproduction (re-establishment) of the IAP until it is fully extirpated (Panetta, 2009). If eradication can be achieved, it may be the most cost-effective form of management, but should only be attempted if eradication is deemed a feasible management option (Panetta et al., 2011). The study illustrates the effects of detectability and search effort on the duration of an eradication program and determines that search speed, kill efficiency, germination rate and seed longevity have the greatest effect on the duration of an eradication program at a given level of detectability and search time. Effort required for eradication is an example of an operational factor (which is intrinsic to considering the economic factor).

A study by Panetta (2015) proposed a new algorithm for assessing eradication feasibility. The study produced eight "eradication syndromes" based upon time to maturation, seedbank persistence and the

membership of one or two contrasting dispersal functional groups. Three alternative modes of dispersal are proposed for the eight eradication syndromes, namely: short distance dispersal (SDD), long distance dispersal (LDD) and human mediated dispersal (HMD). Intuitively one would expect that HMD would play a far more significant role in the dispersal of IAPs in urban areas in comparison to rural areas because of the greater density of people and the existence of infrastructure which may cause the spread of IAPs. In an urban ecosystem one would firstly expect that total infestation area of a target species might be considerably smaller and perhaps more disjointed due to the nature of the urban fabric of cities. If one examines the four different factors used to calculate impedance (logistical considerations, weed detectability, weed biological characteristics and control effectiveness), these factors may also vary between urban and rural areas. For example, accessibility to infestations areas under logistical considerations may be more problematic in urban areas due to private land ownership (lack of sit accessibility) and the use of IAPs as ornamentals (van Wilgen et al., 2020). According to this study, the syndrome with the lowest feasibility of eradication includes plants that have a short juvenile period and high seedbank persistence in combination with abiotic and human-mediated dispersal. The syndrome with the highest feasibility of eradication consists of species that have long juvenile period, low seedbank persistence and short dispersal distances (Panetta 2015).

To date, scientific studies that have been undertaken in urban areas have mostly focused on fragmentation or transformation of natural areas (Saunders et al., 1991; Rouget et al., 2003; Mcdonald et al., 2009; Meng et al., 2015) the role of disturbance or other anthropogenic impacts in urban areas (Alston & Richardson, 2006; Ehrenfeld, 2008), the dispersal of IAPs (Botham et al., 2009) and biotic homogenisation or differentiation (McKinney, 2006; Pauchard et al., 2006; Kuhn & Klotz, 2006; Lososova et al., 2012). However, only a few studies have exclusively compared invasions in urban areas with invasions in peri-urban and rural/semi-natural areas (but see Kuhn et al., 2006; Alston & Richardson, 2006). Certainly, there is a need for understanding invasions in urban areas in South Africa as most of the work done to date is focussed in European cities.

Despite the research that has already been undertaken, distributions of IAPs in urban areas are relatively unknown to both ecologists and urban planners (Gulezian & Nyberg, 2010, Lososova et al., 2012). Enhancing scientific knowledge of IAPs in urban areas is of paramount importance from a conservation perspective (Kowarik, 2011).

In addition to this, a number of nationwide prioritization protocols have been developed for IAPs in South Africa. The majority of these are spatial prioritizations with the exception of a few species-based prioritizations. One of the first species-based prioritizations to be undertaken in South Africa was that of Robertson et al (2003). This study prioritized 61 species based on five aspects (potential invasiveness, actual spatial extent, potential impacts, potential for control and conflicts of interest). Nel et al. (2004) prioritized species by widespread species (117 species categorised into groups based on geographical

range and abundance) and emerging species (84 species categorised into groups based on current propagule-pool size and potentially suitable habitat for invasion). To date, no species-based prioritizations have been undertaken specifically in urban areas in South Africa. In addition to this no prioritization protocols exist at a local (fine) scale (i.e. for the Cape Town Metropolitan area). As such, there is a need to develop prioritization protocols for urban areas.

Many authors have presented theoretical frameworks to determine the feasibility of eradication of IAPs (Panetta & Timmons, 2004; Cacho, 2006; Dodd, 2004; Panetta, 2009; Panetta, 2015) and a few studies aimed to determine the feasibility of eradication of single species (Kaplan et al., 2012, Zenni et al., 2009; Geerts et al., 2013). However, there are no studies which assess the feasibility of eradication of an urban invader. The assessment of feasibility of eradication will be compounded by different factors in urban areas when compared to peri-urban or rural areas. Some of these compounding factors include, but are not limited to, access to sites or private land where IAPs are present, conflicts of interest or buy in from the public, and the ease of control due to human mediated dispersal pathways.

Given this background, the following five chapters will (1) Provide an introduction to urban plant invasions, feasibility of eradication and current management options; (2) explore the differences in Invasive Alien Plant community composition between urban, peri-urban and rural areas (Using the City of Cape Town as a case Study; (3.1) categorize species (emerging, widespread and potential eradication targets) into management categories; (3.2) Prioritize Invasive Alien plants using a Multi-criteria decision model and finally (5) test the feasibility of eradication of the urban invader (*Hypericum canariense*). The final chapter (Chapter 5) summarises the findings of the thesis and provides recommendations for management. Figure 1.1 shows how the proposed outcomes for this thesis aims to address fundamental issues or the current gaps in the literature.

Figure 1: fundemental issues and the gaps in the liturature for this thesis as well as the thesis outcomes.
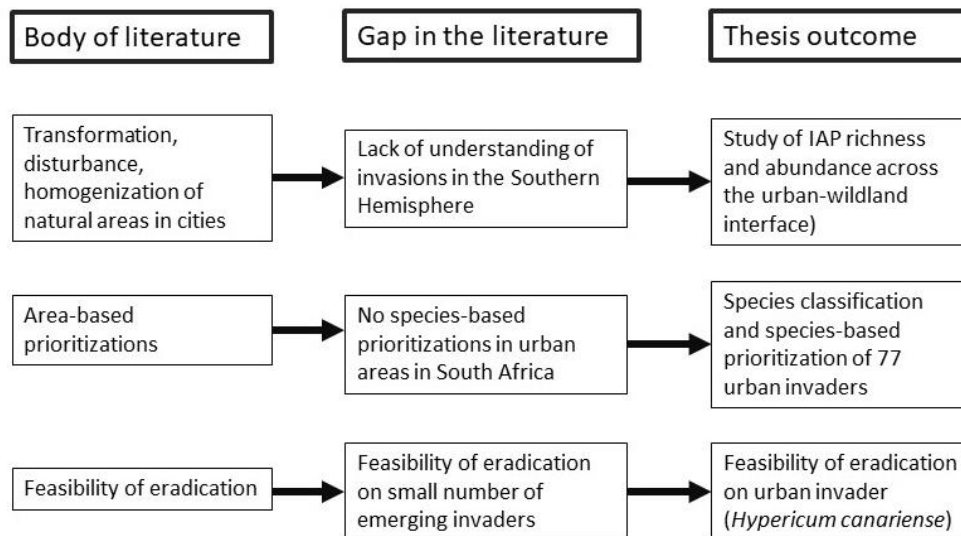
The data chapters are written as stand-alone papers. Chapter 2 is published as a book chapter along with my supervisors as co-authors:

Afonso, L., Esler, K.J., Geerts. S., Gaertner, M. (2020) Comparing invasive alien plant community composition between urban, peri-urban and rural areas; the City of Cape Town as a case study. Chapter 13 In: Verma, P., Singh, P., Singh, R., Raghubanshi, A.S. (eds) Urban Ecology: Emerging Patterns and Social-Ecological Systems Urban Ecology. Elsevier, pp 221-236. ISBN: 978-0-12-820730-7

This study was collectively conceived, I collected the data and analysed it, and my supervisors assisted in co-writing and providing detailed input into the drafts prior to submission as well as helping with data analysis.

Chapter two: Comparing invasive alien plant community composition between urban, peri-urban and rural areas; the city of Cape Town as a case study.[1]

2.1 Abstract

Urban areas are considered hotspots for invasions. Anthropogenic activity within these areas often facilitates the introduction and spread of invasive alien plants, with urban gardens acting as a significant source of propagules. This makes natural areas within and surrounding urban areas particularly vulnerable to invasion by alien plants. The City of Cape Town (CCT), South Africa provides an ideal opportunity to study plant invasions at the urban/wildland interface since urban and sub-urban areas contain a mosaic of natural areas embedded within the urban fabric. We hypothesised that invasive alien plant species richness and abundance would be significantly higher in urban areas and peri-urban areas compared to rural areas. We sampled plant species richness and abundance in fifteen sites – from the Cape Town Biological Network – across the urban-wildland interface (urban, peri-urban and rural). Plant species were grouped according to invasion pathways and adaptive strategies. There was no significant difference in invasive alien plant abundance between urban and rural habitat types, but invasive alien plant abundance was significantly higher in peri-urban habitat types compared to urban and rural habitat types. Species richness was similar between urban, rural and peri-urban habitats. There were equal proportions of competitors and ruderal species in urban areas, but a higher proportion of ruderal species in both the peri-urban and rural habitat types. Agricultural invasive alien plant species were more common throughout, compared to species originating from horticultural or forestry. To secure healthy ecological networks with the urban expansion of the City of Cape Town, management should focus on the peri-urban habitats.

2.2 Introduction

Urbanization influences biodiversity in many different ways (Collins et al., 2000; Pickett et al., 2001; Sukopp and Werner, 1983), e.g., by altering quality of air, water and soil (Gill, 1996; Willig and Walker, 1999); temperature regime; rainfall patterns (Landsberg, 1981; Sheperd et al., 2002), fragmenting and disturbing remaining habitat (Kowarik, 1990). Urban areas are defined as areas surrounded by city development and characterized by high-density human infrastructure such as residential areas, commercial buildings and transport infrastructure (National Geographic, 2016a) are also hotspots for invasions and act as key points of entry for introduced invasive alien plants (IAPs) (Kowarik, 2011). Trade and horticulture are the most prolific pathways of human-mediated dispersal (Dehnen-Schmutz et al., 2007; Dehnen-Schmutz, 2011; Mack et al., 2000; Pysek, 1998; Reichard and Hamilton, 1997; Vitousek, 1997). Consequently, large urban areas often contain a greater proportion of IAPs compared

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[1] Afonso, L., Esler, K.J., Geerts. S., Gaertner, M. (2020) Comparing invasive alien plant community composition between urban, peri-urban and rural areas; the City of Cape Town as a case study. Chapter 13 In: Verma, P., Singh, P., Singh, R., Raghubanshi, A.S. (eds) Urban Ecology: Emerging Patterns and Social-Ecological Systems Urban Ecology. Elsevier, pp 221-236. ISBN: 978-0-12-820730-7

to surrounding rural areas (Khun and Klotz, 2006; Pysek, 1998). The latter is defined as open swathes of land which have a low population density with agriculture being the primary economic activity (National Geographic, 2016b).

Disturbances associated with pulses of available nutrients (Davis and Thompson, 2000) and IAP propagule pressure (Lonsdale, 1999) have been put forward as two of the most significant factors which influence plant invasion in urban areas (Lososova et al., 2012). With increasing levels of human-mediated habitat disturbance, propagule pressure increases and so do the number of non-native plant species (Gaertner et al., 2016; Kowarik, 2011; Lososova et al., 2012). These invasions are often facilitated through disturbances, which are associated with roads, railways and riparian areas. Furthermore, in the urban/wildland interface humans are more likely to disturb natural habitats (natural areas or public open space which is undeveloped and sustains biodiversity) through trail construction, poor vegetation management, vegetation trampling or refuse dumping (Sullivan et al., 2005) and plant collecting (Alston and Richardson, 2006). These areas at the urban/wildland interface are also referred to as peri-urban areas and are at the interface or transition zone between urban and rural areas (Thapa and Murayama, 2008). Typically, they contain both urban and rural features (Allen, 2003).

Natural environments at the urban fringe also experience increased levels of air and water pollution (Struglia and Winter, 2016) and erosion or sedimentation through water runoff from the hard surfaces characteristic of urban areas (Whitford et al., 2001). Many of these factors interact and have a synergistic effect on one another. For example, disturbance leads to increased solar radiation on soils, higher temperatures, altered soil conditions, and may result in increased IAPs and a decrease in native plant diversity (Drayton and Primack, 1996; Parendes and Jones, 2000). Furthermore, parks and gardens act as an important source of propagules for IAPs at the urban fringe (Alston and Richardson, 2006; Anderson et al., 2014). High propagule pressure increases the chance of establishment, naturalization and invasion of IAPs (Lonsdale, 1999; Rouget and Richardson, 2003).

Since urban areas can also act as sources of IAP invasions into natural areas that surround them (Pysek, 1998), natural areas at the urban/wildland fringe are vulnerable to invasion because of their proximity to gardens (Marco et al., 2010; Bell et al., 2003). The abundance and richness of IAPs are known to be linked to the distance to the nearest urban area (Sullivan et al., 2005). Alston and Richardson (2006) found that the distance from putative source populations and human-mediated disturbance influence IAP richness in Newlands forest, South Africa. In contrast, rural areas contain few typical urban features and are strongly related to agricultural land uses or large open swaths of land (National Geographic, 2016a). Cilliers et al. (2008) discovered that IAP invasions penetrate deeper into native urban grasslands when compared to native rural grasslands.

Since urban areas are characterized by high levels of stress and disturbance compared to peri-urban and rural areas, one might expect that IAPs in urban areas may have different plant adaptive strategies to

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those found in peri-urban and rural areas. According to Grime (1977), stress (conditions which restrict production) and disturbance (the destruction of plant biomass) are two external limiting factors which affect plant biomass in any environment. As such, three plant adaptive strategies may occur, i.e., competitive plants (low stress and low disturbance); stress tolerant plants (low disturbance and high stress); and ruderal plants (high disturbance, low stress). The difference in habitat condition between urban areas and rural areas means that plant adaptive strategies and growth forms should differ between these environments. For example, due to the higher levels of nutrients and high levels of human-mediated disturbance in urban areas, one would expect ruderal species to be more prevalent in those areas.

Despite the plethora of research on IAPs, distributions of IAPs within urban areas and surrounding areas are relatively unknown (Gulezian and Nyberg, 2010; Lososova et al., 2012). By investigating the difference in invasive species composition between urban, peri-urban and rural areas, we aimed to explore the mechanisms which promote IAP establishment in urban and surrounding areas and to formulate best-practice management suggestions. We further explore the origin of IAPs (whether horticultural, agricultural or forestry) and their adaptive strategies (Grime, 1977) within urban and surrounding areas.

My study area is the City of Cape Town (CCT). Cape Town is located within the Cape Floristic Region (CFR) of South Africa and is considered a biodiversity hotspot since it has a high native species richness and high levels of endemism (Holmes et al., 2012; Myers et al., 2000). The CFR is also a recognized UNESCO world heritage site, being the smallest but most biologically diverse compared to the other plant kingdoms (Davison and Marshak, 2012). The region has a long history of human activity with resultant disturbance from farming practices, urban development and considerable recreational usage.

Despite the fact that South Africa has some of the most recognizable and unique plant communities in the world, it is also one of the most invaded and threatened (Bromilow, 2010). Many of the prominent IAPs found in Cape Town (acacias and pines) were introduced for multiple reasons. *Acacia saligna* and *A. cyclops* were introduced in the mid-19th century to stabilize sand dunes (Davison and Marshak, 2012). Pine species were introduced for the timber industry, remnants of which are still visible in Cape Town today (Anderson and O' Farrell, 2012). A number of these species have significant impacts on ecosystem services in natural, agricultural and transformed areas (Mangachena and Geerts, 2017; van Wilgen et al., 2008). For hundreds of years many agricultural IAPs were accidentally introduced with the sowing of crop seeds and these now persist as weeds on agricultural and natural land (Bromilow, 2010). Plants are also often intentionally introduced as ornamentals because of their attractive flowers, high growth and germination rate and tolerance to a wide range of environmental conditions, which often leads to a conflict of interests between conservationists, landowners and plant nursery owners (Foxcroft et al., 2008; Geerts et al., 2017; Novoa et al., 2018).

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To determine how natural areas within Cape Town are affected by IAPs, we selected sample sites within the Cape Town Biological Network (BioNet). The Cape Town BioNet consists of a series of interconnected protected areas known as Critical Biodiversity Areas (CBAs) (ranging from pristine habitats to more degraded, but highly threatened ecosystems) and Critical Ecological Support Areas (CESAs) (Davison and Marshak, 2012). The BioNet was developed by the City of Cape Town to identify vegetation remnants of high importance to avoid the loss of endangered and threatened flora to urban development. The Cape Town BioNet is, therefore, an intrinsic feature to the urban fabric of Cape Town. As such the BioNet offers a unique opportunity to study the abundance and composition of IAPs that invade natural areas from urban, peri-urban and rural land uses in the city of Cape Town municipal boundaries.

We aimed to investigate the composition (abundance and richness) of IAPs in urban, peri-urban and rural areas in the City of Cape Town, South Africa. More specifically we determined (1) if there is a difference between IAP species abundance and richness between urban, peri-urban and rural areas; (2) if habitat condition is responsible for promoting the establishment and facilitating invasion of IAPs into natural areas; (3) the reason for introduction and (4) the most prevalent plant adaptive strategies of IAPs in urban environments. Based on our findings, we aim to propose management strategies that focus on conservation at the urban-wildland interface. We hypothesised that invasive alien plant species richness and abundance would be significantly higher in urban areas and peri-urban areas compared to rural areas.

2.3 Methods

2.3.1 Study area

Our study area is located in the greater Tygerberg area of the City of Cape Town where urban, peri-urban and rural sites meet. The area has a Mediterranean climate characterized by winter rainfall and summer drought. The annual rainfall ranges between 100 and 400 mm per annum (Harris et al., 2010). All study sites lie within the BioNet (Davison and Marshak, 2012). All the sites that were selected occur within naturally vegetated areas adjacent to urban, peri-urban or rural habitat types. The urban sites were located within Tygerberg Nature Reserve, Durbanville Nature Reserve and Uitkamp Nature Reserve, which are all urban nature reserves. All sites at all three habitat types are regularly cleared of conspicuous IAPs by both the reserve management as well as the City of Cape Town Invasive Species Management Unit. Moreover, nature reserves are popular recreational areas in an urban matrix of wealthy suburbs with large and diverse garden.

The dominant vegetation type in our study area is Swartland Shale Renosterveld which can be found growing on fertile fine-grained soils in the Swartland and Boland areas of the Western Cape (Walton, 2006). Approximately 95% of this vegetation type has been transformed due to agricultural activities (i.e., vineyards and orchids) with only 0.5% of its original extent being formally protected (Cowen,

2013; Walton, 2006). Swartland Shale Renosterveld is therefore classified as critically endangered (Cowen, 2013). It contains low to tall narrowleaved shrubland growing in more arid areas.

