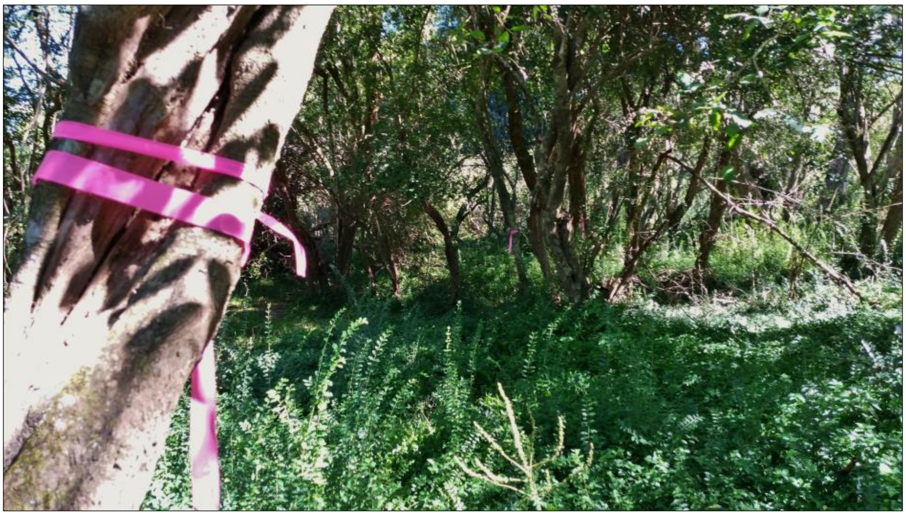
Can oldfield regrowth dominated by non-native privet trees contribute to biodiversity and rainforest regeneration?



Debbie Lynae Rudd B.Sc. Honours Dissertation

A dissertation submitted in partial fulfilment of the requirements for the degree of Bachelor of Science with Honours.

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ABSTRACT

Widespread clearing of tropical and subtropical rainforests is a major threatening process for biodiversity and ecosystem functions worldwide. Deforestation has primarily occurred to create areas of livestock pasture and other agricultural uses, causing diverse and complex forest ecosystems to be replaced with much more simplistic ecosystems dominated by non-native species. To mitigate these impacts, reforestation is needed over large spatial scales. The process of unassisted regrowth on the oldfields that result from retirement of land from livestock grazing provides a potentially important pathway of rainforest restoration. However, oldfield regrowth in Australia is often dominated by non-native pioneer tree species, and the positive and negative roles of these species are strongly debated.

This study investigated the role of a non-native species, small-leaved privet (*Ligustrum sinense*), in oldfield regrowth on the eastern Dorrigo Plateau of subtropical Australia. Three questions were addressed, with a specific focus on the potential conservation value of privet-dominated woody regrowth. (1) What are the ecological values of plant communities in post-pasture regrowth dominated by small-leaved privet? (2) What are the ecological values of bird communities in regrowth dominated by small-leaved privet? (3) What is the potential for future rainforest regeneration in privet-dominated regrowth areas?

These questions were addressed by establishing 21 study sites spread across the study region, and representing three habitat types: regrowth, pasture and forest. There were eight privet-dominated regrowth sites surrounded by livestock pasture; eight sites in livestock pasture; and five sites in conserved old-growth rainforest. As far as possible all three habitat types shared similar soils and elevations and avoided steep slopes. The regrowth vegetation in the privet sites was a strip approximately 20-100 m wide, being part of much longer linear regrowth ribbons that grew bordering streams running across the otherwise-cleared plateau; sites used in this study contained at least 0.4 ha of continuous regrowth habitat.

Vegetation structure, plant communities and bird communities were measured at each site, using a standard quantitative technique previously designed for evaluating the biodiversity values of restored rainforest. Each site had two 20 x 50 m transects within which vegetation structure, and floristic composition and diversity were measured. Bird composition and diversity were measured within a 30 x 100m area which encompassed

the vegetation survey transects. In addition, grazing pressure and levels of fruiting were also measured at each site. The variables measured were specifically chosen to highlight any potential recovery from a pasture-like to a forest-like state in regrowth sites, and to identify any factors that may facilitate rainforest regeneration.

Analyses of the survey results revealed that privet-dominated regrowth of the eastern Dorrigo Plateau had recovered many of the ecological values associated with areas of intact old-growth rainforest, but which were absent from areas of pasture. Bird communities had achieved fast to full recovery, with overall community composition showing greater similarity to forest than pasture and bird species richness and abundance being greatest in regrowth sites. However, the recovery of bird species richness and density was largely attributed to an influx of species that are typically found in both eucalypt forest and rainforest, whereas more specialised, rainforest-dependent, bird species had achieved only a partial similarity to forest reference sites. Vegetation structure had achieved intermediate recovery, with features such as canopy cover and densities of trees with small (<20cm) and medium (20-50cm) stem diameters recovering quickly, while complex features such as the frequency of special life-forms (e.g. epiphytes, robust vines) and trees with large (>50cm) stem diameters had shown little to no recovery. Floristic composition and diversity showed least recovery, and most rainforest-associated tree species and families were absent from the regrowth sites, while the overall floristic composition was more similar to pasture than forest.

The bird and plant survey data was used together with other previously-available information on all species’ ecological characteristics, to conduct further analyses which assessed whether the privet regrowth was more likely to be assisting or inhibiting the future recovery of further rainforest diversity. Despite their comparatively depauperate floristic communities, the privet-dominated regrowth sites contained many of the ecological factors that could facilitate rainforest regeneration: grass cover had significantly decreased from pasture-like levels; canopy cover had increased significantly; there was an abundant supply of fauna-attracting features such as perches and fruiting resources; and species richness and abundance of seed-dispersing birds was high. However, despite evidence that seeds of rainforest trees, including frugivore-dispersed species with large diaspores, were being recruited into the seedling layer of regrowth sites, these species were under-represented or absent from the larger stem diameter classes.

Given the apparent potential for regeneration in regrowth, it would appear that other factors are inhibiting the recovery of floristic diversity and composition characteristic of intact rainforest. Potentially inhibiting factors could include: under-representation of the rainforest-dependent frugivorous bird species that are most effective at dispersing large rainforest seeds; competitive suppression of rainforest seedlings by the privet overstorey; and high mortality or limited growth of recently germinated tree and shrub seedlings as a consequence of grazing pressure. While this study did not directly address factors potentially inhibiting regeneration, high grazing pressure was recorded in regrowth. Since the improved structural complexity and food resources in privet-dominated regrowth (compared with pasture) provide valuable habitat for native fauna, the recommended management strategy is to retain this regrowth while also experimentally investigating the outcomes of interventions aimed at accelerating rainforest regeneration (such as livestock exclusion or privet-thinning) within it.

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STATEMENT OF ORIGINALITY

This work has not previously been submitted for a degree or diploma in any university. To the best of my knowledge and belief, the dissertation contains no material previously published or written by another person except where due reference is made within the dissertation itself.

Signed: Debbie Lynae Rudd

1. INTRODUCTION

1.1 RAINFOREST LAND-COVER CHANGE: FROM DEFORESTATION TO REFORESTATION

Landscape transformation and dominant anthropogenic pastures

Deforestation is one of the major environmental issues facing the world today. Humans have been manipulating ecosystems for thousands of years as a means to increase productivity (FAO, 2016), however current rates of land clearing are unsustainable. Every year significant areas of tropical forest are cleared to create space for agricultural and pastoral land (Laurance, 2008), with three-quarters of all new agricultural land established globally in the 1980s and 1990s coming at the expense of rainforests (Gibbs et al., 2010). By the first decade of the 21st century, mean annual losses of tropical rainforest were estimated at 7 million hectares, while annual increases in agricultural land area were approximately 6 million hectares (FAO, 2016). Evidence suggests that this trend is likely to continue throughout this century. Increasing economic competition and decreasing fertility in existing production lands have been identified as key drivers of land conversion in tropical and subtropical rainforests (Knoke et al., 2014): as the human population continues to expand and food demands increase, rainforest clearing is likely to be an ongoing issue (Jank et al., 2013).

The process of converting rainforest to pasture has significant impacts on biodiversity and ecosystem functions. Complex native forest plant assemblages dominated by diverse and abundant trees and vines are lost when rainforests are converted to agricultural land (Chazdon, 2008). These plant assemblages are replaced with simpler species compositions predominantly comprising grasses, herbs and ferns (Chazdon & Guariguata, 2016; Chazdon, 2014; Holl, 2007). The proliferation of frugivore-dispersed tree and shrub species characteristic of rainforest habitat (McConkey et al., 2012) are typically exchanged for a limited number of wind-dispersed ground cover species. Furthermore, following land conversion managers have often sown non-native pasture grass species which can further simplify plant diversity and structure (Asner et al., 2004). Efforts to improve productivity through the development and planting of high-yield forage cultivars (Jank et al., 2014), often resulting in pastures dominated by single-species, can further impact biodiversity.

Rainforest conversion to pasture is not only affecting the species diversity and composition of plant communities, it is also impacting ecosystem services and releasing significant carbon stores into the atmosphere. Forests provide a wide range of ecosystems services, including erosion reduction and water flow regulation (Bradshaw et al., 2009). The disruption of these services can lead to numerous impacts on the environment and human wellbeing. Flood and drought risks, impacts to global water supplies, and siltation of dams, waterways and coastal environments can all be partially mitigated through the preservation of naturally-functioning rainforest ecosystems (Bradshaw et al., 2009). Forests also contain valuable resource pools, with many traditional medicines, and key ingredients for modern medicines, often derived from rainforest plants (Sodhi et al., 2007). Furthermore, in addition to the loss of goods and services, clearing of intact rainforests can result in significant emissions of carbon dioxide into the atmosphere (van der Werf et al., 2009). The International Panel on Climate Change estimated that, at the time of their fourth Assessment Report, tropical forest loss accounted for approximately 20% of global carbon emissions (IPCC, 2007), with emissions from forestry and other land-uses rising by approximately 40% since 1970 (IPCC, 2014). Thus it is clear that the process of rainforest conversion is contributing to a broad array of environmental impacts globally.

Impacts of rainforest conversion on biodiversity

The conversion of rainforest to pasture has the potential to have considerable negative impacts on global biodiversity. Moist tropical forests are hotspots for floral and faunal species diversity (Laurance et al., 2012; Chazdon et al., 2009). Significant declines in these biodiversity attributes have been linked to both past and present rainforest clearing, and the associated habitat loss and fragmentation (Bradshaw et al., 2009). Conservation status assessments have indicated that many plant species are undergoing rapid declines in the tropics of the Americas, Asia and Africa (IUCN, 2007), largely as result of reductions in rainforest habitat. Laurance and Wright (2009) suggested that if deforestation continues unabated there could be a catastrophic loss of species within this century.

Within tropical and subtropical rainforests the majority of plant species are fleshy-fruited and vertebrate frugivores, particularly birds, are primarily responsible for seed dispersal (Willson et al., 1989; Howe & Smallwood, 1982). Therefore, the ecological interactions between fleshy-fruited plants and seed-dispersing birds are critical to the maintenance of rainforest habitat. However, a global meta-analysis encompassing 138 studies found that birds were the taxonomic group that was most sensitive to the conversion of rainforest into agricultural land (Gibson et al., 2011). Furthermore, BirdLife International has estimated that global bird species diversity may decrease by 13% over the next century, largely as a result of deforestation (BI, 2000). These concurrent declines in rainforest plants and seed-dispersing birds may also reduce the potential for future rainforest regeneration in areas of former forest.

Global biodiversity conservation will require not only the preservation and maintenance of remaining tracts of old-growth rainforest but also large-scale reforestation in areas where rainforests have been converted to pasture. Conserving intact rainforest will help to reduce the rate of biodiversity decline (Gibson et al., 2011) and will be essential for the conservation of some rainforest-dependent species (Gardner et al., 2009). However, only a small proportion (<10%) of remaining tropical forests are currently protected (Schmitt et al., 2008) and these areas are heavily influenced by anthropogenic activities in the surrounding landscape (Harvey et al., 2008; Wittemyer et al., 2008). Therefore, conservation of these areas alone will be inadequate for preserving biodiversity at current levels, maintaining ecosystem services and beginning to reverse the ecological damage already done.

Incorporating human-modified landscapes into the restoration of forest cover over large spatial scales will be essential to the future sustainability of the world’s rainforests and their unique biota (Chazdon & Uriate, 2016; Chazdon, 2014; Chazdon et al., 2009). Expanding forest cover will provide greater resource security for forest-dependent fauna currently restricted to small forest fragments (Chazdon & Uriate, 2016) and aid in the prevention of localised extinctions of deep-forest plant species (Martinez-Garza & Howe, 2003). The cumulative impacts of conserving old-growth forests and secondary forests, and restoring forests on previously forested lands, would also help to reduce atmospheric CO_2 concentrations and significantly offset CO_2 emissions produced by fossil fuel use (Houghton et al., 2015). Therefore, identifying pathways to reforestation will be vital to offset the many environmental impacts arising from ongoing global deforestation trends.

Pathways to reforestation and the potential for oldfield regrowth

Ecological restoration is gaining increasing attention from government agencies that are recognising the importance of restoring vegetation structure and ecosystem functioning, and replenishing biodiversity (SER, 2004). As policy makers acknowledge the need to reinstate rainforests across large spatial scales, cost-effective methods for reforestation are increasingly sought (Sabogal et al., 2015). Potential pathways to reforestation cover a broad spectrum ranging from active restoration to passive, unassisted regrowth. Active restoration methods involve interventions such as direct plantings or direct seeding and eradication or control of invasive species (Rey Benayas & Bullock, 2012; Erskine et al., 2007). However, active restoration requires significant investments of time and labour and is often prohibitively expensive (Rey Benayas & Bullock, 2012; Birch et al., 2010; Choi, 2004), and thus generally only feasible over small spatial scales (Lamb et al., 2005).

Unassisted regrowth, which occurs through the natural dispersal, germination and recruitment of plant species (Rey Benayas & Bullock, 2012; Rey Benayas et al., 2008), may provide a low cost alternative to active restoration. However, the recovery of vegetation structure, floristic and faunal community composition and ecosystem functioning is a gradual process (Chazdon & Guariguata, 2016) and the outcomes of unassisted regrowth can vary in response to different disturbance legacies and spatial scales. When a significant disturbance event, such as rainforest clearing, results in large-scale habitat degradation, species assemblages may be unlikely to recover to pre-disturbance levels through natural regeneration alone (Chazdon & Guariguata, 2016) and enrichment planting may be required for the reestablishment of sensitive species (Bertacchi et al. 2016). Nevertheless, as unassisted regrowth incurs no direct costs, it may provide a viable option for significantly increasing overall forest cover within the landscape while avoiding the considerable costs associated with active restoration (Chazdon & Guariguata, 2016; Shoo et al., 2016; Hobbs & Norton, 1996). Identifying scenarios in which unassisted regrowth is likely to provide the greatest opportunities for rapid rainforest regeneration will be critical to future conservation planning.

