Impacts and control of alien Proteaceae invasion in the Western Cape Province, South Africa

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ABSTRACT

Research focused on ecological impacts and control of invasive alien species (IAS) is gaining attention worldwide. The eradication and control of invasive alien plants (IAP) is essential for the restoration of native plant communities. Understanding ecological impacts and potential invasive risks of IAP is important for their effective management, particularly for prioritisation. Most studies concerning impacts on vegetation structure and plant-pollinator interactions have measured few ecological metrics, resulting in a superficial understanding of plant species invasion. Additionally, most studies related to the control of IAP have focused on major invaders which have demonstrated severe impacts, with less focus on emerging invaders. This study assessed ecological impacts, invasive risks and chemical control options for alien *Hakea drupacea* and *Banksia* species in the Western Cape Province, South Africa. Multiple ecological metrics data on vegetation, soil and plant-pollinator parameters were measured and compared between invaded and uninvaded sites. The invasion risk of fourteen *Banksia* species which have been introduced to South Africa was evaluated by conducting a weed risk assessment (WRA). The herbicide efficacy of resprouting *Banksia integrifolia* and *Banksia serrata* was determined by rating plants response to different treatments, with percentage, height and resprout vigour as measures. Results revealed significant negative impacts of alien *H. drupacea* and *Banksia speciosa* invasion on native plant species richness and diversity and on the abundance of native pollinators. The study demonstrated that 79% of *Banksia* species have a high risk of invading the Fynbos Biome. Chemical control with triclopyr+picloram mix provided effective means of controlling resprouting *Banksia* species. The high invasive risk of *Banksia* species and competitive effects of invasive alien *B. speciosa* and *H. drupacea* with native plant species for biotic and abiotic resources represents a major threat to biodiversity conservation in the Fynbos Biome. The removal of both naturalised and invasive alien *H. drupacea* and *Banksia* populations is recommended in order to conserve native plant communities in the Fynbos Biome.

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General Introduction

1 BIOLOGICAL INVASIONS

Biological invasions have considerable economic and ecological impacts (Richardson and Van Wilgen, 2004; Pimentel et al., 2005) and are considered the second-largest global threat to biodiversity after habitat loss (Wilcove et al., 1998; Sala et al., 2000; Van Wilgen et al., 2008). Expanding global travel and trade have resulted in both accidental and intentional introductions of alien species worldwide and these introductions are linked to various industries such as agriculture, forestry and horticulture (Richardson et al., 2003; Meyerson and Mooney, 2007; Hulme, 2009). Factors such as human population growth, climate change, and both natural and human-induced disturbances contribute to the increased likelihood of invasion in most ecosystems (Meyerson and Mooney, 2007; Richardson et al., 2007; Hulme, 2009).

1.1 INVASIVE ALIEN PLANTS

Alien plant taxa are present in an area due to either intentional or unintentional involvement of humans (Richardson et al., 2000). Invasive alien plants (IAP) are naturalised plants that produce large numbers of viable offspring dispersed far from parent plants with the potential to spread over a large area (Richardson et al., 2000; Pyšek et al., 2004). Naturalised plants are alien plants that reproduce consistently and sustain self-replacing populations for several years without direct anthropogenic influence, but do not necessarily invade ecosystems (Pyšek et al., 2004).

For an alien plant to become invasive, it must pass through a series of stages and overcome various barriers in order to establish and spread to novel environments (Blackburn et al., 2011). From a pool of introduced alien species, only few will emerge as invaders (Williamson and Fitter, 1996; Kolar and Lodge, 2001) and negatively affect native communities by altering ecosystem processes and threatening native biodiversity (Richardson and Van Wilgen, 2004; Strayer et al., 2006). Although invasive alien species (IAS) may represent only a small proportion of introduced species, their numbers and collective impact often outweigh the benefits of having these species in the country (Williamson and Fitter, 1996).

1.2 INVASIVE ALIEN PLANTS IN SOUTH AFRICA

South Africa is vulnerable to alien plant invasions and has one of the largest problems with IAP globally (Richardson and Van Wilgen, 2004). It has been estimated that about 10.2 million hectares (8%) of South Africa has been invaded by alien plant species (Wilson et al., 2013). Out of 8750 alien plant species introduced to South Africa, 660 are naturalised and about 180 are invasive (Wilson et al., 2013). The economic cost of alien plant invasion has been estimated as R 6.5 billion annually (Van Wilgen et al., 2012).

Of the nine South African terrestrial biomes, the Fynbos Biome within the Cape Floristic Region (CFR) in the Western Cape is the most severely invaded (Richardson and Van Wilgen, 2004; Esler et al., 2014a). It is mainly invaded by trees and shrubs from the genera *Acacia* Mill., *Hakea* Schrad. & J.C.Wendl. and *Pinus* L. (Joubert, 2009; Richardson and Van Wilgen, 2004). These invasions have transformed Fynbos ecosystems and invasive species have disproportionately high water use, which has resulted in serious water loss (Le Maitre et al., 1996). Despite ongoing control efforts, IAP remain the largest threat to biodiversity in the Fynbos Biome. Consequently, IAP account for the extinction of up to seven endemic plant species, putting a further 1207 plant species at risk (Esler et al., 2014a).

Given the current spatial extent of these invasions and that there remain climatically uninvaded suitable Fynbos areas that are predicted to be invaded (Rouget et al., 2004), it is essential to quantify impacts posed on native biodiversity by IAP. Moreover, restoring infested areas with native biodiversity through the eradication and control of IAP should be an integral part of managing the Fynbos vegetation to avoid extinctions (Holmes and Richardson, 1999). If left unchecked, further spread of these IAP will replicate impacts already imposed, ultimately leading to further ecosystem transformations.

1.3 IMPACTS OF INVASIVE ALIEN PLANTS

Invasive alien plants have economic (Van Wilgen et al., 2001; Pimentel et al., 2005), social (Shackleton et al., 2007; García-Llorente et al., 2008) and ecological (Richardson and Van Wilgen, 2004; Ricciardi et al., 2013) impacts on the areas they invade. Assessing the economic and social impacts of IAP are beyond the scope of this study but should not be overlooked. From a biodiversity conservation perspective, the focus should be on the ecological impacts (Barney et al., 2013).

Impact is defined as a measurable change (Ricciardi et al., 2013) or significant change to ecosystem properties caused by a non-native species (Hulme et al., 2013; Simberloff et al., 2013). Such changes may be positive, negative or neutral and vary in magnitude both in time and space (Ricciardi et al., 2013). Parker et al. (1999) used a mathematical equation I = R x A x E to define impact. They argued that impact is the product of the geographical range of the invader (R), its abundance or density (A) and the effect of an individual plant invader species (E). The terms impacts and effects will be used interchangeably throughout this thesis.

Impacts of IAP are species and context dependent (Kumschick et al., 2014). Apart from time since invasion (Strayer et al., 2006; Kumschick et al., 2014), the level of impact is determined by an interaction between traits of an alien species, characteristics of the recipient ecosystems and local abiotic factors (Richardson and Pyšek, 2006). There are various hypotheses about which species' life-history traits constitute successful invaders and vulnerable communities, such as the empty niche hypothesis (Mack et al., 2000) and the enemy release hypothesis (Keane and Crawley, 2002).

Some studies have attempted to predict which alien species will have negative impacts based on their life-history traits alone, but with little success (Rejmánek and Richardson, 1996; Kolar and Lodge, 2001). It is difficult to accurately predict when, where and which alien species will pose ecological impacts (Mack, 1996) due to the complexity of mechanisms and mediators of ecological impacts (Thuiller et al., 2006; Hulme et al., 2013; Ricciardi et al., 2013). This highlights the importance of measuring invaders' impact to determine their specific effects. Hence, the need for determining these effects serves as evidence for the justification of control and guiding policies relevant to plant invasions (Simberloff et al., 2013; Jeschke et al., 2014).

1.3.1 Impacts of invasive alien plants on community structure

Impacts of IAP on native plant community structure and ecosystem processes are well documented and include reduction in species richness and diversity (Richardson et al., 1989; Hejda and Pyšek, 2006; Hejda et al., 2009), reduction in surface water run-off (Le Maitre et al., 2000), alteration of soil properties (Vanderhoeven et al., 2005; Heneghan et al., 2006; Dassonville et al., 2008; Li et al., 2014) and changes in fire regimes (Brooks et al., 2004).

However, for studies which have examined impacts on community structure, only a small proportion have presented evidence of mechanisms or pathways generating such impacts

(Levine et al., 2003). One example of an impact mechanism is light exclusion, which only a few authors, for example Braithwaite et al. (1989), have quantified by measuring light availability under *Mimosa pigra* L., an invasive shrub in Australia. Many others largely attributed impacts to competition for light and water but do not provide sufficient evidence (Woods, 1993; Wyckoff and Webb, 1996; Lavergne et al., 1999). Most studies that have examined impacts on ecosystem processes have not tested the consequences of this for community structure (Levine et al., 2003).

1.3.2 Impacts of invasive alien plants on native plant-pollinator interactions

Competition for pollination services between alien and native species has only received attention relatively recently (Moragues and Traveset, 2005; Muñoz and Cavieres, 2008; Dietzsch et al., 2011). Invasive alien plants occurring at high densities and possessing attractive flowers can negatively affect native flowers by disrupting their flower visitations by pollinators (Chittka and Schürkens, 2001; Kandori et al., 2009; Stout and Casey, 2014). This disruption can reduce pollination services to native plant species and result in negative effects on their reproductive success. Alien plants occurring at lower densities may have a neutral (Aigner, 2004; Moragues and Traveset, 2005) or beneficial effects, (Bartomeus et al., 2008; Russo et al., 2016) but the latter has rarely been reported.

Given that about 90% of flowering plant species rely on animal pollinators for their seed production and that they tend to share pollinators (Ollerton et al., 2011), studies on plant-pollinator interactions are of great ecological importance (Menz et al., 2011). However, most such studies have adopted experimental approaches, such as using potted plants, and focusing mainly on insects as a functional group of pollinators (Kandori et al., 2009; Flanagan et al., 2010). This encouraged the present study to use natural communities and focus on birds as pollinators to investigate plant-pollinator interactions.

Many studies concerning impacts on both community structure and plant-pollinator interactions have measured few response variables, resulting in a superficial understanding of the broader features of species' invasion (Hulme et al., 2013). Evaluating more ecological metrics within a single study of impact can enhance the breadth and depth of knowledge of such invasive species' ecological impacts (Barney et al., 2015). The lack of attention to mechanisms (Levine et al., 2003) and lack of standard methodology for quantifying impacts (Kumschick et al., 2014; Wilson et al., 2014; Barney et al., 2015) are some of the shortcomings identified within the invasion literature that warrants attention. The present study will assess multiple ecological metrics implicated in community structure and ecosystem functioning (see Barney et al. 2015) in an attempt to uncover some underlying mechanisms of impacts.

1.4 CONTROL OF INVASIVE ALIEN PLANTS

Approaches to mitigate threats caused by plant invasion involves prevention, eradication, containment and control (Mack et al., 2000; Rejmánek and Pitcairn, 2002; Pyšek and Richardson, 2010). Eradication can be defined as the removal of the whole population of a species from a specific area (Pyšek and Richardson, 2010). Containment involves limiting the species from spreading further from an invaded site (Hulme, 2006). Control is the long-term reduction of the invader population to a low density such that it can be tolerated (Simberloff, 2003).

The control and management of IAP is essential for the successful restoration of indigenous plant populations and communities (Holmes and Richardson, 1999; Van Wilgen et al., 2000; Webster et al., 2007). Most global studies relating to the control of biological invasions have focused on major invaders that have demonstrated severe impacts (Nel et al., 2004; Mgidi et al., 2007). Control measures are often delayed until invasive species are well-established and large infestations are evident (Hobbs and Humphries, 1995). However, emerging invaders should also be a concern due to their potential impacts on the environment (Blossey et al., 2001; Nel et al., 2004).

There is a strong emphasis on pro-active management and targeting emerging invaders for eradication while their population is small (Rejmánek and Pitcairn, 2002; Simberloff, 2009; Wilson et al., 2014). Although alien plants may have no or minor ecosystem impacts at such low densities, justification of their control should be based on feasibility of eradication (Panetta, 2015), low costs of control (Myers et al., 2000; Simberloff, 2009) and concerns over potential negative impacts of a species (Blossey et al., 2001).

Eradication and control of IAP can be expensive for widespread invasions and species with long-lived seed banks (Rejmánek and Pitcairn, 2002; Simberloff, 2003). The cost incurred in clearing invasive species is based on the size of the infestation and life-history traits of species to be cleared. Marais et al. (2004) estimated that about R600 million is spent annually on IAP control by the Working for Water programme (WfW) in South Africa. This highlights the importance of initiating control efforts for both major and emerging invaders (Mgidi et al., 2007) and the use of cost-effective methods in control programmes (Myers et al., 2000).

Different control methods have been developed to reduce or eradicate IAP in order to conserve native communities (Hobbs and Humphries, 1995; Van Wilgen et al., 2000; Tu et al., 2001). Each method has its advantages and disadvantages in terms of efficacy, effects on non-target species and costs (Tu et al., 2001; DiTomaso et al., 2006). The choice of which control method to use is determined by the invasive species' biological attributes, the extent of the infestation and other factors such as budget, manpower and skills (Van Wilgen et al., 2000). Regardless of which methods are used, minimizing impacts on the environment should be a priority (Esler et al., 2014b; Kaiser-Bunbury et al., 2015).

1.4.1 Mechanical control

Mechanical control involves mowing, felling or uprooting of plants and is normally used in conjunction with burning (Hobbs and Humphries, 1995; Van Wilgen et al., 2000). Hand-pulling is suitable for seedlings and saplings occurring in soft and sandy soil (Sheley et al., 1998; Esler et al., 2014b). These techniques have low ecological impact but are labour-intensive and only feasible for small infestations or follow-up clearing (Hobbs and Humphries, 1995; Albrecht et al., 2005). Moreover, mechanical options such as cutting offer a partial and short-term solution only, and require treating stumps with herbicides for resprouting species (Van Wilgen et al., 2000; Cherry et al., 2008). The role of mechanical control is therefore small in control programmes, given that control usually involves large areas (Hobbs and Humphries, 1995).

1.4.2 Chemical control

Chemical control is the use of environmentally safe herbicides (Van Wilgen et al., 2000; Tu et al., 2001). Herbicides remain an important tool to control IAP (Hobbs and Humphries, 1995; Kettenring and Adams, 2011), although concerns remain around their harmful effects on non-target species (Pimentel, 1995; Matarczyk et al., 2002) and potential contamination of water resources (Kolpin et al., 1998; Barbash et al., 2001). However, herbicides can be the most effective and reliable option for controlling IAP, provided that safety procedures are followed (Simberloff, 2009; Kettenring and Adams, 2011). In some cases, it may be the only option available to fight invasions effectively (Simberloff, 2009).

Due to their potential negative side effects, research continues to try to develop more effective and efficient herbicides (Rüegg et al., 2007). Modern herbicides have been reported to have improved properties, are more specific, possess shorter persistent periods and are less toxic (Hobbs and Humphries, 1995; Rüegg et al., 2007; Simberloff, 2009). Main methods of herbicide applications are foliar spray, stem injection and cut-stump treatment (Tu et al., 2001). The latter application is more target-specific and has a low risk of affecting non-target plants and contaminating soil or water (Tu et al., 2001; Cherry et al., 2008). The use of herbicides alone rarely provides long-term control and it is recommended to be used in combination with other methods of control such as mechanical and biocontrol (Goodall and Erasmus, 1996; Motooka et al., 2002).

1.4.3 Biological control

Biological control is the use of introduced natural enemies of invaders to suppress the population density of the specific pest organism to make it less abundant or damaging in its introduced environment (Eilenberg et al., 2001). Biocontrol has historically been considered an environmentally safe method for IAP control (Cory and Myers, 2000). However, increasing cases of its effect on non-target species has been raised concerns and great caution is called for when considering this method (Simberloff and Stiling, 1996; Cory and Myers, 2000). Simberloff (2009) have recommended it to be considered as a last resort. In cases where it has been successfully implemented, biological control can be a long-term, cost-effective and self-sustaining management option (Blossey et al., 1994; Moran et al., 2005).

1.4.4 Integrated weed management

Integrated weed management is an approach that combines two or more methods of controlling IAP (Van Wilgen et al., 2000; Harker and O'Donovan, 2013). The use of a single method typically does not reduce the population of IAP to manageable levels (Swanton and Weise, 1991; Hobbs and Humphries, 1995; DiTomaso, 2000). Hence, a combination of mechanical clearing, application of herbicides and biological control should be used to effectively control target populations in the long-term (Hobbs and Humphries, 1995; Van Wilgen et al., 2000; Van Wyk and Van Wilgen, 2002).

1.5 THE FAMILY PROTEACEAE

The Proteaceae provides an ideal study group to assess invasion impacts, the invasion risks involved and their control methods. This is because many species from this family are common in horticulture which is a primary pathway for introduction and dissemination of IAP globally (Reichard and White, 2001; Richardson and Rejmánek, 2011). Many species from this family are planted for cut-flowers, ornamentals, hedges and food, with a long history of introduction outside their native ranges. The increasing interest in Proteaceae shown by horticultural industries due to their commercial importance (Moodley et al., 2013) provides an additional reason why this family should be studied.

The Proteaceae is a family of woody flowering plants mainly limited to the Southern Hemisphere comprising approximately 1674 species in 79 genera (Moodley et al., 2013). The greatest diversity of genera and species are concentrated in southwestern Australia and the Western Cape Province of South Africa with 16 genera (682 species) and 14 genera (331 species), respectively (Lamont et al., 1985; Cowling and Lamont, 1998). Both regions are similar in most environmental aspects and characterized by a Mediterranean-type climate, nutrient-poor soils and exposure to fire (Lamont et al., 1985; Cowling and Lamont, 1998).

Many Proteaceae species possess morphological and physiological characteristics, which allow them to cope with low nutrient availability and exposure to fire (Bowen, 1981; Venkata, 1971). Unlike families that form symbiotic associations with mycorrhizal fungi, Proteaceae members develop proteoid roots which enhances their uptake of nutrients, particularly phosphorus and nitrogen as well as water (Purnell, 1960; Bowen, 1981; Lamont, 1985; Dinkelaker et al., 1995). Most Proteaceae tend to produce large seeds enriched with nutrients to enable seedling establishment in nutrient-poor soils (Groom and Lamont, 1998; Groom and Lamont, 2010).

The majority of Proteaceae species growing in fire-prone habitats are serotinous (Lamont et al., 1985). Serotiny refers to the retention of mature seeds in fruits or cones on a plant for a year or longer until seed release is stimulated by fire (LeMaitre, 1985). This mechanism ensures that seeds are released after fire when conditions are conducive for germination and establishment (Cowling and Lamont, 1987; Bradstock, 1991). Species have varying degrees of serotiny ranging from weakly serotinous (where most seeds are released upon maturity) to strongly serotinous (where most seeds are retained in the canopy for many years) (Cowling and Lamont, 1987; Enright et al., 1998).

There are 24 alien Proteaceae species introduced and cultivated in South Africa, eleven of which have been recorded as naturalised and eight as invasive (Moodley et al., 2014). There are few aggressive invaders reported in the family and this is partly attributed to their recent introduction (Richardson et al., 1990; Richardson and Rejmánek, 2011). This suggests that most species may still be in the lag phase but with the potential to become invasive in the future. Given the commercial importance of some species and increasing interest in this family for horticulture and floriculture, it is expected that more species will be introduced (Richardson et al., 1990; Moodley et al., 2013).

The genus *Banksia* L.f. has fourteen species introduced to South Africa for floriculture (Geerts et al., 2013), making it the genus of Proteaceae with the largest number of alien species in South Africa (Moodley et al., 2014). Most of these have been identified to have a high risk of invading the Fynbos Biome (Richardson et al., 1990; Honig et al., 1992). Moreover, there is an increasing demand for *Banksia* species in the floriculture industry due to their economic value as cut-flowers (Honig et al., 1992; Sedgley and Janick, 1998; Moodley et al., 2013).

The genus *Hakea* contains one of the most successful invading species in the Fynbos Biome, but there are only five species introduced for use as barriers and firewood (Shaughnessy, 1980). *Hakea* and *Banksia* species possess similar life-history traits such as large viable seed banks, serotinous seeds, short juvenile periods and similar ecological requirements, which make them high invasion-risk species in the Fynbos Biome (Richardson et al., 1987; Richardson et al., 1990).

The commercial importance of this family has been recognized, yet relatively few studies have been undertaken to assess their impacts, risks, or control options for alien Australian Proteaceae in South Africa. Understanding impacts and invasive risks is important for effective management of invasive species, particularly for prioritization purposes (Blossey et al., 2001; Wilson et al., 2013). In terms of risk assessments and control, predictive and proactive management approaches are essential (Wilson et al., 2011). This highlights the importance of carrying out the research reported in this thesis, which serves to fill some of the existing knowledge gaps.

1.6 AIM AND OBJECTIVES

The overarching aim of the study is to assess impacts of alien *Banksia* and *Hakea* invasions on native biodiversity and to determine the effective control method for resprouting *Banksia* species.

I studied this through: 1) quantifying impacts of alien *Hakea drupacea* (C.F.Gaertn.) Roem. & Schult. invasions on native vegetation structure and soil properties; 2) quantifying impacts of alien *Banksia speciosa* R.Br. invasions on native plant-pollinator interactions of an endemic native plant, *Protea compacta* R.Br.; 3) evaluating the invasion risk of fourteen *Banksia* L.f. species which have been introduced to South Africa; and 4) determining the efficacy of different herbicides at varying concentrations as a cut-stump treatment for resprouting *Banksia* species.

1.7 RESEARCH QUESTIONS

The thesis will attempt to answer the following research questions:

1. Does the invasive *Hakea drupacea* alter plant community structure (species diversity and richness) and soil chemical and physical properties?
2. Does the invasive *Banksia speciosa* influence nectar feeding bird abundance, richness and flower visitation rates to *Protea compacta*? What is the nectar volume and concentration produced and does this have any influence on pollinator visitation to the two study species? Does the altered visitation rate, if any, to native *P. compacta* influence its seed set?
3. Which of the fourteen *Banksia* species currently cultivated in South Africa have a low, medium and high risk of invasion?
4. How would *Banksia integrifolia* and *B. serrata* respond to the application of herbicide following cut-stump treatment? Which herbicides at what concentrations are most effective in controlling *B. integrifolia* and *B. serrata* infestations? Is eradication of these species in South Africa feasible at the observed population densities and extent?

1.8 THESIS CHAPTERS

This thesis comprises six chapters of which four are data chapters. Chapter 1 introduces key concepts which developed the thesis' main ideas i.e. invasive alien species, their impacts and control, the Proteaceae family and placing them in context of the relevant literature. It also presents the aims and objectives of the thesis, the research questions it seeks to answer and outlines the thesis chapters.

Chapter 2 assesses whether invasion by *Hakea drupacea* alters plant species richness, diversity and the physical and chemical soil properties of the invaded community. This was determined by comparing these parameters between *H. drupacea*-invaded, uninvaded and cleared sites.

Chapter 3 assesses whether invasive alien *Banksia speciosa* has the potential to compete for pollinators with the co-occurring and co-flowering native, *Protea compacta*. This was determined by comparing bird abundance and richness, visitation rates and nectar properties between *B. speciosa*-invaded and uninvaded sites.

Chapter 4 evaluates the invasion risk of fourteen *Banksia* species cultivated in South Africa, by conducting a weed risk assessment (WRA).

Chapter 5 evaluates the efficacy of different herbicides at different concentrations as a cut-stump treatment to test the efficacy of resprouting *Banksia integrifolia* and *Banksia serrata*.

The concluding chapter (Chapter 6) provides a synthesis of the research findings derived from chapters 2 to 5 and a discussion of implications for management and restoration activities.

Chapter 2

Impacts of an invasive alien *Hakea drupacea* on the vegetation structure and soil properties

Abstract

Invasive alien plants (IAP) can alter vegetation structure and soil properties of invaded ecosystems. This can result in competitive exclusion of native plant species and soil legacy effects, which can hinder the recovery of native plant species. Most studies concerning impacts on vegetation structure and soil properties have measured single or few ecological metrics or limited to single or few study locations. This study assessed effects of *Hakea drupacea* invasion on the vegetation structure and soil physical and chemical properties of an invaded Fynbos community. Native species richness and diversity, litter depth, canopy cover, height and soil properties (nutrients, moisture and pH) were compared in *H. drupacea*-invaded, adjacent uninvaded and cleared sites. The relationship between *H. drupacea* canopy cover and its litter depth was tested. Invaded sites supported significantly lower species richness and diversity than uninvaded sites ($t_{(22)}$ = -4.857; $P < 0.001$) . *Hakea drupacea* litter depth increased as its canopy cover increased with the potential to limit and suppress recruitment and growth of native plant species (r = 0.607, $P = 0.036$). There was no significant effects on soil properties caused by *H. drupacea* invasion. The lack of invasion effects on soil properties suggested that soil conditions could be suitable for native species establishment and recovery without requiring active restoration after local *H. drupacea* removal. Reduction in native plant species diversity and richness constitutes a major threat to the conservation of biodiversity in the Fynbos Biome where IAP have caused local extinction of native plant species. This study emphasized the need for future research to include representative cleared sites to determine legacy effects. *Hakea drupacea* populations should be removed to conserve the species richness and diversity of the Cape Floristic Region.

Keywords: Cape Floristic Region; *Hakea drupacea*; soil properties; species diversity; species richness; vegetation structure

2 INTRODUCTION

Impacts of IAP are a global problem (Vilà et al., 2011) and are gaining increasing attention (Pyšek et al., 2012; Kumschick et al., 2015). Invasive alien plant species compete for both abiotic (Vilà et al., 2006; Hejda et al., 2009; Shackleton et al., 2015; Sharma et al., 2017) and biotic resources, such as pollinators (Brown et al., 2002; Moragues and Traveset, 2005; Ye et al., 2014), which are important for plant survival and reproduction. Alien plant invasions can alter the composition and structure of native communities, ultimately leading to changes in ecosystem processes (Levine et al., 2003). This can have harmful effects on invaded ecosystems and possibly exclude native plant species (Kumschick et al., 2015).

Impacts of IAP on community structure are well documented (Vilà et al., 2011; Tererai et al., 2013; Bravo-Monasterio et al., 2016; Sharma et al., 2017). However, most studies that assess impacts on community structure are limited to measuring few response variables (Barney et al., 2015). Moreover, explanation of the mechanisms behind such impacts are restricted to speculations (Levine et al., 2003). Invasion-associated changes in micro-environments, such as light availability, litter, nutrients and moisture, are some of the mechanisms underlying these impacts but are rarely all tested in a single study (Levine et al., 2003; Skurski et al., 2014). Consequently, a more holistic approach in assessing effects of IAP on multiple response variables is crucial for a complete understanding of the invasion process and its consequences (Barney et al., 2015).

There are fewer impact studies on soil properties, relative to those on community structure (Ehrenfeld and Scott, 2001; Vilà et al., 2006; Dassonville et al., 2008; Li et al., 2014). However, soil properties are equally or even more important when assessing impacts of IAP than other frequently examined effects (Ehrenfeld et al., 2001; Ehrenfeld, 2003). This is because changes in soil properties mediated by invasion affect the recruitment and growth of both native and alien species (Raizada et al., 2008). Soil properties should therefore form an integral part of any study assessing ecological impacts of IAP (Zavaleta et al., 2001). Moreover, a modest number of studies which have examined soil properties have either focused on species which are nitrogen-fixers (Vitousek and Walker, 1989; Witkowski, 1991; Yelenik et al., 2004) or on soil properties alone in a single study without considering other components such as vegetation structure (but see Vilà et al., 2006; Stefanowicz et al., 2017).