2.3.2 Site Selection

For the selection of sites, different layers supplied by the South African National Biodiversity Institute (SANBI) Biodiversity GIS website (http://bgis.sanbi.org/[Biodiversity GIS, hereafter]) were used for analysis in Arc GIS 10.3.1. The City of Cape Town BioNet layer, as well as the Biodiversity GIS Vegetation map of 2012, was used in selecting sampling sites. The sites were selected by extracting the Swartland Shale Renosterveld (SSR) vegetation category from the VegMap 2012 (selected by attributes and exported as a layer) and selecting sampling sites (by location) for all BioNet sites (polygons) that intersected and overlapped with SSR vegetation type layer (Fig. 2.1). Five sites were randomly selected from the full list of VEGMap extracted sites for each habitat type (urban, peri-urban and rural). Each site selected was ground-truthed to ensure they fit the definition of the respective Urban, peri-urban and rural land use types. Sampling was done in one vegetation type (Swartland Shale Renosteveld) in a localized area in order to limit environmental variability.
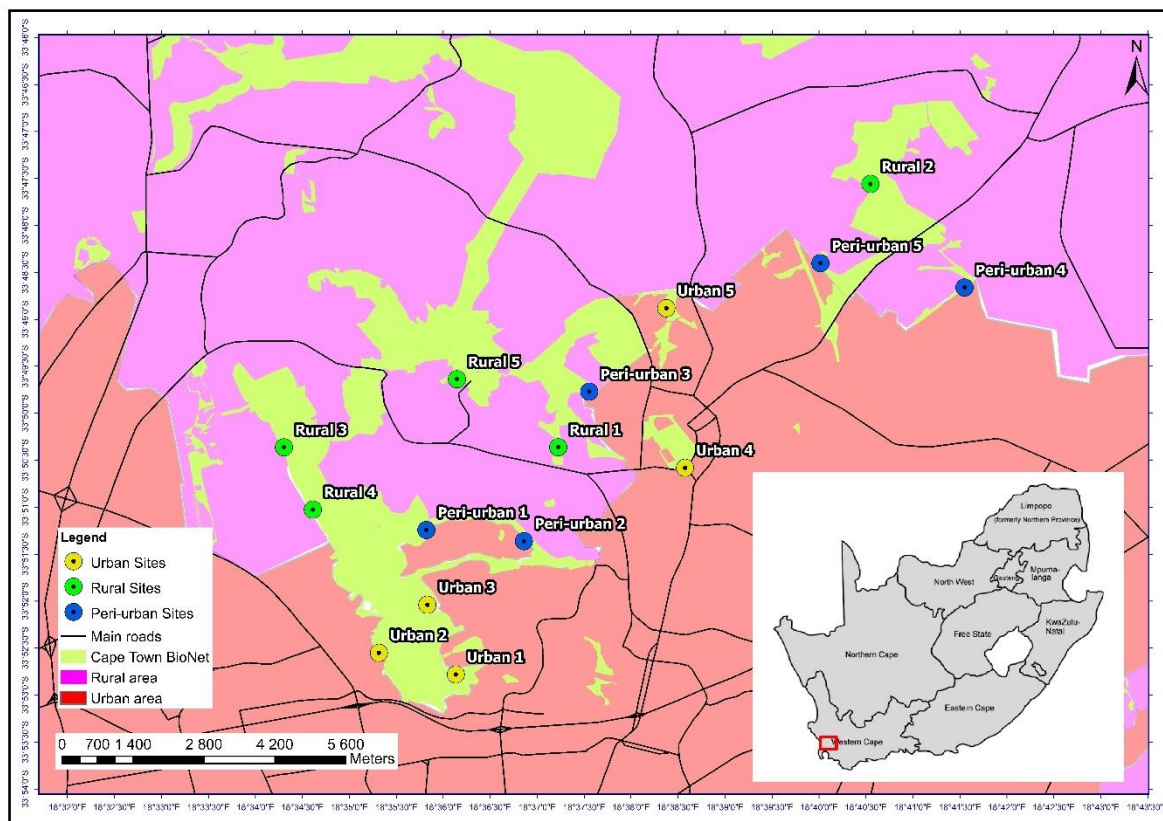
Figure 2.1: Study sites used to identify differences in invasive alien species composition between urban, periurban and rural areas within the greater Tygerberg area of the City of Cape Town. Swartland Shale Renosterveld within the Cape Town Bionet (highlighted in green) is shown in relation to urban land cover (Red) and rural land cover (Purple). The peri-urban zone is present at the interface of the urban and rural zones. The study sites selected across the urban-rural gradient are illustrated on the map as yellow (urban cites), blue (peri-urban) and green (rural sites). The scale of this map is 1:80 000.

Urban sites are defined as areas which are in close proximity (within 100m) to only residential, commercial or industrial land-use types, which represent typical urban land-use types. Peri-urban sites are defined as areas which are in close proximity (within 100m) to residential, industrial or commercial land-use types and open public space or agricultural land-use types. Rural sites are those which are in close proximity (within 100m) to agricultural, public open space and rural land-use types. The land-use types were verified with the use of the City of Cape Town Integrated Zoning Land Parcel layer (City of Cape Town, 2016).

2.2.3 Field sampling and experimental design

We sampled five replicates of each habitat type (urban, peri-urban or rural). Sites were sampled during the summer months of November and December 2016. Within each of these replicates, six 25 m2 plots were sampled (n = 90 plots) to determine IAP species richness and abundance. The number of IAP species as well as their percentage cover was recorded for each plot. Extra-limital species and

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indigenous species were included as they were species that behaved like weeds or are not known to grow in the Swartland Shale Vegetation type. Voucher specimens of all unknown species were collected, pressed and identified to species level.

To identify factors promoting IAPs in urban areas as compared to peri-urban and rural areas we recorded habitat condition for each plot using a scoring system (developed by the City of Cape Town Invasive Species Management Unit) (Table 2.1). Each sub criterion (9 in total) was scored along a scale of 1-5 where 1 is Good and 5 is bad. Specifically, the criteria classes are as follows: 1 = natural; 2 = near-natural; 3 = fair; 4 = modified; 5 = irreversibly degraded. The mean score for each plot was then calculated. According to their habitat condition, plots were classified as containing pristine vegetation (good/natural/near-natural vegetation) or degraded vegetation (fair/degraded/modified/very degraded/irreversibly degraded or modified vegetation). It must be noted that this scoring of criterion is still rudimentary and therefore may be highly subjective to the sampler. The environmental variables collected for each plot were: percentage exposed rock, percentage bare ground and degree of erosion as well as environmental disturbances. Furthermore, it must be noted that indigenous species recorded in the vegetation survey were not linked to the indigenous species sampled or recorded to determine habitat condition for each plot. Indigenous species sampled with the use of table 2.1 were used sorely to estimate habitat condition.

Table 2.1 The criteria used to determine whether habitat condition is a mechanism for IAP species establishment in urban, peri-urban and rural areas.

Criterion	Sub-criterion
Physical disturbance	Top soil lost (e.g. quarry; road)
	Ploughed with or without fertilizing
	Altered hydrology
Biotic impacts	Overgrazing/ brush cutting/over harvesting
Indigenous vegetation cover	Vegetation % cover
	Vegetation post fire age (year)
	Median height (m)
	Strata and spatial patterning of dominant growth forms
Composition	Indigenous species richness

Criteria classes: 1 = natural; 2 =near-natural; 3 = fair; 4=modified; 5=irreversibly degraded

To determine the origin of the IAPs found in this study, they were classified on the basis of the likely reason for their introduction and use in the area, i.e., ornamental horticulture, forestry, agriculture or indigenous. Species origins were obtained from Bromilow (2010) and Fish et al. (2015). In addition, species were classified according to the C-S-R plant functional types (Grime, 1977), i.e., competitors, stress-tolerators and ruderal species. With the use of field guides (Carruthers, 2000; Manning, 2009; van Wyk and Gericke, 2007; Bromilow, 2010), the invasion status (whether invasive, alien, naturalized or extralimital native species) of each species was determined, as was growth form type (trees, shrubs, grasses, herbs and creepers).

2.2.4 Statistical analysis

IAP richness and abundance were analyzed using variance estimation, precision and comparison (VEPAC Module of Statistica 13.5) analyses. VEPAC was used to (1) determine the variance and differences between habitat types for both IAP species richness and IAP species abundance and (2) to determine the difference in IAP species abundance for five different growth forms (grasses, herbs, shrubs, trees and creeper). The VEPAC was a mixed model ANOVA with habitat type as a fixed effect and area as random effect. This was followed by a Fishers Least Significant Difference (LSD) posthoc test (LSD). A mean habitat score was determined for each plot by averaging the score given for each of the subcriteria shown in Table 2.1.

A principal component analysis (done using the R package 'Vegan') was used to determine whether species composition is different between habitat types. In addition, multiple regression analyses were used to determine the relationship between habitat condition, IAP species abundance and richness. All analyses were conducted in STATISTICA 13 (Statistica, 2019) and R (R development core team, 2013).

2.3 Results

In total, 79 alien plant species were recorded in the greater Tygerberg area across all habitat types. Urban, peri-urban and rural habitat types had proportionately more invasive plant species than alien and naturalized plant species (Table 2.2). In urban areas, there were equal proportions of competitors (42%) and ruderal species (46%) with fewer stresstolerators (12%). However, there was a higher proportion of ruderal species (54%) in both the peri-urban and rural habitat types compared to competitors (37%) and stress-tolerators (9%). IAPs from an agricultural origin were more common in all three habitat types than species from a horticultural or forestry origin (Table 2.2).

The two main components of the principal component analysis describe 14% (PC1- x-axis) and 13% (PC2 e Y-axis) of the variation. The species composition for urban and rural habitat types was similar and tightly grouped together (Fig. 2.2). The grouping of peri-urban species was more widely scattered and the species were grouped adjacent to the urban and rural showing that the results for peri-urban sites differ from those of the urban and rural sites. *Bromus hordeaceus, B. diandrus, Lolium multiflorum*

x Lilium perenne, Aevena barbarta and *Cynodon dactylon* were the five major outliers from the PCA and may explain why the two principal components account for relatively little of the variation.

Table 2.2: Proportion and percentage of plant species across the three different habitat types for invasion status, plant adaptive strategies and the potential origins of introductions for all species sampled across the City of Cape Town. Percentages were derived using the total number of species found in each area. IAP = Invasive Alien Plant

		Urban (%)	Peri-urban (%)	Rural (%)
Invasion status	Invasive	31(56)	26(59)	18(53)
	Alien	8(15)	7(16)	4(12)
	Naturalized	11(20)	10(23)	11(32)
	Indigenous	5(9)	1(2)	1(3)
Plant adaptive strategies	Competitor	24(42)	17(37)	13(37)
	Ruderal	26(46)	25(54)	19(54)
	Stress-tolerator	7(12)	4(9)	3(9)
IAP origins	Forestry	3(5)	1(2)	1(3)
	Agricultural	33(58)	28(61)	23(66)
	Horticultural	7(12)	7(15)	6(17)
	Unknown	10(18)	8(17)	3(9)
	Indigenous	4(7)	2(4)	2(6)

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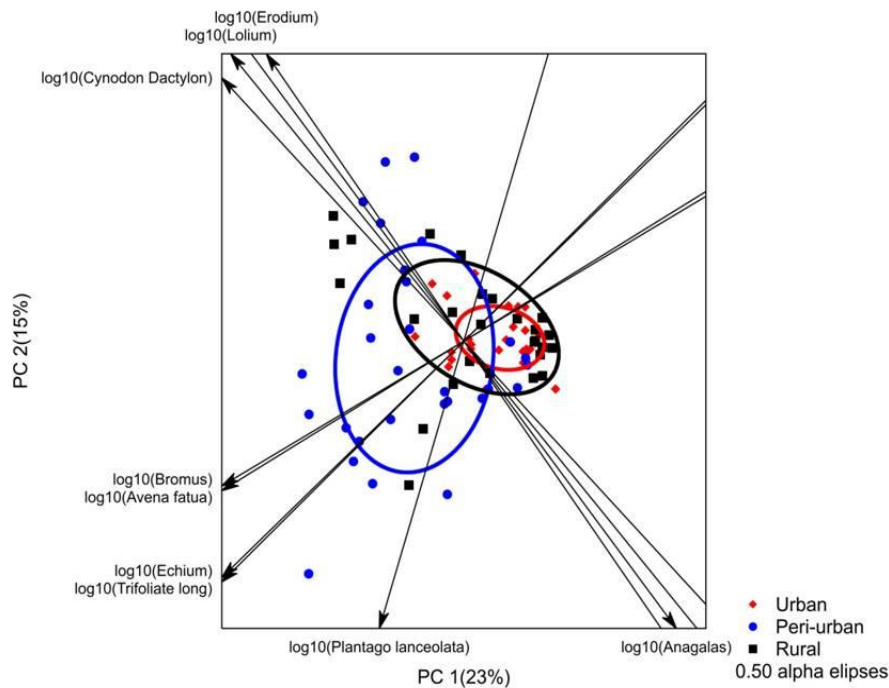
Figure 2.2: Principle component analysis showing the groupings of invasive alien species in urban (red), peri-urban (blue), and rural (black).

There was no significant difference in IAP abundance between urban and rural habitat types (Table 2.3). The peri-urban habitat type had a significantly higher abundance of IAPs compared to urban and rural habitat types (Table 2.3). There was no significant difference in IAP species richness between the different habitat types (Table 2.3).

There was no significant difference in tree, shrub or creeper abundances between the different habitat types (Table 13.3). However, grasses were significantly more abundant in peri-urban habitat compared to the urban and rural habitat types, but there was no significant difference between grasses in the urban and rural habitat types. Herbs found in peri-urban sites were significantly higher in abundance than herbs in the rural habitat type (Table 2.3).

Table 2.3: Table showing the means and P-values for the variance estimation and precision and comparison undertaken for invasive alien plant abundance, invasive alien plant richness as well as the different growth forms sampled across all three habitat types. Abundance is the number of individuals per species in a community and richness is the number of species within a community. P values less than 0.05 are significant. Mean values with the same letter(s) are not significantly different.

	Urban	*Peri-urban*	*Rural*	*P-value*
IAP species abundance	24[b]	57[a]	18[b]	0.000746
IAP species richness	8	9.2	6.2	0.10
Grasses	0.7[b]	1.4[a]	0.7[b]	0.01426
Herbs	1[ab]	1.15[a]	0.75[b]	0.02347
Trees	0.15	0.06	0.04	0.37130
Shrubs	0.056	0.054	0.002	0.41657
Creepers	0	0.12	0	0.39657

The abundance of IAPs was positively correlated to habitat condition (P = .00, Fig. 2.3). Furthermore, the richness of IAPs was positively correlated to habitat condition (P = .0008, Fig. 2.3). Therefore, as the level of degradation increases so does the number and cover of IAPs.
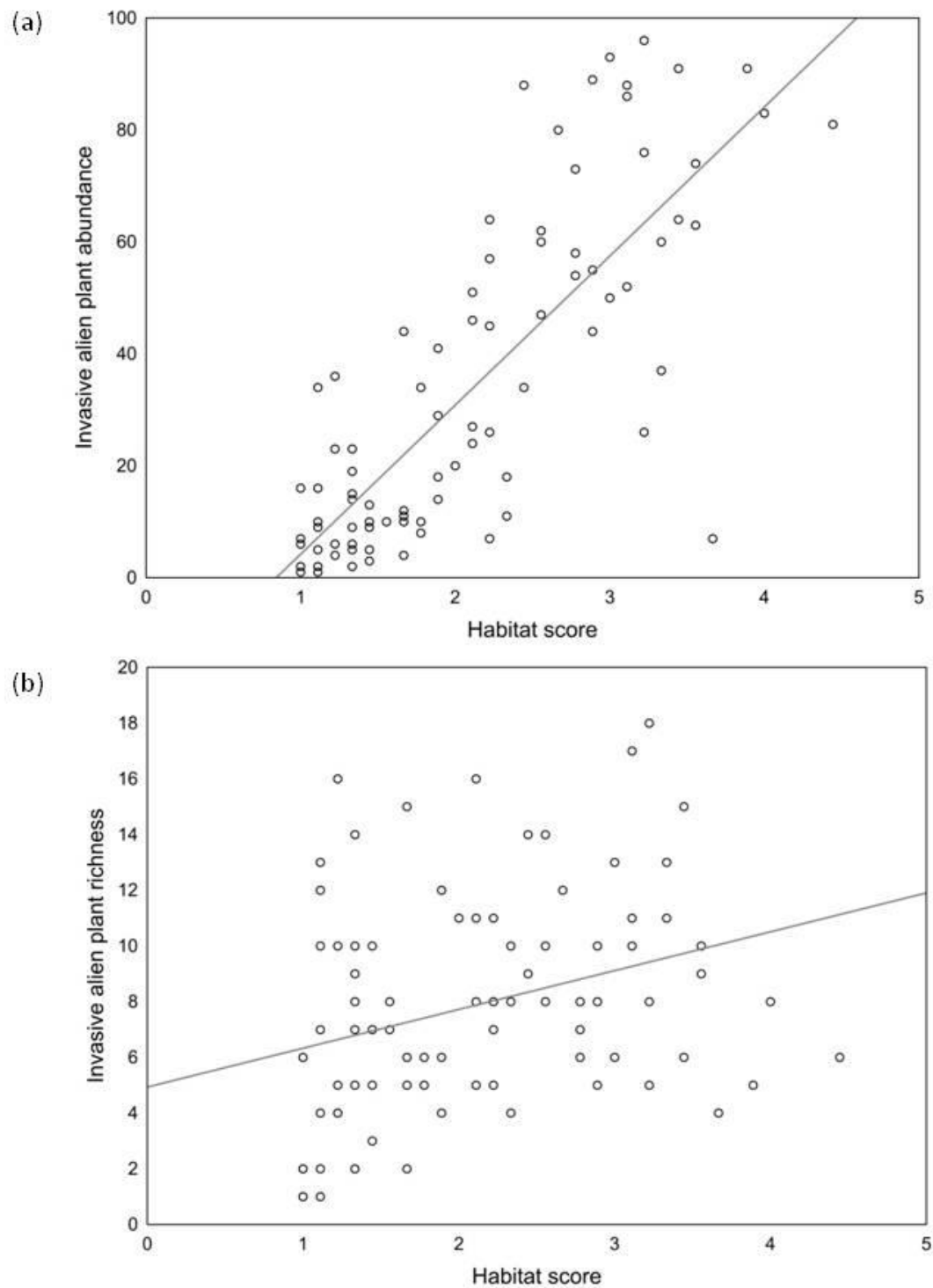
Figure 2.3 Regression analysis of the effect of habitat type on (A) IAP abundance [P = .000; r^2 = 0.637] and (B) richness [P = .003; r2 = 0.096] in urban, peri-urban and rural habitat types. A higher habitat score indicates a higher level of degradation.

2.4 Discussion

In determining the composition of IAPs across the urban-rural gradient, we show that IAP abundance and richness were marginally higher in urban areas compared to rural areas. However, the abundance of IAPs in peri-urban areas were significantly higher than in both the urban and rural areas. However, there was no significant difference in IAP species richness between the different habitat types (Table 2.3). These broad patterns may be associated with the types of IAP source populations in the different areas and/or habitat conditions.

2.4.1 The difference in IAP species abundance and richness between urban, peri-urban and rural areas

Proximity to large propagule sources is an important determinant of successful invasion (Heinrichs and Pauchard, 2015; Richardson and Pysek, 2006). The larger the number of propagules, the higher the chance for establishment, persistence and naturalization of IAPs (Rouget and Richardson, 2003). Because the initial invasion of an IAP into natural areas is often limited by the availability of propagules, in some cases areas with high propagule pressure (for example, from suburban gardens through ornamental horticulture) can be a better predictor of IAP cover than environmental factors (Von Holle and Simberloff, 2005).