Unassisted regrowth in “oldfields” provides one potential avenue for rainforest regeneration over large spatial scales. As agricultural yields decrease and global competition increases (Lambin et al., 2011) farmers are increasingly retiring pastures in areas of low productivity (Knoke et al., 2014). These retired pastures, or oldfields, cover large tracts of land worldwide and provide significant space for potential reforestation (Asner et al., 2004). Field et al. (2008) estimated that oldfields with the potential for future productivity cover an area of close to 400 million hectares worldwide. Enabling or facilitating the process of unassisted regrowth in these oldfields may yield considerable conservation benefits. However, research shows that regeneration trajectories in oldfields can vary signficantly (Catterall, 2016; Norden et al., 2015; Chazdon, 2008; Holl, 2007) and knowledge of natural successional trajectories may not be relevant in the context of oldfields (Hobbs et al., 2009). Therefore, given the need for rapid reforestation over large spatial scales, increasing our understanding of the factors that may inhibit or facilitate rainforest regeneration in oldfields is of mounting importance.

1.2 FACTORS THAT CAN INHIBIT OR FACILITATE RAINFOREST REGENERATION IN OLDFIELDS

Potentially inhibiting and facilitating factors

Oldfields are characterised by a history of intense human land management and the capacity for unassisted regrowth in these areas is influenced by the extent, intensity and duration of ecosystem modification, often referred to as the ‘agricultural legacy’ (Dwyer et al., 2010). For example, factors such as the use of heavy machinery or agrochemicals, frequent harvest, high stocking density and maintenance of dominant non-native pasture plants, can all negatively affect reforestation outcomes (Martinez-Ramos et al., 2016). Furthermore, the agricultural legacy of oldfields can create significantly different abiotic conditions to those found in both the original vegetation and native grasslands (Paul et al., 2012; Hobbs et al., 2009; Holl, 2007).

There are many potential biotic and abiotic limitations to unassisted regrowth in oldfields (Zahawi & Augspurger, 2006; Hooper et al., 2005): soil structure and climatic conditions can influence the regeneration of woody vegetation (Chazdon, 2014); the grasses, herbs and ferns that tend to dominate oldfields can competitively inhibit the recruitment of woody tree seedlings into these ecosystems (Catterall, 2016; Elgar et al., 2014; Hooper et al., 2005; Zimmerman et al., 2007); and the persistence, or reintroduction, of livestock grazing can quickly displace any successful tree recruitment as a result of browsing activities which can damage or destroy seedlings (Catterall, 2016). The rate at which unassisted regrowth can proceed may also vary considerably in relation to both natural factors and human activities. In low productivity environments the establishment of woody vegetation may be limited, or indefinitely suspended, while in high productivity environments regrowth may occur rapidly (Rey Benayas et al., 2008). Revegetation rates may also increase in response to the ceasing of land management aimed at maintaining pasture and the removal of grazing livestock (Catterall, 2016; Asner et al., 2004).

A number of elements are required for the successful regeneration of rainforest vegetation. The availability of rainforest plant propagules in a viable soil seed bank (Loydi et al., 2012; Yates et al., 1995), or an alternative seed source within dispersal proximity (Garcia et al., 2010; Hobbs and Yates, 2000; Holl, 1999), is essential to the colonisation of oldfields by rainforest vegetation. Favourable soil properties and microclimatic conditions are critical to the establishment of many rainforest plants whose growth and survival may require high levels of soil organic matter or shading by established canopy vegetation (Holl, 1999; Nepstad et al., 1996). While the absence of dense grasses and herbs, which can compete for below-ground water and nutrients, may be needed to ensure the germination of many rainforest plants (Catterall, 2016; Elgar et al., 2014; Shoo & Catterall 2013; Zimmerman et al., 2007). Given that animal-mediated seed dispersal is a key feature of the majority of rainforest plants, the presence of suitable seed-dispersing birds or mammals able to travel between intact rainforest and regeneration areas is also a necessity for rainforest regeneration, particularly in the absence of viable soil seed banks (Cubina & Aide, 2001; Holl, 1999).

The occurrence of frugivorous birds in oldfield regrowth will not alone guarantee the establishment of diverse rainforest vegetation, the quality, quantity and composition of dispersed seeds are functions of the biology and behaviour of the species present. Frugivore characteristics such as gape size, seed treatment and digestive process can all affect a seed’s germination viability (Buckley et al., 2006). Gape size has been shown to be positively correlated with the size of fruits consumed by frugivorous birds (Moran & Catterall, 2010), which is particularly pertinent for rainforest plants with large fleshy-fruits. Different treatment of seeds can also result in varying impacts to a seeds’ germination potential: some birds’ drop or cache seeds after consuming the fruit; others chew, crush and expel seeds; while others swallow and then regurgitate or defecate seeds (Buckley et al., 2006). Digestive processes, and the length of time spent in a frugivore’s gut, can also affect the physical and chemical properties of seeds (Buckley et al., 2006). Furthermore, even if gape size, seed treatment and digestive process are all conducive to seed germination, the frugivore’s movement patterns can also influence the distance and location of seed dispersal. Bird species with large home-ranges, and migratory species, can contribute disproportionately to seed dispersal (Fritz & Purvis, 2010), as can large frugivores with a generalist diet. However, the behaviour of frugivorous birds is influenced by the nature of the landscape matrix (Prevedello & Viera, 2010) and native frugivore species diversity and abundance have been shown to decrease along a course gradient ranging from secondary forest to pasture (Gardner et al., 2009).

Due to the many factors required to facilitate rainforest regeneration, and the considerable variation in the abiotic and biotic characteristics of oldfields, these habitats tend to be colonised by a unique suite of pioneer species. The initial stages of unassisted regrowth in oldfields are likely to comprise pioneer tree species that are distinct from those that colonise light gaps in rainforests, both in terms of their origin and their functional attributes (Catterall, 2016). Tree recruitment into oldfields will favour pioneer species for which open landscapes are not a barrier to seed dispersal and for which potential nutrient-poor soils, and competition with dense grasses, herbs and ferns, are not a hindrance to germination and growth (Catterall, 2016). Other factors that can favour ongoing recruitment include prolonged periods, or advantageous timing, of seed production and seed longevity, and seedling resistance to predation and disease, and ability to withstand significant light exposure (Martinez-Ramos et al., 2016). Nevertheless, even with the presence of these factors, the process of rainforest regeneration can require time periods of decades or longer (Goosem et al., 2016), and if facilitating factors are not maintained regeneration may be arrested indefinitely.

The pros and cons of non-native trees as pioneer species

Pioneer tree species that have the ability to withstand the potentially harsh conditions of oldfields are often non-native species which also have a propensity for rapid range expansion (Williams & Jackson, 2007). In particular, fleshy-fruited non-native tree species are regularly the first to colonise oldfields (Elgar et al., 2014), and they often represent the dominant species in the early stages of rainforest regeneration (Lugo & Helmer, 2004). Furthermore, dispersal of these fleshy-fruited non-native pioneers can be readily facilitated by frugivorous birds in disturbed habitats (Buckley et al., 2006).

More generally, there is considerable debate around the potential positive and negative effects of non-native plants in the context of rainforest regeneration (Simberloff et al., 2013; Schlaepfer et al., 2011; Hobbs et al., 2009; Reid et al., 2009; Ewel & Putz, 2004; D’Antonio & Meyerson, 2002). Restoration practitioners have often assumed that the persistence of non-native species would lead to reductions in species richness, decreased structural complexity (Myers & Bazely, 2003) and negative impacts on native tree recruitment (Davis et al., 2011). These assumptions have often resulted in eradication being chosen as the preferred management option (Simberloff et al., 2013;

Ewel & Putz, 2004). In some circumstances non-native pioneer species can inhibit rainforest regeneration and remain the dominant species in secondary forest. However, research is increasingly indicating that non-native plants can also play a positive role in rainforest regeneration.

There are many ways in which non-native trees can contribute to positive environmental outcomes. By attracting frugivores through the provision of habitat and, in the case of fleshy-fruited plants, food resources non-native plants can potentially improve seed dispersal and facilitate rainforest regeneration (Catterall, 2016; Elgar et al., 2014; Zahawi & Augspurger, 2006; Moran et al., 2004; D'Antonio and Meyerson, 2002). As woody vegetation matures, conditions are also likely to become increasingly conducive to rainforest regeneration. Increasing shade cover can contribute to improved microclimatic conditions, and decreasing grass cover can lead to reduced competition for soil water and nutrients, which can enhance the germination of native tree species (Catterall, 2016; Elgar et al., 2014; Neilan et al., 2006). Over time, the cumulative impacts of increased frugivore visitation, and potential seed dispersal, together with decreased competition and favourable microclimate conditions can further facilitate native seedling recruitment (Shoo & Catterall, 2013; Holl, 2007; Zimmerman et al., 2007; Hooper et al., 2005).

1.3 RAINFOREST DEFORESTATION, REFORESTATION AND NON-NATIVE SPECIES AS OLDFIELD PIONEERS IN THE AUSTRALIAN TROPICS AND SUBTROPICS

Extensive clearing of rainforests in the Australian tropics and subtropics occurred following European settlement in the late 19th and early 20th centuries (Erskine et al., 2007; Catterall et al., 2004; Bridger, 1997); commencing with timber harvesting, followed by conversion to small-scale agricultural land-use and eventually culminating with broad-scale livestock grazing, which often comprised planting of non-native pasture grasses (Fisher et al., 1996; Tietzel, 1992). Short pasture grasses came to dominate large areas of the landscape, particularly on shallow slopes and flat terrain, while rainforest remnants were primarily retained on the steeper, less fertile, slopes (Catterall, 2016; Fisher et al., 1996).

From the mid-20th century onwards economic pressures led to the retirement of some pastures and reductions in stocking rates, while increasing interest in conservation led to the commencement of restoration activities in some areas (Goosem & Tucker, 2013;

Parkes et al., 2012; Erskine et al., 2007). In some oldfields across the tropics and subtropics tree recruitment occurred naturally, leading to the establishment of extensive woody regrowth by the late 20th century (Catterall, 2016; Neilan et al., 2006). In other degraded oldfield areas considerable time and effort was invested in reforestation, with restoration activities increasing significantly from the 1980s onwards (Erskine et al., 2007). As reported in the global literature, active restoration efforts in Australia have been constrained by the high costs involved and, as a consequence, have been limited in scale (Erskine et al., 2007). For example, the establishment of 4.4km^2 worth of biodiversity plantings in the Australian Wet Tropics over a two year period cost over $8 million, comprised of government funding and in-kind community support (Catterall et al., 2004), yet restoration activities covered less than 0.2% of the total rainforest area originally cleared (Kanowski et al., 2003).

Across Australia, restoration practitioners are increasingly recognising the need to investigate potential alternative reforestation pathways and researchers are looking to address knowledge gaps in relation to the processes that can facilitate and inhibit rainforest regeneration. Many recent studies have investigated the ecological interactions between rainforest plants and seed-dispersing birds within the Australian tropics and subtropics (Elgar et al., 2014; Moran & Catterall, 2014, 2010; Buckley et al., 2006; Neilan et al., 2006). For example, Moran and Catterall (2010) demonstrated that gape width and level of frugivory were positively associated with both the fruit consumption patterns of frugivore species in subtropical Australia and their seed dispersal potential. In particular, frugivores for which fruit was a major dietary component, and mixed-diet frugivores, tended to consume the largest possible fruits relative to their gape size (Moran & Catterall., 2010). Fruits from a high number of plant species within the Lauraceae family, and the Celastraceae and Sapindaceae families, were well represented in the diets of major frugivores, including fruit-dove species from the *Ptilinopus* genus, and mixed-diet frugivores, including bowerbird species from the Ptilonorhynchidae family, respectively (Moran & Catterall, 2010).

Research into reforestation pathways in the Australian tropics and subtropics is also increasingly focussing on the roles of non-native pioneer species in unassisted regrowth, with studies documenting both positive and negative impacts. For example, in the Big Scrub region of the Australian subtropics, Neilan et al. (2006) demonstrated that non-native camphor laurel (*Cinnamomum camphora*) appeared to be facilitating rainforest regeneration, primarily through the use of this habitat by frugivorous birds which were dispersing the seeds of a wide diversity of native rainforest tree species. Similarly, Elgar et al. (2014) showed that the presence of living and dead woody vegetation, including non-native wild tobacco (*Solanum mauritianum*), was associated with enhanced native tree seedling recruitment in oldfields of the Australian Wet Tropics. Furthermore, the origin and fruiting status of woody vegetation has been shown to be of less importance than its structure, with the provision of suitable perches being the strongest attractant for seed-dispersing birds (Elgar et al., 2014).

Conversely, an earlier field study in the Wet Tropics reported an absence of native rainforest tree recruitment under mature stands of wild tobacco (Florentine et al., 2003) and shade-house experiments indicated shoot and root growth of native rainforest trees may be impaired by wild tobacco leaf leachates (Florentine & Westbrooke, 2003). These conflicting findings led to the proposal by Elgar et al. (2014) that treatment to thin or remove the canopy of wild tobacco may be required to facilitate the ongoing recruitment of native rainforest trees into oldfields dominated by this species (Elgar et al., 2014). These findings highlight the difficulty in determining the net positive or negative benefits of non-native pioneer plants even within a single region. Therefore, increasing knowledge of the ecological values of specific non-native species, and the contexts in which their influence on regeneration can be facilitative or inhibitory, will be essential for ensuring the effectiveness of reforestation efforts in oldfields.

1.4 AIMS AND STRUCTURE OF THIS THESIS

The eastern Dorrigo Plateau is another part of eastern subtropical Australia that was once predominantly covered in rainforest and wet sclerophyll forest (Bridger, 1997, NSW FC 1962). However European settlement of the region resulted in significant conversion of forests into agricultural and pastoral land (Fisher et al., 1996). More recently, a non-native pioneer tree species has been primarily responsible for colonising oldfields in riparian areas throughout the Plateau: by the early 2000s, small-leaved privet (*Ligustrum sinense*) had become structurally dominant in woody regrowth vegetation in these areas. Due to concerns about the numerical dominance and spread of non-native tree species, considerable investments have been made in an attempt to reduce the spread of small-leaved privet (CHCC, 2017; ABC News, 2016). However, that approach to management does not consider whether there may be potential benefits associated with small-leaved privet’s role as a catalyst of regrowth on former pasture. Such benefits could include providing habitat for native fauna or facilitating rainforest regeneration, as has previously been shown for regrowth dominated by camphor laurel in the Big Scrub region (Neilan et al., 2006; Kanowski et al., 2008).

