



# MONASH University

**The ecology, management and restoration of ecological  
communities affected by *Pittosporum undulatum* Vent.  
across Victoria, Australia**

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Master of Philosophy (Environmental Science)**

**A thesis submitted for the degree of (*Doctor of Philosophy*) at Monash  
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## Abstract

The threat and impacts of invasive species worldwide carry a significant environmental, social and economic cost. As a means of protecting the ecological integrity of the natural processes that underpin our society, a strong understanding of the key environmental, societal and economic issues must drive effective management actions. Given the complex and unpredictable range of influences and pressures encompassing the problem of invasive species, this goal is often difficult to achieve. However, when land managers can align theoretical practical and societal considerations, positive outcomes are possible. Through a literature review and field studies, this thesis explores the process of aligning these pillars of successful invasive species management.

Chapter 1 examines the concept of invasive species, along with how perceptions and belief towards invaders have changed over time. Invasive plant species are found throughout almost all ecosystems across the globe. Invaders are often able to establish within novel environments because they possess traits unique to the new community, enabling the species to exploit previously empty niches. Species capable of invading a community without an obvious or unique trait/s present a particularly interesting case study. These species can provide new insights into the invasion process, with potential application for management. An example of this can be found in the Australian native tree *Pittosporum undulatum*, which is viewed as an invader both abroad and at home. Using *P. undulatum* as a study species has allowed examination of the perceptions and values ascribed to native and invasive plants, along with their respective ecological value in an increasingly globalised, disturbed and climatically variable world.

Chapter 2 explores the reproductive life history of *Pittosporum undulatum*. *Pittosporum undulatum* is broadly viewed as a dioecious species, despite observations of population sex ratios that have varied markedly. As a means of informing the management of *P. undulatum* a field-based study was conducted comparing the reproductive ecology of native and invasive populations of *P. undulatum* across Victoria, South Eastern Australia. In contrast to previous studies, results from this

work found *P. undulatum* populations to be gynodioecious, with no strong difference in the proportion of females or hermaphrodites within a population, regardless of its native or invasive origin. Resources, available to *P. undulatum*, determined by the nutrient composition of leaves, did not differ strongly between native and invasive populations. However, relative to native populations, invaders tended to produce seed with a greater mean mass and in similar numbers. Given the strong history of anthropogenic spread of *P. undulatum*, this result may reflect reduced pressure for dispersal. Furthermore, greater seed mass may confer an advantage across the highly variable environments present at the invasion front, particularly in areas of significant human disturbance.

Invasion biologists are often critical that invasive species management programs are regularly coordinated without the monitoring and evaluation needed to achieve desired outcomes. In contrast, practitioners of ecological restoration often feel that research on the topic is too theoretical to provide real value to their work. As a means of bridging this divide I conducted studies evaluating the success of *P. undulatum* restoration efforts and how theory could inform these projects. No study of this kind, measuring the success of *P. undulatum* control works, had been published to date.

Chapter 3 focuses on the response of plant communities to *P. undulatum* control works over time. Results supported the findings of previous studies, which showed that the presence of *P. undulatum* populations correlated strongly with a decline in floral species richness and density. Interestingly it was found that floral species richness and density returned relatively quickly following *P. undulatum* control programs. The species returning to the treated sites were more likely to be indigenous to the area relative to exotic species. Finally, with increasing time populations tended to become increasingly similar in condition when compared to a nearby high quality reference sites. All of these findings together strongly support action to control *P. undulatum* as a means of supporting local biodiversity. Findings from this study were used to explore existing theory and its application to practical restoration.

Chapter 4 focuses on the response of bird communities to *P. undulatum* control works over time. The findings suggest that communities invaded by *P. undulatum* support

bird communities reduced in carnivorous species. This may relate to the dense architecture of *P. undulatum* canopy, restricting carnivorous species and their capacity to sight and hunt prey. Regardless of this finding, the species richness and density of bird communities showed no difference between high quality remnant areas, areas cleared of *P. undulatum* and areas currently invaded by *P. undulatum*. This result may more strongly reflect a general decline in ground dwelling bird species across Australia as opposed to any influence *P. undulatum* populations may exert on bird communities. This work highlights the need to examine species management programs with a holistic view. As my results show, the presence or absence of an invader is only one factor in determining the overall health of a functional ecological community.

Collectively this thesis provides a considered and comprehensive exploration of *P. undulatum* and its role as a native species with the characteristics of an invader. Using *P. undulatum* as a study species this work develops our understanding of what constitutes a native and invasive species, the importance of these terms, how this has fluctuated over time and how this may change in an increasingly globalised, disturbed and climatically uncertain world.

## Declaration

I hereby declare that this thesis contains no material which has been accepted for the award of any other degree or diploma at any university or equivalent institution and that, to the best of my knowledge and belief, this thesis contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

This thesis includes one original publication submitted in a peer review journal, one chapter accepted for publication and another under review. The focus of the thesis is to explore the invasive species process, management and subsequent restoration. Concepts, data collection, field work and write up were all the principal responsibility of myself, the student, working within the School of Biological Sciences under the supervision of Prof. Ros Gleadow, Assoc Prof. Martin Burd and Doc. Susanna Venn.

The inclusion of co-authors reflects the fact that the work came from active collaboration between researchers and acknowledges input into team-based research.

In the case of chapters 2, 3 and 4 my contribution to the work is as follows

<b>Thesis Chapter</b>	<b>Publication Title</b>	<b>Status</b>	<b>Nature and % of student contribution</b>	<b>Co-authors</b>	<b>Co-authors, Monash student Y/N</b>
2	Reproductive investment across native and invasive regions in a range expanding gynodioecious tree	Submitted ( <i>Biological Invasions</i> )	90%	Ros Gleadow Martin Burd Susanna Venn	N

3	Integrating the Passenger-Driver hypothesis and plant community functional traits to the restoration of lands degraded by invasive trees	Published ( <i>Forest Ecology and Management</i> )	90%	Ros Gleadow Martin Burd Susanna Venn	N
4	Response of south eastern Australian bird communities following invasive tree removal	Under review ( <i>Biological Invasions</i> )	90%	Ros Gleadow Martin Burd Susanna Venn	N

I have re-numbered sections of submitted or published papers in order to generate a consistent presentation within the thesis.

Student signature:

Date: 12.05.2019

The undersigned hereby certify that the above declaration correctly reflects the nature and extent of the student's and co-authors' contributions to this work. In instances where I am not the responsible author I have consulted with the responsible author to agree on the respective contributions of the authors.

Main Supervisor signature:

Date: 12.05.2019

## **Publications, conferences and talks during enrolment**

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## **1.0 Invasive plants across Australia and the story of *Pittosporum undulatum***

### **1.1 Introduction**

This review aims to provide a broad background into the history, impacts, and management of invasive flora across Australia. With examples we discuss how perceptions towards invasive species have changed over time, along with the pressures faced by land managers in an increasingly globalised and changing world. The particular and complex role of invasive woody plants is examined, using *Pittosporum undulatum* as a case study; an Australian native tree species that illustrates the challenges with labelling a species as native or invasive, and how this influences corresponding perceptions and management.

#### **1.1.1 Invasive species in Australia**

Australia is widely impacted by invasive species, which ultimately change biodiversity levels, habitat values, disturbance regimes and ecosystem function on a variety of scales. Following the destruction of habitats, the introduction of invasive species are considered to be the second leading cause of species extinction globally (Bellard et al. 2016; Wilcove et al. 1998), with Australia and New Zealand having lost the most native species to invasions over time (Bellard et al. 2016). Over the last 200 years, approximately 10% of Australia's 273 endemic terrestrial mammals have been lost to extinction, with a further 21% now considered threatened (Woinarski et al. 2015). Native plants are also impacted by invasive species, with four plants species now extinct and many more threatened as a result of pressure from invasive plants (Groves & Willis 1999; Woinarski et al. 2015). More broadly, invasive species such as Bitou bush and boneseed (*Chrysanthemoides monilifera*) are capable of threatening whole ecological vegetation classes, and changing the functioning of ecosystems (Adair et al. 2012). Bitou bush and boneseed invasions characteristically result in reduced indigenous seedling germination, growth and recruitment, along with changes

in bird and invertebrate communities (Adair et al. 2012). Invasive species are also able to change nutrient cycling and alter disturbance regimes. Buffel grass (*Cenchrus Ciliaris*) is an invasive grass species that establishes within watercourses throughout central Australia. Buffel grass is capable of producing 2-3 times more flammable material relative to native grasses, and has thereby altered the function of water courses acting as a traditional barrier to fires, to a potential fire corridor (D'Antonio & Vitousek 1992; Miller et al. 2010). Of the 3207 introduced species that have been naturalised within Australia, approximately 500 species and taxa are now listed as noxious weeds (Invasive Plants and Animals Committee 2016). It is estimated that the cost of direct economic losses, combined with the cost of invasive flora and fauna management across Australia was \$13.6 Billion in the financial year 2011-2012 (Hoffmann & Broadhurst 2016). The annual cost of weeds on the Australian agricultural industry alone was estimated at approximately \$2.4 billion (Pimentel et al. 2001). More broadly, the cost of invasive flora management across Australia is suggested to be between \$3.5-4.5 billion (Hoffmann & Broadhurst 2016; Sinden et al. 2004). Data availability, inconsistency in how costs are valued, and difficulty in effectively attributing an economic value to reductions in ecosystem services, makes calculating a true economic impact of invasive species problematic. Therefore the figures presented above are likely to underestimate the true cost (Hoffmann & Broadhurst 2016; Pimentel et al. 2001; Sinden et al. 2004). Collectively these examples underline the strong impact invasive species have across Australia.

### ***1.1.2 Perceptions of invasive plant species in Australia***

The terms used to describe invasive species are influenced by their perceived level of environmental, economic and human health impacts (Beck et al. 2008; Warren et al. 2017). As a result, some of the language used to describe invaders and their management, such as militaristic metaphors and xenophobic expressions, have been criticised by some for being unnecessarily emotive (Bonanno 2016; Davis et al. 2011; Larson 2005; Richardson et al. 2000; Warren et al. 2017). Discussion surrounding basic nomenclature of invasive species has also been divisive. Confusion has surrounded the use of certain terms defining the status of a species (Colautti & Macssa 2004; Richardson et al. 2000). For example, the word “naturalised” refers to a

new species that has established viable, self-perpetuating populations capable of widespread dispersal. Within the literature naturalisation has incorrectly been used as a synonym for invasive (Richardson et al. 2000). Here we follow the direction of Richardson et al. (2000) to define invasive plants as naturalised alien species that reproduce in very large numbers with the capacity to disperse over large distances. Furthermore, a weed is a plant, regardless of origin, that grows in an unwanted location, often with a harmful economic or environmental impact (Richardson et al. 2000).

### ***1.1.3 A history of invasion***

The historical context of Australia's invasive species relates strongly to the cultural perceptions towards how our environment is valued (Kanowski 2017). Prior to the arrival of Europeans, Australia's Indigenous peoples managed the landscape. Through a combination of hunting, gathering, agriculture and use of firestick farming, Indigenous peoples developed a culture of stewardship over the land, and over time were able to directly influence and change Australian ecosystems (Balme & Beck 1996; Bird et al. 2008; Bowman 1998; Flannery 2002; Gammage 2011; McNiven et al. 2012; Pascoe 2014). European colonialists had a starkly different view of the Australian environment. Through the importation of plants and animals for agricultural, economic and sporting purposes, along with those of acclimatisation societies (Cook & Dias 2006; Dunlap 1997; Osborne 2000), Australian ecosystems have been dramatically altered in a relatively short space of time. On a continent with proportionally little forest cover, as much as 50% of Australia's forests have subsequently been cleared or severally modified (Bradshaw 2012; Kanowski 2017).

Along with changes to the species comprising Australian plant and animal communities, societal perceptions towards native and non-native flora and fauna have also changed over time. For example, blackberry *Rubus fruticosus aggregate* was introduced to Australia in the mid 1800's for fruit production. Today blackberry is considered one of the nations most serious invaders, infesting approximately 8.8 million hectares of land, resulting in \$103 million in annual control and production losses (Invasive Plants and Animals Committee 2016).

In contrast to the perceptions of early Europeans, a view amongst some in contemporary society is that native and introduced plants are “good” and “bad” respectively (Bonanno 2016; Brown & Sax 2004; Davis et al. 2011; Selge et al. 2011; Simberloff 2003; Warren 2007). However, as our ecological understanding continues to develop, we have increasingly seen that a starkly contrasting, “good” and “bad” philosophy towards a species based solely upon its origin is not always suitable (Bonanno 2016; Brown & Sax 2004; Davis et al. 2011; Selge et al. 2011; Simberloff 2003; Warren 2007). For example, despite being a high threat weed species that negatively influences biodiversity, in some instances blackberry is also known to provide key habitat for small mammals including the endangered southern brown bandicoot *Isodon obesulus* (Packer et al. 2016). Thus, context around a species and its capacity to influence the richness, density, structure and function of a community is likely to be more relevant than its origin alone in determining the “goodness” of a species (Bonanno 2016; Davis et al. 2011; Rodriguez 2006).

#### **1.1.4 Invasion Pathways**

Introduced species must overcome a series of barriers/filters that ultimately govern the species present within a community (Booth & Swanton 2002; Blackburn et al. 2011; Richardson et al. 2000). Human disturbance factors, globalisation and a changing climate have altered many of these barriers to invasion. Arguably the most extreme change has been the increase of propagule pressure as a result of increased trade (Hulme 2009; Lockwood et al. 2005; Simberloff 2009). This process has helped to assist species overcome dispersal barriers. A range of Australian instances describe the introduction of previously isolated species, either deliberately or without intention. For example, the prickly pear (*Opuntia* spp.) was introduced to Australia from South America upon the arrival of the British first fleet, with the intentions of developing a cochineal dye industry (Freeman 1992). Following its introduction the pear went on to invade an estimated 250,000 hectares of land across Queensland and northern New South Wales, before a biological control program was instigated (Freeman 1992). A changing world, particularly with reference to climate change has the capacity to influence all stages of the invasion process, as described by Walther et al. (2009). Despite being difficult to predict, the geographical range of some species is changing,

which may enable them to spread to regions where abiotic conditions were previous restrictive (Bradley et al. 2010; Hellmann et al. 2008).

Invasive trees and shrubs are some of the species best equipped to take advantage of our changing world. Despite the important role they play, until recently trees and shrubs were not considered prominent among the list of global invasive species (Richardson et al. 2014; Rundel et al. 2014).. However, the threat posed by woody invaders has been widely publicised over the past decade (Richardson & Rejmánek 2011; Richardson et al. 2014; Rundel et al. 2014). Today, woody plants make up approximately 20% of the most well studied invasive species, and of the global top 100 listed invasive flora and fauna, 21 are woody plants (Lowe et al. 2000; Richardson & Rejmánek 2011). Tree and shrub invasions have the capacity to create novel ecosystems. By altering nutrient cycling and the hydrology of a natural system, a single invasive species can influence the surrounding biodiversity, and habitat structure, with the potential to modify disturbance regimes (Richardson et al. 2014; Rundel et al. 2014; Saintilan & Rogers 2015).

Responding to human disturbances such as urbanisation, agriculture, forestry, and horticulture, non-native species can establish within predominantly native remnant environments (Pemberton & Liu 2009; Richardson et al. 2014). These species are often characterised as weeds. These same disturbance factors can also enable the density and distribution range of native species to expand, resulting in altered species compositions, and changed ecosystem process (Carey et al. 2012a; Davis et al. 2011; Nackley et al. 2017; Simberloff et al. 2012). The line between exotic invaders and problematic native species is likely to become further blurred as the distribution range of species change, following rapid global climate change (Webber & Scott 2012). A well-studied example of a native tree species that has undergone intense demographic changes over the past two centuries is *Pittosporum undulatum*. Despite its prominence in the literature no comprehensive review of the species has been compiled to date. Below we go into detail about the species, its history, ecology, and reproductive habit, and how these factors have interacted with human changes to the environment, enabling the species to become a weed (Gleadow & Ashton 1981; Richardson et al. 2000).



## **1.2 *Pittosporum Undulatum*: the native invader**

### **1.2.1 *Phylogeography***

The name *Pittosporum* derives from the Greek *pittos* meaning pitch, and *sporos* meaning seeds, in reference to the black mucus coated seeds characteristically found within *Pittosporum* fruits (Bacon et al. 2011; Cayzer et al. 2000). *Pittosporum undulatum* is one of approximately 200-240 flowering species found within the Pittosporaceae family (Bacon et al. 2011; Chandler et al. 2007). Seven of the nine genera comprising the family are thought to be endemic to Australia. In contrast, species of the genera *Pittosporum* are found globally, across tropical, arid and temperate regions, with species located in Africa, across Asia, Australia and as far east as Hawaii (Bacon et al. 2011; Chandler et al. 2007; Cooper 1956). A comprehensive review of the *Pittosporum* genus suggests in excess of 100 species can be found within the group, with 20 considered native to Australia (Cayzer et al. 2000; Chandler et al. 2007). Endemism is characteristic of this group.

### **1.2.2 *Distribution and Ecology***

*Pittosporum undulatum* is the most commonly grown species within the *Pittosporum* genus (Cayzer et al. 2000). *Pittosporum undulatum* is a tree species considered native to temperate rainforests of coastal southeastern Australia (Gleadow & Ashton 1981). The species range extends along the coast, east of the Great Dividing Range from subtropical regions around Brisbane in Queensland, to the cool temperate areas of western Gippsland in Victoria (Cayzer et al. 2000; Cooper 1956; Gleadow & Ashton 1981). Native populations of *P. undulatum* are thought to spread as far as 280km inland across New South Wales, but are restricted to 120km inland, and altitudes of below approximately 400m across Southern Victoria (Gleadow & Ashton 1981). *Pittosporum undulatum* is replaced in drier monsoonal areas in Northern Australia by velvet pittosporum (*Pittosporum venulosum*) (Cayzer et al. 2000) and is more commonly found in wet sclerophyll forests of New South Wales and Victoria (Gleadow & Ashton 1981).

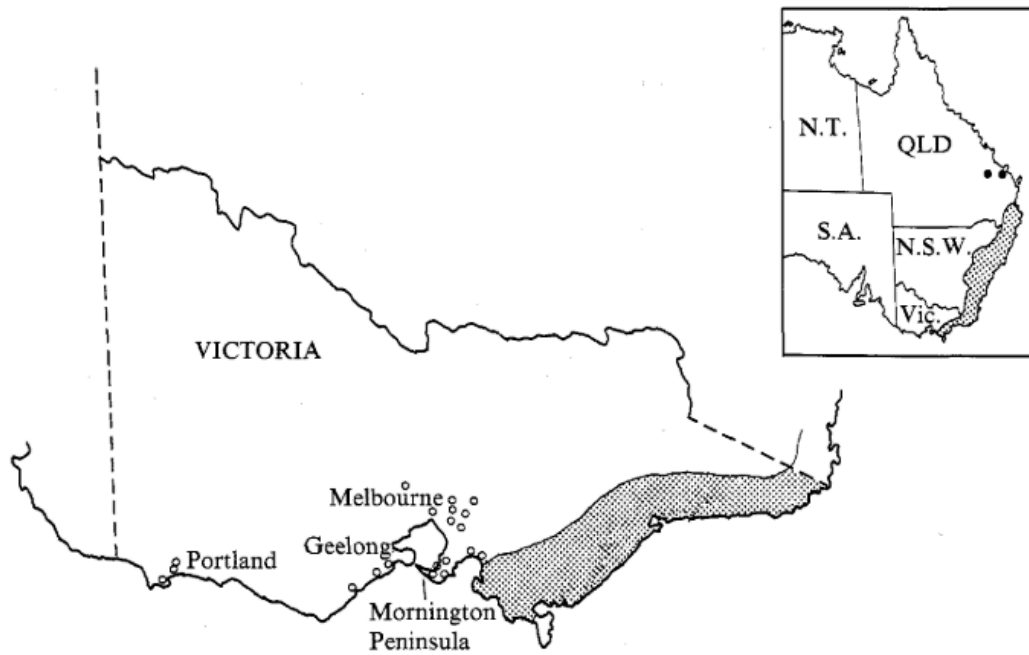


Fig 1.1. Extract from Gleadow and Ashton 1981. Distribution of *Pittosporum undulatum*. Native distribution shown in grey, following Cooper 1956. Circles indicate invaded areas in 1981.

Several reports have suggested that *P. undulatum* displays the characteristics of an invasive species (Gleadow & Ashton 1981; Goodland & Healey 1996; Hortal et al. 2010; Mullett & Simmons 1995). Presently, *P. undulatum* is considered invasive within many regions across Australia as well as Lord Howe Island and Norfolk Island (Eurobodalla Council 2017; Mornington Peninsula Shire et al. 2012; Yarra Ranges 2017). *Pittosporum undulatum* has also become a major invader globally, with invasive populations in New Zealand, Portugal, Jamaica, Hawaii, and is an emergent weed in South Africa (Gleadow & Ashton 1981; Goodland & Healey 1996; Hortal et al. 2010; Lourenço et al. 2011; Mokotjomela et al. 2013). *Pittosporum undulatum* is known to establish quickly after disturbance (Bellingham et al. 2005; Rose & Fairweather 1997), although it may also invade at undisturbed locations (Gleadow & Ashton 1981; Gleadow & Walker 2014; Rose & Fairweather 1997). Once established, mature trees can reach heights of 8-30m (Mullett 2001). Individuals form dense canopies, shading out the undergrowth and reducing structural diversity, floristic composition and the integrity of ecological systems (Gleadow & Ashton 1981; Mullett 2001).

*Pittosporum undulatum* has been observed as a shrub of 2-3 m in harsher and particularly drier locations, whilst in favourable conditions has been reported to reach heights of between 10-30 m (Cooper 1956; Gleadow & Ashton 1981; Mullett & Simmons 1995). In open environments *P. undulatum* is often observed to produce several branches at the base of the plant in addition to the main trunk (Cooper 1956). Borges Silva et al. (2017) examined growth ring variation in *P. undulatum* populations across the Azores, Portugal. Findings showed a mean annual diameter growth rate of 0.38 cm/year, with individuals requiring an average of 8 years to reach breast height (Borges Silva et al. 2017). Works conducted in Victoria, Australia further suggest a strong correlation between tree age and circumference (Gleadow & Walker 2014). Seedling leaf form is entire and lanceolate-oblong in shape, whilst adult leaves are entire and elliptical-oblong in shape (Cooper 1956). Adult leaves reach a size of 6-16 cm in length, and 1.5-5 cm in width, with colouration dark green on top and paler below (Cooper 1956). Stem growth begins in late winter/early spring with the expansion of closed buds (Gleadow & Ashton 1981). Provided the right growing conditions this process can occur again in early/mid summer. Apical dominance is weak, resulting in a bushy habit (Cayzer et al. 2000; Gleadow & Ashton 1981).

As the distribution of *P. undulatum* varies widely, so to do the environmental conditions under which it exists. Though able to endure some degree of shade, *P. undulatum* is not considered a shade tolerate plant, but instead has the capacity to exploit high light environments (Gleadow et al. 1983). *Pittosporum undulatum* seedlings were considered very drought tolerant when grown under low temperatures and in deep shade. Recovery of *P. undulatum* seedlings exposed to water stress, together with a thin leaf cuticle and greater investment in security over efficiency for xylem water transport are characteristics of a drought tolerant, rather than a drought avoiding species (Gleadow & Rowan 1982; Longui et al. 2011). However, growth rates, plant dry weight and survival have been show to drop significantly when water stressed plants were exposed to both higher photon flux densities and temperatures (Gleadow 1982; Gleadow & Rowan 1982). The survival rate of *P. undulatum* seedlings established in exposed sites has been observed at less than 5 %, relative to

the 75 % of plants grown under an established *P. undulatum* canopy (Gleadow 1982; Gleadow & Narayan 2007). These results suggest that *P. undulatum* has the capacity to withstand high light levels and water stress, but may struggle with a combination of both. Though not tested specifically for *P. undulatum*, the positive link between increased soil nutrient resources in urban bushlands and the spread of invasive woody weeds, including *P. undulatum* has been made (King & Buckney 2002; Leishman et al. 2004; Rose & Fairweather 1997).

### **1.2.3 Reproduction**

*Pittosporum undulatum* individuals reach reproductive maturity at 4-5 years old (Gleadow & Ashton 1981). Flowering takes place during spring and under suitable conditions again in autumn (Gleadow & Ashton 1981). Flowering sees a profuse display of creamy white, scented flowers emerge (Gleadow & Walker 2014; Mullett 1996). Flowering structure is terminal, with 4-15 flowers completing each inflorescence (Cooper 1956; Goodland & Healey 1996). Petals average a length of 11-17 mm, with stamens 5-11 mm long (Cooper 1956; Goodland & Healey 1996). Flowers are insect pollinated (Goodland & Healey 1996). *Pittosporum undulatum* produce small round orange fruits with a mean diameter of 10-14 mm. Dehiscent fruits are seen in autumn, with each fruit producing on average between 12 and 30 red seeds coated in a thick, dark, sticky mucilage (Cooper 1956; Gleadow & Ashton 1981). Seeds are stored predominantly in the canopy, with poor viability if buried to a depth of more than 4cm within the soil (Gleadow & Rowan 1982; Gleadow & Narayan 2007).

*Pittosporum undulatum* has been described as a dioecious species, with populations comprising male and female individuals (Gleadow & Walker 2014). However, some observers have noted variation to this pattern, with functionally female flowers on what appears to be predominantly male trees (Mullett 1996), and in other cases populations instead comprised males and hermaphrodites (Goodland & Healey 1996; Cooper 1956). The study undertaken by Gleadow and Walker (2014) showed that males comprised approximately 60-70 % of sampled *P. undulatum* populations. Current management practises focus on removing females from invasive populations,

as these are the individuals that promote further colonising spread via seed dispersal (Gleadow & Walker 2014). This strategy has the benefit of reducing the workload and spreading limited management resources, but requires further examination, and will be addressed in chapter two of this thesis.

#### **1.2.4 Dispersal and invasion**

A species must first overcome dispersal constraints in order to establish as an invader within a new location. Direct human dispersal removed this constraint for *P. undulatum*. As a profuse flowering large shrub or tree producing deep shade, sweet scented flowers, and bird attaching fruits, *P. undulatum* has long been a valued as an ornamental and hedging plant by gardeners, farmers, and those in the plant nursery industry (Cooper 1956; Lourenço et al. 2011; Gleadow & Walker 2014; Goodland & Healey 1996). Subsequently *P. undulatum* seed was spread across Australia and the globe. In addition to humans, bird species are considered the primary vector for *P. undulatum* seed (Gleadow 1982). Key vector species include Silvereye (*Zosterops lateralis*), Pied Currawong (*Strepera graculina*), Satin Bowerbird (*Ptilonorhynchus violaceus*), Red Whiskered Bulbus (*Pycnonotus jocosus*) and in particular the European Blackbird (*Turdus merula*) (Gleadow 1982; Mullett 1999). The typical pattern of *P. undulatum* establishment beneath larger trees fits this mode of dispersal, as it is from the trees branches above that resting birds defecate and deliver the seed (Gleadow 1982).

Because *P. undulatum* plants are capable of fruiting twice a year under favourable conditions, fruit may be available to frugivorous birds outside of autumn, when fruiting peaks for fleshy fruited species across temperate regions (French 1992). Whilst seasonal variations make fleshy fruit availability scarce at certain times, *P. undulatum* may represent a relatively persistent food resource for these birds, which as a result may promote the species dispersal (Stefan Rose 1997). It has been suggested mammals such as possums, foxes, fruit bats and pigs may also play a role as dispersal agents, though no studies on this have been conducted to date (Goodland & Healey 1996; Stefan Rose 1997).

### 1.2.5 Invasion impacts

The negative impacts of *P. undulatum* range expansion have long been recognised. Gleadow and Ashton (1981) first reported on the invasive capacity of *P. undulatum* in bushland across peri-urban Melbourne, and the corresponding reduction of species richness and diversity. From that time onwards several authors have noted these same impacts, both locally and internationally (Lourenço et al. 2011; Mullett 2001; O’Leary et al. 2018; Rose & Fairweather 1997). The dense shade cast as the *P. undulatum* canopy coalesces dramatically reduces light penetration to the ground, to as little as 11 % and in some case 2 % of full daylight (Gleadow & Ashton 1981; Gleadow & Narayan 2007). This reduction in light levels reduces a vital resource for understorey species, which subsequently are outcompeted (Gleadow & Ashton 1981; Mullett 2001; O’Leary et al. 2018). A dramatic reduction in species richness is often coupled with a reduction in structural function and ecosystem services (O’Leary et al. 2018). The leaves of *P. undulatum* plants are known to contain a range of saponins, oils and resins (Chandler et al. 2007; Gleadow & Ashton 1981). The presence of these compounds may have allelopathic affects on competitors (Gleadow & Ashton 1981). This is an area of study that requires further exploration.

*Pittosporum bicolor* is another *Pittosporum* species native to southeastern Australia. *Pittosporum bicolor* is most commonly found throughout higher altitude regions of Victoria and Tasmania (Cooper 1956). The range expansion of *P. undulatum* means that the distribution of the two species now overlap (Cayzer et al. 2000). As a result the two species are now hybridizing, particularly throughout the Dandenong Ranges and South Gippsland (Gleadow & Ashton 1981; Cayzer et al. 2000). Given the increased genetic material available to these hybrids, the potential exists for greater evolutionary capacity to adapt to their surrounding environment, though this is not guaranteed (Rius & Darling 2014). Little is known of the ecology of the hybrid population, including if they are reproductively viable. Further study is suggested for these hybrid populations.

### 1.2.6 Perceptions, management and climate change

*Pittosporum undulatum* typifies the spectrum of perceptions people can hold towards a plant, and particularly a non-native species (Dickie et al. 2014). Indigenous aboriginal communities of Gippsland in eastern Victoria valued the medicinal properties of *P. undulatum* and its capacity to treat insect bites (Monash School of Biological Sciences 2010). In more contemporary times *P. undulatum* has been viewed positively by some as a native species under the lens of its ecological role as a coloniser within warm and cool temperate rainforests (Melick & Ashton 1991). In contrast, many people view *P. undulatum* poorly across the area of its range expansion, with a focus placed upon the resulting loss of species richness and ecological function (Eurobodalla Council 2017; Gleadow & Ashton 1981; Walker 2015; Yarra Ranges 2017). Complicating this apparent positive - negative dichotomy are examples such as the locally endangered Powerful Owl (*Ninox strenua*), which is known to roost and feed within the *P. undulatum* canopy, both within and beyond the species native range (McNabb & McNabb 2011). Competing perceptions therefore exists between people who view the tree as an invader, and therefore seek its removal, and those keen to see the species retained as a means of supporting the owl population (Gleadow & Walker 2014; McNabb & McNabb 2011; Nilar et al. 2019).

Through invasive trees are often viewed poorly within contemporary society due to the reduction of floristic richness and diversity often observed in their presence, invasive tree species can provide valuable structure to an environment, in addition to ecosystem services, nutrient cycling, erosion control and economic development (Dickie et al. 2014; Pejchar & Mooney 2009; Richardson 1998). When it comes to the management of invasive trees, a clear felling approach is not always the best option (Finn et al. 2009; Kirkman et al. 2007). A slower, and more sensitive approach, which may potentially be resource intensive, can provide a more ecologically effective way to manage woody weeds (Kirkman et al. 2007; McNabb & McNabb 2011). A sensitive approach when removing invasive *P. undulatum* populations is likely to aid the management of species such as the Powerful Owl (McNabb & McNabb 2011).

As with any invasive species control program, regardless of the approach, follow up is needed to ensure the programs effectiveness (Funk et al. 2008; Suding 2011; Wortley et al. 2013). Though increasingly common this aspect of invasive plant management remains limited. O’Leary et al. (2018) have recently completed a studying monitoring the impacts, outcomes and effectiveness of *P. undulatum* control programs overtime. Results indicate *P. undulatum* control has had a positive overall effect on the local environment and greater depth on these results is provided throughout chapter three of this thesis.