Ornamental horticulture is one of the most important pathways for IAP introductions worldwide and spread of introduced IAPs often begins in urban areas (Mayer et al., 2017). However, this is not the case in our study. Only 12%, 15% and 17% of the IAPs found within urban, peri-urban and rural habitat types, respectively, were of horticultural origin. It is key to note that our study area is not within the core of the City of Cape Town but generally has a much higher proportion of gardens compared to the more densely developed inner city. The low proportion of horticultural IAPs found across all three sites indicates that the successful invasion of horticultural IAPs into natural areas may be dependent on other factors (e.g., elevation, aspect or disturbance). The majority (more than 50%) of the IAPs found in all three habitat types are of agricultural origin, possibly a legacy of agricultural land uses that occurred in the area prior to development. Therefore, a potential explanation for the highest abundance of IAPs in peri-urban areas is that these are receiving environments for both urban/ suburban (horticultural) IAPs and those of agricultural origin.

The distance from IAP source populations was found to be an important factor in structuring IAP richness in Cape Town forests (Alston and Richardson, 2006). Similarly, the abundance of IAPs in New Zealand native forest fragments correlates with the distance to the nearest large town (Sullivan et al., 2005). In Southern Wisconsin (USA) rural housing variables and the distance to forest edge had the strongest association to the richness and abundance of IAPs in native forests (Gavier-Pizzaro et al., 2010). However, some studies contradict these general findings; for example, Fornwalt et al. (2003)

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concluded that anthropogenic disturbances (logging, grazing and other similar disturbances) have less of an impact on IAP establishment than topographic positioning.

2.4.2 Habitat condition is responsible for promoting the establishment and facilitating the invasion of IAPs into natural areas

Historic and current anthropogenic activities are important influences on the degree of invasion in natural areas (inside urban areas and beyond) and may actually diminish the influence of proximity to the urban edge. The City of Cape Town has a long history of human influence, including the presence of permanent settlements, contour farming and recreational usage; these disturbances are likely to have had a profound effect in increasing IAPs in the area.

Where intact native vegetation occurs adjacent to urban centres, competition among species may hamper the invasion of IAPs into these areas (Stajerova et al., 2017; Wania et al., 2006). Swartland Shale Renosterveld demonstrates some resistance to invasion (biotic resistance) by horticultural IAPs since the plots in the urban habitat types which were classified as good/natural or near-natural had lower abundances and richness of IAPs when compared to plots which were classified as modified/fairly degraded or irreversibly degraded. Similarly, biotic resistance, likely due to the saturation of the natural vegetation patch by a very competitive native species (Funk et al., 2008), was responsible for hindering invasion of IAPs in native grassland remnants in the North West, South Africa (Cilliers et al., 2008).

Our results show that the more degraded the habitat condition of the sites, the more likely IAPs occur in higher abundances and richness. As such, the higher abundances and richness of IAPs in peri-urban areas is most likely because these peri-urban areas are the most disturbed habitats in the sample area. Previous studies support our findings. Alston and Richardson (2006) found evidence that disturbance expedites IAP invasions while investigating the roles of habitat features, distance from alien source populations and disturbance structure IAP invasions at the urban/wildland interface. Invasive alien plant invasions were highest in areas of anthropogenic activity and near vegetation fragment edges (Aiko et al., 2012) as well as roadside and sparse habitats (Alston and Richardson, 2006; Catford et al., 2011; Geerts et al., 2016; Johnston and Johnston, 2004). Sites without disturbance did not support invasive alien plants in urban bushland, while sites which were disturbed by trampling did support invasive alien plants (Lake and Lieshman, 2004). Furthermore, Buonopane et al. (2013) found that noxious and exotic IAP frequency and cover significantly decreased with the distance from roads.

Roads and trails aid in increasing a site's invasibility through the creation of open spaces, which effectively fragment natural areas and change local ecosystem microclimates (Delgado et al., 2007; Gelbard and Belnap, 2003). Through the alteration of light availability, disturbance adds to the competitive advantage of invasive species by opening up spaces in the canopy for IAPs to potentially grow and become established among native plant species (Yates et al., 2004). Natural areas and surrounding areas in the City of Cape Town are very popular recreational areas creating disturbances

and facilitating dispersal of IAPs. Furthermore, the firebreaks found on all of the study sites provide a stepping stone habitat for IAPs which have the potential to spread further into the natural areas.

2.4.3 Practical implications for management

The accelerated rural land transformation brought on by the process of urbanization means that managers urgently need to appropriately manage and conserve natural and seminatural habitat remnants (Kowarik, 2011). It is crucial to protect these remnants against further degradation and IAP invasion.

While horticultural IAPs were not the most prevalent in this study, a critical step in preventing the spread of IAPs across the urban/wildland interface would be to remove species that are known or suspected to be invasive from suburban gardens (Alston and Richardson, 2006). Furthermore, further housing developments adjacent to natural areas of high conservation importance should be discouraged from using IAPs in landscaping.

Since habitat condition was a key predictor of IAP abundance and richness, it follows that the susceptibility of invaded ecosystems can be managed to minimise factors that facilitate alien plant invasion such as reducing anthropogenic disturbance. Firebreaks, or any source of potential disturbance (roads, paths and rivers) must receive attention to ensure that IAPs are not allowed to disseminate into natural areas. Firebreaks and the associated disturbance have been shown to have an increased frequency of IAP plants (Aanderud et al., 2017). This would demand a considerable investment of resources, but is indicative of the level of management that is required if small remnants of natural or seminatural vegetation are to be conserved.

Natural areas that are already heavily invaded must be monitored and buffer zones should be established in order to protect nature reserves (Heinrich and Pauchard, 2015). Furthermore, when planning and identifying BioNet areas which will be affected by the process of urbanization, long-term goals must be set in order to maintain already critically endangered vegetation types from becoming infested by IAPs. Habitat condition standards should be forecast and IAP control programmes implemented to maintain the ecological integrity of potential future conservation sites.

Swartland Shale Renosterveld and the Cape Town BioNet are under considerable threat from IAP invasion and from anthropogenic activities and immediate management intervention is required to maintain ecosystem integrity throughout the mosaic of natural areas found within urban areas. Most importantly, management should be prioritized for areas at the urban-wildland interface and beyond to prevent the degradation of potential future conservation sites as the city expands. Further research on the intrinsic characteristics of peri-urban areas may provide a better understanding of invasion success and spread of ornamental IAPs and biotic resistance.

Conclusions

An investigation into the IAP species richness and abundance across the urban-wildland interface resulted in the rejection of both our two original hypotheses. There is no significant difference in IAP species abundance between urban and rural areas. However, there is a significant difference between peri-urban areas compared to urban and rural areas. These is no significant difference in IAP species richness between urban, peri-urban and rural areas. Management should focus on areas at the urban-wildland interface and beyond to prevent the degradation of potential future conservation sites as the city expands.

2.5 Acknowledgements

Chapter 3: The management and prioritization of invasive alien plants in urban areas: Testing a multi-criteria decision model.

3.1 Abstract

Eradication applies to species that have been recently introduced and still have limited distributions but pose a potential environmental and economic threat. Eradication requires the elimination of every individual plant from an area in which recolonization is unlikely to occur. According to Panetta (2015), eradication occurs via two processes: (i) extirpation (the elimination of the target in both space and time) and (ii) containment, which is the prevention of geographical expansion (preventing spread). The aims of this study are threefold: Firstly, to develop a concept/ framework for categorising species into widespread IAPs, emerging IAPs and potential eradication targets. Secondly, to group species from the City of Cape Town into these management categories. Thirdly, to prioritise potential eradication targets within urban areas with the use of a multi-criteria decision model. The classification protocol provides a succinct method for orderly allocation of different types of species to different management efforts (even when limited data is available). The Analytical Hierarchical Process (AHP) model provides a list of ranked species upon which land managers can focus management or monitoring programmes. Furthermore, the model outlines the significance of obtaining species level information for effective prioritization of IAP species in urban areas.

3.2 Introduction

Land managers and decision makers are facing significant challenges in that they are required to make use of limited funding to manage invasive alien plants (IAPs) to achieve the most benefits (van Wilgen et al., 2008). Due to a lack of funding or general limited resources, land managers and conservationists must carefully choose which species control efforts must be focussed on (Nel et al., 2004). Furthermore, decision makers, planners and land managers are required to prioritize management and control activities in an environment where information varies in its nature and quality (van Wilgen et al., 2008). Prioritization of areas and IAPs is essential to ensure the allocation of resources is most efficient and cost-effective in order to support good decision-making (McGeoch et al., 2016). The necessity to respond effectively to biological invasions has thus promoted the development of decision-making tools to assist in setting priorities for management (Roura-Pascual et al., 2009). These tools can be used to identify the order in which species should be controlled.

Invasive species can be grouped into widespread invasive species, emerging species and targets for early detection and rapid response. This categorisation is based on a framework proposed by Blackburn et al. (2011, adapted from Richardson et al. 2000). The framework illustrates that the invasion process can be divided into a series of stages (ranging from transport and introduction to establishment and spread). The framework also illustrates that at each stage of the invasion process there are barriers (geographical, environmental, reproductive and dispersal) that need to be overcome in order for a

species to pass onto the next stage of invasion. According to a study by McGeoch et al. (2016), prioritization can be undertaken across the distinctive stages of invasion (Blackburn et al., 2011). Firstly, the prioritization of pathways can be undertaken to prevent the introduction or reduce the risk of introduction of potentially harmfully IAPs.

A widespread invasive alien plant (IAP) is an alien plant which has successfully overcome the major barriers (geographical, environmental, reproductive and dispersal) which limit the spread of introduced plants, to become fully invasive (Richardson et al., 2000). An emerging species is an IAP that has already become naturalised in South African ecosystems, but which currently has restricted populations and causes less of an impact than major invaders; however, the species has attributes and potentially suitable habitat that could lead to its further proliferation and a more significant impact in the future (Mgidi, 2004). Eradication targets or Early Detection and Rapid Response (EDRR) species are a subset of emerging species, which are still feasible for eradication (e.g. those IAPs which have a small number of populations or a limited distribution or slow rate of spread or are easy to control). Species which exhibit all these attributes should be considered for eradication no matter what the risk to the ecological and economic environment is. This study focuses on the prioritization of naturalized species (potentially eradicable IAPs/ so-called early detection and rapid response (EDRR) species).

It is complex to group species into management categories, as managers and decision makers simultaneously need to consider a number of IAPs at all stages of invasion (Pysek and Richardson, 2010). They also need to make decisions with limited knowledge of spatial extent of infestations of IAPs (Panetta, 2009) which may be rapidly expanding (Simberloff, 2003). Additionally, there often exists poor population level information. This lack of information also makes it challenging to identify potential and practically feasible eradication targets.

In urban ecosystems the situation is even more complex than in rural ecosystems. Alternative frameworks for IAP management are necessary because land use management, accessibility to IAPs (private land managers), scale (city as a local scale as compared to nation-wide management), pathways of introduction and vectors of IAP dispersal may differ substantially between urban and peri-urban areas and rural areas (Gaertner et al., 2016).

Emerging species are currently afforded lower priority in terms of management (Mgidi et al., 2007) largely because of limited funds and resources (Olckers, 2004). This is because in many cases, when an IAP is introduced to an ecosystem, there are no discernible impacts during the lag phase of invasion (i.e. between establishment of a species and a species becoming invasive), especially over short time scales (Richardson et al., 2000). However, this may be evasive as impacts from IAPs may be small at first but may have calamitous consequences in the long term (Mgidi et al., 2004; Pysek & Richardson, 2010). It is therefore important that management programs give equal attention to targeting emerging species to prevent further spread (Nel et al., 2004; Panetta and Timmons, 2004; Brunel et al., 2010).

For each of the different categories/groups of invasive species, different management options apply: Mitigation is the only practically feasible management approach for widespread invasive species (van Wilgen et al., 2011). In contrast, containment as a management approach can be utilised for species where eradication is not feasible, but where there is still considerable risk of geographical expansion to pristine natural areas (van Wilgen et al., 2011). The focus of this management approach should be on preventing spread of IAPs, with limited distributions, to new areas thereby limiting potential impacts that may arise with an increase in infestation area of an IAP.

Eradication applies to species which have been recently introduced and still have limited distributions but pose a potential environmental and economic threat. Eradication requires the elimination of every individual plant from an area in which recolonization is unlikely to occur (Myers et al., 1998). According to Panetta (2015), eradication occurs via two processes: (i) extirpation (the elimination of the target in both space and time) and (ii) containment, which is the prevention of geographical expansion (preventing spread).

Several nationwide prioritization protocols have been developed for IAPs in South Africa. The majority of these are spatial prioritizations with the exception of a few species-based prioritizations. One of the first species-based prioritizations to be undertaken in South Africa was that of Robertson et al. (2003). This study prioritized 61 species based on five aspects (potential invasiveness, actual spatial extent, potential impacts, potential for control and conflicts of interest). Nel et al. (2004) prioritized species by widespread species (117 species categorised into groups based on geographical range and abundance) and emerging species (84 species categorised into groups based on current propagule-pool size and potentially suitable habitat for invasion).

Other species-focused prioritisations include: Kumschick et al. (2012) who developed a prioritization framework which includes scientific impact assessment and the evaluation of impact importance by affected stakeholders; and Roura-Pascual et al. (2009) who developed a robust conceptual framework for effective management of the most important IAP species. South African spatial prioritizations undertaken to date include Mgidi et al. (2007) who sought to develop a protocol for predicting the potential distribution of emerging IAPs species using distribution records from other regions to augment existing records within the country of concern. To date, no species-based prioritizations have been undertaken specifically in urban areas in South Africa. In addition to this no prioritization protocols exist at a local (fine) scale (i.e. for the Cape Town Metropolitan area). As such, there is a need to develop prioritization protocols for urban areas.

The primary aim of this study is to develop a classification tool for the grouping of widespread, emerging and eradication target species. The first step to achieve this aim is to develop a concept/framework for categorising species into widespread IAPs, emerging IAPs and potential eradication targets. The secondary aim of this study is to utilize a multi-criteria decision model to prioritise a larger

number of potential eradication target IAPs within urban areas. The findings of this study provide suitable frameworks for the management of IAPs in other urban areas worldwide.

3.2 Methods

3.2.1 Classification of invasive alien plants

The section below outlines the process of developing a classification protocol.

3.2.1.1 A classification tool for the management of Invasive Alien Plants

To develop the classification tool (Aim 1), a desktop review of current classification frameworks was undertaken. This classification framework aimed to group different IAPs into three types (widespread IAPs, emerging IAPs and potential eradication targets). The desktop review undertaken was with the use of scholarly search engines between 2016 and 2019 (i.e. google scholar and the web of science) to find all classification/categorisation tools and protocols or studies that assess the feasibility of eradication that have been developed or undertaken worldwide to date. The keywords used were: classification, categorization, feasibility, protocols, grading, eradication, allocation, apportionment, grouping, arrangement, containment, allocation, grading, designation, allotment, sequencing, organization and distribution. These keywords were used in conjunction with keywords such as: alien, invasive, exotic, incongruous, plants, weeds and non-native. The papers selected were first chosen by scrutinizing the title, then by reading the abstract. Only applicable studies were chosen and assessed.

3.2.1.2 Managing the database of invasive alien plants in South Africa

Once the classification framework had been developed, each type of IAP within the City of Cape Town was then assigned to alternative management categories (mitigation, containment and eradication).

To do so, we compiled a database of invasive alien plants that have been introduced into Cape Town. Data were taken from South African National Biodiversity Institute (SANBI) list of, invasive and extra-limital species (species which originate from outside of the Cape Floristic Region) (~623 spp.) and the City of Cape Town (CCT) list of invasive species and management protocols.

We used data from known plant lists and ad hoc information gathered from the City of Cape Town invasive species unit. Notes on most of the species found were obtained from The Flora of the Cape Peninsula (Adamson and Salter, 1950). Information on a total of 623 species was gathered, however, this number included native and extralimital species. For the purpose of this study we excluded these species to yield a total of 515 species from which we classified widespread IAPs, emerging IAPs and potential eradication targets using the classification protocol developed in aim 1 of this study. Detailed information on the distribution and abundance of these species was available (Adamson and Salter, 1950) despite the fact that some of these species are at an early stage of invasion.

3.2.2 Prioritization of potential eradication targets

3.2.2.1 Multi-criteria decision making.

A multi-criteria decision making approach known as the analytical hierarchical process (AHP; Saaty, 1990) was used to determine and weight criteria to undertake the prioritization. In 2013, two workshops were organised by Greg Forsyth and David Le Maître (CSIR, Stellenbosch) in which managers, researchers and other stakeholders agreed on a goal and on criteria for prioritisation (Forsyth, 2013).

Workshops were attended by 20 managers and researchers which were selected from the Table Mountain National Park, South African Nurserymen's Association/ SAGIC, the CSIR, the South African National Biodiversity Institute (SANBI), the Centre for Invasion Biology (CIB -Stellenbosch University) and the City of Cape Town (CCT). The result was the development of a model that was then presented to the same workshop stakeholders in a follow-up workshop to refine the model further and scrutinize the goal. At the end of this follow-up workshop a goal was agreed upon, namely; to prioritize the control of Early detection and Rapid response (ED&RR) target plant species in the Cape Metropole.

AHP involves three steps. The first step is to define a goal. In this case the goal is to "rapidly control invasive alien plants, both present and emerging, so as to maintain ecosystems in the Cape Town metro". Interested and affected parties opted to exclude the development of an area-based prioritization model in addition to the species-based model as such a model would include spatial data. The reasoning behind this is that species selected for this prioritization pose a high threat irrespective of where they occur as they have the potential to spread throughout the city of Cape Town.

The second step in the AHP process is to identify the criteria and sub-criteria and to agree on the relative weighting of criteria that will be used in prioritising the target plant species. The Analytic Hierarchy Process (AHP) was used to compare individual criteria within the hierarchical structure and to assign weights to each of these according to their relative importance at each level (Saaty, 1990). Criteria and sub-criteria that are given low weights in an AHP model have little influence on the outcome. The aim of this synthesis was to limit the final prioritisation model to as few as possible criteria at each level in the hierarchy, while reflecting the diversity of views among the managers and researchers who attended the workshop. Expert Choice 11.5 decision support software (Anon. 2009) was used to facilitate this process.

Once the goal, criteria and sub-criteria for prioritising the ED&RR target plant species (Appendix B) had been identified and incorporated in the AHP model the third step was to populate the underlying data matrix (values found in Appendix C).

3.2.2.2 Comparisons of species in their applicability for achieving the goal.

Once the criteria had been agreed upon by stakeholders, we identified plant datasets that would permit for comparisons to be made between the alternative species with regard to each of the identified criteria and subsequent sub-criteria. Following the workshops, the weighting of criteria were completed manually using the identified data sets supplied by the City of Cape Town (Weights shown in Appendix C). These calculations of numbers (species which were scored upon a scale for each criteria and sub-criteria) were completed using Excel spreadsheets. The weights were then imported into the AHP model with the use of Expert Choice (Anon, 2009) which then calculated the weights for each criteria, sub-criteria and for the overall goal for each species.

3.2.2.3 Description and scoring of criteria.

Once the data matrix was completed it was imported into the AHP model (Shown in Appendix A). The results of running the data through the prioritisation model were presented for the overall goal and different criteria (Appendix A). The data used, the rationale, and the values assigned to each criterion and associated sub-criteria in the model are discussed below and presented in Appendix A.

3.2.2.3.1 Spread.

This criterion consists of two sub-criteria. These are the current rate of spread and current distribution of the ED&RR target IAPs.