Therefore, this study aims to assess the potential for rainforest regeneration in oldfield regrowth dominated by small-leaved privet, on the eastern Dorrigo Plateau. It will achieve this aim by addressing three component questions.

First, the thesis asks: what are the ecological values of plant communities in post-pasture regrowth dominated by small-leaved privet? Plant communities in regrowth sites were evaluated by measuring vegetation structure and floristic composition and diversity, and similar measurements were made in reference sites of pasture and old-growth remnant rainforest. The regrowth values were then compared with values in reference sites to assess their relative degree of recovery from a pasture-like towards a rainforest-like state.

The second question is: what are the ecological values of bird communities in post-pasture regrowth dominated by small-leaved privet? From bird survey data at the same sets of sites, species composition and diversity in regrowth were compared with values in pasture and old-growth remnant rainforest, to likewise assess their relative degree of recovery from a pasture-like towards a rainforest-like state.

The third question is: what is the potential for future rainforest regeneration in privet-dominated regrowth areas? This question is addressed by further analysis of the plant and bird data, incorporating additional information about the ecological characteristics of the various species involved, and in particular those characteristics related to bird-assisted dispersal and recruitment of fleshy fruited trees and shrubs, and other relevant ecological information. The findings are interpreted by considering factors that may facilitate or inhibit rainforest regeneration in privet-dominated regrowth

The remainder of this thesis is structured as follows:
Chapter 2 describes the study area and the study design.
Chapter 3 investigates the ecological values of the regrowth plant communities.
Chapter 4 investigates the ecological values of the bird communities.
Chapter 5 considers how the plant and bird communities may create a potential for current and future rainforest regeneration in the regrowth.
Chapter 6 discusses the findings and their broader management implications.

2. STUDY AREA AND EXPERIMENTAL DESIGN

2.1 STUDY AREA

2.1.1 Location

The study area was located on the eastern Dorrigo Plateau within the Coffs Harbour Local Government Area of New South Wales, Australia. Study sites on the plateau were distributed between Lowanna (30° 13′ 0″ S, 152° 54′ 0″ E) in the north and Brooklana (30°18'02.0"S 152°53'25.2"E) in the south (Figure 2.1).

Figure 2.1: Location of the study region. The inset map shows the general location of the study region in northern New South Wales, Australia. The yellow box shows the extent of the study region on the eastern Dorrigo Plateau (Map layer source: Google Earth 2016).

2.1.2 Physical characteristics

The eastern Dorrigo Plateau comprises undulating terrain, with slight to moderate slopes, and is bounded by the steep hills of the Dorrigo and Orara sections of the Great Escarpment (NSW OEH, 2012). A number of rivers and creeks intersect the landscape of the eastern Dorrigo Plateau.

The geology of the eastern Dorrigo Plateau region consists of Carboniferous metasediments which have been formed by the deposition of ancient marine and/or riverine sediments (NSW OEH, 2012). Soil mapping recognised two soil landscapes in the region, Erosional Ulong and Colluvial Bobo (Milford, 1996), both of which were characterised by low fertility and high erosion potential (NSW NPWS, 2002). These soil landscapes comprised five different soil types which were differentiated according to topography: Red Earths, Yellow Earths, Kraznozems and Red Podsolics generally occurred on the mid to lower slopes of the study region and were typically well-drained and moderately deep to deep; while Lithosols occurred on the steeper slopes and tended to be very shallow (NSW NPWS, 2002). Vegetation is greatly influenced by local soil landscape (Fisher et al., 1996), with soils of the eastern Dorrigo Plateau predominantly supporting wet and dry sclerophyll forest, interspersed with rainforest in more protected, higher rainfall areas (NSW OEH, 2012).

The climate in the region ranges from subtropical to warm temperate (Fisher et al., 1996), with lower temperatures and higher rainfall typical in higher altitude areas (NSW OEH, 2012; NSW FC, 1962). Seasonal patterns result in warm, humid summers and dry, moderate to cool winters (Fisher et al., 1996). Bureau of Meteorology data for the nearby town of Dorrigo show that average temperatures, based on records spanning from 1997-2017, range from approximately 14-24°C in summer and 4-16°C in winter (BoM, 2017). Rainfall data for the same period indicate that the driest months coincide with winter and the wettest with late summer to early autumn, with mean rainfall ranging from approximately 58 mm in July to 313 mm in March (BoM, 2017).

2.1.3 Flora and fauna

The eastern Dorrigo Plateau falls within the diverse North Coast Bioregion of New South Wales which contains large numbers of threatened flora and fauna, including several species that are endemic to the region (NSW OEH, 2016).

Pre-European vegetation

At the time of European settlement, the eastern Dorrigo Plateau was covered in tree-dominated vegetation communities, particularly rainforests and sclerophyll forests (NSW FC, 1962). Rainforests of the region have been defined as multi-layered plant communities associated with high moisture levels, common features included: closed canopies; mixed compositions of typically broad-leaved and evergreen species; specialist life-forms such as vines, epiphytes and stranglers; and, sometimes, abundant buttressed trees (NSW FC, 1965). Four rainforest subtypes have been recognised, with Warm Temperate Rainforest identified as the dominant subtype across the eastern Dorrigo Plateau and Cool Temperate Rainforest recorded in small, fragmented patches along waterways in the area (NSW FC, 1965; Fisher et al., 1996).

Sclerophyll forests were defined by NSW FC (1965) as vegetation communities dominated by eucalyptus species, or eucalypt-like trees, common features included: a continuous canopy; a bole length greater than the crown depth; and, in most cases, a single tree layer. Three sclerophyll forest subtypes were recognised, with Wet Sclerophyll Forest identified as the dominant subtype across the eastern Dorrigo Plateau and defined as tall forest (height typically >30 m), often comprising an understorey of small scattered trees, a mesomorphic shrub layer and ground herbs (NSW FC, 1965).

Fisher et al. (1996) noted that the most abundant trees in intact Warm Temperate Rainforest were coachwood *(Ceratopetalum apetalum)*, black wattle *(Callicoma serratifolia)* and/or sassafras *(Sassafras* spp.*)*, while regeneration areas were dominated by Australian blackwood *(Acacia melanoxylon)* and black wattle. A number of plant families with high species diversity have also been recorded within the rainforest and wet sclerophyll plant communities of the region, including Lauraceae, Myrtaceae, Fabaceae and Proteaceae (NSW OEH, 2012). Surveys in 2012 indicated that the Lauraceae family was particularly diverse and abundant in rainforest and wet sclerophyll habitats, with 24 taxa recorded in the region, including species of *Cryptocarya, Endiandra* and *Neolitsea* (NSW OEH, 2012).

2.1.4 Land use history

At the time of European settlement the eastern Dorrigo Plateau was covered in extensive swathes of subtropical rainforest (Bridger, 1997). Prior to this period, the indigenous Australian practice of using fire as a habitat management tool may have altered the vegetation of the region to some degree (Fisher et al., 1996). However, it was the land modification activities that came with European settlement that really left their mark on the landscape.

Historical accounts indicate that timber harvesting activities in the Coffs Harbour region commenced as early as the mid to late 19th century, with foresters harvesting hoop pine *(Araucaria cunninghamii)*, ash *(Eucalyptus regnans)* and Antarctic beech *(Nothofagus moorei)* on the eastern Dorrigo Plateau (Coffs Harbour Heritage Study, cited in Fisher et al., 1996). From a harvest perspective, coachwood was recognised as an important rainforest tree in the region (NSW FC, 1962). Large saw mills were constructed in the late 19th to early 20th centuries, and in 1924 the Dorrigo Railway was established to enable the transport of timber; these activities contributed to increased logging pressure in the region (Fisher et al., 1996).

Mining also impacted the natural landscapes of the Coffs Harbour region in the past, with the discovery of gold and copper in the region in the mid to late 19th century resulting in rapid population growth and considerable habitat degradation (Fisher et al., 1996). At least one historic mine was located within the vicinity of Lowanna on the eastern Dorrigo Plateau.

In the late 19th to early 20th centuries, shortly after timber harvesting opened up the area, increasing numbers of people began to settle across the eastern Dorrigo Plateau to establish agricultural and grazing activities, and consequently cleared more land (Fisher et al., 1996). By 1920, a pattern had been well-established, whereby sheep and beef cattle graziers were distributed across the tablelands and dairying activities occupied the high-moisture areas with good soil (NSW FC, 1962). These activities resulted in the conversion of large areas of tree-dominated vegetation to pasture by the mid-20th century.

However, by the mid-20th century a number of protected forest reserves, including both dedicated State Forests and National Parks, had also been established across the Coffs Harbour region (NSW FC, 1962). Most of these protected areas were located on broken topography with poor soils and many of them were designated as multiple-use, meaning timber harvesting, research and recreation activities were all potentially permissible (Fisher et al., 1996).

From the late-20th century onwards, many pastoral areas across the eastern Dorrigo Plateau were retired, particularly along riparian zones, allowing fragmented patches of self-organised regrowth to establish. Erskine et al. (2007) noted that in tropical and sub-tropical Australia more generally, agricultural practices were sometimes abandoned shortly after rainforest was cleared due to a lack of productivity. However, retirement of pastures has increasingly occurred following more recent (late 20th century) declines in key agricultural industries and changes to trade policies (Erskine et al., 2007). The specific processes that led to the retirement of pastures in the study region were not identified during this study; however discussions with local residents indicated that there had been a shift from dairying towards beef cattle and sheep grazing in the late-20th century, which may have facilitated the establishment of woody regrowth in some areas.

In 2011, the major industries of employment in the area still included sheep and beef cattle farming, and forestry and logging (ABS, 2011a; ABS, 2011b). The census for 2011 showed that population density in the region remained low in the 21st century, with the eastern Dorrigo Plateau (including Lowanna, Ulong and Brooklana) supporting a total population of 587 people (ABS, 2011a; ABS, 2011b).

At the time of this study, the eastern Dorrigo Plateau was characterised by a fragmented landscape with scattered remnant forest patches, including both protected areas and active logging coupes, embedded in a matrix of human modified land uses and small regrowth patches varying in size, age and proximity to intact rainforest. Many patches of regrowth were dominated by non-native invasive trees, especially small-leaved privet *(Ligustrum sinense),* a large shrub or small tree that was originally planted as a popular hedge plant or windbreak due to its frost resistant characteristics (Bellinger Landcare Inc., 2006). Small-leaved privet appeared to be the most prolific non-native species on the eastern Dorrigo Plateau, with Coffs Harbour City Council mapping showing significant areas across the region (Figure 2.2).

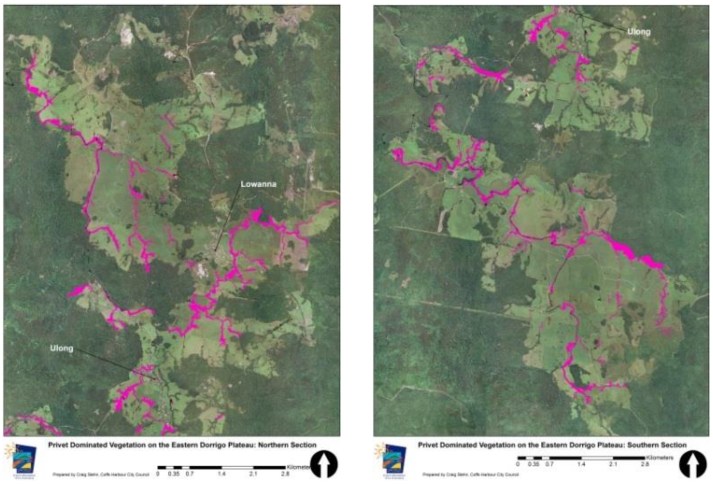

Figure 2.2: Small-leaved privet distribution on the eastern Dorrigo Plateau, northern (left) and southern (right) maps. Map extent covers an area of <20 x 10 km (map layer source: Coffs Harbour City Council, 2014).

2.2 EXPERIMENTAL DESIGN

2.2.1 Site selection and study design

The study design comprised 21 survey sites dispersed across the region, placed within each of three habitat types (Figures 2.3 and 2.4; see also Appendix I): pasture (8 sites), regrowth (8 sites) and rainforest (henceforth also termed “forest”; 5 sites).

Forest site selection criteria were: rainforest vegetation with a closed canopy, with possible gaps created by tree falls or other disturbances; the presence of three vegetation layers, comprising canopy, shrubby understorey and ground layer; the presence of multiple functionally-diverse life-forms, such as vines, epiphytes, ferns or cycads; a diversity of tree species with varying stem classes and leaf sizes; the presence of leaf litter and woody debris; and eucalypt species not commonly represented as emergent trees. All forest sites were protected within State Forest or National Park tenure.

Figure 2.3: Examples of the three habitat classes used in the study: (a) pasture, site P1 located near Lowanna on the eastern Dorrigo Plateau; (b) regrowth, site R2 located in the vicinity of Brooklana; (c) forest, site F3 located in the middle of the study region near Ulong.

Regrowth site selection criteria were: historical clearing of original forest vegetation and use of the land as livestock pasture during much of the twentieth century; the presence of a linear riparian strip of dense and spatially continuous woody regrowth about 5-10 m tall, and dominated by small-leaved privet *Ligustrum sinense* (riparian areas being the main parts of the landscape in which post-pasture regrowth occurred); and the apparent absence of any active management, such as weed control activities or restoration works.

Pasture site selection criteria were: the occurrence of stock grazing, either on a continuous or rotational basis; dominance by short, dense pasture grasses over a continuous area of 0.5 ha, with trees and shrubs absent except for scattered individuals; and location of each site on the same property as a selected regrowth site.

Additionally, all sites met the following overarching criteria: located at elevations 500-650 m asl; within a minimum local patch size of 0.4 ha of contiguous habitat; and easily accessible from a roadway. A minimum distance of 100 m separated regrowth sites from pasture sites, whereas forest sites were spatially separated from all other sites by 500-3000m, because continuous remnant rainforest was located at the margins of the study region (Figure 2.4). All forest and regrowth sites were within 50 m of a waterway (one edge of three forest sites and six regrowth sites being within 10 m of the waterway), whereas pasture sites were necessarily between 50 and 250 m of a waterway, due to the presence of woody regrowth along streams.