The aim of restoration programs is to remove invasive species in the hope that native ones will take their place, to ultimately re-establish a natural and functioning ecosystem (McDonald et al. 2016). Though this aim is certainly admirable, for a variety of reasons, including the advance of global change through globalisation, urbanisations and human induced climate change, in many instances this aspiration may be impossible to achieve (Ehrenfeld 2000; Hobbs 2007). International trade and human movement have rapidly increased the rate of exotic species introductions to natural environments (Hulme 2009; Simberloff 2009). Even as invasive plants are removed, high propagule pressure can enable the re-establishment of invasive populations (Simberloff 2009). Furthermore, as global temperatures increase, climates change and major disturbances such as cyclones and droughts intensify, the distribution of plants is changing (Thomas 2010; Webber & Scott 2012). As this happens the suitable environmental envelope for known invasive species is changing (Thomas 2010; Webber & Scott 2012). Species shifting their range as a result of climate change do not fit the traditional mould of a native or invasive plant, and new terms are already being suggested for them (Webber & Scott 2012). If the goal of restoration ecology is to restore functioning plant and animal communities, any management actions will need to consider these varied anthropogenic influences (Ehrenfeld 2000; Hobbs 2007). The role of invasive *P. undulatum* populations in degrading plant and animal communities will also need to be considered with respect to these broader anthropogenic management issues. Furthermore, species such as *P. undulatum* that are able to grow quickly on marginal lands may need to be examined for their benefits as much as the problems they present (Borges Silva et al. 2017; Jakubowski et al. 2010).



As the world continues to increase in disturbance and complexity, the lessons learned from case studies will become increasingly valued. The history of *P. undulatum* and its range expansion presents a clear example applicable to the management of many other ranges expanding species. The ornamental introduction of *P. undulatum*, its ongoing distribution through nurseries, responses to disturbance and avian dispersal means that management actions for *P. undulatum* will be directly relatable to many similar species. The strength of *P. undulatum* as a case study is likely to grow as the prominence of native species becoming problematic becomes more common (Carey et al. 2012a; Davis et al. 2011; Nackley et al. 2017; Simberloff et al. 2012).

### ***1.2.7 Conclusion and future perspectives***

Invasive species, perceptions towards them and their management have played a strong part in the history of Australia. Here we have described how these factors have changed over time, leaving many parts of Australia's ecosystems highly degraded. Judgements of the value of invasive species will continue to evolve, hopefully with less emphasis placed upon the species origin, and more upon the potential ecological impacts as well as benefits they may provide.

Species such as *P. undulatum* typify the degree of change that can occur following human disturbances. Understanding the patterns of range expansion and population growth allows us to observe the same patterns in other species, leading to improved management outcomes. Clearly major societal; economic and environmental adjustments are needed to properly manage the threat of invasive species such as *P. undulatum*. The fast pace of global change means that these adjustments will not only need to be well considered, monitored and evaluated, but done so rapidly.

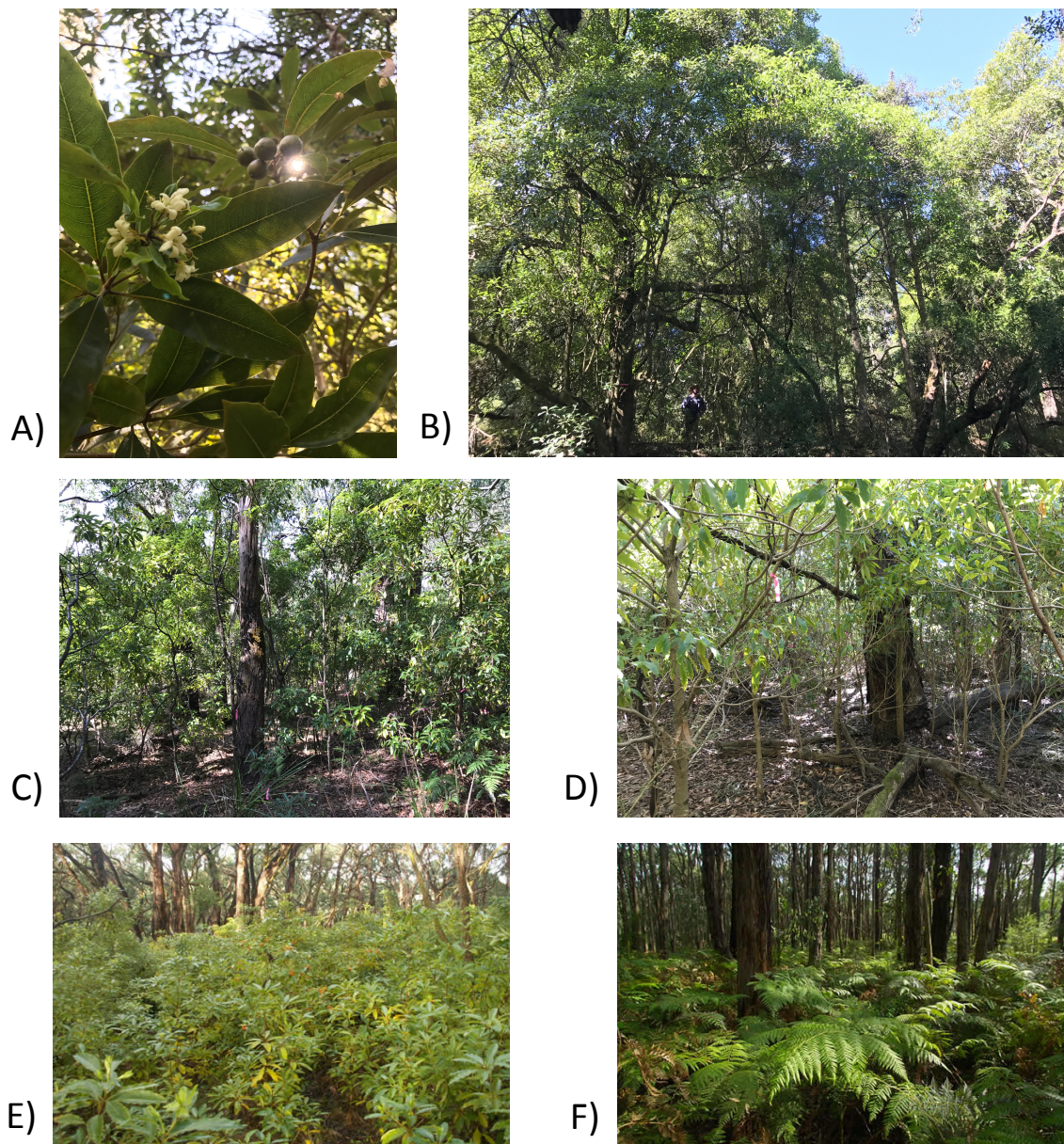


Fig.1.2. Images of *Pittosporum undulatum*. A) Flowers and unripened fruit of *P. undulatum* B) *Pittosporum undulatum* population at Morwell National Park, note person at base of tree for size reference C) Depauperate undergrowth beneath *P. undulatum* population at Red Hill D) Depauperate undergrowth beneath *P. undulatum* population at Glenfern Valley Bushlands in Upwey D) Condition of site at Red Hill prior to *P. undulatum* removal works E) Condition of site At Red Hill approximately 10 years post *P. undulatum* removal works. Refer to table 2.2 for further site details.

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## **2.0 Reproductive investment across native and invasive regions in a range expanding gynodioecious tree**

This chapter has been submitted to the journal *Biological Invasions*

### **2.1 Abstract**

The success of invasive species relies heavily on the production, dispersal and genetic composition of propagules. Therefore, for range expanding species, breeding strategy and level of reproductive investment will strongly influence their capacity to establish and invade new areas. We examined the sexual breeding strategy and fruit output of *Pittosporum undulatum*, a dioecious tree species native to coastal south eastern Australia that is frequently viewed as an invader beyond its native range, both within Australia and globally. Observations of population floral sex ratios and fruit investment were contrasted between populations located within the species native and invasive ranges. Variability of resource availability was also considered, as this is a factor that can influence reproduction and gender ratios. Observations contrast with the findings of previous studies, with *P. undulatum* populations sampled consisting primarily of females and hermaphroditic individuals rather than females and males. Main findings suggest that though hermaphrodites are capable of fruiting, a significantly greater proportion of females produced fruit and in significantly higher quantity. Trends were consistent regardless of a populations native or invasive origin. The overall level of reproductive investment was similar across native and invasive populations, however invaders appeared to produce seeds with significantly greater mass. Results are discussed with respect to established theory focused on breeding strategies in invasive species.

### **2.2 Introduction**

The success of invasive plant species relies heavily on the production, dispersal and genetic composition of propagules (Barrett et al. 2008). Relative to their native range, invasive species commonly show increases in fecundity (Burns et al. 2013) and self-compatibility (Petanidou et al. 2012). Thus, a focus on the mating systems and

reproductive allocations of invasive species may improve our understanding of the processes that promote the establishment and expansion of invasive populations (Barrett et al. 2008; Burns et al. 2013) and potentially guide control strategies (Lambrinos 2004; Barrett et al. 2008; Barrett 2011; Albert et al. 2015).

Baker (1955) considered the role of self-compatibility in the colonisation of islands. Due to the haphazard nature of long distance dispersal of propagules to islands, the likelihood of establishing within close proximity to a potential mate is thought to be low. Thus, hermaphroditic species with a self-compatible breeding system should predominate among island colonisers, as they have the capacity for uniparental reproduction through self-pollination (Grossenbacher et al. 2017). The same concept can be extended to an invasion front, where individuals may be establishing in relative isolation (Pannell et al. 2015). Baker's Law, as this idea is now referred (Pannell et al. 2015), implies, therefore, that in invasive plants self-compatible hermaphroditic reproductive systems might predominate, especially at the margins of an invasive range.

Any initial advantage that hermaphrodites may have in the founding of new populations need not persist as a population grows (Eppley & Pannell 2007a). Metapopulation dynamics of an invasion may mean that female plants or hermaphroditic plants that have female-biased sexual allocation are favoured during the initial phase of population expansion, as females will tend to contribute disproportionately to early population growth (Pannell 1997; Eppley & Pannell 2007b). Only later in consolidated populations with higher population density and female availability would the selective advantage of males and male allocation increase. Consistent with this, high seed fecundity is also a notable features of individuals in invasive populations (Bossdorf et al. 2005; Burns et al. 2013). We might also expect invasive populations to shift their breeding systems toward gynodioecy or toward a greater proportion of female plants in already gynodioecious species (Pannell et al. 2008).

In large, well-established invasive populations, novel resource environments may alter the sex expression of plants or skew flowering sex ratios (Abe et al. 2002).

Individual plants in diclinous species can often change sex expression according to their size or resource status (Freeman et al. 1980; Delph & Wolf 2005), and individuals of monoecious species are known to shift their floral sex ratios in response to resource availability (Freeman et al. 1981). In these cases female expression was almost universally favoured by greater moisture, soil nutrients, or sunlight. Some degree of segregation of male and females along resource gradients has been noted in dioecious species, with males tending to predominate at the drier, poorer end of the gradient and females at more fertile (Freeman et al. 1976; Bierzychudek & Eckhart 1998). If invasive ranges offer a more favourable resource environment, for example through greater resource availability (Davis et al. 2000), we might expect to find a female bias in sex expression or sex occurrence among plants within those invasive populations. Alternatively, if permissive environments allow small plants to survive, and small size favours male expression, invasive populations may show male-biased sex expression or sex ratios (Abe et al. 2002).

It has been recognized that a suite of morphological and life history traits that promote dispersal are likely to be exaggerated at an invasion front because the best dispersers are most likely to arrive at the front, mate with other recent arrivals, and, to the extent that traits are heritable, pass them to the next generation that itself may extend the front (Travis & Dytham 2002; Simmons & Thomas 2004; Phillips et al. 2010; Tabassum & Leishman 2018). This idea, which has been called the “Olympic Village effect” in the context of movement adaptations for animal dispersal (Phillips et al. 2010), suggests that, in plants, seed or fruit traits that affect dispersal and establishment might differ between the native and introduced ranges of an invasive species. Such processes seem to account for lower wing loadings (thus, slower descent and more horizontal dispersal) of the winged seeds in more recently derived populations of Lodgepole Pine (*Pinus contorta*) following its post-glacial range expansion in North America (Cwynar & MacDonald 1987). For species with animal-dispersed fruits, we might expect selection for dispersal ability to favour greater fruit load and higher probability of fruiting (Burns 2002; Naniwadekar et al. 2015). Large seed size, in contrast, is thought to diminish dispersal ability but may enhance competitive ability and stress tolerance during seedling establishment (Turnbull et al. 1999; Westoby et al. 1996).

We explored the reproductive traits of a long-lived woody invader, sweet pittosporum (*Pittosporum undulatum* Vent.), both within and beyond its original range.

*Pittosporum undulatum* is a shade tolerant tree that has many characteristics of an invasive species within and beyond its native range of coastal southeastern Australia (Gleadow & Ashton 1981; Gleadow et al. 1983). Previous studies indicated that individual *P. undulatum* trees can be male, female or hermaphroditic, but how this relates to its environment or invasive status remains unknown (Gleadow & Walker 2014; Goodland & Healey 1996; Mullett 1996). Here we measured the frequency of sexual types in native and non-native populations of *P. undulatum* that varied in density and resource availability. We also assessed the probability of fruit set, fruit load and seed size, traits that affect the ability to reach and establish in novel sites. . Finally, as resource availability may influence sex expression and reproductive output we also measured the concentration of total nitrogen, carbon and the ratio of stable isotopes  $^{13}\text{C} : ^{12}\text{C}$  ( $\delta^{13}\text{C}$ ) in leaves as a proxy for resource status of trees (Freeman et al. 1980; Delph & Wolf 2005; Sofo et al. 2012). Further details on nutrient measurements are supplied in the method section. We hypothesise that *P. undulatum* from invasive populations will 1) have a higher proportion of females than native populations; 2) produce relatively more fruit and 3) have a greater number and/or smaller seeds, relative to native populations

Investigating how the breeding system and sexual expression of invasive species differs between native and novel environments may improve our understanding of pest species and their management (Blossey & Notzold 1995; Keane & Crawley 2002; Hejda et al. 2009; Phillips et al. 2010; Etten et al. 2017). Furthermore, contrasting the performance of species within and beyond their native ranges can be use useful in testing ecological theory, as it provides an initial baseline to examine the response of a species to novel environments (Hierro et al. 2006; Sax et al. 2007; Hierro et al. 2005).

## 2.3 Methods

### 2.3.1 *Pittosporum undulatum*

Present across a range of habitat types, *P. undulatum* Vent. (sweet pittosporum) is most commonly found in wet and temperate rainforests (Gleadow & Ashton 1981). *Pittosporum undulatum* is a shade tolerant tree native to coastal areas of southeastern Australia, stretching from the tropical Gold Coast in Queensland (mean rainfall 1259.2 mm, mean temperature 25.3°C) to Gippsland in South Eastern Victoria (mean rainfall 735.5 mm, mean temperature 20.1°C) (Bom 2019). Though variable, the native range of *P. undulatum* is thought to extend inland approximately 280 km throughout New South Wales, but only around 120 km inland throughout Victoria, where it is limited by altitudes greater than 400 m (Gleadow & Ashton, 1981). See Fig 2.1 for map approximating the species range across Victoria, Australia. The species has been observed to exhibit many characteristics of an invasive species within and beyond its native range (Gleadow et al. 1983; Gleadow & Ashton 1981).

Following British arrival to Australia altered fire regimes, introduced vectors, peri-urban disturbance and horticultural propagation have all contributed to the spread of *P. undulatum* across Australia and the globe (Gleadow & Ashton 1981; Gleadow & Rowan 1982; Gleadow 1982; Gleadow et al. 1983; Gleadow & Narayan 2007; Rose & Fairweather 1997). *Pittosporum undulatum* is treated as an invader throughout many regions of Australia, extensively across mainland southeastern Australia, including the Yarra Ranges and Mornington Peninsula, along with and several surrounding islands (Yarra Ranges 2017; Eurobodalla Council 2017; Mornington Peninsula Shire et al. 2012). *Pittosporum undulatum* has also become a major problem globally, with invasive populations in New Zealand, Portugal, Jamaica, Hawaii, and is an emergent weed in South Africa (Gleadow & Ashton 1981; Goodland & Healey 1996; Lourenço et al. 2011; Hortal et al. 2010; Mokotjomela et al. 2013). *Pittosporum undulatum* is known to establish quickly after disturbance, although it may also become invasive at undisturbed locations (Bellingham et al. 2005; Gleadow & Ashton 1981; Gleadow & Walker 2014; Rose & Fairweather 1997). Once established, mature trees can reach heights of 8-30 m (Mullett 2001). Individuals form dense canopies, shading out the undergrowth and reducing structural



diversity, floristic composition and the integrity of ecological systems (Gleadow & Ashton 1981; Mullett 2001; O’Leary et al. 2018).



Fig. 2.1. *Pittosporum undulatum* native (black), invasive (hatched) and spreading regions (grey) across Victoria, Australia.

### 2.3.2 Site description

We investigated variations in tree density, sex, resource availability and fruit load for native and invasive populations of *P. undulatum*. Seven populations within temperate *Eucalyptus* forests of East Gippsland in southeastern Victoria, Australia, were selected to represent native populations (Table 2.1). A further seven populations across peri-urban areas of Melbourne, in southeastern Australia, were selected to represent invasive populations (Table 2.1).

Table 2.1. Location, elevation, climate data (BOM 2019) and vegetation data (DELWP, 2019) of sampled populations along with population's status as a native or invader to that location. Climate data was taken from the closest available weather station with a history of over 20 years in operation wherever possible. Ecological Vegetation Class (EVC) data was determined through a combination of ground trothing and the use of the Victorian Department of Environment Water Land and Planning (DELWP) Nature Kit.

Site	Origin	Location	Elevation (m)	Mean Annual Rainfall ( mm )	Mean maximum temperature (°C)	Ecological Vegetation Class (EVC)
Morwell National Park 1	Native	Lat:-38.36 Lon:146.40	184	735.5	20.1	Warm temperate rainforest
Morwell National Park 2	Native	Lat:-38.36 Lon:146.40	184	735.5	20.1	Warm temperate rainforest
Lakes Entrance 1	Native	Lat: -37.88 Lon: 147.96	40	713.7	19.1	Limestone box forest
Lakes Entrance 2	Native	Lat: -37.88 Lon: 147.96	40	713.7	19.1	Limestone box forest
Lake Tyers State park	Native	Lat:-37.76 Lon:148.07	89	815.4	19.0	Limestone box forest

Marlo	Native	Lat: -37.79 Lon: 148.55	22	769.1	20.5	Coast banksia woodland
Mallacoota	Native	Lat: -37.56 Lon: 149.76	19	933.9	19.5	Lowland Forest
Red Hill	Invaded	Lat: -38.39 Lon: 145.02	131	900.1	19.2	Herb rich foothill forest
Bittern 1	Invaded	Lat: -38.30 Lon: 145.12	81	719.8	19.2	Lowland forest
Bittern 2	Invaded	Lat: -38.30 Lon: 145.12	81	719.8	19.2	Lowland forest
Upwey 1	Invaded	Lat: -37.90 Lon: 145.31	291	1029.3	19.7	Valley grassey forest
Upwey 2	Invaded	Lat: -37.90 Lon: 145.31	291	1029.3	19.7	Valley grassey forest
Montrose	Invaded	Lat: -37.84 Lon: 145.33	222	944.6	15.2	Grassey dry forest
Silvan	Invaded	Lat: -37.83 Lon: 145.42	293	1200.7	15.6	Lowland Forest

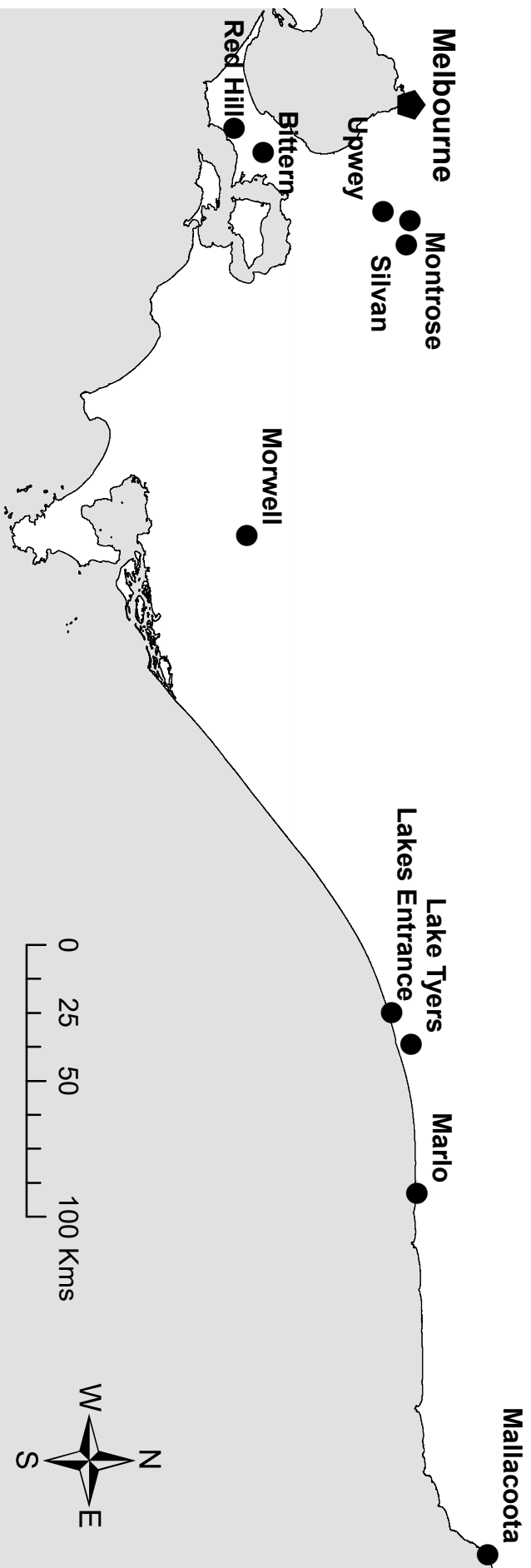


Fig. 2.2. Map of sampling sites across Victoria, Australia. Morwell, Lakes Entrance, Lake Tyers, Mario and Mallacoota are all considered native populations. Remaining locations represent invasive populations.

### **2.3.3 Sex and resource analysis**

Sex expression of individual *P. undulatum* plants was determined throughout September and October of 2016. At each site a 20 m x 20 m quadrat was established, with mature trees growing within the quadrat examined and labelled according to the presence/absence of male and female floral structures. Within each quadrat, five leaves from five randomly selected individuals of each sex were sampled for nutrient analysis, specifically total leaf nitrogen, total leaf carbon and  $\delta^{13}\text{C}$ . Nitrogen is a limiting resource important to growth and reproduction. Increased soil nitrogen availability is generally seen in greater leaf nitrogen stores (Fusaro et al. 2017). Leaf carbon (as an indicator of structural and non structural carbohydrates) can be indicative of sclerophylly and photosynthetic output, respectively (Funk 2013). These attributes can reflect a plants photosynthetic capacity and adaptation to shade. Finally  $\delta^{13}\text{C}$  is an indicator of water use efficiency over the lifetime of the leaf (Retuerto et al, 2000). Drought stress is considered the leading cause of reduced tree growth and productivity in Mediterranean plants and is therefore an important resource consideration when assessing population productivity (Sofa et al. 2012). Leaves were selected from approximately the third stem of a branch. Leaves were dried in an oven for 48 hours at 60 degrees before being ground in a homogenizing tissue mill. Total elemental nitrogen and carbon were measure on finely ground freeze-dried leaf samples using a LECO CNS2000 analyser (Environmental Analysis Laboratory, Southern Cross University, NSW, Australia). After combustion, samples were analysed for carbon isotopes by continuous flow isotope ratio mass spectrometry at the EAL. Carbon isotopes were expressed as  $\delta^{13}\text{C}$  using the PeeDee belemnite standard.

### **2.3.4 Fruit load and seed mass analysis**

From the 1<sup>st</sup> of February to the 20<sup>th</sup> of March 2017, the original sample plots were re-examined, and total fruit loads recorded for individual plants. Plants were ranked and categorised by fruit load intervals of 50, with 0 equating to no fruit observed, 1 equating to 50 fruits or less, 2 equating to 100 fruit or less, up to 11 equating to 500-550 fruits. Mature fruits (orange in colour) were collected from with the canopy.

Seeds from collected fruits were removed from their fruit casing and cleaned of mucilage using tissue paper. All seeds from each fruit were weighed together and the mean seed weight calculated.

### **2.3.5 Statistical Analysis**

All analysis was conducted using the R statistical program through the base package (R Core Team 2017). Variation in mature plant density, proportion of females within populations across native and invasive populations, and differences in seed number and weight across populations were analysed through unpaired  $t$  tests.

Differences between native and invasive sites were compared using equal variance  $t$  tests. Differences between each reproductive type in the proportion of fruiting trees were examined through a generalised linear model. Comparison of the mean fruit load rank between reproductive types across populations was also examined via a generalised linear model.

Linear modelling was conducted to investigate variation in the proportion of individuals producing fruit, and the mean rank of fruit production across sex and origin. Data were arcsine square root and cube root transformed respectively prior to analysis to meet the conditions of normality.

## **2.4 Results**

All populations were comprised of either females or hermaphrodites, with no male trees observed within the study. Native and invasive sites did not differ significantly, either in the mean density of *P. undulatum* trees (Table 2.2 and Table 2.3), or in the proportion of females (Table 2.2 and Table 2.3). No difference in the proportion of fruiting individuals was detected between native and invasive populations (Table 2.3). However, a significant difference in the proportion of fruiting individuals was detected across sexes, with females more frequently found to produce fruit (Table 2.4, Fig. 2.3A). A significant interaction was also observed between plant sex and invasion status, with females in native sites more likely to fruit comparative to

females in invasive populations (Table 2.4, Fig 2.3A). When examining the quantity of fruit produced by each tree, female individuals appeared more likely to produce higher quantities of fruit (Table 2.5, Fig. 2.3B). Though no dissimilarity in  $\delta^{13}\text{C}$  and carbon availability was observed among populations regardless of a populations native or invasive status, marginal significance was found suggesting that higher mean population leaf nitrogen availability corresponded with greater fruit production in females, (slope estimate= 5.540, adjusted  $r^2 = 0.23$ ) (Table 2.6, Fig. 2.4A, 2.4B, 2.4C and Fig. 2.5). The mean mass of seeds produced in invasive populations (0.0062 g) was approximately twice that of seeds from native populations (0.0033 g) (Table 2.7, Fig. 2.6)

Table 2.2. Variation in tree density, sex, mean number of seeds per fruit capsule, mean seed weight and standard deviation across sampled populations. Fruit availability limited sampling for Silvan and Bittern sites.

	Total mature trees in sampled 20 x 20m plot	Number of Females	Mean seed number	Mean seed number (sd)	Mean seed weight (g)	Mean seed weight (sd)
Morwell 1 (Native)	30	10	32	1.82	0.0023	0.0003
Morwell 2 (Native)	8	5	28	3.62	0.002	0.0001
Lakes Entrance 1 (Native)	103	46	33.8	1.30	0.0036	0.0003
Lakes Entrance 2 (Native)	34	16	27.08	3.64	0.0038	0.0001
Lake Tyers (Native)	22	8	25	1.41	0.0029	0.0026
Marlo (Native)	62	34	27.25	4.89	0.0040	0.0014
Mallacoota (Native)	143	71	28.66	5.69	0.0027	0.0006

Red Hill (Invasive)	159	72	27.6	4.00	0.0082	0.0026
Bittern 1 (Invasive)	51	15	N/A			
Bittern 2 (Invasive)	89	44	N/A			
Upwey 1 (Invasive)	30	14	26.84	4.20	0.0071	0.0017
Upwey 2 (Invasive)	29	12	28.73	5.42	0.0066	0.0016
Montrose (Invasive)	84	31	27.92	4.69	0.0028	0.0009
Silvan (Invasive)	27	10	N/A			

Table 2.3. *t* test comparing differences in total tree density, the proportion of females and proportion of fruiting individuals among native and invasive populations of *P. undulatum*.

	T	df	P
Population density	0.368	12	0.718
Proportion of females	- 1.3031	12	0.217
Proportion of Fruiting individuals	-1.8365	10	0.09614



Table 2.4. Generalised linear model comparing variation in the proportion of fruit producing individuals between female and hermaphroditic trees from native and invasive populations. \* Denotes significance at an alpha of 0.05

Effect	Estimate	Std Error	t value	p value
Intercept	0.290	0.066	4.384	<0.01*
Sex	0.582	0.136	4.272	<0.01*
Origin	-0.092	0.126	-0.737	0.469
Sex:Origin	0.526	0.178	2.949	<0.01*

Table 2.5. Linear model comparing variation in ranked fruit production between female and hermaphrodite trees from native and invasive populations. \* Denotes significance at an alpha of 0.05

	Estimate	Std. Error	t value	p value
Intercept	0.441	0.149	2.952	<0.01*
Sex	0.646	0.211	3.056	<0.01*
Origin	- 0.08	0.195	- 0.428	0.672
Origin:Sex	0.389	0.276	1.405	0.175

Table 2.6. Linear model examining the relationship between female fruit rank and population mean leaf nitrogen values.

	Estimate	Std. Error	t value	p value
Intercept	0.144	0.111	1.300	0.222
Leaf nitrogen	5.540	2.642	2.099	0.062

Table 2.7. T test results examining variation in mean seed number and seed weight between native and invasive populations. \* Denotes significance at an alpha of 0.05

	T	df	P
Variation in seed number by origin	-0.167	34.645	0.868
Variation in seed mass by origin	4.7009	24.422	<0.01*

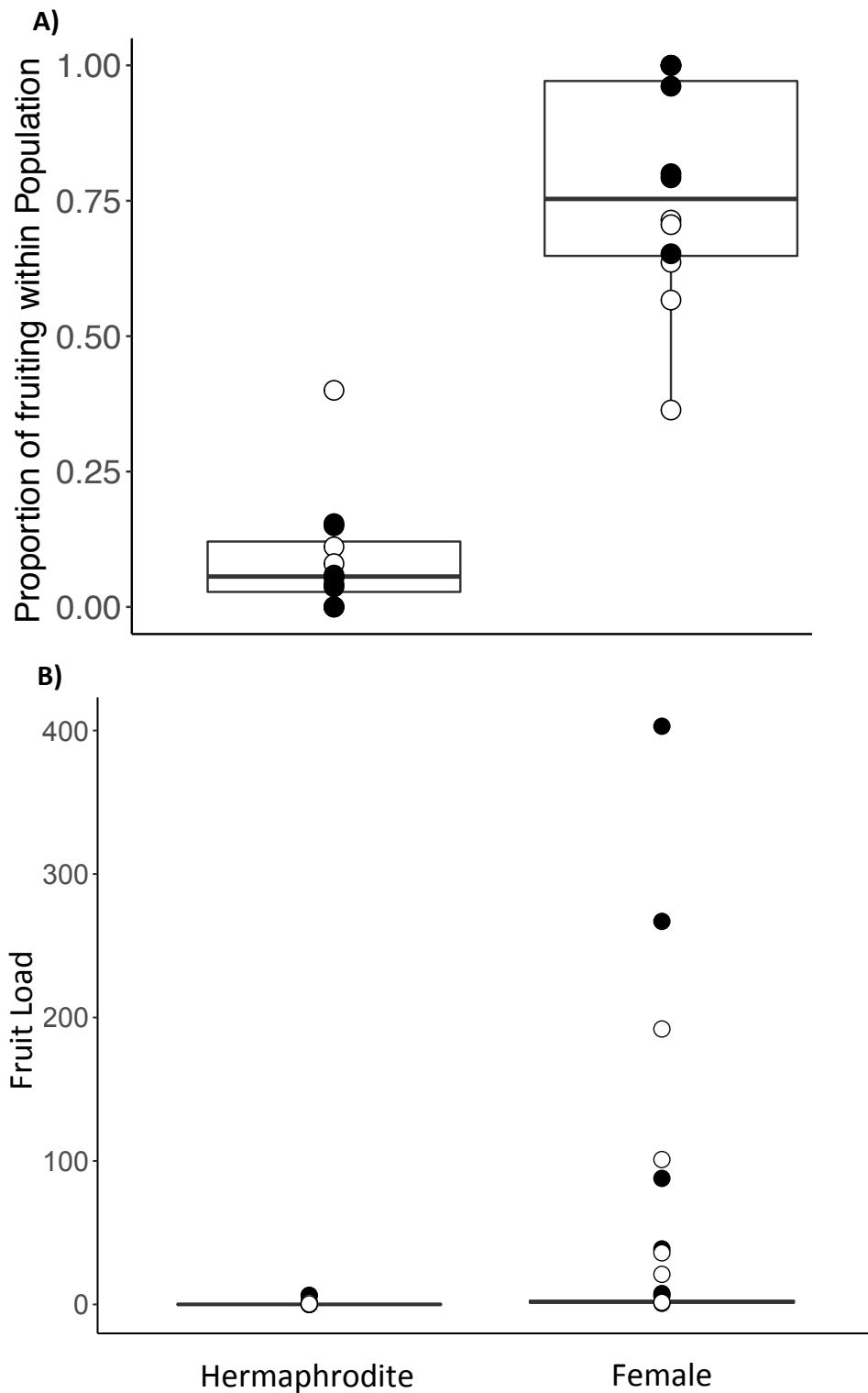


Fig. 2.3. A) Variation in the proportion of female and hermaphrodite individuals fruiting across native and invasive populations. B) Variation in the mean fruit rank for females and hermaphrodites within native and invasive populations. Native populations = black circles, invasive populations = white circles. For full statistical analysis see tables 2.4 and 2.5.