3.2.2.2.1.1 Current rate of spread.

We obtained data on the spread of invasive alien plants from Dr Tony Rebelo of SANBI. He had calculated this by taking the linear distance between the locations of species recorded on the Cape Peninsula by Adamson and Salter (1950) and where these species occur in 2010. One of his assumptions was that the North –South gradient along the Cape Peninsula is proportional to the area in the City that is invaded. Using an ordinal scale of 1 to 3 we assigned a score of 1 to species that had not, or had hardly spread since 1950; 2 to plants that had spread less than 10 kilometres; and 3 to species that had spread up more than 10 kilometres.

3.2.2.3.1.2 Current distribution.

Once again this information was obtained from Dr Tony Rebelo who used the square of the linear distance referred to under the previous sub-criteria. We supplemented this with locality information on the City of Cape Town's IAP website (http://www.capetownivasives.co.za). In this case plants were ranked on a scale of 1 to 4 based on the estimated area that these plants had invaded by 2010. We assigned a value of 1 to those species that already infest more than 2500 ha, 2 to plants infesting < 400 ha, 3 to plants infesting < 25 ha and 4 to plants occupying < 1 ha. The rationale for this is that plants

that are found in fewer localities or only occupy a limited area should be controlled before large infestations are tackled.

3.2.2.3.2 Negative impacts on biodiversity process and pattern, and ecosystem services.

We do not have data on specific impacts for most species but we do know that species vary in the degree of impact (Robertson et al., 2003). However, threats posed to ecosystem functioning by ED&RR species are not easy to quantify. A combination of known impacts and this expert knowledge and opinion was used to give each species a value for each of these criteria using an ordinal ranking scale as set out below. The assigning of weights can be subjective, however this process has been used successfully in the past to prioritize large data sets (Forsyth et al., 2012; van Wilgen et al., 2020).

3.2.2.3.2.1 Negative impacts on biodiversity process

This sub-criterion captures many of the features that enable a species to transform the ecosystems it invades which is why many of these species are identified as 'transformers' by Henderson (2001). Known effects of the acacias which fix nitrogen that can leach into soil and groundwater (which then discharges into streams), and other species such as Eucalyptus species which invade and alter riparian habitats and, thus, biogeochemical processes.

Scoring was done on an ordinal scale of 1 to 3 taking into account which species typically transform the ecosystems they invade (i.e. they are transformers as defined by Henderson 2001). Species having high impacts (transformers) were given a score of 3, special effect and potential transformers a score of 2, and non-transformers and score of 1. Special effect species includes those that are poisonous to livestock or unpalatable. The rationale is that plants that potentially could have marked negative impacts on biodiversity process ought to be brought controlled or eradicated first.

3.2.2.3.2.2 Negative impacts on biodiversity patterns

This sub-criteria explains where (what habitats) and how plants change the structure of the indigenous plant communities. For example, the extent to which the species alters the fuel load and thus the fire behavior or the ability to introduce fire into relatively fire free communities. An example is *Chromolaena odorata* which is flammable and invades forest-grassland ecosystems increasing fire intensities and damaging the forest edge. We used Dr Rebelo's interpretation of the data contained in the "i-nuturalist" database (https://www.inaturalist.org/) as a surrogate for this sub-criterion.

He used a scale of 0 to indicate where plants had not invaded the natural veld, 3 for occasionally invaders, 5 for minimal effect, 7 for some environmental impacts and 9 if the plant species was a known ecosystem transformer. The rationale for this was that species having the most detrimental effect on biodiversity pattern would be scored highest.

3.2.2.3.2.3 Ecosystem services

These are products and process derived from ecosystems that benefit people; these services are produced by the processes that sustain ecosystems and by the roles that species and communities play in those ecosystems (Boyd and Banzhaf, 2006). These services can include production services which yield useful goods including grazing and browsing and harvestable products; cultural and recreational services which contribute to human wellbeing such as recreational activities; and regulatory services which are ecosystem processes and functions that regulate the dynamics and the flows of services. The degree to which ED&RR plant species can potentially impact on health, water resources and economics was identified during the workshops as being important and they are discussed further below:

3.2.2.3.2.3.1 Health

Negative impacts on the health of humans and animals were ranked from 1 to 3 based on whether or not they were poisonous or irritates and pose a risk. Non-poisonous or non-irritant plants were assigned a score of one, plants which were either poisonous or an irritant were scored 2 and those that were both poisonous and an irritant were scored 3. Information was obtained from Henderson (2001), Dufour-Dror (2012) and Bromilow (2010). The rationale being that plants that are poisonous or irritants to humans or animals should be controlled ahead of plants that pose no danger.

3.2.2.3.2.3.2 Water

This criterion recognizes the critical importance of water as a resource for both ecosystems and human livelihoods and welfare and, therefore, of clearing species that have high water usage. There is little information on this especially in the case of emerging weeds. We reasoned that plants with greater biomass would use more water than small herbaceous plants. We therefore used a scale of 1 to 3 with little or no known effect being given a score of 1, a score of 2 where some negative effect on water resources is known and 3 when this negative impact is high.

3.2.2.3.2.3.3 Economics

This gives an indication of whether or not a plant species has the potential to disrupt certain ecosystem services (used as a surrogate for economics). A score of 1 was allocated to the plant species where such impacts are negligible and 2 where we believe there will be a negative effect. The rationale for this was that plants having known negative impacts need to be controlled first.

3.2.2.3.3 Ease of control

Ease of control reflects how easily the species can be controlled. So, effective control can be achieved in a single clear-felling operation followed by a well-timed fire. For example, many wattles sprout vigorously, have persistent seed banks and mature rapidly (Gibson et al., 2011). If the density of these

species is to be reduced it will require many follow-up control operations spread over several years, or possibly decades.

3.2.2.3.3.1 Control effort

Scores for these were based on the traits of the species which makes them particularly difficult to control. Ease of control does not include dispersal but does include whether or not they are fast maturing and require herbicides to prevent coppicing or reproducing. Species were given ordinal scores so that species that are easy to control got a high score. As such, the scores ranged from 1, where both mechanical and chemical control and follow-up treatments are needed, to 3 where little or no intervention is needed. These scores were based on inputs from Dr Rebelo. The rationale is that the easier a plant is to control the higher it scores. Therefore plants that can be successfully controlled using simply mechanical methods receive a higher score than plants that require chemical control involving many follow-ups.

3.2.2.3.3.2 Biocontrol available

Bio-control is an effective means of containing the spread and densification of invasive alien plants. Our rationale was that plants under effective biocontrol would be less of a priority than plants were no bio-control agent was available. At this stage, there is no bio-control for any of the ED&RR target plant species. For this reason all the target plants were weighted equally and given a score of one.

3.2.2.3.3.3 Fire promoted

This sub-criterion has two components. Firstly longevity of soil stored seedbanks or whether the plant is serotinous, and secondly if the plant resprouts (coppices) or produces suckers. Although some species can invade mature veld between fires many species depend on fire to create opportunities for establishment.

3.2.2.3.3.3.1 Resprouting

Species that sprout or sucker are able to survive disturbances such as fire and are more difficult to control as they often need to be treated with herbicides, often more than once. Sprouting species were given a score of 1 while non-sprouting plants were allocated a score of 2. The rational is that nonsprouting plants typically produce more seed and should be removed before they have the opportunity of establishing dense stands.

3.2.2.3.3.3.2 Seedbanks

Species having long lived soil stored seedbanks or that are serotinous are promoted by fire. Plants were scored from 1 to 3 depending on whether they had long-lived soil stored seedbanks (1), whether they were serotinous (2) or whether they had short-lived or no soil stored seedbanks (3) which are known to

be stimulated by fire. Our rationale is that plants that can survive fire either through serotiny or long lived soil stored seedbanks are more problematic and should not be given attention ahead of plant species that have short-lived seeds.

3.2.2.3.4 Invasive potential

A plant species ability to spread is determined primarily by the way its seeds are dispersed and the numbers of seeds it produces. Its ability to establish viable populations is largely due to its progeny being able to successfully overcome competition in this new environment.

3.2.2.3.4.1 Dispersal rate

Plants having well dispersed seed either by means of water, wind, birds or mammals (excluding humans) were given a score of 2, while plant having only local or short-range seed dispersal were given a score of 1. The rationale for this is that invasive plants that have short dispersal distances are less of a priority than those that can be dispersed over long distances.

3.2.2.3.4.2 Potential habitat envelope

This sub-criterion refers to the number of different habitats that a plant species normally occurs in. A score of 1 was given to those plant species that are confined to one habitat and a 2 to those plants that occur in more than one habitat. The rationale is that plants that occur in more than one habitat in their native environments are more likely to invade more than one habitat within the Cape Metropole.

3.2.2.3.4.3 Pre-adapted to climate

Scoring was done by means of an ordinal scale of 1 to 3. Plants originating from Mediterranean climates were given a score of 3, those from temperate climates 2 and those from sub-tropical or tropical areas 1. The rationale for this scoring is that plants that originate elsewhere but from a similar climate to that of Cape Town pose the greatest invasion risk.

The following results section is sub divided into two main sections. The first section, "*Classification tool for the management of Invasive Alien Plants*" presents data that can be used to classify IAPs into management categories and details which data is used to develop this classification protocol. The classification protocol is then used to identify widespread, emerging and eradication target IAPs from a large database of Cape Town IAPs. The second section presents the final ranked list of 77 ED&RR species which need to be prioritised for management which were prioritized from a model developed by Greg Forsyth and David Le Maitre (CSIR, Stellenbosch). The highest priority species occur at the top of the list and the lowest priority species occur lower down or at the bottom of the list.

3.3 Results

3.3.1 Classification tool for the management of Invasive Alien Plants

The desktop review yielded six different classification tools or studies based on a IAPs feasibility of eradication that have been developed and presented across the globe for the classification of widespread and emerging invaders. The first framework identified was that of Richardson et al., (2000) - the rate of IAP spread. This framework identifies "invasive species or "widespread plants" as naturalized plants (plants which have surmounted various biotic and abiotic barriers to irregular reproduction) that begin to proliferate from their original site of introduction. The study stipulates that a plant becomes invasive when it spreads more than 100m over or less than a 50 year period (by plants which spread via seed or other propagules).

The second classification protocol identified was that of the "infestation area rule" (1000ha rule [10km2]) (Rejmanek & Pitcairn, 2002). This study showed that professional eradication of IAPs smaller than one hectare is usually possible. Furthermore, the studies illustrated that one third of all infestations between 1 hectare and 100 hectares and one quarter of infestations between 101 and 100 hectares have been successfully eradicated. The costs of eradication becoming unrealistic to achieve successful eradication beyond 1000 hectares, eradication should not be attempted if the target IAP plant species exceeds this number.

Thirdly, Panetta and Timmons (2004) evaluated the feasibility of eradication for terrestrial weed incursions. The study undertaken on a number of IAP eradication projects, showed that the feasibility of eradication declines rapidly with increasing area of IAP infestation. In addition to this, the study outlines a number of factors, which influence the effort required to accomplish successful eradication. These factors are logistic, detection, biological and control factors which contribute to the impedance of full eradication.

Cacho et al., (2006) propose that IAP detectability can play a crucial role in the success of an IAP/ weed eradication program. Knowledge of the target species biology (seeders vs re-sprouters) is important in providing additional information relevant in designing a search and detection program for the target species to determine how widespread a search should be to encompass any potential escapees – this may potentially influence the length of an IAP species eradication program. Furthermore, the value of detection of an IAP at an early stage may be undervalued because denser and more widely spread infestations require more time to be successfully eradicated once located.

Pannetta and Cacho, (2014) refined the strategies for containing weed incursions with the use of theoretical and semi-quantitative models. This study identified two parameters (seed persistence and the length of juvenile period/ time to reproduction) as crucial in determining the suitability of eradication as a strategy for managing outliers surrounding an IAP infestation.

Lastly, Panetta, (2015) presents an alternative algorithm for assessing the feasibility of eradication based on data acquired from two previously attempted eradication programs which had failed. This study proposed with "eradication syndromes" which have alternative eradication feasibilities. The syndrome with the highest probable successful feasibility of eradication are IAPs with a long juvenile period, low seedbank persistence, short-distance and or human-mediated dispersal. The syndrome with the lowest feasibility of eradication are IAPs with a short juvenile period, a high seedbank persistence and non-human- mediated biotic dispersal.

The two main criteria used to classify IAPs are the infestation area (Rejmanek and Pitcairn, 2002) and the IAP rate of spread (Richardson et al., 2000). These two specific criteria are the most pertinent criteria that are repeatedly identified as being important criteria to measure IAP eradication success. Furthermore, specific data on species traits such as "seed persistence; length of juvenile period; IAP detectability, seedbank persistence, mode of dispersal" are not readily available or easily attainable for such a large number of species in the City of Cape Town. However, IAP infestation area as well as rate of spread can be easily obtained and generated with the aid of The *Flora of the Cape Peninsula* (Adamson and Salter, 1950). As such, infestation area and rate of spread were used to classify widespread, emerging and eradication targets in this study (Fig. 3.1).
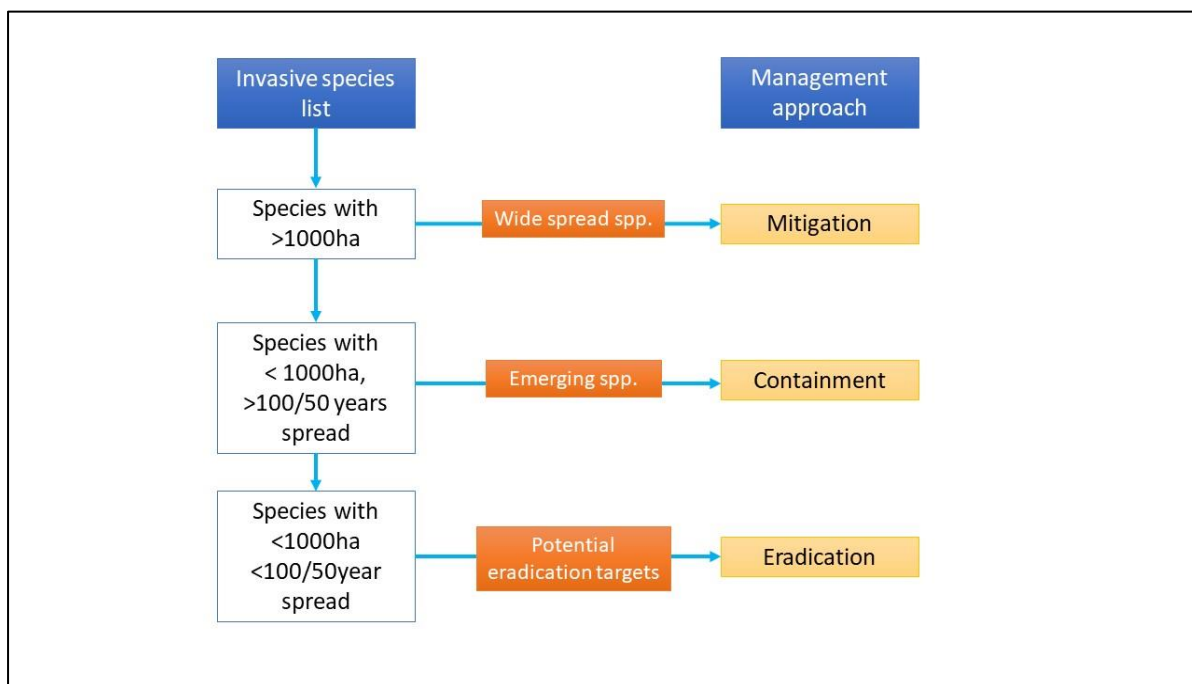
Figure 3.1: Classification framework developed for the grouping of Invasive Alien Plants into three types of IAPs and their subsequent management approaches.

3.3.1.1 Managing the database of invasive alien plants in South Africa (applying the new framework)

3.3.1.1.1 Widespread Invasive Alien Plants

A preliminary list of widespread invaders was obtained by applying filtering criteria to the peninsula database: the number of populations and/ or the size of the infestation area (occupied area in ha). We excluded all species that had an abundance of less than 1000ha (Rejmanek and Pitcairn, 2002). The applied filter reduced the original list of 515 IAPs to 483 IAPs (Fig. 3.1).

A total of 94% (N=483) of all the species recorded in our database were classified as widespread species (Table 3.1) and included well-known IAPs such as *Acacia mearnsii, A. cyclops* and *A. saligna*.

Table 3.1: The top 30 widespread invasive alien plants ranked from largest to smallest based on the size of the infestation. Spread was determined by subtracting the amount of linear spread of 2010 from the infested area in 1950 for that species. Spread is the linear distance of recorded infestations where 1 locality is equal to one hectare. Where the spread – recent change is equal to zero, no spread over a 50 year period is recorded. The study area covered the entirety of the City of Cape Town municipality.

Name	Growth form	Spread: Recent change (d/km)	Area invaded (2010) (ha)
Acacia cyclops	Tree	0	2500
Acacia longifolia	Tree	0	2500
Acacia saligna	Tree	0	2500
Ammophila arenaria	Grass	0	2500
Briza maxima	Grass	0	2500
Canna indica	Shrub	50	2500
Conyza albida	Herb	50	2500
Conyza bonariensis	Herb	0	2500
Cortaderia selloana	Grass	30	2500
Corymbia ficifolia	Tree	50	2500
Eucalyptus conferruminata	Tree	0	2500
Hakea drupacea	Tree	45	2500

Hakea gibbosa	Tree	45	2500
Hypericum canariense	Shrub	50	2500
Hypochaeris radicata	Herb	20	2500
Leptospermum laevigatum	Shrub	0	2500
Myoporum montanum	Tree	0	2500
Opuntia monacantha	Succulent	50	2500
Pinus pinaster	Tree	0	2500
Pinus pinea	Tree	0	2500
Pittosporum undulatum	Tree	50	2500
Quercus robur	Tree	0	2500
Sesbania punicea	Tree	50	2500
Yucca gloriosa	Succulent	50	2500
Acacia mearnsii	Tree	20	900
Acacia melanoxylon	Tree	10	900
Agave americana var. americana	Succulent	30	900
Agave americana var. expansa	Succulent	30	900
Agave sisalana	Succulent	30	900
Avena barbata	Grass	0	900

3.3.1.1.2 Emerging invasive alien plants

In order to construct the list of emerging invaders we first excluded the 483 widespread invaders from the original peninsula database. This reduced the list to a total of 33 IAPs. The remaining database was then subsequently filtered according to two criteria: (a) the total abundance of that species (occupied area in Ha); and (b) the rate of spread (refer to fig. 3.1). As the list of widespread IAPs (483 species) had already been excluded we subsequently excluded all species with a spread rate of less than 100ha per 50 year period. This reduced the original list of 33 IAPs to 18 IAPs.

A total of 18 emerging species were classified from a list of 515 IAPs.   This represents only 3.5% of the IAPs from our original database. Interestingly, of the emerging IAPs classified from the original database, only five were recorded in 1950 (*Chamaecytisus prolifer var. palmensis, Sequoia sempervirens, Stipa papposa, Taxodium distichum, Telopea speciosissima*).

Table 3.2: The 18 emerging invasive alien plants classified according to our classification protocol. Spread was determined by subtracting the amount of linear spread of 2010 from the infested area in 1950 for that species. Spread is the linear distance of recorded infestations where 1 locality is equal to one hectare. The study area covered the entirety of the City of Cape Town municipality. All species which occur in this table had a had a spread -recent change of 1km and 1ha area invaded.