Previous studies that have examined impacts of IAP on invaded ecosystems on both plant community structure and soil properties have reported negative (Heneghan et al., 2006; Hejda et al., 2009; Bravo-Monasterio et al., 2016; Afreen et al., 2017), positive (Dassonville et al., 2008; Lankau et al., 2009; Zhang et al., 2009; Dong et al., 2015) or neutral (Hejda and Pyšek, 2006; Vanderhoeven et al., 2006; Vilà et al., 2006; Dong et al., 2015) effects. The magnitude and direction of an impact outcome is largely determined by an invader's density with more pronounced effects as both invader cover and age of invasion increase (Strayer et al., 2006; Gooden et al., 2009; Dostál et al., 2013). It is further determined by the differences and or similarity in main functional traits between alien and native plant species (Scharfy et al., 2010).

Previous research on impacts of IAP on vegetation structure in the Fynbos Biome have focused on groups of IAP species, rather than on single species (for example Richardson et al., 1989). This results in the generalization of impacts and difficulty in attributing such effects to a specific invader species, which is necessary for management purposes. Such impact studies were further limited to either a single or few study locations, hence restricting the perspective of the invasion impact. The present study focuses on a single IAP species and across several locations, because impacts are often species-specific (see Hejda et al 2009; Simberloff, 2013).

There is a lack of quantitative data on the effects of *Hakea drupacea* (C.F.Gaertn.) Roem. & Schult. on the vegetation structure and soil properties of ecosystems it has invaded, despite it having invaded large areas of the Fynbos Biome (Richardson and Van Wilgen, 2004; Joubert, 2009; Esler et al., 2014). As with most other IAP worldwide (Schmitz et al., 1997; Skurski et al., 2014), available information on this species remains anecdotal and have been largely extrapolated from other IAP, particularly *H. sericea* Schrad. & J.C.Wendl.

Limited studies conducted on *H. sericea*'s impacts have demonstrated that it can reduce native species richness (Richardson et al., 1989), alter fire regimes (Van Wilgen and Richardson, 1985) and reduce water resources (Enright, 2000; Van Wilgen et al., 2006). Consequently, there is a need to assess ecological impacts of other invasive alien *Hakea* species on the Fynbos Biome to help conserve its rich flora and fauna.

In South Africa, *H. drupacea* was proclaimed as a noxious weed in 1937 along with other two *Hakea* species (Fugler, 1982) and is classified as a category 1b invader under the NEM:BA regulations (NEM:BA, 2014). It has a widespread distribution across the Western Cape

Province, largely occurring in sandstone Fynbos but less common in granite, limestone and alluvium Fynbos (Erckie, 2014). *Hakea drupacea* has been identified a major invader in the Fynbos Biome (Richardson and Van Wilgen, 2004) and this was supported by its high invasive risk (Erckie, 2014). Factors contributing to its success are its efficient seed dispersal over long distances and high seed longevity in the canopy (Richardson et al., 1987).

The present study will evaluate important ecological metrics implicated in impacts of IAP. It represents one of the composite quantitative measures of the impacts of *H. drupacea* on the vegetation structure and soil properties in the Fynbos Biome. Specifically, it sought to answer the following questions: 1) Does the invasive alien *H. drupacea* reduce native Fynbos species diversity and richness? 2) Does it alter soil chemical and physical properties of the invaded area?

2.1 MATERIALS AND METHODS

2.1.1 Study species

Hakea drupacea (C.F.Gaertn.) Roem. & Schult. formerly known as *H. suaveolens* R.Br., is a woody perennial shrub or tree up to 6 m tall (Henderson, 2001). Native to Australia, it was introduced to South Africa in 1850 (Shaughnessy, 1986) as a hedge plant to prevent animals from entering pine plantations and to stabilize sand dunes on the Cape Flats (Fugler, 1982).

Leaves of *H. drupacea* are up to 100 mm long and divided into upright, sharp-pointed needles (Fig. 2.1A). It produces clusters of white to cream fragrant flowers (Fig. 2.1B) between June and September in South Africa (Henderson, 2001; Bromilow, 2010). It is serotinous, has two-winged seeds that are covered in woody capsules (Fig. 2.1C) and they are primarily dispersed by wind. It forms dense, impenetrable stands (Fig. 2.1D) which can suppress native vegetation (Fugler, 1982; Richardson and Van Wilgen, 2004).

Figure 2.1. Morphology of *Hakea drupacea* (A) sharp-pointed needle leaves (B) white flowers occurring in axillary clusters (C) woody follicles containing serotinous seeds (D) dense monospecific stand. Photographs: A-C google images; D- Ernita van Wyk.

2.1.2 Study area

The study was conducted at 12 sites across the *H. drupacea* vegetation range in the Western Cape Province, South Africa (Appendix 1). The area experiences a Mediterranean-type climate with hot dry summers and cold wet winters. Average annual rainfall is about 300 mm of which most falls during winter months (Lamprecht et al., 2006). The mean temperature ranges between 15°C and 27°C for the cool and warm months, respectively. Soils are classified as well-drained, acidic and nutrient-poor (Mucina and Rutherford, 2006).

2.1.3 Experimental design

An observational approach of comparing invaded, uninvaded and cleared sites was employed. Twelve plots each measuring 5 x 5 m ($25m^2$) were randomly located in invaded, adjacent uninvaded and cleared sites, comprising a total of 27 plots. This is the representative number and size of plots recommended by Barney et al. (2015). Due to low availability of cleared sites in close proximity to invaded and uninvaded sites for comparative purposes, only three plots represented cleared sites. Invaded sites referred to those predominantly invaded by *H. drupacea* with a cover of between 25-100% and uninvaded sites constituted sites where *H. drupacea* was absent. In cases where a site free of *H. drupacea* could not be obtained, few individuals were allowed to occur in uninvaded plots. Such percentage cover was as low as between 1-5 %, since IAP at such low density have little or no effects upon native vegetation composition and structure (Hejda and Pyšek, 2006; Catford et al., 2012). Cleared sites referred to recovered sites where *H. drupacea* and other alien plants have been removed.

Sites were located in close proximity to each other such that they had similar ecological conditions i.e. similar topography (altitude, slope, aspect), vegetation, soil type and land-use history (Hejda and Pyšek, 2006; Barney et al., 2015). A distance ranging between 20 m and 50 m was maintained between sites. This ensured that uninvaded or cleared plots were not affected by impacts such as shade from the invader. Most invaded sites considered were those with old invasions since that is where impacts could be assessed with a high level of confidence (Tererai et al., 2013). But due to lack of old site availability, three sites that were moderately invaded with 25-30% *H. drupacea* cover were included in the sampling.

2.1.4 Data collection

2.1.4.1 Vegetation survey

A vegetation survey was carried out between July and November 2016. For each site, plots of 5 x 5 m were temporarily demarcated using a measuring tape and plot corners marked with steel rods. The location for each plot was marked using a handheld Global Positioning System (Garmin GPS map 60CSx).

All individual plant species encountered in each plot were counted and identified up to species level where possible. Plants present in plots and all taxonomic authorities for plant names are given in Appendix 2. They were assigned to their origin status (alien or native) and growth forms in which they occurred. Plant specimens which could not be identified in the field were pressed for identification at Compton Herbarium, Kirstenbosch Botanical Gardens. Heights for all plant species were measured using a measuring tape and that of large trees were visually estimated. Woody plants found along plot margins were included if any of their parts fell inside the plot. In such cases, only the height of the part found in the plot was measured. For multi-stemmed plants, only the height of the tallest stem was recorded. Canopy cover percentage for each woody species was visually estimated. Percentage cover for grasses was estimated according to Braun-Blanquet (1972) cover classes as follows: 5: 75-100 %; 4: 50-75 %; 3: 25-

50 %; 2: 5-25 %; 1: 1-5 % cover. The presence and absence of leaf litter was noted, percentage cover visually estimated and depth of accumulated leaf litter in five subsamples per plot was recorded (Barney et al., 2015). The level of invasion for each plot was characterized by the cover of *H. drupacea.*

2.1.4.2 Soil sampling

Soil sampling was done at the same time and on the same plots as the vegetation survey. Five soil samples from four edges and the centre of each plot were collected. Soil was sampled 48 hours without rain (Barney et al., 2015). Any surface litter present was removed and the top 10 cm of the soil was collected with a standard soil auger. Five soil samples from each plot were combined and mixed thoroughly to form a bulk sample for the plot. Soil was sieved with a 2 mm mesh to remove large particles and stones. Any root fragments and debris were removed by hand. A representative sample of about 500 g was placed in labelled, clean, air-proof polythene plastic bags.

Soil samples were analysed for soil texture, soil moisture, pH_{KCL}, organic carbon (C), nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg) and sodium (Na). Nitrogen content was determined following the Kjeldahl method (Bremmer & Mulvaney 1982). Phosphorus, K, CA, Mg, Na and C were determined by using 1% citric acid. Soil was weighed, oven-dried at 105 °C for 48 hours and re-weighed to obtain percent soil moisture. The soil analysis was carried out at Western Cape Department of Agriculture, Elsenburg Laboratory. Their analysis methods are in compliance with standard procedures used by member laboratories of the inter-laboratory control scheme Agri Laboratory Association of Southern Africa (AgriLASA) and methods of the Association of Official Analytical Chemists (AOAC) (Guthrie, 2007).

2.2 DATA ANALYSIS

Species richness was calculated as the number of native species per plot. The Shannon-Weiner diversity index (H') was used to measure species diversity and calculated using the following formula: $H' = -\sum P_i \ln P_i \ (S)$, where P_i is the proportion of individuals found in the i^{th} species, ln is the natural logarithm of P_i, and S is the number of species in the community (Magurran, 2004). The Shannon-Weiner diversity index was calculated for each plot and per invasion category. Species collected in different plots for each invasion category were pooled to calculate the overall Shannon-Weiner diversity index.

The Shapiro-Wilk test was performed on all data to test for normality assumptions. The differences in vegetation parameters and soil properties between invaded and uninvaded sites was tested using an independent T-sample test for data following a normal distribution. Data for calcium deviated from a normal distribution and was log-transformed before analysis to meet normality assumptions. Data for species diversity deviated from normal distribution and could not be log transformed on account of the zeros present, and thus were tested using a non-parametric Mann-Whitney U test. The relationship between litter depth and canopy cover percentage of *H. drupacea* as an invader was tested using the Pearson correlation. Data from cleared sites were excluded from statistical analyses due to the relatively small sample size rendering the data not representative to merit general conclusions. All statistical analyses were conducted using the software IBM SPSS Statistics, Version 24.

2.3 RESULTS

2.3.1 Effects of *Hakea drupacea* invasion on species richness and diversity

A total of 129 species belonging to 75 genera and 35 families comprising mostly of shrubs were found (Appendix 2). The number of species recorded in invaded (n=12), uninvaded (n=12) and cleared (n=3) vegetation was 49, 94 and 27, respectively. The 49 species recorded in invaded vegetation comprised of 41 native (84%) and 8 alien (16%) species. The 94 species in uninvaded vegetation consisted of 89 native (95%) and 5 alien (5%) species and cleared sites had 27 species of which 26 (96%) were natives and 1 (4%) was an alien (Fig. 2.1).

Native species such as *Passerina corymbosa* (Thymelaeaceae) and *Erica* species (Ericaceae) dominated the uninvaded and cleared sites, but were absent where *H. drupacea* occurred. *Osyris compressum* (Santalaceae) was the main native species co-existing with the invader *H. drupacea*. Other IAP such as *Acacia saligna*, *A. longifolia*, *A. cyclops* (Fabaceae) and *Leptospermum laevigatum* (Myrtaceae) co-occured with *H. drupacea* in the invaded sites.

Mean species richness in uninvaded sites was nearly three times higher (10.50±3.18 mean±SD) than in invaded sites (3.83±3.54) (Table 2.1; Fig. 2.2). The difference was statistically highly significant ($t_{(22)} = -4.857$; $P < 0.001$). The uninvaded sites were more diverse than the invaded sites and the difference was significant (Mann-Whitney U test: $P < 0.008$). The species diversity mean rank declined from 16.29 in uninvaded to 8.71 in invaded plots (Table 2.1). Shannon-Weiner diversity indices were 3.69, 2.99 and 2.94 for uninvaded, invaded and cleared sites respectively, when the index was calculated for each invasion category rather than for each plot. The canopy cover of *H. drupacea* in invaded sites was significantly higher than that of dominant native species in the uninvaded sites ($t_{(22)} = 5.992$, $P < 0.001$; Fig. 2.4).

Plant heights in invaded sites were higher than those in uninvaded sites and the difference was highly significant ($t_{(247)} = 18.575$, $P < 0.001$; Fig. 2.5). Invaded plots had greater mean litter depth (11.75±5.07 cm) than litter from uninvaded plots did (0.43±0.35 cm) and the difference was highly significant ($t_{(22)} = 7.721$, $P < 0.001$; Fig. 2.6). A positive relationship existed between *H. drupacea* litter depth and its percentage canopy cover. ($r = 0.607$, $P = 0.036$; Fig. 2.7).

Table 2.1. Means (±SD) of vegetation parameters measured in *Hakea drupacea*-invaded and adjacent uninvaded sites at 12 sites across the *H. drupacea* vegetation range

Vegetation parameters	Invaded	Uninvaded	df	*t*	*P*
Height (m)	3.12±1.43(161)	0.94±0.25(83)	242	13.785	<0.001
Canopy cover (%)	62.92±24.07	17.92±9.88	22	5.992	<0.001
Litter depth (cm)	11.75±5.07	0.43±0.35	22	7.721	<0.001
Species richness	3.83±3.54	10.50±3.18	22	-4.857	<0.001
Species diversity	8.71	16.29	-	-	<0.008

The sample size is 12 unless otherwise indicated in parentheses. Percentage canopy cover and height values in invaded sites are for the invader *Hakea drupacea*. Percentage canopy cover in uninvaded sites is for the dominant native species in each plot. Height values in the uninvaded sites are for the dominant native species *Passerina corymbosa*. Data for species diversity are mean ranks. All statistical tests (at $P < 0.05$ significant level) were significant between the two sites.

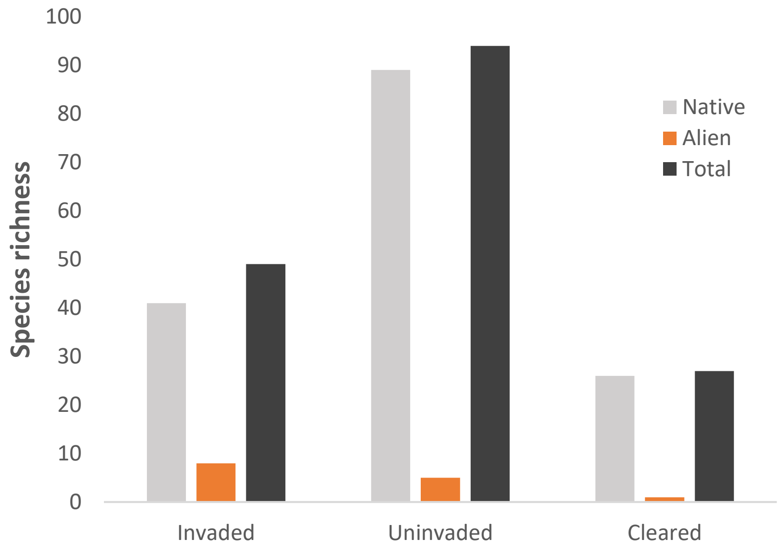

Figure 2.2. Plant species richness in *Hakea drupacea*-invaded (n=12), adjacent uninvaded (n=12) and cleared (n=3) sites in a 25 m^2 plot across the *H. drupacea* vegetation range.

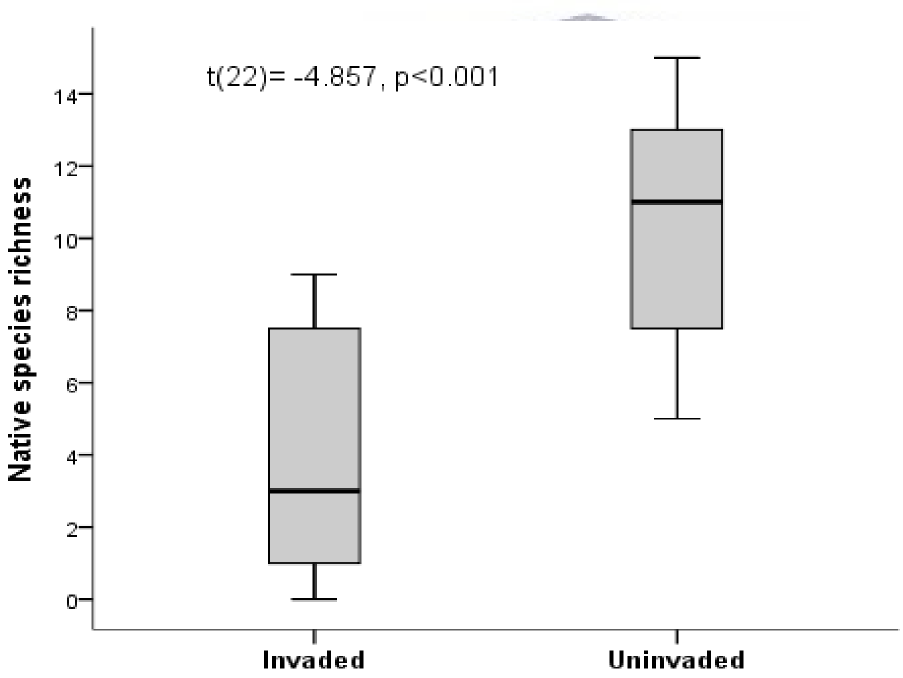

Figure 2.3. Mean native plant species richness in *Hakea drupacea*-invaded and adjacent uninvaded sites (n= 12) in a 25 m^2 plot across the *H. drupacea* vegetation range. The grey box represents the 1st and 3rd quartiles, dark horizontal line represents the mean and whiskers represent minimum and maximum values.

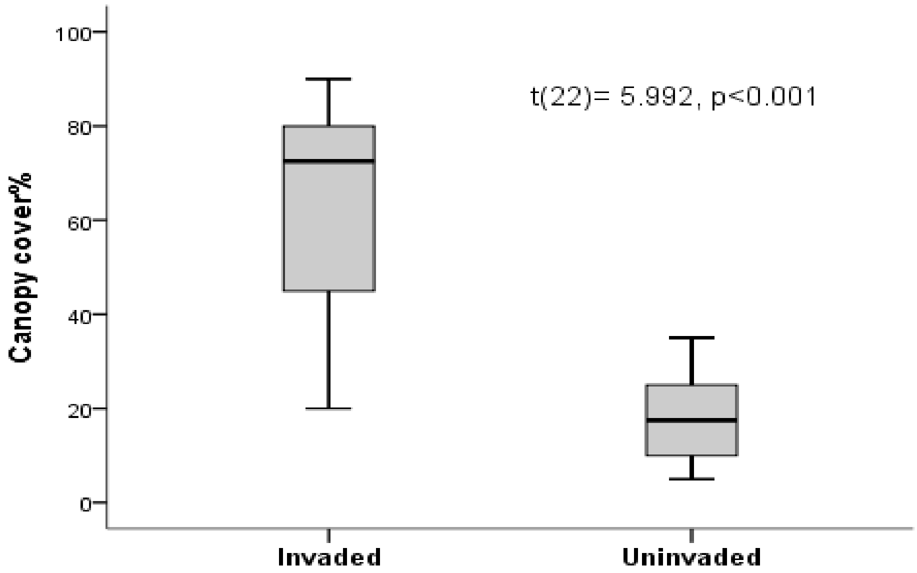

Figure 2.4. Mean canopy cover of *Hakea drupacea* in invaded sites and of the dominant native species in an adjacent uninvaded sites (n=12) in a 25 m^2 plot across the *H. drupacea* vegetation range.

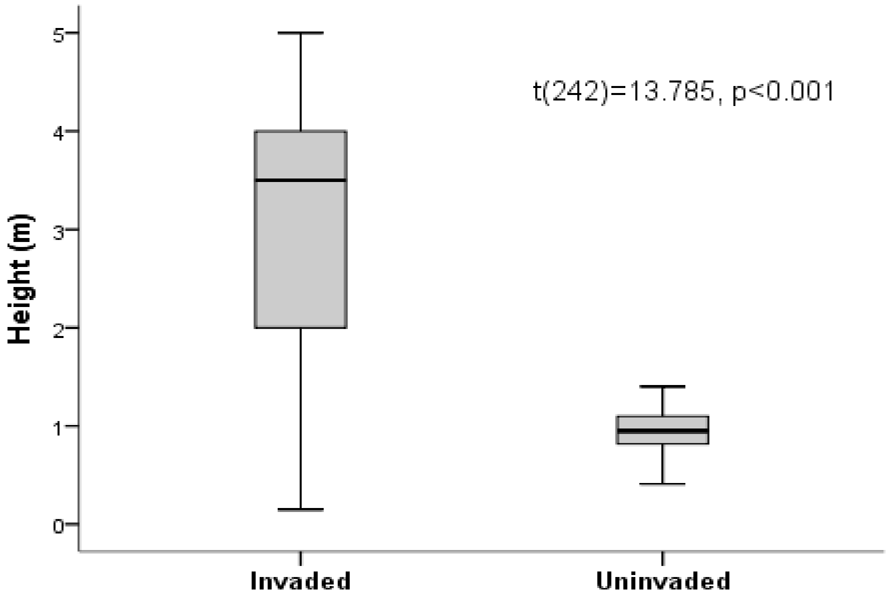

Figure 2.5. Mean heights of *Hakea drupacea* and the dominant species (*Passerina corymbosa*) in invaded and adjacent uninvaded sites (n= 12) in a 25 m^2 plot across the *H. drupacea* vegetation range.

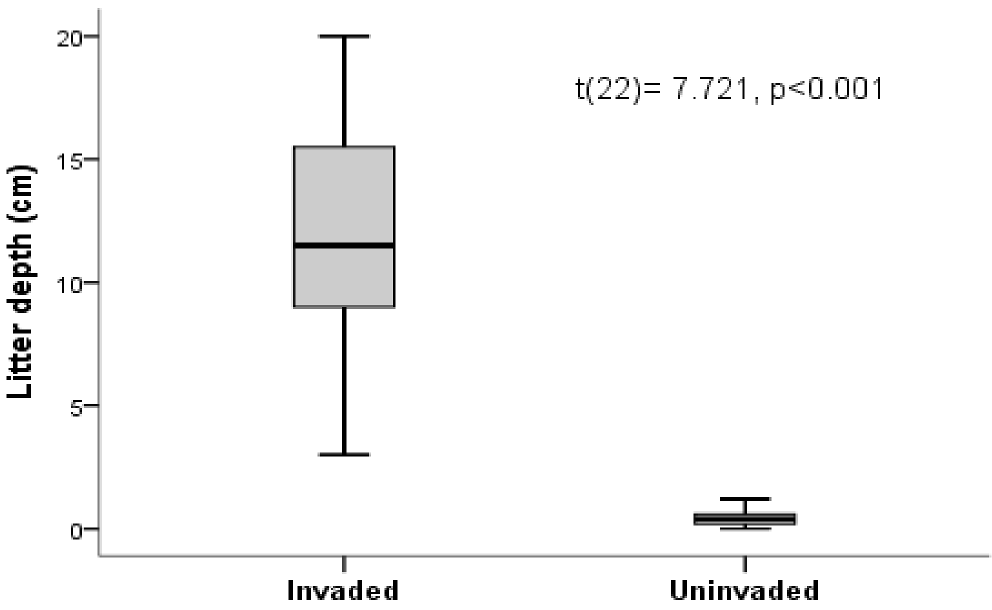

Figure 2.6. Mean litter depth of *Hakea drupacea* in invaded sites and native Fynbos plants in adjacent uninvaded sites (n=12) in a 25 m^2 plot across the *H. drupacea* vegetation range.

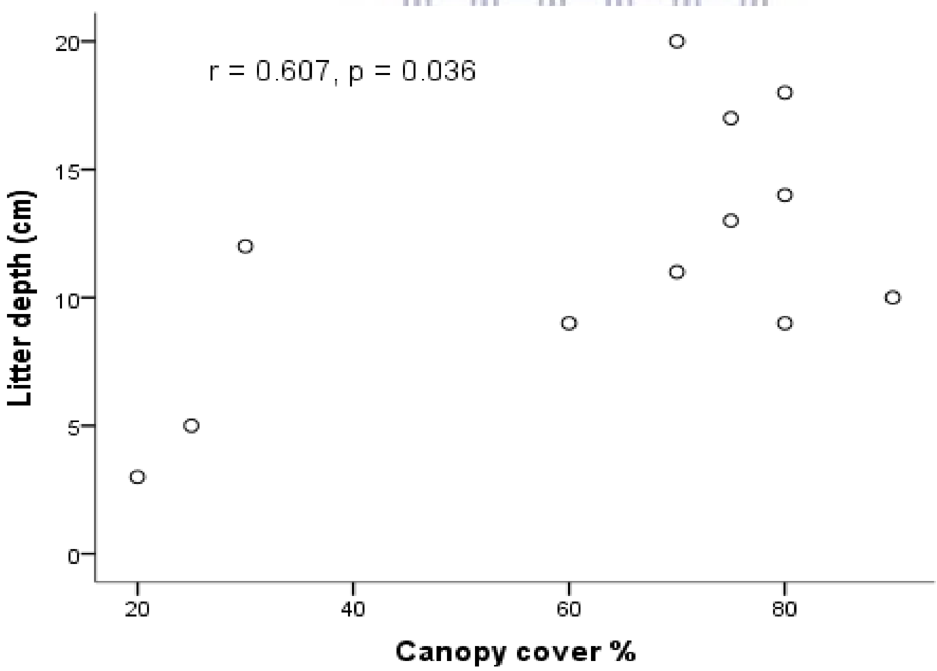

Figure 2.7. Scatter plot showing the relationship between *Hakea drupacea* canopy cover and its litter depth in invaded sites (n=12) in a 25 m^2 plot across the *H. drupacea* vegetation range.

2.3.2 Effects of *Hakea drupacea* invasion on soil properties

All plots across all sites had a sandy soil texture. All topsoil properties were unaffected by site type and had similar values in both invaded and uninvaded sites. Topsoil of invaded sites contained a slightly higher moisture, carbon, nitrogen, sodium and potassium content than the uninvaded sites, but the difference was not significant (Table 2.2).

Table 2.2. Means (±SD) of levels of topsoil chemical and physical properties in *Hakea drupacea*-invaded and adjacent uninvaded 12 sites across the *H. drupacea* vegetation range.