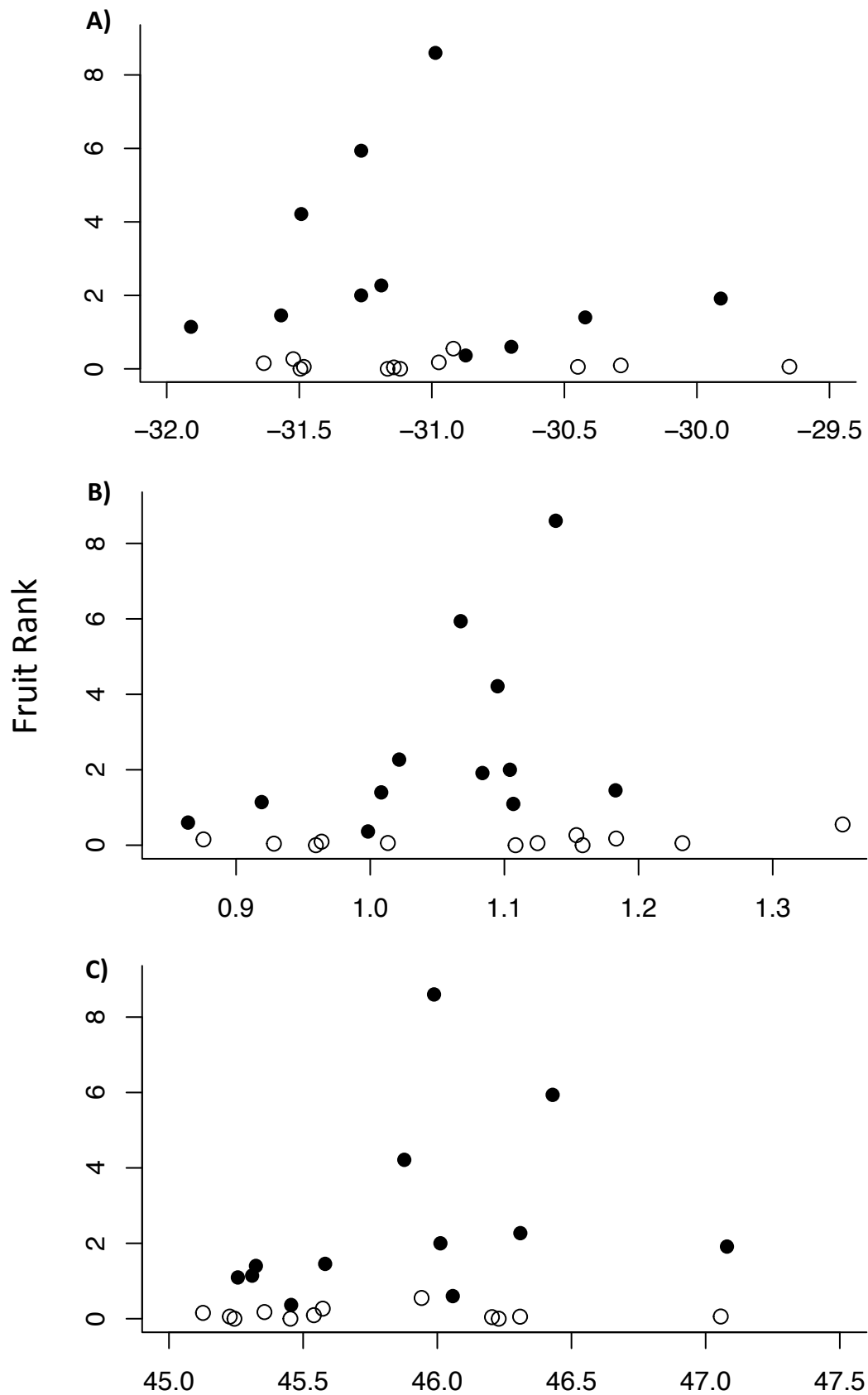


Fig. 2.4. Mean site leaf nutrient values for A)  $\delta^{13}\text{C}$  (‰), B) Leaf Nitrogen (% dry weight), C) Leaf Carbon (% dry weight) contrasted against the mean site fruit rank values. Females = black circles, Hermaphrodites = white circles.

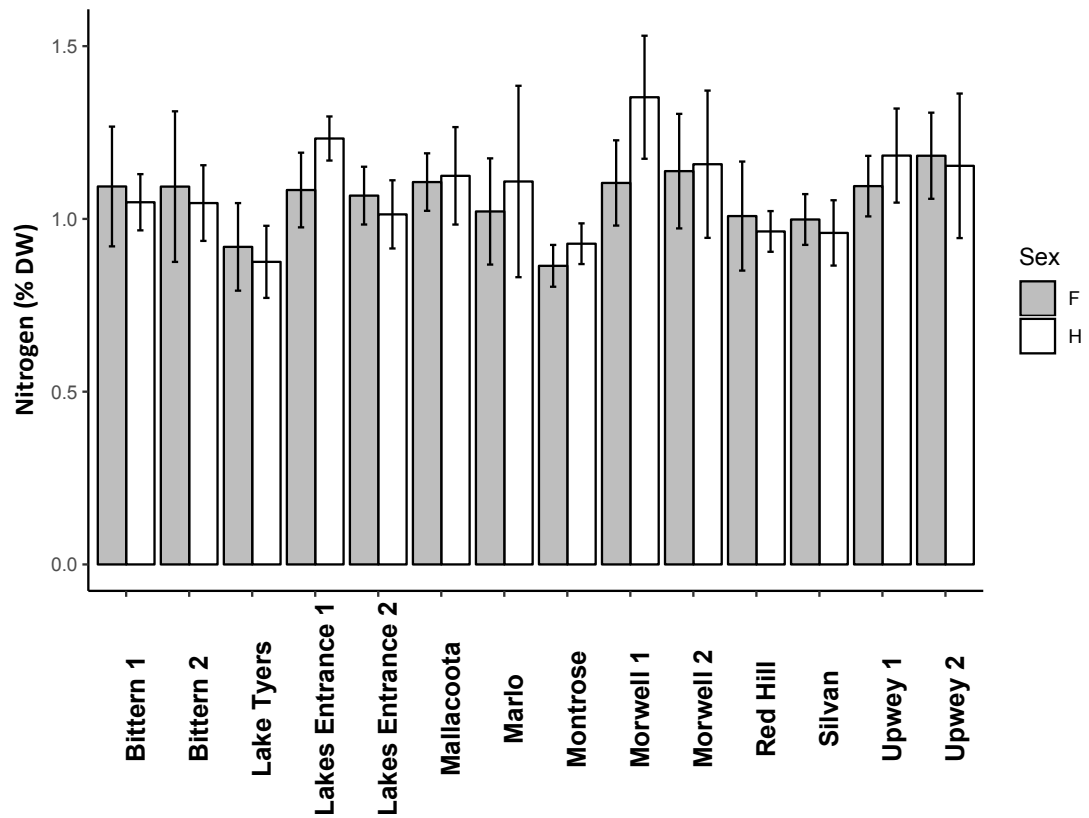


Fig. 2.5. Variation in mean leaf nitrogen content for *P. undulatum* populations. Grey bars = female, white bars = hermaphrodite. Error bars display standard deviation

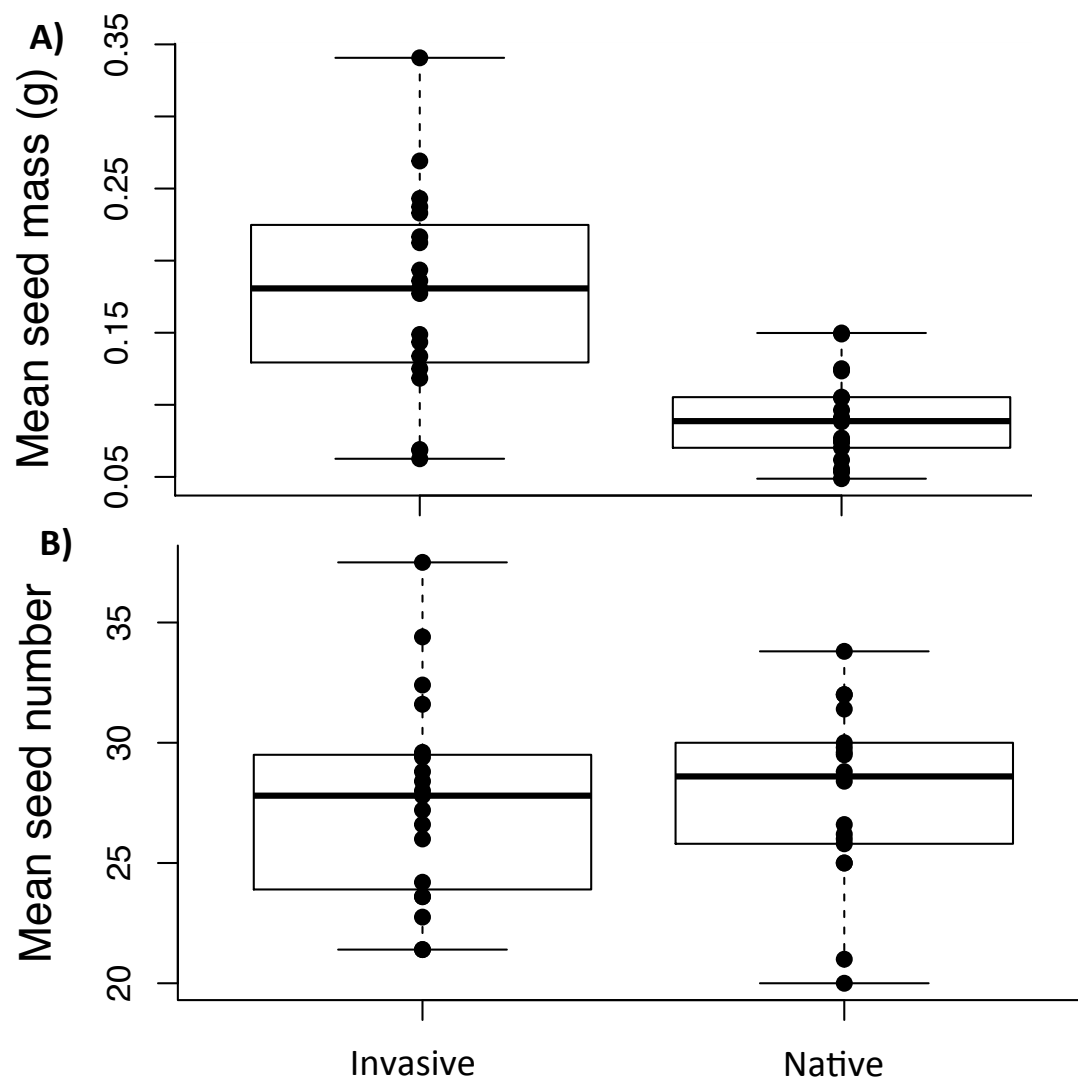


Fig. 2.6. Plot of mean seed mass A) and mean seed number per fruit B) for native and invasive populations. Circles represent values for individual trees, with all samples taken from female trees. Significant difference found for A) at  $\alpha = 0.05$ .

## 2.5 Discussion

This study explored the breeding strategy and reproductive ecology of a range expanding invasive tree and considered how these factors varied between native and invasive populations. We expected to find a high level of female sex expression among plants in invasive populations, because higher numbers of females are likely to enhance population expansion at the invasion front through seed production (Pannell 1997). We also anticipated a higher investment in reproductive traits for invasive

populations, as these traits are likely to both extend the dispersal capacity of seeds and improve their chance of survival to maturity within novel environments.

### **2.5.1 Variations in density and sex ratios for *P. undulatum* populations across native and invasive origins.**

The most striking result found was the lack of male trees within any population regardless of its native or invasive origin. Populations were comprised of either females or hermaphroditic individuals. This finding is in stark contrast to previous studies exploring *P. undulatum* populations within southeastern Victoria, which instead showed a male bias within invasive populations (Gleadow & Walker 2014; Mullett 1996). A hermaphroditic bias has been observed within an invasive population of *P. undulatum* in Jamaica (Goodland & Healey 1996). Of the 60 trees sampled in their study, Goodland and Healey (1996) found 78.4 % of individuals to be hermaphroditic, with the remainder female. Results from a study of *P. undulatum* floral sex ratios at three sites across the foothills of the Dandenong Ranges, Victoria, has suggested females make up approximately 40 % of invasive *P. undulatum* populations, with the remainder being male (Gleadow & Walker 2014). Furthermore, in a study of the *P. undulatum* population at Woods reserve within the Mornington Peninsula, Victoria, Mullet (1996) found approximately 9 % of predominately male flowering plants within five 10 x 30 m plots went on to produce fruit. These distinct results suggest that the proportion of males, females and hermaphrodites making up *P. undulatum* populations may be highly variable and certainly less consistent than previously proposed. The previous works noted above together with results from this study present a Victorian focus. Further sampling across *P. undulatum*'s native range along with greater international sampling of invasive population would be of value.

Theory suggests hermaphroditism should be more common in populations with younger age cohorts and in lower densities (Eppley & Pannell 2007a; Pannell et al. 2008). This theory could imply that as invasive populations of our study are relatively young (approximately 10-30 years) they should be less dense and therefore support higher proportions of hermaphrodites relative to older aged native populations. Instead our sample populations showed an approximate even proportion of female and hermaphroditic individuals, and though variable, a similar density of individuals

across populations, in both cases regardless of their native or invasive origin. Our prediction of greater female representation in invasive populations has therefore not been met. One possibility is that despite the younger age of invasive *P. undulatum* populations, the equivalent density of trees at native and invasive sites means that the selective pressure for a higher proportion of females may not be as strong for the established invasive populations of this study, comparative to a population at the very early stages of invasion/range expansion, where tree density is lower.

### **2.5.2 The influence of tree sex on fruit set and fruit quantity**

Female trees are far more likely to produce fruit relative to hermaphrodites. Female production of significantly higher fruit loads suggests that hermaphrodites are predominately filling the role of males as pollen producers. Baker's law postulates the selective advantage hermaphrodites may have due to their capacity to self-pollinate in environments where mates are sparse (Baker 1955). The strong persistence of females together with observations of hermaphrodites generally fulfilling the male role within *P. undulatum* populations, suggest there is no strong pressure for self-fertilisation within the populations sampled. Given the high stand density that *P. undulatum* populations can reach along with the consistent and ongoing introduction of *P. undulatum* to invasion sites (Gleadow & Ashton 1981; Gleadow & Rowan 1982; O'Leary et al. 2018), mate proximity may not be an issue, which would therefore reduce selective pressure for self fertilisation. In this instance, enhanced reproductive and growth traits may present a stronger selective advantage, improving the capacity to establish, develop and reproduce within varied and disturbed environments.

Contrary to expectations, females from native populations were more likely to fruit relative to those from the invasive range. As  $\delta^{13}\text{C}$  results were non-significant, variation in access to water is unlikely to reflect variation in fruit production. Similarly, a lack of variation in total leaf carbon for trees of different sex and origin suggests that light availability, needed for carbohydrate stores in resources intensive fruit production is a consistent within and across populations (Funk 2013). Though fruit production was not strongly influenced by total carbon and  $\delta^{13}\text{C}$  values, marginal significance was found for the influence of mean leaf nitrogen levels on female fruit



load. The slope of 5.54 found when contrasting fruit production and leaf nitrogen availability is much higher than unity, and indicates the dramatic benefits increased nitrogen availability can have towards fruit production. Exponent values of this magnitude have been found in other plant species (Hubbell 1980). The relationship between nitrogen availability and fruit production has implications on management of the species, as populations downstream from agricultural areas may be more prone to invasion due to the presence of nitrogen based fertilisers. Production of fleshy fruits is considered to be a resource intensive process, and is often attributed to females within dioecious populations incurring higher reproductive costs (Sinclair et al. 2012). However, pollen production may also require high nitrogen resources, which may in part explain why leaves from both female and hermaphroditic individuals recorded similar levels of nitrogen (Harris & Pannell 2008).

Higher fruit set for females in native populations, may indicate that individuals within invasive populations might not be reaching their full fruiting capacity. A variety of factors, such as the level of disturbance, reduced pollination services or presence of facilitator species could contribute to this (Nuñez et al. 2009; Eckert et al. 2010). Regardless, the result may be of concern for land managers, as the ongoing naturalisation of invasive *P. undulatum* populations may potentially reduce this fruiting constraint and therefore the expansion of invasive populations.

Current management practises focus on removing females from invasive populations throughout the range expansion, as these are the individuals that promote further colonising spread via seed dispersal (Gleadow & Walker 2014). This action has the benefit of reducing the workload and spreading limited management resources. Though our work has observed evidence of hermaphroditic individuals producing fruit, a general trend for significantly greater fruit production in females would support the targeted removal of females, as proposed by Gleadow and Walker (2014). However, this policy assumes sex ratios are consistent across all populations, and that the breeding system displayed by individuals remains fixed, regardless of fluctuations in resources, time and stochastic factors. Anecdotal reports suggest sexual lability in *P. undulatum* may present a factor in the species control, with the removal of all females from a population in one season, followed by the production of fruit on trees

previously considered “male”. Further study of this matter is advocated as a means of improving management efforts.

### **2.5.3 *The importance of population origin and sex to reproductive traits***

Seeds produced in the invasive range were significantly larger than in the native range. Seed mass reflects maternal investment, with larger mass representing a potential establishment advantage through greater stored resources. Findings of greater seed mass within invasive populations throughout the literature are mixed, with data showing comparatively larger (Crawley et al. 1996; Daws et al. 2007; Correia et al. 2016) and smaller seeds (Rejmánek 1996; Mason et al. 2008) within invasive populations. Depending upon the mechanism/s in place larger seeds may not disperse as far as smaller seeds. Our findings contrast with the aprior expectation that invasive populations would be made of individuals with smaller mean seed mass. This may reflect an advantage heavier seeds provide to invasive populations when competitiveness or tolerance is favoured over dispersal capacity, as a larger seed mass may improve the capacity for a plant to withstand the unpredictable environments found at novel and disturbed sites (Muller-landau 2010; Van Kleunen et al. 2015). Because humans have assisted the introduction of *P. undulatum* to novel environments, there may not have been the strong selection on dispersal traits that we expected. However, traits that enhance seedling establishment and thus population growth may be favoured in young populations (Pannell 1997; Eppley & Pannell 2007b), and this might occur for differences in seed mass. Analogous to enhanced dispersal capability in cane toads at the invasion front, increased seed mass may also reflect an advantageous genotype at the range margin (Phillips et al. 2010). Avian seed dispersal may be important to *P. undulatum* dispersal which variation in seed size is unlikely to impact (Gleadow 1982). Collectively this emphasises the benefits of enhanced establishment traits over dispersal characteristics.

### **2.5.4 *Conclusion***

Our results highlight the value in understanding the breeding strategy employed by a focal invasive species as a means of developing improved and more targeted control methods. Fruiting characteristics of females and hermaphrodites reinforces the

practise of targeting female individuals for removal in invasive populations, provided subsequent follow up management is employed. This work emphasises the value of a study focused on the reproductive characteristics of an invader and how it can be enhanced with analysis through a biogeographical lens. More broadly, the study of invasive populations in the context of invasion theory helps to expand our understanding of ecological concepts and their application to invasion management.

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### **3.0 Applying community assembly theory to the restoration of lands degraded by woody weeds**

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#### **3.1 Abstract**

Our study examined the response of ten plant communities across Victoria, Australia to the infestation and subsequent removal of *Pittosporum undulatum*, a tree native to south eastern Australia that is increasingly viewed as an invader within and particularly beyond its native range. At sites where *P. undulatum* removal has occurred over a 1-14 year period, species richness, canopy cover and functionality were compared against nearby invaded and remnant temperate Eucalyptus bushland, so as to gauge the direction and magnitude of community change over time. There are four main findings: 1) Low levels of native and non-native species richness and canopy cover recorded at communities impacted by dense *P. undulatum* populations; 2) very low densities of *P. undulatum* at all cleared areas after removal; 3) removing *P. undulatum* caused an increase in species richness, particularly for native species; and 4) over time, management intervention lead to increasing similarity in community composition and function between cleared areas and remnant controls. Our case study demonstrates how community assembly theory and the Passenger-Driver hypothesis can be used effectively to understand the mechanisms at play between native and exotic drivers of community composition and function. Results are discussed in relation to how ecological theory can be applied to inform and improve invasive species management and restorative actions.

## 3.2 Introduction

Rapid globalisation is affecting the composition and structure of ecological communities through the reduction of barriers to biological invasions (Hulme 2009; Pyšek et al. 2012). Invasive exotic species can result in dramatic biodiversity loss, diminished community function, reduced ecosystem services and altered disturbance regimes (Brooks et al. 2004; Liao et al. 2008; Pejchar & Mooney 2009; Simberloff 2011). Native species may also possess invasive characteristics, placing further stress upon ecological communities within their natural environment (Adair 2008; Carey et al. 2012b; Simberloff et al. 2012; Taylor et al. 2016). Restoration ecology is increasingly being viewed as a tool to combat the damaging effect of invasive species, and in doing so it is also being used to improve the integrity and resilience of native ecological communities (CBD 2011; Chazdon 2008; Suding 2011). Significant global interest and investment through the Convention for Biological Diversity and initiatives such as the Bonn Challenge, strongly support restorative practices (Aronson & Alexander 2013; Verdone & Seidl 2017). Despite this, a disjunction between on-ground restoration effort and scientific theory is apparent. Several authors have called for greater integration of established ecological theory into the practice of ecological restoration (Dickens & Suding 2013; Giardina et al. 2007; Hobbs & Norton 1996; Matzek et al. 2014). Furthermore, the practice of follow up monitoring, although strongly advocated, has until recently been infrequent at best (Funk et al. 2008; Suding 2011; Wortley et al. 2013). Greater understandings of the capacity for restoration programs to restore ecological integrity are needed in order to efficiently and effectively rehabilitate degraded landscapes (Suding 2011). The challenge for researchers is to synthesise theory into a format applicable to practical on-ground management (Dickens & Suding 2013; Matzek et al. 2014). Here, we explore community assembly theory, the Passenger-Driver Hypothesis and functional richness and its capacity to enhance restoration projects through a wide-scale invasive species treatment and monitoring program.

Application of community assembly theories has been suggested as a means of bridging the divide between ecological theory and restoration practice (Booth & Swanton 2002; Funk et al. 2008; Hulvey & Aigner 2014; Ryan et al. 2010). Community assembly theories propose that the composition and structure of a

particular community is a consequence of a series of filters, affecting which individuals from a regional pool of species are able to disperse into and survive the specific abiotic and biotic conditions of the area (Díaz et al. 1998; Keddy 1992; Lars et al. 2012). Management focused at the species level has in some cases seen the removal of one invasive species, only to have it return or be replaced by another. For example, once cleared of the invasive Reed Canarygrass (*Phalaris arundinacea* L.), wetland communities of North America are frequently prone to reinvasion from the same species due to the remnant seed bank and dispersal from nearby sources (Adams & Galatowitsch 2006). Community assembly theory broadens the focus of invasion management from the species to the community level, which is suggested to improve long-term management outcomes (Booth & Swanton 2002; Ryan et al. 2010). Funk et al. (2008) proposed that adjusting community assembly filters for local conditions should favour a return of native community composition, structure and function over time, provided a sufficient seed source is available. Adjustment of filters could be used as treatments to increase community resistance to the establishment of invaders and facilitate long-term community resilience (Cleland et al. 2013; Funk et al. 2008; Hulvey & Aigner 2014; Palmer et al. 1997). Whether or not this is successful depends upon a thorough understanding of the focal weed/s ecology, local community, disturbance regime and the abiotic conditions (Hobbs 2007).

The effectiveness of any management intervention depends on whether or not the conditions that favour the indigenous plant community, and encumber the invasive species, can be established. The Passenger-Driver hypothesis (PDH) draws from this idea, distinguishing species that have become invasive as a result of environmental change, human induced or otherwise, from those capable of driving change in the absence of environmental alteration (MacDougall & Turkington 2005). The PDH provides a useful perspective for placing a focal invasive species in a whole community management context. The PDH categorises species according to their response to change. Species that become invasive as a direct result of environmental change are considered Passengers; species that drive change in ecosystem function, species composition and functional diversity independently of any environmental alteration are considered Drivers; species that establish after environmental change but then proceed to drive plant community change independently from the initial disturbance are considered Backseat Drivers (Bauer 2012; Chabrierie et al. 2008). The

response of the native plant community after invasive species removal can indicate the category that the invader falls under. Removal of a Passenger species is analogous to treating a symptom, with the target species and/or other invaders likely to reinvade (Fig. 3.1). A more successful approach will be to treat the underlying environmental change that enabled the invader to first establish. Removal of both Backseat Driver and Driver species should see a direct return of native plant communities, though in the case of the Backseat Driver this may take longer due to initial disturbances that allow the weed to establish (Fig.3.1). Finally at the most extreme end of the spectrum, Transformer species can alter community dynamics beyond local thresholds, pushing the community to a new stable state (Fig. 3.1) (Catford et al. 2012; Fukami & Nakajima 2011; Richardson et al. 2000). Neither the removal of Transformer or Passenger species is expected to promote the reestablishment of native communities, and for this reason the PDH cannot distinguish between these two types of invaders. However, defining the role of a target invasive species as a Passenger, Driver, Backseat Driver or Transformer of a system, should better enable land managers to modify conditions more favourable to native species (Lindenmayer et al. 2015).

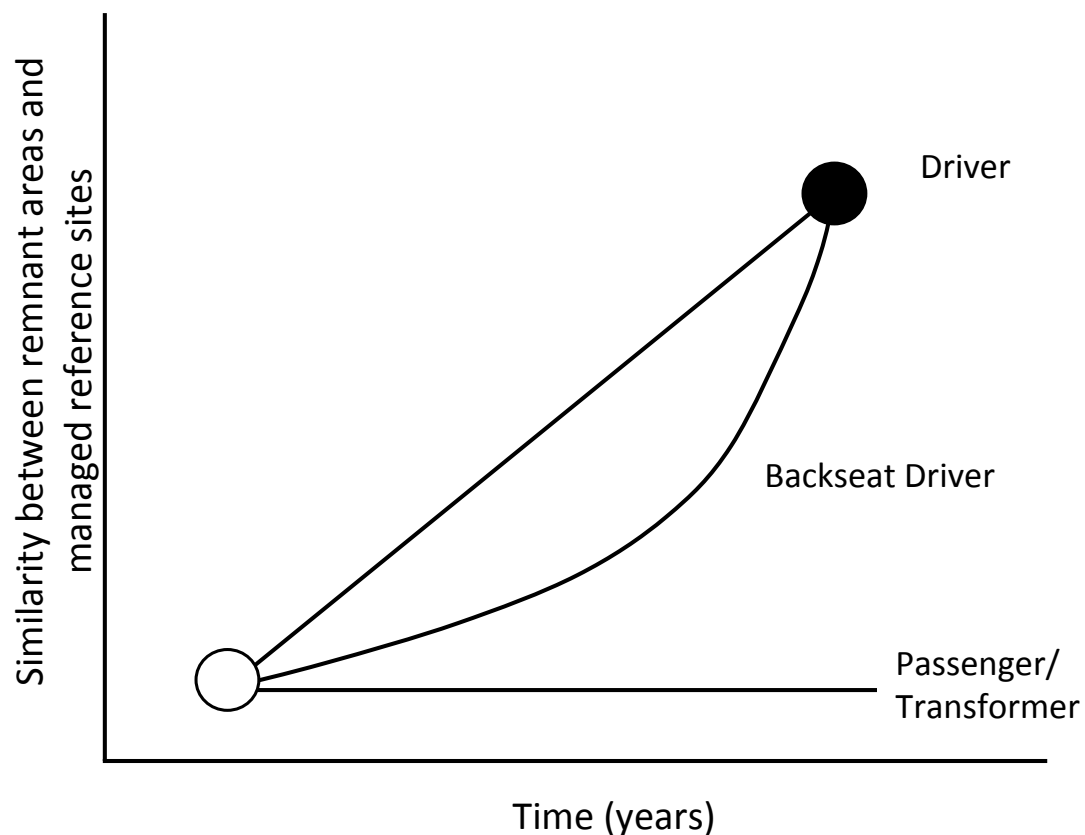


Fig. 3.1. Conceptual diagram depicting the response of a heavily invaded plant community (white circle) to invasive species removal. Plant community response is considered relative to a biodiverse high quality reference community (black circle). Removal of Passenger weeds is expected to alter the invaded community but not to a state similar that of the remnant reference community. Removal of Driver and Backseat Driver weeds is expected to lead to ecosystem recovery. However, if the disturbance initially enabling the establishment of a backseat driver is not addressed, the recovery of affected communities may be delayed. Removal of Transformer species will not lead directly to ecosystem recovery and for this reason the response of native plant communities to Transformer and Passenger species removal cannot be distinguish. Transformation of the ecosystem can make it difficult for native species to re-establish. Therefore altered stable states must be factored into management planning.

The response of a species to abiotic and biotic constraints along with its role as a Passenger, Driver, Backseat Driver or Transformer within a community is ultimately governed by its functional traits. Traits reflect a species' resource capture, reproduction, dispersal and environmental strategies/tolerances (McGill et al. 2006; Reich et al. 2003; Westoby et al. 2002). Species trait values can therefore be used to characterise the functional diversity of a community. Despite traditional focus on increasing species richness, greater emphasis should be placed upon improving a community's functional diversity. Increasing functional diversity within a community can promote greater invasion resistance, long-term stability and improved ecosystem functioning (Díaz & Cabido 2001; Funk et al. 2008; Montoya et al. 2012; Pokorný et al. 2005). This approach provides a new perspective to view community processes. Trait based analysis has been advocated as a method to reveal if a community is recovering from a functional standpoint (Cadotte et al. 2011; Mouillot et al. 2013). Plant height, photosynthetic performance and reproduction are three traits considered particularly informative when examining a species' life history strategy (Díaz et al. 2016; Westoby 1998; Westoby et al. 2002). Height indicates plant competitive ability at maturity, seed mass represents the trade-off between dispersal ability and resource available to a germinating seedling, and SLA (fresh leaf area divided by oven dry mass) indicates the productivity of a leaf (Cornelissen et al. 2003). Together with an understanding of the community and its environment, these traits can help to gauge

restoration success and guide the strategic direction of future works (Drenovsky et al. 2012; Funk et al. 2008; Funk & McDaniel 2010).