IAPs Name	Growth form
Acacia adunca	Tree
Acacia stricta	Tree
Acacia subporosa	Tree
Acacia viscidula	Tree
Buddleja davidii	Shrub
Buddleja madagascariensis	Shrub
Chamaecytisus prolifer var. palmensis	Tree
Eucalyptus regnans	Tree
Eucalyptus rudis subsp. rudis	Tree
Leucojum aestivum	Herb
Melaleuca hypericifolia	Tree
Melaleuca nesophila	Tree
Opuntia imbricata	Tree
Sequoia sempervirens	Tree
Stipa papposa	Grass
Taxodium distichum	Tree

Telopea speciosissima	Shrub
Wigandia urens var. caracasan	Shrub

3.3.1.1.3 Potential eradication targets

To obtain the final list of eradication targets the same two filter criteria from the classification of emerging IAPs were applied to the remaining list of IAPs (once widespread invaders as well as emerging invaders had all been excluded). However, we excluded all IAPS with a rate of spread that exceeds 100ha per 50 year period. The original list of 515 IAPs was reduced to 33 IAPs with the exclusion of widespread and emerging IAPs. The final list of eradication targets totals at 15 IAPs.

From the original list of 515 IAPs, only 15 (3% of the original database) of these species were classified as potential eradication targets. Of the 15 classified IAPs only seven (*Acacia paradoxa, Amaranthus muricatus, Barbarea vulgaris var. Arcuata, Erodium cicutarium, Gastrodia sesamoides, Lathyrus angulatus, Ranunculus repens*) were recorded in 1950. These species should make relatively soft targets from the City of Cape Town as the distribution of the total infestation and rate of spread is limited, making eradication an attainable management goal.

Table 3.3: The 15 Potential eradication target plants classified to our classification protocol. Spread was determined by subtracting the amount of linear spread of 2010 from the infested area in 1950 for that species. Spread is the linear distance of recorded infestations where 1 locality is equal to one hectare. The study area covered the entirety of the City of Cape Town municipality. P indicates the presence of the species that have not yet invaded more than 1 hectare (ha). All species presented in this table have not spread in the last 50 years. As such, they all display zero for their spread- recent change.

IAPs Name	Growth form	Area invaded (2010) (ha)
Acacia paradoxa	Tree	1
Ageratum houstonianum	Herb	P
Amaranthus muricatus	Shrub	1
Barbarea vulgaris var. arcuata	Shrub	1
Bryophyllum delagoense	Succulent	P

Cardiospermum grandiflorum	Creeper	P
Crotalaria agatiflora	Shrub	P
Erodium cicutarium	Herb	1
Gastrodia sesamoides	Herb	1
Lathyrus angulatus	Herb	1
Melaleuca ericifolia	Tree	P
Passiflora edulis	Creeper	P
Pennisetum villosum	Grass	P
Pinus roxburghii	Tree	P
Ranunculus repens	Herb	1

3.3.2 *Prioritization of the control of ED&RR species in the Cape Town Metro.*

The following table (Table 3.4) details the final rankings of the 77 early detection and rapid response species. The species which received the highest overall score are the species which have the highest rank.

Table 3.4 Final ED&RR target plant species in the Cape Metropol. The Following table details the top 50 IAP species in ascending order of priority or by order of rank.

Species	Rank
Schinus terebinthifolius	2.14
Melaleuca hypericifolia	2.07
Hakea sericea	2.07
Acacia paradoxa	2
Ailanthus altissima	2
Sesbania punicea	2
Casuarina equisetifolia	2
Cereus jamacaru	2
Acacia pycnantha	2

Hakea drupacea (suaveolens)	1.92
Hakea gibbosa	1.92
Schinus molle	1.92
Hedera helix	1.92
Celtis sinensis	1.92
Acacia mearnsii	1.92
Prosopis glandulosa var. torreyana	1.92
Cinnamomum camphora	1.92
Robinia pseudoacacia	1.92
Solanum mauritianum	1.92
Acacia elata	1.85
Acacia stricta	1.85
Cortaderia selloana	1.85
Acacia baileyana	1.85
Cirsium vulgare	1.85
Grevillea banksii	1.85
Grevillea robusta	1.85
Acer negundo	1.85
Cardiospermum grandiflorum	1.78
Lythrum salicaria	1.78
Spartium junceum	1.78
Psidium guajava	1.78
Melia azedarach	1.78
Eriobotrya japonica	1.78
Echinopsis spachiana	1.78
Cotoneaster pannosus	1.78
Cinnamomum camphora	1.78

Alnus glutinosa	1.78
Hedychium flavescens	1.78
Acacia implexa	1.71
Centranthus ruber	1.71
Genista (Cytisus) monspessulana	1.71
Rivina humilis	1.71
Opuntia ficus-indica	1.71
Cotoneaster franchetii	1.71
Tecoma stans	1.71
Cestrum laevigatum	1.71
Eucalyptus camaldulensis	1.71
Egeria densa	1.71
Pyracantha angustifolia	1.71
Pittosporum undulatum	1.64
Canna indica	1.64
Betula alba	1.64
Araujia sericifera	1.64
Hedychium coronarium	1.64
Hakea salicifolia	1.64
Pyracantha coccinea	1.64
Prunus armeniaca	1.64
Lolium perenne	1.64
Tropaeolum majus	1.64
Syzygium paniculatum	1.5
Hedychium coccineum	1.57
Ageratina adenophora	1.57
Acacia podalyriifolia	1.57

Prunus persica var. persica	1.57
Anredera cordifolia	1.57
Atriplex lindleyi subsp. inflata	1.57
Myoporum insulare	1.57
Atriplex nummularia subsp. nummularia	1.5
Pennisetum purpureum	1.5
Cotoneaster lacteus	1.5
Cuscuta campestris	1.5
Catharanthus roseus	1.5
Conyza albida (syn. sumatrensis)	1.42
Hypericum canariense	1.42
Crataegus pubescens	1.42
Cestrum aurantiacum	1.42
Plectranthus comosus	1.35

3.4 Discussion

3.4.1 Classification of widespread, and emerging and eradication targets

The classification of widespread and emerging invaders and eradication targets has been generally based on a plant's feasibility (where feasibility may be considered in terms of IAP management achieving success [Panetta and Timmons, 2004]) of eradication or ease of control. As such, Cape Town and the Cape Floristic Kingdom (CFR) would be an important area of focus for IAP eradication programmes, which should always maximise the net benefits. The fact that some successful weed eradication programmes have taken decades to be seen to fruition is indicative that the risk failure to IAP eradication will always remain high without procedures in place to ensure long-term funding and maintenance of staff capacity (Panetta and Timmons, 2004).

When we study the resulting set of widespread IAPs of the classification protocol we see that 94% of the original database were determined to be widespread IAPs (Table 3.1), including well-known IAPs such as *Acacia mearnsii, A. cyclops* and *A. saligna.* More funds have been designated to controlling *A. mearnsii* than all other IAPs together (Nel et al., 2004). Expectedly, grasses (agricultural weeds) such as *Briza maxima*, *Avena barbata* and *Avena fatua* all display massive infestation areas. *Pinus pinea* and *P. pinaster* as well as *Hakea gibbosa* and *H. drupacea* are typical widespread species found not only in

Cape Town but throughout most of South Africa. According to Richardson and Van Wilgen (2004) the genera *Hakea, Pinus* and *Acacia* are all principle invaders of the fynbos biome. However, unexpectedly *Hypericum canariense* is considered one of the widespread IAPs (information emerged after the selection of this species for study). This species is considered by most land managers within the City of Cape Town and the South African National Biodiversity Institute (SANBI) (personal coms) to be an emerging IAP.

When we study the list of emerging invaders (Table 3.2) it is important to note the presence of four acacia species *A. adunca, A. stricta, A. subporosa* and *A. viscidula* as emerging IAPs. If containment is possible for these IAPs, it is paramount that it be implemented in order to prevent these species from becoming widespread invaders of the fynbos biome (and potentially other parts of South Africa). Furthermore, monitoring programmes should be developed and implemented to ensure that these species do not progress to being widespread IAPs.

Table 3.3 presents a list of potential eradication targets. It is important to note the presence of *Pinus roxburghii* and *Acacia paradoxa* and two species that fall within the aforementioned *Pinus* and *Acacia* genera, which typically show high levels of invasion in the fynbos biome (Richardson and Van Wilgen, 2004). On the other hand, species such as *Erodium cicutarium* may pose a potential risk in the future as the closely related *E. moschatum* is already considered a widespread agricultural weed (Bromilow, 2010). Additionally *Agertum houstonianum* is also considered a serious weed of agricultural crops even though it occurs in such small numbers.

The classification described in this chapter should allow a focused IAP management when dealing with large numbers of species where funding is limited. The classification at a municipal scale allows plant species to be prioritized in terms of management and circumnavigates the issues when prioritizing species across multiple spatial scales and different ecosystems (Nel et al., 2004). Classification and subsequent management of IAPs should always be chosen for their likelihood where maximum net benefits are attained through management actions (Cacho et al., 2008). These net benefits can be compared to the "do nothing scenario" to determine whether a particular management strategy will be worth-while.

Management and resources can then be focussed on broad range management across larger scales where widespread species are more prevalent, or at smaller scales where eradication or habitat specific control or extirpation may still be possible before the species surmounts barriers to invasion and becomes widespread. Emerging invaders are often overlooked as they do not yet pose a major threat or have few biological, and economic consequences in comparison to wide-spread invaders (Nel et al., 2004) but have the potential to create significant impacts in the future if not controlled at the earliest stage of invasion. Cacho et al. (2008) highlights the importance of having successful weed control programmes,

managers must be able to identify IAPs at an early stage of invasion, to determine the invasive potential of that species and have the capacity (economic and time) to see the program to completion.

These classification categories also aid in defining specific and focuses management guidelines (Nel et al., 2004). For example, emerging invaders with high potential for proliferation through propagules and a large area of invisible habitat should be prioritised for further research or halting plants at the earliest stage of invasion (Olckers, 2004). Furthermore, species that are listed under National Environmental Management; Biodiversity Act, 2004 (Act no. 10 of 2004) (NEMBA) category 1 species and category 2 species which fall into the "eradication targets and "emerging species" should be targeted for immediate eradication. NEMBA is the South African Act which lists IAP species which should be eradicated.

In the City of Cape Town (CCT), 483 well established/ naturalised species widespread species which are considered major invaders were classified. Of these, 18 are considered, using the classification protocol as emerging invaders and 15, eradication targets. It is important to induct these species into monitoring programmes where the necessary resources for eradication are not available. The use of species characteristics data to classify species is helpful in reducing the subjectivity and ad-hoc management by experts alone. In turn, local experts and land managers can collectively provide species data on lesser-known species in order to better identify how to manage species in the short and long term in the context of looking at a number of species altogether (Cacho et al., 2008).

Biological control is also an option for emerging invaders and eradication targets as research has shown that biological control is most effective during the earliest stages of invasion (Olckers, 2004). However, biological control of emerging species has not been practiced world-wide due to the fact that biological control is often used as a "last resort" where traditional management was determined to be unsustainable or unsuccessful and that biocontrol can be costly and take a lot of research prior to release of biocontrol agents (Olckers, 2004). The targeting of emerging invaders and eradication target species identified in this study should be used as an intuitive list of species which can be targeted for eradication and possibly be studied further for the use of biocontrol if the IAP species has already spread beyond what is feasible in terms of successful eradication upon more detailed investigation.

Some of the limitations of this study is the general lack of information of species generally (Cacho et al., 2008) and in the City of Cape Town. As such the classification tool is rudimentary. Information of the longevity of the seedbank for species (Panetta et al., 2011; Panetta et al 2007), their ease of detectability (Panetta and Timmons, 2004) and the length of juvenile period (Panetta and Cacho, 2014) is information that can only be attained through species specific studies. Without such information, IAP eradication programmes can become lengthy and costly which may eventually lead to their failure (Panetta et al., 2011). However, the more information becomes available on a wider variety of species, the easier it will become to classify species for management and target potential high-impact species

for eradication. As more information about a species becomes available, IAP control and eradication decisions should be refined (Cacho et al., 2008).

Furthermore, since the City of Cape Town is a mosaic of natural areas, an area based approach for the prioritization and management of species may need to be undertaken secondary to this study to ensure sensitive ecosystems are kept free of IAPs. This area-based approach should identify areas which have a high susceptibility to invasion by emerging invaders and eradication targets. Furthermore, this should ensure that land managers can identify sites where future management will be most cost-effective in the future (Nel et al., 2004). In light of the development of this classification tool, land managers need to consider a broad range of management actions simultaneously. Eradication targets and emerging species should not be the sole focus of management, but species which are widespread will also need to be contained in order to limit their impact on natural areas within the city.

3.4.2 The prioritization of ED&RR plant species in the Cape Town Metro.

A number of approaches have been developed and implemented over the last few years in South Africa in order to usher in the effective prioritization of IAPs (E.g. Forsyth et al. 2012; Nel et al., 2004 and Potgieter et al., 2018). This study is a species-based approach which aims to build on similar studies undertaken in the past. We adopted and applied a larger database for prioritizing IAP species across a complex urban landscape. The purpose of this prioritization is to streamline management and change the focus of ad hoc species management in the City to a more focused management strategy where IAPs can be targeted with the best return on efforts across an urban landscape such as Cape Town (Appendix A). Furthermore, this prioritization framework identified species that pose significant risks in terms of their ability to successfully overcome biotic and abiotic barriers to dispersal and that may pose significant risks to natural and semi-natural areas within Cape Town.

The intentional control of IAPs can lessen the adverse environmental and economic impacts whereas delays in funding IAP management strategies may be very costly in the long run (Funk et al., 2014). The allocation of funding within cities for IAP management is a real challenge due to conflicting allocation priorities and the absence of long term and focused IAP management strategies (Potgieter et al., 2018). This prioritization is specifically important because IAP species management is mostly reliant on available short term operational funding from different departments within the City of Cape Town instead of abundant, independent funding (Ilrich et al., 2017). Sufficient funding should be directed at controlling the highest priority species first rather than following an approach of trying to address all species simultaneously but with insufficient resources.

The species-based IAP prioritization approach adopted and tested within this study is collaborative, reproducible, systematic, and transparent. The application of the Analytical Hierarchical Approach used here made use of expert opinion and science-based methods to find practical solutions where the optimal solution is often impossible or overly complex. Furthermore, the AHP system can reproduced and

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expanded upon when new information, additional data and more stake holders become available. This process can establish a set of clear and transparent priorities that can be used to guide the prioritization of IAP species for management (Potgieter et al., 2018). The expert workshop identified the most important criteria and sub-criteria needed to when prioritising emerging IAPs and potential eradication target species. It is therefore important that the data needed to support these criteria and sub-criteria be systematically included in the "I-Naturalist" and City of Cape Town's databases or at the least such data should be recorded when identifying and measuring the extent of invasions in the City of Cape Town.

The highest ranked species from the AHP model include *Hakeas* such as *Hankea sericea* (rank 3). *Acacias* such as *Hakea drupacea* (suaveolens) (rank 10) and *Hakea gibbosa* (rank 11). According to the results of the rankings, a number of *Acacia* species also top the list. These acacias are *Acacia paradoxa* (rank 4) *Acacia pycnantha* (rank 9) and *Acacia mearnsii* (rank 15). Furthermore, a number of acacias occur further down the list but remain far from the bottom priority IAP species. Acacia species were introduced into Cape Town at the turn of the century for dune stabilisation and have since posed major invaders across South Africa (van Wilgen et al., 2020). Since its introduction for the use of producing tannins from bark, *Acacia mearnsii* has become one the most widespread IAP species in South Africa (Nel et al., 2004). Hakeas were introduced into South Africa as early as the 1850s where species such as *Hakea gibbosa* spread into many coastal water catchments in the Western and Eastern Cape (Gordon and Fourie, 2011). At least four species of hakea have become important invaders in South Africa (Le Maitre et al., 2008). It is therefore critically important to prioritize these species for management.

In conclusion, the priorities assigned to the 90 emerging IAPS and potential eradication target species (Table 3.4) should be used as to guide to focus immediate future management of important or on the highest priority species. As new or revised datasets become available, they should be incorporated into the AHP model and used to generate a revised set of rankings. This is critically important as this model has only now been tested with a larger number of species and has shown that it is effective in prioritizing IAPs in the City of Cape Town. In the future, the City of Cape Town could potentially populate this model with hundreds of species to get a better understanding of which IAPs to manage first and where to allocate limited funding. In addition, as understanding on these important species improves, the weightings assigned to the criteria and sub-criteria.

This new approach in managing or considering many species or data resulted in an intuitive framework that encompasses all the characteristics of potentially threatening urban invaders. It mitigates the complexities involved in the decision-making process in urban environments and allows land managers and conservationists to generate priority IAPs for management across a complex urban landscape. The development and testing of this prioritization framework proved useful in guiding and empowering

decision makers and conservationists especially where funding is limited across the globe (Potgieter et al., 2018).

Chapter four: The invasive alien *Hypericum canariense* in South Africa: Management, cost and eradication feasibility

4.1 Abstract

The shrub *Hypericum canariense* (*Hypericaceae*) is an emerging invader in several parts of the world (United States of America, New Zealand and Hawaii). In this chapter, I determined the current distribution in South Africa, assessed soil seedbanks, determined size at reproduction, evaluated the current management protocols, provide recommendations for control and assessed the feasibility of eradication. In South Africa *H. canariense* has naturalised at seven localities in the South Western Cape. The species is considered a rare ornamental and naturalised populations have escaped cultivation. All current populations are in disturbed areas, mostly along roadsides and informal dumping sites. Once established, *H. canariense* forms large persistent seed banks with between 20 and 2040 seeds/m2. This species resprouts after cutting, with about half the plants resprouting if no herbicide or 3% Garlon (Ismapyr) is applied. If 2% Hatchet (Triclopyr) is applied, resprouting still occurs, but resporuting is significantly lower. Annual follow-up clearing is required to control resprouting but should continue for an estimated 10 years until the seedbank is exhausted. I estimate that initial clearing of the naturalised populations of *H. canariense* would take 17.5 person days and from thereon 4.5 person days per year for the following 10 years to eradicate this species form South Africa. The initial cost of clearing all populations in South Africa will amount to ZAR35000, with a total cost off eradication of ZAR57300. I recommend that the City of Cape Town eradicates this species in line with National Environmental Management: Biodiversity act (Act 10 of 2004)

4.2 Introduction

Urban areas have been identified as hotspots of invasions (Kowarik, 2011). Whether intentional or inadvertent, anthropogenic activity is responsible for most introductions of invasive alien plants (IAPs) due to significant land use transformation and the increase in trade associated with the development of the global economy (Vitousek et al., 1997; Kent et al., 1999; Dehnen-Shmutz et al., 2007). This high volume of human mediated dispersal has brought about an urgent need to explain where and how IAPs might invade native ecosystems (Dlugosch and Parker, 2008a). The understanding of pathways and movements of IAPs is crucial to inform management of IAPs (Rouget et al., 2015).

Ornamental horticulture has been identified as a major pathway for invasions world-wide (Dehnen-Schmutz, 2011). Some species have been transported accidentally as stow-aways, but largely and most importantly, many plant species are introduced for horticultural or agricultural purposes. These intentionally introduced species benefit from human assistance during cultivation and some of these species cause problems as invaders where they escape from their site of introduction and proliferate into disturbed and natural areas (Afonso et al., 2020; Geerts et al., 2017; Richardson et al., 2006). It is

therefore crucial to detect and determine the potential threat of newly introduced species in urban areas or areas with high anthropogenic activity and land use change.