Soil properties	Invaded	Uninvaded	df	*t*	*P*
Texture	Sand	Sand	-	-	-
Moisture %	8.67±7.70	6.42±5.49	22	0.824	=0.419
pH_{KCL}	5.33±1.12	5.39±0.94	22	-0.138	=0.892
Carbon %	2.38±0.95(10)	1.72±0.82(10)	18	1.652	=0.116
Nitrogen %	0.11±0.07	0.09±0.05	22	0.657	=0.518
Phosphorus mg/kg	9.50±9.67	8.42±6.09	22	0.328	=0.746
Potassium mg/kg	79.08±37.68	59.08±32.48	22	1.393	=0.178
Sodium mg/kg	136.00±46.99	88.25±103.37	22	1.457	=0.159
Magnesium cmol/kg	1.79±0.82	1.45±0.54	22	1.221	=0.235
Calcium cmol/kg	0.69±0.57	0.68±0.43	22	0.037	=0.971

Data are means±SD of 12 replicates unless otherwise indicated in parentheses. All statistical tests (at $P < 0.05$ significant level) were not significant between the two sites.

2.4 DISCUSSION

Effects of *H. drupacea* observed on the vegetation structure supported our hypothesis relating to the reduction in species richness and diversity of native plant species in the presence of *H. drupacea* stands. This was generally supported by findings in previous studies. In contrast, no effect on soil properties was detected as predicted and this contrasted with a high proportion of previous studies.

2.4.1 Effects of *H. drupacea* invasion on species richness and diversity

Hakea drupacea invasion significantly reduced species richness and diversity. This indicates that it has a negative effect on the native plant community structure. Such reduction can be attributed to *H. drupacea* high canopy cover, height and litter production that are either acting individually or in combination.

Hakea drupacea forms high canopy cover that produces shading effects thereby reducing light availability to the understory species that ultimately limits and suppresses the growth of short-statured native species such as those reported by Bravo-Monasterio et al. (2016). The Fynbos

Biome is dominated by herbaceous and low shrubs species and light availability is naturally high due to the lack of trees as growth forms (Esler et al., 2014). The presence of a tall alien shrub or tree, such as *H. drupacea*, can therefore be detrimental to short-statured native species due to high shading effects. Reduction in light availability can alter environmental conditions within the site whereby certain species will tolerate such new conditions created by an invasion (Bravo-Monasterio et al., 2016). Light availability was not quantified in this study, but it is one of the mechanisms responsible for native plant species displacement (Braithwaite et al., 1989; Jäger et al., 2007; Bravo-Monasterio et al., 2016).

Findings of negative effects exerted by *H. drupacea* in this study are consistent with the results of Richardson et al. (1989) who reported dense stands of IAP, including *Hakea* species, suppressing understory species richness and diversity in the Fynbos Biome. This effect was largely attributed to canopy closure caused by IAP. Similar negative effects of a *Pinus contorta* Douglas invasion, which possess morphological and ecological characteristics similar to *Hakea* species, was reported in treeless ecosystems in Chile (Bravo-Monasterio et al., 2016). There is accumulating evidence supporting the premise that tree invasion into areas naturally deprived of trees can create novel conditions which may exclude native species (Jäger et al., 2007; Rundel et al., 2014; Bravo-Monasterio et al., 2016).

The accumulation of litter observed under *H. drupacea* canopies in this study has likely, partly contributed to the reduction of species richness and diversity in invaded sites. This finding agrees with previous studies which reported litter to be responsible for a reduction in native species richness (Williams and Wardle, 2007; Mitchell et al., 2011; Dostál et al., 2013; Bravo-Monasterio et al., 2016). A litter layer of 5-10 cm deep has been reported to limit recruitment of other species (Olson and Wallander, 2002). The thick litter layer averaging 11.75±5.066 cm observed under *H. drupacea* stands in the present study, and reaching up to 20 cm thick in some sites, is 3-4 times greater than that reported to limit plant recruitment by Olson and Wallander (2002). It should therefore be expected to have negatively influenced the plant community structure in the present study.

The presence of a thick litter layer under the canopy of IAP is a common occurrence (Evans et al., 2001; Yelenik et al., 2004; Williams and Wardle, 2007; Hata et al., 2010). It has been reported to influence the structure of native plant communities by changing light and water availability (Facelli and Pickett, 1991), inhibiting seed germination and also growth of seedlings (Olson and Wallander, 2002; Hata et al., 2010). Furthermore, other micro-environmental changes produced by litter accumulation beneath IAP include influencing nutrient supply rates (Versfeld, 1981; Richardson and Kluge, 2008). In the present study, the depth of litter in sites invaded by *H. drupacea* increased with the increase in canopy cover percentage. This indicates that more litter is expected to accumulate with increasing age of invasion and canopy cover closure. This will potentially increase fire risk by increasing biomass of flammable material leading to an increase in fuel loads (Van Wilgen and Richardson, 1985; Williams and Wardle, 2007).

Susceptibility to invasion differs between native species, with some being excluded because of invasion, and others not (Stinson et al., 2007; Bravo-Monasterio et al., 2016). This may be the case with native species such as *Passerina corymbosa* and *Erica* species which occurred in uninvaded and cleared sites, but not in invaded sites. These plant species may have attributes incongruous with new conditions created by invasion and they are gradually filtered out (Weiher and Keddy, 1995). They may, for example, not be adapted to higher shade levels created by *H. drupacea*. However, species such as *Osyris compressum* tolerated the presence of *H. drupacea*, especially in sites where the canopy cover was low. It is unknown if such native species will continue to co-exist under high canopy cover. Presently, canopy closure in the study sites was not completely attained in most populations. Evidence has shown that only ruderal species are expected to co-exist with invaders in invaded sites (Hejda and Pyšek, 2006; Bravo-Monasterio et al., 2016). These can be shade-tolerant species that require high humidity, and that are favoured by conditions created by invasives such as *H. drupacea*, e.g. increases in canopy cover.

The presence of *H. drupacea* not only reduced the richness and diversity of indigenous species, but it also appeared to have facilitated other IAP, particularly *Acacias* and *L. laevigatum* by providing disturbed conditions favourable for their establishment. This concurs with the invasional meltdown hypothesis, which states that alien species facilitate one another's invasion (Simberloff and Von Holle, 1999). This will increase the invasibility of the community and ultimately increase the number of IAP and their impacts (Simberloff, 2006; O'Loughlin and Green, 2017).

2.4.2 Effects of *Hakea drupacea* invasion on soil properties

Our results did not reveal a significant difference in topsoil characteristics between the invaded and uninvaded areas. The lack of effects on soil properties in this study are in contrast with a high proportion of previous studies which reported alteration of soil properties in areas invaded by IAP (Vanderhoeven et al., 2005; Heneghan et al., 2006; Dassonville et al., 2007; Zhang et al., 2009; Afreen et al., 2017).

Changes in soil properties do not necessarily always occur with invasion. Failure to detect significant impacts on soil properties have been reported in other studies (Hook et al., 2004; Vanderhoeven et al., 2006; Stefanowicz et al., 2017). Moreover, bidirectional and neutral effects have been reported in the literature, suggesting that there is no specific pattern in the response of soil properties to alien invasion. Effects vary with soil type and the identity of invading species (Dassonville et al., 2008; Scharfy et al., 2010) and may be pronounced at any stage of an invasion (Medina-Villar et al., 2016).

Explanations on the lack of significant invasion effects observed on soil properties in this study may be three-fold: Firstly, it may be too early in the invasion stage to exert any effect, particularly as some sites were not heavily invaded causing results to be insignificant. Change in soil properties require a longer time in some species (Vilà et al., 2006; Medina-Villar et al., 2016). Secondly, the lack of altered soil properties can stem from native Fynbos species having more functional traits similar to *H. drupacea*. Soil conditions are altered when alien and native species differ in main functional traits, such as quality and quantity of litter (Levine et al., 2003; Scharfy et al., 2010). Thirdly, it may suggest that the direction of either an increase or decrease differs among replicate sites, causing lack of patterns in soil properties changes caused by *H. drupacea* invasion.

The pre-invasion site history such as disturbance (e.g. fire) and age of invasion was unknown in the present study. This uncertainty is common in many invasion studies using the multi-scale comparison approach (Adair and Groves, 1998; Hejda et al., 2009; Stricker et al., 2015). It could therefore be possible that impacts observed were mediated by other factors than *H. drupacea* invasion, causing it to be a passenger and not the driver of changes observed (Macdougall and Turkington 2005). Moreover our results spanned different seasons and may have been affected by seasonal changes. This involves changes concerning soil properties, such as moisture and nutrient concentrations, since these are largely influenced by seasonal changes.

Future studies should be conducted within the shortest possible time period to avoid differences among sites which may arise from seasonal vegetation dynamics.

2.5 MANAGEMENT IMPLICATIONS

Results of this study demonstrated that *H. drupacea* invasion has a significant impact on the Fynbos vegetation structure, but not on the soil properties of the invaded ecosystem. Such a strong impact on the vegetation structure at the community level is expected to be manifested at the landscape level (Hulme and Bremner, 2006). This constitutes a threat to the conservation of native biodiversity in the Fynbos Biome where alien invasive plants have been implicated in the extinction of seven plant species with a further 1207 at risk (Esler et al., 2014). This suggests that there is a need to control *H. drupacea* and prevent its further establishment and spread.

In the absence of any control measures, its density will increase and even cause more severe effects. It may also facilitate the establishment and spread of other alien invasive species such as *A. saligna*, *A. cyclops* and *L. laevigatum* that were observed to co-occur with *H. drupacea* at the sites studied. Similar attention should be paid to these alien invasive plants co-occuring with *H. drupacea* since they may pose equally negative effects on native vegetation.

Hakea drupacea is already widespread in the Western Cape Province that it would be difficult to eradicate (Erckie, 2014). Control attempts conducted by the Working for Water Programme and various nature reserves in the Western Cape Province are necessary to mitigate impacts posed and should be encouraged. *Hakea drupacea* possess biological features such as being a serotinous (Richardson et al., 1987) non-resprouter (Fugler, 1982), which can enhance successful control. Control strategies should involve clearing and burning afterwards to remove its litter layer in order to promote vegetation recovery (Richardson and Kluge, 2008).

The recovery of native vegetation after clearing *H. drupacea* without actively restoring soil conditions is expected since no changes in soil properties occurred in this study. However, the potential ability of native vegetation to recover on its own after clearing is generally considered low when it involves transformer species, such as *Acacia* species, which cause greater changes in soil properties (Yelenik et al, 2004; Mostert et al 2017) than *H. drupacea.*

2.6 CONCLUSION

Findings of this study revealed that the presence of *H. drupacea* can suppress the establishment of native plant species. The low species richness and diversity observed in invaded plots can be ascribed to taller heights, high canopy cover and litter accumulation beneath *H. drupacea* stands. Lack of effects associated with *H. drupacea* invasion on soil properties can potentially be attributed to the young age of invasion or possession of similar functional traits between *H. drupacea* and native plant species. This suggests that it may not be a requirement to facilitate active ecological restoration. It is necessary to reduce the invasive *H.drupacea* density to a manageable level in order to conserve native plant species richness and diversity. The removal of *H. drupacea* litter from invaded areas should also be considered to enhance vegetation recovery. Future studies should quantify nutrients concentration in plant biomass and include representative cleared sites to determine legacy effects. By assessing the impacts of the invasive *H. drupacea*, this study provided a set of impact variables that significantly alters vegetation structure.

Chapter 3

Impacts of an invasive alien *Banksia speciosa* on a native plant-pollinator interaction

Abstract

Invasive alien flowering plants can disrupt native pollinator networks, affecting pollination and reproductive success of native plants. These effects could be either positive or negative with few studies reporting neutral effects. Most studies concerning biotic pollination have focused on insects and adopted an experimental approach. This study compared native *Protea compacta* and invasive alien *Banksia speciosa*, both bird-pollinated, to determine whether the invasive species competes with the native for pollinators or acts as a magnet species. Nectar-feeding bird abundance and richness, flower visitation rates, nectar properties and seed set were measured and compared between *B. speciosa*-invaded and uninvaded sites in the Agulhas National Park, South Africa. Pollinator exclusion and breeding system experiments were conducted to determine autonomous seed production and the degree of pollen limitation. *Banksia speciosa* flowers produced nectar with low sucrose concentration but supported a significantly higher abundance of sugarbirds ($F_{2,38}$=19.395, P < 0.001) but a lower abundance of sunbirds ($F_{2,38}$ = 10.903, P < 0.001) than *P. compacta*. There was no significant reduction in nectar-feeding bird species richness in *P. compacta* site. Study species displayed different levels of self-compatibility, but both relied on pollinators to enhance seed set. Pollen limitation was not responsible for the low seed set observed in inflorescences to which pollen was added. Results suggest that the presence of a flowering invasive *B. speciosa* has a competitive effect on the attraction of sugarbirds compared with native *P. compacta*. The attraction of sugarbirds by the invasive *B. speciosa* may adversely affect the reproductive success of the native *P. compacta* due to the latter's dependence on sugarbirds for pollination.

Keywords: *Banksia speciosa*; Cape Floristic Region; Cape sugarbirds; nectar-feeding birds; plant-pollinator interaction; *Protea compacta*

3 INTRODUCTION

Impacts of invasive alien plant (IAP) species have become a worldwide concern (Levine et al., 2003; Kandori et al., 2009; Vilà et al., 2011). Invasive alien plant species compete with native plant species for both abiotic resources such as water, nutrients, light, space (Braithwaite et al., 1989; Vanderhoeven et al., 2006; Vilà et al., 2006; Dassonville et al., 2008) and biotic resources such as seed dispersal (Bass et al., 2006) and pollination (Chittka and Schürkens, 2001; Brown et al., 2002; Moragues and Traveset, 2005; Ye et al., 2014), which are important for plant species' survival and reproduction. Their invasion into natural and protected areas is undesirable due to the importance of these areas in biodiversity conservation (Skurski et al., 2013). It is thus essential that ecological processes such as pollination and seed dispersal of native species should remain undisturbed in these areas (Rebelo, 1987).

Competition between alien and native plant species for abiotic resources (Braithwaite et al., 1989; Allsopp and Holmes, 2001; Ehrenfeld, 2003; Hejda et al., 2009; Chamier et al., 2012) and between native species for biotic resources (Campbell and Motten, 1985; McGuire and Armbruster, 1991; Gardner and Macnair, 2000; Bell et al., 2005) have been well studied. Conversely, studies concerning competition for pollination between native and alien species have been limited until relatively recently (see for example Lopezaraiza-Mikel et al., 2007; Bartomeus et al., 2008). Furthermore, competition for pollinators between native and alien species can be more extreme due to the absence of evolutionary mechanisms allowing them to co-exist (Kandori et al., 2009).

Invasive alien plants with showy flowers that share similar pollinator guilds with native plants may facilitate or disrupt native flower visitations (Chittka and Schürkens, 2001; Brown et al., 2002; Traveset and Richardson, 2006; Dietzsch et al., 2011). Their presence can cause native pollinators to include floral resources from invasive plants in their diets (Memmott and Waser, 2002; Goodell and Parker, 2016). This can cause changes in flower visitation (Kandori et al., 2009; Powell et al., 2011; Goodell and Parker, 2016), alteration in seed set (Brown et al., 2002; Muñoz and Cavieres, 2008) and reduced pollen quality and quantity (Brown and Mitchell, 2001; Da Silva and Sargent, 2011) with the potential to affect reproductive success of native plants.

Most studies on plant-pollinator interactions have focused on insects, but those concerning birds have been relatively rare (but see Geerts et al., 2013). Such studies have found positive

(Moragues and Traveset, 2005; Bartomeus et al., 2008; Russo et al., 2016), negative (Chittka and Schürkens, 2001; Brown et al., 2002; Moragues and Traveset, 2005; Dietzsch et al., 2011; Goodell and Parker, 2016) and neutral effects (Aigner, 2004; Moragues and Traveset, 2005) on competition for pollination between invasive alien and native plant species. The direction of such outcomes have been reported to depend on the density of the IAP (Dietzsch et al., 2011; Stout and Casey, 2014), the composition of native flowering plants and pollinators within the invaded community (Larson et al., 2006; Bartomeus et al., 2008; Stout and Morales, 2009).

Most previous pollination studies concerning competition between native and IAP have adopted experimental approaches to study these interactions by using potted plants (Kandori et al., 2009; Flanagan et al., 2010) and rarely tested effects of IAP in a natural setting. There is thus a need for studies using natural field experiments. Moreover, a dense stand of *Banksia speciosa* R.Br. was selected for the present study, since it is not only the presence, but also the density of IAP that is important in assessing pollinator dynamics (Flanagan et al., 2010; Dietzsch et al., 2011; Stout and Casey, 2014).

Relatively little is known about impacts of IAP on competition for pollination with native plants in the Fynbos Biome (Rebelo, 1987), despite the Fynbos having high species richness and endemism (Manning and Goldblatt, 2012). The genus *Banksia* has 14 species introduced for floriculture (Moodley et al., 2013) and studies have demonstrated that most of them have a high risk of becoming major invaders in the Fynbos Biome (Richardson et al., 1990; Honig et al., 1992). The invasion of *Banksia ericifolia* L.f. has already shown impacts at the ecosystem level with a potential to disrupt networks of indigenous pollination (Geerts et al., 2013). Consequently, there is a need to assess the ecological impacts of invasive alien *B. speciosa* to help conserve the rich Fynbos flora and associated fauna.

Banksia speciosa and *Protea compacta* R.Br. serve as ideal species for this study as they flower simultaneously, share habitat and pollinators'guilds, and are from the same family. Moreover, *B. speciosa* possesses large and showy inflorescences (George, 1981) with the potential to attract many pollinators. Both species are reported to be primarily bird-pollinated (Paton and Turner, 1985; Collins and Rebelo, 1987; Steenhuisen et al., 2012) particularly by Cape sugarbirds and sunbirds (Mostert et al., 1980; Moodley et al., 2016).

It is hypothesized that *B. speciosa* is able to increase its density and spread with the potential to attract more pollinators than the native *P. compacta*, as well as increase its own seed set. Its increased density may attract potential pollinators away from *P. compacta* particularly if it offers more rewarding nectar resources, thereby negatively impacting its reproductive success. On the other hand, the presence and density of *B. speciosa* may have no effect on the native *P. compacta* although this is unlikely (see Aigner, 2004; Moragues and Traveset, 2005).

The objective of this study was to quantify impacts of *B. speciosa* invasion on a plant-pollinator interaction of the native *P. compacta* in the Agulhas National Park. Specifically, it sought to answer the following questions: 1) Does the invasive alien *B. speciosa* influence nectar-feeding bird abundance, richness and flower visitation rates to native *P. compacta*? 2) What is the nectar volume and concentration produced and does this have any influence on pollinator visitation to the two study species? 3) Does the altered visitation rate to *P. compacta*, if any, influence its seed set?

3.1 MATERIALS AND METHODS

3.1.1 Study species

Banksia speciosa is a perennial woody shrub, that grows up to 8 m tall (George, 1981; George, 1987), and was introduced from Australia to South Africa for the cut-flower industry (Richardson et al., 1990). The cream to pale yellow hermaphroditic and protrandrous flowers are grouped into a long inflorescence 4-12 cm (George, 1981; Fig. 3.1A). They are visited by birds, insects (Moodley et al., 2016) and small mammals (Collins and Rebelo, 1987) for pollen and nectar. It flowers throughout the year both in its native range (George, 1981; George, 1987) and in South Africa (Erckie, pers. obs.).

Protea compacta is a non-sprouting serotinous shrub of up to 3.5 m tall that is endemic to South Africa (Rourke, 1982; Rebelo, 2001). Pink and, on rare occasions, white flowers of *P. compacta* are grouped into large inflorescences (Rebelo 2001; Fig. 3.1B). Flowers have reportedly been visited by birds and insects (Collins and Rebelo, 1987), with birds as effective pollinators (Mostert et al., 1980). It is known to flower during winter months from April - September with a peak in May and June (Rebelo, 2001).

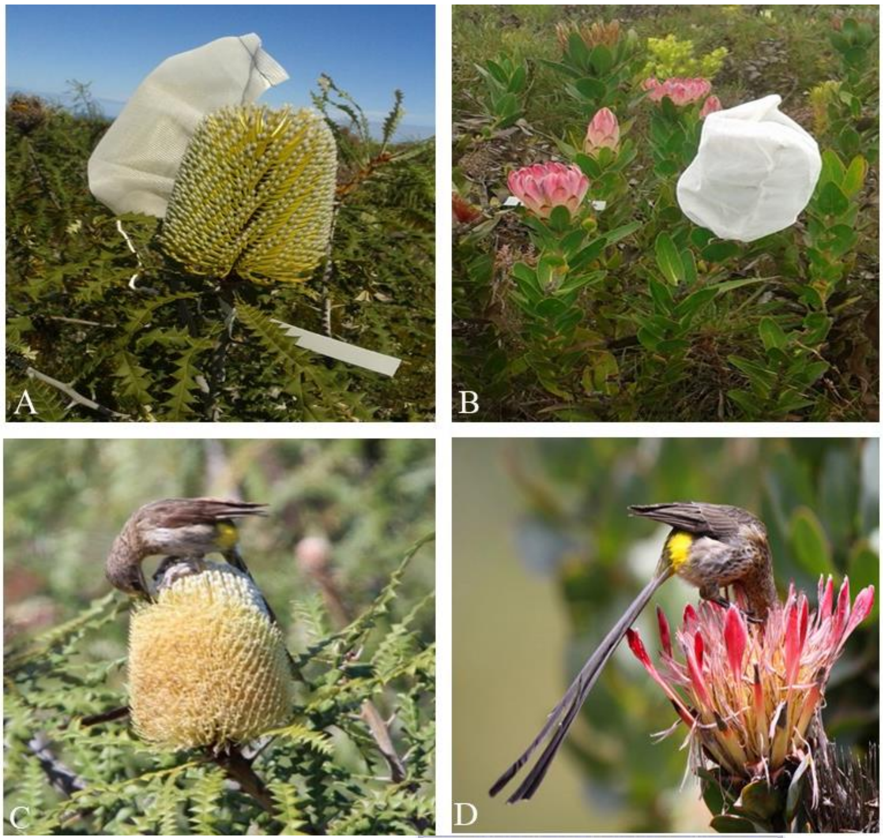

Figure 3.1. Illustration of the experimental design of plants allocated to three pollination treatments: (A) bagged and un-manipulated to test for auto fertility; (B) tagged and cross-pollinated (C) tagged and left open to receive natural pollination (D) Male Cape sugarbird feeding on *P. compacta* nectar. Photographs: A-Ernita van Wyk; B, D- Laimi Erckie; C-Sjirk Geerts

3.1.2 Study area

Study sites were located at Soetanysberg (34°44'667 S; 19°52'931 E) and Bergplaas (34°43'585 S; 19°52'488 E) in the Agulhas National Park, Western Cape Province, South Africa. The area experiences a Mediterranean-type climate characterized by cold wet winters and warm dry summers. It receives mean annual rainfall of 452 mm. The mean temperature ranges between 13.5 °C and 20.6 °C for the cool and warm months, respectively (Richards et al., 1995). Soils are derived from Table Mountain sandstone and are acidic and nutrient-poor (Thwaites and Cowling, 1988).

Local fire records of the South African National Parks (SANParks) indicate that the study site at Soetanysberg was last burnt in 2009. The age class of both study species was therefore similar across stands since they are both strongly serotinous species (George, 1981; Rourke, 1982; George, 1987; Rebelo, 2001) with recruitment occurring only after fire (LeMaitre, 1985). Sites were previously cut-flower farms where *B. speciosa* and *P. compacta* were cultivated before the land was acquired and incorporated into the Agulhas National Park (Kraaij et al., 2009).

Field observations were conducted at three sites during the peak flowering months of *B. speciosa* and *P. compacta*. Both plant species are pollinated by the same functional group of pollinators (Collins and Rebelo, 1987) and their flowering seasons overlap. Native vegetation invaded by a large dense monospecific stand of *B. speciosa* (the *Banksia* site) and an adjacent large monospecific stand of native *P. compacta*, referred to as *P. compacta* adjacent site, were located 10 m apart at Soetanysberg. *Banksia speciosa* and the *P. compacta* populations cover an area of 2 and 1.5 ha, respectively. Another stand of *P. compacta* referred to as the *P. compacta* control site located 2.6 km away from *P. compacta* adjacent, served as a control plot and was located at Bergplaas. *Protea compacta* was the most abundant native bird-pollinated flowering species at both Soetanysberg and Bergplaas.

3.1.3 Data collection

3.1.3.1 Bird observations

To determine nectar-feeding bird abundance and richness in invaded and uninvaded sites, a point count method was employed. This is a widely used approach to get estimates of relative abundance of terrestrial birds (Bibby, 2000). Due to small stands' size, only one fixed point was established within each stand and marked with flagging tape and a Global Positioning System (GPS). Bird observations were conducted for 8 days between 7:00-12:00 when nectar-feeding birds are most active, during good weather conditions without rain and little or no wind. The species and number of nectar-feeding birds seen within a 40 m radius from the observer were recorded. Bird observations were carried out by two observers, with Nikon 8 x 42 binoculars, and lasted for 10 minutes at each point. Sites were observed twice per day and were sampled in an alternating manner. The *Banksia*, *Protea* adjacent and *Protea* control sites were sampled for 14, 15 and 12 times, respectively.

3.1.3.2 Flower visitations

To determine bird visitation rates to the flowering *B. speciosa* and *P. compacta* inflorescences, 20 focal inflorescences were randomly selected within each site. The number of bird visits to focal inflorescences were recorded for 30 minutes at each point. Birds were only considered as potential pollinators once they made contact with the flower's receptive part (Gibson et al., 2013; Stout and Casey, 2014). Sites were observed at least twice per day and were sampled in an alternating manner to eliminate any temporal bias in observations. A total of 420, 450 and 360 minutes for flower visitations were accumulated for the *Banksia*, *Protea* adjacent and *Protea* control sites, respectively.

3.1.3.3 Nectar properties and availability

To determine the quantity and quality of floral nectar rewards available to nectar-feeding birds, a 5 x 5 m plot was demarcated in each of the three sites. The number of individual plants and inflorescences in each plot were counted. Nectar from between 12-17 open flowers on ten flowering inflorescences of *P. compacta* was measured. For *B. speciosa*, nectar from all open flowers within the inflorescences was measured. Nectar was extracted from flowers of *B. speciosa* and *P. compacta* using 5 µL and 40 µL micro-capillary tubes, respectively. Nectar sugar concentration was determined with a 0-50 % handheld refractometer. Nectar volume and concentration were converted to milligrams of sucrose per flower and the total amount of sucrose produced per inflorescence obtained. The number of inflorescences per plot were counted and multiplied with the average milligrams of sucrose per inflorescence to calculate total sugar per plot.

3.1.3.4 Breeding system experiments

To determine the capability for autonomous seed production and degree of pollen limitation of *B. speciosa* and *P. compacta*, exclusion of pollinators and pollen addition experiments were conducted. A method similar to that of Moodley et al. (2016) was used. Between 15 and 30 plants of each species were randomly selected in *B. speciosa* and *P. compacta* stands and randomly allocated to one of the three treatments: a) one inflorescence still in a bud phase was bagged with fine-mesh nylon bags to exclude pollinators (Fig. 3.1A & B), b) another inflorescence with about 70-90% receptive flowers was tagged and pollen was added to all open flowers from multiple pollen donors within the same population by rubbing it onto receptive stigma with a 25 mm paint brush, and c) one inflorescence as a control was tagged and left uncovered to allow access by pollinators (Fig. 3.1C & D).