Potential study sites were initially identified through the study of Google Earth satellite imagery and small-leaved privet distribution maps (Figure 2.2), then screened against the selection criteria described above and refined through consultation with a Coffs Harbour City Council environmental officer familiar with the study region. Selection of pasture and regrowth sites was also influenced by the necessity for access agreements with private landholders. Each site was then ground-truthed prior to final selection.

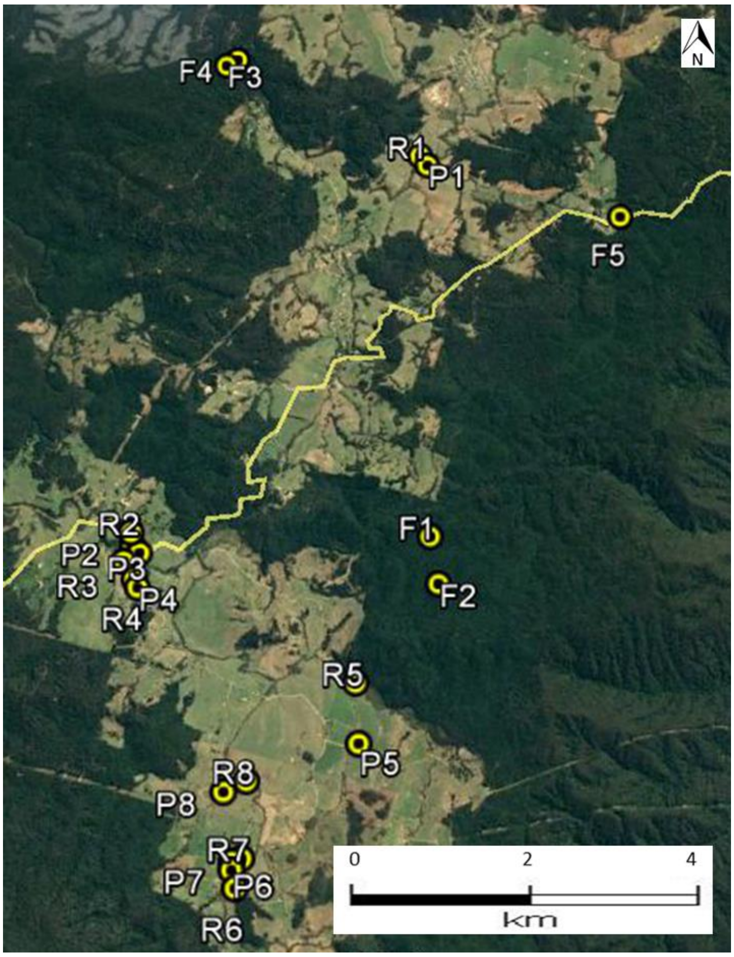

Figure 2.4: Location of the 21 study sites on the eastern Dorrigo Plateau. Yellow circles show the location of each study site, with labels P1-P8 representing pasture sites, labels R1-R8 representing regrowth sites and labels F1-F5 representing forest (map layer source: Google Earth, 2016).

3. ECOLOGICAL VALUES OF REGROWTH PLANT COMMUNITIES

3.1 INTRODUCTION

The vegetation structure of pasture is markedly different from that of rainforest. Pasture is typified by a simple ground layer of grasses or herbs and an absence of trees and shrubs, or occasionally a small number of scattered trees. Conversely, rainforest habitat is associated with high levels of canopy cover, canopy height, tree density, large trees, tree basal area and woody debris, a high frequency of specific life forms that occur mainly in rainforest vegetation and little or no cover of grass (Kanowski et al., 2010). Special life forms of rainforest include robust vines, epiphytes, hemi-epiphytes and strangler figs (Kanowski et al., 2005). Mature rainforests typically harbour a variety of functionally diverse plant life forms and a high diversity of plant species with differing stem classes and leaf sizes (McDonald & Hunter, 2010, Kanowski et al, 2003 and; Catterall et al, 2004).

The structural features of rainforest are also correlated with the provision of habitat for rainforest fauna (Kanowski et al., 2010), and rapid assessments of vegetation structure can be useful as an aid to determining the relative ecological values of different areas (McElhinney et al., 2005). However, to gain a deeper understanding of ecological value, it is necessary to also investigate the floristic composition of an area's vegetation.

The present chapter addresses the first of this study's three key research questions: what are the ecological values of the plant communities in areas of post-pasture regrowth vegetation that are dominated by small-leaved privet?

It does so through quantitative analysis of data on vegetation structure and floristic composition, collected at all sites, and analysed to reveal differences between pasture, regrowth and forest habitats. Detailed investigations of a range of structural attributes, plant species richness, and the density and taxonomic composition of the flora, enable an assessment of the extent to which the areas of regrowth have recovered those structural and floristic attributes associated with intact remnant forest which had been lost on its conversion to pasture.

3.2 METHODS

3.2.1 Field measurements: vegetation structure

Data on vegetation structure were derived from transect surveys undertaken within each of the 21 study sites (Forest N = 5, Regrowth N = 8, Pasture N = 8) from 14/04/2015-26/06/2015. Vegetation structure measurements followed the methodology of Kanowski et al. (2010) and comprised measurements of ground cover, occurrence of special life forms, canopy characteristics, size specific tree stem densities and basal area. Each site contained a pair of variable width (maximum 20 m for vegetation surveys) transects, each measuring 50 m in length. Wherever possible, transects were run end to end in a single 100 m line, however in some narrow regrowth sites transects were split into multiple sections to account for local spatial constraints.

Ground cover measurements involved estimating the proportion (percentage) of ground covered by: vegetation <1 m high (recorded separately for grasses, herbs, ferns, vines and moss); litter (comprising leaf litter and woody debris); bare ground (comprising rock and bare soil); and other (comprising tree trunks, roots and other). Ground cover up to 1 m above the surface was assessed in each of six 1 m x 1 m plots (with three plots present per transect, placed at the 5 m, 25 m and 45 m marks around the transect mid-line, and the total ground cover equalling 100% for each plot). Woody debris was measured in fine (<10 cm) and coarse (≥10 cm) diameter classes by counting all fallen logs and branches, lying on or within 1 m above the ground, that intercepted the mid-line of each transect.

Special life forms of rainforest comprised: strangler figs; slender vines (<5 cm diameter); robust vines (≥5 cm diameter); vine tangles; individual thorny scramblers; thicket-forming scramblers; small palms; tree ferns; ground ferns; epiphytic ferns; hemi-epiphytes; other epiphytes; strap-leaf herbs (such as lilies); and cordylines. Special life forms were measured as either present or absent in each of six 10 m x 10 m quadrats at each site (with three quadrats per transect, placed at the 5 m, 25 m and 45 m marks around the transect mid-line). Other special life forms listed in Kanowski et al (2010) that were absent from all the study sites were: vine towers; palm trees; wide-leaf herbs (such as gingers); cycads with above-ground stems; cycads with below-ground stems; and pandanus.

Canopy characteristics (average canopy height and canopy cover) were measured through visual assessment within each of the six 10 m x 10 m quadrats described above. Canopy height was the height of the tallest tree in the canopy, excluding gaps and scattered emergent trees. Canopy cover was the projective cover of vegetation (leaves, branches and trunks) >2 m above ground level (equivalent to the shade cast by all vegetation >2 m high, if the sun was directly overhead).

Tree stem density measurements were counts of free-standing woody-stemmed plants >1 m high, classified into size classes according to dbh (the trunk diameter at 1.3 m), within the following survey areas: within 2.5 m either side of the transect mid-line for stems <10 cm (500 m^2 per site); within 5 m of the mid-line for stems 10-20 cm and 20-50 cm (1,000 m^2 per site); within 10 m for stems >50 cm (2,000 m^2 per site). Multi-stemmed plants were assigned to a representative dbh class using: combined dbh = $\sqrt{\sum dbh_i^2}$; where dbh_i is the dbh of each stem. Stags (dead free-standing woody-stemmed plants >1 m high and >10 cm dbh) were also counted, in the same dbh classes and survey areas.

Additionally, tree basal area per hectare was measured as the cross-sectional area of trees, calculated from dbh classes, as follows: basal area (m^2 per ha) = $\Sigma\, n_i\, \pi\, (dbh_i)$ 2/40000, where n_i is the number of stems in the i^{th} size class/ha, and dbh_i is the notional mean diameter (cm) of the i^{th} size class (= midpoint of class – 5% of class range; from Kanowski et al. 2008). Stag basal area was measured in the same manner.

Prior to data analyses, each structural attribute was expressed as a single overall value for each of the 21 sites, as follows: (a) each ground cover and canopy variable was averaged across the six within-site measurements; (b) the frequency of each special life-form was the number of within-site quadrats where it was present (range 0-6) and special life-form richness was the number of different life forms present (maximum potential value = 14); and (c) all tree and stag density counts were converted to stems per hectare (stems/ha <10cm dbh = stem count x 20; stems/ha 10-20cm and 20-50cm dbh = stem counts x 10; stems/ha >50cm dbh = stem count x 5) and these stem densities were then condensed into three categories <20cm dbh, 20-50 cm dbh and >50 cm dbh.

3.2.2 Field measurements: floristics

Floristic data collection followed the methodology of Kanowski et al. (2010), and comprised species-level identification (wherever possible) and abundance measurements of all vascular plants – tree seedlings, established (>1m tall) trees and shrubs, and other life forms. Data were collected in the same pair of variable width transects used for vegetation structure surveys, with floristic surveys also undertaken from 14/04/2015-26/06/2015. All floristic identifications were by an experienced field botanist (Dr. Bill McDonald). When species level identifications were not possible, the finest possible taxonomic resolution was used.

Tree and shrub seedlings were defined as living, free-standing, woody-stemmed plants <1 m tall, and were identified as "present" within 2.5 m either side of the transect mid-line. Larger (> 1 m tall) trees and shrubs were counted as the numbers of stems per species, within the same size classes as used for the vegetation structure (i.e. the stems in the vegetation structure data were also identified to species level). Prior to floristic analyses, all tree and shrub species stem counts were converted to stems per hectare, as described for vegetation structure, and then grouped into the following stem classes: <2.5cm dbh, 2.5-10cm dbh and >10cm dbh (these differed from the classes used for vegetation structure analyses, due to low representation of larger stem size classes within many species).

Other life forms of vascular plants were identified as species "present" in each of the six 10 m x 10 m quadrats. For each ground cover species (ferns, herbs and grasses), the percent cover of each was estimated in each of the 10 m x 10 m quadrats, and these values were then averaged across the six quadrats to obtain a site-level value for each species. Vines and epiphyte species were recorded as "present" in the same quadrats, with records expressed as site level frequency scores (0-6), and then converted to fractions, prior to analysis.

General qualitative comments on vegetation health, habitat disturbance, dominant or notable species and common recruits were also recorded for each site.

3.2.4 Data analyses

Taxonomic and functional attributes

All vascular plant species were classified according to three sets of taxonomic or functional criteria (as described by Shoo et al. (2016): family; life-form (trees/shrubs, vines, epiphytes, ferns, herbs, grasses); and origin (native or non-native).

Vegetation structure

There were 32 individual site-level vegetation structure variables (Appendix II), from which those recorded at four or more of the 13 combined forest and pasture sites were included in univariate analyses of the extent of their relative recovery in regrowth sites, using single-factor ANOVA to compare their values among the three habitats. In cases where the ANOVA $P<0.10$, the variable's 'relative recovery distance' (RRD) in regrowth was also calculated, as follows. RRD = 100 x (R mean – P mean) / (F mean – P mean), where R = regrowth, P = pasture and F = forest.

Four overall recovery categories were then allocated to attributes that differed significantly ($p<0.05$) or near-significantly ($0.05<p<0.10$) among the three habitats, as follows: F (fast) 71%-130%; I (intermediate) 30-70%; S (slow) between -29% and 29%; D (deflected) ≤ -30% or >130% or F and P both zero but R values are larger.

The pattern of variation in vegetation structure was also analysed using multi-dimensional scaling (MDS) ordination to visualise the overall pattern of similarity or difference among individual sites, and analysis of similarity (ANOSIM) to test for significant differences among habitat types (first among all three habitat types and then between habitat pairs). Biplot vectors were added to the ordination, to identify the key attributes associated with differences among sites. These analyses used all vegetation structure attributes which had non-zero values at two or more of the 21 sites, with all values range standardised to remove the effect of the variety of different measurement scales, using: (V - min V) / (max V - min V); where V = the site-specific value of a variable, min V = its smallest value across all sites and max V = its largest value.

All ordinations and ANOSIMs (for vegetation structure and other attributes) were conducted using the "vegan" package (Oksanen et al., 2013), in R software version 3.1.1 (R Foundation for Statistical Computing, 2014), and used the Euclidian distance to measure inter-site differences.

Floristic composition of trees and shrubs

All floristic tree and shrub data were log-transformed prior to statistical analyses to reduce the influence of outliers and improve normality. Recovery rates of tree/shrub species richness and density in regrowth were assessed by comparing values of the following nine variables among the three habitats using ANOVAs, and by calculating their RRD values and assigning recovery categories (as described previously for vegetation structure): species richness of seedling trees/shrubs (<1m tall); species richness and density (stems/ha) of larger trees/shrubs (>1m tall), both in total and within each of the three stem size classes (<2.5 cm, 2.5-10cm, >10cm dbh). These analyses were repeated three times: for all species, for native species, and for non-native species, resulting in 27 sets of tests.

The ten most abundant tree families in remnant rainforest, based on their average density of large stems (>10cm dbh) per site across the five forest sites, were identified. Their recovery rates in regrowth were then assessed by comparing the site-specific stem densities (>10 cm dbh) of each family among the three habitats using ANOVAs, and by calculating their RRD values and assigning recovery categories as described previously for vegetation structure. Multivariate MDS ordination and ANOSIM analyses of overall floristic differences were also undertaken, as previously described for vegetation structure, using site-specific stem densities (>10 cm dbh) of all tree and shrub families with non-zero values at two or more sites. A similar series of analyses was also conducted at species level, in this case using the site-specific values of stem density per species for all stems >1m tall.

Occurrence and floristic composition of other life forms

Recovery rates in regrowth of five other life forms (ferns, epiphytes, vines, herbs and grasses) were likewise each assessed using among-habitat ANOVAs, together with RRD measurements and assignment of recovery classes. Analyses used relative abundances, measured as site-level frequencies for vines and epiphytes, and percent cover for ferns, herbs and grasses. Within each life form there were three potential analyses: for all species, native species, and non-native species. ANOSIM tests of difference in species composition between habitat pairs were also undertaken for ferns, vines, herbs and grasses (epiphytes were restricted to forest sites only), each time using site-specific relative abundances of all species.