Detecting and controlling invasive alien plants during the early stages of an invasion not only prevents potential ecological damage, but it also means that fewer resources will be needed than waiting to control established and widely dispersed plant invasions (Forsyth, 2013). Furthermore, the naturalization stage (when a species is emerging) is considered to be the most effective time to attempt to reduce the ultimate impact of an invasive species (Panetta & Timmons, 2004). Similarly, in South Africa the feasibility of eradicating a single species has been investigated (see for example Kaplan et al., 2012, Zenni et al., 2009; Geerts et al. 2013). However, these studies largely focus on areas outside of urban areas (but see Geerts et al. 2017). The assessment of feasibility of eradication will be compounded by different factors in urban areas. Some of these compounding factors include, but are not limited to, access to sites where IAPs are present, conflicts of interest or buy in from the public, and the ease of prevention due to human mediated dispersal pathways (Irlich et al., 2017).

To address some of these issues I focus on a recent naturalising garden escapee, *Hypericum canariense L.* (Hypericaceae). The genus *Hypericum* is easily cultivated and grown from seeds as ornamentals for borders, rock gardens and ground covers (Star et al., 2003). *Hypericum canariense* is a fast-growing, multi-stemmed shrub reaching about 4m in height. *Hypericum canariense* is native (and endemic) to Canary Islands and Madeira which is an unusually small native range (Starr et al., 2003; Dlugosch & Parker, 2008a) for an invasive species. It is widely cultivated as an ornamental and is now an emerging invader in California (United States of America), Maui - Hawaii (United States of America), New Zealand and Western Australia (Dlugosch and Parker, 2008b, Starr et al., 2003). *Hypericum canariense* is not the only *Hypericum* species found in South Africa; *Hypericum perforatum* and *H. revolutum Vahl subsp. revolutum* are examples of other *Hypericum* species that are invasive. *Hypericum perforatum* has already been successfully controlled via the use of bio-control agents after being abundant – mostly in the Western Cape – around the mid-1900s (Van Wilgen et al. 2020). Several species within this genus thus have the ability to become invasive in parts of the world with a Mediterranean climate.

Little is known about the control of *Hypericum canariense*. However, the invasions of this species have come to the attention of conservation agencies in the United States of America (e.g. the Nature Conservancy weed alert) and it has subsequently been placed on IAP watch lists (Dlugosch and Parker, 2008b). In addition to this, the plants in California are being cut at the base and treated with glyphosate at high concentrations (Starr et al., 2003). On Maui, plants are being controlled by being cut at the base, rebounded and then being sprayed with foliar herbicide (Starr et al., 2003). Control can be complicated by persistent seed banks since *H. perforatum,* for example, has substantial seed banks (Buckley et al., 2003a) that can persist up to 30 years (Buckley et al., 2003b). Moreover, another *Hypericum* species (*Hypericum maculatum*) has also shown significant soil seed bank production

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(Milberg, 1995). Although Hansford and Iaconis (2004) suggest that *H. canariense* create soil seed banks, no studies have investigated the soil seed bank size of *H. canariense*.

The aim of this study is to determine whether *H. canariense* has the potential to become a major invader in South Africa and whether it should be targeted for eradication. To achieve this I (1) investigate the introduction history and current distribution of *H. canariense* in South Africa; (2) determine size at reproduction, reproductive output and soil seedbank; (3) determine the most effective management option (mechanical or chemical or a combination thereof); (4) calculate the cost of eradication from South Africa; and (5) provide recommendations for legislation and national management strategies.

4.3 Methods

4.3.1 Study species

Hypericum canariense L. (Hypericaceae) is a large perennial shrub native to the Canary Island and Madeira (Starr et al., 2003). In South Africa it produced bright yellow flowers (Fig. 4.2d) during Spring and Summer and produces seedpods (Fig 4.2c) in the Autumn. *Hypericum canariense* produces numerous small pale-reddish brown seeds which are easily dispersed via wind and water (Starr et al., 2003). The seedpods are reticulate with each seed pod containing hundreds of potentially viable seeds (Hear, 2016).

4.3.2 Hypericum canariense distribution in South Africa

The distribution of *H. canariense* in South Africa was recorded with the use of the Global Biodiversity Information Facility (GBIF), the I-naturalist database and the Southern African Plant Invaders Atlas, SAPIA (Henderson, 2007), the database of herbarium records, old plant lists, and unpublished records of the authors. Local experts (SANBI, CapeNature, SANParks and the City of Cape Town) and conservation officers were consulted to determine all the potential localities of *H. canariense*. Localities were visited to confirm presence and map the populations.

I surveyed all known *H. canariense* infestations between October 2016 and May 2017. The mapping of *H. canariens*e populations involved walking parallel transects extending at least 100m in all directions beyond the last plant encountered, except along streams (where the search distance was increased to 500m). Each plant encountered was recorded with the use of a Global Positioning System (GPS). The current occupied area was calculated by taking an average canopy area (calculated by determining the average canopy radius for all the plants in each population) for each population (taken from a transect for the big populations and from the total number of plants from the smaller infestations).

4.3.3 Population structure, reproductive size and reproductive output

Population structure as well as the reproductive size and output of *H. canariense* were determined at each infestation site. To assess population structure a single transect spanning 3m wide and 20m long was walked through the densest part for all populations. For the smaller populations, every single plant

was measured. Stem diameter, perpendicular canopy diameter, plant height, the presence of flowers and / or seed pods were recorded. These measurements were taken during the months of October and November 2016 when *H. canariense* is flowering.

Reproductive output per plant was calculated by removing all the seed pods from ten randomly selected plants at the Redhill section of the TMNP and the Kenilworth conservation area. Twenty seed pods were randomly selected from the sample and the seeds contained in each seed pod was tallied and averaged to indicate the average number of seeds produced by any given seed pod. The average number of seed pods was then multiplied by the average number of seed pods to determine the average reproductive output per individual plant. The reproductive output of each population was then taken from the product of the average reproductive output and the number of plants per population.

4.3.4 Seed viability and soil seed bank dynamics

To determine the size of the seed bank, five soil samples per population were randomly taken under the canopy of an adult *H. canariense* plant and combined. At each sample the organic-horizon material was removed, a soil core of depth of 20cm and width of 10cm, was collected and taken to the greenhouse and spread out in seedling trays. The trays were arranged through a factorial arrangement in a randomized block design and kept at 10-32^{0}C and watered every second day. Once the seedlings had germinated the emergent seedlings were identified and counted. The green house experiment ran for a duration of five months (December 2016-April 2017).

4.3.5 Effectiveness of different control methods

To date there are no herbicides registered for the treatment or control of *H. canariense* in South Africa. Therefore, I compared the effectiveness (measured by mortality rate) of two alternative and most commonly used herbicides by the City of Cape Town Invasive Species Unit (Hatchet [Ticlopyr] and Garlon [Ismapyr]). To determine the efficiency of clearing methods a cut stump (Fig. 4.2a) chemical trial was undertaken at the Redhill section of the TMNP population and at the Kenilworth Conservation areas as these two infestations were the largest. The chemical trial involved setting out thirty random 25m^2 (5m x 5m) plots in three distinctive blocks (one block each for two different herbicide applications and one block as a control) at the densest parts of the infestation. The same experimental design was used at the Kenilworth Conservation Area with the exception that the plots were 6.25m^2 (2.5m x 2.5m) in size. Within each plot, plants were cut at the base as close to ground level with loppers, machetes or saws. Plants that were small enough were hand pulled (Fig. 4.2b)

Stumps were treated with either a 3% triclopyr (Garlon 4) or a 2% imazapyr (Hatchet) solution. The third management block consisted of cut stump with no herbicide application (control). The concentration levels were based on standard operating cut stump application procedures by the City of Cape Town Invasive Species Management Unit. Efficacy of the different cut stump herbicide applications were assessed three and six months after control operations by tallying the number of

emerging seedlings and re-sprouting individuals. The mortality rate was then determined for each of the different chemical treatments and the control. Results were similar for the three and six month survey, so the six-month data are used.

4.3.6 Costs to eradicate Hypericum canariense from South Africa

In order to calculate the projected cost for eradication of *H. canariense*, I calculated the effort required to extirpate all known infestations of *H. canariense*. To calculate "effort", I monitored a team from the City of Cape Town Invasive Species Unit and recorded the amount of herbicide used as well as the number of person-days required. This effort was transformed into effort per person per day as a standard unit of measurement to which cost in ZAR was applied. The cost of the "effort" was calculated and determined for the initial clearing of infestations as well as for follow up clearing. The daily rate is taken from the Expanded Public Works Program (EPWP) rates, which are currently being used by the City of Cape Town (ZAR110.00/day for general workers [2019])

No adjustments were made to the cost calculations because I did not consider the complexities of the sites (general topography and slope of the site), travel time to and from site (by vehicle and walking) and the time taken to access the site. This is because all infestations are easily accessible. Cost of herbicides, as well as any equipment expenses is included in the cost of eradication of this species. Costs were increased by 7% per annum based on recent rates of inflation in South Africa.

4.3.7 Risk Assessment of potential invasiveness

I assessed the risk of *Hypericum canariense* in South Africa using the Australian Weed Risk Assessment (Pheloung et al., 1999) as well as the guidelines for the implementation of this system outside of Australia (Gordon et al., 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). The Australian Weed Risk Assessment was developed to determine the risk of species introduced into Australia and New Zealand, but has proven to be accurate across a broad geographic range (Gordon et al., 2010; Kaplan et al., 2012). It is important to note that I applied the guidelines for answering the questions for areas of the world outside Australia (Gordon et al., 2010).

4.3.8 Statistical analysis

Analysis of variance was used to determine the difference in the effectiveness of different chemical treatments. The analysis of variance was a mixed model ANOVA with mortality as a fixed effect. A one-way ANOVA was used to determine the effect that fire vs no fire had on seedling recruitment after 6 months from clearing. All the analysis were done in Statistica 13 (Statistica, 2019) and R (lmer package) (R development core team, 2013).

4.4 Results

4.4.1 Current distribution of Hypericum canariense in South Africa

I recorded a total of 4534 *H. canariense* plants across all seven known naturalized infestations. In the Redhill section of the Table Mountain National Park (TMNP), the total number of plants recorded was 2537 spread over and area of 910m^2 (Fig. 4.1a). Kenilworth had a total of 1826 individuals over 347m^2 (Fig. 4.1b). In Tygerberg I recorded only 34 plants. The Odendaal park, Lionshead, M3 and Tokai forest infestations were limited to a small number of individuals over a small area (Table 4.1, Fig. 4.1a).

At Redhill Nature section of the TMNP the distribution of *H. canariense* is along the roadside and adjacent to human habitation where disturbance to the natural veld is high. However, the area invaded at Kenilworth had pockets of high disturbance where illegal dumping and other invasive alien plants were present. Furthermore, *H. canariense* distribution can be associated with disturbed road verges at the M3, lions head and Odendaal park populations. The largest *H. canariense* plants were found at the Redhill section of the TMNP with stem diameters of up to 25cm. *Hypericum canariense* populations have a high proportion of smaller plants, suggesting a high rate of population expansion at Redhill, Kenilworth and the M3 populations.

4.4.2 Population structure, reproductive size and reproductive output

The different populations of *H. canariense* had substantially different size distributions and all comprised of multiple generations. The size of the plants at the onset of reproduction also differed between the two main infestations; 0.25m was the minimum size observed for a reproductive plant at Kenilworth (Fig. 4.3B) and 0.3m was observed at Redhill (Fig. 4.3A). Fewer than 3% of plants taller than 0.5m carried seeds (Fig. 4.3).

4.4.3 Seed viability and soil seedbank dynamics

Seedbanks were present in all seven naturalised infestations except for Odendaal park (Table 4.1). The number of *H. canariense* viable seeds ranged from 0 to 2040 (mean of 469) seeds/m^2 per site. The site that has the largest number of seedlings was Lions Head.

4.4.4 Effectiveness of different control methods

Hatchet (Triclopyr) application induced a higher mortality rate after 6 months compared to Garlon (Ismapyr) and the control (Fig. 4.4). For Hatchet about 10% of plants resprouted, whilst for Garlon and the control this was significantly higher.

4.4.5 Future costs for eradication of Hypericum canariense

The initial clearing of *H. canariense* infestation at the Redhill section of the TMNP of 813 plants took three person days and cost ZAR1.62 per plant. The total cost to extirpate all seven infestations (at 2020 price level) is estimated at ZAR35000. I therefore estimate that it will take ZAR57300 over 10 years to eradicate *H. canariens*e from South Africa. A total of 17.5 person days will be required for *H.*

canariense extirpation in the first year with at least 4.5 person days a year for the next 10 years. This number is initially high because *H. canariense* has such a large soil seedbank.

4.4.6 Australian weed risk assessment

The risk assessment score of 14 indicates that *H. canariense* has the potential to become invasive in South Africa (Appendix D). Species with a score exceeding six are considered within the "high risk" category of invasiveness. The traits which make this species so invasive are high rate of reproduction, high number of seeds produced and the fact that it can be pollinated by general pollinators. This species is also climatically suitable for the Fynbos biome of South Africa as the species home range has a Mediterranean climate.

4.4.7 Effect of fire on recruitment of Hypericum canariense.

The effect of fire was measured at the Kenilworth Nature Reserve and Redhill Nature Reserve (part of the Table Mountain National Park). Fire seems to have an adverse effect on seedlings as recruitment is significantly lower in areas that experienced a fire after clearing versus areas with no fires) (Fig. 4.5). Average seedlings per plot: $p = 0.00$.

Table 4.1: The locality, number of plants, size of population, average plant height, seedlings /m2, year first recorded, record origin and the land use of the *H. canariense* populations found in South Africa. Seedlings/m^2 was calculated by sampling 0.05m^2 bulk soil from the seven naturalized localities and converted to viable seeds per m^2.