Infructescences were harvested 6 months after the completion of flowering to determine seed production. Follicles of *B. speciosa* were opened by heating infructescences in the oven for between 2-30 days at 120 °C. Some follicles that were not completely dry took longer to open, lengthening their period in the oven until all seeds had been released. Seeds were removed from follicles using tweezers and were counted. Seeds of *P. compacta* were counted immediately upon harvesting their infructescences. The ability for each species to set seed without a pollinator and degree of pollen limitation was measured by comparing seeds per infructescence between pollinator excluded, natural pollination and pollen addition treatments (Goldingay and Carthew, 1998; Powell et al., 2011).

3.2 STATISTICAL ANALYSIS

The Shapiro-Wilk test was performed on all data to test for normality. The difference in nectar-feeding bird abundance, flower visitation and nectar volume per inflorescence among different sites was tested using a one-way Analysis of Variance (ANOVA) since data met ANOVA assumptions. A Tukey post-hoc multiple comparison test was used to evaluate differences among sites. Due to a low number of different species of sunbirds observed, all sunbirds were pooled in order to conduct statistical tests.

A non-parametric test, Kruskall-Wallis was used to compare the difference in the number of seeds between pollinator excluded, natural pollination and pollen addition treatments as the data was non-parametric. For significant results, a Mann-Whitney U test was used to determine which treatments were significantly different from each other. To assess the effects of pollination treatments, the proportion (%) of inflorescences which produced seeds and number of seed produced per inflorescence were used as measures. All statistical analyses were conducted using the software IBM SPSS Statistics, Version 24.

3.3 RESULTS

3.3.1 Bird observations

A total of four species of nectar-feeding birds were observed: Cape sugarbird (*Promerops cafer*), Orange-breasted sunbird (*Anthobaphes violacea*), Southern double-collared sunbird (*Cinnyris chalybea*) and Malachite sunbird (*Nectarinia famosa*). The opportunistic nectar feeding Cape weavers (*Ploceus capensis*) were also observed but were excluded from the analysis due to their low visitation rates. Cape sugarbirds were the most frequent visitors and accounted for 90% of visits in the *Banksia* site followed by the Orange-breasted sunbirds with 19%, with no Southern double-collared sunbirds or Malachite sunbirds observed (Table 3.1). There was significant differences in Cape sugarbirds and sunbirds visits between *Banksia* and both *Protea* sites with significantly higher numbers of sugarbirds visiting the *Banksia* site (one-way ANOVA: $F_{2,38}$=19.395, P < 0.001) while sunbird visits to the *Banksia* site were significantly lower ($F_{2,38}$ = 10.903, P < 0.001; Fig. 3.2). No significant difference in visitations were observed between *Protea* adjacent and *Protea* control sites for both Cape sugarbirds ($F_{2,38}$ = 19.395, P = 0.261) and sunbirds ($F_{2,38}$ = 10.903, P = 0.136).

Nectar-feeding bird species richness between sites varied from 2 to 4 (Table 3.1). The *Protea* control site had the highest number of species and was the only site where Southern double-collared sunbirds was recorded. Only two species of nectarivorous birds visited the *Banksia* site and no species was exclusive to this site.

Table 3.1. Average number of nectar-feeding birds observed during the 10-minute point count at three sites in the Agulhas National Park

Bird species	Site		
	Banksia	*Protea* adjacent	*Protea* control
Promerops cafer (Cape sugarbird)	13 (90)	4 (48)	6 (52)
Anthobaphes violacea (Orange-breasted sunbird)	1 (10)	2 (49)	3 (29)
Cinnyris chalybea (Southern double-collared sunbird)	0	0	1 (11)
Nectarinia famosa (Malachite sunbird)	0	0.3 (3)	1 (8)

Percentages (%) of nectar-feeding birds are indicated in parentheses. Number of 10-minutes observation periods were 14, 15 and 12 for *Banksia*, *Protea* adjacent and *Protea* control, respectively.

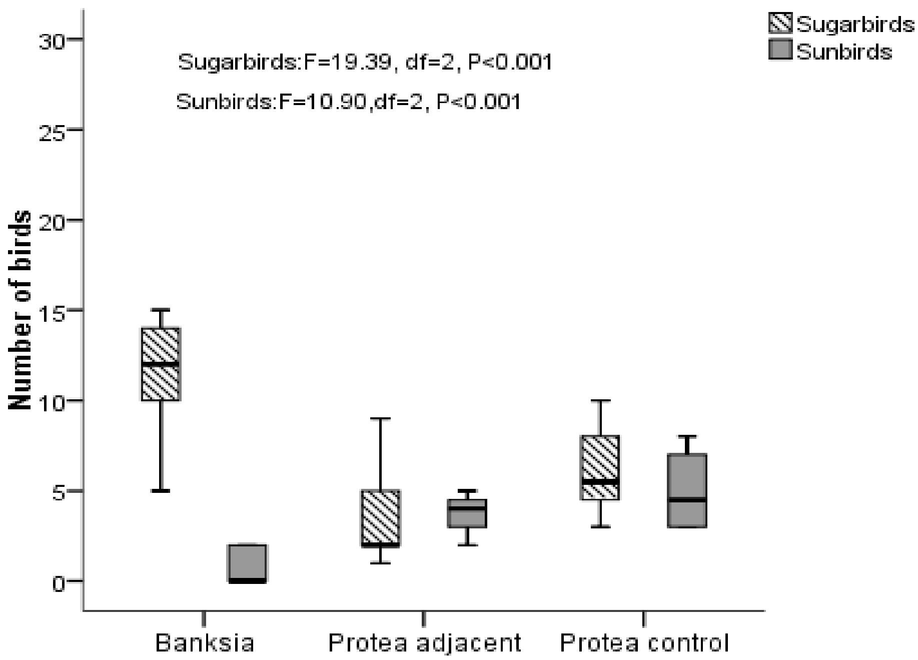

Figure 3.2. Mean number of sugarbirds and sunbirds observed during the 10-minute point count at three sites in the Agulhas National Park, *Banksia* (n=14), *Protea* adjacent (n=15) and *Protea* control (n=12). n is the number of 10-minutes observation periods. The box plot shows the 1st and 3rd quartiles (grey and patterned box), median (dark horizontal bar) and whiskers represent minimum and maximum values.

3.3.2 Flower visitations

Cape sugarbirds used flowers in buds as a perch since flowers open from bottom to top in *B. speciosa* (Fig. 3.1C). In *P. compacta* they perched on the inner buds as flowers open from the outside to the inside (Fig. 3.1D). Sunbirds perched on *P. compacta* in a similar manner as sugarbirds and made contact with reproductive parts of both study species.

Cape sugarbirds visited significantly more *B. speciosa* flowers than native *P. compacta* flowers in *Protea* adjacent and *Protea* control (one-way ANOVA: $F_{2,38} = 66.66$, $P < 0.001$). Cape sugarbird flower visitation did not differ significantly between *Protea* adjacent and *Protea* control ($F_{2,38} = 66.663$, $P = 0.473$; Fig. 3.3). Visitation rates to flowers by sunbirds differed between *Banksia* and *Protea* control and between *Protea* adjacent and *Protea* control ($F_{2,38} = 16.891$, $P < 0.002$), with significantly more sunbirds visits at the *Protea* control site and significantly low sunbird visits at the *Banksia* site ($F_{2,38} = 16.891$, $P < 0.001$). Visitation rates to flowers by sunbirds did not differ significantly between *Banksia* and *Protea* adjacent sites ($F_{2,38} = 16.891$, $P = 0.080$; Fig. 3.3).

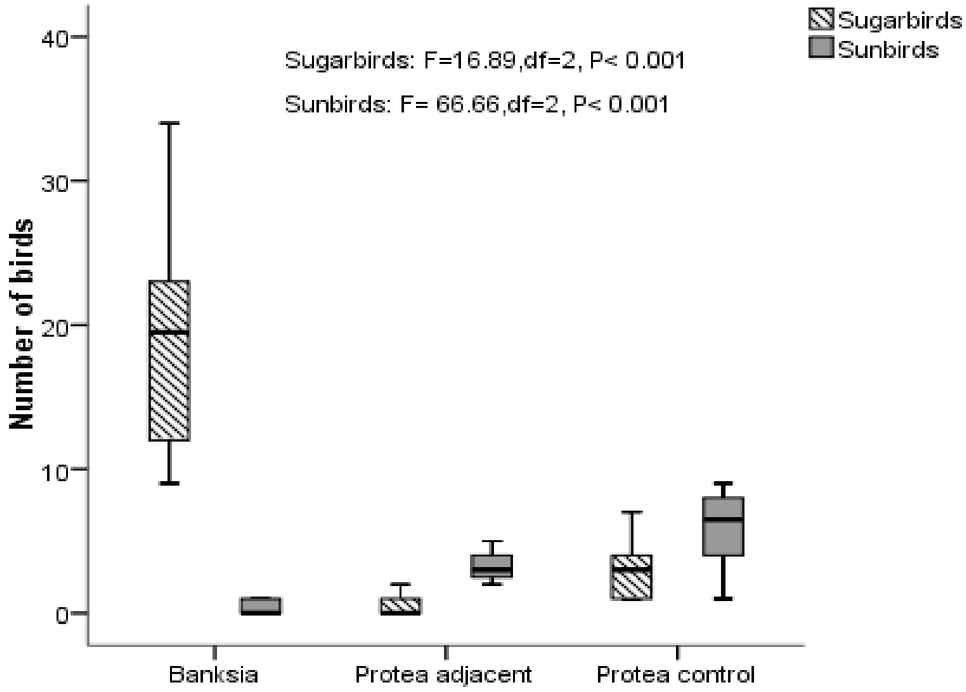

Figure 3.3. Mean number of sugarbirds and sunbirds which contacted floral reproductive parts during the 30-minute flower visitation observation at three sites in the Agulhas National Park, *Banksia* (n=14), *Protea* adjacent (n=15) and *Protea* control (n=12). n is the number of observations that lasted for 30-minutes. The box plot shows the 1st and 3rd quartiles (grey and patterned box), median (dark horizontal bar) and whiskers represent minimum and maximum values.

3.3.3 Nectar properties and availability

Nectar volume per flower ranged from 0.5 to 6 µL with a mean of 3.33 in *B. speciosa*, between 5 to 40 µL with a mean of 18.84 in *P. compacta* adjacent and between 12 to 58 µL with a mean of 29.24 in *P. compacta* control site (Table 3.2). Nectar volume per inflorescence varied among sites with *B. speciosa* having the lowest mean of 52.90 µL (range 12-142) and *P. compacta* in control site having the highest of 441.50 µL (range 286- 684; Table 3.2).

Nectar sucrose concentration per flower ranged between 4-30 %w/w with a mean of 17.39 in *B. speciosa* site, between 11-27 %w/w with a mean of 19.58 in *P. compacta* adjacent and between 9-31 %w/w with a mean of 19.92 for *P. compacta* control site. Total amount of sucrose per hectare was 0.01 g, 2.18 g and 9.55 g for *B. speciosa*, *P. compacta* adjacent and *P. compacta* control, respectively (Table 3.2).

Table 3.2. Means (±SD) volume, concentration and sucrose in nectar of *Banksia speciosa* and *Protea compacta* flowers at three sites in the Agulhas National Park

Site	Nectar volume per flower (µL)	Nectar volume per inflorescence (µL)	Nectar concentration per flower (%w/w)	Sucrose per flower (mg)	Sucrose per hectare (g)
Banksia	3.3±1.61(158)	52.90±37.05(10)	17.39±5.95(158)	0.61±0.37	0.01
Protea adjacent	18.84±7.52(139)	291.00±53.71(9)	19.58±3.39(139)	4.01±1.86	2.18
Protea control	29.24±10.52(151)	441.50±131.05(10)	19.92±3.69(151)	6.45±3.08	9.55

The number of flowers and inflorescences sampled per site are indicated in parentheses.

3.3.4 Breeding system experiments

All *B. speciosa* follicles examined contained two seeds each. The overall seed set per inflorescence (Fig. 3.4) and percentage of inflorescences producing seeds was low for both species (Table 3.3). The number of seeds per infructescence varied among different treatments although no statistically significant difference was detected in all. There was a significant difference in the number of seeds produced per inflorescence between natural, pollinator excluded and pollen addition treatments for *B. speciosa* (Kruskal-Wallis: $X^2 = 15.59$, df=2, $P < 0.001$). Significant differences in the number of seeds produced per inflorescence were also observed between natural and pollinator excluded treatments and pollinator excluded and pollen addition treatments (Mann-Whitney U test: U= 234.500, $P = 0.002$). No significant difference was detected between natural and pollen addition treatments (U= 109.500, $P = 0.204$).

Results in the *Protea* control site followed a similar pattern to the *Banksia* site. There was a significant difference in the number of seeds produced per inflorescence among treatments found in *Protea* control site (Kruskal-Wallis: $X^2 = 14.361$, df=2, $P < 0.001$) with a difference observed between natural and pollinator excluded as well as pollinator excluded and pollen addition treatments (U= 44.000, $P = 0.002$). No significant difference was observed between natural and pollen addition treatments (U= 101.500, $P = 0.646$).

Seed set was low for the pollinator excluded treatment compared to natural and pollen addition treatment except for *Protea* adjacent where no significant difference was detected. There was no significant differences between treatments in the number of seed per inflorescence across sites: natural ($X^2 = 3.687$, df = 2, $P = 0.158$); pollinator excluded ($X^2 = 3.752$, df = 2, $P = 0.153$) and pollen addition ($X^2 = 5.344$, df= 2, $P = 0.069$; Fig. 3.4)

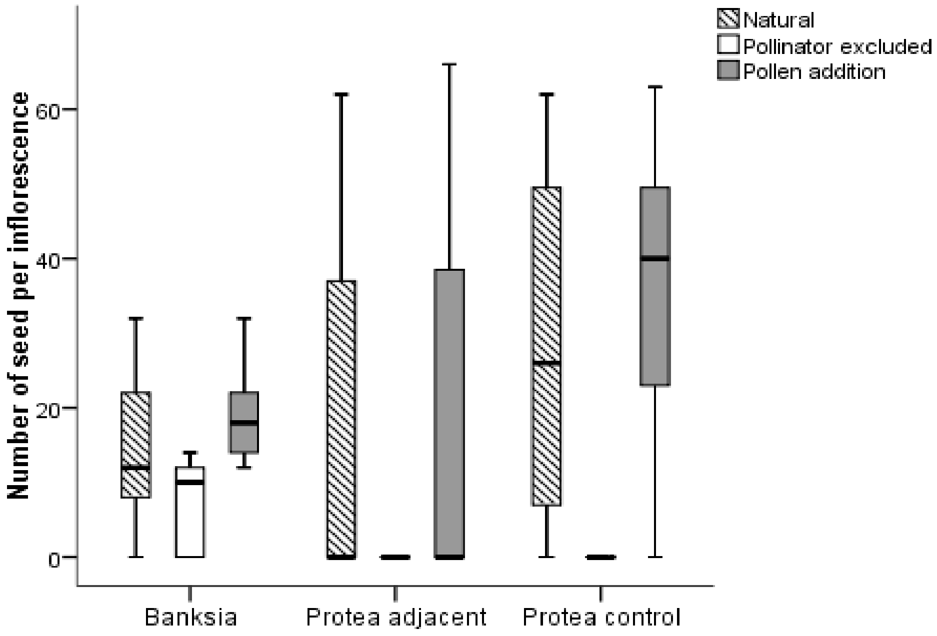

Figure 3.4. A comparison of seed set following three pollination treatments across three sites in the Agulhas National Park.

Table 3.3. Seed production as measured by the number of seeds per inflorescence and percentage (%) of inflorescences which produced seeds for natural, pollinator excluded and pollen addition treatment for *Banksia speciosa* and two *Protea compacta* sites

	Pollination treatment		
Site	Natural	Pollinator excluded	Pollen addition
Banksia	83 (30)	51 (29)	90 (10)
Protea adjacent	47 (15)	20 (15)	47 (15)
Protea control	73 (15)	20 (15)	80 (15)

Percentage (%) of inflorescences which produced seeds are indicated in parentheses.

3.4 DISCUSSION

3.4.1 Bird observation and flower visitations

This study found that *B. speciosa* and *P. compacta* share pollinators to some extent. Their similarity in pollinators suggests they can be potential competitors for pollination services. *Banksia speciosa* can interact with native nectar-feeding birds, particularly Cape sugarbirds, in its introduced range. This agrees with the findings of Moodley et al. (2016) who found introduced Australian *Banksia* and *Hakea* species to have established interactions with native pollinators in the South African Fynbos.

Banksia speciosa attracted fewer sunbirds but a higher abundance of sugarbirds than native *P. compacta*. The significantly lower number of sunbirds observed on *B. speciosa* may be attributable to the low nectar volume secreted by this species. However, the significantly high abundance of sugarbirds in the *Banksia* site, which is nectar-poor, is rather surprising. A high abundance of bird species was expected in both *P. compacta* sites where nectar resources are abundant.

Banksia speciosa displays large attractive inflorescences bearing numerous flowers. However, nectar on most flowers concentrate only on few rows towards the upper flowers (Erckie, pers. obs.) which cannot compensate for the small volume in individual flowers. If each flower were to contain nectar, one could argue that high abundance of sugarbirds is due to the high number of *B. speciosa* flowers contained within an inflorescence. Perhaps the availability of low volume nectar in *B. speciosa* flowers induced a greater number of flower visitations since the low nectar volume may have resulted in more flowers being visited to compensate for the low nectar volume.

High abundance of sugarbirds in such a low-nectar *B. speciosa* stand may be explained by several factors: Firstly, preference of sugarbirds for sucrose-rich nectar provided by *B. speciosa* flowers over hexose-rich nectar provided by *P. compacta* due to the former's high energy content which maximizes sugarbirds' rate of energy intake. This is driven by sugarbirds high energetic needs to maintain their large body size of about 37 g (Tjørve and Scholtz, 2007), relative to sunbirds that have smaller size and lower daily energy requirements. This becomes critical during their breeding time, which occurs concurrently with the flowering season, when more energy is required by males to defend their territory (Mostert et al., 1980). This is in line with both the optimal foraging (Real, 1983) and optimal diet (Schaefer et al., 2003) theories which suggest that pollinators will forage where floral rewards are greater and discriminate on food items based on their energetic value. Secondly, availability of insects may be greater on *B. speciosa* than *P. compacta* site. Sugarbirds supplement their diet with insects found on Proteaceae inflorescences (Mostert et al., 1980). Observations were made during the breeding season when intake of insects is necessary, particularly for females since they need proteins for egg formation (Mostert et al., 1980). In addition to food availability, *B. speciosa* may provide better nesting sites due to its formation of dense thickets. Cape sugarbirds have been reported to prefer more dense vegetation, which increases protection against harsh weather and predators (Burger, 1976).

Contrary to expectations, high numbers of sugarbirds in the *B. speciosa* stand cannot be linked to high nectar volumes since *B. speciosa* flowers secrete relatively low volume of nectar (this study). Hence, other than speculating about sucrose-dominant nectar and the potential availability of nesting sites, reasons for high visitation rates by sugarbirds to *B. speciosa* warrant further study.

Sunbirds responded positively to the high volume of nectar offered by *P. compacta*. This finding is consistent with previous studies, which reported sunbirds preference for flowers with high nectar volume and their avoidance of those with low nectar volume (Gill and Wolf, 1977; Kalinganire et al., 2001). Similar patterns of nectar preference and avoidance have been demonstrated for other pollinators (Pyke, 1980; McDade, 1983; Real, 1983; Waddington and Gottlieb, 1990). This is in accord with the optimal foraging theory, which predicts that pollinators are likely to forage on high rewarding flowers (Real, 1983). Nectar volume rather than concentration have a stronger effect on nectarivorous bird behaviour (Wolf, 1975; Burd, 1995) although Schmid et al. (2015) found the opposite for Fynbos nectarivorous birds.

Sunbirds preference for *P. compacta* flowers can also be attributed to less competition for food and possibly nesting sites. Nectarivorous birds compete for food resources when there is an overlap in their feeding niches (Rebelo et al., 1984). Sugarbirds were reported to often dominate and outcompete sunbirds for nectar resources (Wooller, 1982; Schmid et al., 2016). Concentration of high numbers of sugarbirds on the *B. speciosa* site allowed sunbirds' access to nectar and other resources at the *P. compacta* site where sugarbird abundance was relatively low.

Sunbirds preference for *P. compacta* flowers may also be ascribed to the dilute nectar with less sucrose produced by its flowers in comparison to that of *B. speciosa* that is concentrated with 69 % sugar as sucrose (Nicolson and Van Wyk, 1998). It is thus reasonable to argue that sunbirds discriminated against *B. speciosa* flowers due to the high percentage of sucrose found in their nectar. This concurs with the hypothesis that passerines prefer hexose-rich rather than sucrose-rich nectar because they cannot tolerate sucrose (Baker and Baker, 1983; Del Rio et al., 1992). Conversely, both sugarbirds and sunbirds do not have sucrose aversion as described for passerine species as they have been reported to assimilate both hexose and sucrose with more than 99 % efficiency (Lotz and Nicolson, 1996; Jackson et al., 1998b; Jackson et al., 1998a). They have possibly evolved digestive adaptations to feed on sucrose-rich nectar.

3.4.2 Breeding system experiments

Banksia speciosa is self-compatible and capable of producing seeds to a certain degree through autogamy. However, pollinators are required to transfer pollen between plants to enhance seed set in both study species. The high levels of *B. speciosa* self-compatibility found in the present study are consistent with the findings of Moodley et al. (2016) who reported high levels of self-compatibility in *B. speciosa*.

Banksia species have been reported to have mixed breeding systems. Self-compatibility has been reported for most *Banksia* species (Collins and Rebelo, 1987; Vaughton, 1988; Smith and Gross, 2002). Collins and Rebelo (1987) reported high incidence of self-incompatibility in *Banksia* but this was considered to be an overestimate since they based their data on pollen tube growth rather than on seed set (Steenhuisen and Johnson, 2012b). Findings of the present study along with those of Moodley et al (2016) suggest that *B. speciosa* may expand its population. Self-compatibility is one of the crucial components, although not necessarily a requirement, for successful invasions (Rambuda and Johnson, 2004; Pyšek et al., 2011) since it offers reproductive assurance (Richardson et al, 2000). In contrast, low levels of self-compatibility in *P. compacta* suggests that it relies heavily on pollinators to enhance seed set. This finding is in agreement with Schmid et al. (2015) who reported low self-compatibility in *P. compacta* and four other *Protea* species.

Pollen addition failed to significantly increase the percentage of inflorescences producing seeds and the number of seeds per inflorescence for both species. This suggests that pollen availability is not a limiting factor for seed set in both species. High abundance of pollinators observed, especially sugarbirds, ensured sufficient pollination of these plants. The absence of a significant increase in seed set through pollen addition was also reported in both *Banksia* (Paton and Turner, 1985; Copland, 1987; Goldingay and Whelan, 1990) and *Protea* species (Hargreaves et al., 2004; Steenhuisen and Johnson, 2012a).

Seed set for both *B. speciosa* and *P. compacta* was low relative to the number of flowers contained in an inflorescence. Even if adequate or effective pollination had taken place, relatively few flowers had successfully set seed. Low seed set observed in the present study is a common occurrence within the Proteaceae in both Australia and southern Africa (Rebelo and Rourke, 1985; Collins and Rebelo, 1987; Ayre and Whelan, 1989; Steenhuisen and Johnson, 2012a). *Banksia* species have the lowest fruit set among all genera, with 10 % or less of flowers setting fruits (Lamont et al., 1985; Collins and Spice, 1986; Collins and Rebelo, 1987). *Protea compacta* was reported to have only 8 % of flowers setting seeds (Rebelo and Rourke, 1985).

Low seed set in *Banksia* and *Protea* species can be attributed to factors such as nutrient availability (Collins and Spice, 1986; Paton and Turner, 1985; Wallace and O'Dowd, 1989), seed predation (Zammit and Hood, 1986; Carlson and Holsinger, 2010; Steenhuisen and Johnson, 2012a; Schmid et al., 2015) and spatial availability (Collins and Spice, 1986; Rebelo and Rourke, 1985; Trueman and Wallace, 1999). Not all factors have been investigated in this study but have either alone or in combination produced the observed effect.

3.5 MANAGEMENT IMPLICATIONS

At a local scale, the attraction of sugarbirds from *P. compacta* by *B. speciosa* may have implications on the reproductive success of the former. This is due to the dependence of *Protea* species on sugarbirds for effective pollination (Broekhuysen, 1959; Mostert et al., 1980) rather than on sunbirds (Skead, 1967). There were fewer sugarbirds and also lower seed set in both *P. compacta* stands. The presence of sunbirds in *P. compacta* stands even in high numbers may therefore not contribute to the reproductive success of *P. compacta*.

The flowering of *P. compacta* is restricted to winter (Rebelo, 2001) whilst that of *B. speciosa* occurs throughout the year (George, 1981). By flowering throughout the year, *B. speciosa* may provide a consistent supply of nectar availability to birds when no other nectar sources are available. However, *B. speciosa* population should be removed, given that it attracts sugarbirds away from *P. compacta* and that the former can spread faster with any fire occurrence.

3.6 CONCLUSION

This study served as a unique opportunity to investigate plant-pollinator interactions in a natural system. It revealed that invasive alien *B. speciosa* did not attract higher numbers of nectar-feeding bird species than its native congener *P. compacta*. However, its ability to attract high abundance of sugarbirds can negatively influence the native *P. compacta*'s reproductive success. Factors that appeared to contribute to sunbird preference for *P. compacta* flowers include large volume of hexose-rich nectar and less competition for food resources. It remains to be established if sugarbird preference for *Banksia* flowers can be explained by its sucrose-dominant nectar offering energetic benefits or availability of suitable nesting sites.

The invasive alien *B. speciosa* and native *P. compacta* have different levels of compatibility but both rely on pollinators for seed set. The low seed set observed in the present study can be attributed to factors other than pollen limitation. Future studies assessing impacts of IAP on native plant-pollinator interactions should include pre and post-clearing observations as well as the rate at which nectar in flowers is replenished.

Chapter 4

Invasion risk of *Banksia* species (Proteaceae) in South Africa

Abstract

Risk assessments serve as useful tools to identify potential invasive species and to assist in prioritising management efforts for naturalised species. This study used the weed risk assessment (WRA) tool to evaluate the invasion risk of 14 introduced *Banksia* species presently cultivated as cut-flowers in South Africa. This tool predicts invasiveness of alien plant species based on geographical, ecological and biological traits. Eleven species out of the 14 (79%) assessed for risk were predicted to have high invasion risk and three species (21%) were classified as requiring further evaluation. Traits which contributed to species scoring a high invasion risk included climate suitability, history of invasion elsewhere, prolific seed production and dense thicket formation. None of the *Banksia* species obtained scores indicating a low probability of invasion and other species not yet introduced in the country may need to be evaluated for use in the floriculture industry. *Banksia* populations existing outside cut-flower farms should be prioritised for control and nation-wide eradication. A precautionary approach for using *Banksia* species in floriculture by adopting best management practices such as containment and monitoring to prevent species from escaping from cultivation is emphasised.

Keywords: Banksia; Fynbos Biome; floriculture; invasion risk; Proteaceae; weed risk assessment

4 INTRODUCTION

The introduction of non-native plant species has increased with expanding global travel and trade (Meyerson and Mooney, 2007; Hulme, 2009). This has resulted in an increase of new invasion cases with potentially negative ecological and economic impacts (Richardson and Van Wilgen, 2004; Pimentel et al., 2005). Horticulture has been identified as being amongst the main pathways responsible for the introduction and dissemination of most invasive alien plant (IAP) species worldwide (Reichard and White, 2001; Richardson et al., 2003; Dehnen-Schmutz et al., 2007; Richardson and Rejmánek, 2011).