Here, we present the results of a study of plant communities overtime following the removal of *Pittosporum undulatum* Vent (sweet pittosporum). *Pittosporum undulatum* is a shade tolerant tree native to coastal areas of southeastern Australia, which demonstrates many of the characteristics of an invasive species within and well beyond its native range (Gleadow et al. 1983). Present across a range of habitat types, *P. undulatum* is most commonly found in wet and temperate rainforests (Gleadow & Ashton 1981). Altered fire regimes, introduced vectors, peri-urban disturbance and horticultural propagation have all contributed to the spread of this species after European arrival (Gleadow 1982; Gleadow & Ashton 1981; Gleadow & Rowan 1982; Gleadow et al. 1983). Presently, *P. undulatum* is considered invasive within many regions across Australia as well as Lord Howe Island and Norfolk Island (Eurobodalla Council 2017; Mornington Peninsula Shire et al. 2012; Yarra Ranges 2017).

*Pittosporum undulatum* has also become a major issue globally, with invasive populations in New Zealand, Portugal, Jamaica, Hawaii, and is an emergent weed in South Africa (Gleadow & Ashton 1981; Goodland & Healey 1996; Hortal et al. 2010; Lourenço et al. 2011; Mokotjomela et al. 2013). *Pittosporum undulatum* is known to establish quickly after disturbance (Bellingham et al. 2005; Rose & Fairweather 1997), although it may also become weedy at undisturbed locations (Gleadow & Ashton 1981; Gleadow & Walker 2014; Rose & Fairweather 1997). Once established, mature trees can reach heights of 8-30 m (Mullett 2001). Individuals form dense canopies, shading out the undergrowth and reducing structural diversity, floristic composition and the integrity of ecological systems (Gleadow & Ashton 1981; Mullett 2001).

Specifically we aim to: 1) Determine the capacity for communities to re-establish and resist reinvasion following woody weed removal; 2) Use *P. undulatum* as a case study to test the Passenger-Driver hypothesis, examining the role of the weed species as a Passenger, Driver, Backseat Driver or Transformers within plant communities; and, 3) examine the response of functional traits in communities after weed removal. We hypothesise that if *P. undulatum* is a Passenger/Transformer, then communities will display both low species and functional richness following weed removal. However,

if the target weed is acting as a Backseat Driver/Driver we expect that woody weed removal will be sufficient to make significant improvements to community richness and function through time.

### 3.3 Methods

#### 3.3.1 Site Description

We investigated the ways in which intact native plant communities responded to the infestation and subsequent removal of sweet pittosporum (*Pittosporum undulatum* Vent.). Ten sites across peri-urban areas of Melbourne, in southeastern Australia, were identified to evaluate the success of *P. undulatum* removal and its influence on community richness and function (Table 3.1 and Fig. 3.2). The Dandenong Ranges, 45 km east of Melbourne, and the Mornington Peninsula 85 km southeast from Melbourne, both represent a temperate, wet climate (Table 3.1). Sites were selected based on the following three conditions (1) presence of a high quality area of remnant vegetation that would act as a positive control – “remnant control”; (2) an area where work had been conducted to remove *P. undulatum* – “cleared treatment”; and (3) an area currently infested by *P. undulatum* to act as a negative control – “invaded control” (not included at Birdland, Ferntree Gully and Sherbrooke Forest). Positive and negative controls enable an assessment of both the direction and magnitude of any change in the treated community (Guido & Pillar 2017). Site selection was refined to ensure a comparison of similar vegetation within any one site. Ecological Vegetation Class (EVC) mapping (supported by on-ground observations) was utilised to ensure each sampling condition within a site supported similar vegetation (DELWP 2017). Sampling conditions within each site ranged in size from 1ha-12 ha. Information on the density of *P. undulatum* at the cleared areas, weed management practises and disturbance regime was included in the analysis. The management area at each site was characterised as having a severe *P. undulatum* infestation prior to removal work. *Pittosporum undulatum* density at each site ranged from 30 %-70 % canopy cover (Table 3.1). To remove the effect of different management practices, sites were chosen based upon the intensity of follow up weed monitoring. With the exception of the Wonga Park site, which was cleared in 2016, all sites have



experienced some degree of follow up weed maintenance within 12 months of the initial *P. undulatum* removal. Maintenance was conducted at least twice after *P. undulatum* removal for sites aged three years and older. Approximately half of the sites have been exposed to control burns within the past 15 years. Control burns were used on remnant conditions only as a means of maintaining the natural disturbance regime and stimulating biodiversity (Penman et al. 2011). Unlike other studies e.g. (Gleadow & Narayan 2007), fire was not used in the initial control of *P. undulatum* at the study sites. Cleared areas and invaded controls had not been burnt within the timeframe of this study (14 years).

### **3.3.2 Vegetation Sampling**

Sampling was conducted from late October to early December of 2016. To determine the effect of *P. undulatum* on local plant diversity, 6-10 randomly stratified quadrats were established at each sampling condition within each site. Using the species area curve method, plant identification was performed across an increasing large area to a point where no new species were recorded. It was at this point that the bounds of the quadrat were set. This process was conducted initially at remnant control areas to determine the quadrat size applied to cleared and invaded areas of the same site. Quadrats were either 3 x 3 m<sup>2</sup> or 4x4 m<sup>2</sup> in size. Each individual plant within a quadrat was identified to species level wherever possible (VicFlora 2017) and assigned a percentage score for canopy cover, including projected foliage cover (based upon visual estimates).

To examine the functional recovery of sites following *P. undulatum* removal, plant functional trait values were assigned to 205 of the 348 observed species. Trait data availability limited our capacity to examine all species present. The trait values recorded for each species were maximum height (Bull 2014), seed mass (Meers et al. 2010; Royal Botanic Gardens Kew 2017; Williams et al. 2005) and specific leaf area (SLA) (collected from sources in Appendix 3a). These three traits are considered particularly influential to the overall plant species life strategy.

### 3.3.3 *Statistical analysis*

All analysis was conducted using R statistical program (R Core Team 2017). Multiple linear regression was utilised to examine the effect of time since clearing on the relative species richness at cleared treatments within each site. Principal components analysis (PCA) ordination techniques were used to visualise variability in species presence/absence and percentage canopy cover between conditions and across sites. The functional response of communities to *P. undulatum* infestation and its subsequent removal was also investigated through PCA, using mean site trait values, weighted by mean percentage cover scores. Differences in observed values between conditions within each site were used for inter-site comparison. All data were centred and scaled to unit variance prior to analysis. Species with canopy cover scores of  $< 1\%$  were given a value of 0.5 to enable statistical analysis. Euclidian distances were calculated in multidimensional space, with vectors extracted as the basis for principal components. Multiple linear regression was used to examine relationships between time since management action, and the relativised Euclidian distance between remnant controls and cleared treatments on the graph.

Table 3.1. Site details including location, vegetation type, *Pitiosporum undulatum* density prior to removal, year of removal and climate. Ref no refers to labelling format in Figs. 3.6a, b and c.

Site name	Ref No	Latitude	Longitude	Ecological Vegetation Complex (EVC)	Initial <i>P. undulatum</i> density (%)	Year of <i>P. undulatum</i> removal	Mean Annual rainfall (mm)	Elevation (m)
Wonga Park (WP)	1	-37.755709	145.283738	Grassy dry forest	50	2016	807.5	141
Greens Bush (GB)	2	-38.418634	144.958019	Damp sands herb rich woodlands	50	2015	779.4	176
Panton Hill (PH)	3	-37.642608	145.242843	Grassy dry forest	70	2014	688.5	181
Woods Reserve (WR)	4	-38.288326	145.091165	Lowland forest	50	2012	904.3	91
Birdsland Reserve (BR)	5	37.924444	145.340278	Grassy dry forest	30	2011	1113.6	170
Glenfern Valley Bushlands (GFVB)	6	-47.909783	145.314540	Valley Grassy Forest	60	2010	1056.8	187
Ferntree Gully (FTG)	7	-37.879164	145.306283	Grassy Dry Forest	50	2006	928.4	276
Red Hill (RH)	8	-38.401103	145.040113	Herb Rich Foothill Forest	60	2006	1008.9	114
Montrose (M)	9	-37.820394	145.346866	Grassy dry forest	60	2005	1031.9	409
Sherbrooke Forest (S)	10	-37.905239	145.369618	Wet Forest	50	2002	1261.5	495



Fig. 3.2. Map depicting the distribution of sites across peri-urban Melbourne, Australia, and the Mornington Peninsula. Black circles denote site locations in reference to Melbourne (black pentagon). White and grey sections indicate land and ocean respectively. WP- Wonga Park, GB – Greens Bush, FTG – Ferntree Gully, PH – Panton Hill, M – Montrose, BL – Birdsland Reserve, WR – Woods Reserve, RH – Red Hill, GFVB – Glenfern Valley Bushlands, S – Sherbrooke Forest.

### 3.4 Results

We compared the species richness, percent cover and functional diversity of plant communities where *Pittosporum undulatum* was present, where it had been removed and in areas historically free of this species. There were four main findings: 1) low levels of native and non-native species richness and canopy cover were recorded at communities impacted by dense *P. undulatum* populations; 2) a very low density of *P. undulatum* was recorded at all cleared treatments after removal, regardless of years since removal (1-14 years); 3) removing *P. undulatum* resulted in an increase in species richness, particularly for native species, and; 4) increasing time since management intervention saw increasing similarity in community composition and function between cleared treatments and remnant controls.

#### 3.4.1 Species richness and canopy cover

Invaded controls, where *P. undulatum* was in high density, supported low species richness, with an average of 21.1 species present across 10 sites (see supplementary data – Appendix 3b). Very few species were recorded growing under a *P. undulatum* canopy and these species were regularly observed at low densities. The removal of *P. undulatum* resulted in an increase in species richness (Fig. 3.3). Species richness in quadrats at cleared treatments increased to levels comparable to remnant control communities (Fig. 3.3). A strong relationship was detected between relative species richness at cleared treatments and the time since weed treatment took place ( $F = 4.575$ ,  $p = 0.014$ ) (Table 3.2, Fig. 3.4). The distance between cleared treatments and remnant controls did not appear to have a significant impact on the relative species richness of restored plant communities ( $p = 0.887$ ) (Table 3.2). The proportion of native species at cleared areas was higher than that of exotic species (Fig. 3.3). Cleared treatments did not see an increase in *P. undulatum* canopy density over time (Fig. 3.5).

Upon visual inspection principal components analysis (PCA) detected a clustering effect across sites for invaded controls, but varied across sites for cleared and remnant areas (Figs. 3.6a, 3.6b). However, there was a trend in the direction of the remnant

control i.e. cleared areas appeared closer to their corresponding positive remnant controls. Two exceptions were sites 3 and 7, both of which were exposed to higher levels of disturbance. When examining the effect of time since management action on the relativised Euclidian distance between remnant and cleared areas, a significant negative linear trend was detected, with the state of cleared treatments becoming more similar to remnant conditions with increasing time since management intervention (Figs. 3.7a, 3.7b:  $F = 3.752$ ,  $p = 0.119$  and  $F = 2.735$   $p = 0.03$ , Table 3.2).

### **3.4.2 Functional trait analysis**

To examine the community response to weed removal from a functional perspective, species were classified according to functional traits (Fig. 3.6c). A clustering pattern was again found for communities at invaded controls. When examining the relative Euclidian distance between remnant controls and cleared treatments a non-significant negative linear trend was detected such that cleared treatment areas became increasingly similar in function to remnant controls, with greater time since *P. undulatum* removal (Fig. 3.7c) ( $p = 0.149$ ,  $F = 1.024$ ).

Table 3.2. Statistical analysis of the effect of time since clearing on the relative species richness observed at cleared treatments across sites (data presented in Fig. 3.3), along with the relativized Euclidian distance between remnant control and cleared areas within sites (data presented in Fig. 3.7a,b,c). \* Denotes significance at 95% confidence. Species Richness: Adjusted  $r^2 = 0.442$ , F statistic = 4.575. Presence/Absence: Adjusted  $r^2 = 0.478$ , F statistic = 3.752. Canopy Cover: Adjusted  $r^2 = 0.366$ , F statistic = 2.735. Functional Richness: Adjusted  $r^2 = 0.007$ , F statistic = 1.024.

		Estimate	Std. Error	T value	p value
Species	Intercept	0.663	0.137	4.812	0.001*
Richness	Time since clearing	0.052	0.016	3.227	0.014*
	Distance to remnant site (m)	0.000	0.000	0.147	0.887
Presence	Intercept	1.421	0.515	2.754	0.033*
Absence	Time since clearing	0.122	0.067	-1.814	0.119
	Distance to remnant site (m)	0.000	0.000	1.025	0.345
Canopy	Intercept	3.495	0.869	4.021	0.006*
Cover	Time since clearing	-0.309	0.113	-2.722	0.034*
	Distance to remnant site (m)	0.001	0.000	-1.617	0.156
Functional	Intercept	2.923	1.244	2.349	0.057*
Richness	Time since clearing	-0.269	0.162	-1.655	0.149
	Distance to remnant site (m)	0.000	0.001	-0.554	0.599

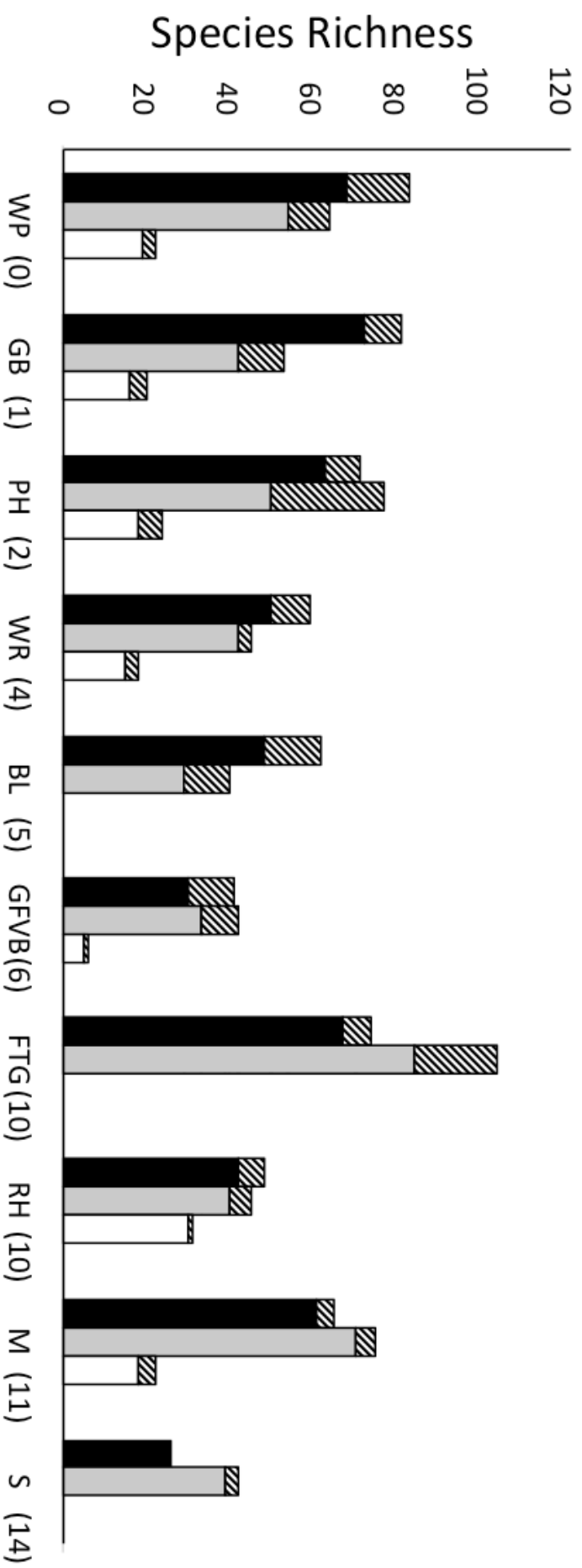


Fig. 3.3. Variation in species richness across sites and within sampling areas. Site labels relate to site names and years since *Pittosporum undulatum* removal found in table 1; WP- Wonga Park, GB – Greens Bush, PH – Panton Hill, WR – Woods Reserve, BL – Birdsland Reserve, GFVB – Glenfern Valley Bushlands, FTG – Ferntree Gully, RH – Red Hill, M – Montrose, S – Sherbrooke Forest. Remnant control – remnant bush land (black), Cleared treatment – *Pittosporum undulatum* removed (grey), Invaded control – high-density *Pittosporum undulatum* (white). Hatched sections indicate the proportion of non-native species recorded within each sampling condition. For statistical analysis see Table 3.2.



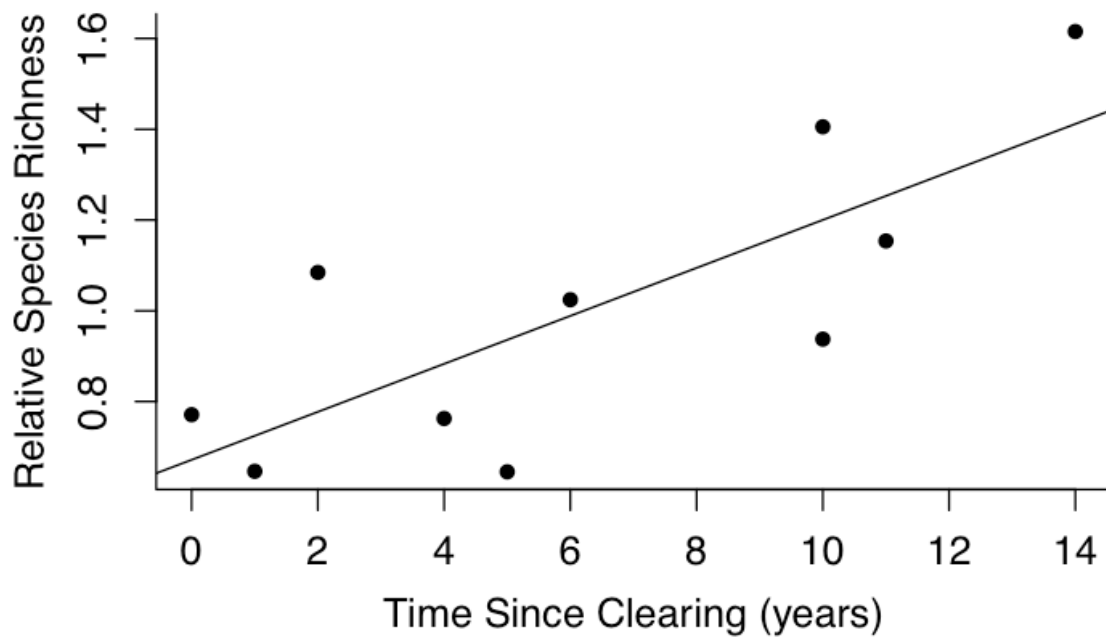


Fig. 3.4. Relationship between time since *Pittosporum undulatum* removal at cleared treatments and relative species richness (species richness at cleared treatment divided by species richness at remnant control). Adjusted  $r^2 = 0.442$ ,  $p = 0.014^*$ . For full statistical analysis see Table 3.2.

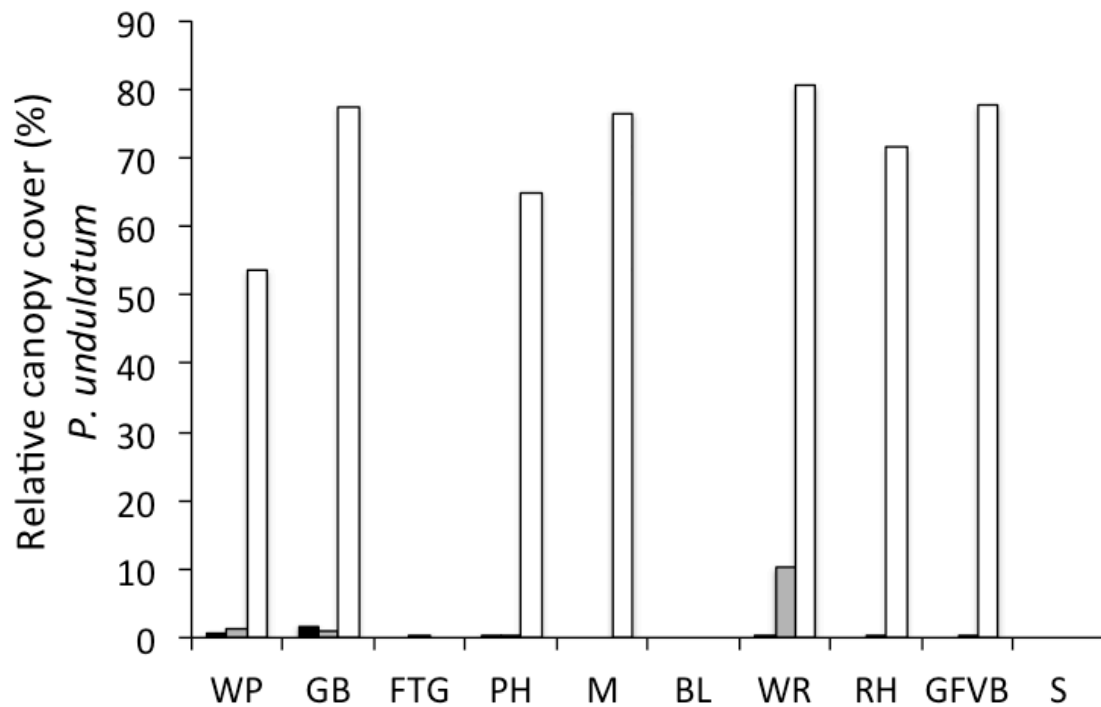


Fig. 3.5. Mean percentage cover of *Pittosporum undulatum* within treatments and across sites. Remnant control – remnant bushland (black), cleared treatment – *Pittosporum undulatum* removed (grey), negative control – high-density *Pittosporum undulatum* (white). A negative treatment was not included at FTG, BL and S treatments. WP- Wonga Park, GB – Greens Bush, PH – Panton Hill, WR – Woods Reserve, BL – Birdsland Reserve, GFVB – Glenfern Valley Bushlands, FTG – Ferntree Gully, RH – Red Hill, M – Montrose, S – Sherbrooke Forest. For full statistical analysis see Table 3.2

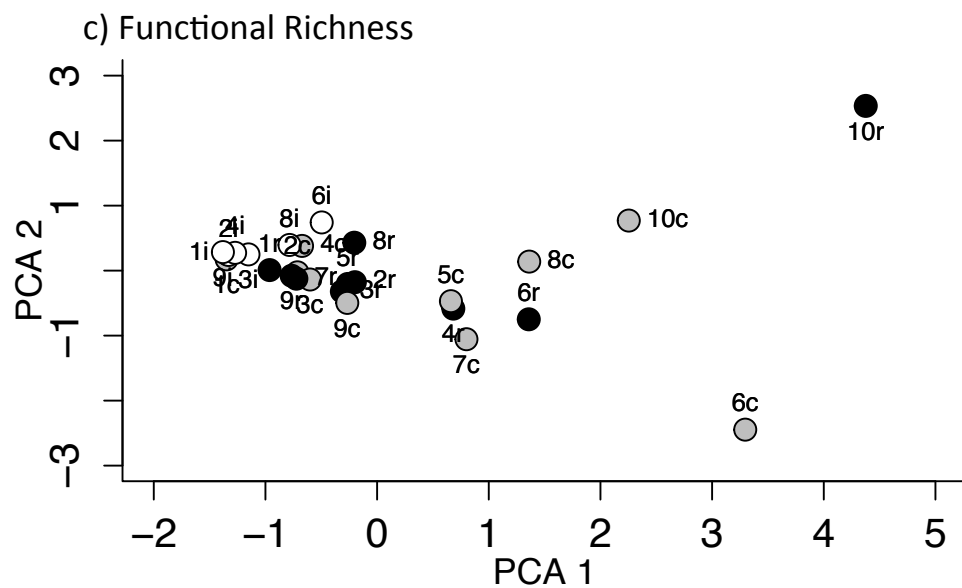
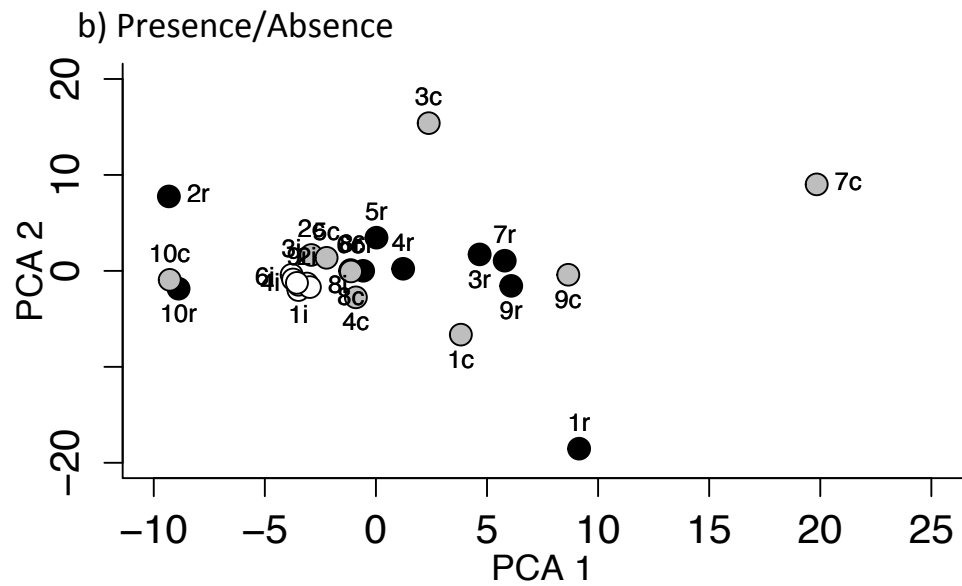
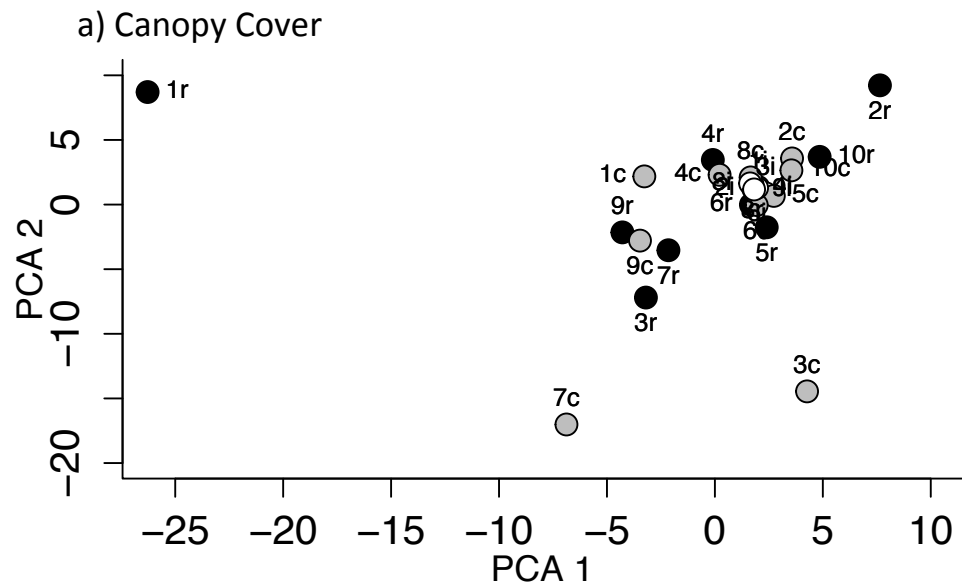


Fig. 3.6. Principal components analysis showing the change in community similarity in terms of a) mean canopy cover, PCA 1 variation explained 10.75 %, PCA 2 variation explained 9.00 % b) species presence/absence PCA 1 variation explained 11.31 %, PCA 2 variation explained 9.02 % and c) the relative functional similarity of all sampling conditions within sites PCA 1 variation explained 70.65 %, PCA 2 variation explained 21.96 %. Points were composed of mean weighted trait values for each sampling condition across all sites based upon the trait values of 205 species. Sampling conditions are labelled by site “ref no” found in Table 1 and treatment type. 1- Wonga Park, 2 – Greens Bush, 3 – Pantom Hill, 4 – Woods Reserve, 5 – Birdslan Reserve, 6 – Glenfern Valley Bushlands, 7 – Ferntree Gully, 8 – Red Hill, 9 – Montrose, 10 – Sherbrooke Forest. r = Remnant control – remnant bushland (black circles), c = Cleared treatments – *Pittosporum undulatum* cleared (grey circles), i = invaded control – high density *Pittosporum undulatum* (white circles). For full statistical analysis see Fig. 3.7 a,b,c and Table 3.2.

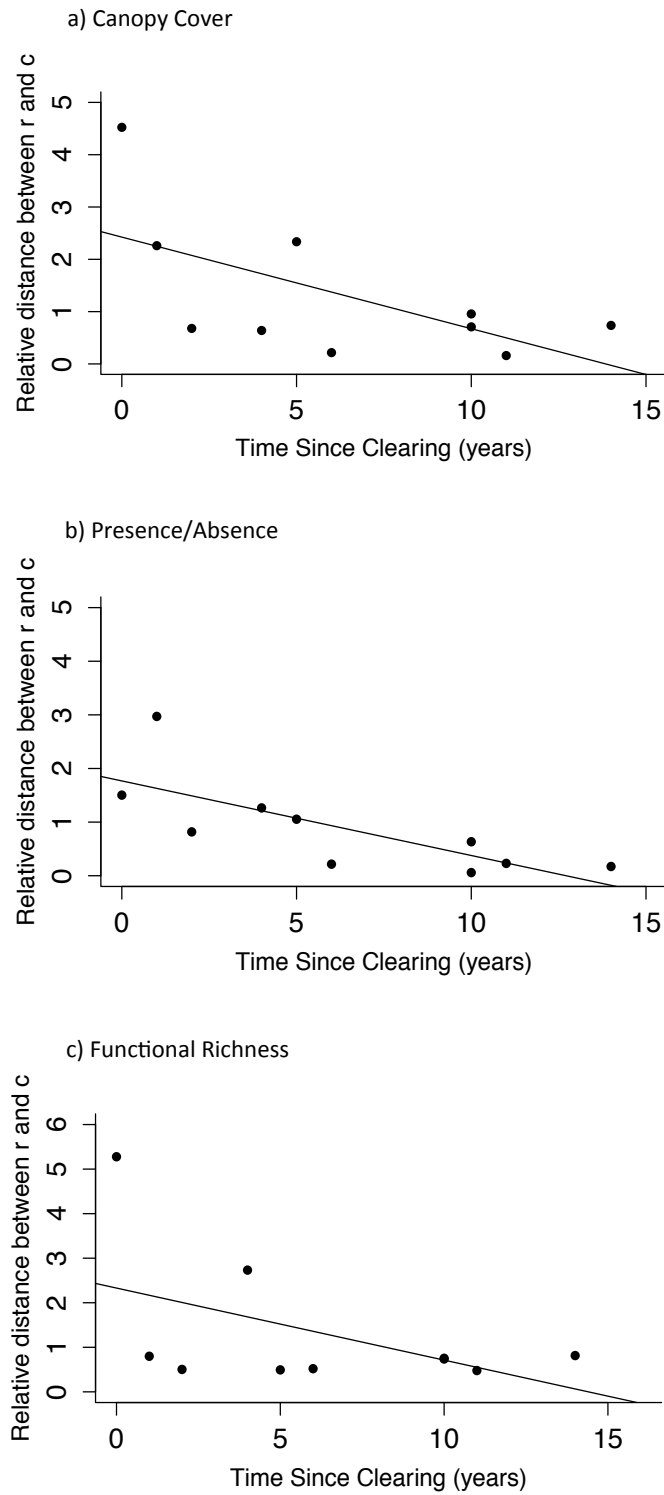


Fig. 3.7. The relative Euclidian distance between remnant controls (r) and cleared treatments (c) across time since management action for a) Canopy Cover, adjusted  $r^2 = 0.3664$ ,  $p = 0.034^*$ , b) Presence/Absence, adjusted  $r^2 = 0.478$ ,  $p = 0.119$ ; and c) functional richness  $r^2 = 0.007$ ,  $p = 0.149$ . Distances are relative to the distance between remnant and invaded conditions within a site. For full statistical analysis see Table 3.2.