Infestation	Coordinates	Number of plants	Size of the infestation (m^2)	Avarage plant height (cm)	Viable seeds per/m^2	Year first recorded	Record origin	Land use types and activities
Redhill section TMNP	34° 9'35.69"S 18°24'36.03"E	2537	910	96	400	1950	Experts	Roadsides and dumping areas and paths
Kenilworth	34° 0'12.02"S 18°28'59.16"E	1826	347	67	60	2010	Cape Town Invasive Species Unit	Dumping areas and conservation area
Tygerberg	33°52'45.60"S 18°35'37.15"E	79	68	58	20	2016	Cape Town Invasive Species Unit	Conservation area
M3	34° 1'8.78"S 18°29'17.44"E	49	123	71	480	2010	Cape Town Invasive Species Unit	Roadside
Lionshead	33°56'0.73"S 18°23'26.30"E	8	7	69	2040	2017	This study	Roadside
Odendaal park	33°50'10.24"S 18°35'1.35"E	16	2.4	102	0	2017	This study	Roadside
Tokai	34° 5'26.99"S 18°26'30.49"E	28	34	55	280	2001	Experts	Roadside and path
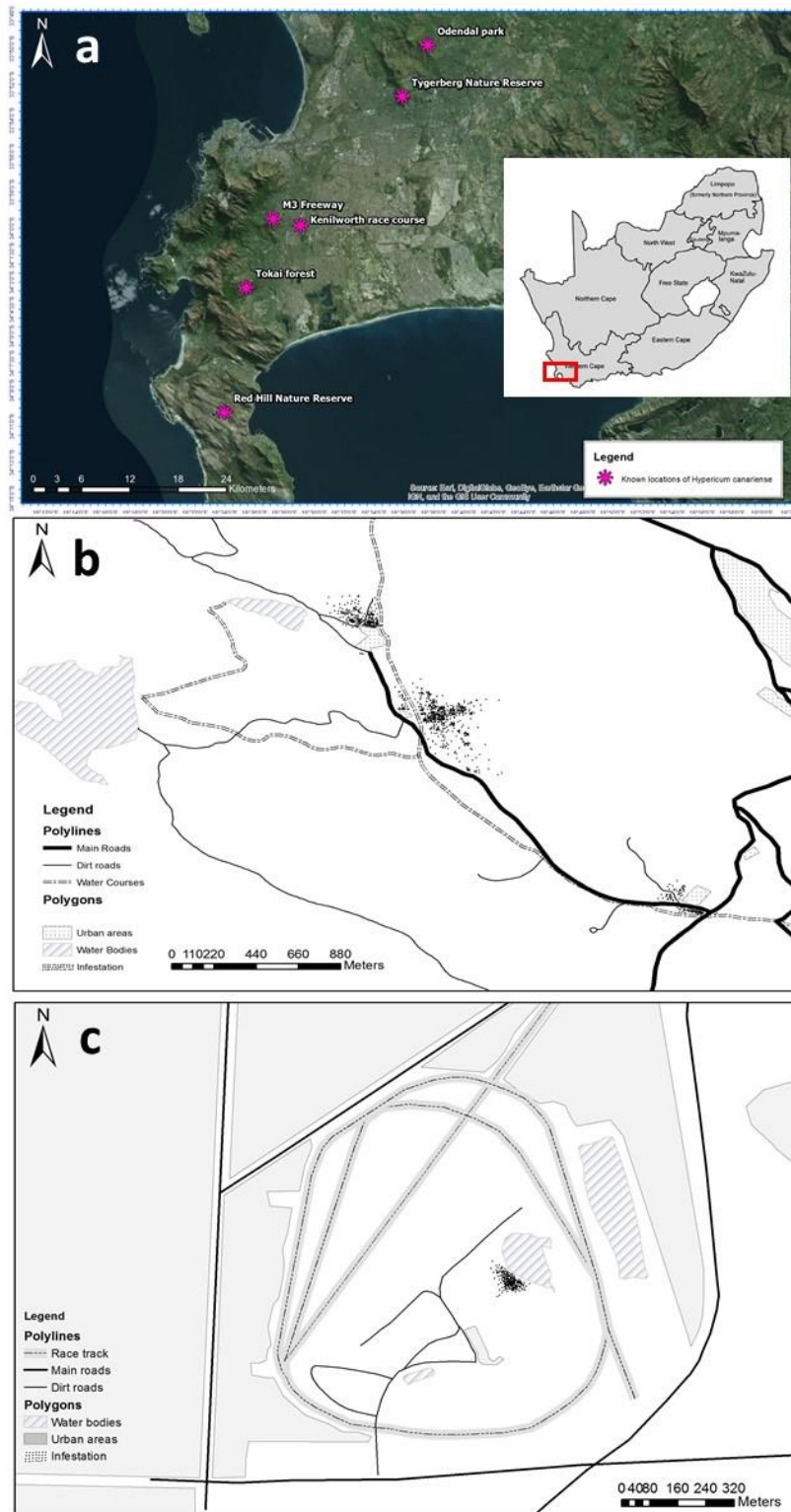
Figure 4.1: Known distribution of *H. canariense* at a (a) local scale (b) Redhill section of the TMNP (indicated by black dots) and (c) Kenilworth Nature Reserve.
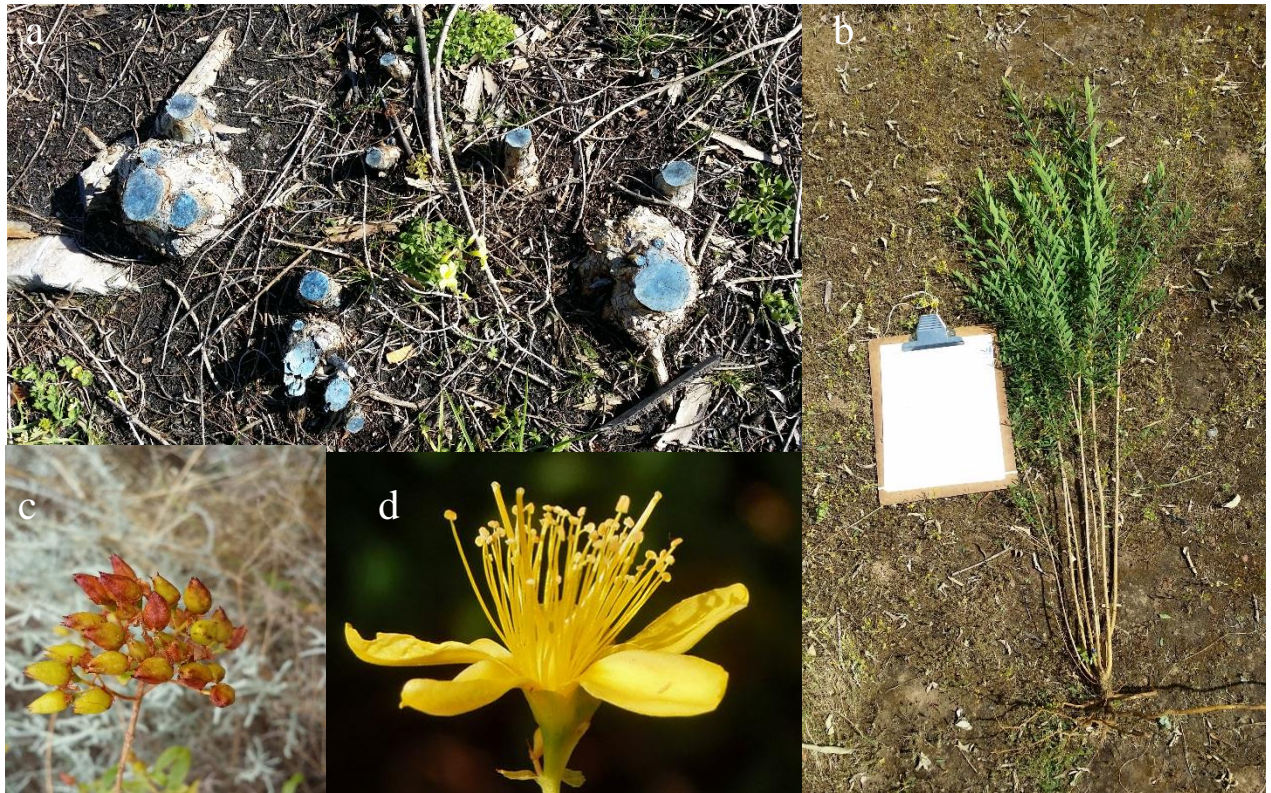
Figure 4.2. *Hypericum canariense* in South Africa. a) *H. canariense* cut stump treatment at the Redhill section of the TMNP reserve. b) a small *H. canariense* plant that was hand pulled at Tygerberg nature reserve. c) Typical *H. canariense* seedpods. d) *H. canariense* floral structure.
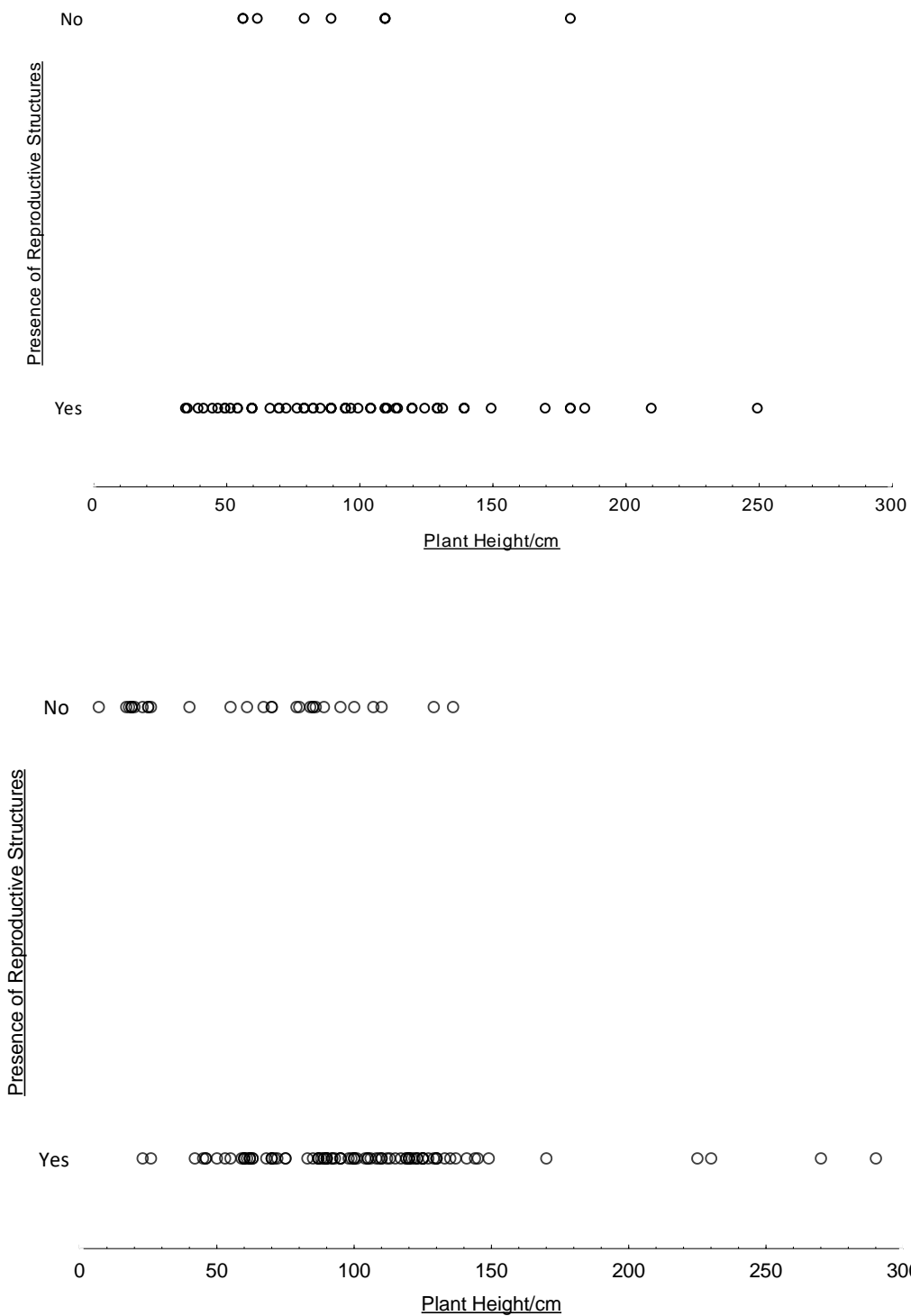
Figure 4.3: Size at reproduction for *H. canariense* infestations (a) Redhill section of the TMNP, (b) Kenilworth. At Redhill plants can reproduce at a height of 25cm whereas at Kenilworth plants reproduce at approximately 30cm.

60
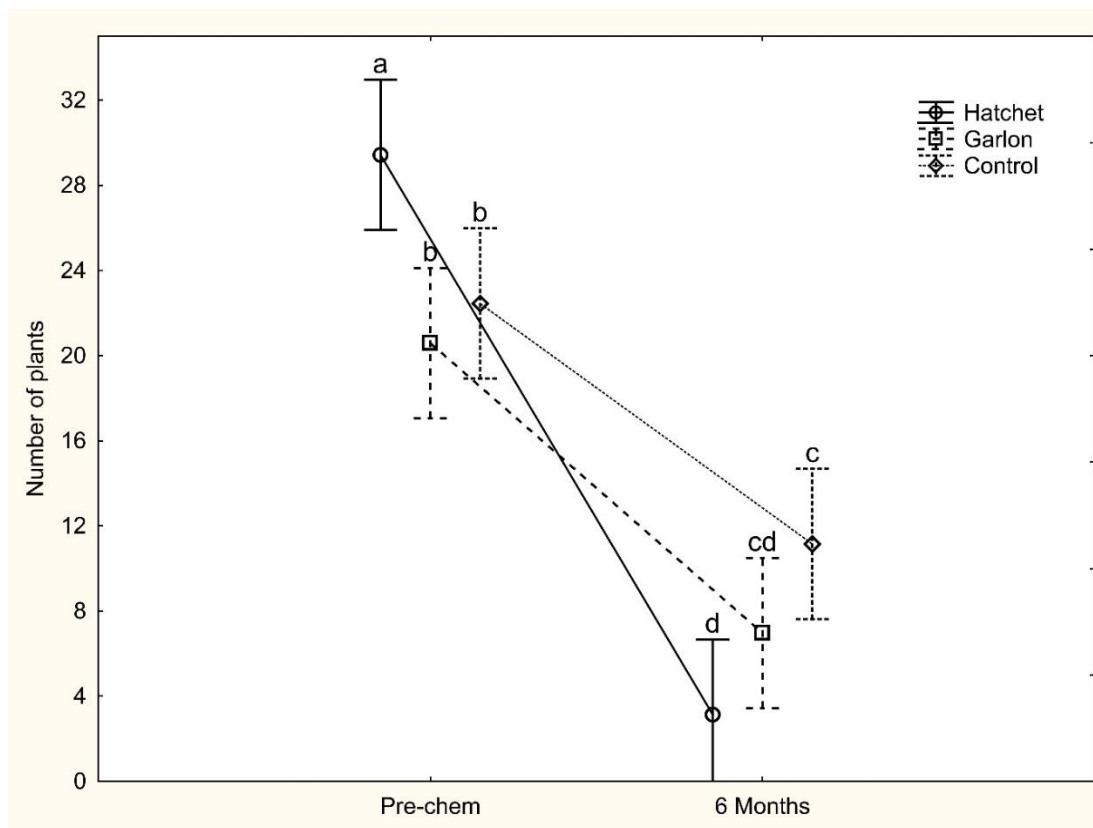
Figure 4.4: Whisker plot showing the difference between treatments before cut stump treatments and 6 months following treatment ($F_{(2,48)}$ = 10.29, P<0.05). These treatments were combined for both Kenilworth Nature Reserve and the Redhill section of the Table Mountain National Park.

Table 4.2: Mixed model ANOVA in R (lmer package) showing the combined effect of site, time and treatment have on the mortality of *Hypericum canariense* at Kenilworth Nature Reserve and Redhill Nature Reserve (part of the Table Mountain National Park).

	Sum sq	Mean sq	NumDF	DenDF	F value	p value
site	128.58	128.58	1	48	2.29	0.14
Treatment	186.09	93.04	2	48	1.66	0.20
time	7771.02	7771.02	1	48	138.24	<0.01
site*Treatment	384.57	192.28	2	48	3.42	0.04
site*time	364.24	364.24	1	48	6.48	0.01
Treatment*time	1156.65	578.33	2	48	10.29	<0.01
site*Treatment*time	150.54	75.27	2	48	1.34	0.27
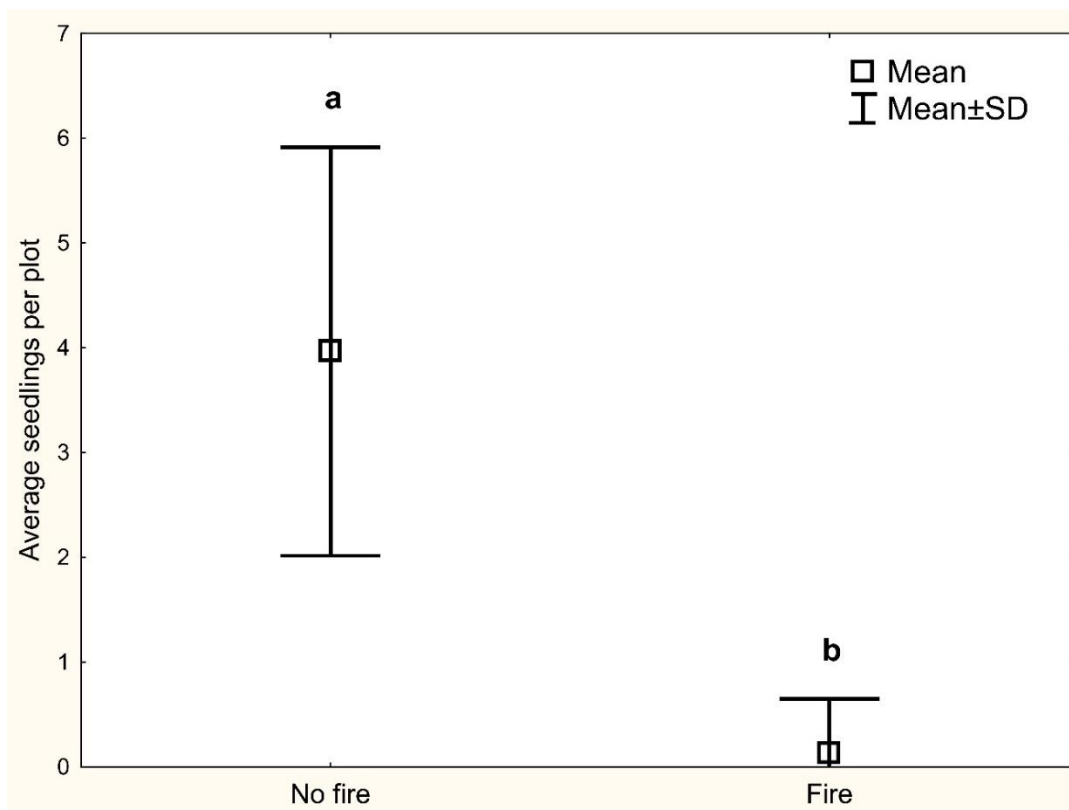
Figure 4.5: The average number of seedlings were significantly lower in areas that experienced a fire after clearing Redhill versus areas without a fire (Kenilworth). Different letters indicate significant differences.

4.4 Discussion

I show that the ornamental species *H. canariense* is an invader of disturbed areas at the urban-wildland interface and the two largest known infestations occur in conservation areas, TMNP at Redhill and the Kenilworth conservation area. Indicators are that *H. canariense* requires disturbance to successfully invade. Fire is an example of such a disturbance, however despite *H. canariense* re-sprouting with some vigour following a fire, fire does not stimulate the seedbank and fires decreases seedling recruitment (Fig. 4.2).

H. canariense was introduced through the ornamental horticulture with the first populations being recorded in 1950. The fact that *H. canariense* is not recognised as a popular horticultural species in South Africa and that the City of Cape Town has already attempted to control *H. canariense* prior to the undertaking of this study could explain why the distribution of *H. canariense* in South Africa is so limited. Despite this, *H. canariense* has the potential to become a widespread invader. This is enhanced by copious seed production and a large and persistent seed bank.

The seeds of *H. canariense* are small and easily transported by wind and water. The proximity of all naturalising populations to roads and watercourses highlights this – these movement pathways may

facilitate long-distance dispersal (Gelbard and Belnap, 2003). Within populations, dispersal is more consistent with stochastic dispersal through seed drop and gravity, potentially aided by wind. This, in combination with occasional short-jump dispersal due to accidental human movement or human disturbance, this may be sufficient to explain current patterns of spread at these infestations.

From the findings of this study, *H. canariense* is known to form persistent seed banks. The distribution of the seed banks is suspected to be mostly limited to beneath the canopies of mature shrubs. However, it is important to note that with plants growing in close proximity or within water courses there is the potential to spread rapidly if not contained (Cilliers and Siebert, 2012). As there is no evidence of *H. canariense* forming a seedbank in the literature, this study effectively confirms that *H. canariense* has the potential to develop a significant seedbank, but how long seeds remain viable needs to be determined.

The distribution of *H. canariense* appears to be rather limited with only seven naturalised populations being confirmed from 47 reported cases. The rest of the reported cases fall within city gardens or manmade green spaces. However, this species has been shown to be very invasive in other parts of the world with a similar Mediterranean climate and it has the ability to invade natural vegetation (Dlugosch and Parker, 2008b; Starr et al., 2003). Therefore, I propose listing of the species as category 1a under the National Environmental Management: Biodiversity Act. In particular, since I determined it to be potential invasive according to the Australian weed risk assessment. Given the current limited distribution, the potential threats posed and the success of control to date, I consider eradication a feasible and desirable management goal since the current size and distribution of *H. canariense* means it can still be potentially eradicated from South Africa (Kaplan et al. 2014).

The Redhill population covered by the survey was burnt in an intense wildfire January 2018. While *H. canariense* is able to re-sprout from cut stems (presumably if the stem is not treated or treated with the wrong herbicide application), I saw significant evidence of root re-sprouts, and vegetative reproduction (even on plants that were previously treated using the cut stump method). Because this species produces such a large seedbank, the significantly lower recruitment in areas with fires is positive and will make management considerably easier. Despite this, comprehensive annual follow ups are required because IAPs reach reproductive maturity in just a few months and previously standard clearing operations have "missed" flowering plants – making previous control operations ineffective in eradicating this species.

I found that the herbicide Hatchet had the highest efficiency upon mortality on the existing stands of *H. canariense*. Hatchet had an 87% combined mortality rate at Kenilworth Nature Reserve and Redhill Nature Reserve. As such I recommend that Hatchet at 2% concentration should be used to initially clear all naturalising populations.

In conclusion, *H. canariense* is an alien invasive species that has the potential to become a widespread invader in South Africa. Fortunately, it is not widely planted, only occurs at seven sites, has no benefits,

and seed banks are not stimulated by fire. Given its short juvenile period, potentially long-distance seed dispersal, ineffectiveness of current herbicides and large seed banks *H. canariense* should be prioritised for management.

Chapter five: Conclusions and implications for Management

The end goal of the work presented in this thesis was to provide succinct management recommendations on important IAP species in the City of Cape Town, with a particular focus on emerging IAPs and potential eradication targets within an urban environment. To achieve these aims, I conducted a literature review, traditional field sampling across the urban-rural gradient and species-specific experiments. This chapter summarises the main findings of this study and highlights management actions to improve management outcomes and ensure funding is optimised at a municipality level.

5.1 Thesis conclusions

Invasive alien plant species abundance was found to be significantly higher in natural areas at the interface between urban and rural areas (peri-urban zone) (Chapter two). The majority of these IAPs were determined to be agricultural weeds or to have originated from agricultural origins, with the second highest source of IAPs originating from horticultural activities. Habitat condition was established as an important factor in predicting the level of invasion, where sites which had a higher level of degradation were positively correlated with higher levels of IAP species richness and abundance. As such I recommend that management efforts should be focused on peri-urban areas in order to limit the impact IAP species have on natural and semi-natural areas which have been demarcated for conservation use within the Cape Town biological network. In addition to this, I suggest that firebreaks should be closely monitored to ensure that invasion of IAPs is hampered at these open sites to limit the chance of their proliferation into natural and semi-natural areas. Housing estates adjacent to all natural and semi-natural areas should be discouraged from using horticultural plant species, which may pose a significant risk of invasion into natural areas.