The response to alien invasion involves various interventions with prevention as the most cost-effective option (Wittenberg and Cock, 2001; Leung et al., 2002; Pyšek and Richardson, 2010). However, legislation prohibiting and or limiting introduced species are either lacking or hardly enforced (Genovesi, 2005; Simberloff et al., 2005). It is usually not considered practical to conduct risk assessments due to the high number of alien candidate species, the low levels of inspection capacity and the expense of individual species risk assessment (Hulme, 2006). Formal risk assessment protocols are, however, being considered to be included in legislations in many parts of the world (McGeoch et al., 2010; Vanderhoeven et al., 2017).

Research focusing on species' biological traits associated with invasiveness (Rejmánek and Richardson, 1996; Kolar and Lodge, 2001) and characteristics of the invaded habitats (Richardson and Pyšek, 2006) have provided some information to predict alien plants invasion success in new regions. Additionally, different tools have been developed to predict the likelihood of alien species establishment in new ranges, such as the Australian weed risk assessment (A-WRA) protocol (Pheloung et al., 1999), climate modelling (Guisan and Thuiller, 2005), expert system (Tucker and Richardson, 1995) and habitat modelling (Zalba et al., 2000). Predictions based on climatic modelling alone do not consider the biology and ecology of species, which are also determinants of species distribution (Trethowan et al., 2011). Other tools, particularly the WRA, and the expert system developed for screening Fynbos species are considered more powerful since they integrate both the biology/ecology of a species and climatic suitability.

The genus *Banksia* L.f. and other members of the Proteaceae have attracted scientific and horticultural attention over several decades. *Banksia* species are grown commercially on flower farms for the international cut-flower markets both in its native range (Burgman, 1982) and in

South Africa (Richardson et al., 1990). There is a high probability that there will be more introductions and cultivation of *Banksia* species in South Africa due to their importance to the floricultural industry (Richardson et al., 1990).

Banksia is an ideal genus for the possible selection of study species for evaluating invasive potential in the Fynbos Biome for several reasons. The genus has many species which are of both conservation concern and economic importance, and has the highest number of species introduced to South Africa out of the Proteaceae family (Moodley et al., 2014). Most of the species are cultivated for the cut-flower industry with several farms situated in the Western Cape Province (Geerts et al., 2013). They are characterised by large and colourful inflorescences which make them desired in the cut-flower industry (George, 1987). Some *Banksia* species have been identified as high risk species and potential invaders of the Fynbos Biome (Richardson et al., 1990). Moreover, the genus has well-documented biological and ecological information in its native range, serving as source of data for WRA responses.

There is limited information on the invasive potential of *Banksia* species in the Fynbos Biome, despite the fact that they are already grown commercially. The few studies conducted so far have either focused on a single species (Honing et al., 1992; Geerts et al., 2013) or relatively few species (Tucker and Richardson, 1995). The only known comprehensive study currently is that of Richardson et al., (1990), although several species present in the country were not included. These will be addressed in the present study. Predicting the invasion risk of all introduced *Banksia* species would be beneficial to prevent their further spread and potential impacts as well as understanding the most important factors affecting the invasion success of *Banksia* species.

This study represents the comprehensive assessment of the invasive potential of *Banksia* species introduced and cultivated for floriculture in South Africa. It aims to evaluate the risk of invasion of *Banksia* species by specifically addressing the following research question: 1) Do the 14 *Banksia* species already present in South Africa pose a low, medium or high risk of invasion?

4.1 MATERIALS AND METHODS

4.1.1 Weed Risk Assessment Protocol

The WRA tool developed in Australia by Pheloung et al., (1999) and modified for South Africa environments was used. This approach has been adapted for use by several different countries (McGregor et al., 2012) and has been applied in both pre-border screening of plants proposed for new introductions and post-border assessment of naturalised species (Pheloung et al., 1999; Hulme 2012). It has been tested in various climates and geographies and has proved accurate to 90-95% in its predictive power (Gordon et al., 2008; Gordon et al., 2012; Kumschick and Richardson, 2013).

This semi-quantitative system has 49 questions comprising of three sections which address biogeographical, biological and undesirable attributes of the species. The system requires a minimum of ten questions from three sections to be addressed in order to give a recommendation. The WRA scores for questions range between 0 and 5. Scores for all questions answered are totalled to give an overall score for each species. The WRA classifies species into three categories by using a pre-defined threshold score. A score below one indicates that the species has a low invasion risk (accept); scores between one and six indicate that further evaluation is required before conclusions can be drawn; a score above six indicates that the species has a high invasion risk and should be rejected.

Responses to the questions were only provided where information was available. Where evidence was lacking, questions were assigned unknown as a response. Lack of information therefore did not bias assessment towards a low risk category. Responses were entered into the WRA model and scores were automatically generated by the system. A published guide for the application of the WRA was followed to answer questions (Gordon et al., 2010). Information on which *Banksia* species are present in South Africa was obtained from the literature (Moodley et al., 2014). Authorities for scientific names are given in Table 4.1 and not repeated elsewhere in the text.

4.1.2 Source of data

Questions for the WRA were answered based on information gathered from primary literature and internet databases such as the Global Compendium of Weeds (http://www.hear.org/gcw/) and Plants for a Future (http://www.pfaf.org/database/plants.php). Personal observations made during the *Banksia* control study (Chapter 5) contributed to WRA responses for selected species.

4.2 DATA ANALYSIS

The Shapiro-Wilk test was performed on all data to test for normality assumptions. The difference between scores partitioned between responses to questions related to agriculture and the environment was tested using an independent T-sample test. The relationship between the total WRA score and the number of questions answered for each species was tested using the Pearson correlation. All statistical analyses were conducted using the software IBM SPSS Statistics, Version 24.

4.3 RESULTS

Out of the 14 *Banksia* species evaluated, 11 (79%) were found to be of high risk and 3 (21%) required further evaluation. Scores for species ranged from 5-17 and no single species scored below one (Table 4.1). Species which obtained scores between 1-6 and required further evaluation were *B. baxteri*, *B. burdettii* and *B. menziesii*. The rest of the species scored above 6 with the highest score obtained by *B. speciosa* (17) followed by *B. serrata* (15) and *B. integrifolia* (14).

An average of 33 questions (range 29-39) out of 49 questions were answered (Appendices 3-16). This was based on the amount of information available for each species. Factors and traits which contributed to the high risk recommendation included climate suitability and invasion elsewhere, dense thicket formation, prolific seed production and wind dispersal mechanism. There was no significant relationship between WRA score and the number of questions answered ($r = 0.049$, $P = 0.869$) (Fig. 4.1). The environmental sector of the questionnaire was more affected with a mean score of 11.79±3.77 than the agricultural sector with a mean score of 6.86±2.38. The difference was highly significant ($t_{(26)} = -4.139$, $P < 0.001$).

Table 4.1. Invasion risk of 14 *Banksia* species introduced to South Africa evaluated using the weed risk assessment (Pheulong et al., 1999) modified for use in South Africa

Species name	Score	Invasion risk level
Banksia baxteri R.Br	5	evaluate further
Banksia burdettii Baker f.	5	evaluate further
Banksia coccinea R.Br.	8	high
Banksia ericifolia L.f.	13	high
Banksia formosa (R.Br.) A.R.Mast & K.R. Thiele	9	high
Banksia hookeriana Meisn.	12	high
Banksia integrifolia L.f.	14	high
Banksia menziesii R.Br.	5	evaluate further
Banksia prionotes Lindl.	12	high
Banksia quercifolia R.Br.	8	high
Banksia serrata L.f.	15	high
Banksia speciosa R.Br.	17	high
Banksia spinulosa Sm.	8	high
Banksia sphaerocarpa R.Br.	8	high

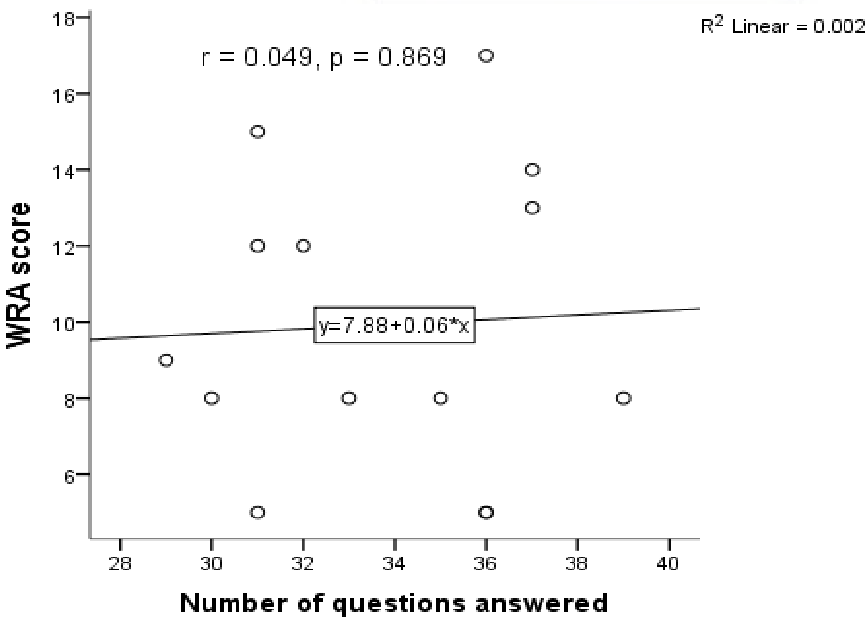

Figure 4.1. Relationship between the WRA score obtained and the number of questions answered for *Banksia* species using the Australian weed risk assessment (Pheulong et al., 1999) modified for use in South Africa.

4.4 DISCUSSION

This study revealed that most *Banksia* species have a high invasive risk and would have been rejected if screened prior to their introduction to South Africa. This suggests that most of the environmental conditions and attributes a species needs to possess in order to exploit invasion windows are met. Species which possess life-history attributes that lead to establishment and invasion success resulted in higher scores than those less associated with invasion. A high number of questions were answered for each species, which resulted in improving the reliability of the recommendations (Pheulong et al., 1999). The weak correlation found between WRA score and the number of questions answered suggests that risk prediction does not depend on the amount of information available for each species. This is consistent with findings of previous studies that investigated the relationship between WRA scores and number of questions answered (Daehler and Carino, 2000; Daehler et al., 2004; Gordon et al., 2012).

Results of high invasion risk of most *Banksia* species in South Africa are consistent with findings of previous studies for *B. ericifolia* (Geerts et al., 2013; Honig et al., 1992); *B. hookeriana*, *B. prionotes*, *B. coccinea*, *B. quercifolia* (Richardson et al., 1990; Tucker and Richardson 1995) and *B. speciosa* (Tshilingalinga, 2014). These species were reported to have high invasive risk due to taller heights, large seed banks, short juvenile period and formation of dense thickets. The WRA has also addressed such traits, except height, as determinants of invasiveness.

However, findings of the present study did not agree with previous studies on the low invasive risk of *B. menziesii* since it obtained a score above one. Previous studies have attributed its low invasive risk in the Fynbos Biome to low production of seeds and long juvenile period (Richardson et al., 1990; Tucker and Richardson, 1995). Main questions which contributed to *B. menziesii* attaining a further evaluation category in the present study are whether the species is a congeneric weed or has naturalised beyond its native range. Being assigned a further evaluation score indicates that it is subjected to undergo secondary screening.

The secondary screening consists of a short subset of WRA questions in the form of a short decision tree (Daehler et al 2004; Křivánek and Pyšek, 2006; Koop et al., 2012). It uses key questions that are the main predictor of invasiveness and its results lead to the same outcome as in WRA i.e. accept, reject and further evaluation (Gordon et al., 2008; Koop et al., 2012). While some authors claimed that species in the further evaluation category may represent minor invaders (Koop et al., 2012), such preliminary conclusions cannot be made for the three *Banksia* species placed in the further evaluation category in the present study. There is presently no secondary screening tool developed or modified for use in South Africa which could be used to determine the fate of the species placed in the further evaluation category. Environmental Impact Classification for Alien Taxa (EICAT) can however be used to quantify and categorise their impacts. EICAT is another scoring system which predicts the magnitude of impacts into five levels of impact ranging from minimal to major concerns (Blackburn et al., 2014; Hawkins et al., 2015).

A striking finding is that most *Banksia* species which obtained high scores are resistant to the *Phytophora cinnamomi* root-fungus, a major disease in the country (McCredie et al., 1985). In addition, *P. cinnamomi* susceptibility was also found to be a clear mechanism of invasion success in Proteaceae (Moodley et al., 2013). This includes species such as *B. ericifolia*, *B. integrifolia* and *B. serrata*. The exception is *B. speciosa* which received the highest score but is highly susceptible to the disease. Species that are resistant to this disease should be considered priorities for control and or eradication since resistance may increase their chances of becoming invasive. The population growth of species that are highly susceptible such as *B. hookeriana*, *B. coccinea*, *B. prionotes*, *B. baxteri* and *B. sphaerocarpa* may be regulated by the disease albeit they still have a greater chance of naturalising (Richardson et al., 1990; Moodley et al., 2014). *Banksia menziesii* and *B. burdettii* are amongst the three that were placed on the further evaluation category and are moderately susceptible to the disease.

The WRA is designed primarily as a precautionary method to prevent entry of potential invaders (Pheloung et al., 1999). However, it can also be used as a management tool for species that entered new areas without being screened (Hulme, 2012; Kumschick and Richardson, 2013). Given that *Banksia* species are already present in South Africa and half of them are naturalised and or invasive, this risk assessment can be used as a management tool for prioritising control efforts against *Banksia* species by ranking them according to their level of risk. It will further inform and guide policy with regards to legal listing of the species. None of the *Banksia* species is listed under the current NEM:BA legislation (NEM:BA 2014).

There is accumulating evidence suggesting that prediction of species' invasive potential remains a difficult task (Williamson and Fitter, 1996; Hulme et al., 2013; Ricciardi et al., 2013). This is because it is not only species' life-history traits but also the vulnerability or features of the community which determine the likelihood of a species becoming invasive (Richardson and Pysek, 2006; Ricciardi et al., 2013). The predictive power of any risk assessment must therefore be very high to identify potential invaders reliably (Smith et al., 1999). The WRA screening system used in this study has a high accuracy and predictive power of about 90-95% (Pheulong et al., 1999; Gordon et al., 2008). It further does not only consider the biology of the species, but also its invasion history elsewhere and its climatic suitability. However, unless environmental conditions and all factors that determine invasion success are known, risk assessments cannot accurately predict the invasion success of alien plant species (Moodley et al., 2014). It should therefore be noted that WRA alone is not sufficient to predict the likelihood of a plant species becoming invasive. But the high level of invasion risk revealed in this study sufficiently justifies a focused management response of *Banksia* species in South Africa.

4.5 CONCLUSION AND RECOMMENDATIONS

This study revealed that most *Banksia* species have a high risk of invasion in the South African Fynbos with potential to pose damage to native vegetation. Three species with intermediate risk scores placed on the further evaluation category are subjected to further assessment. This study did not find evidence of any *Banksia* species with a low probability of invasion that could be prioritised by the floriculture industry.

Given the growing interest and commercial demand for these species, good management practices such as containment and monitoring of populations in cut-flower farms is required to minimize invasion risk. Populations of strongly serotinous species should be protected from fire and efforts should be made to remove all abandoned plantations. Existing populations of *Banksia* species outside cut-flower farms should be prioritised for control and eradication. Conducting risk assessment for other *Banksia* species which are not yet introduced to South Africa should be an avenue for further research. This should include predicting their potential distribution range so that areas susceptible to invasion countrywide can be identified. This will assist in developing prohibited and permitted list of species to restrict further introduction of *Banksia* species with high invasion risk.

Chapter 5

Chemical control of *Banksia integrifolia and Banksia serrata* (Proteaceae) in South Africa: A preliminary assessment of efficacy

Abstract

The chemical control of two resprouting *Banksia* species is reported for the first time in South Africa. Field experiments were conducted to evaluate the efficacy of various herbicides for the control of resprouting *Banksia integrifolia* and *B. serrata*. The herbicides metsulfuron, imazapyr and a picloram/triclopyr mix applied at different concentrations to cut-stumps were tested. Evaluation was done six months after cut-stump treatment application. Response variables measured were percentage of plants resprouting and not resprouting, average resprouting height and resprouting vigour. The triclopyr/picloram mix provided the best results with 100% stump mortality followed by imazapyr at 5% concentration with a 91% stump mortality. Metsulfuron at 1% concentration was the least effective treatment providing 45% stump mortality. A trend of effective control with increasing herbicide concentration was observed on stumps treated with imazapyr and metsulfuron. Average resprouting height ranged from 2 to 18 cm. Resprouting vigour varied from poor to good with some resprouts displaying some form of deformation. Results suggest that concentrations above 5% for imazapyr and metsulfuron is required for effective control. A full-scale trial is needed to determine the most effective herbicide with minimum dosage to minimize negative effects on non-target species before the triclopyr/picloram mix is registered as an effective herbicide for the control of *B. integrifolia* and *B. serrata.* Given the current spatial extent of infestation and biological features of the two species, eradication is considered feasible and should form part of management objectives. The invasive *Banksia* species should become a target for eradication under NEM:BA regulations.

Keywords: *Banksia integrifolia*; *Banksia serrata*; cut-stump; eradication target; herbicide treatment; triclopyr/picloram mix

5 INTRODUCTION

The invasion of natural and semi-natural areas by invasive alien plants (IAP) is a worldwide problem (Wilcove et al., 1998; Levine et al., 2003). These invasions can have detrimental impacts on native species such as altering community structure (Richardson et al., 1989; Hejda et al., 2009) and disrupting ecosystem processes (Vitousek, 1990; Chittka and Schürkens, 2001; Heneghan et al., 2006). Their control and management is essential to mitigate their negative impacts (Holmes and Richardson, 1999; Van Wilgen et al., 2000; Webster et al., 2007).

Approaches to mitigate threats caused by plant invasions involves prevention, eradication, containment and control (Simberloff, 2003; Pyšek and Richardson, 2010). Prevention is the most cost-effective option (Wittenberg and Cock, 2001; Leung et al., 2002), but legislation limiting the introduction of alien species are rarely enforced (Genovesi, 2005; Simberloff et al., 2005). Eradication is considered the second best option once prevention fails whereas control should be used once eradication has failed or is not considered feasible (Myers et al., 2000; Simberloff, 2003; Simberloff et al., 2013).

Any emerging IAP species detected should be controlled with any effective means (Olckers, 2004; Mgidi et al., 2007; Simberloff, 2009). This should be undertaken even if impacts have not been quantified (Wittenberg and Cock, 2001). Although alien plants may have minor or no noticeable ecosystem impacts at low densities (Blossey et al., 2001), justification of their control is based on low costs of control, its effectiveness (Rejmánek and Pitcairn, 2002; Simberloff, 2009) and concerns over potential negative impacts of a species (Blossey et al., 2001). Populations which are left unmanaged may expand their ranges with the potential to cause harmful effects and displace native species (Myers and Bazely, 2003; Pluess et al., 2012). There are different control methods available to deal with plant invasions including mechanical, chemical and biocontrol, used alone or in combination (Hobbs and Humphries, 1995; Tu et al., 2001). The choice of which method to use depends on the size of the infestation, the biology of the species and resource availability (Van Wilgen et al., 2000).

The genus *Banksia* L.f. contains about 170 species (Mast and Thiele, 2007), most of which are endemic to Australia (George, 1981). *Banksia* species and other taxa from fire-prone environments are characterised by life history traits which allow them to be classified as either reseeders or resprouters based on their response to fire and other forms of disturbance which destroy above-ground parts (Bell, 2001; Bond and Midgley, 2001; Lamont and Wiens, 2003).

Reseeders or non-sprouters refer to plants that are killed by fire and entirely depend on seed for regeneration. Resprouters or fire-tolerant species refers to those with the ability to survive fire and reproduce vegetatively (Gill, 1981). Resprouters possess a thick bark (1-3 cm) or lignotuber from which new shoots emerge (George, 1981; James, 1984; George, 1987). The thick bark protects buds and vascular systems from the heat of a fire and enable resprouting (George, 1987; Lawes et al., 2011).

Since 1970, fourteen *Banksia* species have been introduced to South Africa. This is the earliest date of occurrence, specifically for *B. ericifolia* L.f. which was introduced for floriculture (Geerts et al., 2013). Five species (*B. ericifolia*, *B. formosa* R.Br., *B. integrifolia* L.f., *B. serrata* L.f. and *B. speciosa* R.Br.) have been reported to be invasive or naturalised at different sites in the Western Cape Province (Moodley et al, 2014). Thirty six percent (5 out of 14) of *Banksia* species introduced are resprouters (*B. integrifolia*, *B. serrata*, *B. menziesii* R.Br., *B. sphaerocarpa* R.Br. var. *sphaerocarpa* and *B. spinulosa* Sm. var. *spinulosa*). The remaining 64% (9) are reseeders and exhibit different degrees of serotiny.

Banksia species have not been targeted for control in South Africa, primarily due to their recent introduction and commercial value (Ernita van Wyk, pers. comm.). However, studies have demonstrated that most have a high risk of becoming major invaders in the Fynbos Biome (Richardson et al., 1990; Chapter 4). Invasion by *B. ericifolia* and *B. speciosa* have already demonstrated impacts with a potential to disrupt networks of indigenous pollination (Geerts et al., 2013; Chapter 3). It has further been described that their demand in floriculture is likely to increase (Honig et al., 1992; Moodley et al., 2013). However, some of these species should be listed in the South African legislation which governs the control and management of invasive alien species (IAS) (NEM:BA, 2014). This makes it essential to develop effective control methods to avoid repetition of demonstrated impacts of *Banksia* invasion.

Registration of herbicides for use on specific invader species is compulsory in South Africa (Erasmus, 1988). Herbicides for use against *Banksia* species are yet to be evaluated and or registered and little information is available to guide the control of resprouting *Banksia* species (Ernita van Wyk, pers. comm.). Hand cutting alone as a mechanical method of control for species with vigorous resprouting ability will not be effective in the long-term, indicating that chemical treatment is required for their effective control (Cherry et al., 2008; Lemola, 2014). Moreover, using mechanical control only as a control method is unfeasible as it is timeconsuming and expensive because it involves repeated follow-ups (Hobbs and Humphries, 1995; Van Wilgen et al., 2000; Simberloff, 2003). The current infestation size of most *Banksia* species is too small to consider using biocontrol which is ideal for large-scale invasions. Furthermore, despite biocontrol agent releases in South Africa having a good track record of low to no risk (Lotter and Hoffmann, 1998; Esler et al., 2010), biocontrol is considered to be a high risk option as it cannot be reversed and should be considered a last resort (Simberloff, 2009).

The purpose of this study was to determine the efficacy of different herbicides at varying concentrations as a chemical control method for resprouting *B. integrifolia* and *B. serrata* in the Western Cape Province, South Africa. Specific questions addressed were: 1) How would *B. integrifolia* and *B. serrata* respond to the application of herbicide treatments following cut-stump treatment? 2) Which herbicides and at what concentrations will be most effective in controlling *B. integrifolia* and *B. serrata* infestations? 3) Is eradication of these species in South Africa feasible at the observed population densities and extent?

5.1 MATERIALS AND METHODS

5.1.1 Study species

Banksia integrifolia is a multiple-stemmed tree that grows up to 25 m tall (George, 1981; George, 1987). Three subspecies are recognised (George, 1981) but only one has been introduced to South Africa (Moodley et al., 2013). *Banksia integrifolia* is non-serotinous in its native range (George, 1981), although Moodley et al. (2014) reported it to be strongly serotinous in the South African Fynbos. George (1987) described it as a fire-tolerant tree that resprouts from epicormic shoots but did not indicate whether a lignotuber is present. It possesses a thick bark of < 2 cm that protects it from hot fires and enables it to resprout (George, 1981). It flowers from 4 to 6 years with flowering occurring from January to July (George, 1987; Fig. 5.1A).

Banksia serrata is a shrub or single-stemmed tree that grows up to 16 m tall (George, 1981; George, 1987). It is fire-tolerant and resprouts from epicormic shoots (George, 1987). It is weakly serotinous in its native range (George, 1981) but Moodley et al. (2014) reported it to be strongly serotinous in the South African Fynbos. There is uncertainty as to whether or not *B. serrata* possesses a lignotuber. Some reported its presence (Bradstock and Myerscough, 1988; Whelan et al., 1998; Renshaw, 2005) while others questioned it (Taylor and Hopper, 1988). George (1987) did not indicate whether or not a lignotuber occurs. It possesses a thick bark of < 3 cm which protects it from hot fires and enables it to resprout (George, 1981). It is reported to flower from January to June (George, 1987; Fig. 5.1B).

Figure 5.1. Study species in flower: (A) *Banksia integrifolia* (B) *Banksia serrata*; Herbicide treatment of *Banksia integrifolia* at Pringle Bay: (C) application of herbicide to a cut-stump with a paint-brush; (D) tag nailed onto the treated cut-stump. Photographs: A, D- Ernita van Wyk; B- google image; C- Mark Mauldin.

5.1.2 Study area

The study was conducted at Pringle Bay (34°21'105 S; 18°49'064 E) and Betty's Bay (34°21'037 S; 18°55'169 E) in the Western Cape Province, South Africa. The study area has a Mediterranean-type climate with hot dry summers and cold wet winters. Average annual rainfall is about 300 mm of which most falls during winter months (Lamprecht et al., 2006). The invasive *B. integrifolia* population at Pringle Bay is one of the nine populations present in South Africa (Moodley et al., 2014). This population emerged from a single tree planted in 1980 (Moodley et al., 2014) and currently cover approximately four hectares. The naturalised *B. serrata* population at Betty's Bay emerged out of nine individuals planted in the natural Fynbos. A fire around 2000 has caused the population to spread to an area of about 0.24 hectares (Moodley et al., 2014).

5.1.3 Experimental layout and treatments

Seven plots of varying sizes were established within the Pringle Bay *B. integrifolia* population (Fig. 5.2). Due to the spatial position of the *B. integrifolia* patches, plot sizes were not identical within the invaded site. Each treatment plot contained between 12 - 26 plants of *B. integrifolia*. The choice of herbicide active ingredients to test was informed by consultation with a licenced herbicide expert who is familiar with the genus *Banksia* and conditions in the Western Cape (Graham Harding, pers. comm.).

The seven herbicide treatments selected comprised of the following active ingredients: Metsulfuron (methyl2-[(4-methoxy-6-methyl-1,3,5-triazin-2-yl)] carbamoylfamoyl/benzoate) at three concentrations: 1%, 3% and 5%; Imazapyr ((±)-2-[4,5-dihydro-4-methyl-4-1(1-methylethyl)-5-oxo-1-H-imidazol-2-yl]-3-pyridinecarboxylic acid) at three concentrations: 1%, 3% and 5%, and triclopyr ([(3,5,6-trichloro-2pyridinyl)-oxy]aceticacid)+picloram (4-amino-3,5,6-trichloro-2-pydinecarboxylic acid) herbicide gel formulation at one dosage. Since this experiment formed part of an eradication attempt, no plot was kept herbicide free to serve as a control.

Mixing and application of herbicides started from lowest to highest concentration of the same active ingredient within each treatment. A dye was added to herbicides to improve visibility on stumps and avoid repeated applications. Clean equipment was used for each treatment. Herbicide application was made on dry weather days (without rain) and was performed in spring when herbicide function is maximized (MacDonald et al., 2013).