### 3.5 Discussion

The targeted removal of invasive species such as *Pittosporum undulatum* can greatly influence the composition, structure and functionality of native plant communities. Here we show that *Pittosporum undulatum* removal resulted in an increase in native flora richness and a proportionally small increase in the presence of exotic species. Once *P. undulatum* was cleared, very low rates of re-establishment were detected. Importantly, the removal of this species appears to have altered the state of cleared treatments. Control of *P. undulatum* produced a trend whereby the species richness, canopy cover and functionality of plant communities within cleared areas became increasingly similar to nearby positive reference controls over time.

#### ***3.5.1 How has the removal of *Pittosporum undulatum* influenced the capacity for communities to re-establish and resist reinvasion?***

Our results highlight how the removal of a targeted woody weed can alter community composition and structure whilst influencing community assembly filters. More broadly, the results demonstrate the capacity for ecological theory to strengthen practical on-ground restoration efforts through the implementation of a long-term comparative monitoring program. Reduction of *P. undulatum* reduces its presence within cleared treatments. Whilst this is a relatively basic process it has far reaching implications for the regulation of community assembly filters (Booth & Swanton 2002; Funk et al. 2008; Hulvey & Aigner 2014; Ryan et al. 2010). Initial effects of woody weed removal are strongest on the biotic assembly filter, which operates at local scales on direct competition. Removal of a dominant woody weed such as *P. undulatum* diminishes its capacity to compete with resident native species, whilst alleviating pressure for nutrients, moisture and light (Gleadow & Ashton 1981; Gleadow et al. 1983; Levine et al. 2003). It has been suggested that *Pittosporum undulatum* may also have an allelopathic effect on its neighbors (Gleadow 1982; Gleadow & Ashton 1981). Thus, removal of this species from the system may reduce any suppressive effect *P. undulatum* has on competitors, in addition to increasing the availability of resources (Del Fabbro & Prati 2015).

The same processes that affect the biotic assembly filter also impact the abiotic assembly filter, operating at the landscape scale. Here, the abiotic environment was altered principally through the removal of *P. undulatum*'s dense shading canopy (Devine et al. 2007). Removal of the canopy influences light and moisture levels, in turn changing the local microclimate (Roundy et al. 2014). Woody weed presence and removal can also change the regulation of microbial processes, leading to altered nutrient availability (Lindsay & French 2005; Marchante et al. 2009). Recent studies have shown that the restoration of plant communities at degraded landscapes can have a strong positive effect on native pollination systems (Baskett et al. 2011; Kaiser-bunbury et al. 2017). An indirect increase in the competitive success of native species may therefore be as a result of increased pollinator richness. There is no way to separate the effect of an adjusted biotic filter from abiotic filter. However, distinguishing between these two effects and the adjustments to dispersal filters is possible. Bird species such as the Silver eye (*Zosterops lateralis*), Pied Currawong (*Strepera graculina*) and particularly the introduced European Blackbird (*Turdus merula*) are considered important dispersers of *P. undulatum* seed (Gleadow 1982; Mullett 1996). Observations of these species at all 10 sites (ALA 2017) suggests that their role as vectors may not strongly influence the recovery of native plant communities after *P. undulatum* management.

A near complete absence of *P. undulatum* regrowth at all cleared treatments, even after 1-14 years, underlines the importance of knowing the ecology of the target species. *Pittosporum undulatum* holds most of its seed in the canopy, thus once mature trees are removed from the system, so is the majority of its seed (Gleadow & Narayan 2007). Furthermore, viability of the remaining *P. undulatum* seed bank is considered poor if exposed to hot dry conditions in the absence of a shading canopy, or if buried more than 4cm deep in the soil (Gleadow & Rowan 1982; Gleadow & Narayan 2007).

Beneath the deep shading *P. undulatum* canopy full sunlight has been recorded at just 24 % and 8 % through summer and winter respectively (Gleadow 1982; Gleadow & Narayan 2007). Low light levels may preclude the recruitment of other invasive species, and may help to explain the low level of exotic species observed at cleared

areas. Furthermore, the deep shade may also inhibit the germination of indigenous species present within the soil seed bank. Thus, native species that are in situ are able to quickly re-establish after removal of *P. undulatum*.

### **3.5.2 To what degree does the target weed act as a driver of community change?**

Communities within this study cleared of *P. undulatum* displayed not only a rapid increase in species richness, but also a trend to shift back towards a state similar to that of remnant controls. A shift such as this is more typical of a Driver species (Lindenmayer et al. 2015; MacDougall & Turkington 2005). Monitoring the ecological response of native communities after invasive species removal enables the assessment of the target species along the Passenger - Driver spectrum (Lindenmayer, Wood, et al. 2015; MacDougall & Turkington 2005). Following this process helps to indicate the importance of disturbance to invader establishment (Hobbs 2007; Lindenmayer et al. 2015). Information such as this can assist land managers to decide if invasive species treatment is likely to lead to ecosystem recovery, define expectations, form management programs and to set realistic goals (Guido & Pillar 2017; Lindenmayer et al. 2015).

Defining *P. undulatum* as a Driver of change helps to improve our understanding of the species as a woody weed. Furthermore it underlines the suitability for investment and targeted removal of *P. undulatum* to restore high quality remnant bush land. If communities had responded to *P. undulatum* removal with a Passenger/Transformer response, careful amelioration of the disturbance regime and abiotic processes would have been suggested, along with the adjustment of restoration goals see (Lindenmayer et al. 2015, see Fig. 3.1). This process can be evaluated only after the removal of invasive species and therefore cannot inform initial management actions. However, dominant weed species are often targeted for removal by a range of stakeholder groups e.g. Government bodies, Parks associations and volunteer groups. Researching the results of these efforts via a basic floristic assessment, together with nearby remnant and invasive controls (in a similar fashion to this study) would be highly informative to the development of further action plans.



### **3.5.3 Do floristic communities recover after restoration work from a functional standpoint?**

The analysis of community recovery from a functional traits perspective provides an additional lens to judge and evaluate the effect of restoration actions. Greater functional diversity means that available resources are utilised more efficiently across an area, improving community resilience, long term stability and invasion resistance (Cadotte et al. 2011; Díaz & Cabido 2001; Funk et al. 2008). A non-significant negative relationship regarding differences between remnant controls and cleared areas suggests that removal of *P. undulatum* may have altered the functional richness of managed plant communities (Fig. 3.6c, 3.7c). This relationship suggests plant communities may become more functionally similar to positive remnant conditions with increasing time since management action took place. Though a strong relationship was not found, any pattern for increased functionality, together with increases in species diversity could support the idea that the rate at which plant functionality increases is limited by the rate at which new species can return and re-establish (Aerts & Honnay 2011). As the result is nonsignificant we suggest further research to investigate this possibility. The level of disturbance, soil stored seed bank and time also constrain the natural re-establishment of highly functional native plant communities. Activities that cause high levels of disturbance such as woody weed removal can promote the establishment of colonizing species, which outcompete less vigorous growers and late successional species (Bohn et al. 2014; Palma et al. 2017). Colonizing species often share many functional attributes e.g. short lifespan, small seed size and rapid growth strategies (Bohn et al. 2014; Webster et al. 2006). Thus, whilst functional richness may increase, if functionally similar species are entering the region, overall community functionality may not rise significantly (Aerts & Honnay 2011). The rate of increasing species richness and by extension the rate of increasing functional richness will therefore depend upon both the time it takes for seed from more distant areas to reach the site, along with the time it takes for successional processes to diversify niche space within the environment.

### **3.5.4 Conclusions**

Comparing restoration successes and failures is made difficult by the paucity of monitoring after intervention. Despite this, some evidence exists suggesting that given

sufficient time, often in the order of decades, and the necessary human will, ecosystems can recover from serious disturbance. These concepts are supported by the data presented here. Our study demonstrates how ecological theory can be used to understand native communities response to invasive species and ecosystem change, leading to more informed restoration projects in the future. As global demand for restoration solutions increases, application of theory-based approaches will become increasingly important. We therefore encourage greater monitoring and data collection integrated within each restoration project and support a whole ecosystem approach to monitoring restorative effects including fauna, ecosystem process, soil microbes and soil nutrient analysis.

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## 4.0 Response of south eastern Australian bird communities following invasive tree removal

This chapter has been accepted for publications by the journal *Biological Invasions* subject to minor revision.

### Abstract 4.1

Examining faunal responses to invasive plant removal expands our understanding of the capacity for disturbed communities to be restored, beyond the conventional lens of simply focusing on vegetation recovery. We sought to explore how changes in local bird communities over time can be used to assess the efficacy of restoration programs with time, focusing on removal of the invasive tree/shrub *Pittosporum undulatum* Vent (Sweet Pittosporum) across temperate south eastern Australia. To do this we compared the species richness, density and functionality of local resident bird communities across remnant and invaded areas, together with those cleared of invasive *P. undulatum* populations. Presence of *P. undulatum* appeared to reduced the number of carnivorous bird species observed at invaded areas, but did not appear to strongly influence bird community species richness, or density among sampling conditions. Strong differences were observed between those birds present within or below the dense *P. undulatum* canopy, relative to birds also within invaded areas but utilising other habitat features such as the canopy above. We discuss the value in understanding the impacts of invasive tree species and their removal, particularly in light of the long-term decline in ground dwelling bird communities across temperate Australia. Our work demonstrates the habitat values that *P. undulatum* populations provide to bird communities and their functional groups, along with the effect of invasive tree removal on bird communities, and the response of those bird communities with time. We show that studies of this nature including avian monitoring programs can enhance restoration projects.

## 4.2 Introduction

The practice of restoration ecology is increasingly used to counter the damaging effects of invasive plant species and to restore the integrity of native communities. Successful restoration is, however, more complex than the simple removal of invasive species. In particular, the value of restored vegetative communities to the wildlife they contain remains poorly understood (Munro et al. 2014). Birds are strongly influenced by the characteristics of their environment, such as vegetation structure and floristic richness (Carignan & Villard 2002; Ortega-Álvarez & Lindig-Cisneros 2015; Munro et al. 2014; Padoa-schioppa et al. 2005). Moreover, birds contribute to a functioning ecosystem through seed dispersal, increased germination via gut passage, pollination, pest species control, and soil formation (Reid et al. 2014; Sekercioglu 2006; Lindenmayer et al. 2017). Birds, therefore, make excellent bio-indicators of environmental health and fluctuations in bird communities measured against suitable controls are likely to reflect the success of restoration efforts.

Birds are increasingly incorporated into restoration monitoring globally. For example, studies across Central America (Reid et al. 2012; Reid et al. 2014), Africa (Aerts et al. 2008) and the Caribbean (Zahawi & Augspurger 2006) have measured bird community richness and abundance in response to different restoration treatment types. Despite these examples, the practise remains limited, particularly in Australia. In the review by Ortega-Álvarez & Lingig-Cisneros (2015) of 109 avian related publications from the journal *Ecological Restorations*, only five were from Australia, of which only one, based in the tropics, explored the response of communities to invasive weeds (White et al. 2009).

Functional traits ultimately govern the ecological behaviour of all species. Such traits reflect a species' opportunity for resource capture, reproduction, dispersal and environmental strategies/tolerances (McGill et al. 2006; Reich et al. 2003; Westoby et al. 2002) (Table 4.1). Species trait values can therefore be used to characterise the functional diversity of a community. Despite the traditional focus on increasing species richness, greater emphasis might therefore be placed upon improving a community's functional diversity (Lindenmayer, Blanchard, et al. 2015). Examining changes in bird functional diversity in the context of restoration ecology provides a

different perspective to assess the sensitivity and recovery of community processes, which may subsequently promote greater community resilience, long-term stability and improved ecosystem functioning (Fischer et al. 2007; Karp et al. 2011; Lindenmayer, Blanchard, et al. 2015).

Here, we investigate fluctuations in resident bird community species richness, density and functional traits over time following the removal of the native woody weed tree, sweet pittosporum (*Pittosporum undulatum*). *Pittosporum undulatum* is a shade tolerant tree native to coastal areas of southeastern Australia from the Gold Coast in Queensland to Gippsland in South Eastern Victoria. The species has been observed to exhibit many characteristics of an invasive species within and beyond its native range (Gleadow et al. 1983; Gleadow & Ashton 1981). O’Leary et al. (2018) suggested that *P. undulatum* removal is likely to enable native plant communities to recover. Whilst the response of birds to this tree has been discussed only for specific species until now (Gleadow 1982; McNabb & McNabb 2011), there have been no examinations of how *P. undulatum* removal may affect bird communities more broadly. Through this study we investigate how invasive *P. undulatum* populations affect the species richness and density of bird communities. Furthermore, this study investigates if bird communities at invaded areas utilise *P. undulatum* infestations as habitat. Finally we examine how the removal of *P. undulatum* populations influences bird communities, and if those communities change with increasing time since removal.

We hypothesise that the presence of dense *P. undulatum* populations and the subsequent reduction in floristic structure will make these areas less suitable for a wide variety of birds, thereby reducing avian species richness and density. We also expect the reduction in vegetation structure created by dense *P. undulatum* infestations to narrow the functional range of bird species present, particularly when compared to the vegetation overstorey of the upper canopy. Finally, we postulate greater bird species richness, density and functionality will occur following *P. undulatum* removal and the subsequent recovery of vegetation communities.

## 4.3 Methods

### 4.3.1 *Pittosporum undulatum*

Present across a range of habitat types, sweet pittosporum (*Pittosporum undulatum* Vent.) is most commonly found in wet and temperate rainforests (Gleadow & Ashton 1981). Altered fire regimes, introduced vectors, peri-urban disturbance and horticultural propagation have all contributed to the spread of this species after European arrival (Gleadow 1982; Gleadow et al. 1983; Gleadow & Ashton 1981; Gleadow & Narayan 2007; Gleadow & Rowan 1982). Presently, *P. undulatum* is treated as an invader within many regions across Australia including Lord Howe Island, Norfolk Island, the Yarra Ranges and Mornington Peninsula (Eurobodalla Council 2017; Mornington Peninsula Shire et al. 2012; Yarra Ranges 2017).

*Pittosporum undulatum* has also become a major issue globally, with invasive populations in New Zealand, Portugal, Jamaica, Hawaii, and is an emergent weed in South Africa (Gleadow & Ashton 1981; Goodland & Healey 1996; Hortal et al. 2010; Lourenço et al. 2011; Mokotjomela et al. 2013). *Pittosporum undulatum* is known to establish quickly after disturbance (Bellingham et al. 2005; Stefan Rose 1997), although it may also become weedy at undisturbed locations (Gleadow & Ashton 1981; Gleadow & Walker 2014; Stefan Rose 1997). Reports from Gleadow (1982) suggest that seeds from *P. undulatum* fruit are not widely eaten by native birds, though they represent a major winter food source for the introduced European blackbird (*Turdus merula*). Once established, mature trees can reach heights of 8-30m (Mullett 2001). Individuals form dense canopies, shading out the undergrowth and reducing structural diversity, floristic composition and the integrity of ecological systems (Gleadow & Ashton 1981; Mullett 2001).

### 4.3.2 *Site Selection*

Ten peri urban sites across the Yarra Ranges and Mornington Peninsula near Melbourne, in southeastern Australia, were identified to evaluate the success of *P. undulatum* removal and its influence on bird community richness and function (Table

4.2 and Fig. 4.2). The vegetation at these sites has been analysed in a previous study (O’Leary et al. 2018) (Table 4.2). Sites were selected based upon local vegetation health and biodiversity, with each site comprising the following three conditions (1) presence of a high quality area of remnant vegetation – “remnant control”; (2) an area where work had been conducted to remove *P. undulatum* – “cleared treatment”; and (3) an area currently infested by *P. undulatum* – “invaded control” (not included at Ferntree Gully and Sherbrooke Forest). Through direct observation and call, bird species found at invaded controls were further categorised by those present within or below the dense *P. undulatum* population (no canopy), relative to birds also within invaded areas but utilising other habitat features throughout the overstorey above (canopy). A strong contrast in vegetation between no canopy and canopy conditions made this distinction relatively straightforward. No canopy vegetation density is considered to provide significantly lower habitat heterogeneity relative to the understorey conditions of nearby remnant sites (O’Leary et al. 2018; Stirnemann et al. 2014). Remnant, cleared and invaded conditions enable an assessment of both the direction and magnitude of any change in the managed community (Guido & Pillar 2017). Site selection was refined to ensure a comparison of bird communities across similar vegetation within any one site. Ecological Vegetation Class (EVC) mapping (supported by on-ground observations) was utilised to ensure each sampling condition within a site supported similar vegetation (DELWP 2017). Each site ranged in size from 1ha-12ha. Information on the density of *P. undulatum* at the cleared areas, weed management practises and disturbance regime was included in the analysis. The management area at each site was characterised as having a severe *P. undulatum* infestation prior to removal work. *Pittosporum undulatum* density at each site ranged from 30%-70% canopy cover (Table 2). To remove the effect of different management practices, sites were chosen based upon the intensity of follow up weed monitoring. With the exception of the Wonga Park site, which was cleared in 2016, all sites have experienced some degree of follow up weed maintenance within 12 months of the initial *P. undulatum* removal. Maintenance was conducted at least twice after *P. undulatum* removal for sites aged three years and older. Approximately half of the sites have been exposed to control burns within the past 15 years. Control burns were used on remnant conditions only as a means of maintaining the natural disturbance regime and stimulating biodiversity (Penman et al. 2011). Unlike other studies e.g. (Gleadow & Narayan 2007), fire was not used in the initial control of *P.*

*undulatum* at study sites. Cleared areas and invaded controls had not been burnt within the timeframe of this study (14 years). Further details of the vegetation of each site can be found in O’Leary et al. (2018).

#### **4.3.3 Bird Surveys**

Bird surveys were performed in order to assess the effect of *P. undulatum* and its subsequent removal on local bird communities. Sampling was conducted from mid May to late June of 2017. Surveys were conducted for each site on three separate mornings during the period of dawn to 10am. Following a modified version of the process established by Loyn (1986) and outlined in Loyn et al. (2007), 10 minutes survey time was allocated per hectare of sampling area within each site, to a maximum of 20 minutes. Sites were visited three times each, with species presence/absence at a condition comprising all bird species observed across the three visits. Bird density was taken as the total number of observations of each species averaged across the three visits. Birds observed by sight and sound within and below the vegetation canopy were identified to species level. Birds flying overhead were not included in the study.

#### **4.3.4 Functional Traits**

Trait values reflecting the functionality of bird communities were extracted from the Handbook of Australian, New Zealand and Antarctic Birds (Higgins et al. 2006). The trait types identified in Table 4.1. have been selected as they provide additional measures of community response to the restoration process beyond simply species richness and density. In particular these response traits focusing on bird morphology, foraging behaviour and habitat preferences, reflect environmental changes following *P. undulatum* removal and subsequent restoration of vegetation communities. These traits enable a deeper exploration of those bird types more sensitive to these ecological changes (Stirnemann et al. 2014; Hatfield et al. 2018).

Table 4.1. Functional traits, habitat and feeding preferences considered important to the regulation of bird communities observed within the study.



Trait Type	Trait Description	Reference
Body mass (g)	Smaller birds are associated with complex, heterogeneous, fine grained vegetation structure, whereas larger birds are more commonly found in open environments	(Fischer et al. 2008; Stirnemann et al. 2014)
Foraging behaviour (arboreal or non – arboreal)	Ground dwelling species are associated with heterogeneous vegetation and high ground cover, potentially as protection against predation	(Antos et al. 2008; Stirnemann et al. 2014)
Nest location (arboreal or non – arboreal)	Arboreal relative to ground nests are more commonly depredated within southeastern Australian Eucalypt forests, particularly by bird species.	(Piper & Catterall 2004; Whyte et al. 2005)
Interspecific aggression (aggressive or non aggressive)	Many Honeyeater species, along with some carnivorous species are strongly territorial, and can suppress overall bird diversity	(Ford 1979; Ford et al. 2001; Fulton & Ford 2002)
Habitat preference	Preference for open sclerophyll forests, closed sclerophyll forests or variable habitat including both open, closed and disturbed environments	(Higgins et al. 2006)
Feeding preference	Feeding guilds including insectivores, omnivores, carnivores and nectivores/granivores/frugivores	(Higgins et al. 2006)

#### 4.3.5 Statistical analysis

All analysis was conducted using the R statistical program (R Core Team 2017). Nonmetric multi dimensional scaling and principal components analysis (PCA) ordination techniques were used to visualise variability in species presence/absence

and density between conditions and across sites. As both techniques were in agreement PCA results will be discussed throughout.

The functional response of communities to *P. undulatum* infestation and its subsequent removal was also investigated through PCA, using mean site trait values of bird species present at each site, weighted by mean bird species density scores. Differences in observed values between conditions within each site were used for inter-site comparison. All data were centred and scaled to unit variance prior to analysis. Euclidian distances between points were calculated on the plane of the first two principal components.

Euclidian distances between remnant conditions and managed sites on PCA graphs were assessed relative to their proximity to invaded conditions. These Euclidian distances were then explored through linear and multiple linear analyses as a means of assessing the level of community recovery following *P. undulatum* removal, given the initial level of disturbance caused by the *P. undulatum* infestation

Linear and multiple linear regression was used to examine relationships between time since management action, and the relativised Euclidian distance between remnant controls and managed sites on the graph. One-way ANOVA along with Tukey's post hoc test was employed to examine differences between conditions in terms of species richness and density. PCA was also used for variable reduction in the analysis of feeding guild variation and habitat suitability. Analysis took place using the `prcomp` function from the Base package. All variables with eigenvalues greater than 0.9 were used in the analysis. PCA scores were then compared between conditions using ANOVA and Tukey's post hoc test.

Table 4.2. Site details including location, *Pittosporum undulatum* density prior to removal, year of removal and vegetation type. Ref no refers to labelling format in fig. 4.6

Site name	Ref No.	Latitude	Longitude	Ecological Vegetation Complex (EVC)	Initial <i>P. undulatum</i> canopy cover (%)	Year of <i>P. undulatum</i> removal	Mean Annual rainfall (mm)	Elevation (m)
Wonga Park (WP)	1	-37.755709	145.283738	Grassy dry forest	50	2016	807.5	141
Greens Bush (GB)	2	-38.418634	144.958019	Damp sands herb rich woodlands	50	2015	779.4	176
Ferntree Gully (FTG)	3	-37.879164	145.306283	Grassy Dry Forest	50	2006	928.4	276
Panton Hill (PH)	4	-37.642608	145.242843	Grassy dry forest	70	2014	688.5	181
Montrose (M)	5	-37.820394	145.346866	Grassy dry forest	60	2005	1031.9	409
Birdsland Reserve (BR)	6	37.924444	145.340278	Grassy dry forest	30	2011	1113.6	170
Woods Reserve (WR)	7	-38.288326	145.091165	Lowland forest	50	2012	904.3	91
Red Hill (RH)	8	-38.401103	145.040113	Herb Rich Foothill Forest	60	2006	1008.9	114
Glenfern Valley Bushlands (GFVB)	9	-47.909783	145.314540	Valley Grassy Forest	60	2010	1056.8	187
Sherbrooke Forest (SF)	10	-37.905239	145.369618	Wet Forest	50	2002	1261.5	495



Fig. 4.1. Map depicting the distribution of sites across peri-urban Melbourne, Australia, and the Mornington Peninsula. Black circles denote site locations in reference to Melbourne (black pentagon). White and grey sections indicate land and ocean respectively. WP- Wonga Park, GB – Greens Bush, FTG – Ferntree Gully, PH – Panton Hill, M – Montrose, BL – Birdsland Reserve, WR – Woods Reserve, RH – Red Hill, GFVB – Glenfern Valley Bushlands, S – Sherbrooke Forest

## 4.4 Results

No significant differences in bird community species richness or density were observed among remnant, cleared or invaded conditions (Table 4.3, see supplementary material - Appendix 4a – presence absence table, Fig. 4.3). Of birds detected within invaded conditions, significant differences were identified between those birds observed within and under the *P. undulatum* canopy (no canopy), and those species not within the *P. undulatum* infestation (canopy) e.g. in the *Eucalyptus* overstorey. Significantly fewer bird species (49 % less) and a lower overall density of birds (64 % lower density) were observed within no canopy areas, when compared against canopy habitat (Table 4.4).

Analysis of bird feeding guilds, habitat preferences and life history traits indicate that areas invaded by *P. undulatum* support birds with similar habitat preferences and trait values relative to remnant and cleared bird communities, but support significantly fewer carnivorous species (Table 4.5, Fig. 4.4a and 4.4b, See supplementary material - Appendix 4b). PCA results relating to feeding guild and habitat preferences showed significantly lower scores for no canopy communities relative to the canopy bird communities, underlining the relatively low level of bird species richness and density within no canopy communities (Table 4.5, Fig. 4.5a and 4.5b, see supplementary material - Appendix 4b).

Overall low numbers of exotic bird species were observed (3 exotic species across all sites), and with regard to these exotic birds no difference in species richness or density was detected between sampling conditions or across sites (refer to supplementary material - Appendix 4a). The Powerful Owl (*Ninox strenua*) was the only threatened species recorded (Webster et al. 1999), with two birds observed within remnant areas of the Woods Reserve site and a further two birds observed within the cleared area of the Birdsland Reserve site.

Although the removal of *P. undulatum* does appear to influence the species richness, density and functionality of bird communities (Fig. 4.4a and 4.4b), Euclidian distance analysis indicates no pattern to suggest that bird communities at cleared areas become any more or less similar to corresponding communities at remnant areas (Table 4.6

and Fig 4.6a, 4.6b, 4.6c). Despite this, a result approximating significance suggests bird species richness ( $p = 0.0822$ ) and density ( $p = 0.0722$ ) increase with the length of time elapsed since management action to clear areas invaded by *P. undulatum* (Table 4.4 and Fig. 4.7a, 4.7b). No apparent change to functionality over time was identified.

Table 4.3. ANOVA analysis comparing bird species richness and bird density across conditions.

	df	Sum Sq	Mean Sq	F Value	p Value
Species Richness					
Condition	2	101.3	50.67	2.444	0.107
Residuals	25	518.4	20.73		
Bird Density					
Condition	2	316.2	158.10	2.397	0.112
Residuals	25	1649.1	65.96		

Table 4.4. Single factor ANOVAs examining differences in bird species richness and density for no canopy and canopy components of invaded areas. \* Denotes significance at 95 % confidence

		df	Sum sq	Mean Sq	F	p
ANOVA (species richness)	Condition	1	60.06	60.06	7.074	0.018*
	Residuals	14	118.88	8.49		
ANOVA (species density)	Condition	1	3.719	3.719	10.62	0.005*
	Residuals	14	4.904	0.350		

Table 4.5. Tukey's test examining different scores for PCA 2 based upon species across feeding preferences. R - Remnant control – remnant bushland, C - Cleared treatments – *Pittosporum undulatum* cleared, I - invaded control – high density *Pittosporum undulatum*. In addition, Tukey's test examining differences in feeding and habitat preferences for birds at invaded areas within no canopy and canopy communities. \* Denotes significance at 95 % confidence

	Diff	Lwr	Upper	P adj
I-C	1.1326	0.1386	2.1267	0.0233*
R-C	-0.0713	-1.0085	0.8658	0.9803
I-R	-1.2040	-2.1980	-0.2099	0.0154*
Invaded feeding preferences	-2.2989	-3.6765	-0.9213	0.0032*
Invaded habitat preferences	-1.9909	-2.9796	-1.0022	0.0007*

Table 4.6. Statistical analysis of the effect of time since clearing on both the relative species richness and densities of remnant, cleared, and invaded bird communities observed across sites (data presented in Fig. 4.3), along with the Euclidian distances between remnant and cleared treatments across sites (data presented in Fig. 4.5). \* Denotes significance at 95% confidence. Species Richness: Adjusted  $r^2 = 0.1939$ , F statistic = 2.082. Species Richness Euclidian Distance: Adjusted  $r^2 = 0.36$ , F statistic = 3.531. Species Density:  $r^2 = 0.2383$ , F statistic = 2.408, Species Density Euclidian Distance:  $r^2 = -0.01672$ , F statistic = 0.926

		Estimate	Std. Error	T value	p Value
Species Richness ANOVA	Intercept	0.6652	0.1739	3.826	0.0064*
	Time since clearing	0.0417	0.0205	2.027	0.0822
	Distance to remnant site (m)	-0.0000	0.0001	-0.443	0.6712
Euclidian Distance (PCA scores based upon species richness values)	Intercept	1.1976	0.4980	2.405	0.0471*
	Time since clearing	-0.1047	0.0589	-1.778	0.1186
	Distance to remnant site (m)	0.0006	0.0002	2.149	0.0687
Species Density ANOVA	Intercept	0.6778	0.2652	2.555	0.0378*
	Time Since Clearing	0.0663	0.0313	2.115	0.0722
	Distance to remnant site (m)	-0.0001	0.0001	-0.802	0.4491
Euclidian Distance (PCA scores based upon species density values)	Intercept	1.4301	0.7659	1.867	0.104
	Time since clearing	-0.0562	0.0906	-0.620	0.555
	Distance to remnant site (m)	0.0005	0.0004	1.269	0.245
Euclidian Distance (PCA scores based upon community trait values)	Intercept	-0.0035	0.5950	-0.006	0.995
	Time since clearing	-0.0516	0.0703	-0.734	0.487
	Distance to remnant site (m)	0.0002	0.0003	0.755	0.475



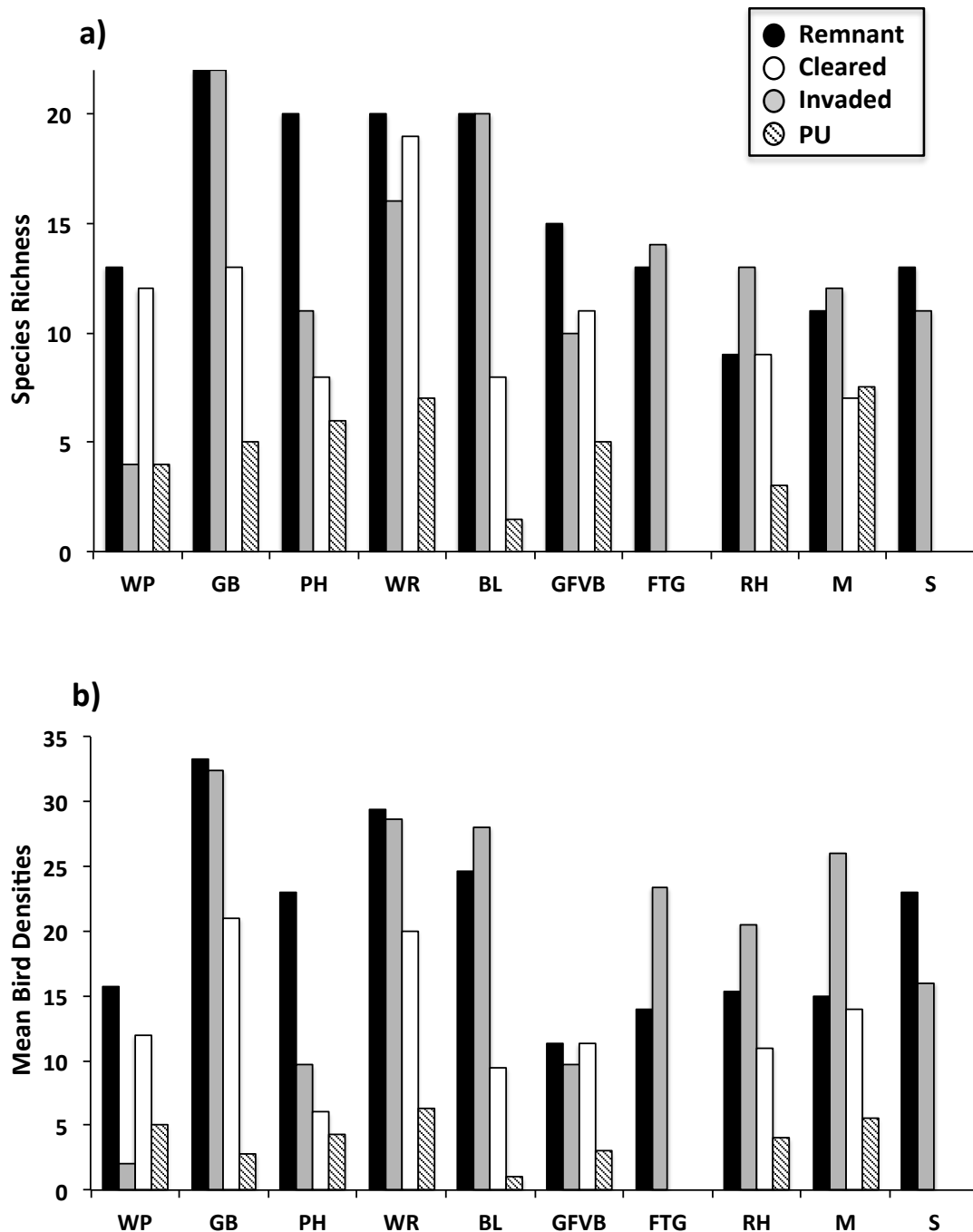


Fig. 4.3. Species richness of bird communities within conditions and across all sites (a). Densities of birds observed within conditions and across sites relative to sampling effort (b). Remnant control – remnant bush land (black), Cleared treatment – *Pittosporum undulatum* removed (grey), Invaded control – high-density *Pittosporum undulatum* (whole community) (white), no canopy – birds observed specifically within the *P. undulatum* canopy at invaded controls (hatched). Sites have been ordered by time since removal, with WP being the most recently cleared site and S the most distant. Further details on time since removal are available in table 4.2.