3.3 RESULTS

3.3.1 Vegetation structure

Among the 24 vegetation structure variables tested, 23 differed significantly ($p<0.05$) or strongly ($0.05<p<0.10$) among the three habitats (Table 3.1). Only herb cover lacked strong differences among the habitats. A further eight vegetation structure variables (all special life forms of rainforest) were not analysed as they occurred at fewer than four sites: strangler figs, hemi-epiphytes, robust vines (≥5 cm diameter); vine tangles, individual scramblers, thicket-forming scramblers, other epiphytes and cordylines.

One variable, percent bare ground, had a “deflected” recovery pattern (Table 3.1; Fig. 3.1). Six variables showed fast recovery: grass percent cover, ground ferns, canopy cover, densities of stems <20cm dbh and 20-50cm dbh, and tree basal area all had values in regrowth sites that were close to those in the forest (Table 3.1; Fig. 3.2), but with forest values that differed strongly from those in pasture. Percent grass cover was the only variable which had much greater values in pasture than in forest.

Nine variables showed intermediate recovery: percent ground cover of ferns, mosses, litter and other, fine woody debris, slender vines, life form richness, canopy height and density of stems >50cm dbh all had values in regrowth sites that were greater than those in pasture but less than forest (Table 3.1; Fig. 3.3). Seven variables showed slow or no recovery: percent ground vine cover, coarse woody debris, small palms, tree ferns, epiphytic ferns, strap-leaf herbs and stag basal area all had values in regrowth sites that were close to those in pasture, together with forest values that greatly exceeded those in pasture (Table 3.1; Fig. 3.4).

The three habitats differed significantly in overall vegetation structure (ANOSIM global $R = 0.77$, $p = 0.0002$; pasture vs forest $R = 1.0$, $p = 0.001$; regrowth vs pasture $R = 0.66$, $p = 0.0008$; regrowth vs forest $R = 0.72$, $p = 0.002$). Ordination revealed that regrowth sites were intermediate between forest and pasture (Fig. 3.5). Only grass cover was associated with pasture sites, while other variables tended to be associated with forest sites to varying degrees, except for herb cover which was not associated with any specific habitat but which had high values in one pasture and one regrowth site (Fig. 3.5; see also Figs. 3.1-3.4).

Table 3.1: Values (mean, SE) of all vegetation structure variables in pasture (P), regrowth (R) and forest (F) and the results of ANOVAs among habitat types; significant differences ($p<0.05$) are bolded. In cases where ANOVA $p<0.10$, the relative recovery distances (RRD) and recovery rates are also shown; N = 8, 8, 5 sites in P, R, F respectively.

Variable type	**Measurement**	**P mean (SE)**	**R mean (SE)**	**F mean (SE)**	**ANOVA *p value***	**RRD[1]**	**Rate[1]**
Ground							
Grass cover	% cover	86.6 (7.3)	19.8 (7.6)	0.4 (0.3)	**<0.0001**	77%	F
Fern cover	% cover	0.0	1.0 (0.5)	1.8 (0.7)	**0.05**	59%	I
Vine cover	% cover	0.00	0.13 (0.05)	1.10 (0.35)	**0.0002**	12%	S
Herb cover	% cover	7.2 (5.2)	8.1 (4.3)	1.1 (0.6)	0.57	-	-
Moss cover	% cover	0.0	0.9 (0.6)	2.1 (0.9)	0.06	40%	I
Litter cover	% cover	0.3 (0.2)	31.4 (3.6)	69.2 (6.5)	**<0.0001**	46%	I
Bare ground	% cover	5.6 (2.3)	33.2 (5.7)	14.9 (4.9)	**0.0008**	296%	D
Other cover	% cover	0.3 (0.2)	5.6 (1.1)	9.8 (2.2)	**0.0001**	58%	I
Fine woody debris	no./site	0.1 (0.1)	26.0 (7.3)	49.8 (8.5)	**0.0001**	52%	I
Coarse woody debris	no./site	0.0	1.9 (0.7)	9.0 (3.0)	**0.0006**	21%	S
Special life forms							
Slender vines	% frequency	0.0	0.6 (0.2)	1.0 (0.0)	**<0.0001**	60%	I
Small palms	% frequency	0.00	0.00	0.97 (0.03)	**<0.0001**	0%	S
Tree ferns	% frequency	0.00	0.01 (0.01)	0.67 (0.14)	**<0.0001**	1%	S
Ground ferns	% frequency	0.0	0.7 (0.1)	0.9 (0.1)	**<0.0001**	77%	F
Epiphytic ferns	% frequency	0.00	0.06 (0.06)	0.70 (0.11)	**<0.0001**	9%	S
Strap-leaf herbs	% frequency	0.0	0.0	1.0 (0.0)	**<0.0001**	0%	S
Life form richness	no./site	0.0	2.9 (0.3)	8.6 (0.8)	**<0.0001**	33%	I
Canopy and trees							
Canopy height	metres (m)	0.0	7.1 (0.4)	23.9 (3.4)	**<0.0001**	30%	I
Canopy cover	% cover	0.0	45.6 (3.6)	63.0 (2.9)	**<0.0001**	72%	F
Stems <20cm dbh[2]	stems/ha	0.1 (0.1)	3.8 (0.2)	3.8 (0.1)	**<0.0001**	105%	F
Stems 20-50cm dbh[2]	stems/ha	0.0	2.3 (0.1)	2.3 (0.2)	**<0.0001**	101%	F
Stems >50cm dbh[2]	stems/ha	0.0	0.8 (0.2)	1.4 (0.2)	**<0.0001**	59%	I
Tree basal area	m^2/ha	0.02 (0.02)	36.38 (6.16)	50.18 (7.75)	**<0.0001**	73%	F
Stag basal area	m^2/ha	0.0	0.4 (0.4)	7.6 (3.1)	**0.002**	5%	S

[1] RRD = 100 X (R mean – P mean)/(F mean–P mean); rate = F fast, I intermediate, S slow, respectively RRD 71-130%, 30-70%, -29-29%; D deflected, RRD <= -30% or >130% or F and P both zero but R values are larger.

[2] Log$_{10}$-transformed

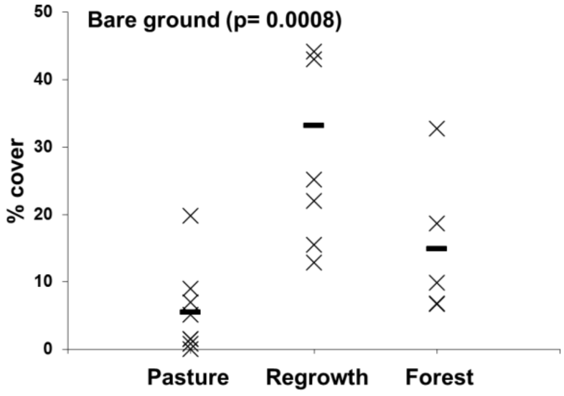

Figure 3.1: Bare ground cover had a "deflected" recovery (>130%) in regrowth (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and the p value was obtained from the ANOVA.

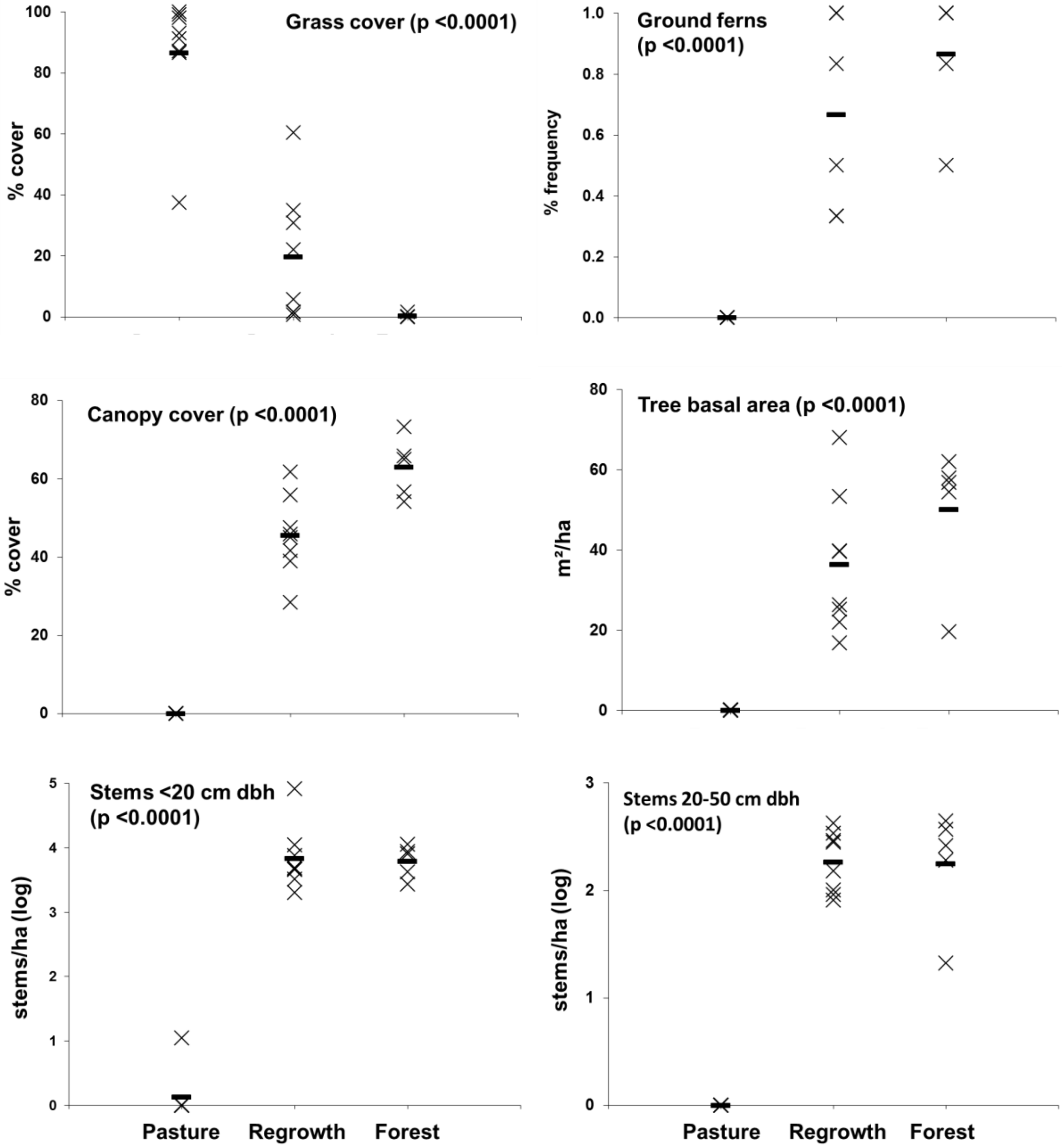

Figure 3.2: Vegetation structure attributes which had achieved fast or full recovery (71-130%) in regrowth (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and p values are from ANOVAs.

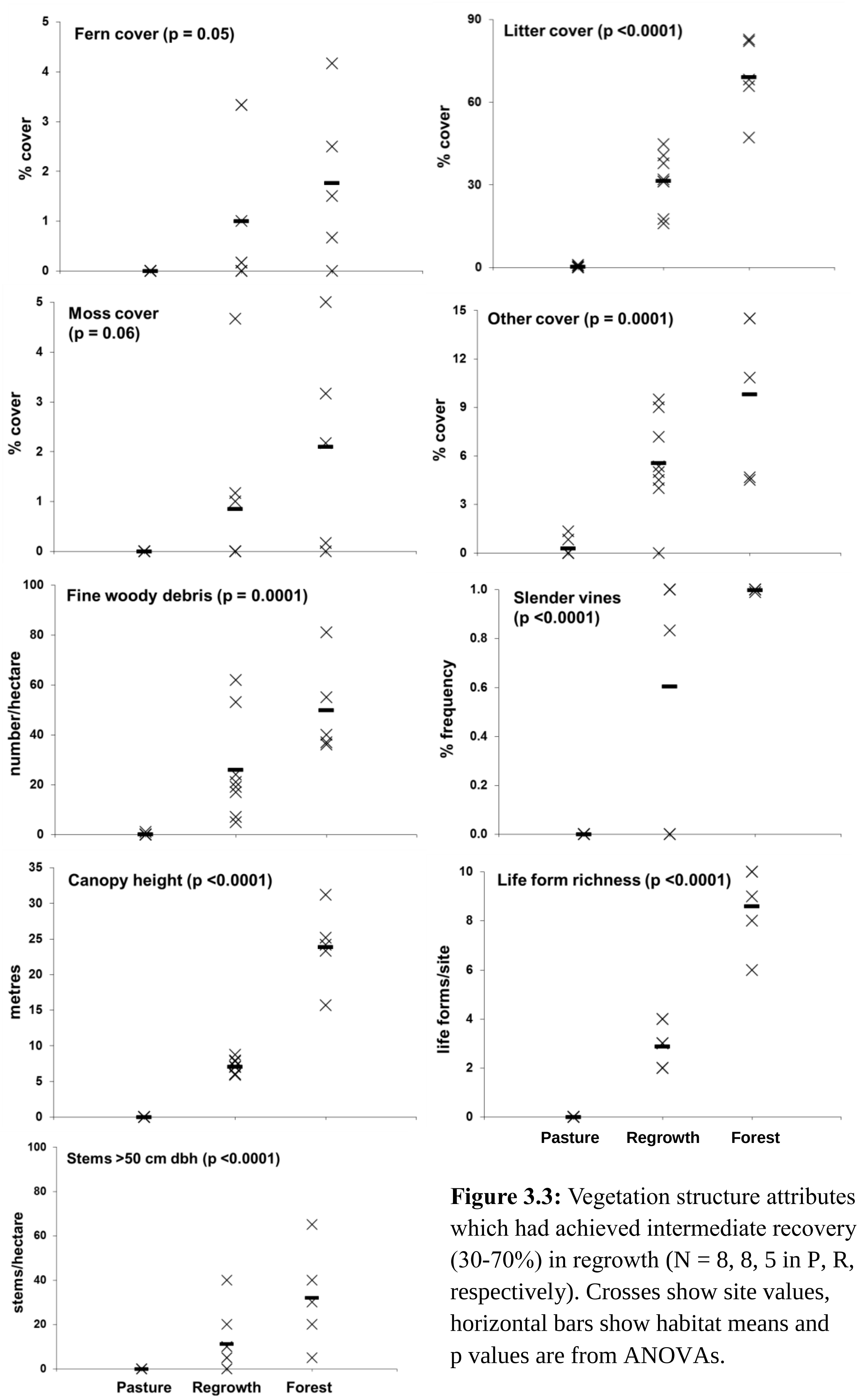

Figure 3.3: Vegetation structure attributes which had achieved intermediate recovery (30-70%) in regrowth (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and p values are from ANOVAs.