Plants were cut close to the base at a height of 5-10 cm with a chainsaw. Herbicides were applied immediately to fresh cut stems with a paint brush (Fig. 5.1C). Immediate application is necessary to prevent the stump layer from getting dry and inhibiting herbicide absorption and translocation throughout the root system (Cuddihy et al., 1991; Cherry et al., 2008; MacDonald et al., 2013). Caution was exercised to cause any spillage or run-off from the stump onto the ground to prevent harming non-target species and ground contamination. Stumps in each plot were marked with a unique colour tag and identity number and each tag was nailed onto the cut-stump (Fig. 5.1D). Slashes were stacked at one place for later removal and to create space for native vegetation recovery.

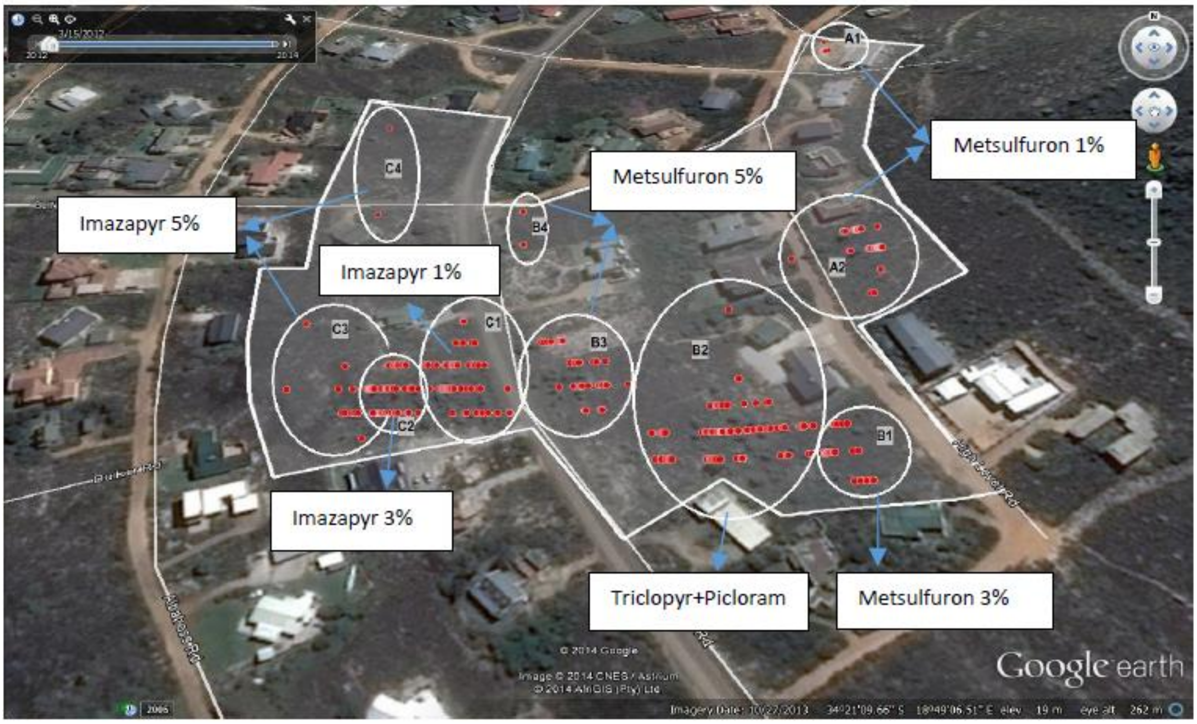

Figure 5.2. Schematic representation of the experimental layout of *Banksia integrifolia* showing seven treatment plots at Pringle Bay, Western Cape Province, South Africa.

The population of *B. serrata* at Betty's Bay comprised of relatively few individuals. This population was too small to conduct an experiment similar to that of *B. integrifolia*. Methods as described above were followed but all plants were treated with triclopyr/picloram mix, due to the ease of application of the gel formulation and reduced risk of contaminating non-target species.

5.1.4 Herbicide efficacy evaluation

Evaluation of the response of plants to herbicide treatments was conducted six months after herbicide application. To determine treatment effects, the following parameters were recorded: number of plants resprouting, average resprouting height and resprouting vigour. The height of resprouts was measured with a tape measure. Resprout vigour was visually estimated as excellent if it appeared exceptionally healthy; good if appeared normal; fair if there were signs of discolouration or deformity and poor if such signs appeared more pronounced (Santos et al., 1991). Plants were considered dead if no resprouting occurred and no other living tissues were visible at the time of evaluation.

5.1.5 Effects of herbicides on non-target native species

Native plants were not scientifically evaluated for effects by herbicide treatments but a rapid visual assessment was made with nearby plants within a 2 m radius of each herbicidal treatment.

5.2 RESULTS

A total of 143 *B. integrifolia* plants were treated with different herbicides. Different effective control potential was observed with regards to both type and concentration of herbicides (Table 5.1). Fifty five percent of stumps treated with metsulfuron 1% resprouted with an average height of 18 cm. Resprouting vigour appeared poor with most new growth slightly yellow and others showed deformed growth. This treatment had the highest mean resprouting height and the lowest stump mortality (Fig. 5.3 and 5.4). For the metsulfuron 3% treatment, 39% of stumps treated resprouted with 14 cm average height. Resprouting vigour was rated fair as some of the coppiced stumps showed signs of deformation in the form of leaves curling. Metsulfuron 5% treatment data could not be obtained since tags had disappeared by the time of evaluation.

Imazapyr 1% treatment resulted in 33% of stumps to resprout with 6 cm average height. Resprouting vigour was poor, most coppiced stems showed broccoli growth and yellowing of leaf margins. Imazapyr 3% treatment resulted in 27% of treated stumps resprouting with an average height of 3 cm. Resprouting vigour was poor as more than half of coppiced stems showed broccoli and compact growth. Only few plants had no signs of deformation. Nine percent of stumps treated with imazapyr 5% treatment coppiced with an average height of 2 cm. Resprouting vigour was good as no signs of deformation was observed. This treatment had the lowest mean resprout height and yielded the lowest number of resprouting stumps.

The triclopyr/picloram mix treatment resulted in the death of all 26 stumps since none resprouted (Table 5.1). Similar results were obtained for *B. serrata* plants, which were all treated with the triclopyr/picloram mix (data not shown).

Height of resprouts varied and followed a similar trend with herbicide effectiveness as indicated by stump mortality. Resprouting heights treated with imazapyr 5% concentration, the second most effective herbicide, were shorter and the least effective treatment, metsulfuron 1%, had the tallest height. There was a significant difference observed in mean resprouting heights between treatments as determined by one-way ANOVA: $F_{4,31} = 7.412$, $P < 0.001$; Fig.

5.4). There was a significant difference between metsulfuron 1% and all three imazapyr treatments; imazapryr 1% and metsulfuron1%; imazapyr 3% and metsulfuron 1% as well as imazapyr 5% and metsulfuron 1%. The rest treatments showed no significant difference.

Table 5.1. Herbicide treatments and their efficacy on *Banksia integrifolia* cut-stumps during a field experiment at Pringle Bay, Western Cape Province, South Africa

Herbicide	Concentration %	Treated stumps	Sprouting stumps	Non-sprouting stumps	Mean sprout height (cm)	Resprouting vigour
Metsulfuron	1	22	12	10	18	Poor
	3	23	9	14	14	Fair
	5	22	-	-	-	-
Imazapyr	1	26	9	17	7	Poor
	3	11	3	8	3	Poor
	5	11	1	10	2	Good
Tri/Pic mix		26	0	26	0	0

Tri/Pic= Triclopyr/Picloram mix; (-) represents missing data.

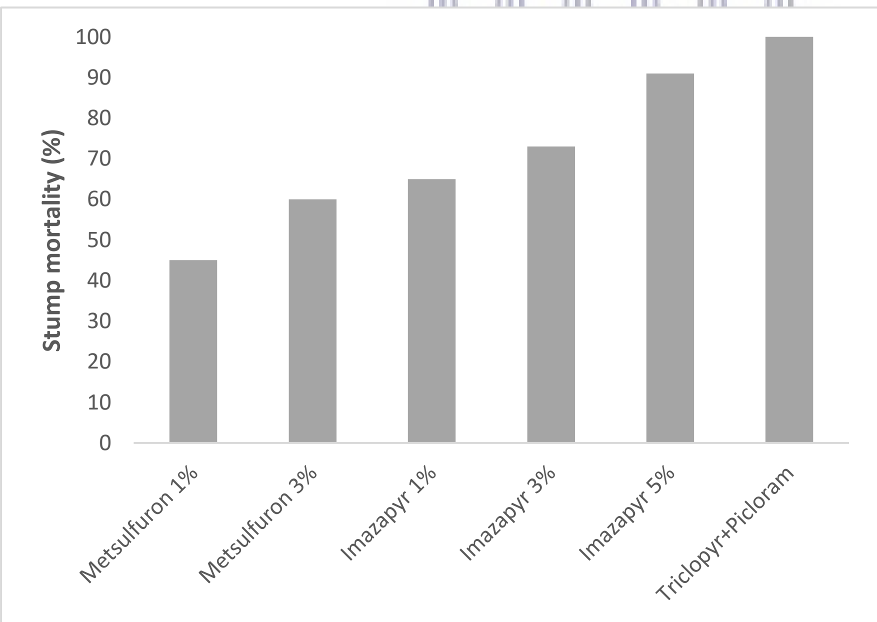

Figure 5.3. Stump mortality percentage of *Banksia integrifolia* with different herbicides evaluated 6 months post-treatment at Pringle Bay, Western Cape Province, South Africa.

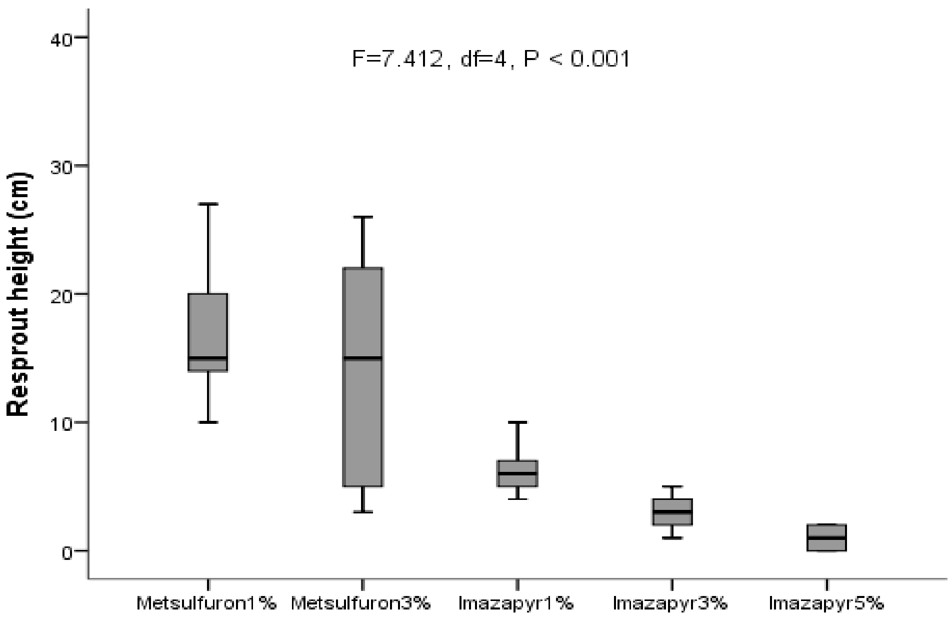

Figure 5.4. Mean resprouting height of *Banksia integrifolia* treatment with different herbicides assessed 6 months post-treatment at Pringle Bay, Western Cape Province, South Africa. Metsulfuron 1% (n=12), Metsulfuron 3% (n=10), Imazapyr 1% (n=9), Imazapyr 3% (n=3) and Imazapyr 5% (n=2). The box plot shows the 1st and 3rd quartiles (grey box), median (dark horizontal bar) and whiskers represent minimum and maximum values.

5.3 DISCUSSION

5.3.1 Herbicide treatment efficacy

The response of *B. integrifolia* to different herbicide treatments tested varied with regards to stump mortality percentage, resprouting height and resprouting vigour. The variation observed among treatments suggested a difference in herbicide absorption and translocation as well as in susceptibility to herbicides.

The triclopyr/picloram mix was the most effective for the control of both *B. integrifolia* and *B. serrata* with 100% stump mortality (this was the only treatment tested for *B. serrata*). This suggests that it penetrated readily and maximized absorption and translocation, resulting in a greater degree of control than other herbicides. Equally, it could mean that it is more resistant to breakdown within the plant. This formulation requires only a small amount of herbicide to be effective making it the most cost-effective method of application (Tu et al., 2001). Moreover, this herbicide mixture is considered safe for the environment and poses minimal risks to non-target species (Cuddihy et al., 1991). Triclopyr as an active ingredient degrades quickly in the soil, reducing its potential to reach groundwater (Bovey, 1995; Piirto et al., 1996), whereas the thick gel form of the mixture minimises spillage (Cherry et al., 2008).

Both triclopyr and picloram belong to the pyridine family and have similar modes of action with which they kill susceptible plants (Retzinger Jr and Mallory-Smith, 1997). They are selective herbicides which mimic the plant growth hormone auxin, resulting in unregulated growth and ultimately the death of the susceptible plant. It disrupts the balance of plant growth hormones upon absorption (Gunsolus and Curran, 2007).

The second most effective treatment was imazapyr at 5% concentration, which provided 91% stump mortality. The fact that only one treated stump resprouted suggests that such concentrations may be effective at controlling the target species. There could be several reasons for one treated stump to resprout: Firstly, the cambium layer of the stump may have not been covered with adequate amounts of herbicide. Secondly, the stump may not have been treated immediately with the herbicide resulting in the stump layer drying, thereby preventing absorption and translocation of the herbicide to the root system. Good coverage of the cambium with enough herbicide and its application within an appropriate time of 5-10 minutes can positively influence the response of plants to herbicide treatment (Bovey, 1995). Thirdly, the stump could have been cut higher than recommended, resulting in poor translocation of herbicides. Stumps that are cut high are more likely to resprout than those cut nearer ground level (Leonard and Murphy, 1965).

Mortality of stumps treated with the concentration of 1% and 3% imazapyr was between 65% and 73%. However, given the poor resprouting vigour with abnormalities and deformations observed on resprouts, their survival is doubtful. The eventual death of all resprouts with poor vigour was observed by Santos et al. (1991) during the control of raspberry with imazapyr cut-stump treatment in Hawaii. Imazapyr is a non-selective herbicide which controls plant growth by blocking the enzyme acetolase synthase (ALS) responsible for protein formation and cell growth (Shaner et al., 1984; Brown, 1990; Stidham, 1991; Retzinger Jr and Mallory-Smith, 1997). Unlike other herbicides, imazapyr is a slow acting herbicide and treated plants tend to die slowly (Bovey, 1995). Its persistence in the soil differs with both soil properties and moisture content (Bovey, 1995; Terry et al., 1996).

Metsulfuron 1% was found to be the least effective herbicide treatment, providing minimum stump mortality (45%) of *B. integrifolia*. Decrease in resprouting percentage with increased concentration suggests that zero percent resprouting can be achieved with higher concentrations. The low efficacy provided by metsulfuron 1% may be because such low concentration resulted in poor absorption and translocation to the root system. It may also be due to metsulfuron often being used as a foliar spray, for which it was designed, rather than a cut-stump treatment (MacDonald et al., 2013). Its inclusion in this trial was based on reports of its accidental drift over and effect on *Banksia* species while treating *Chrysantheimodes monilifera* subsp. *rotundata* (DC.) in Australia (Cherry et al., 2008). Metsulfuron kills susceptible plants in a similar manner as imazapyr by inhibiting the enzyme responsible for producing essential amino acids (Hay, 1990; Wepplo, 1990; Retzinger Jr and Mallory-Smith, 1997; MacDonald et al., 2013). Similar to triclopyr, metsulfuron has a short persistence in the soil and does not bio-accumulate in non-target species (Brown, 1990).

Rate response was evident on plants treated with metsulfuron and imazapyr with resprouting height decreasing with increasing concentration and vice versa. Results suggest that both imazapyr and metsulfuron can provide effective results at controlling *B. integrifolia* when applied at a concentration greater than 5%. This may be because increasing concentrations enhance maximum absorption and translocation to the roots system. Further research should test these herbicides at higher concentrations to determine if greater control can be achieved and test their effects on non-target native species.

Herbicides used in this study are generally used in the control of woody species in both natural and agricultural systems (Bovey, 1995; Macdonald et al., 2013). They are from three different herbicide families and represent two different modes of actions. Given that only chemicals from three families were used in this study, other herbicides from other families with different modes of actions should also be tested. Practitioners of weed control encourage the use of herbicides with various modes of action to increase success rates, especially in trial experiments (Harker and O'Donovan, 2013).

5.3.2 Effects of herbicides on non-target native species

Although effects of tested herbicides on native nearby plant species were not scientifically evaluated, no negative effects were observed. However, this does not imply that these herbicides have no negative effects. This could be attributed to the fact that the cut-stump method used during this study has a low probability of contaminating the environment and affecting non-target species because it is target-specific (Tu et al., 2001).

5.4 MANAGEMENT

5.4.1 Eradication of resprouting *Banksia* species

Eradication of *B. integrifolia* and *B. serrata* is considered feasible since populations are not yet widespread. Both populations are less than five hectares in size and there are success stories associated with small infestations which are less than 10 hectares compared to widespread populations (Simberloff, 2001; Rejmánek and Pitcairn, 2002). Small infestations are easier to eradicate because control measures can be applied thoroughly and are cheaper (Rejmánek and Pitcairn, 2002; Pluess et al., 2012). Eradication failure has mainly been reported for large infestations which are more than 1000 hectares in size (Rejmánek and Pitcairn, 2002).

The population of *B. integrifolia* at Pringle Bay is currently the only known invasive South African population (Moodley et al., 2014). The remaining eight populations in the Western Cape Province, some of which are on flower farms, are young and not yet spreading (Moodley et al., 2014). *Banksia integrifolia* increases its population with long residence time rather than by fire (Moodley et al., 2014) and fire occurrence in the Fynbos might not trigger its spread. The eradication of the existing single population of *B. serrata* should be considered before any fires break out, since its spread is triggered by fire (Moodley et al., 2014).

The fecundity life-form hypothesis predicts that resprouters have low seed production and seedling recruitment (Bell, 2001; Lamont and Wiens, 2003). They are therefore expected to take longer to spread since they require more time to increase their population. They are further likely to be self-incompatible (Lamont et al., 1998), although self-incompatibility is common in the genus *Banksia* (Lamont et al., 1985; Collins and Rebelo, 1987). Self-incompatibility can partly contribute to limiting a successful invasion since it reduces reproductive assurance (Richardson et al., 2000; Pyšek et al., 2011). This suggests that *B. integrifolia* and *B. serrata* spread can be impeded with the potential to increase eradication success (Panetta, 2009). Moreover, the feasibility and speed with which eradication can be achieved is increased since no persistent seed bank is formed (Simberloff, 2003; Panetta, 2009).

Resprouting *Banksia* species tend to live longer, for about 300-500 years (Enright and Lamont, 1992), since they can withstand many fire cycles (Head and Lacey, 1988). In addition, both *B. serrata* and *B. integrifolia* have high resistance towards drought (Lamont & Markey, 1995) and to the *Phytophthora cinnamomi* root-fungus (George, 1987), an attribute which is associated with resprouters (Moore et al., 2007). These are all fitness advantages that can make their populations stable and increase their invasive capacity. Climate predictions for the Western Cape indicated that it will become drier (Tsedu, 2017) with more potential fires which will favour drought-resistant and fire-dependent *Banksia* species. This makes it essential to devote efforts and resources to achieve their eradication from the infested area.

5.4.2 Eradication of reseeding (non-sprouters) *Banksia* species

Nine *Banksia* species that are reseeders can be eradicated by mechanical means through felling trees. Application of herbicides will not be required since these species do not resprout, except in the case of *B. ericifolia*. Resprouting in this species is restricted to juveniles (Taylor and Hopper, 1988; Renshaw, 2005; Geerts et al., 2013), due to the presence of a lignotuber during its juvenile stages which disappears when approaching the adult stage (Renshaw, 2005). Herbicides are therefore only required to control younger plants. Out of the nine reseeding *Banksia* species present in South Africa, *B. ericifolia* and *B. speciosa* have the largest populations of 18 and 7, respectively. The remaining species have between 1 to 4 populations of which some have been reported to spread with or without fire occurrence (Geerts et al., 2013).

Reseeders tend to have high seed output and recruitment rates, and take less time to reach maturity (Enright and Lamont, 1989; Bell, 2001; Bond and Midgley, 2003). This increases propagule pressure and consequently the likelihood of invasion success (Higgins et al., 2008). Propagule pressure is one of the important factors contributing to a successful invasion (Lockwood et al., 2005; Colautti et al., 2006), including that of Proteaceae species in Fynbos (Moodley et al., 2013). Reseeders have an advantage in areas such as Fynbos with short fire-return intervals, which can enhance their invasion capacity (Moodley et al., 2013). The fecundity life-form hypothesis predicts that they are likely to be self-compatible (Lamont and Wiens, 2003). This suggests that their spread can be more rapid than the co-occurring resprouters which necessitates speedy eradication actions for reseeders.

5.4.3 Follow-up treatment

Follow-up treatment of cleared sites is essential since some plants may have been missed or were not treated with adequate concentration of herbicides (Esler et al., 2014), especially in trials such as the present study. Seedlings emergence from seeds present in the soil is not expected from the study species since they do not form a soil seed bank (Le Maitre, 1992). However, the possibility that some cones may have remained on site can result in seedlings emergence, particularly for the weakly serotinous *B. serrata* whereby some cones are expected to open upon maturity (Moodley et al., 2014; Erckie, pers. obs.).

Follow-ups are necessary to treat any re-infestation occurring in cleared areas (Erasmus and Clayton, 1992). Seedlings of both *B. serrata* and *B. integrifolia* can be hand-pulled whereas resprouting shoots can be foliar-sprayed if plants are below 1 m or otherwise cut if taller than 1 m (Blanchard and Holmes, 2008). Due to the risk involved in using this method of herbicide application, care should be taken to avoid spray drift and prevent damage to non-target species. If no regrowth or seedling recruitment is observed in the infested area for at least three consecutive years, eradication can be considered successful and monitoring discontinued (Rejmánek and Pitcairn, 2002).

5.5 CONCLUSION AND RECOMMENDATIONS

Evidence from this study suggests that a tricopyr+picloram combination (herbicide active ingredient) should be used in an effort to control *B. integrifolia*, *B. serrata* and possibly other resprouting *Banksia* species. Not only has it proven effective, but also observed to pose minimal risk to non-target plant species.

Resprouters have a low spread rate compared to reseeders, due to their reproductive traits and plants can be controlled if treated properly. Given the limited distribution range and restricted infestation size of *Banksia* species, coupled with a lack of accumulated seed banks for the studied and most other *Banksia* species, eradication is feasible. *Banksia* species should therefore be considered targets for eradication under the NEM:BA regulations subject to risk and eradication feasibility considerations as well as their value to the floricultural industry. The success of the eradication will largely depend on the availability of sufficient resources for manual clearing and herbicide treatments and commitment to the eradication programme to do follow-ups to prevent re-invasion.

An eradication programme should be initiated as soon as effective herbicides are confirmed and control efforts should be extended to other invasive populations of *Banksia* in the Western Cape Province. This excludes populations on cut-flower farms, which only require containment to prevent spread from commercial farms. Care must be taken both with cleared materials from these sites and with monitoring effects when these areas are burnt. This will avoid invasion of natural vegetation and replication of impacts already caused by *Banksia* species.

This study suggests that efforts should be focused on the development of experimental methodologies that can provide robust statistical results with treatment sample sizes smaller than typical herbicide trials which are conducted with extensive invasions and therefore with a large number of available plants. Nascent populations of high risk species require robust experimentation with typically small populations whilst eradication is being attempted at the same time.

Future research should conduct a longer term trial using triclopyr+picloram and address the correct concentration of imazapyr and metsulfuron herbicides required to effectively control *B. integrifolia* and possibly other resprouting *Banksia* species. Since only three herbicides were tested in this study, other herbicides with different modes of action should be tested for control efficacy. Moreover, this study only evaluated the efficacy of herbicides and comparison of chemical costs may be necessary in further trials. An assessment of the effects these herbicides may have on native plant species should also be performed.

Chapter 6
General Conclusion

6 SUMMARY

Invasive alien plant (IAP) species affect various components of an ecosystem, such as vegetation structure and functions. Ecosystems invaded by IAP that compete with native species for both biotic and abiotic resources lose their productivity and stability to the detriment of biodiversity (Adair and Groves, 1998; Levine et al., 2003). Ecosystems free from invasion and with high species richness are likely to be productive and may be protected from disturbances (Kennedy et al., 2002). There has been little success in predicting which invaders impose large impacts and which communities are susceptible to invasion due to the complexity of mechanisms involved (Mack, 1996; Ricciardi et al., 2013). Quantifying impacts of specific plant invaders on native biodiversity and the mechanisms behind such effects therefore generates a better understanding, which contributes to the proper management of IAP. This provides justification for expensive control measures of such IAP.

The present study examined the response of native Fynbos vegetation, soil properties (Chapter 2) and native pollinators' richness and abundance (Chapter 3) to the presence of invasive alien *Hakea drupacea* and *Banksia speciosa*. For *Banksia* species, the invasion risk (Chapter 4) and efficacy of chemical control methods (Chapter 5) were evaluated in South Africa. Findings revealed that competition for light and pollination services were the underlying mechanisms responsible for the effects observed. This is in support of the hypothesis that IAP threaten biodiversity and that competition for abiotic and biotic resources are among the mechanisms responsible for alteration of plant and animal community structure observed in many invaded ecosystems (Levine et al., 2003).

6.1 RESEARCH FINDINGS

6.1.1 Impacts of *Hakea drupacea* invasion

This represented the first study concerning impacts of invasion by *H. drupacea* on native plant species diversity, richness and soil properties of invaded communities in the Fynbos Biome of South Africa. In support of the hypothesis tested, sites invaded by *H. drupacea* had lower species richness and diversity than uninvaded sites. This was attributed to reduced light availability resulting from the formation of dense canopy cover by tall *H. drupacea* plants and the accumulation of litter underneath these stands. *Hakea drupacea* further facilitated the establishment of other IAP by creating environmental conditions conducive for invasion.

Contrary to the hypothesis tested, nutrient concentrations, pH and moisture in soils in invaded and adjacent uninvaded sites were similar. This indicated that *H. drupacea* had not significantly altered any of the soil parameters. Lack of significant effects associated with *H. drupacea* invasion on soil properties can be attributed to the invasion being recent and or possession of similar functional traits between *H. drupacea* and native plant species.

6.1.2 Impacts of *Banksia speciosa* invasion

This study investigated bird-pollinated invasive alien *B. speciosa* and native *Protea compacta* to determine whether the invasive species act as a competitor or magnet species. The invasive *B. speciosa* produced flowers with low volumes of sucrose-rich nectar, yet supported significantly higher abundance of sugarbirds and lower abundance of sunbirds. The native *P. compacta* produced flowers with high volumes, but hexose-rich nectar and harboured high abundance of sunbirds and low abundance of sugarbirds.