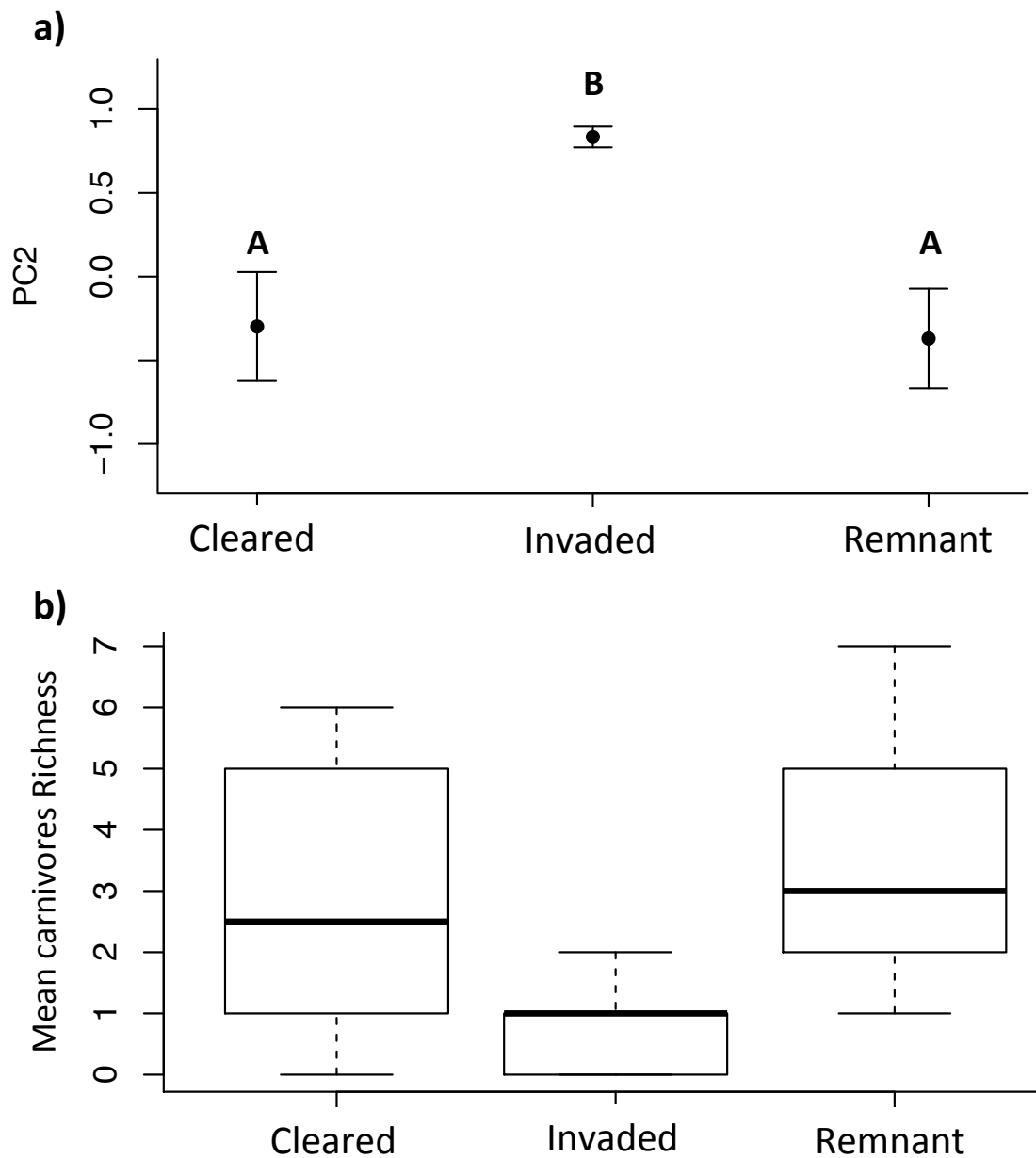


Fig. 4.4. a) Line plot across different sampling conditions displaying values for PCA 2 based upon bird feeding guilds. A – Both Cleared and Remnant conditions have statistically similar scores for PCA 2, B – Invaded conditions are significantly different to Cleared and Remnant conditions with respect to PCA 2. Confidence intervals represent +/- mean standard error. b) Mean number of carnivores observed across each sampling conditions

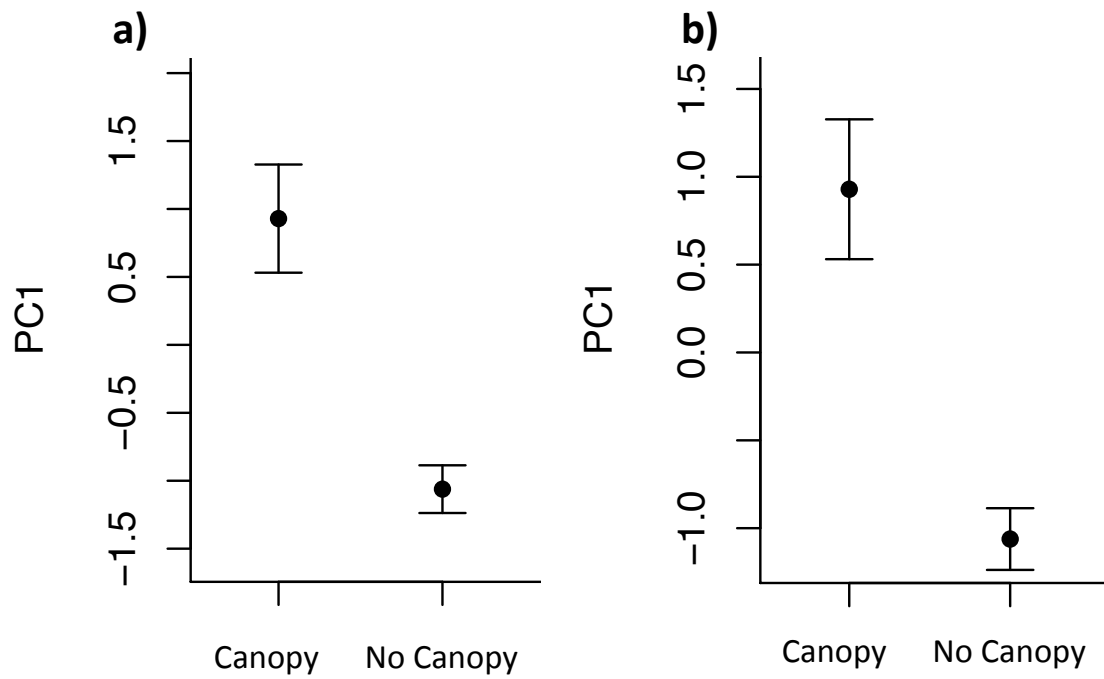


Fig. 4.5. Line plots distinguishing bird communities observed utilising *dense P. undulatum* infestation within invaded areas (No Canopy) against those using other habitat features also within invaded areas (Canopy). a) Comparison of canopy and no canopy scores for PCA 1 based upon feeding guilds ( $p = 0003$ ). b) Comparison of canopy and no canopy scores for PCA 1 based upon habitat preferences ( $p = <0.01$ ). Confidence intervals represent  $\pm$  mean standard error.

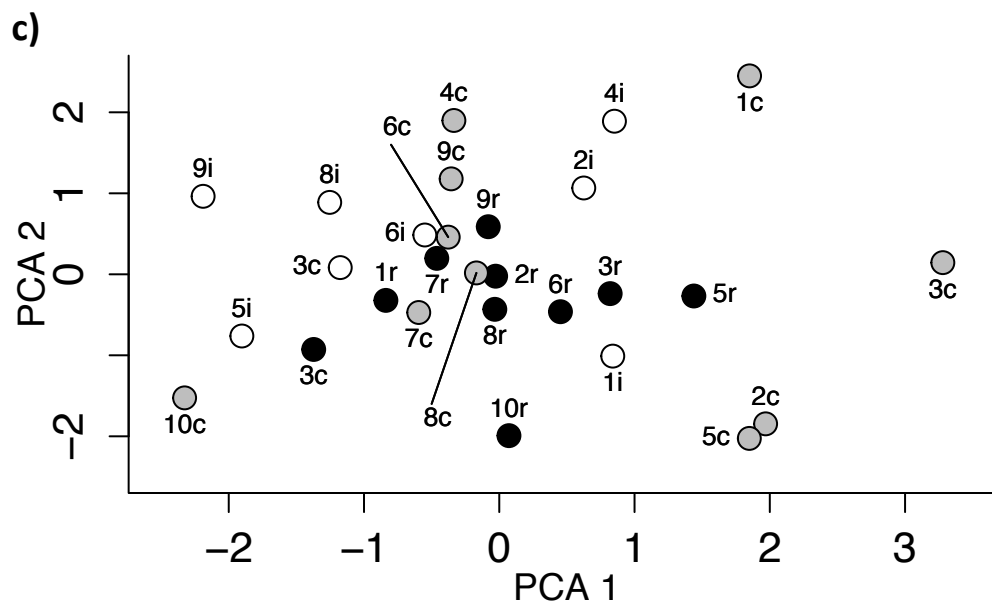
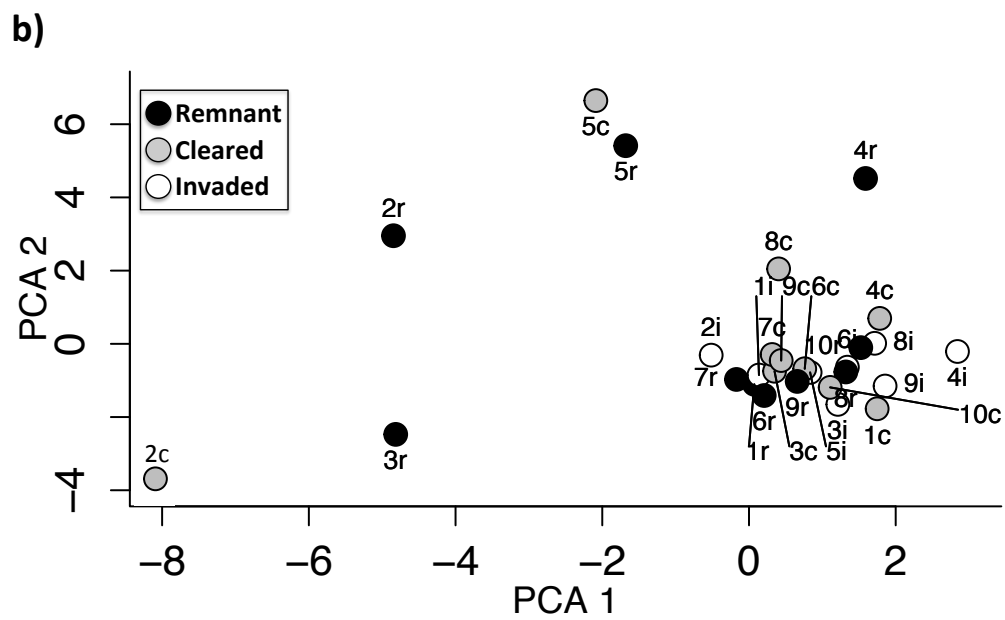
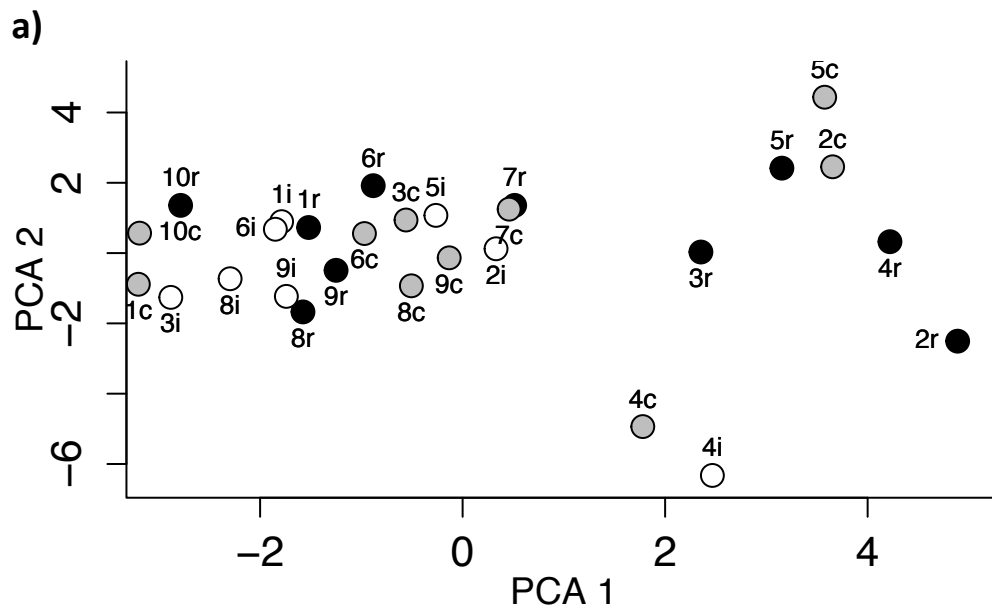


Fig. 4.6. Principal components analysis showing the change in community similarity in terms of a) total bird species richness, PCA1 variation explained 12.08 %, PCA2 variation explained 9.77 % b) total bird species density, PCA1 variation explained 12.11 %, PCA2 variation explained 11.39 %, c) mean trait values across all species PCA1 variation explained 34.73 %, PCA 2 variation explained 26.22 %. 1- Wonga Park, 2 – Greens Bush, 3 – Pantom Hill, 4 – Woods Reserve, 5 – Birdsland Reserve, 6 – Glenfern Valley Bushlands, 7 – Ferntree Gully, 8 – Red Hill, 9 – Montrose, 10 – Sherbrooke Forest. r = Remnant control – remnant bushland (black circles), c = Cleared treatments – *Pittosporum undulatum* cleared (grey circles), i = invaded control – high density *Pittosporum undulatum* (white circles).

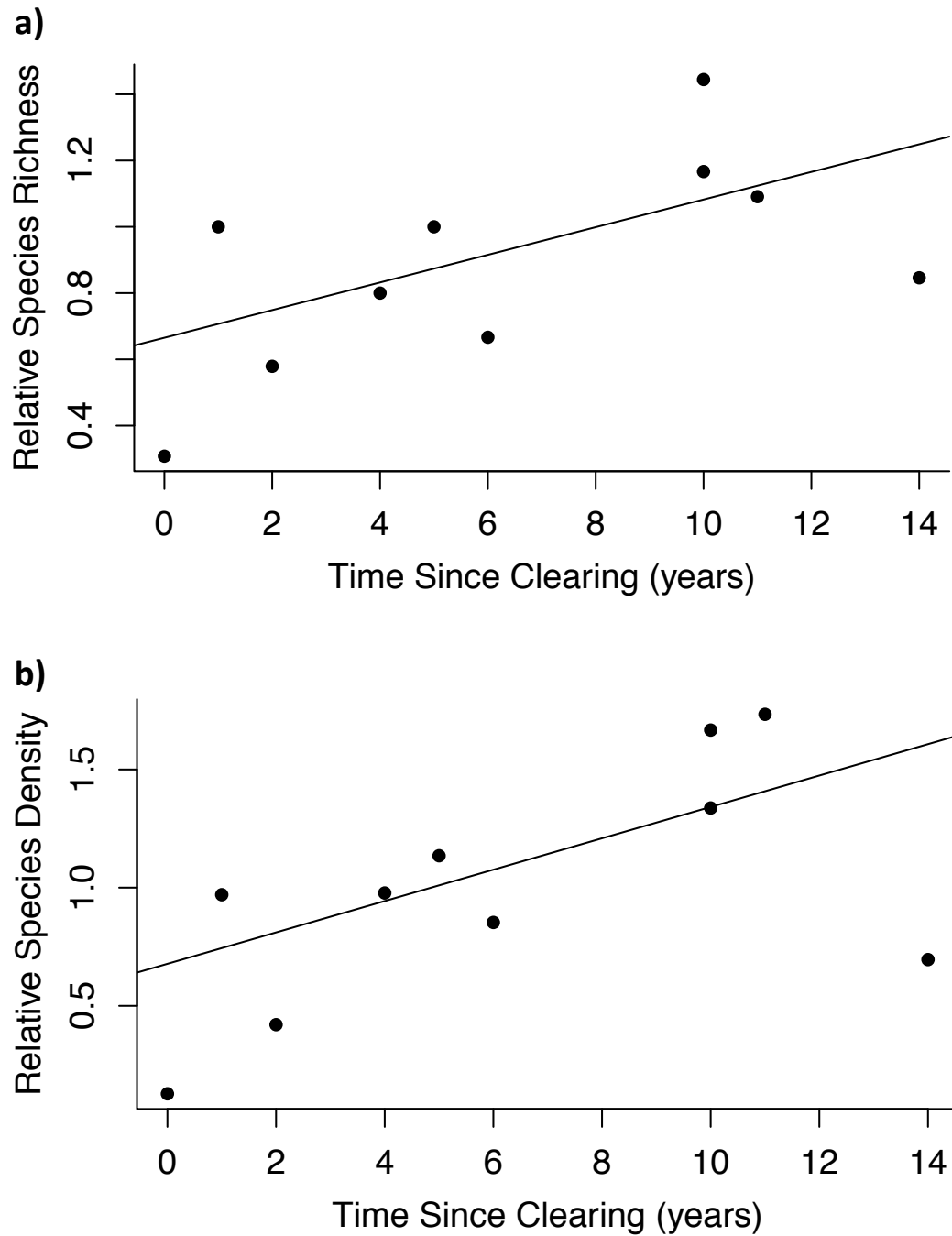


Fig. 4.7. Relationship between time since *Pittosporum undulatum* removal at cleared treatments and a) relative species richness (species richness at cleared treatment divided by species richness at remnant control) adjusted  $r^2 = 0.2749$ ,  $p = 0.068$ , b) relative species density (species density at cleared treatment divided by species density at remnant control) adjusted  $r^2 = 0.272$ ,  $p = 0.070$ . For full statistical analysis please see table 4.6.

## 4.5 Discussion

Our results showed that though species richness and the abundance of bird communities did not differ between sampling conditions, a significant difference in the functionality of bird communities at invaded areas was uncovered. The presence of dense *P. undulatum* infestations was found to reduce the number of carnivorous bird species observed at invaded areas relative to remnant and cleared conditions. A finer scale analysis focused on no canopy and canopy bird communities at invaded sites suggests that the presence of dense *P. undulatum* infestations may reduce the density of not only carnivores but also all bird feeding types. Finally, there was a suggestion of a trend for greater bird species richness and density over time following *P. undulatum* removal, but no suggestion that these communities were reverting back to a remnant state. These topics will be discussed in detail below.

### ***4.5.1 Does the functionality, richness and density of bird communities vary among sampling conditions?***

The functionality of remnant bird communities was affected by the presence of dense *P. undulatum* infestations, with fewer carnivorous species detected at invaded conditions. No difference in bird species richness and density was found among remnant, cleared and invaded conditions for resident bird communities within this study. The lack of effect of richness and density suggests native bird communities are able to sustain a similar level of species richness at invaded, cleared and remnant areas relative, regardless of any initial or legacy effects *P. undulatum* and its removal may be having on the structure and diversity of native vegetation (Gleadow & Ashton 1981; O'Leary et al. 2018). The effect of *P. undulatum* invasion on carnivores suggests a complex suite of causal interactions between flora and fauna.

A carnivore's view of prey is likely to be restricted by the dense foliage produced by the *P. undulatum* canopy (Stirnemann et al. 2014). Vegetation heterogeneity in general has been identified as being important to avian carnivore habitat, something that invasive woody trees/shrubs reduce (Stirnemann et al. 2014). This finding

supports the textual discontinuity hypothesis, which suggests that smaller birds are more commonly associated with complex, heterogeneous and fine-grained vegetation (Fischer et al. 2008; Holling 1992; Stirnemann et al. 2014). The dense architecture of the *P. undulatum* canopy is likely to restrict aggressive native temperate Australian birds, which tend more often to be larger species such as territorial honeyeaters e.g. Bell minors and Noisy minors (Ford 1979; Ford 2011), along with carnivorous Magpies and Pied Currawongs. Collectively these results suggest that forests invaded by dense *P. undulatum* populations do not provide high quality habitat for carnivorous birds. Though it is recognised that prey species such as possums are known to nest within the *P. undulatum* canopy, depauperate groundstorey habitat may limit smaller prey species such as lizards, gastropods, frogs. This in turn may also contribute to the diminished number of carnivorous bird species. The lack of predators could therefore mean *P. undulatum* infestations may support higher richness and density of smaller frugivorous, nectivorous insectivorous and grainivorous bird species, but our results suggest the effective absence of ground level habitat for many bird species beneath the *P. undulatum* canopy may also restrict these functional bird types.

Given the relative absence of undergrowth vegetation for many bird species beneath the *P. undulatum* canopy, it is striking that no significant difference in bird species richness and density was found between communities at remnant controls with suitable ground habitat, relative to bird communities at invaded and cleared areas with ground level components either absent or recovering respectively. However, with the exception of the shrub/ground layer, plant communities invaded by *P. undulatum* would be expected to provide many of the other habitat components required by native birds via other native vegetation structures, particularly through the intact *Eucalyptus* overstorey.

Previous studies examining the habitat requirements for bird communities have identified a clear relationship between avian diversity and habitat structure (Ikin et al. 2012; MacArthur & MacArthur 1961; Recher 1969; Stirnemann et al. 2014). Several important habitat components, including foliage, flowers, bark, the ground layer, air spaces and hollow bearing trees have been identified (Antos et al. 2008; Ikin et al. 2012; McElhinny et al. 2006; Stirnemann et al. 2014). These components are expected



to be present at all conditions, but with an absent shrub/ground layer for invaded sites and a regenerating ground layer in cleared conditions.

The ground layer of most temperate Australian forests are dominated by vegetation, woody debris, and logs, all which can support invertebrate communities and act as foraging substrate for native birds (Antos & Bennett 2005; McElhinny et al. 2006). *P. undulatum* is known to shade out low-level vegetation, producing a homogenous and relatively bare ground level that is unlikely to be suitable to many native temperate Australian bird species. (Gleadow & Ashton 1981; Mullett & Simmons 1995; Recher et al. 2002; Stirnemann et al. 2014). Ground foraging birds can form some of the most abundant bird communities in temperate Australia (Antos et al. 2008; McElhinny et al. 2006). Similar bird species richness and density among remnant, cleared and invaded conditions may reflect the long-term decline of ground dwelling bird species within the woodlands of Southern Australia (Antos et al. 2008; Ford et al. 2001; Ford 2011; Stirnemann et al. 2014). Fragmentation of landscapes along with urbanisation and the introduction of invasive mammalian predators has seen the gradual reduction of ground dwelling bird species from many temperate *Eucalyptus* forest and woodlands (Antos et al. 2008; Antos & Bennett 2005; Ford 2011). It is therefore possible that any loss of habitat through the deterioration of suitable ground level substrate via the presence of *P. undulatum*, may not have a great effect on ground dwelling bird species, as they may already be diminished or absent within many remnant bushland areas. Following from this, the removal of *P. undulatum* and subsequent improvement in the quality of ground level vegetation structure is unlikely to result in an increase in ground dwelling bird species richness.

It is important to note that whilst a *Eucalyptus* overstorey tends to remain present at areas of high *P. undulatum* density, invasive populations of this species which in some instances have been established for several decades, dramatically limits the germination of *Eucalyptus* seedlings, resulting in the virtual absence of younger *Eucalyptus* age classes growing to replace the mature stock (Gleadow & Walker 2014). Therefore, the quality of habitat structure utilised by bird communities across temperate southeastern Australia may become increasingly simplified as the overstorey tree species continue to age.

#### **4.5.2 Does *P. undulatum* present suitable habitat for bird species at invaded areas?**

At a more local scale within invaded sites there was significantly lower bird species richness, density and functionality within *P. undulatum* habitat (no canopy) relative to the overstorey above (canopy). These results suggest that the majority of resident bird species observed within invaded conditions preference canopy vegetation for habitat, particularly overstorey species as opposed to the simplified habitat provided by the *P. undulatum* infestation. The finding of significantly fewer insectivores, omnivores, carnivores and nectivore/granivore/frugivores, supports the hypothesis that dense *P. undulatum* populations do not act as high quality habitat for a variety of bird species, which may be found in other vegetative components of invaded areas.

Despite reports suggesting European Blackbirds may be a common vector of *P. undulatum* (Gleadow & Rowan 1982), no difference in exotic species presence or density was found across all conditions. Only 3 out of the 47 bird species observed (6.38 %) throughout the study were exotic species, and no significant difference in exotic species density was detected across conditions. In addition to an overall low density of birds found within the *P. undulatum* canopy, this result suggests that *P. undulatum* does not act as important habitat for exotic bird species.

Reports have also suggested that *P. undulatum* may provide important habitat to the Powerful Owl, a native species that is currently considered threatened across Victoria (McNabb & McNabb 2011). Four powerful owls were sighted in two separate locations throughout the survey process. At each location Powerful Owls were perched on trees considered locally indigenous to the region. At Woods Reserve Owls were sighted within the remnant control (roosting in a mature Messmate, *Eucalyptus Obliqua*), and also within the cleared area of Birdsland Reserve (roosting in mature blackwoods, *Acacia melanoxylon*). These results suggest that Powerful Owls are capable of returning to areas where invasive tree control has occurred. The retention of established tall eucalypt canopy and subcanopy trees suitable for roosting is likely an important habitat component for this species. A more specific, targeted study examining the effects of *P. undulatum* removal on the Powerful Owl is suggested. It is

possible that other rare native resident bird species may utilise *P. undulatum* as a resource, although that was not observed in this study

#### **4.5.3 How do bird communities respond to *P. undulatum* removal?**

Though not significant to the convention of  $\alpha = 0.05$ , results provide a suggestion of higher bird species richness and the overall density with time since *P. undulatum* control. No correlation was observed between the relative similarity of bird communities present at the remnant control and cleared areas across sites. Given the limited number of exotic species observed, an increase in bird species richness and density implies that *P. undulatum* removal may increase native avian biodiversity over time. As native vegetation re-establishes it can potentially provide more resources, habitat and over time greater niche diversity for native bird species. Although overall bird species richness and density does appear to increase with time since *P. undulatum* removal, this increase does not necessarily produce a bird community more similar to that found within remnant controls. Two possible explanations for this response indicate either the removal of *P. undulatum* has altered the bird community in such a way as to increase overall bird richness and density, but not in such a way as to render cleared communities more similar to remnant controls, or, alternatively bird communities at remnant and invaded conditions are similar prior to the removal of *P. undulatum*, and though the removal of this invasive tree may increase the richness and the density of bird communities, remnant and cleared bird communities maintain a consistent level of similarity post *P. undulatum* control. If the later alternative is correct, it would support the idea that both *P. undulatum* invasion and its removal does not have a strong impact on the composition and structure of resident bird communities within temperate *Eucalyptus* forests. If correct, this alternative would add further weight to the idea that shrub/ground level habitat may be relatively less important to the persistence of local bird communities, due to the longterm decline of ground based bird species, particularly in urban and agricultural environments (Antos et al. 2008; Ford 2011; Ford et al. 2001; Stirnemann et al. 2014).

#### **4.5.4 Conclusion**

Information gathered here offers details distinct from those uncovered from a monitoring program focusing specifically on vegetation alone. While the outcomes from this study have potential to improve the management of other restoration programs focused on invasive trees, the effect of invasive tree removal will differ depending upon the habit and ecology of the tree/s in question, together with their relationship with the surrounding environment. Application of the monitoring program presented here, or something similar is suggested to improve the capacity of invasive tree removal projects to monitor the effects of restoration efforts.

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## 5.0 General discussion and conclusion

This thesis explores the issues surrounding the history and management of invasive species across Australia and the world with application to *Pittosporum undulatum* as a model. Globally, invasive species research and management has evolved in response to the growing ecological, economic and social risks posed by invasive species. The work presented here, focused on *P. undulatum* management, complements and develops upon much of this earlier research (Gleadow & Ashton 1981, Mullet 1996, Rose 1997). Context is also given for future directions in an increasingly globalised, interconnected world with a changing climate.

This thesis has focused on four main research aims. The first aim, addressed in chapter one was to introduce the reader to the concepts of native and invasive species, and how perceptions towards native and invasive species have changed over time. With specific reference to Australia and *P. undulatum* this chapter discussed how these changing perceptions have resulted in fluctuating management of native and non-native species. With a warming climate and increasing human disturbance, further changes will be required to manage ecological communities into the future. The second aim, addressed in chapter two was to examine variability in the breeding ecology and reproductive traits of *P. undulatum* across native and invasive regions. Findings of approximate parity in the sex ratio of native and invasive *P. undulatum* populations across southeastern Australia were in contrast to earlier research. This outcome has implications for the way in which established theory is incorporated into invasive species management. The third and fourth aims, addressed in chapters three and four were to observe the response of plant and bird communities respectively, following the removal of invasive *P. undulatum* populations over time. The re-establishment of native plant communities following *P. undulatum* removal suggests that the species is acting as a driver of change. However, bird communities did not differ strongly across sampling conditions. These findings underline the importance of incorporating theory into the practise of restoration, along with the value of goal-orientated projects with the potential to achieving multiple benefits. Implications of these results are discussed broadly below.

Australia's biogeographic remoteness has allowed it to evolve in relative isolation from invasive species (Woinarski et al. 2015). As illustrated in chapter one, European arrival and the ensuing introduction of thousands of exotic species have resulted in significant and widespread ecological degradation (Bellard et al. 2016; Fensham & Laffineur 2019; Woinarski et al. 2015). Over the past 200 years, approximately 10 % of Australia's 273 endemic terrestrial mammals have been lost to extinction, with a further 21 % now considered threatened (Woinarski et al. 2015). Acclimatization societies that sought to establish species in different locations all over the world were popular in the 19th century (Cook & Dias 2006; Osborne 2000). Although such societies now no longer operate, the practise of importing and establishing populations of exotic and in some cases invasive species remains prevalent across Australia (Groves & Willis 1999; Invasive Plants and Animals Committee 2016).

It can be argued that agricultural processes have generated species that possess at least some invasive characteristics, e.g. Johnsongrass (*Sorghum halepense*) (Sezen et al. 2016), weed beet (*Beta vulgaris subsp. vulgrais*) (Ellstrand et al. 2010) and several forestry species (Barney & DiTomaso 2008, Richardson et al. 2008). These species have been artificially selected for their capacity to grow rapidly in monoculture populations and outcompete rival species (Barney & DiTomaso 2008). Moreover, through land clearing and farming practices humans have actively favoured these crops over native species. This practice has had extraordinary economic and health benefits (Australian Department of Agriculture and Water Resources 2018; Hawkes & Ruel 2006), but at the same time has been detrimental to our environment through processes such as habitat loss, reduced connectivity, species loss and altered disturbance regimes (Bradshaw 2012; Kanowski 2017). Increased efforts to grow native crop plants that are suited to local conditions, including the use of wild crop relatives should be explored (Pascoe 2014, Warschefsky et al. 2010), (see chapter one). Beyond agricultural practices, the world of horticulture continues to transport known invasive species across the planet (Reichard & White 2001). Australians have few restrictions as to what they can grow in their backyards. As discussed in chapter one, even though billions are spent to conserve and protect remnant bushland sites across the globe, particularly in western nations such as Australia, backyard escapees

will continue to present an ongoing threat to these sites and overall conservation efforts (Invasive Plants and Animals Committee 2016; Pemberton & Liu 2009; Reichard & White 2001). Real action is needed to conduct research and educate people about the benefits of their local biodiversity and risks that backyard escapees pose (van Kleunen et al. 2018). Scientific processes and frameworks such as a simple species-monitoring program and the Passenger Driver Hypothesis demonstrated throughout this thesis, in particular chapter three, help to provide examples of pathways to better connect people with their local environment.