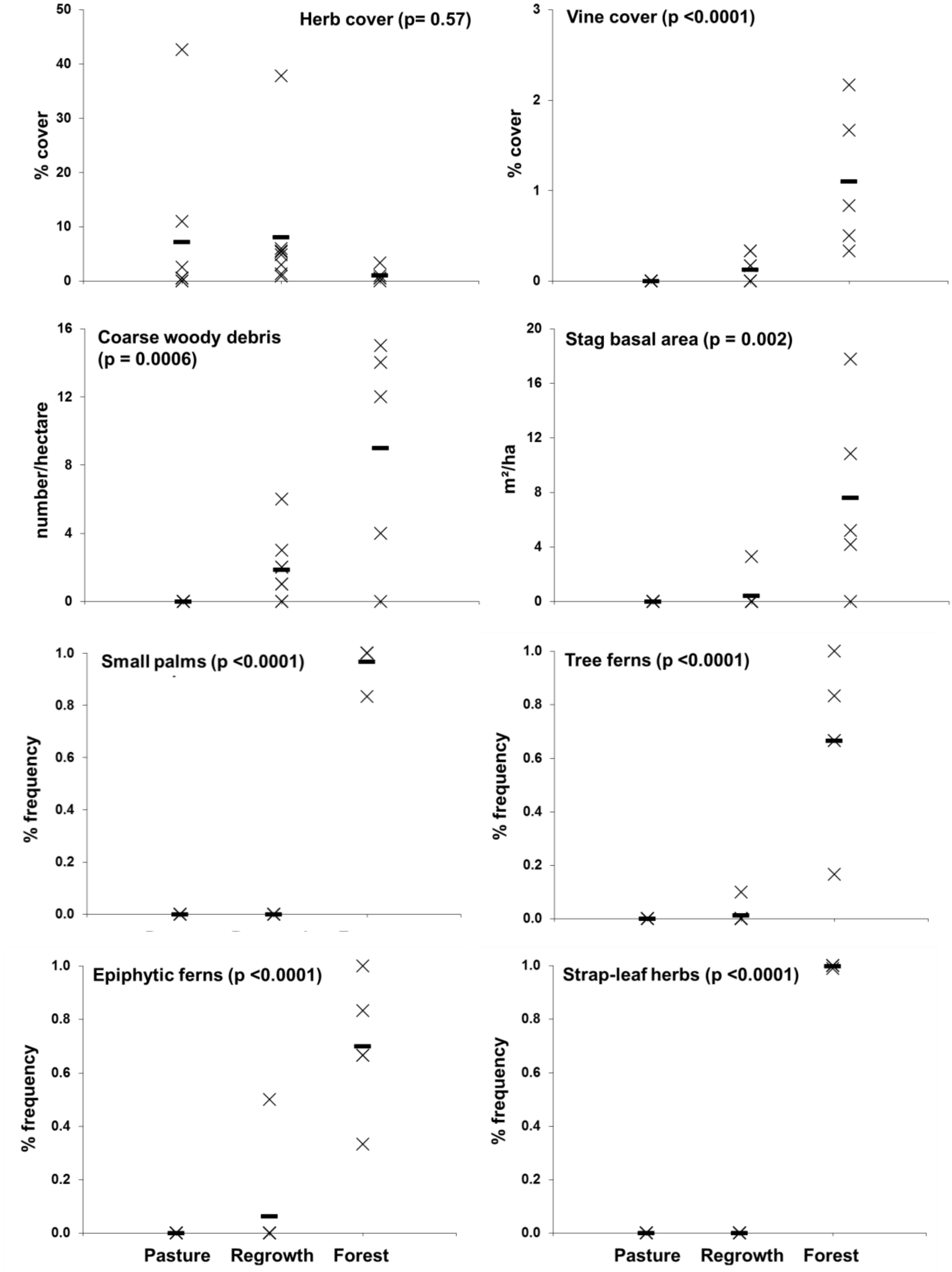

Figure 3.4: Vegetation structure attributes which had achieved slow or no recovery (-29%-29%) in regrowth (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and p values are from ANOVAs.

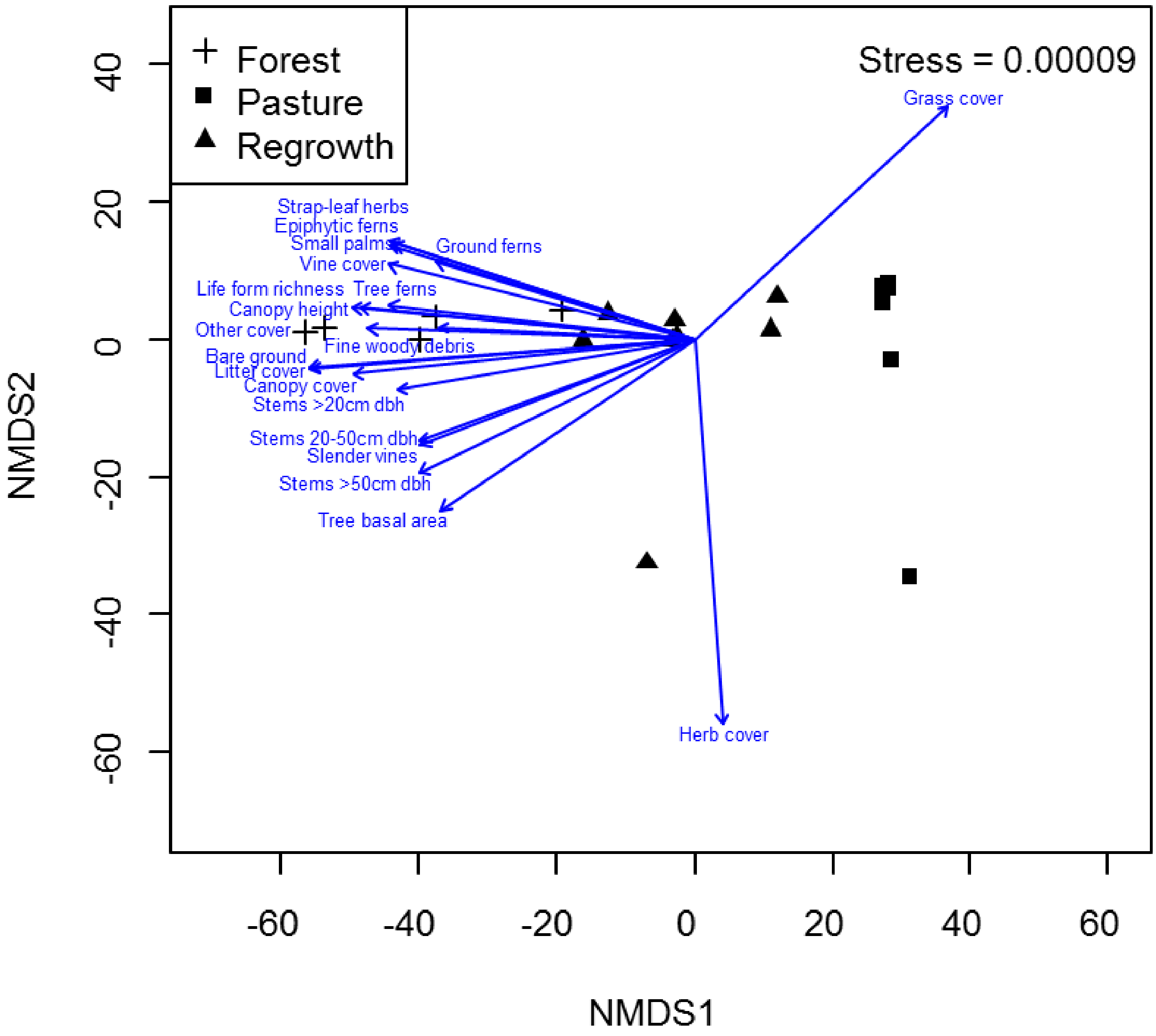

Figure 3.5: MDS ordination of 21 sites (N = 8, 8, 5 for P, R, F respectively) based on Euclidean dissimilarity of 30 vegetation structure variables with non-zero values at ≥2 sites. Arrows show biplot vectors for all intrinsic variables whose site specific values were very significantly associated ($p < 0.01$) with sites' locations in the plot.

3.3.2 Floristic diversity and composition of trees and shrubs

A total of 94 tree and shrub species, belonging to 38 families, were recorded across all 21 sites (Appendix III). Recovery rates in regrowth of native trees and shrubs >1m tall were slow for all variables except total density of stems >10 cm dbh, which had an intermediate recovery rate (Table 3.2). A similar pattern was seen when native and non-native species were combined (Table 3.2). In contrast, non-native species richness and density of all stems had a "deflected" pattern of difference, because non-native stems were a component of regrowth but were absent from both pasture and forest (Table 3.2). For seedlings (<1 m high), the pattern was broadly similar, except that the species richness of non-native seedlings did not differ significantly among habitats; some non-native species occurred in both pasture and regrowth, leading to an “intermediate” recovery rate for total species richness (Table 3.2).

Eight of the ten most abundant forest tree and shrub families had achieved slow or no recovery in regrowth (Table 3.3): Lauraceae, Cunoniaceae, Grossulariaceae, Sapindaceae, Monimiaceae, Winteraceae, Araucariaceae and Celastraceae, with Myrtaceae and Proteaceae showing intermediate recovery. More broadly, the three habitats differed significantly in overall community composition of tree and shrub families, based on stems >10 cm dbh (ANOSIM global R = 0.46, p = 0.0002; pasture vs forest R = 0.73, p = 0.0008; regrowth vs pasture R = 0.25, p = 0.004; regrowth vs forest R = 0.62, p = 0.001).

Family-level ordination revealed that sites in regrowth were more similar to pasture than forest habitat (Figure 3.6), with both regrowth and pasture sites being tightly clustered, indicating low variability among sites, whereas forest sites were more widely separated in the ordination, indicating much greater variability. Nineteen of the 29 analysed families were significantly ($p<0.05$) associated with this pattern (12 having $p<0.01$). Most families were associated with forest habitat, while Myrtaceae tended to be aligned between forest and regrowth (Figure 3.6; see also Table 3.3).

At the individual species level, the ten most abundant forest tree and shrub species (>1 m tall) all showed slow or no recovery in regrowth (Table 3.4). More broadly, the community composition of tree and shrub species >1 m tall differed significantly among the three habitats (ANOSIM global R = 0.56, p = 0.0002; pasture vs forest R = 0.86, p = 0.001; regrowth vs pasture R = 0.28, p = 0.0004; regrowth vs forest R = 0.90, p = 0.0008), with ordination (Figure 3.7) again showing a pattern in which regrowth was clearly distinct from forest habitat, and more similar to pasture, with forest having greatest among-site variability. Of the 58 analysed tree and shrub species, 40 were significantly ($p<0.05$) associated with the ordination pattern (18 having $p<0.001$), and all being associated with the forest region of the ordination plot.

Table 3.2: Species richness and density (mean, SE) of trees and shrubs in pasture (P), regrowth (R) and forest (F), overall and by size class and origin, and the results of ANOVAs among habitat types; significant differences (p<0.05) are bolded. In cases where ANOVA p<0.10, the relative recovery distances (RRD) and recovery rates are also shown. Species richness measured as species/site; density as stems/ha; N = 8, 8, 5 sites in P, R, F respectively.

Variable*	**P mean (SE)**	**R mean (SE)**	**F mean (SE)**	**ANOVA *p value***	**RRD[1]**	**Rate[1]**
Species richness – seedlings (<1 m tall)						
Total species richness	0.4 (0.1)	5.4 (0.6)	18.2 (1.0)	**<0.0001**	30%	I
Native species richness	0.0	4.4 (0.5)	18.2 (1.0)	**<0.0001**	24%	S
Non-native species richness	0.4 (0.1)	0.9 (0.1)	0.0	0.11	-	-
Species richness – all stems >1 m tall						
Total species richness	0.13 (0.04)	4.75 (0.25)	36.00 (1.79)	**<0.0001**	13%	S
Native species richness	0.13 (0.04)	3.38 (0.21)	36.00 (1.79)	**<0.0001**	9%	S
Non-native species richness	0.0	1.4 (0.1)	0.0	**<0.0001**	-	D
Species richness – stems >1 m tall and <2.5 cm dbh						
Total species richness	0.0	2.9 (0.3)	28.6 (1.9)	**<0.0001**	10%	S
Native species richness	0.0	1.8 (0.2)	28.6 (1.9)	**<0.0001**	6%	S
Non-native species richness	0.00	1.13 (0.04)	0.00	**<0.0001**	-	D
Species richness – stems 2.5-10 cm dbh						
Total species richness	0.0	3.0 (0.2)	19.8 (1.2)	**<0.0001**	15%	S
Native species richness	0.0	1.8 (0.2)	19.8 (1.2)	**<0.0001**	9%	S
Non-native species richness	0.0	1.3 (0.1)	0.0	**<0.0001**	-	D
Species richness – stems >10 cm dbh						
Total species richness	0.13 (0.04)	3.63 (0.20)	16.20 (1.07)	**<0.0001**	23%	S
Native species richness	0.13 (0.04)	2.63 (0.20)	16.20 (1.07)	**<0.0001**	16%	S
Non-native species richness	0.0	1.0 (0.0)	0.0	**<0.0001**	-	D
Density – all stems >1 m tall						
Total density	0.1 (0.1)	16.0 (0.8)	101.0 (5.8)	**<0.0001**	16%	S
Native density	0.10 (0.01)	8.80 (0.70)	101.0 (5.80)	**<0.0001**	9%	S
Non-native density	0.0	7.2 (0.2)	0.0	**<0.0001**	-	D
Density – stems >1 m tall and <2.5 cm dbh						
Total density	0.0	10.5 (0.7)	73.0 (5.8)	**<0.0001**	14%	S
Native density	0.0	4.7 (0.7)	73.0 (5.8)	**<0.0001**	6%	S
Non-native density	0.0	5.7 (0.3)	0.0	**<0.0001**	-	D
Density – stems 2.5-10 cm dbh						
Total density	0.0	10.8 (0.6)	47.1 (2.9)	**<0.0001**	23%	S
Native density	0.0	4.7 (0.5)	47.1 (2.9)	**<0.0001**	10%	S
Non-native density	0.0	6.1 (0.2)	0.0	**<0.0001**	-	D
Density – stems >10 cm dbh						
Total density	0.1 (0.1)	9.7 (0.3)	31.3 (1.0)	**<0.0001**	31%	I
Native density	0.1 (0.1)	5.5 (0.4)	31.3 (1.0)	**<0.0001**	18%	S
Non-native density	0.0	4.2 (0.2)	0.0	**<0.0001**	-	D

[1] RRD = 100 X (R mean – P mean)/(F mean–P mean); rate = F fast, I intermediate, S slow, respectively RRD 71-130%, 30-70%, -29-29%; D deflected, RRD <= -30% or >130% or F and P both zero but R values are larger.