Contrary to predictions, *B. speciosa* invasion caused no significant reduction in nectar-feeding bird species richness in the *P. compacta* site. The two study species shared only two species of nectar-feeding birds, Cape sugarbirds and Orange-breasted sunbirds. However, its invasion caused a significant decline in the abundance of Cape sugarbirds in the *P. compacta* site. High abundance of sugarbirds in the nectar-poor *B. speciosa* site can be ascribed to its sucrose-rich nectar of high-energy content required by the large-bodied sugarbirds and better nesting sites. Low sunbird abundance observed on *B. speciosa* site can be attributed to its low-nectar volume and suggests that this invasive species does not serve as a valuable resource for other nectar-feeding birds besides sugarbirds.

The competitive effect of invasive *B. speciosa* for sugarbirds with the native *P. compacta* may adversely affect the latter's reproductive success, due to its dependence on sugarbirds for effective pollination (Mostert et al., 1980). Results further demonstrated that the native and invasive species have different levels of compatibility but both relied on pollinators for seed production. No evidence of pollen-limitation was observed that could explain low seed set observed in the present study.

6.1.3 Invasion risk of *Banksia* species

This study evaluated the invasion risk of 14 *Banksia* species present and cultivated for cut-flowers in South Africa, using the weed risk assessment (WRA) tool. It revealed that 79% of *Banksia* species have a high invasive risk and would have been rejected if screened prior to introduction. The remaining 21% obtained intermediate risk scores, placing them in the 'further evaluation' category. These would either require secondary screening to determine their fate or the use of Environmental Impact Classification for Alien Taxa to categorise their impacts.

This study did not find evidence of any *Banksia* species with a low probability of invasion which the floricultural industry could target for commercial use. Most *Banksia* species, such as *Banksia ericifolia*, *Banksia integrifolia* and *Banksia serrata*, which are high-risk species are resistant to the *Phytophora cinnamomi* root-fungus that may increase their chances of becoming invasive.

6.1.4 Chemical control of resprouting *Banksia* species

This study evaluated the efficacy of metsulfuron, imazapyr and triclopyr+picloram herbicides for the control of resprouting *Banksia integrifolia* and *Banksia serrata*. A mechanical method of control, which involved cutting alone, is not a viable control option since *B. integrifolia* and *B. serrata* can resprout. Application of a triclopyr/picloram mix to cut-stumps proved the most effective herbicide since it provided 100% stump mortality whilst metsulfuron 1% was found to be the least effective herbicide treatment. Results indicated that both imazapyr and metsulfuron can provide effective results at controlling *B. integrifolia* when applied at concentration greater than 5%. Results were based on a small sample size and limited to one replicate; and may not be conclusive but they provide an illustrative pattern and are indicative of the potential efficacy of different herbicide concentrations.

Eradication is considered a feasible option since the size of most *Banksia* populations fall within the recommended size optimal for successful eradication (Rejmánek and Pitcairn, 2002). In addition, these species have low spread rate due to lack of persistent soil-stored seed banks.

6.2 MANAGEMENT IMPLICATIONS

There are a number of implications for conservation management that emerged from this study: The reduction in native species diversity and richness, facilitation of establishment of other IAP (Chapter 2), reduction in the abundance of sugarbirds as pollinators (Chapter 3) and the high invasive risk potential (Chapter 4) caused by alien *H. drupacea* and *Banksia* invasions provided evidence for its negative ecological impacts. This constitutes a major threat to the conservation of native biodiversity in the Fynbos Biome where IAP are implicated in the extinction of native plant species, with many others at risk (Esler et al., 2014). This indicates the need for control measures to mitigate impacts and prevent their further establishment and spread. This will also reduce the potential establishment of other IAP that are facilitated by alien invasive Proteaceae. In the absence of control measures, their density will increase and cause more severe effects with ecological and economical losses.

Control of resprouting *B. integrifolia* and *B. serrata* requires the application of herbicides as tested in this study and proved to be effective for the species studied and possibly those that will emerge in future research. Effects of herbicides may be none or small relative to the long-term effects *Banksia* species may have on the native vegetation. In addition, prioritisation of control for *Banksia* species should be based on their potential risk of invasiveness and their resistance to the *Phytophora cinnamomi* root-fungus disease.

Hakea drupacea and *B. speciosa,* which the present study found to pose significant impacts on native biodiversity of the Fynbos Biome, do not resprout hence no herbicide application will be required in their control attempts. These species can be controlled by the use of mechanical methods of control and inspection for seedlings emergence. Follow-up clearing will require less effort with possible low costs due to the absence of soil-stored seed banks for these species.

6.3 RECOMMENDATIONS AND FUTURE RESEARCH

This study provided baseline data and generated future potential research directions. Populations of invasive alien *H. drupacea* and *B. speciosa* should be removed due to the significant impacts demonstrated in the present study. Suitable control methods should be based on the biology of the species and extent of infestation for each species as discussed in detail within the relevant chapters of this thesis.

In this study, the limited sample size of cleared sites placed limitations on the identification of potential legacy effects of *H. drupacea* as an invader. Future studies assessing impacts on the vegetation structure and soil properties should include representative cleared sites to determine legacy effects and identify invasion effects from disturbance effects. Studies assessing impacts of IAP on native plant-pollinator interactions should include both pre- and post-clearing observations and the rate at which nectar in flowers is replenished.

Populations of *Banksia* species outside cut-flower farms should be considered targets for eradication under the NEM:BA regulations. Good management practices, such as containment and monitoring of *Banksia* populations in existing cut-flower farms, should be adopted to minimize and prevent species to escape from cultivation and thus posing threats to native vegetation. Conducting weed risk assessments and climatic modelling of other *Banksia* species currently not present in South Africa to predict their invasiveness and potential distribution range should form part of future research. This will provide opportunities to prevent the introduction of potentially harmful species.

Future research should conduct full-scale and longer term trial using triclopyr+picloram and address the correct concentration of imazapyr and metsulfuron herbicides required for effective control of *B. integrifolia* and possibly other resprouting species of *Banksia*. Estimation of clearing and chemical control costs and an assessment of the effects these herbicides may have on native plant species is necessary in further trials.

The present study has led to new insights into ecological impacts, invasive risks and control methods of invasive alien *H.drupacea* and *Banksia* species present in South Africa. Results generated from this study are useful in making informed decisions about prioritising which *Hakea* and *Banksia* species should receive attention and resource allocation towards their management in the Fynbos Biome.

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APPENDICES

Appendix 1

Sites, locations and characteristics where *Hakea drupacea* data collection was conducted across the *H. drupacea* vegetation range in the Western Cape Province, South Africa.

Site	**Location**	**Invasion category**	***Hakea* cover (%)**
Glencairn 1	S34.15.627;E18.41.138	invaded	90
	S34.15.546;E18.41.026	uninvaded	-
	S34.05.162;E18.23.810	cleared	-
Glencairn 2	S34.15.922;E18.40.627	invaded	75
	S34.15.844;E18.40.638	uninvaded	-
Elim	S34.46.554;E19.83.914	invaded	25
	S34.46.564;E19.83.868	uninvaded	-
Napier	S34.46.046;E19.88.984	invaded	60
	S34.45.995;E19.45.998	uninvaded	-
Stonehaven	S34.08.095;E18.23.733	invaded	70
	S34.08.208;E18.23.737	uninvaded	-
Fishhoek	S34.08.233;E18.24.948	invaded	80
	S34.08.216;E18.24.956	uninvaded	<1
Houtbay	S34.08.216;E18.24.955	invaded	20
	S34.02.341;E18.22.055	uninvaded	-
Fisherhaven	S34.22.006;E19.08.610	invaded	30
	S34.21.844;E19.08.592	uninvaded	-
Hawston	S34.23.038;E19.08.386	invaded	70
	S34.23.412;E19.08.148	uninvaded	-
Vermont	S34.24.313;E19.08.930	invaded	80
	S34.24.310;E19.08.900	uninvaded	-
Raimondo farm	S34.24.017;E19.08.233	invaded	75
	S34.24.019;E19.08.311	uninvaded	-
	S34.24.016;E19.08.859	cleared	-
Onrus	S34.24.255;E19.09.859	invaded	80
	S34.24.259;E19.09.858	uninvaded	-
	S34.24.259;E19.09.876	cleared	-

(-) indicates no *Hakea drupacea* cover

Appendix 2

List of species encountered during the vegetation sampling in 12 sites across the *Hakea drupacea* vegetation range in the Western Cape Province, South Africa.

Species	**Family**	**Growth form**	**Origin**
Acacia cyclops A. Cunn.	Fabaceae	shrub/tree	alien
Acacia longifolia Willd.	Fabaceae	tree	alien
Acacia saligna H.L.Wendl.	Fabaceae	tree	alien
Agathosma capensis (L.) Dümmer	Rutaceae	shrub	native
Anthospermum aethiopicum L.	Rubiaceae	shrub	native
Anthospermum L. species	Rubiaceae	shrub	native
Anthospermum spathulatum Spreng	Rubiaceae	shrub	native
Aspalathus chenopoda L.	Fabaceae	shrub	native
Aspalathus hispida Thunb.	Fabaceae	shrub	native
Aspalathus L. species	Fabaceae	shrub	native
Asparagus retrofractus L.	Asparagaceae	shrub	native
Asparagus rubicundus P.J.Bergius	Asparagaceae	shrub	native
Athanasia trifurcata (L.) L.	Asteraceae	shrub	native
Aulax umbellata (Thunb.) R.Br.	Proteaceae	shrub	native
Berkheya coriacea Harv.	Asteraceae	shrub	native
Berzelia abrotanoides (L.) Brongn.	Bruniaceae	shrub	native
Bolusafra bituminosa (L.) Kuntze	Fabaceae	creeper	native
Briza maxima L.	Poaceae	grass	alien
Campylostachys cernua (L.f.) Kunth	Stilbaceae	shrub	native
Carpobrotus edulis (L.) L.Bolus	Aizoaceae	creeper	native
Cassine peragua L.	Celastraceae	tree	native
Cliffortia atrata Weim.	Rosaceae	shrub	native
Cliffortia brevifolia Weim.	Rosaceae	shrub	native
Cliffortia falcata L.f.	Rosaceae	shrub	native
Cliffortia phyllanthoides Schltr	Rosaceae	shrub	native
Cliffortia ruscifolia L.	Rosaceae	shrub	native
Cliffortia stricta Weim	Rosaceae	shrub	native
Clutia pulchella L.	Euphorbiaceae	shrub	native
Clutia L. species	Euphorbiaceae	shrub	native
Conyza bonariensis (L.) Cronquist	Asteraceae	herb	alien
Cullumia reticulata (L.) Greuter, M.V.Agab. & Wagenitz	Asteraceae	shrub	native
Diosma hirsuta L.	Rutaceae	shrub	native
Diospyros glabra (L.) De Winter	Ebenaceae	tree/shrub	native

Disa bracteata Sw.	Orchidaceae	herb	native
Ehrharta brevifolia Schrad.	Poaceae	grass	native
Ehrharta calycina Sm.	Poaceae	grass	native
Ehrharta villosa Schult.f.	Poaceae	grass	native
Elytropappus rhinocerotis (L.f.) Less.	Asteraceae	shrub	native
Eragrostis capensis (Thunb.) Trin.	Poaceae	grass	native
Erepsia aspera (Haw.) L.Bolus	Aizoaceae	herb	native
Erica corifolia L.	Ericaceae	shrub	native
Erica imbricata L.	Ericaceae	shrub	native
Erica plukenetii L.	Ericaceae	shrub	native
Erica Engl. species	Ericaceae	shrub	native
Erica tristis Bartl.	Ericaceae	shrub	native
Eucalyptus conferruminata S.G.M.	Myrtaceae	tree	alien
Eucalyptus species L'Her	Myrtaceae	tree	alien
Euclea polyandra (L.f.) E.Mey. ex Hiern	Ebenaceae	tree/shrub	native
Ficinia bulbosa (L.) Nees	Cyperaceae	sedge	native
Geochloa rufa (Nees) N.P.Barker & H.P.Linder	Poaceae	grass	native
Gnidia juniperifolia Lam.	Thymelaeaceae	shrub	native
Hakea drupacea R.Br.	Proteaceae	tree/shrub	alien
Helichrysum dasyanthum (Willd.) Sweet	Asteraceae	shrub	native
Helichrysum cymosum (L.) D.Don	Asteraceae	shrub	native
Helichrysum litorale Bolus	Asteraceae	shrub	native
Helichrysum patulum (L.) D.Don	Asteraceae	shrub	native
Helichrysum species Mill.	Asteraceae	shrub	native
Helichrysum teretifolium (L.) D.Don	Asteraceae	shrub	native
Hellmuthia membranacea (Thunb.) R.W.Haines & Lye	Cyperaceae	shrub	native
Hermannia cuneifolia Jacq.	Malvaceae	shrub	native
Indigofera L. species	Fabaceae	dwarf shrub	native
Kiggelaria africana L.	Achariaceae	tree	native
Lobelia L. species	Campanulacea	shrub	native
Lachenalia Aiton species	Hyacinthaceae	geophyte	native
Lebeckia meyeriana Eckl. & Zeyh	Fabaceae	shrub	native
Leptospermum laevigatum (Gaertn.)F.Muell.	Myrtaceae	shrub/tree	alien
Leucadendron linifolium (Jacq.) R.Br.	Proteaceae	shrub	native
Leucadendron modestum I.Williams	Proteaceae	shrub	native
Leucadendron salignum P.J.Bergius	Proteaceae	shrub	native
Leucadendron xanthoconus (Kuntze) K.Schum.	Proteaceae	shrub	native

Lobelia linearis Thunb.	Campanulaceae	shrub	native
Lobostemon glaucophyllus (Jacq.) H.Buek	Boraginaceae	shrub	native
Manulea cheiranthus (L) L.	Scrophulariaceae	shrub	native
Metalasia muricata (L) D.Don	Asteraceae	shrub	native
Metalasia densa (Lam.) P.O.Karis	Asteraceae	shrub	native
Mimetes cucullatus (L.) R.Br.	Proteaceae	shrub	native
Myrsine africana L.	Primulaceae	shrub	native
Nidorella ivifolila (L.) J.C.Manning & Goldblatt	Asteraceae	shrub	native
Oftia africana (L.) Bocq.	Scrophulariaceae	shrub	native
Osteospermum moniliferum L.	Asteraceae	shrub	native
Osyris compressum P.J.Bergius	Santalaceae	tree/shrub	native
Passerina corymbosa Eckl. ex C.H.Wright	Thymelaeaceae	shrub	native
Pelargonium capitatum (L.) L'Hér.	Geraniaceae	dwarf shrub	native
Pelargonium cucullatum (L.) L'Hér.	Geraniaceae	dwarf shrub	native
Pelargonium L'Her. ex Aiton species	Geraniaceae	dwarf shrub	native
Pelargonium tabulare (Burm.f.) L'Hér.	Geraniaceae	dwarf shrub	native
Penaea mucronata L.	Penaeaceae	dwarf shrub	native
Pentameris pallida (Thunb.) Galley & H.P.Linder	Poaceae	grass	native
Pentameris curvifolia (Schrad.) Nees	Poaceae	grass	native
Phylica buxifolia L.	Rhamnaceae	shrub/tree	native
Phylica ericoides L.	Rhamnaceae	shrub	native
Phylica nigrita Sond.	Rhamnaceae	shrub	native
Phylica species L.	Rhamnaceae	shrub	native
Polypogon monspeliensis (L.) Desf.	Poaceae	grass	native
Polygala myrtifolia L.	Polygalaceae	shrub/tree	native
Protea repens (L.) L.	Proteaceae	shrub	native
Restio bifidus Thunb.	Restionaceae	dwarf shrub	native
Restio paludosus Pillans	Restionaceae	dwarf shrub	native
Restio triflora Rottb.	Restionaceae	dwarf shrub	native
Restio species Rottb.	Restionaceae	dwarf shrub	native
Salvia africana L.	Lamiaceae	shrub	native
Searsia cuneifolia (L.f.) F.A.Barkley	Anacardiaceae	shrub	native
Searsia glauca (Thunb.) Moffett	Anacardiaceae	shrub	native
Searsia laevigata (L.) F.A.Barkley	Anacardiaceae	shrub	native
Searsia lucida (L.) F.A.Barkley	Anacardiaceae	shrub	native
Searsia tomentosa (L.) F.A.Barkley	Anacardiaceae	shrub	native
Senecio pterophorus DC.	Asteraceae	shrub	native

Senecio burchellii DC.	Asteraceae	shrub	native
Senecio erosus L.f.	Asteraceae	shrub	native
Senecio pubigerus L.	Asteraceae	shrub	native
Sporobolus africanus (Poir.) Robyns & Tournay	Poaceae	grass	native
Staberoha distachyos (Rottb.) Kunth	Restionaceae	dwarf shrub	native
Stoebe capitata P.J.Bergius	Asteraceae	shrub	native
Stoebe cyathuloides Schltr.	Asteraceae	dwarf shrub	native
Stoebe plumosa (L.) Thunb.	Asteraceae	shrub	native
Struthiola striata Lam.	Thymelaeaceae	shrub	native
Tarchonanthus littoralis P.P.J.Herman	Asteraceae	shrub	native
Tetragonia L. species	Aizoaceae	shrub	native
Thamnochortus fruticosus P.J.Bergius	Restionaceae	shrub	native
Thamnochortus insignis Mast.	Restionaceae	shrub	native
Thamnochortus lucens (Poir.) H.P.Linder	Restionaceae	shrub	native
Thamnochortus species P.J.Bergius	Restionaceae	shrub	native
Tribolium uniolae (L.f.) Renvoize	Poaceae	grass	alien
Trichocephalus stipularis (L.) Brongn.	Rhamnaceae	shrub	native
Watsonia borbonica (Pourr.) Goldblatt	Iridaceae	bulb/herb	native
Wahlenbergia capensis (L.) A.DC.	Campanulaceae	herb	native
Wahlenbergia Schrad. ex Roth species	Campanulaceae	herb	native

Appendix 3

A weed risk assessment of *Banksia baxteri* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	1		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	1		0,1or 2
Broad climate suitability (environmental versatility)	no	4	0	0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	yes	3,10		0 or 2
Naturalized beyond native range	yes	10	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	12	0	0 or 1
Weed of agriculture/horticulture/forestry	no	12	0	0,1,2,3,4
Environmental weed	unknown			0,1,2,3,4
Congeneric weed	yes	5,10	2	0,1,2
Produces spines, thorns or burrs	no	7	1	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	7	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	11,14	0	0 or 1
Host for recognised pests and pathogens	yes	9	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	11	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	17	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	7	0	0 or 1
Grows on infertile soils	yes	6	1	0 or 1
Climbing or smothering growth habit	no	7	0	0 or 1
Form dense thickets	unknown	8		0 or 1
Aquatic	no	7	0	0 or 5
Grass	no	7	0	0 or 1
Nitrogen fixing woody plant	no	13	0	0 or 1
Geophyte	no	7	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	17	0	0 or 1
Produce viable seed	yes	16	1	-1 or 1
Hybridises naturally	unknown			-1 or 1

Self-fertilisation	yes	16	1	-1 or 1
Requires specialist pollinators	no	16	0	0 or -1
Reproduction by vegetative propagation	no	18	-1	-1 or 1
Minimum generative time (years)	3	15	0	0,1,-1
Propagules likely to be dispersed unintentionally	no		-1	-1 or 1
Propagules dispersed intentionally by people	yes		1	-1 or 1
Propagules likely to disperse as a produce contaminant	no	1	-1	-1 or 1
Propagules adapted to wind dispersal	yes	10	1	-1 or 1
Propagules buoyant	no	1	-1	-1 or 1
Propagules bird dispersed	yes	15	1	-1 or 1
Propagules dispersed by other animals (externally)	no	1	-1	-1 or 1
Propagules dispersed by other animals (internally)	no	16	-1	-1 or 1
Prolific seed production	no	15	-1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	15	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	no	7	-1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Biological Resources Study (1999)
2. Australian Native Plants Nursery (2016)
3. Breitwieser et al. (2016)
4. Dave's garden (2016)
5. Fraser (2010)
6. Gardens Online (2016)
7. George (1987)
8. McCaw (2008)
9. McCredie et al. (1985)
10. Moodley et al. (2014)
11. Quattrochi (2012)
12. Randall (2012)
13. USDA, ARS, Germplasm Resources Information Network (2016)
14. Wagstaff (2008)
15. Witkowski et al. (1991)
16. Wooller and Wooller (2001)
17. Wooller et al. (2002)
18. Wooller and Wooller (2004)

Appendix 4

A weed risk assessment of *Banksia burdettii* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	1		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	1		0,1or 2
Broad climate suitability (environmental versatility)	no	4	0	0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	10	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	11	0	0 or 1
Weed of agriculture/horticulture/forestry	no	11	0	0,1,2,3,4
Environmental weed	unknown			0,1,2,3,4
Congeneric weed	yes	10,12	2	0,1,2
Produces spines, thorns or burrs	no	5,6	0	0 or 1
Allelopathic	no	2		0 or 1
Parasitic	no	5,6	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	7	0	0 or 1
Host for recognised pests and pathogens	yes	9	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	2	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	12,13	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	8	0	0 or 1
Grows on infertile soils	yes	7	1	0 or 1
Climbing or smothering growth habit	no	5,6	0	0 or 1
Form dense thickets	yes	7,12	1	0 or 1
Aquatic	no	5,6	0	0 or 5
Grass	no	5,6	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	5,6	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	2	0	0 or 1
Produce viable seed	yes	13	1	-1 or 1
Hybridises naturally	unknown			-1 or 1

Self-fertilisation	yes	14	1	-1 or 1
Requires specialist pollinators	yes	15	-1	0 or -1
Reproduction by vegetative propagation	no	6	-1	-1 or 1
Minimum generative time (years)	2	12	0	0,1,-1
Propagules likely to be dispersed unintentionally	no	13	-1	-1 or 1
Propagules dispersed intentionally by people	yes	13	1	-1 or 1
Propagules likely to disperse as a produce contaminant	no	1	-1	-1 or 1
Propagules adapted to wind dispersal	no	7	-1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	7	-1	-1 or 1
Propagules dispersed by other animals (externally)	no	1	-1	-1 or 1
Propagules dispersed by other animals (internally)				-1 or 1
Prolific seed production	yes	7	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	7,12	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	no	6	-1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Biological Resources Study (1999)
2. Australian Native Plants Nursery (2016)
3. Flora of Australian (2011)
4. Gardens Online (2016)
5. George (1981)
6. George (1987)
7. Lamont and Barker (1988)
8. Matthews (2002)
9. McCredie et al. (1985)
10. Moodley et al. (2014)
11. Randall (2007)
12. Richardson et al. (1990)
13. Tucker and Richardson (1995)
14. Vaughton (1993)
15. Whelan et al. (1980)
16. Witkowski et al. (1991)

Appendix 5

A weed risk assessment of *Banksia coccinea* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	11		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	11		0,1or 2
Broad climate suitability (environmental versatility)	no	1	0	0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	9	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	10	0	0 or 1
Weed of agriculture/horticulture/forestry	no	10	0	0,1,2,3,4
Environmental weed	unknown			0,1,2,3,4
Congeneric weed	yes	11	2	0,1,2
Produces spines, thorns or burrs	no	6,7	0	0 or 1
Allelopathic				0 or 1
Parasitic	no	6,7	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	12	0	0 or 1
Host for recognised pests and pathogens	yes	8	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	12	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	11	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	7	0	0 or 1
Grows on infertile soils	yes	11	1	0 or 1
Climbing or smothering growth habit	no	7	0	0 or 1
Form dense thickets	yes	11	1	0 or 1
Aquatic	no	7	0	0 or 5
Grass	no	7	0	0 or 1
Nitrogen fixing woody plant	unknown		0	0 or 1
Geophyte	no	7	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	2	0	0 or 1
Produce viable seed	yes	11	1	-1 or 1
Hybridises naturally	unknown			-1 or 1

Self-fertilisation	no	4	-1	-1 or 1
Requires specialist pollinators	no	3	0	0 or -1
Reproduction by vegetative propagation	no	7	-1	-1 or 1
Minimum generative time (years)	3	13	0	0,1,-1
Propagules likely to be dispersed unintentionally	no		-1	-1 or 1
Propagules dispersed intentionally by people	yes		1	-1 or 1
Propagules likely to disperse as a produce contaminant	no	1	-1	-1 or 1
Propagules adapted to wind dispersal	yes	11	1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	yes	13	1	-1 or 1
Propagules dispersed by other animals (externally)	no	13	-1	-1 or 1
Propagules dispersed by other animals (internally)				-1 or 1
Prolific seed production	yes	11	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	11	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	no	7	-1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Biological Resources Study (1999)
2. Australian Native Plants Nursery (2016)
3. Collins and Rebelo (1987)
4. Fuss and Sedgley (1991)
5. Gardens Online (2016)
6. George (1981)
7. George (1987)
8. McCredie et al. (1985)
9. Moodley et al. (2014)
10. Randall (2007)
11.Tucker and Richardson (1995)
12.Wagstaff (2008)
13. Witkowski et al. (1991)

Appendix 6

A weed risk assessment of *Banksia ericifolia* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	20		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	20		0,1or 2
Broad climate suitability (environmental versatility)	yes	6	0	0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	11		-2,-1,0,1,2
Garden/amenity/disturbance weed	no	18	0	0 or 1
Weed of agriculture/horticulture/forestry	no	18	0	0,1,2,3,4
Environmental weed	yes	12	4	0,1,2,3,4
Congeneric weed	yes	11	2	0,1,2
Produces spines, thorns or burrs	no	7	0	0 or 1
Allelopathic	no	1		0 or 1
Parasitic	no	21	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	22	0	0 or 1
Host for recognised pests and pathogens	no	14	0	0 or 1
Causes allergies or is otherwise toxic to humans	no	22	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	24	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	2	0	0 or 1
Grows on infertile soils	yes	20	1	0 or 1
Climbing or smothering growth habit	no	8	0	0 or 1
Form dense thickets	yes	25	1	0 or 1
Aquatic	no	8	0	0 or 5
Grass	no	8	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	8	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	1	0	0 or 1
Produce viable seed	yes	26	1	-1 or 1
Hybridises naturally	yes	8,19		-1 or 1

Self-fertilisation	no	10	-1	-1 or 1
Requires specialist pollinators	no	16	0	0 or -1
Reproduction by vegetative propagation	no	8	-1	-1 or 1
Minimum generative time (years)	3	12	0	0,1,-1
Propagules likely to be dispersed unintentionally	no	4	-1	-1 or 1
Propagules dispersed intentionally by people	yes	6	1	-1 or 1
Propagules likely to disperse as a produce contaminant	no	1	-1	-1 or 1
Propagules adapted to wind dispersal	yes	12	1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	13	-1	-1 or 1
Propagules dispersed by other animals (externally)	no	13	-1	-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	yes	12	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	12,26	1	-1 or 1
Well controlled by herbicides	no	9		-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	no	8,25	-1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Native Plants Society (2007)
2. Australian National Botanic Gardens (2002)
3. Bradstock and Bedward (1992)
4. Benson and McDougall (2000)
5. Collins and Rebelo (1987)
6. Elliot (2003)
7. Fraser (2010)
8.George (1987)
9. Geerts et al. (2013)
10. Goldingay et al. (1991)
11. Henderson (2007)
12. Honing et al. (1992)
13. Hughes et al. (1994)
14. McCredie et al. (1985)
15. Moodley et al. (2014)
16.Paton and Turner (1985)
17. Protea Atlas Project (2012)
18. Randall (2007)
19. Sedgley et al. (1994)
20.Tucker and Richardson (1995)
21. USDA, ARS.National Genetic Resources Programme (2016)
22. Wagstaff (2008).
23. Witkowski et al. (1991)
24. Zammit and Westoby (1988)
25. Zammit and Westoby (1987)
26. Zammit and Westoby (1987

Appendix 7

A weed risk assessment of *Banksia formosa* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	10		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	10		0,1or 2
Broad climate suitability (environmental versatility)	unknown			0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	13,14	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	14	0	0 or 1
Weed of agriculture/horticulture/forestry	no	14	0	0,1,2,3,4
Environmental weed	unknown			0,1,2,3,4
Congeneric weed	yes	13	2	0,1,2
Produces spines, thorns or burrs	no	6	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	unknown			0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	16	0	0 or 1
Host for recognised pests and pathogens	yes	1,8	1	0 or 1
Causes allergies or is otherwise toxic to humans	unknown			0 or 1
Creates a fire hazard in natural ecosystems	yes	3	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	2	0	0 or 1
Grows on infertile soils	yes	4	1	0 or 1
Climbing or smothering growth habit	no	2	0	0 or 1
Form dense thickets	yes	15	1	0 or 1
Aquatic	no	2,4	0	0 or 5
Grass	no	2,4	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	2,4	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	2	0	0 or 1
Produce viable seed	yes	13	1	-1 or 1
Hybridises naturally	unknown			-1 or 1

Self-fertilisation	yes	12	1	-1 or 1
Requires specialist pollinators	no	12	0	0 or -1
Reproduction by vegetative propagation	no	17	-1	-1 or 1
Minimum generative time (years)	5	3	-1	0,1,-1
Propagules likely to be dispersed unintentionally	no	2	-1	-1 or 1
Propagules dispersed intentionally by people	yes	11	1	-1 or 1
Propagules likely to disperse as a produce contaminant	unknown			-1 or 1
Propagules adapted to wind dispersal	yes	5,9	1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	9	-1	-1 or 1
Propagules dispersed by other animals (externally)	unknown			-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	yes	13	-1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	13	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	no	9	-1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Alcoa of Australia (2002)
2. Australian Native Plants Society (2007)
3. Barret (2002)
4. Cochrane et al. (2002)
5. Fitzpatrick et al. (2008)
6. George (1987)
7. Geerts et al. (2013)
8. Hart (1983)
9. Lamont and Groom (1998)
10. Lamont et al. 1985
11. Leonhardt and Richard (1999)
12. Matthew and Sedgley (1998)
13. Moodley, et al. (2016)
14. Randall (2007)
15. Sinclair et al. (1996)
16. Wagstaff (2008)
17. Gardening with angus.http://www.gardeningwithangus.com.au (accessed 19/10/2016).