An invasive alien plant's area of infestation and rate of spread were presented as the two main criteria for classifying emerging IAPs and eradication targets in the City of Cape Town. The criteria presented were taken from a body of literature which is focussed on the feasibility of eradication and the success of weed eradication programs. With the use of these two criteria, IAPs can be classified as either widespread IAPs, emerging IAPs and potential eradication targets and can be subsequently grouped into management categories (i.e. mitigation, containment and eradication) (chapter three). This classification identified a subset of IAP species which are important species to management. For these species, effective management would see a high return on investment because they could be monitored and potentially eradicated before they exert a significant environmental and economic impact (by traversing the barriers to invasion and maximising management efforts in the face of limited funding).

The Analytical Hierarchical Process was used to develop a model for the prioritization of emerging IAPs [known as early detection and rapid response species for the City of Cape Town]) and adopted and tested in this study (chapter two and Appendix A). The main criteria presented (by order of importance and weight within the model) were IAP spread, negative impacts, ease of control and

invasive potential. These main criteria have a subset of criteria which are also weighted for their importance in achieving the goal (effective management of emerging IAPs in the Cape Town Metro). The priorities and final list of emerging IAPs should be used to guide future management of emerging IAP species going forward. Sufficient funding should be made available and directed to manage the highest priority species rather than the arbitrary selection and management of species or trying to manage all species at once.

Taken from a list of emerging IAP species, *Hypericum canariense* was studied in detail and its feasibility of eradication determined within the City of Cape Town (Chapter four). Of more than 50 reported cases of *H. canariense* reported across the City of Cape Town, only seven naturalising populations were confirmed and those were within natural and semi-natural areas around the city. Of two herbicide cut stump applications, Hatchet (Triclopyr) at a 2% concentration was determined to be the most effective at killing this species. It should take 17.5 person days to extirpate all *H. canariense* populations from the city with an additional four and a half days a year for ten years to exhaust the seedbank (the presence of a seedbank also determined in this study). The initial cost of clearing all populations is ZAR35 000 and a total of ZAR57 300. I recommend that it is within reach that the City of Cape Town can eradicate *H. canariense* in line with the National Environmental Management: Biodiversity act (Act 10 of 2004).

5.2 Implications for management and future research

Practicing land managers and conservation staff within the City of Cape Town are encouraged to focus management of IAPs on peri-urban zones to ensure ecosystem integrity is maintained as the city expands into rural areas (chapter two). Impacts such as natural vegetation harvesting, recreational activities and, most importantly, overgrazing should be addressed to lessen the anthropogenic impact of these areas. Further research into the dynamics of peri-urban areas and the stressors upon them should be undertaken to better understand the dynamics of these areas and to ensure healthy biological networks as the city continues to expand to the north and north east of the Cape Town metro.

Information on the longevity of seedbanks, the ease of detectability of an IAP species, and the length of an IAP's juvenile period should be gathered and stored in plant species databases in order to better classify IAP species (chapter two). This level of information would better inform management of species as a whole. Without such information, overly ambitious eradication programs may be undertaken which may be costly in the long run.

The results of the prioritisation (Chapter 2) are only as good as their data. As such, I encourage land managers and practicing conservationists to collect data that has been identified as important in the prioritization model for a far larger number of species. As these new data become available, they can be easily accommodated by the hierarchical model and used to generate a revised set of rankings of IAP species. An example of a lack of data includes the effects of these emerging IAPs on biodiversity

processes. Where data are completely lacking it is prudent to rely on the knowledge of experienced researchers, land managers and conservationists. Furthermore, the use of the I-Naturalist database (https://www.inaturalist.org/) could be indispensable in better informing this hierarchical model.

My final data chapter focused in on a specific emerging invader. Because *Hypericum canariense* is still limited in its distribution across the City of Cape Town (chapter 4), it responds well to cut-stump herbicide application and is not dispersed by birds or insects, this species should be targeted for eradication. I suggest that further research of the feasibility of eradication should be undertaken on firstly, the list of potential eradication targets identified in chapter 2 and secondly on the list of emerging IAPs. Moreover, I suggest that monitoring programmes be developed and implemented to better understand the rate of spread and potential negative impacts of emerging IAPs within the City of Cape Town. With direct knowledge on the feasibility of eradication of other species, it becomes easier to allocate funding to target these species before they become widespread and pose significant risks to natural and semi-natural areas within the city of Cape Town.

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Appendix A: A Final modified model for prioritising the control of 15 ED&RR target plant species in the Cape Metropole.

Criteria are given with weights assigned. G values indicate the contribution of criteria and sub-criteria to the goal while L values show the contribution of sub-criteria to covering criteria. In both cases these sum to one.

Goal: To rapidly control IAPs (present & emerging) so as to maintain ecosystems within the Metro

Criteria	Sub criteria
Spread (L: .4784 G: .4784)	• Current rate of spread (L: .7500 G: .3588)
	• Current distribution (L: .2500 G: .1196)
Negative impacts (L: .2577 G: .2577)	• Biodiversity process (L: .6370 G: .1642)
	• Biodiversity pattern (L: .2583 G: .0666)
	• Ecosystem services (L: .105 G: .027)
	o Health (L: .3333 G: .0090)
	o Water (L: .3333 G: .0090)
	o Economics (L: .3333 G: .0090)
Ease of control (L: .1921 G: .1921)	• Control effort (L: .600 G: .115)
	• Bio-control available (L: .2000 G: .0384)
	• Fire promoted (L: .2000 G: .0384)
	o Seed banks (L: .7500 G: .0288)
	o Resprouting (L: .2500 G: .0096)
Invasive potential (L: .0718 G: .0718)	• Dispersal rate (L: .555 G: .040)
	• Potential habitat envelope (L: .356 G: .026)
	• Pre-adapted to climate (L: .0889 G: .0064)

Appendix B: Species List for Chapter 3.1 (list of all widespread species, emerging species and eradication targets)

Acacia elata; Acacia implexa; Acacia paradoxa; Acacia stricta; Ailanthus altissima; Cardiospermum grandiflorum; Centranthus ruber; Cortaderia selloana; Genista (Cytisus) monspessulana; Hakea drupacea (suaveolens); Lythrum salicaria; Melaleuca hypericifolia; Pittosporum undulatum ; Rivina humilis; Spartium junceum; Sesbania punicea; Conyza albida (syn. sumatrensis); Canna indica; Schinus terebinthifolius; Casuarina equisetifolia; Psidium guajava; Hakea gibbosa; Hypericum canariense; Schinus molle; Hakea sericea; Melia azedarach; Opuntia ficus-indica; Hedera helix; Acacia baileyana; Celtis sinensis; Cirsium vulgare; Eriobotrya japonica; Syzygium paniculatum; Acacia mearnsii; Atriplex nummularia subsp. Nummularia; Cereus jamacaru; Echinopsis spachiana; Betula alba; Araujia sericifera; Pennisetum purpureum; Acacia pycnantha; Cotoneaster franchetii; Cotoneaster lacteus; Cotoneaster pannosus; Crataegus pubescens; Hedychium coccineum; Hedychium coronarium; Cuscuta campestris; Tecoma stans; Hakea salicifolia; Cinnamomum camphora; Cestrum laevigatum; Prosopis glandulosa var. Torreyana; Catharanthus roseus; Ageratina adenophora; Alnus glutinosa; Cinnamomum camphora; Eucalyptus camaldulensis; Robinia pseudoacacia; Solanum mauritianum; Egeria densa; Grevillea banksii; Acacia podalyriifolia; Pyracantha angustifolia; Pyracantha coccinea; Prunus armeniaca; Prunus persica var. Persica; Grevillea robusta; Lolium perenne; Acer negundo; Anredera cordifolia; Atriplex lindleyi subsp. inflate; Tropaeolum majus; Myoporum insulare; Plectranthus comosus; Cestrum aurantiacum; Hedychium flavescens;

Appendix C: Final data set used in results for the prioritization of IAPs in the Cape Town Metro.

This table displays the raw data used to populate the AHP model developed in section 3.2. Different scales were used for each of the sub criteria (see Section 3.2.3). The final list of priority species is exhibited in Table 3.4.

#	Species	Rate of spread	Current Distrobution	Biodiversity Process	Biodiversity Pattern	Health	Ecosystem Services	Economics	Control Effort	Biocontrol	Seed banks	Resprouting	Dispersal rate	Pot Hab env	Climate adaption
1	*Acacia elata*	2	2	2	3	1	3	2	2	1	1	1	1	2	3
2	*Acacia implexa*	1	3	3	3	1	2	2	1	1	1	1	1	2	2
3	*Acacia paradoxa*	1	4	4	3	1	2	2	2	1	1	2	1	2	2
4	*Acacia stricta*	1	4	4	3	1	2	2	2	1	1	0	1	2	2
5	*Ailanthus altissima*	2	2	2	3	3	2	2	1	1	3	1	2	2	2
6	*Cardiospermum grandiflorum*	1	4	4	0	1	1	1	3	1	3	2	2	1	1
7	*Centranthus ruber*	3	2	2	2	1	1	1	2	1	3	0	2	1	3
8	*Cortaderia selloana*	3	1	1	3	2	2	2	1	1	3	1	2	2	2
9	*Genista (Cytisus) monspessulana*	1	2	2	3	2	2	2	1	1	1	1	2	1	3
10	*Hakea drupacea (suaveolens)*	3	1	1	3	1	2	2	3	1	2	2	2	1	3
11	*Lythrum salicaria*	1	3	3	3	1	2	1	2	1	3	2	0	1	2
12	*Melaleuca hypericifolia*	1	4	4	3	1	2	2	1	1	2	2	2	2	2

13	*Pittosporum undulatum*	3	1	1	3	1	2	1	1	1	3	1	2	1	2
14	*Rivina humilis*	1	3	3	3	2	1	1	2	1	3	0	2	1	1
15	*Spartium junceum*	3	2	2	3	2	2	2	1	1	1	1	1	1	3
16	*Sesbania punicea*	3	2	2	3	2	3	1	1	1	3	2	2	2	1
17	*Conyza albida (syn. sumatrensis)*	3	1	1	2	1	1	1	1	1	2	2	1	2	1
18	*Canna indica*	3	1	1	3	1	1	1	1	1	3	1	2	2	2
19	*Schinus terebinthifolius*	3	2	2	3	3	3	2	1	1	3	1	2	2	2
20	*Casuarina equisetifolia*	3	2	2	3	2	3	2	1	1	3	1	2	2	1
21	*Psidium guajava*	3	2	2	2	1	3	1	1	1	3	1	2	2	1
22	*Hakea gibbosa*	3	1	1	3	1	3	2	2	1		2	1	2	3
23	*Hypericum canariense*	3	1	1	2	1	1	1	2	1	1	1	2	2	1
24	*Schinus molle*	3	2	2	3	3	3	2	1	1	3	1	1	1	1
25	*Hakea sericea*	3	2	2	3	1	3	2	2	1	2	2	1	2	3
26	*Melia azedarach*	3	2	2	2	3	3	2	1	1	1	1	2	1	1
27	*Opuntia ficus-indica*	3	2	2	2	2	1	2	1	1	1	1	2	2	2
28	*Hedera helix*	3	2	2	2	3	1	2	1		3	1	2	2	2
29	*Acacia baileyana*	3	2	2	3	2	3	1	2	1	1	1	2	2	2
30	*Celtis sinensis*	3	2	2	3		3	2	1	1		1	2	2	
31	*Cirsium vulgare*	3	2	1	2	2	1	1	1	1	3	2	2	2	3
32	*Eriobotrya japonica*	3	2	1	2	1	2	2	1	1	2	2	2	2	2

33	*Syzygium paniculatum*	3	2	2	3	1	2	1	1	1	1	1	1	2	1
34	*Acacia mearnsii*	3	2	3	3	1	3	2	1	1	1	1	2	2	2
35	*Atriplex nummularia subsp. nummularia*	3	2	2	2	1	1	1	1	1	1	2	1	2	1
36	*Cereus jamacaru*	3	2	3	3	2	1	2	3	1	2	1	2	2	1
37	*Echinopsis spachiana*	3	2	2	3	2	1	2	2	1	1	1	1	2	2
38	*Betula alba*	2	2	2	3	1	2	1	2	1	1	1	1	2	2
39	*Araujia sericifera*	3	2	1	2	3	1	1	2	1	1	1	2	2	1
40	*Pennisetum purpureum*	3	2	3	2	1	1	1	1	1	1	1	1	2	1
41	*Acacia pycnantha*	2	2	3	3	1	3	2	3	1	2	1	1	2	2
42	*Cotoneaster franchetii*	2	2	2	3	2	1	2	1	1	2	1	1	2	2
43	*Cotoneaster lacteus*	2	2	1	3	1	1	1	1	1	2	1	2	2	1
44	*Cotoneaster pannosus*	2	2	2	3	2	1	2	1	1	1	2	2	2	2
45	*Crataegus pubescens*	2	2	2	3		1	1	1	1	1	1	1	1	1
46	*Hedychium coccineum*	2	2	3	3	1	1	1	1	1	1	1	1	2	2
47	*Hedychium coronarium*	2	2	3	3	1	1	2	1	1	1	1	1	2	2
48	*Cuscuta campestris*	3	2	1	2	1	1	1	1	1	1	1	1	2	3
49	*Tecoma stans*	3	2	2	2	1	1	2	1	1	3	1	2	2	1

50	*Hakea salicifolia*	1	3	2	3	1	2	1	1	1	2	2	2	1	1
51	*Cinnamomum camphora*	2	2	3	3	2	3	1	1	1	1	1	2	2	1
52	*Cestrum laevigatum*	3	2	3	3	2	1	1	1	1	2	2	1	1	1
53	*Prosopis glandulosa var. torreyana*	3	2	2	3	3	3	2	1	1	2	1	1	2	1
54	*Catharanthus roseus*	3	2	2	1	2	1	2	1	1	1	1	1	2	1
55	*Ageratina adenophora*	3	2	1	2	2	1	2	2	1	1	1	2	1	1
56	*Alnus glutinosa*	3	2	1	2	1	3	2	1	1	1	1	2	2	3
57	*Cinnamomum camphora*	2	2	3	3	1	2	2	2	1	3	1	2	1	2
58	*Eucalyptus camaldulensis*	2	2	3	3	1	1	2	3	1	1	0	1	1	3
59	*Robinia pseudoacacia*	2	2	3	3	2	2	2	3	1	1	1	2	2	1
60	*Solanum mauritianum*	2	2	3	3	3	2	2	1	1	1	1	2	2	2
61	*Egeria densa*	1	3	2	3	1	2	1	2	1	1	1	1	2	2
62	*Grevillea banksii*	1	3			3	1	1	1	1	3	2	2	2	3
63	*Acacia podalyriifolia*	1	2	2	2	1	3	1	1	1	1	0	2	2	3
64	*Pyracantha angustifolia*	2	2	3	2	2	3	2	1	1	1	0	2	1	2
65	*Pyracantha coccinea*	2	2	2	2	1	3	1	1	1	1	0	2	2	3
66	*Prunus armeniaca*	2	2	3	3	1	2	1	2	1	2	0	1	1	2
67	*Prunus persica var. persica*	2	2	3	1	1	2	1	2	1	2	0	2	1	2

68	*Grevillea robusta*	3	2	2	3	3	2	1	2	1	1	0	2	2	3
69	*Lolium perenne*	3	2	2	1	1	1	1	3	1	3	0	1	1	3
70	*Acer negundo*	2	2	2	2	1	3	1	3	1	3	0	2	2	2
71	*Anredera cordifolia*	2	2	2	1	1	3	2	1	1	1	1	1	2	2
72	*Atriplex lindleyi subsp. inflata*	2	2	3	2	1	2	1	1	1	1	0	2	1	3
73	*Tropaeolum majus*	2	2	2	3	1	1	1	3	1	1	0	1	2	2
74	*Myoporum insulare*	1	3	1	1	1	3	1	2	1	1	0	2	2	3
75	*Plectranthus comosus*	2	2	1	2	1	2	1	2	1	1	0	2	1	1
76	*Cestrum aurantiacum*	1	3	1	2	2	2	1	1	1	1	0	2	1	2
77	*Hedychium flavescens*	1	3	3	3	1	2	2	1	1	3	0	1	2	2

Appendix D: Australian weed risk assessment and supplementary table on the amount of effort required to initially extirpate *Hypericum canariense* from Cape Town.

Species: *Hypericum canariense*			
	Question	**Answer**	**Score**
1	Is the species highly domesticated?	No	1
2.01	Species suited to South African climates	Yes, Mostly the Western Cape	2
2.02	Quality of climate match data	Unknown	2
2.03	Broad climate suitability (environmental versatility)	No	-1
2.04	Native or naturalised in regions with extended dry periods	Yes, where native in the Canary Islands and Madeira	1
2.05	Does the species have a history of repeated introductions outside its natural range?	Yes	1
3.01	Naturalised beyond native range	Yes	1
3.02	Garden/amenity/disturbance weed	Yes	1
3.03	Weed of agriculture/horticulture/forestry	No	-1
3.04	Environmental weed	Yes	1
3.05	Congeneric weed	Yes	1
4.01	Produces spines, thorns or burrs	No	-1
4.02	Allelopathic	No	-1
4.03	Parasitic	No	-1
4.04	Unpalatable to grazing animals	Unknown	0
4.05	Toxic to animals	Unknown	0
4.06	Host for recognised pests and pathogens	No	-1
4.07	Causes allergies or is otherwise toxic to humans	No	-1
4.08	Creates a fire hazard in natural ecosystems	Yes	1
4.09	Is a shade tolerant plant at some stage of its life cycle	Yes	1
4.10	Grows on infertile soils	Yes	1
4.11	Climbing or smothering growth habit	Yes	1
4.12	Forms dense thickets	Yes	1
5.01	Aquatic	No	-1
5.02	Grass	No	-1
5.03	Nitrogen fixing woody plant	No	-1

5.04	Geophyte	No	-1
6.01	Evidence of substantial reproductive failure in native habitat	No	-1
6.02	Produces viable seed	Yes	1
6.03	Hybridizes naturally	Unknown	0
6.04	Self-fertilisation	Yes	1
6.05	Requires specialist pollinators	No	-1
6.06	Reproduction by vegetative propagation	Yes	1
6.07	Minimum generative time (years)	1	1
7.01	Propagules likely to be dispersed unintentionally	Yes	1
7.02	Propagules dispersed intentionally by people	Yes, as an ornamental	1
7.03	Propagules likely to disperse as a produce contaminant	No	-1
7.04	Propagules adapted to wind dispersal	Yes	1
7.05	Propagules water dispersed	Yes	1
7.06	Propagules bird dispersed	No	-1
7.08	Propagules dispersed by other animals (internally)	Unknown	0
8.01	Propagules dispersed by other animals (externally)	Unknown	0
8.02	Prolific seed production	Yes	1
8.03	Evidence that a persistent propagule bank is formed (>1 yr)	Yes	1
8.04	Well controlled by herbicides	Yes	1
8.05	Tolerates or benefits from mutilation, cultivation or fire	Yes	1
8.06	Effective natural enemies present in South Africa	No	-1
Total score:			**14**

Supplementary table: Time to clear for initial extirpation at all sites

Site	Size of population.	Estimated time in person days
Redhill section TMNP	2537	9 person days
Kenilworth	1826	6 Person days
Tygerberg	79	0.5 Person days
M3	49	0.5 Person days
Lionshead	8	0.5 Person days
Odendaal park	16	0.5 Person days
Tokai	28	0.5 person days
Total		17.5 person days