Australia is recognised for developing one of the most successful biosecurity programs worldwide (Hoffmann & Broadhurst 2016). Although the program is well established to prevent the importation of threatening plants and animals, the targeted focus of protecting agricultural industries means that Australia's native plant and animal communities are particularly exposed to those introduced species that remain undetected (Alacs & Georges 2008; Tovar et al. 2017). For example, Mexican Feathergrass (*Nassella tenuissima*) is thought to have been illegally introduced to Australia and sold online in 2008 (Invasive Species Council 2019). Introductions such as this are not uncommon and result in an ongoing reduction to the ecological integrity of Australian ecosystems (Alacs & Georges 2008). More work is required to properly acknowledge the role of globalisation in facilitating the spread of invaders, and develop adequate policies to address it (Hulme 2009; van Kleunen et al. 2018). Australia is not alone in the globalised spread of invaders, with all nations across the planet exposed to this threat. *Myrica faya* for example, has invaded throughout the Hawaiian montane forests, elevating canopy nitrogen and water content (Asner & Vitousek 2005), whilst *Fallopia spp.* invasion through European riparian environments have seen a reduction in plant richness and invertebrate biomass (Gerber et al. 2008). Greater incorporation of biosecurity principals into infrastructure projects, along with education surrounding the value of native species and the impact of invaders will assist in reducing the spread of invasive species (Ding et al. 2008).

An important aspect of invasive plants is their high reproductive potential (Simberloff 2009). This is not necessarily straight forward, as there may initially be only a few individuals at the invasion front limiting the possibility of out crossings. Baker's Law

states that self-compatible species should predominate as island colonisers, as these species have the capacity for uniparental reproduction through self-pollination (Baker 1955). This same concept can be applied to the front of an expanding invasive population (refer to chapter two). Following Baker's Law, the facilitated spread of invasive species through globalisation should see the growing prominence of self-compatible individuals at the invasion front (Barrett et al. 2008; Grossenbacher et al. 2017; Pannell et al. 2015). Studies of metapopulations support this hypothesis, with some proposing that the prominence of hermaphroditism in certain species may be the result of evolutionary selective pressure for reproductive assurance in individuals colonising new environments (Pannell 1997; Pannell et al. 2015). However, in circumstances where mates are available this pressure may be reduced. Despite the rate of first recorded arrivals of invasive species continuing to rise (Seebens et al. 2017), the majority of invasive propagules transported through globalisation are more likely to be common invaders that have already spread across the local environment. It is therefore possible that Baker's Law may have less application today for these common invaders, as the high level of propagule pressure associated with globalisation means that new arrivals are likely to establish as part of an existing population rather than in isolation. Support for this can be observed in the results of chapter two, where findings of a strong persistence of females within all sampled *P. undulatum* populations, together with observations of hermaphrodites generally fulfilling the male role, suggest there is no strong pressure for self-fertilisation.

The pressures of globalisation, disturbances from land use change and a warming climate are not only likely to exacerbate the problem of invasive species, but may also result in native species spreading beyond their historical distributions (Carey et al. 2012a; Pivello et al. 2018; Simberloff et al. 2012). As climates change, the distribution envelope of many native species is shifting (Niskanen et al. 2019). This is exemplified by the spread of many alpine species towards the poles (Mason et al. 2015). The geographical shift of suitable climate envelopes can create the potential for native species to spread into previously unsuitable territories (Niskanen et al. 2019; Webber & Scott 2012). At the regional level changing climates, particularly in conjunction with other disturbances can therefore reduce dispersal barriers to spread for native species in a similar way to globalisation (Cunningham et al. 2016; Seebens

et al. 2018). One example of an apparent native invader is that of *Pittosporum undulatum* (Gleadow & Ashton 1981; S Rose 1997; O’Leary et al. 2018), the subject of this thesis (see Chapter one). It is important to note that a changing climate is just one of several drivers, including altered disturbance regimes, land clearing, introduced vectors and horticultural propagation, behind the continued spread of *P. undulatum* across Australia and the globe (O’Leary et al. 2018) (see Chapter one). As climates change it is worth considering how these same drivers may influence the emergence of new native invaders and if the responses of native invaders are different to those of exotic invaders.

Increasing pressure is being placed on restoration and conservation programs to sustain global biodiversity and the services they provide. A growing view within the academic community is that management should not be directed towards the restoration of pristine remnant bushland, or allocating precious resources to unrealistic conservation programs (Bottrill et al. 2008; Hagerman & Satterfield 2014; Hobbs 2007). Rather, they should identify key community structures, ecological functions, processes and threats, along with how they can best be managed to maintain resilient communities that provide ecosystem services, community structure and function (Funk 2008; Rohr et al. 2018; Suding 2011). Work completed throughout chapters three and four demonstrate practical pathways to identify some of these functions and processes, and provide suggestions towards their management. Successful conservation and restoration programs are not always aiming to achieve the same thing, however, integrating these programs can help to both restore key habitat whilst also preserving endangered species (Harvey et al. 2017; Jones et al. 2016; Volis 2019). For example, a program to conserve three South American turtle species has not only helped to improve the status of the turtles, but also provided stability to the surrounding ecological food web through a range of unintended benefits for non-target species (Campos-Silva et al. 2018). Equally, work to remove invasive mammals from 181 islands across the globe seeks to redefine the structure of local communities (Jones et al. 2016). An outcome of this work is the likely benefit to several highly threatened bird, mammal and reptile species. The goals of both endangered species conservation and ecological community restoration can therefore

achieve universally beneficial outcomes by approaching a common problem from different perspectives (Volis 2019).

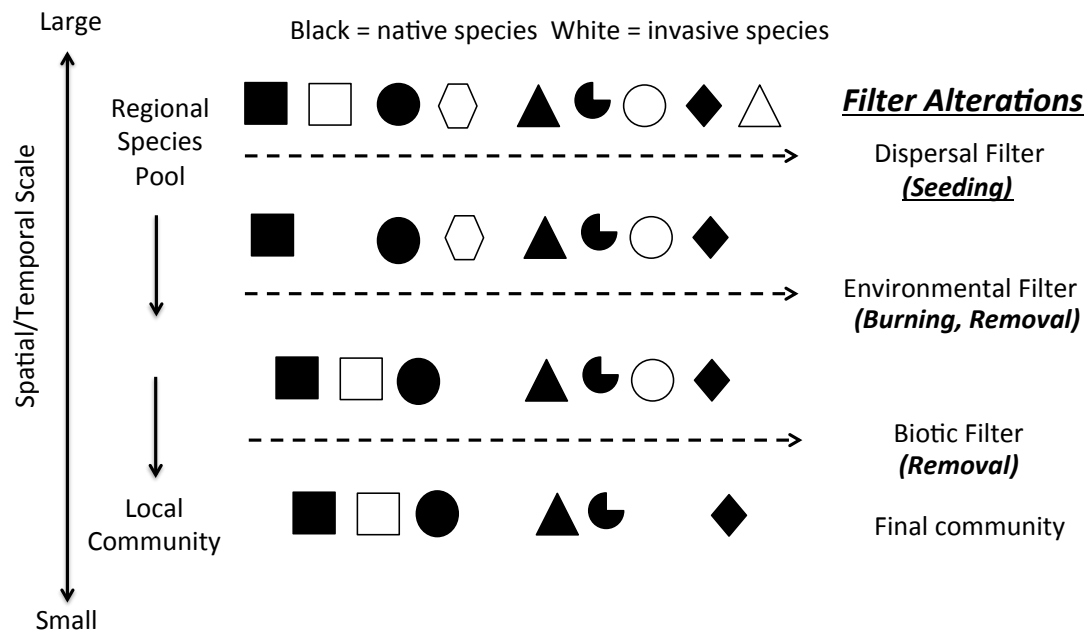


Fig 5.1. Adaptation from Funk (2008). Figure demonstrates the barriers to invasion described in community assembly rules, along with how these can be manipulated (italicised text) through community restoration programs.

A key component of any restoration or conservation program must be the implementation and development of appropriate scientific theory (Catford et al. 2009). Chapters three and four discuss the historic disjunction between the science of invasive species management, and the practice of ecological restoration. Research should provide greater direction toward management action (Dickens & Suding 2013; Hobbs & Norton 1996; Matzek et al. 2014; Temperton et al. 2004). At the same time, practical application of ecological restoration must be based upon scientific principles, with adequate follow up monitoring to ensure programs can be evaluated and improved upon (Hobbs & Norton 1996; Suding 2011). An example of theory developed for management purposes can be seen in Temperton et al. (2004), and advanced by Funk (2008), (See figure 5.1). The development of community assembly theory, and its advancement as a methodology to restrict invasive species, whilst encouraging native species and specific trait types demonstrates a practical, theory

based process to develop resilient communities (Funk 2008). In order to develop this more broadly, additional work is required on a range of topics, including an improved understanding of the impacts of ecological restoration practices on soil e.g. carbon sequestration, water retention (Filoso et al. 2017), soil structure, micro biota (Strickland et al. 2017) etc., along with how to restore communities resilient to climate change (Locatelli et al. 2015).

Finally, in light of the fluctuating perceptions towards invasive species and native invaders discussed in chapter one, it is worth considering how current management actions towards species such as *P. undulatum* may be viewed in the near future. An apparent dichotomy of views exist regarding *P. undulatum*, with some considering the species a key component of local communities (Lester 2016; Miles 2006), whilst others demonise it as an destructive invader (Gleadow & Ashton 1981; Gleadow & Walker 2014; Hall 2004). Regardless of these perceptions, the species is well recognised for its high degree of plasticity and high growth rate across a variety of conditions, including marginal lands (Gleadow 1982; Mullett 1996). Climate change is already placing significant pressure on plant communities globally. Based upon current trajectories possible temperatures are expected to rise by 3.7°C to as much as 7.8°C above pre industrial levels by 2100 (Edenhofer et al. 2014). Temperatures of this magnitude are likely to dramatically impact all aspects of life, with many species expected to be unable to withstand the associated changes in climate, weather extremes and natural disasters (Wiens 2016; Urban 2015). Additional focus on the importance of greening urban areas as a climate change mitigation strategy (Norton et al. 2015; Pramova et al. 2012) and to improve human health (Mills et al. 2019) is already challenging cities, planners and communities to find adaptive plant species suitable to urban environments. This challenge will likely see societies across the global prioritise shade bearing plant species such as *P. undulatum*, that are capable of surviving in the hostile environment of marginal and urban areas. Some propose the use of climate-adjusted provenancing to select local species best adapted to future climates (Butterfield et al. 2017; Prober et al. 2015). However, for some regions it may be that there are few trees suitable to future climates and the presence of a vigorous tree such as *P. undulatum* to provide shade may be highly sought after. As global warming intensifies climate stressors, and the expansion of urban areas impact



the health of increasingly more people across the globe, species selection in urban forests will comprise the most suitable species, regardless of their origin (Sjöman et al. 2016).

This work shows that global changes to climate and rates of propagule pressure will place increasing demands on native communities to maintain ecological value and services. Both native and invasive species alike will face these demands. The value of native species and their adaptations to local conditions are likely to grow, though movement of native species may see the rise of native invaders. In contrast, the threat posed by invasive species is expected to be exacerbated by global changes, though as we move into an era of increasing climate instability the benefits these species provide may exceed their costs. Collectively this work prompts us to consider what constitutes a native and invasive species, how we value these labels, and how the importance of native and invasive species is likely to change in the future.

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## 6.0 Appendices

### *Appendix 3a*

#### References for leaf SLA trait values

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### Appendix 3b

Presence and absence of plant species observed at conditions within each study site. Column headings relate to site name in table 3.1.

Scientific name	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Acacia_acletoestissima</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia_baileyana</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia_dealbata</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Acacia_decurrens</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Acacia_elata</i>	1	1	0	0	0	0	0	0	0	0	1	0	0	0	0
<i>Acacia_genistifolia</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0
<i>Acacia_leprosa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia_longifolia</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia_meunsi</i>	1	0	0	0	1	0	0	1	1	1	0	0	0	0	1
<i>Acacia_melanoxylon</i>	1	0	1	1	1	1	0	1	0	1	1	1	1	1	1
<i>Acacia_mucronata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia_myrtifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia_paradoxa</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0
<i>Acacia_pycnantha</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Acacia_stricta</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Acacia_verticillata</i>	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Acacia_novae_zelandiae</i>	0	0	0	1	1	1	1	1	0	1	1	0	1	0	0
<i>Acrotriche_prostrata</i>	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0
<i>Acrotriche_serrulata</i>	1	1	1	1	1	0	1	0	0	1	1	1	0	0	1
<i>Adiantum_aethiopicum</i>	1	1	1	0	1	1	0	0	0	0	0	0	1	0	0
<i>Agapanthus_praecox</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Aira_caryophylla</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Aira_spp.</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0
<i>Allocasuarina_littoralis</i>	0	0	1	0	0	0	0	0	1	0	0	1	0	0	0
<i>Amperea_xiphoclada</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Amyema_pendula</i>	0	0	0	1	1	0	1	0	0	0	1	0	0	0	0
<i>Anagallis_arvensis</i>	0	0	0	0	0	0	0	1	0	0	0	0	1	1	1
<i>Anthoxanthum_odoratum</i>	1	1	0	1	1	1	1	1	0	1	1	0	1	1	1
<i>Arthropodium_milleflorum</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0

<i>Scientific_name</i>	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Arthropodium_strictum</i>	1	1	0	0	0	0	1	1	0	1	1	0	1	0	
<i>Asparagus_asparagoides</i>	0	0	0	0	0	0	0	1	1	0	0	0	0	0	
<i>Asparagus_scandens</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Asperula_conferta</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Asperula_eurphylla</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Asperula_scoparia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Asplenium_bulbiferum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Asplenium_flabellofolium</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Astroloma_humifusum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Atherosperma_moschatum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Australina_pusilla</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Austrocynoglossum_latifolium</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	1	
<i>Austrostipa_mollis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Austrostipa_puinodis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Austrostipa_rudis</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	1	
<i>Austrostipa_spp.</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Austrostipa_stipoides</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Avena_fatua</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	
<i>Bedfordia_arborescens</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Billardiera_heterophylla</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Billardiera_mutabilis</i>	1	1	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Billardiera_scandens</i>	0	0	0	1	0	0	1	0	0	1	1	0	0	0	
<i>Blechnum_nudum</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Bossiaea_cinerea</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Bossiaea_prostrata</i>	0	0	0	0	0	0	0	0	0	1	0	0	1	0	
<i>Briza_maxima</i>	0	0	0	0	0	0	1	1	0	1	0	0	0	0	
<i>Briza_minor</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Brunonia_australis</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0	
<i>Bulbine_bulbosa</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0	
<i>Burchardia_umbellata</i>	1	1	0	1	0	0	1	0	0	1	0	0	0	0	
<i>Bursaria_spinosa</i>	0	0	0	0	1	0	1	1	1	0	0	0	1	0	
<i>Caladenia_camea</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Caladenia_latifolia</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Caladenia_transitoria</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Calectasia_spp.</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	

<i>Scientific_name</i>	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Calochaena_dubia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Calystegia_marginata</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Cardamine_tenuifolia</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Carex_breviculmis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Carex_inversa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Carex_lynx</i>	0	0	0	1	1	0	0	0	0	0	0	0	0	0	
<i>Cassinia_aculeata</i>	1	1	0	1	1	0	1	1	0	1	0	1	1	1	
<i>Cassinia_longifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Cassinia_trinerva</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Cassythamelantha</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Cassythapubescens</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Cenchrus_dandestinus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Centaurium_erythraea</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Centaurium_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Centaurium_tenuiflorum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Centella_cordifolia</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Centrolepis_fascicularis</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Cerastium_glomeratum</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Cerastium_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	1	0	
<i>Chamaescilla_corymbosa</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Chiloglottis_valida</i>	1	0	0	1	0	0	0	0	0	0	1	0	0	0	
<i>Chrysanthemoides_monilifera</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Chrysoccephalum_spp.</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Clematis_aristata</i>	1	1	1	1	1	1	0	0	1	0	0	0	0	0	
<i>Clematis_microphylla</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Comesperma_volubile</i>	1	1	0	1	0	0	1	0	0	1	0	0	0	0	
<i>Coprosma_quadrijfida</i>	1	1	1	1	1	1	0	0	1	0	0	1	1	1	
<i>Coprosma_repens</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Coronidium_monticola</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Coronidium_scorpiodies</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Correa_reflexa</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Corybas_incurvus</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Corybas_spp.</i>	0	0	0	1	0	0	0	0	1	0	0	0	0	0	
<i>Craspedia_variabilis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Crategeus_monogyna</i>	0	1	0	0	0	0	0	0	1	0	0	0	0	0	

<i>Scientific_name</i>	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Cryptostylis_subulata</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Gyathea_australis</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Cynoglossum_suaevolens</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Cytisus_scoparius</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Dactylis_glomerata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Daucus_carota</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Daucus_glochidiatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Daviesia_leptophylla</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Desmodium_gunnii</i>	0	0	0	1	1	0	0	0	0	0	0	0	1	0	
<i>Desmodium_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Deyeuxia_quadriseta</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Deyeuxia_spp.</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Dianella_admixta</i>	1	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Dianella_laevis</i>	1	1	0	0	0	0	0	0	0	0	1	0	1	0	
<i>Dianella_revoluta</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0	
<i>Dianella_tasmanica</i>	0	0	0	1	1	0	0	1	0	0	0	0	0	1	
<i>Dichelachne_micrantha</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Dichelachne_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Dichondra_repens</i>	1	1	0	1	1	1	0	1	1	0	0	0	1	1	
<i>Dicksonia_antarctica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Dilwynia_cinereascens</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Dilwynia_glaberrima</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Dilwynia_spp.</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Dipodium_punctatum</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Ditrichia_graveolens</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Drosera_aberrans</i>	1	1	0	1	0	0	1	0	0	0	0	0	0	0	
<i>Drosera_auriculata</i>	1	1	1	0	0	0	1	1	0	0	0	0	0	0	
<i>Drosera_peltata</i>	0	0	0	1	0	0	1	0	1	1	0	0	0	0	
<i>Drosera_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Echinopogon_ovatus</i>	0	0	0	0	1	0	0	1	0	0	0	0	0	0	
<i>Ehrharta_erecta</i>	1	1	0	0	1	0	0	0	0	0	0	0	0	0	
<i>Ehrharta_longifolia</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Epacris_impresa</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Eragrostis_brownii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Erica_lusitanica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	

Scientific_name	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Eucalyptus_cypellocarpa</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Eucalyptus_dives</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Eucalyptus_goniocalyx</i>	0	1	1	0	0	0	1	1	1	0	0	0	1	0	
<i>Eucalyptus_macrohyrcha</i>	1	1	0	0	0	0	1	0	1	0	0	0	0	0	
<i>Eucalyptus_meliodora</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	1	
<i>Eucalyptus_obliqua</i>	1	1	1	1	1	1	1	0	1	1	1	1	1	0	
<i>Eucalyptus_ovata</i>	0	0	0	1	1	0	0	0	0	1	1	0	1	1	
<i>Eucalyptus_polyanthemos</i>	1	0	0	0	0	0	1	1	0	0	0	0	0	0	
<i>Eucalyptus_pryoriana</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Eucalyptus_radiata</i>	0	1	1	1	1	0	0	0	1	1	1	0	1	0	
<i>Eucalyptus_regans</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Eucalyptus_viminalis</i>	0	0	0	0	0	1	0	0	0	1	1	0	0	0	
<i>Euchiton_involucratu</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Euchiton_japonicus</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Exocarpos_cupressiformis</i>	1	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Exocarpos_strictus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Ficinia_nodosa</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Gahnia_radula</i>	1	1	1	0	0	0	1	0	0	0	1	1	1	0	
<i>Gahnia_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Galium_aparine</i>	0	0	0	1	1	0	0	1	0	0	0	0	0	1	
<i>Galium_gaudichaudii</i>	0	1	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Galium_propinquum</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Galium_spp.</i>	0	0	0	1	1	0	1	1	1	0	0	0	0	0	
<i>Geranium_potentilloides</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Geranium_solanderi</i>	0	0	0	1	1	0	0	0	0	0	0	0	0	0	
<i>Geranium_sp._2</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Geranium_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	1	1	
<i>Gladiolus_undulatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Glossodia_majior</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Glycine_clandestina</i>	0	0	0	1	1	0	0	0	0	0	0	0	1	1	
<i>Glycine_microphylla</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Glycine_tabacina</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Gonocarpus_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Gonocarpus_tetragynus</i>	1	1	0	1	0	0	1	1	0	1	1	0	1	0	
<i>Goedenia_lanata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	



Scientific_name	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Goodenia_ovata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Hakea_decurrens</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Hardenbergia_violacea</i>	0	1	0	0	0	0	1	1	0	0	0	0	0	0	
<i>Hedera_helix</i>	0	1	1	0	0	0	0	1	0	0	0	0	0	0	
<i>Hedycarya_angustifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Hemarthra_uncinata</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Hibbertia_riparia</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Histiopteris_incisa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Holcus_lanatus</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	1	1
<i>Hovea_heterophylla</i>	1	1	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Hydrocotyle_foveolata</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Hydrocotyle_geraniifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Hydrocotyle_hirta</i>	1	1	0	0	0	0	0	1	0	0	0	0	1	0	
<i>Hydrocotyle_laxiflora</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Hydrocotyle_sibthorpioides</i>	0	0	0	1	1	0	0	0	0	0	0	0	0	0	
<i>Hydrocotyle_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Hypericum_gramineum</i>	1	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Hypochaeris_glabra</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Hypochaeris_radicata</i>	1	1	0	1	0	0	1	0	0	1	0	0	1	1	
<i>Hypolepis_glandulifera</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Ilex_aquifolium</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Imperata_cylindrica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Indigofera_australis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Isolepis_cernua</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Juncus_pallidus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Kunzea_leptospermoides</i>	1	1	0	1	0	0	1	0	0	0	0	0	0	0	
<i>Lachnagrostis_spp.</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Lagenophora_gracilis</i>	1	0	0	0	0	0	0	0	0	1	1	0	1	0	
<i>Lagenophora_stipitata</i>	1	1	0	1	0	0	0	1	0	1	1	0	0	0	
<i>Leontodon_saxatilis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Lepidosperma_concavum</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Lepidosperma_elatus</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	1	
<i>Lepidosperma_filiforme</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Lepidosperma_laterale</i>	0	0	0	0	0	1	0	0	0	0	0	1	0	0	
<i>Lepidosperma_longitudinale</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	

Scientific_name	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Lepidosperma_spp.</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0	
<i>Leptospermum_continentale</i>	0	0	1	1	1	0	0	0	0	1	1	1	1	0	
<i>Leptospermum_lanigerum</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	0	
<i>Leptospermum_scoparium</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Leptospermum_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Leucochrysium_albicans</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Leucopogon_parviflorus</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Leucopogon_virgatus</i>	1	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Lobelia_anceps</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Lomandra_filiformis</i>	1	1	1	1	1	1	1	1	0	1	1	1	1	1	
<i>Lomandra_longifolia</i>	1	1	0	1	1	1	1	0	0	0	0	0	0	0	
<i>Lomandra_micrantha</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	0	
<i>Lomandra_multiflora</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Lomatia_ilicifolia</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	
<i>Luzula_campensis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Luzula_meridionalis</i>	0	1	0	0	1	0	0	0	0	0	0	0	0	0	
<i>Luzula_spp.</i>	0	0	0	1	1	0	0	1	0	0	0	0	0	0	
<i>Medicago_polymorpha</i>	0	0	0	0	0	0	0	0	1	0	0	0	1	0	
<i>Medicago_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Melaleuca_squarrosa</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Microlaena_stipoides</i>	1	1	1	1	1	1	1	1	1	1	1	1	1	1	
<i>Microseris_sp.3</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Microsorium_pustulatum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Microtis_arenaria</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Microtis_spp.</i>	0	0	0	0	0	0	0	0	1	1	0	0	1	0	
<i>Microtis_unifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Muellerina_eucalyptoides</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Myosotis_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Myosotis_sylvatica</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0	
<i>Nicotiana_glauca</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Olearia_argophylla</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Olearia_lirata</i>	1	1	0	1	1	0	0	0	0	0	1	0	0	0	
<i>Olearia_myrsinoides</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0	
<i>Olearia_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Onopordum_acanthium</i>	0	0	0	1	0	0	0	1	0	1	0	0	1	0	

Scientific_name	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Opercularia_ovata</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Opercularia_varia</i>	1	1	0	0	0	0	0	1	0	1	1	0	0	0	
<i>Oxalis_corniculata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Oxalis_exilis</i>	1	0	0	0	0	0	0	0	0	1	1	0	0	0	
<i>Oxalis_incarната</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Oxalis_perennans</i>	1	1	0	1	1	1	1	1	1	1	1	1	1	1	
<i>Oxalis_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	1	1	
<i>Ozothamnus_ferrugineus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Pandorea_pandorana</i>	0	0	1	0	0	0	1	1	1	0	0	0	0	1	
<i>Parsonsia_brownii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Paspalum_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Paterersonia_occidentalis</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Pelargonium_inodorum</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	
<i>Pimelea_axiflora</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Pimelea_cuviffora</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Pimelea_humilis</i>	1	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Pinus_radiata</i>	1	1	1	0	0	0	0	1	1	0	0	1	0	0	
<i>Pitiosporum_undulatum</i>	1	1	1	1	1	1	1	1	1	1	1	1	0	0	
<i>Plantago_lanceolata</i>	1	0	0	0	1	0	0	0	0	1	0	0	1	1	
<i>Plantago_spp.</i>	0	0	0	1	0	0	0	0	0	0	0	0	1	0	
<i>Plantago_varia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Platylodium_montanum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Platylodium_obtusangulum</i>	0	0	0	1	0	0	1	0	0	0	0	0	0	0	
<i>Poa_annua</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Poa_clelandii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Poa_ensiformis</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	1	
<i>Poa_labillardierei</i>	0	0	0	1	1	0	1	1	0	1	0	0	1	1	
<i>Poa_morrisii</i>	0	1	0	1	1	0	0	1	0	0	0	0	1	0	
<i>Poa_rodwayi</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	
<i>Poa_sieberiana</i>	1	1	1	1	0	1	1	0	0	1	0	0	0	0	
<i>Poa_spp.</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	0	
<i>Poa_tenera</i>	0	0	0	0	1	0	0	0	0	0	0	0	1	0	
<i>Polyphlebium_venosum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Polyscias_sambucifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Polystichum_proliferrum</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	

<i>Scientific_name</i>	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Pomaderris_aspera</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Paranthera_microphylla</i>	1	0	0	1	1	0	1	1	0	1	0	0	1	1	1
<i>Paranthera_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Potentilla_indica</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Prostanthera_lasianthos</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Prunella_vulgaris</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Pteridium_esculentum</i>	1	1	1	1	1	1	1	0	0	1	1	1	0	0	0
<i>Pteris_tremula</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Pterostylis_alpina</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Pterostylis_melagramma</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Pterostylis_nutans</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Pterostylis_spp.</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Pultenaea_gunnii</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Pultenaea_scabra</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Quercus_robur</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Ranunculus_lappaceus</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Ranunculus_plebeius</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Ranunculus_spp.</i>	0	0	0	1	0	0	0	1	0	1	0	0	0	0	0
<i>Ranunculus_rosea</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Rosa_rubiginosa</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Rubus_fruitosus_spp._agg.</i>	1	1	0	1	1	1	1	1	1	1	0	1	1	1	1
<i>Rubus_parvifolius</i>	0	0	0	0	1	1	1	0	0	0	0	0	1	1	1
<i>Rytidosperma_caespitosum</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Rytidosperma_pallidum</i>	1	1	0	0	0	0	1	1	0	1	0	0	0	0	0
<i>Rytidosperma_pilosum</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Rytidosperma_racemosum</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Rytidosperma_spp.</i>	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0
<i>Sambucus_gaudichaudiana</i>	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0
<i>Schoenus_apogon</i>	1	0	0	0	0	0	0	0	0	1	1	0	1	1	1
<i>Senecio_glomeratus</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
<i>Senecio_hispidulus</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Senecio_linearifolius</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Senecio_minimus</i>	1	1	0	0	0	0	1	1	0	0	0	0	1	0	0
<i>Senecio_phelleus</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Senecio_prenanthoides</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0

Scientific_name	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Senecio_quadridentatus</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0	
<i>Sigesbeckia_orientalis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Solanum_aviculare</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Solanum_nigrum</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Solanum_prinophyllum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Sonchus_oleraceus</i>	0	0	0	0	1	0	0	0	0	0	0	0	0	0	
<i>Sonchus_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Sporobolus_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Stackhousia_monogyne</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Stellaria_flaccida</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Stellaria_pungens</i>	0	0	0	1	1	0	0	0	0	0	0	0	0	0	
<i>Stylidium_graminifolium</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Taraxacum_spp.</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Tetrarhena_junceae</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Tetratheca_ciliata</i>	0	1	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Thelymitra_ixioides</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Thelymitra_spp.</i>	0	0	0	0	0	0	1	1	0	1	0	0	0	0	
<i>Themeda_triandra</i>	0	0	0	0	0	0	0	0	0	1	0	0	1	0	
<i>Thysanotus_pateronii</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Thysanotus_tuberosus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Tricoryne_elatior</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Ulmus_spp.</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	0	
<i>Velleia_paradoxa</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Veronica_calycina</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Veronica_gracilis</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Veronica_plebelia</i>	0	0	0	1	0	0	1	1	1	0	0	1	0	0	
<i>Veronica_spp.</i>	0	0	0	0	0	0	0	1	1	0	0	0	0	0	
<i>Vicia_sativa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	
<i>Viola_hederacea</i>	1	1	1	1	1	0	1	0	1	1	1	0	1	1	
<i>Wahlenbergia_gracilentia</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Wahlenbergia_gracilis</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0	
<i>Wahlenbergia_stricta</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Watsonia_meritana</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Wurmbea_dioica</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
<i>Xanthorrhoea_australis</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	

<i>Scientific_name</i>	WP-R	WP-C	WP-I	GB-R	GB-C	GBI	PH-R	PH-C	PH-I	WR-R	WR-C	WR-I	BL-R	BL-C	BL-I
<i>Xanthorrhoea_minor</i>	1	0	0	0	0	0	0	0	0	1	1	0	0	0	
<i>Xanthosia_dissecta</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Xanthosia_spp.</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	
<i>Zantedeschia_aethiopica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1	

<i>Scientific_name</i>	GFVB-R	GFVB-C	GFVB-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C	
<i>Acacia_aculeatissima</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_baileyana</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_dealbata</i>	0	1	0	0	0	0	0	0	1	0	0	0	0	0	
<i>Acacia_decurrens</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_elata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_genistifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_leprosa</i>	0	0	0	1	1	1	0	0	0	0	0	0	0	0	
<i>Acacia_longifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_meunsi</i>	0	1	1	0	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_melanoxylon</i>	0	1	0	0	0	0	0	1	1	0	1	1	0	1	
<i>Acacia_mucronata</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0	
<i>Acacia_myrtifolia</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	0	
<i>Acacia_paradoxa</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_pycnantha</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Acacia_stricta</i>	0	0	0	1	1	1	1	0	0	1	1	0	0	0	
<i>Acacia_verticillata</i>	0	0	0	0	1	1	1	1	1	0	1	0	0	0	
<i>Acaena_novae_zelandiae</i>	0	0	0	1	0	0	1	1	1	0	1	0	0	1	
<i>Acrotriche_prostrata</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Acrotriche_serrulata</i>	1	1	0	1	0	0	1	0	0	0	1	0	0	1	
<i>Adiantum_aethiopicum</i>	0	1	0	1	1	1	1	1	1	1	1	0	0	0	
<i>Agapanthus_praecox</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0	
<i>Aira_caryophylla</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Aira_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Allocasuarina_littoralis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Amperea_xiphoclada</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Amyema_pendula</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
<i>Anagallis_arvensis</i>	0	1	0	0	1	1	0	0	0	0	0	0	0	0	
<i>Anthoxanthum_odoratum</i>	1	1	0	1	1	1	1	1	0	0	1	0	0	0	
<i>Arthropodium_milleflorum</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0	