Appendix 8

A weed risk assessment of *Banksia hookeriana* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	14		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	14		0,1or 2
Broad climate suitability (environmental versatility)	unknown			0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	11	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	10	0	0 or 1
Weed of agriculture/horticulture/forestry	no	10	0	0,1,2,3,4
Environmental weed	unknown			0,1,2,3,4
Congeneric weed	yes	14	2	0,1,2
Produces spines, thorns or burrs	no	3	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	3	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	unknown			0 or 1
Host for recognised pests and pathogens	yes	9	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	15		0 or 1
Creates a fire hazard in natural ecosystems	yes	11	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	3	0	0 or 1
Grows on infertile soils	yes	7	1	0 or 1
Climbing or smothering growth habit	no	3	0	0 or 1
Form dense thickets	yes	11	1	0 or 1
Aquatic	no	3	0	0 or 5
Grass	no	3	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	3	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	16	0	0 or 1
Produce viable seed	yes	2,11	1	-1 or 1
Hybridises naturally	yes	3,12,13	1	-1 or 1

Self-fertilisation	yes	13	1	-1 or 1
Requires specialist pollinators	no	5	0	0 or -1
Reproduction by vegetative propagation	no	14	-1	-1 or 1
Minimum generative time (years)	3	2	0	0,1,-1
Propagules likely to be dispersed unintentionally	no		-1	-1 or 1
Propagules dispersed intentionally by people	yes	16	1	-1 or 1
Propagules likely to disperse as a produce contaminant	unknown			-1 or 1
Propagules adapted to wind dispersal	yes	6,11	1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	yes	4	1	-1 or 1
Propagules dispersed by other animals (externally)	no	4	-1	-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	yes	11,14	-1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	2,11	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	no	3	-1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Native Plants Society (2007)
2. Enright et al. (1996)
3. George (1987)
4. He et al. (2004)
5. Krauss et al. (2009)
6. Lamont et al. (1993)
7. Lamont et al. (1985)
8. Leonhardt and Richard (1999)
9. McCredie et al. (1985)
10. Randall (2007)
11. Richardson et al. (1990).
12. Sedgley and Janick (1998)
13. Sedgley et al. (1996)
14. Tucker and Richardson (1995)
15. Wagstaff (2008)
16. Witkowskie et al. (1994)

Appendix 9

A weed risk assessment of *Banksia integrifolia* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes			0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high			0,1or 2
Broad climate suitability (environmental versatility)	unknown			0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	yes	2,5,11		0 or 2
Naturalized beyond native range	yes	2,5,11	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	10	0	0 or 1
Weed of agriculture/horticulture/forestry	no	10	0	0,1,2,3,4
Environmental weed	yes	15		0,1,2,3,4
Congeneric weed	yes	15	2	0,1,2
Produces spines, thorns or burrs	no	4	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	13	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	14	0	0 or 1
Host for recognised pests and pathogens	yes	7	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	13	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	9	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	4	0	0 or 1
Grows on infertile soils	yes	1	1	0 or 1
Climbing or smothering growth habit	no	4	0	0 or 1
Form dense thickets	yes	pers.obs	1	0 or 1
Aquatic	no	4	0	0 or 5
Grass	no	4	0	0 or 1
Nitrogen fixing woody plant	yes	6	1	0 or 1
Geophyte	no	4	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	1	0	0 or 1
Produce viable seed	yes	9	1	-1 or 1
Hybridises naturally	yes	4, 8	1	-1 or 1

Self-fertilisation	yes	3	1	-1 or 1
Requires specialist pollinators	yes	3	-1	0 or -1
Reproduction by vegetative propagation	no	4	-1	-1 or 1
Minimum generative time (years)	3	3	0	0,1,-1
Propagules likely to be dispersed unintentionally	no	9	-1	-1 or 1
Propagules dispersed intentionally by people	yes	9	1	-1 or 1
Propagules likely to disperse as a produce contaminant	no		-1	-1 or 1
Propagules adapted to wind dispersal	no	3	-1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	3	-1	-1 or 1
Propagules dispersed by other animals (externally)	no	3	-1	-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	yes	9	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	9	1	-1 or 1
Well controlled by herbicides	yes	Pers.obs.	-1	-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	yes	4	1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Native Plants Society (2016)
2. Cameron (2000)
3. Cunningham (1991)
4. George (1987)
5. Henderson 2007
6. Logan et al. (1988)
7. McCredie et al. (1985)
8. Molyneux and Forester (2007)
9. Moodley et al. (2014)
10. Randall (2007)
11. Wagner et al. (2016)
12. Plants For a Future Database: http://www.pfaf.org/database/plants.php (accessed 17/10/2016).
13. Flora of Australia: http://www.anbg.gov.au/abrs/online-resources/flora/redirect.jsp (accessed 19/10/2016).
14. National Centre for Biotechnology Information: http://www.ncbi.nlm.nih.gov/sites/etrez (accessed 17/10/2016).
15. Global Compendium of Weeds: http://www. hear.org/gcw/species/banksia_intergrifolia (accessed 18/10/2016).

Appendix 10

A weed risk assessment of *Banksia menziesii* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	10		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	10		0,1or 2
Broad climate suitability (environmental versatility)	unknown		0	0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	10,16	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	9	0	0 or 1
Weed of agriculture/horticulture/forestry	no	9	0	0,1,2,3,4
Environmental weed	unknown			0,1,2,3,4
Congeneric weed	yes	10	2	0,1,2
Produces spines, thorns or burrs	no	3	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	3	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	6	0	0 or 1
Host for recognised pests and pathogens	yes	5	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	6,14	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	13	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	3	0	0 or 1
Grows on infertile soils	yes	10,13	1	0 or 1
Climbing or smothering growth habit	no	3	0	0 or 1
Form dense thickets	no	13	0	0 or 1
Aquatic	no	3	0	0 or 5
Grass	no	3	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	3	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	1,15	0	0 or 1
Produce viable seed	no	10	-1	-1 or 1
Hybridises naturally	yes	3,12	1	-1 or 1

Self-fertilisation	no	7,8,11	-1	-1 or 1
Requires specialist pollinators	no	7	0	0 or -1
Reproduction by vegetative propagation	yes	15	1	-1 or 1
Minimum generative time (years)	5	2,10	-1	0,1,-1
Propagules likely to be dispersed unintentionally	no	10	-1	-1 or 1
Propagules dispersed intentionally by people	yes	10	1	-1 or 1
Propagules likely to disperse as a produce contaminant	unknown			-1 or 1
Propagules adapted to wind dispersal	yes	13	1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	13	-1	-1 or 1
Propagules dispersed by other animals (externally)	unknown			-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	no	2,10	-1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	no	10	-1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	yes	2,3	1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Biological Resources (1999)
2. Cowling et al. (1987)
3. George (1987)
4. Lamont et al. (1985)
5. McCredie et al. (1985)
6.Quattrochi (2012)
7. Ramsey (1988)
8. Ramsey and Vaughton (1991)
9. Randall. (2012)
10. Richardson et al. (1990)
11. Scott (1980)
12. Sedgley, et al. (1996)
13. Tucker and Richardson (1995)
14. Wagstaff (2008)
15. Australian Native Plants Society.*Banksia menziesii*.http://www.anpsa.org.au/b-men.html (accessed 23/10/2016)
16. Global Compendium of Weeds.http://www.hear.org/gcw/species/banksia_menziesii (accessed 18/10/2016)

Appendix 11

A weed risk assessment of *Banksia prionotes* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	12		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	12		0,1or 2
Broad climate suitability (environmental versatility)	unknown			0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	7,15	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	8	0	0 or 1
Weed of agriculture/horticulture/forestry	no	8	0	0,1,2,3,4
Environmental weed	yes	9	4	0,1,2,3,4
Congeneric weed	yes	9	2	0,1,2
Produces spines, thorns or burrs	no	4	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	4	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	13	0	0 or 1
Host for recognised pests and pathogens	yes	6	1	0 or 1
Causes allergies or is otherwise toxic to humans	unknown			0 or 1
Creates a fire hazard in natural ecosystems	yes	9,12	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	4	0	0 or 1
Grows on infertile soils	yes	9,12	1	0 or 1
Climbing or smothering growth habit	no	4	0	0 or 1
Form dense thickets	yes	9	1	0 or 1
Aquatic	no	4	0	0 or 5
Grass	no	4	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	4	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	14	0	0 or 1
Produce viable seed	yes	3,9	1	-1 or 1
Hybridises naturally	yes	4,10,11	1	-1 or 1

Self-fertilisation	no	2	-1	-1 or 1
Requires specialist pollinators	no	1,2	0	0 or -1
Reproduction by vegetative propagation	no	14	-1	-1 or 1
Minimum generative time (years)	3	3	-1	0,1,-1
Propagules likely to be dispersed unintentionally	no	9	-1	-1 or 1
Propagules dispersed intentionally by people	yes	9	1	-1 or 1
Propagules likely to disperse as a produce contaminant	unknown			-1 or 1
Propagules adapted to wind dispersal	yes	12	1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	12	-1	-1 or 1
Propagules dispersed by other animals (externally)	unknown			-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	yes	3,12	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	3,12	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	no	4	-1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Collins and Rebelo (1987)
2. Collins and Spice (1986)
3. Cowling et al. (1987)
4. George (1987)
5. Lamont et al. (1985)
6. McCredie et al. (1985)
7. Moodley et al. (2014)
8. Randall (2012
9. Richardson et al. (1990)
10. Sedgley (1998)
11. Sedgley, et al. (1996)
12. Tucker and Richardson (1995)
13.Wagstaff (2008)
14. Australian Native Plants Society.*Banksia prionotes*.http://www.anpsa.org.au/b-pri.html (accessed 23/10/2016)
15. Global Compendium of Weeds.http://www.hear.org/gcw/species/banksia_prionotes (accessed 18/10/2016)

Appendix 12

A weed risk assessment of *Banksia quercifolia* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	7		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	7		0,1or 2
Broad climate suitability (environmental versatility)	unknown			0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	6,1	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	8	0	0 or 1
Weed of agriculture/horticulture/forestry	no	8	0	0,1,2,3,4
Environmental weed	unknown			0,1,2,3,4
Congeneric weed	yes	10	2	0,1,2
Produces spines, thorns or burrs	no	3	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	3	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	9	0	0 or 1
Host for recognised pests and pathogens	yes	5	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	9	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	7	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	yes	3	1	0 or 1
Grows on infertile soils	yes	4	1	0 or 1
Climbing or smothering growth habit	no	3	0	0 or 1
Form dense thickets	yes	7	1	0 or 1
Aquatic	no	3	0	0 or 5
Grass	no	3	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	3	0	0 or 1
Evidence of substantial reproductive failure in native habitat	yes	1	1	0 or 1
Produce viable seed	yes	7	1	-1 or 1
Hybridises naturally	unknown			-1 or 1

Self-fertilisation	unknown			-1 or 1
Requires specialist pollinators	yes	2	-1	0 or -1
Reproduction by vegetative propagation	no	1,3	-1	-1 or 1
Minimum generative time (years)	3	7	0	0,1,-1
Propagules likely to be dispersed unintentionally	no	7	-1	-1 or 1
Propagules dispersed intentionally by people	yes	7	1	-1 or 1
Propagules likely to disperse as a produce contaminant	no	1	-1	-1 or 1
Propagules adapted to wind dispersal	yes	7	1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	unknown			-1 or 1
Propagules dispersed by other animals (externally)	no	7	-1	-1 or 1
Propagules dispersed by other animals (internally)	no	7	-1	-1 or 1
Prolific seed production	yes	7	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	7	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	no	3	-1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Native Plants Society (2008). Banksia quercifolia anpsa.org.au/b-que.html (accessed 23/10/2016)

2. Collins and Rebelo (1987)

3. George (1987)

4. Lamont et al. (1985)

5. McCredie et al. (1985)

6. Moodley et al. (2014)

7. Tucker and Richardson (1995)

8. Randall (2007)

9. Wagstaff (2008)

10. Global Compendium of weeds:http://www.hear.org/gcw/species/banksia_quercifolia (accessed 18/10/2016)

Appendix 13

A weed risk assessment of *Banksia serrata* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	9		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	9		0,1or 2
Broad climate suitability (environmental versatility)	unknown			0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	8,18	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	12	0	0 or 1
Weed of agriculture/horticulture/forestry	no	12	0	0,1,2,3,4
Environmental weed	yes	18	4	0,1,2,3,4
Congeneric weed	yes	18	2	0,1,2
Produces spines, thorns or burrs	no	5	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	5	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	unknown			0 or 1
Host for recognised pests and pathogens	no	11	0	0 or 1
Causes allergies or is otherwise toxic to humans	unknown			0 or 1
Creates a fire hazard in natural ecosystems	yes	4	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	1,17	0	0 or 1
Grows on infertile soils	yes	9	1	0 or 1
Climbing or smothering growth habit	no	5	0	0 or 1
Form dense thickets	yes	pers.obs	1	0 or 1
Aquatic	no	5	0	0 or 5
Grass	no	5	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	5	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	16	0	0 or 1
Produce viable seed	yes	2	1	-1 or 1
Hybridises naturally	yes	5,16	1	-1 or 1

Self-fertilisation	no	6	-1	-1 or 1
Requires specialist pollinators	no	6	-1	0 or -1
Reproduction by vegetative propagation	yes	2,4,5	-1	-1 or 1
Minimum generative time (years)	2	4	0	0,1,-1
Propagules likely to be dispersed unintentionally	no	13	-1	-1 or 1
Propagules dispersed intentionally by people	yes	10,13	1	-1 or 1
Propagules likely to disperse as a produce contaminant	unknown			-1 or 1
Propagules adapted to wind dispersal	no	7	-1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	7	-1	-1 or 1
Propagules dispersed by other animals (externally)	unknown		-1	-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	yes	14	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	14	1	-1 or 1
Well controlled by herbicides	yes	pers.obs	-1	-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	yes	3,15	1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Benson and McDougal (2000)
2. Bradstock (1990)
3. Bradstock (1991)
4. Bradstock and Myerscough (1988)
5. George (1987)
6. Goldingay and Carthew (1998)
7. Hammill et al. (1998)
8. Henderson (2007)
9. Lamont et al. (1985)
10. Moodley et al. (2014)
11. McCredie et al. (1985)
12. Randall (2012)
13. Sedgley and Janick (1998)
14. Vaughton et al. (1998)
15. Whelan et al. (1998)
16. Australian Native Plants Society.http://www.anpsa.org.au/b-ser.html (accessed 23/10/2016).
17. Australian Native Plants.http://www.australianplants.com/plants.asp (accessed 23/10/2016).
18. Global compendium of Weeds.http://www.hear.org/gcw/species/banksia_serrata (accessed 18/10/2016).
19. Available at:http://www.pinterest.com/pin/22166223139282908/Banksia serrata (accessed 19/10/2016).

Appendix 14

A weed risk assessment of *Banksia speciosa* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	6		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high	10		0,1or 2
Broad climate suitability (environmental versatility)	no	4	0	0 or 1
Native or naturalized in regions with extended dry periods	no	12		0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	7	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	9	0	0 or 1
Weed of agriculture/horticulture/forestry	no	9	0	0,1,2,3,4
Environmental weed	yes	14		0,1,2,3,4
Congeneric weed	yes	14	2	0,1,2
Produces spines, thorns or burrs	no	3	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	3	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	unknown			0 or 1
Host for recognised pests and pathogens	yes	8	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	3	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	2	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	yes	12	1	0 or 1
Grows on infertile soils	yes	11,13	1	0 or 1
Climbing or smothering growth habit	no	3	0	0 or 1
Form dense thickets	yes	10	1	0 or 1
Aquatic	no	3	0	0 or 5
Grass	no	3	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	3	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	12	0	0 or 1
Produce viable seed	yes	10,11	1	-1 or 1
Hybridises naturally	unknown			-1 or 1

Self-fertilisation	yes	7	1	-1 or 1
Requires specialist pollinators	no	1	0	0 or -1
Reproduction by vegetative propagation	no	11	-1	-1 or 1
Minimum generative time (years)	3	11,12	0	0,1,-1
Propagules likely to be dispersed unintentionally	no	7	-1	-1 or 1
Propagules dispersed intentionally by people	yes	7	1	-1 or 1
Propagules likely to disperse as a produce contaminant	no	13	-1	-1 or 1
Propagules adapted to wind dispersal	yes	11	1	-1 or 1
Propagules buoyant				-1 or 1
Propagules bird dispersed	no	11	-1	-1 or 1
Propagules dispersed by other animals (externally)	unknown			-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	yes	10	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	10	1	-1 or 1
Well controlled by herbicides	no	3,pers.obs	1	-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	yes	3	1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Collins and Rebelo (1987).
2. Cowling and Lamont (1985)
3. George (1987)
4. Kirton (2007)
5. Lamont et al. (1985)
6. Moodley et al. (2014)
7. Moodley et al. (2015)
8. McCredie et al. (1985)
9. Randall (2012)
10. Richardson et al. (1990)
11.Witkwoskie et al. (1991)
12. Australian Native Plants Society.http://www.anpsa.org.au/APLO30/jun03-4.html (accessed 23/10/2016)
13. Australian Native Plants.http://www.australianplants.com/plants.asp (accessed 23/10/2016)
14. Global compendium of Weeds.http://www.hear.org/gcw/species/banksia_speciosa (accessed 18/10/2016)
15. Flora of Australia. http://www.anbg.gov.au/abrs/online-resources/flora/redirect.jsp (accessed 19/10/2016)

Appendix 15

A weed risk assessment of *Banksia spinulosa* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no	12	0	0 or -3
Species suited to South African climate	yes	18		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high			0,1or 2
Broad climate suitability (environmental versatility)	yes	20	0	0 or 1
Native or naturalized in regions with extended dry periods	yes	20		0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	19	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	19	0	0 or 1
Weed of agriculture/horticulture/forestry	no	19	0	0,1,2,3,4
Environmental weed	no	19	0	0,1,2,3,4
Congeneric weed	yes	2,6	2	0,1,2
Produces spines, thorns or burrs	no	4	0	0 or 1
Allelopathic	unknown	8		0 or 1
Parasitic	no	12	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	8	0	0 or 1
Host for recognised pests and pathogens	no	10	0	0 or 1
Causes allergies or is otherwise toxic to humans	no	8	0	0 or 1
Creates a fire hazard in natural ecosystems	yes	7	1	0 or 1
Is a shade tolerant plant at some stage of its life cycle	no	18	0	0 or 1
Grows on infertile soils	yes	1	1	0 or 1
Climbing or smothering growth habit	no	4	0	0 or 1
Form dense thickets	yes	9	1	0 or 1
Aquatic	no	4	0	0 or 5
Grass	no	4	0	0 or 1
Nitrogen fixing woody plant	unknown			0 or 1
Geophyte	no	4	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	1,17	0	0 or 1
Produce viable seed	yes	13	1	-1 or 1
Hybridises naturally	yes	1	1	-1 or 1

Self-fertilisation	no	5	-1	-1 or 1
Requires specialist pollinators	no	16	0	0 or -1
Reproduction by vegetative propagation	no	18	-1	-1 or 1
Minimum generative time (years)	2	18	0	0,1,-1
Propagules likely to be dispersed unintentionally	no	1	-1	-1 or 1
Propagules dispersed intentionally by people	yes	18	1	-1 or 1
Propagules likely to disperse as a produce contaminant	no	20	-1	-1 or 1
Propagules adapted to wind dispersal	yes	13	1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	17	-1	-1 or 1
Propagules dispersed by other animals (externally)	no	1	-1	-1 or 1
Propagules dispersed by other animals (internally)	unknown			-1 or 1
Prolific seed production	no	13	-1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	13	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	yes	4	1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Benson and McDougall (2000)
2. Fraser (2010)
3. Carthew et al. (1996)
4. George (1987)
5. Goldingay and Whelan (1990)
6. Honing et al. (1992)
7. Knox and Morrison (2005)
8. Matthews (2002)
9. Myerscough et al. (2001)
10. McCredie et al. (1985)
11.Scott (2009)
12. Spencer, R. (2002)
13.Vaughton and Ramsey (1997)
14. Vaughton (1998)
15.Vaughton (1993)
16. Vaughton (1992)
17. Australian Native Plants Society. http://www.anpsa.org.au/b-spi.html (accessed 23/10/2016)
18. Australian National Botanic Gardens http://www..anbg.gov.au/gnp7/banksia-spinulosa.html. (accessed 23/10/2016)
19. Global Compendium of Weeds. http://www.hear.org/gcw/species/banksia_spinulosa (accessed 18/10/2016)
20. Sunny Gardens. http://www.sunnygardens.com/garden_plants/banksia_0287.php (accessed 19/10/2016)

Appendix 16

A weed risk assessment of *Banksia sphaerocarpa* adapted from Pheloung et al. (1999) modified for use in South Africa.

Question	Answer	Reference	Score	score range
Is the species highly domesticated?	no		0	0 or -3
Species suited to South African climate	yes	12		0,1or 2
Quality of climate match data (0-low; 1-intermediate; 2-high)	high			0,1or 2
Broad climate suitability (environmental versatility)	no	15	0	0 or 1
Native or naturalized in regions with extended dry periods	unknown			0 or 1
Does the species have a history of repeated introductions outside its natural range	unknown			0 or 2
Naturalized beyond native range	yes	5	2	-2,-1,0,1,2
Garden/amenity/disturbance weed	no	8,14	0	0 or 1
Weed of agriculture/horticulture/forestry	no	8,14	0	0,1,2,3,4
Environmental weed	no	8	0	0,1,2,3,4
Congeneric weed	yes	11	2	0,1,2
Produces spines, thorns or burrs	no	2	0	0 or 1
Allelopathic	unknown			0 or 1
Parasitic	no	1	0	0 or 1
Unpalatable to grazing animals	unknown			-1 or 1
Toxic to animals	no	7,10	0	0 or 1
Host for recognised pests and pathogens	yes	6	1	0 or 1
Causes allergies or is otherwise toxic to humans	no	7,10	0	0 or 1
Creates a fire hazard in natural ecosystems	unknown			0 or 1
Is a shade tolerant plant at some stage of its life cycle	yes	2	1	0 or 1
Grows on infertile soils	yes	9	1	0 or 1
Climbing or smothering growth habit	no	2	0	0 or 1
Form dense thickets	unknown			0 or 1
Aquatic	no	2	0	0 or 5
Grass	no	2	0	0 or 1
Nitrogen fixing woody plant	no	1	0	0 or 1
Geophyte	no	2	0	0 or 1
Evidence of substantial reproductive failure in native habitat	no	12	0	0 or 1
Produce viable seed	yes	9	1	-1 or 1
Hybridises naturally	unknown			-1 or 1

Self-fertilisation	yes	4	1	-1 or 1
Requires specialist pollinators	no	3	0	0 or -1
Reproduction by vegetative propagation	no	1	-1	-1 or 1
Minimum generative time (years)	5	9	0	0,1,-1
Propagules likely to be dispersed unintentionally	no	1	-1	-1 or 1
Propagules dispersed intentionally by people	yes	9	1	-1 or 1
Propagules likely to disperse as a produce contaminant	no	1	-1	-1 or 1
Propagules adapted to wind dispersal	yes	9	-1	-1 or 1
Propagules buoyant	unknown			-1 or 1
Propagules bird dispersed	no	9	-1	-1 or 1
Propagules dispersed by other animals (externally)	no	1	-1	-1 or 1
Propagules dispersed by other animals (internally)	no	1		-1 or 1
Prolific seed production	yes	9	1	-1 or 1
Evidence that persistent propagule bank is formed (>1 yr)	yes	9	1	-1 or 1
Well controlled by herbicides	unknown			-1 or 1
Tolerates or benefits from mutilation, cultivation or fire	yes	2,9	1	-1 or 1
Effective natural enemies present in South Africa	unknown			-1 or 1

1. Australian Biological Resources (1999)
2. George (1987)
3. Liber and Collins (2009)
4. Llorens et al. (2012)
5. Moodley et al. (2014)
6. McCredie et al. (1985)
7. Quattrochi (2012)
8. Randall (2012)
9. Richardson et al. (1990)
10.Wagstaff (2008)
11.Williams (2008)
12. Australian Native Plants Society. *Banksia sphaerocarpa*.http://wwwanspa.org.au/b-sph.html (accessed 23/10/2016).
13. Australian Native Plants Nursery. *Banksia sphaerocarpa*.http://www.australianplants.com/plants (accessed 23/10/2016).
14. Global Compendium of Weeds. http://www.hear.org/gcw/species/banksia_sphaerocarpa (accessed 18/10/2016).
15. Gardening with Angus. http://www.gardeningwithangus.com.au.banksia-sphaerocarpa-fox-banksia (accessed 19/10/2016).