Table 3.3: Density (mean, SE) of the ten most abundant forest tree and shrub families, in pasture (P), regrowth (R) and forest (F), and results of ANOVAs among habitat type; significant differences (p <0.05) are bolded. Density measured as log stems/ha, for stems >10 cm dbh. Number of species present in forest sites and, in cases where ANOVA p<0.10 the relative recovery distances (RRD) and recovery rates, are also shown; N = 8, 8, 5 sites in P, R, F respectively.

Family	**No. spp in F**	**P mean (SE)**	**R mean (SE)**	**F mean (SE)**	**ANOVA *p value***	**RRD[1]**	**Rate[1]**
Lauraceae	13	0.0	0.8 (0.1)	7.2 (1.3)	**0.002**	11%	S
Cunoniaceae	7	0.0	0.4 (0.1)	3.7 (0.2)	**<0.0001**	11%	S
Myrtaceae	11	0.0	1.7 (0.3)	3.1 (0.5)	**0.02**	55%	I
Proteaceae	5	0.0	1.0 (0.2)	2.6 (0.6)	**0.03**	38%	I
Grossulariaceae	2	0.0	0.0	2.2 (0.4)	**0.0002**	0%	S
Sapindaceae	2	0.0	0.2 (0.1)	1.4 (0.3)	**0.01**	14%	S
Monimiaceae	2	0.0	0.0	1.4 (0.4)	**0.02**	0%	S
Winteraceae	1	0.0	0.0	1.3 (0.3)	**0.004**	0%	S
Araucariaceae	1	0.0	0.2 (0.1)	1.3 (0.3)	**0.01**	15%	S
Celastraceae	1	0.0	0.0	1.1 (0.2)	**0.002**	0%	S

[1] RRD = 100 X (R mean – P mean)/(F mean–P mean); rate = F fast, I intermediate, S slow, respectively RRD 71-130%, 30-70%, -29-29%.

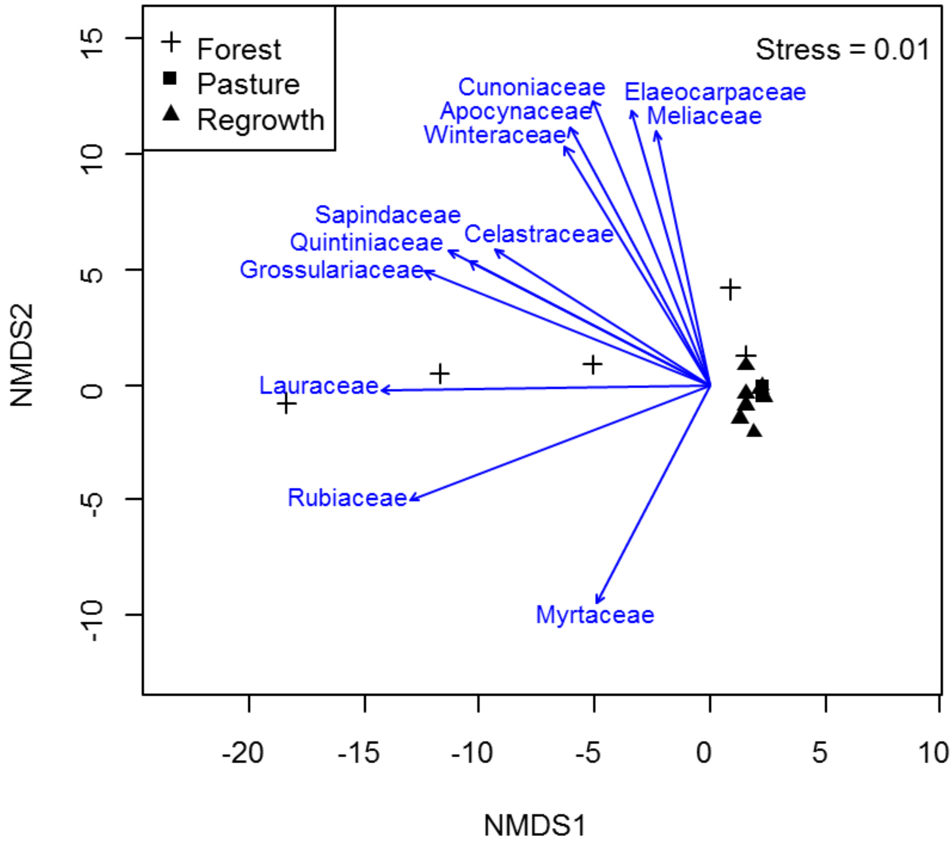

Figure 3.6: MDS ordination of 21 sites (N = 8, 8, 5 for P, R, F respectively) based on Euclidean dissimilarity of the 29 tree and shrub families (stems/ha) with non-zero values at ≥2 sites, based on densities of stems > 10 cm dbh. Arrows show biplot vectors for all intrinsic variables whose site specific values were very significantly associated (p <0.01) with sites' locations in the plot.

Table 3.4: Density (mean, SE) of the ten most abundant forest tree/shrub species in pasture (P), regrowth (R) and forest (F) and the results of ANOVAs among habitat types; significant differences ($p<0.05$) are bolded. Density measured as stems/ha (log), of stems >1 m tall (all dbh classes). In cases where ANOVA $p<0.10$, relative recovery distances (RRD) and recovery rates are shown; N = 8, 8, 5 sites in P, R, F respectively.

Species	**P mean (SE)**	**R mean (SE)**	**F mean (SE)**	**ANOVA *p value***	**RRD[1]**	**Rate[1]**
Araucariaceae						
Araucaria cunninghamii	0.00	0.20 (0.07)	3.29 (0.29)	**<0.0001**	6%	S
Cunoniaceae						
Callicoma serratifolia	0.0	0.0	3.0 (0.4)	**<0.0001**	0%	S
Ceratopetalum apetalum	0.00	0.13 (0.05)	5.83 (0.24)	**<0.0001**	2%	S
Ericaceae						
Trochocarpa laurina	0.0	0.0	3.3 (0.3)	**<0.0001**	0%	S
Lauraceae						
Cryptocarya glaucescens	0.0	0.0	3.6 (0.4)	**<0.0001**	0%	S
Cryptocarya meissneriana	0.0	0.0	4.2 (0.2)	**<0.0001**	0%	S
Myrtaceae						
Acmena smithii	0.00	0.00	3.05 (0.47)	**0.0002**	0%	S
Proteaceae						
Orites excelsa	0.00	0.00	3.05 (0.20)	**<0.0001**	0%	S
Rubiaceae						
Atractocarpus benthamianus	0.0	0.0	2.8 (0.6)	**0.003**	0%	S
Sapindaceae						
Sarcopteryx stipata	0.0	0.0	3.7 (0.2)	**<0.0001**	0%	S

[1] RRD = 100 X (R mean – P mean)/(F mean–P mean); rate = F fast, I intermediate, S slow, respectively RRD 71-130%, 30-70%, -29-29%.

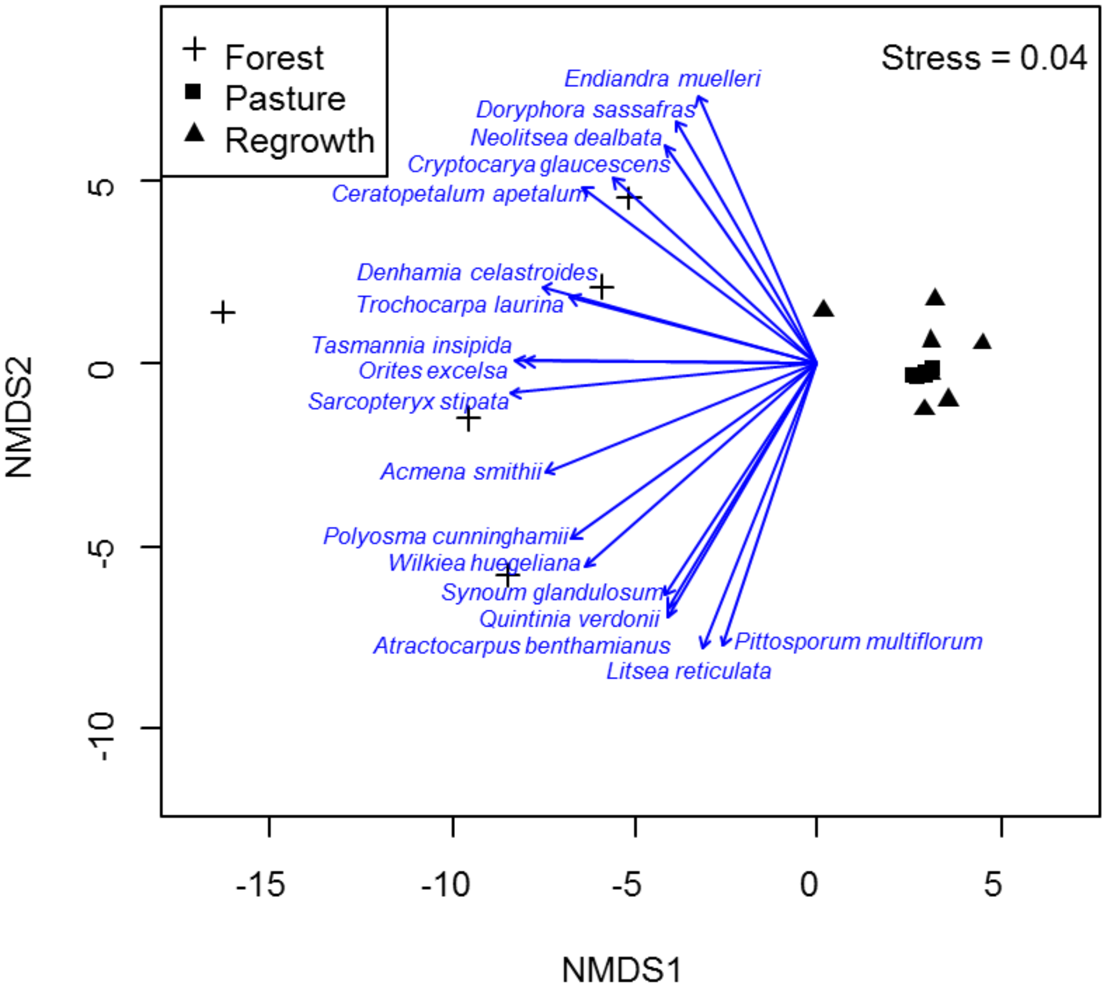

Figure 3.7: MDS ordination of 21 sites (N = 8, 8, 5 for P, R, F respectively) based on Euclidean dissimilarity of the 58 tree and shrub species (stems/ha of stems >1 m tall in all size classes) with non-zero values at ≥ 2 sites. Arrows show biplot vectors for all intrinsic variables whose site specific values were very significantly associated ($p<0.001$) with sites' locations in the plot.

3.3.3 Occurrence and floristic composition of other life forms

Across all 21 sites, 128 species of other life forms were recorded, comprising 21 ferns from 12 different families, 45 herbs from 25 families, 23 grasses from 3 families, 30 vines from 18 families and 9 epiphytes from 6 families (Appendix IV). For these life forms, 19 of 22 tested species richness and relative abundance attributes differed significantly ($p<0.05$) among the three habitats (Table 3.5).

All epiphyte species were native, and epiphytes occurred only in forest sites (Table 3.5), with two species (*Asplenium australasicum, Platycerium bifurcatum*) being recorded at more than two sites. All ferns were also native, and ferns were absent from pasture sites, and showed an intermediate recovery in regrowth coupled with fast recovery of relative abundance. *Blechnum cartilagineum*, *Pteridium esculentum* and *Adiantum silvaticum* were the most common species, all occurring at five or more sites.

For vines, forest sites contained a diverse range of native species, and regrowth showed little recovery of both species richness and relative abundance. Non-native vine species richness and relative abundance had a “deflected” pattern, due to their presence in regrowth but absence from both pasture and forest habitat (Table 3.5). Seven vine species occurred at five or more sites: the natives *Melodinus australis, Embelia Australiana*, *Petermannia cirrosa*, *Rubus moluccanus*, *Ripogonum discolor* and *Cissus hypoglauca*, and the non-native *Lonicera japonica* (*L. japonica* and *R. moluccanus* occurred only in regrowth sites, while the remaining species occurred almost entirely in forest sites).

Both grasses and herbs occurred in all three habitats, and both had an overall pattern in which pasture sites supported high overall abundance and were overwhelmingly dominated by non-native species whereas forest sites contained low overall abundance, of only native species. Regrowth sites supported an intermediate total abundance, comprising a diverse mixture of natives and non-natives (Table 3.5). Native herbs had highest species richness and relative abundance in regrowth, while non-native herbs also had highest species richness in regrowth, even though their abundance did not differ among habitats (Table 3.5). Only the native *Lomandra spicata* and the non-natives *Prunella vulgaris* and *Plantago lanceolata* occurred at five or more sites (*L. spicata* was restricted to forest and the other two were recorded in pasture and regrowth).

Native grasses (“grasses” included both grasses and sedges) were most abundant in regrowth, where they showed a “deflected” pattern of recovery of native species richness (to 866% RRD, Table 3.5). For non-native grasses, overall relative abundance showed fast recovery (greatly reduced abundance) in regrowth, although species richness remained similar to that of pasture (Table 3.5). The native sedge *Carex breviculmis,* the native grasses *Entolasia marginata*, *Microlaena stipoides* and *Oplismenus imbecillis,* and the non-native grasses *Axonopus affinis, Paspalum urvillei* and *Pennisetum clandestinum* each occurred at five or more sites.

The species composition of herbs, vines, ferns and grasses differed significantly between all testable habitat pairs, except in the case of grass species composition which was similar in regrowth and forest habitats (Table 3.6).

Table 3.5: Species richness and relative abundance values (mean, SE) for other life-forms, overall and by species origins, in pasture (P), regrowth (R) and forest (F), and the results of ANOVAs among habitat types; significant differences (p <0.05) are bolded. Species richness was measured as species/site; relative abundances as frequencies (vines, epiphytes) or percent cover (ferns, herbs, grasses), from six 10 m x 10 m quadrats per site. In cases where ANOVA p<0.10, relative recovery distances (RRD) and recovery rates are also shown; N = 8, 8, 5 sites in P, R, F respectively.