Scientific_name	GFV-B-R	GNV-B-C	GFCE-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Arthropodium_strictum</i>	1	1	0	1	1		0	0	0	1	1	0	0	0
<i>Asparagus_asparagoides</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Asparagus_scandens</i>	0	0	0	0	1		1	1	0	0	0	0	0	0
<i>Asperula_conferta</i>	0	0	0	0	0		0	0	0	1	0	0	0	0
<i>Asperula_eurphylla</i>	0	0	0	0	0		0	0	0	0	0	0	1	1
<i>Asperula_scoparia</i>	0	0	0	0	0		0	1	1	0	0	0	0	0
<i>Asplenium_bulbiferum</i>	0	0	0	0	0		0	0	0	0	0	0	1	0
<i>Asplenium_flabellifolium</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Astroloma_humifusum</i>	0	0	0	0	1		0	0	0	0	0	0	0	0
<i>Atherosperma_moschatum</i>	0	0	0	0	0		0	0	0	0	0	0	1	0
<i>Australina_pusilla</i>	0	0	0	0	0		0	0	0	0	0	0	1	1
<i>Austrocynoglossum_latifolium</i>	0	0	0	0	0		0	0	0	0	0	0	1	1
<i>Austrostipa_mollis</i>	1	0	0	0	0		0	0	0	0	0	0	0	0
<i>Austrostipa_pubinodis</i>	0	1	0	0	0		0	0	0	0	0	0	0	0
<i>Austrostipa_rudis</i>	1	1	0	0	0		0	0	0	0	0	0	0	0
<i>Austrostipa_spp.</i>	0	0	0	0	0		0	0	0	0	1	0	0	0
<i>Austrostipa_stipoides</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Avena_fatua</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Bedfordia_arborescens</i>	0	0	0	0	0		0	0	0	0	0	0	1	1
<i>Billardiera_heterophylla</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Billardiera_mutabilis</i>	0	0	0	0	0		1	0	0	0	0	0	0	0
<i>Billardiera_scandens</i>	0	0	0	1	1		0	1	0	1	1	0	0	0
<i>Blechnum_nudum</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Boschia_cinerea</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Boschia_prostrata</i>	1	0	0	0	1		0	0	0	1	1	0	0	0
<i>Briza_maxima</i>	1	0	0	1	1		0	0	0	0	0	0	0	0
<i>Briza_minor</i>	0	0	0	0	1		0	0	0	0	0	0	0	0
<i>Brunonia_australis</i>	0	0	0	0	1		0	0	0	1	1	0	0	0
<i>Bulbine_bulbosa</i>	0	0	0	0	1		0	0	0	0	0	0	0	0
<i>Burchardia_umbellata</i>	0	1	0	1	1		0	0	0	0	1	0	0	0
<i>Bursaria_spinosa</i>	1	1	0	1	1		0	0	0	0	0	0	0	0
<i>Caladenia_camea</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Caladenia_latifolia</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Caladenia_transitoria</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Calactasia_spp.</i>	0	0	0	0	0		0	0	0	0	0	0	0	0

<i>Scientific_name</i>	GFV-B-R	GNV-B-C	GFCH-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Calochaena_dubia</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0
<i>Calystegia_marginata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Cardamine_tenuifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Carex_breviculmis</i>	0	0	0	0	1	0	0	0	0	1	1	0	0	0
<i>Carex_inversa</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Carex_lynx</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Cassinia_aculeata</i>	0	0	0	1	0	1	1	1	0	1	1	0	0	0
<i>Cassinia_longifolia</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Cassinia_trinerva</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0
<i>Cassythamelantha</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Cassythapubescens</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Cenchrus_dandestinus</i>	0	1	0	0	0	0	0	0	0	0	0	0	0	0
<i>Centaurium_erythraea</i>	1	1	0	1	0	1	0	0	0	0	1	0	0	0
<i>Centaurium_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Centaurium_tenuiflorum</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0
<i>Centella_cordifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Centrolepis_fascicularis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cerastium_glomeratum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Cerastium_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Chamaescilla_corymbosa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Chiloglottis_valida</i>	0	0	0	0	0	0	0	1	0	0	0	0	0	0
<i>Chrysanthemoides_monilifera</i>	1	1	0	0	0	0	0	0	0	1	0	0	0	0
<i>Chrysoccephalum_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Clematis_aristata</i>	0	0	0	1	0	0	1	1	0	1	0	0	1	1
<i>Clematis_microphylla</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Comesperma_volubile</i>	0	0	0	1	1	1	1	0	0	1	0	0	0	0
<i>Coprosma_quadrijfida</i>	0	0	0	1	0	0	1	1	1	1	1	1	1	1
<i>Coprosma_repens</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Coronidium_monticola</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Coronidium_scorpoides</i>	0	0	0	0	0	1	0	0	0	1	1	0	0	0
<i>Correa_reflexa</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	0
<i>Corybas_incurvus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Corybas_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Craspedia_variabilis</i>	0	0	0	0	0	0	0	0	0	1	1	0	0	0
<i>Crataegus_monogyna</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0



Scientific_name	GFV-B-R	GNV-B-C	GFCS-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Cryptostylis_subulata</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Gyathea_australis</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	1
<i>Cynoglossum_suaevolens</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Cytisus_scoparius</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dactylis_glomerata</i>	1	0	0	0	1	0	1	0	0	0	0	0	0	0
<i>Daucus_carota</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Daucus_glochidiatus</i>	0	0	0	1	0	0	0	0	0	0	1	0	0	0
<i>Daviesia_leptophylla</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Desmodium_gunnii</i>	1	0	0	1	1	1	1	1	1	0	0	0	0	0
<i>Desmodium_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Deyeuxia_quadriseta</i>	0	1	0	0	0	0	1	0	0	0	1	0	0	0
<i>Deyeuxia_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dianella_admixta</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Dianella_laevis</i>	1	0	0	0	1	1	0	0	0	0	0	0	0	0
<i>Dianella_revoluta</i>	0	0	0	0	1	1	0	1	0	1	1	0	0	0
<i>Dianella_tasmanica</i>	0	0	0	0	0	0	1	1	0	0	0	1	0	0
<i>Dichelachne_micrantha</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dichelachne_spp.</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0
<i>Dichondra_repens</i>	1	1	0	0	1	1	1	1	1	0	1	0	0	0
<i>Dicksonia_antarctica</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0
<i>Dilwynia_cinereascens</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0
<i>Dilwynia_glaberrima</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dilwynia_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Dipodium_punctatum</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Dittrichia_graveolens</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Drosera_aberrans</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Drosera_auriculata</i>	1	1	0	1	0	0	1	1	1	1	0	1	0	0
<i>Drosera_peltata</i>	0	0	0	0	1	1	1	0	0	0	1	0	0	0
<i>Drosera_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Echinopogon_ovatus</i>	0	0	0	0	1	1	0	1	0	0	0	0	0	0
<i>Ehrharta_erecta</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0
<i>Ehrharta_longifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Epacris_impresa</i>	0	0	0	0	1	1	0	1	1	0	1	0	0	0
<i>Eragrostis_brownii</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Erica_lusitanica</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0

Scientific_name	GFV-B	GNV-B	GFCS-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Eucalyptus_cypellocarpa</i>	0	0	0	1	0	0	0	0	0	0	0	1	0	0
<i>Eucalyptus_dives</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus_goniocalyx</i>	1	1	0	0	1	1	0	0	0	1	1	1	0	0
<i>Eucalyptus_macrohyrcha</i>	0	0	0	1	1	1	0	0	0	1	1	0	0	0
<i>Eucalyptus_meliodora</i>	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus_obliqua</i>	0	0	0	0	1	1	1	1	1	0	1	1	0	0
<i>Eucalyptus_ovata</i>	0	1	0	0	0	0	0	0	1	0	0	0	0	0
<i>Eucalyptus_polyanthemos</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus_pryoriana</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Eucalyptus_radiata</i>	1	1	0	1	0	0	1	0	0	1	0	1	0	0
<i>Eucalyptus_regnans</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	1
<i>Eucalyptus_viminalis</i>	0	0	0	0	0	0	0	0	1	0	0	0	0	1
<i>Euchiton_involucratu</i>	0	0	0	1	1	1	0	0	0	0	0	0	0	0
<i>Euchiton_japonicus</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0
<i>Exocarpos_cupressiformis</i>	0	1	0	1	1	1	0	0	0	1	1	0	0	0
<i>Exocarpos_strictus</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Ficinia_nodosa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Gahnia_radula</i>	0	0	0	0	1	1	0	0	0	1	1	1	0	0
<i>Gahnia_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Galium_aparine</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Galium_gaudichaudii</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	1
<i>Galium_propinquum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Galium_spp.</i>	0	0	0	0	1	1	1	1	0	0	0	0	0	0
<i>Geranium_potentilloides</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	1
<i>Geranium_solanderi</i>	0	0	0	0	0	0	1	1	0	0	0	0	0	0
<i>Geranium_sp._2</i>	0	0	0	0	0	0	0	1	1	0	0	0	0	0
<i>Geranium_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Gladiolus_undulatus</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Glossodia_majior</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Glycine_clandestina</i>	1	1	0	1	1	1	1	1	0	1	1	0	0	0
<i>Glycine_microphylla</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Glycine_tabacina</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Gonocarpus_spp.</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0
<i>Gonocarpus_tetragynus</i>	1	0	0	1	1	1	1	0	0	1	1	1	0	0
<i>Goedenia_lanata</i>	0	0	0	1	0	0	0	0	0	1	1	0	0	0

Scientific_name	GFV-B-R	GNV-B-C	GFCS-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Goodenia_ovata</i>	0	0	0	1	1		1	0	0	1	1	0	0	0
<i>Hakea_decurrens</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Hardenbergia_violacea</i>	0	0	0	0	0		0	0	0	0	1	0	0	0
<i>Hedera_helix</i>	0	0	0	1	0		0	0	0	0	0	0	0	1
<i>Hedycarya_angustifolia</i>	0	0	0	0	0		0	0	0	0	0	0	1	1
<i>Hemarthra_uncinata</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Hibbertia_riparia</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Histiopteris_incisa</i>	0	0	0	0	0		0	0	0	0	0	0	1	1
<i>Holcus_lanatus</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Hovea_heterophylla</i>	0	0	0	0	1		0	0	0	0	1	0	0	0
<i>Hydrocotyle_foveolata</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Hydrocotyle_geraniifolia</i>	0	0	0	0	0		0	0	0	0	0	0	0	1
<i>Hydrocotyle_hirta</i>	0	0	0	1	1		0	1	1	0	0	0	1	0
<i>Hydrocotyle_laxiflora</i>	0	0	0	0	1		0	0	0	0	0	0	0	0
<i>Hydrocotyle_sibthorpioides</i>	0	0	0	0	0		1	0	0	0	0	0	0	0
<i>Hydrocotyle_spp.</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Hypericum_gramineum</i>	1	1	0	1	1		0	0	0	1	1	0	0	0
<i>Hypochaeris_glabra</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Hypochaeris_radicata</i>	1	1	0	1	1		1	0	0	1	1	0	0	0
<i>Hypolepis_glandulifera</i>	0	0	0	0	0		0	0	0	0	0	0	0	1
<i>Ilex_aquifolium</i>	0	0	0	0	0		0	0	0	0	0	1	0	1
<i>Imperata_cylindrica</i>	0	0	0	0	1		0	0	0	0	0	0	0	0
<i>Indigofera_australis</i>	0	0	0	1	0		0	0	0	0	0	0	0	0
<i>Isolepis_cernua</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Juncus_pallidus</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Kunzea_leptospermoides</i>	1	1	0	0	0		0	0	0	0	0	0	0	0
<i>Lachnagrostis_spp.</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Lagenophora_gracilis</i>	0	0	0	0	1		0	1	0	0	0	0	0	0
<i>Lagenophora_stipitata</i>	0	0	0	1	1		1	1	0	1	1	0	0	0
<i>Leontodon_saxatilis</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Lepidosperma_concavum</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Lepidosperma_elatius</i>	0	0	0	1	0		1	1	1	1	0	0	1	1
<i>Lepidosperma_filiforme</i>	0	1	0	0	0		0	0	0	0	0	0	0	0
<i>Lepidosperma_laterale</i>	0	0	0	1	1		0	0	0	0	1	0	0	0
<i>Lepidosperma_longitudinale</i>	0	0	0	0	0		0	0	0	0	0	0	0	0

Scientific_name	GFV-B-R	GNV-B-C	GFCH-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Lepidosperma_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Leptospermum_continentale</i>	0	1	0	0	0	0	0	0	0	0	0	1	0	0
<i>Leptospermum_lanigerum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Leptospermum_scoparium</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Leptospermum_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Leucochrysum_albicans</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Leucopogon_parviflorus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Leucopogon_virgatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Lobelia_anceps</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Lomandra_filiformis</i>	1	1	1	1	1	1	1	1	1	1	1	1	0	0
<i>Lomandra_longifolia</i>	0	0	0	1	1	1	1	0	1	1	1	0	0	0
<i>Lomandra_micrantha</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Lomandra_multiflora</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Lomatia_ilicifolia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Luzula_campetris</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Luzula_meridionalis</i>	1	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Luzula_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Medicago_polymorpha</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Medicago_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Melaleuca_squarrosa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Microlaena_stipoides</i>	1	1	1	1	1	1	1	1	1	1	1	1	0	1
<i>Microseris_sp.3</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Microsorium_pustulatum</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	0
<i>Microtis_arenaria</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0
<i>Microtis_spp.</i>	0	0	0	0	0	1	0	0	0	0	1	0	0	0
<i>Microtis_unifolia</i>	1	1	0	0	0	0	0	0	0	0	0	0	0	0
<i>Muellerina_eucalyptoides</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Myosotis_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Myosotis_sylvatica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Nicotiana_glauca</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Olearia_argophylla</i>	0	0	0	1	0	0	0	0	0	0	0	0	1	1
<i>Olearia_lirata</i>	0	0	0	0	0	1	0	0	0	1	1	0	0	0
<i>Olearia_myrsinoides</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Olearia_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Onopordum_acanthium</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	1

Scientific_name	GFV-B-R	GNV-B-C	GFGB-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Opercularia_ovata</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Opercularia_varia</i>	0	0	0	1	1		0	0	0	1	1	0	0	0
<i>Oxalis_corniculata</i>	0	0	0	0	0		0	0	0	0	1	0	0	0
<i>Oxalis_exilis</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Oxalis_incarната</i>	0	0	0	0	1		0	0	0	0	0	0	0	0
<i>Oxalis_perennans</i>	1	1	0	1	1		1	1	1	1	1	0	0	1
<i>Oxalis_spp.</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Ozothamnus_ferrugineus</i>	0	0	0	0	0		0	0	0	0	1	0	0	0
<i>Pandorea_pandorana</i>	1	0	1	1	1		0	0	1	0	1	1	1	1
<i>Parsonsia_brownii</i>	0	0	0	0	0		0	0	0	0	0	0	1	0
<i>Paspalum_spp.</i>	1	0	0	0	0		0	0	0	0	0	0	0	0
<i>Paterersonia_occidentalis</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Pelargonium_inodorum</i>	0	0	0	0	1		0	0	0	0	0	0	0	0
<i>Pimelea_axiflora</i>	0	0	0	0	0		0	0	0	0	0	0	0	1
<i>Pimelea_cuviffora</i>	0	0	0	0	1		0	0	0	0	0	0	0	0
<i>Pimelea_humilis</i>	1	0	0	1	1		0	0	0	1	1	0	0	0
<i>Pinus_radiata</i>	0	0	1	0	0		0	0	0	0	0	1	0	0
<i>Pitiosporum_undulatum</i>	0	1	1	0	1		0	1	1	0	0	1	0	0
<i>Plantago_lanceolata</i>	0	0	0	0	1		0	0	0	1	0	0	0	0
<i>Plantago_spp.</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Plantago_varia</i>	0	0	0	1	1		0	1	0	1	1	0	0	0
<i>Platylobium_montanum</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Platylobium_obtusangulum</i>	0	0	0	0	0		0	0	0	1	1	0	0	0
<i>Poa_annua</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Poa_clelandii</i>	0	0	0	0	0		0	1	0	0	0	0	0	0
<i>Poa_ensiformis</i>	0	0	0	1	1		0	1	0	0	0	0	0	0
<i>Poa_labillardierei</i>	0	0	0	0	0		1	0	0	0	0	0	0	0
<i>Poa_morrisii</i>	1	1	0	1	1		0	0	0	1	1	1	0	0
<i>Poa_rodwayi</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Poa_sieberiana</i>	0	0	0	0	1		0	0	0	1	0	0	0	0
<i>Poa_spp.</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Poa_tenera</i>	0	0	0	1	1		1	0	1	0	1	0	0	0
<i>Polyphlebium_venosum</i>	0	0	0	0	0		0	0	0	0	0	0	1	0
<i>Polyscias_sambucifolia</i>	0	0	0	0	0		0	0	0	1	0	1	0	0
<i>Polystichum_proliferrum</i>	0	0	0	0	0		0	0	0	0	0	0	1	1

<i>Scientific_name</i>	GFV-B-R	GNV-B-C	GFCH-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Pomaderris_aspera</i>	0	0	0	0	1		0	0	0	0	0	0	1	1
<i>Paranthera_microphylla</i>	1	1	0	1	1		0	1	0	1	1	0	0	0
<i>Paranthera_spp.</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Potentilla_indica</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Prostanthera_lasianthos</i>	0	0	0	0	0		0	0	0	0	0	0	0	1
<i>Prunella_vulgaris</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Pteridium_esculentum</i>	0	0	0	1	0		1	1	1	1	1	1	0	0
<i>Pteris_tremula</i>	0	0	0	0	0		0	0	0	0	0	0	0	1
<i>Pterostylis_alpina</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Pterostylis_melagramma</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Pterostylis_nutans</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Pterostylis_spp.</i>	0	0	0	1	0		0	0	0	1	0	0	0	0
<i>Pultenaea_gunnii</i>	0	0	0	0	0		1	1	0	0	1	0	0	0
<i>Pultenaea_scabra</i>	0	0	0	0	0		0	0	0	1	0	0	0	0
<i>Quercus_robur</i>	0	0	0	0	1		0	0	0	0	1	0	0	0
<i>Ranunculus_lappaceus</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Ranunculus_plebeius</i>	0	0	0	1	0		0	0	0	1	0	0	0	1
<i>Ranunculus_spp.</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Ranuncula_rosea</i>	0	1	0	0	0		0	0	0	0	0	0	0	0
<i>Rosa_rubiginosa</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Rubus_fruticosus_spp_agg.</i>	0	1	0	0	1		0	1	0	0	0	1	0	0
<i>Rubus_parvifolius</i>	1	1	0	1	1		1	1	0	1	0	0	0	0
<i>Rytidosperma_caespitosum</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Rytidosperma_pollidum</i>	0	0	0	1	1		0	0	0	1	1	0	0	0
<i>Rytidosperma_pilosum</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Rytidosperma_racemosum</i>	1	1	0	0	0		0	0	0	0	0	0	0	0
<i>Rytidosperma_spp.</i>	0	0	0	0	1		0	1	1	0	0	0	0	0
<i>Sambucus_gaudichaudiana</i>	0	0	0	0	0		0	0	0	0	0	0	1	1
<i>Schoenus_apogon</i>	0	1	0	0	1		0	0	0	0	1	0	0	0
<i>Senecio_glomeratus</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Senecio_hispidulus</i>	0	0	0	0	0		0	0	0	0	0	0	0	0
<i>Senecio_linearifolius</i>	0	0	0	1	1		0	0	0	0	0	0	0	0
<i>Senecio_minimus</i>	0	0	0	0	1		0	1	0	0	0	0	0	1
<i>Senecio_phelleus</i>	0	0	0	0	0		0	0	0	1	0	0	0	0
<i>Senecio_prenanthoides</i>	0	0	0	1	1		0	0	0	0	1	0	0	0

<i>Scientific_name</i>	GFV-B-R	GNV-B-C	GFCH-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Senecio_quadridentatus</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Sigesbeckia_orientalis</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	1
<i>Solanum_aviculare</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Solanum_nigrum</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Solanum_prinophyllum</i>	0	0	0	0	0	0	0	0	0	0	0	0	1	1
<i>Sonchus_oleraceus</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Sonchus_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Sporobolus_spp.</i>	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Stackhousia_monogyne</i>	0	0	0	0	0	1	0	0	0	0	1	0	0	0
<i>Stellaria_flaccida</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	1
<i>Stellaria_pungens</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Stylidium_graminifolium</i>	0	0	0	1	0	1	0	0	0	1	1	0	0	0
<i>Taraxacum_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Tetrarhena_junceae</i>	0	0	0	0	0	0	0	1	1	0	1	1	1	1
<i>Tetratheca_ciliata</i>	0	0	0	0	0	1	0	0	0	1	0	0	0	0
<i>Thelymitra_ixioides</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Thelymitra_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Themeda_triandra</i>	1	1	0	1	1	1	0	0	0	1	1	0	0	0
<i>Thysanotus_pateronii</i>	0	0	0	0	0	1	0	0	0	1	0	0	0	0
<i>Thysanotus_tuberosus</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Tricoryne_elatior</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0
<i>Ulmus_spp.</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Velleia_paradoxa</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Veronica_calycina</i>	0	0	0	0	0	1	0	0	0	0	0	0	0	0
<i>Veronica_gracilis</i>	0	0	0	1	0	0	0	0	0	0	0	0	0	0
<i>Veronica_plebelia</i>	0	0	0	0	0	0	1	0	0	0	0	0	0	0
<i>Veronica_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Vicia_sativa</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0
<i>Viola_hederacea</i>	0	0	0	1	1	1	1	1	1	1	1	0	0	1
<i>Wahlenbergia_gracilentia</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Wahlenbergia_gracilis</i>	0	0	0	0	0	1	0	0	0	1	1	0	0	0
<i>Wahlenbergia_stricta</i>	0	0	0	1	1	1	0	0	0	0	0	0	0	0
<i>Watsonia_meriana</i>	0	0	0	0	1	1	0	0	0	0	0	0	0	0
<i>Wurmbea_dioica</i>	0	0	0	1	1	1	0	0	0	0	1	0	0	0
<i>Xanthorrhoea_australis</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0

<i>Scientific_name</i>	GFV-B-R	GNV-B-C	GFCE-I	FTG-R	FTG-C	FTG-I	RH-R	RH-C	RH-I	M-R	M-C	M-I	S-R	S-C
<i>Xanthorhoea_minor</i>	0	0	0	0	0	0	0	0	0	1	0	0	0	0
<i>Xanthosia_dissecta</i>	0	0	0	0	0	0	0	0	0	0	1	0	0	0
<i>Xanthosia_spp.</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Zantedeschia_aethiopica</i>	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix 4a

Presence and absence of bird species observed at conditions within each study site. Column headings relate to site name in table 4.1.

Scientific Name	1r	1c	1i	2r	2c	2i	3r	3c	3i	4r	4c	4i	5r	5c	5i	6r	6c	6i	7r	7c	8r	8c	8i	9r	9c	9i	10r	10c
Striated Thornbill	0	0	0	1	1	1	0	0	0	0	1	1	0	0	0	0	0	0	0	1	0	1	1	1	1	1	0	0
Brown Thornbill	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	0	1	0	1	1	1	1	1
Eastern Spinebill	1	0	1	0	1	0	1	1	0	0	0	1	0	1	1	0	0	0	0	1	0	0	0	1	1	0	0	0
Australian King Parrot	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Red Wattlebird	1	0	0	1	0	0	1	1	1	1	0	0	1	1	0	0	0	0	0	1	0	0	0	1	1	0	0	0
Sulphur Crested Cockatoo	0	0	0	0	1	0	0	0	0	1	0	0	0	1	0	0	0	0	1	1	0	0	0	1	0	0	1	1
Little Corella	0	0	0	1	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Yellow Tailed Black Cockatoo	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spotted Wood duck	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Grey Shrike thrush	1	1	1	1	1	1	1	1	0	1	1	1	1	1	0	1	1	1	1	0	0	1	0	0	1	0	1	0
White throated Treecreeper	1	1	1	1	1	0	1	1	1	0	0	1	1	1	0	1	1	1	1	0	1	1	1	0	0	0	1	1
Australian Raven	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Little Raven	1	0	1	0	1	0	1	1	0	1	0	0	1	1	0	1	0	0	1	0	0	0	1	0	1	0	1	0
Australian Magpie	0	0	0	1	1	1	0	1	0	1	0	1	1	1	0	0	1	0	1	1	1	1	1	0	0	1	0	0
Grey Butcherbird	0	0	0	1	1	1	0	1	0	1	1	0	1	1	1	0	0	0	1	1	0	1	1	0	0	0	0	1



Scientific Name	1r	1c	1i	2r	2c	2i	3r	3c	3i	4r	4c	4i	5r	5c	5i	6r	6c	6i	7r	7c	8r	8c	8i	9r	9c	9i	10r	10c
Laughing Kookaburra	0	0	0	1	1	0	1	0	0	1	0	0	1	1	1	1	1	0	0	1	0	0	0	0	0	0	1	0
Varied Cettella	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mistletoebird	0	0	0	1	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Galah	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0
Eastern Yellow Robin	1	1	1	1	1	1	1	1	0	1	1	1	1	1	0	0	0	1	0	0	1	1	0	1	1	1	1	1
Magpie Lark	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Yellow Faced Honey Eater	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
White eared Honeyeater	1	0	1	1	1	1	1	1	0	1	1	1	1	0	0	1	1	0	1	0	1	1	0	1	1	1	0	0
Superb Fairy wren	0	0	0	1	0	0	1	0	0	1	1	1	1	1	0	0	0	0	1	1	1	1	0	0	0	1	0	0
Noisy Miner	0	0	0	0	1	1	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
White Naped Honeyeater	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Superb Lyrebird	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Powerful Owl	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Golden Whistler	0	0	1	1	1	0	1	0	0	1	1	1	1	1	1	1	1	0	1	0	0	0	0	0	0	0	0	0
Rufous Whistler	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spotted Pardalote	0	0	0	1	1	1	1	1	0	1	1	1	1	1	1	1	1	1	1	1	1	0	0	1	1	1	0	1
Striated Pardalote	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0	0	0	0	0
Scarlet Robin	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
White Cheeked Honeyeater	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
New Holland Honeyeater	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crimson Rosella	1	0	0	1	1	1	1	1	0	1	1	0	1	1	1	1	1	1	0	1	1	1	1	1	1	1	1	1
Eastern Rosella	0	0	0	1	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Whip Bird	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Grey Fantail	1	1	1	1	1	1	1	1	1	1	1	1	1	0	0	1	1	1	0	0	1	1	1	1	1	1	1	1
White Browed Scrubwren	1	0	0	1	0	0	0	0	0	1	0	1	0	0	0	1	0	1	0	1	0	1	0	1	0	0	1	1
Pied Currawong	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0

Scientific Name	1r	1c	1i	2r	2c	2i	3r	3c	3i	4r	4c	4i	5r	5c	5i	6r	6c	6i	7r	7c	8r	8c	8i	9r	9c	9i	10r	10c
Grey Currawong	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Spotted Dove	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Common Starling	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Rainbow Lorikeet	1	0	1	0	1	1	0	0	0	0	0	0	0	1	0	1	0	1	0	0	0	0	0	0	0	0	0	
Common Blackbird	0	0	0	0	0	1	0	0	0	1	0	1	1	1	1	0	0	0	0	1	0	0	0	0	0	1	0	
Silvereye	0	0	0	0	0	0	1	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	

## Appendix 4b

Appendix 4b.1. Tukey's test examining different PCA scores based upon species richness for bird communities across feeding preferences. Site name relate to column 2 of Table 4.2 in the supplementary data. PC1 – Principal component one, PC2 – Principal component two, R - Remnant control – remnant bushland, C - Cleared treatments – *Pittosporum undulatum* cleared, I - invaded control – high density *Pittosporum undulatum*.

Site	PC1	PC2
WP-R	0.57177286	1.14219426
WP-C	-2.8984586	0.15765433
WP-I	0.12697494	0.97096534
GR-R	2.99415064	0.42293434
GB-C	3.29543214	0.75194111
GB-I	0.39796527	0.95493952
PH-R	1.38827381	-0.94928398
PH-C	-1.62226051	-2.02735739
PH-I	-1.62529772	0.89060077
WR-R	1.72049702	-1.18456949
WR-C	1.08346236	0.76511003
WR-I	1.76781616	0.61004801
BL-R	1.0401659	-2.19835084
BL-C	1.83307173	-0.77805506
BL-I	-1.38574183	0.56430983
GFVB-R	0.49465528	-0.12289843
GFVB-C	-0.95633587	0.12574376
GFVB-I	-0.35262958	0.74898982
FTG-R	-0.74707115	-0.54557976
FTG-C	-0.39543329	-1.33991366
RH-R	-1.27365986	0.09626687
RH-C	-0.18969852	0.03216359
RH-I	-0.90950946	1.04580387
M-R	-1.06439514	-0.57505665
M-C	-1.06488788	-1.44961404
M-I	-1.62529772	0.89060077
S-R	-0.01589094	0.21941832
S-C	-0.58767004	0.78099478

Appendix 4b.2. Tukey's test examining different PCA scores based upon species richness feeding preferences for birds within no canopy and canopy communities. Site name relate to column 2 of Table 4.2 of the supplementary data. PC1 – Principal component one, PC2 – Principal component two, PU – (no canopy) dense *Pittosporum undulatum* and understory below, NPU – (canopy) Non *Pittosporum undulatum* vegetation, particularly the *Eucalyptus* dominated overstory.

	PC1	PC2
WP-PU	-1.9346891	-0.09633895
GB-PU	-1.3215938	-0.42282124
PH-PU	-0.6250069	0.96705744
WR-PU	-0.6250069	0.96705744
GFVB-PU	-0.8089685	0.72985238
RH-PU	-2.2885035	-0.1102451
M-PU	-0.9788214	0.9531513
WP-NPU	1.6957851	0.07245462
GB-NPU	2.0023327	-0.09078653
PH-NPU	-0.5386456	-0.97260244
WR-NPU	4.094963	0.42282866
BL-NPU	0.205208	0.59442353
GFVB-NPU	1.4172899	-0.51904503
RH-NPU	0.2443025	-1.52238364
M-NPU	-0.5386456	-0.97260244

Appendix 4b.3. Tukey's test examining different PCA scores based upon species richness habitat preferences for bird within no canopy and canopy communities. Site name relate to column 2 of Table 4.2 of the supplementary data. PC1 – Principal component one, PC2 – Principal component one, PU – (no canopy) dense *Pittosporum undulatum* and understory below, NPU – (canopy) Non *Pittosporum undulatum* vegetation, particularly the *Eucalyptus* dominated overstory.

	PC1	PC2
WP-P	-1.4944391	0.20022532
GB-P	-1.0054868	0.04757057
PH-P	-0.5351405	-0.32518965
WR-P	-0.6480756	-0.17697322
GFVB-P	-0.8925517	-0.10064585
RH-P	-1.8518503	0.42476912
M-P	-1.0054868	0.04757057
WP-N	1.9836472	2.24153873
GB-N	1.6456904	0.72697326
PH-N	-0.1777292	-0.54973344
WR-N	2.7373785	-1.6857674
BL-N	-0.1591233	-0.32962797
GFVB-N	1.0260455	-1.15147579
RH-N	0.6677858	1.03228276
M-N	-0.2906643	-0.40151702