Variable	**P mean (SE)**	**R mean (SE)**	**F mean (SE)**	**ANOVA *p value***	**RRD[1]**	**Rate[1]**
Ferns						
Native species richness	0.0	3.1 (0.3)	4.4 (0.4)	**0.0009**	70%	I
Native relative abundance	0.00	1.50 (0.26)	1.70 (0.12)	0.05	88%	F
Epiphytes						
Native species richness	0.0	0.0	2.8 (0.2)	**<0.0001**	0%	S
Native relative abundance	0.00	0.00	1.13 (0.06)	**<0.0001**	0%	S
Vines						
Total species richness	0.0	2.3 (0.1)	13.0 (0.9)	**<0.0001**	18%	S
Total relative abundance	0.00	1.15 (0.05)	5.40 (0.32)	**<0.0001**	21%	S
Native species richness	0.0	1.3 (0.2)	13.0 (0.9)	**<0.0001**	10%	S
Native relative abundance	0.00	0.31 (0.04)	5.40 (0.32)	**<0.0001**	6%	S
Non-native species richness	0.0	1.0 (0.0)	0.0	**<0.0001**	-	D
Non-native relative abundance	0.00	0.83 (0.02)	0.00	**<0.0001**	-	D
Herbs						
Total species richness	3.75 (0.37)	8.00 (0.55)	2.80 (0.09)	**0.02**	-447%	D
Total relative abundance	10.1 (2.7)	4.2 (0.5)	1.5 (0.1)	0.5	-	-
Native species richness	0.9 (0.2)	4.0 (0.4)	2.8 (0.1)	**0.04**	163%	D
Native relative abundance	0.25 (0.05)	1.69 (0.20)	1.47 (0.06)	**0.03**	118%	F
Non-native species richness	2.6 (0.2)	3.8 (0.3)	0.0	**0.01**	-46%	D
Non-native relative abundance	9.6 (2.7)	2.5 (0.6)	0.0	0.4	-	-
Grasses						
Total species richness	3.6 (0.2)	5.9 (0.3)	1.2 (0.2)	**0.002**	-95%	D
Total relative abundance	79.1 (3.8)	10.9 (1.4)	0.3 (0.1)	**<0.0001**	86%	F
Native species richness	0.9 (0.1)	3.5 (0.2)	1.2 (0.2)	**0.002**	866%	D
Native relative abundance	1.3 (0.4)	6.5 (1.1)	0.3 (0.1)	0.11	-	-
Non-native species richness	2.8 (0.2)	2.3 (0.2)	0.0	**0.0005**	18%	S
Non-native relative abundance	77.8 (3.8)	4.4 (1.1)	0.0	**<0.0001**	94%	F

[1] RRD = 100 X (R mean – P mean) / (F mean–P mean); rate = F fast, I intermediate, S slow, respectively RRD 71-130%, 30-70%, -29-29%; D deflected, RRD <= -30% or >130% or F and P both zero but R values are larger.

Table 3.6: Results of pairwise ANOSIM tests of among-habitat differences in species composition between pasture (P), regrowth (R) and forest (F) for each life form present in >1 habitat. Significant differences (p<0.05) are bolded; input data were relative abundances of all recorded species; N = 8, 8, 5 sites in P, R, F respectively.

Life form	**No. of species**	**P vs F**		**R vs P**		**R vs F**	
		R	**p**	**R**	**p**	**R**	**p**
Ferns	21	-	**-**	-	**-**	0.33	**0.01**
Vines	30	-	**-**	-	**-**	0.78	**0.0008**
Herbs	45	0.39	**0.005**	0.09	**0.04**	0.25	**0.05**
Grasses	23	0.38	**0.01**	0.52	**0.0006**	-0.0007	0.37

4. ECOLOGICAL VALUES OF BIRD COMMUNITIES IN REGROWTH

4.1 INTRODUCTION

The bird species associated with pasture habitat are highly dissimilar to those associated with forest habitat (Gardner et al., 2009). Pasture is typified by bird communities of limited diversity often comprising grassland species, whereas rainforest is typified by diverse bird community assemblages featuring a number of rainforest-dependent species (Chazdon, 2014; Holl, 2007).

Global deforestation for the creation of agricultural land has led to the local extinction of many forest-associated bird species (Gardner et al., 2007) but knowledge of the capacity of reforested habitats to support the recovery of forest-dependent fauna is varied, and research on unassisted regrowth has produced largely site-specific outcomes (Chazdon et al., 2009; Bowen et al., 2007; Gardner et al., 2007). Furthermore, many studies have used broad species richness measures to determine faunal recovery (Reid et al., 2014; Bowen et al., 2007), a metric that fails to account for rainforest-dependent species which can be the slowest to recover (Chazdon et al., 2009; Bowen et al., 2007; Gardner et al., 2007).

Habitat-specialisation is increasingly being identified as a likely key factor inhibiting the colonisation of reforested sites by forest fauna (Chazdon et al., 2009; Bowen et al., 2007). Thus to understand whether reforested sites are in the process of recovering biodiversity, Freeman et al. (2015) proposed that it is useful to conduct analyses which consider the functional traits of species, such as their habitat associations.

The present chapter addresses the second of this study’s three key research questions: what are the ecological values of the bird communities in areas of post-pasture regrowth vegetation that are dominated by small-leaved privet?

It does so through quantitative analysis of data on bird species richness, diversity and community composition, collected at all sites, and analysed to reveal differences between pasture, regrowth and forest habitats. Detailed investigations of bird communities at the species-level, functional habitat group level and landscape-level, enable an assessment of the extent to which areas of regrowth have recovered those bird community attributes associated with intact remnant forest, which had been lost on its conversion to pasture.

4.2 METHODS

4.2.1 Field measurements: birds

Data on bird species richness, abundance and community composition were derived from area searches undertaken within each of the 21 study sites (Forest N = 5, Regrowth N = 8, Pasture N = 8), comprising 30 minute searches of a 0.3 ha area, repeated six times per site across a period of five months. Wherever possible surveys were conducted in a 100 x 30 m plot encompassing the same habitat in which vegetation structure and floristics were measured; however dimensions had to be modified for some regrowth sites, as for vegetation structure (see Section 3.2.1). During each survey, a single observer walked quietly through the plot in a zig-zag manner, recording all birds that were seen or heard. Each record consisted of species name, number of individuals on-site or on-patch (outside of the 0.3 ha search area), and height above ground. Careful effort was made to ensure that individuals were not recorded more than once during each visit; in cases where there were many individuals of a species, the total recorded individuals was an estimate of the number of different individuals present on site. Additional notes on behaviour, including foraging and breeding activities were also made.

Two surveys were conducted in winter (19/08/2015-27/08/2015), two in spring (12/10/2015-21/10/2015) and two in summer (10/12/2015-20/12/2015) for each study site. All surveys were conducted in the morning (between 7.00-12.00 pm in winter and 6.00-11.00 am in spring/summer). In the event of heavy rain, strong wind or excessive heat surveys were rescheduled. A rotating schedule was implemented to ensure no site was disproportionately surveyed at a more advantageous time for bird observations and surveys for any given site were conducted a minimum of one week apart.

Surveys were conducted by two different observers; one an experienced professional ornithologist (Dr Greg Clancy) and the other a student trained in visual and audial identification of birds of the region. Prior to undertaking surveys the two observers worked together to test and refine the survey methodology, and visited each individual site to agree on the specific survey area, including differentiating between site and patch area. For the first winter and spring surveys, to avoid observer bias and confirm that survey methodology was consistent, both observers visited sites together and alternated between the roles of survey leader and companion. Following each survey, Dr Clancy also offered training on local bird identification. The second winter and spring surveys were conducted by the student alone, with all records and queries subsequently quality-checked by Dr Clancy. The two summer surveys were conducted by Dr Clancy alone, with all records quality-checked by the student.

4.2.2 Data analyses

Calculation of bird variables for each site

Prior to analyses “site” records (birds within the 0.3 ha survey area) and “patch” records (birds within 20 m of a survey area boundary in the same habitat type) were separated. Data analyses used only site records of birds located at heights within 10 m above the habitat’s typical canopy height. For each identified bird species, a single ‘site’ abundance value was calculated, as the average number of individuals recorded per 30-minute survey (across 0.3 ha) from the six repeat surveys. The bird species richness (total number of species recorded across six repeat 30-minute surveys of a 0.3 ha area) was also calculated.

Thus, all “patch-only” records, together with “flyovers” (birds flying at heights >10 m above the average canopy height for each habitat type) and three single records of birds that could not be identified to species were excluded from all quantitative treatments of the data, and were treated as incidental supplementary observations.

Functional grouping of bird species by habitat preference

All bird species were classified according to three sets of taxonomic or functional criteria: family; origin (native or non-native); and functional habitat group. Functional habitat groups were based on the categories utilised in Catterall et al. (2012), with five categories used in this study. “*Rainforest-dependent* (RF): species are largely confined to, or dependent on, rainforest. *Mixed Forest* (MF): species occur mainly in a wider range of forested habitats spanning both rainforest and the more open-canopied eucalypt forests and woodlands. *Eucalypt Forest* (EF): species are typically found in eucalypt forest or woodland, and only occasionally occur in denser forest (including rainforest), or less wooded habitats. *Grassland/ Wetland* (GW): species occur mainly in grassland, wetland or water, although they may also occur within lightly-timbered open habitat, or be dependent on dense swampy vegetation; includes aerial feeding species. *Non-native* (XX): species are introduced species which have established free-living populations since European settlement” (Catterall et al. 2012). For each functional habitat group, a single ‘site’ abundance value was calculated, as described above for individual bird species.

Assessing recovery rates of bird species and communities in regrowth

Recovery rates in regrowth were assessed by comparing values of species richness and abundance; both overall and within each functional habitat group (12 variables in total) among the three habitats using ANOVAs, and by calculating their RRD values and assigning recovery categories, as described in Section 3.2.4. Additionally, similar analyses were conducted for individual species that were recorded at four or more of the 13 combined forest and pasture sites.

To investigate the overall pattern of bird community variation among sites and among habitats, multivariate MDS ordination and ANOSIM analyses were also undertaken, using the vegan package of R software, as described in Section 3.2.4, but in this case using the Bray-Curtis inter-site dissimilarity measure instead of Euclidean distance. Biplot vectors were added to ordinations, to identify habitat groups associated with differences among sites. Two sets of multivariate analysis were conducted: first, using the site-specific abundances of birds within each of the five functional habitat groups; and second, using species-specific abundances of all species present at two or more of the 21 sites.

4.3 RESULTS

A total of 70 bird species, belonging to 34 families, were recorded across all 21 sites (Appendix V). Additionally, a further 16 species were recorded as patch-only, fly-over or unidentified species (Appendix VI).

The 12 avian species richness and abundance variables tested all differed significantly ($p<0.05$) or strongly ($0.05<p<0.10$) among the three habitats (Table 4.1). Three variables showed fast or full recovery: non-native species richness and abundance had values in regrowth that were closer to those in forest than pasture, as did grassland/wetland abundance (Table 4.1; Fig. 4.1). Three variables showed intermediate recovery: rainforest-dependent species richness and abundance values were greater in regrowth than pasture but still less than forest, while grassland/wetland species richness was lower in regrowth than pasture but still greater than forest (Table 4.1; Fig. 4.1). The remaining six variables displayed a “deflected” recovery pattern: total species richness and abundance, mixed forest species richness and abundance and eucalypt forest species richness and abundance were all greatest in regrowth, with values greater in pasture than forest for the eucalypt forest variables but greater in forest than pasture for the other variables (Table 4.1; Figures 4.1 and 4.2).

The three habitats differed significantly in the composition of bird functional habitat groups (ANOSIM global R = 0.74, p = 0.0002; pasture vs forest R = 0.88, p = 0.0006; regrowth vs pasture R = 0.80, p = 0.0002; regrowth vs forest R = 0.95, p = 0.001). Ordination revealed that regrowth sites were positioned in closer proximity to forest than pasture (Fig. 4.3). Most forest and regrowth sites were moderately well clustered, while pasture sites were scattered widely, suggesting greater site to site variability in pasture. Rainforest-dependent species were strongly associated with forest, mixed forest species were predominantly associated with regrowth and both grassland/wetland species and non-native species were associated with particular pasture sites (see also Fig. 4.2).

Table 4.1: Values (mean, SE) of bird species richness and abundance in pasture (P), regrowth (R) and forest (F), for all species and by functional habitat group, and the results of ANOVAs among habitat types; significant differences (p<0.05) are bolded. In cases where the ANOVA p<0.10, the relative recovery distances (RRD) and recovery rates are also shown; N = 8, 8, 5 sites in P, R, F respectively.

Variable[1]	**P mean (SE)**	**R mean (SE)**	**F mean (SE)**	**ANOVA *p value***	**RRD**[2]	**Rate**[2]
Species richness						
Total species richness	9.00(0.70)	22.00(0.38)	16.80(1.04)	**<0.0001**	145%	D
Rainforest-dependent	0.0	2.3(0.1)	5.0(0.3)	**<0.0001**	46%	I
Mixed forest	2.3(0.3)	14.9(0.3)	10.8(0.7)	**<0.0001**	148%	D
Eucalypt forest	1.63(0.20)	2.88(0.16)	1.00(0.20)	0.06	-198%	D
Grassland/wetland	4.4(0.3)	1.9(0.2)	0.0	**0.0004**	57%	I
Non-native species	0.75(0.09)	0.12(0.04)	0.00	**0.02**	84%	F
Abundance						
Total abundance	6.25(1.04)	15.50(0.37)	9.83(0.60)	**0.01**	258%	D
Rainforest-dependent	0.0	1.7(0.2)	4.3(0.6)	**<0.0001**	40%	I
Mixed forest	1.5(0.4)	10.9(0.3)	5.2(0.2)	**<0.0001**	254%	D
Eucalypt forest	0.65(0.09)	1.90(0.14)	0.27(0.07)	**0.007**	-329%	D
Grassland/wetland	3.8(0.6)	0.9(0.1)	0.0	0.07	76%	F
Non-native species	0.40(0.06)	0.02(0.01)	0.00	**0.03**	95%	F

[1] Species richness is the total no. of species recorded across six repeat 30-minute surveys; abundance is the average no. of individuals per survey; both within 0.3 ha.

[2] RRD = 100 X (R mean – P mean)/(F mean – P mean); rate = F fast, I intermediate, S slow, respectively RRD 71-130%, 30-70%, -29-29%; D deflected, RRD <= -30% or >130%.

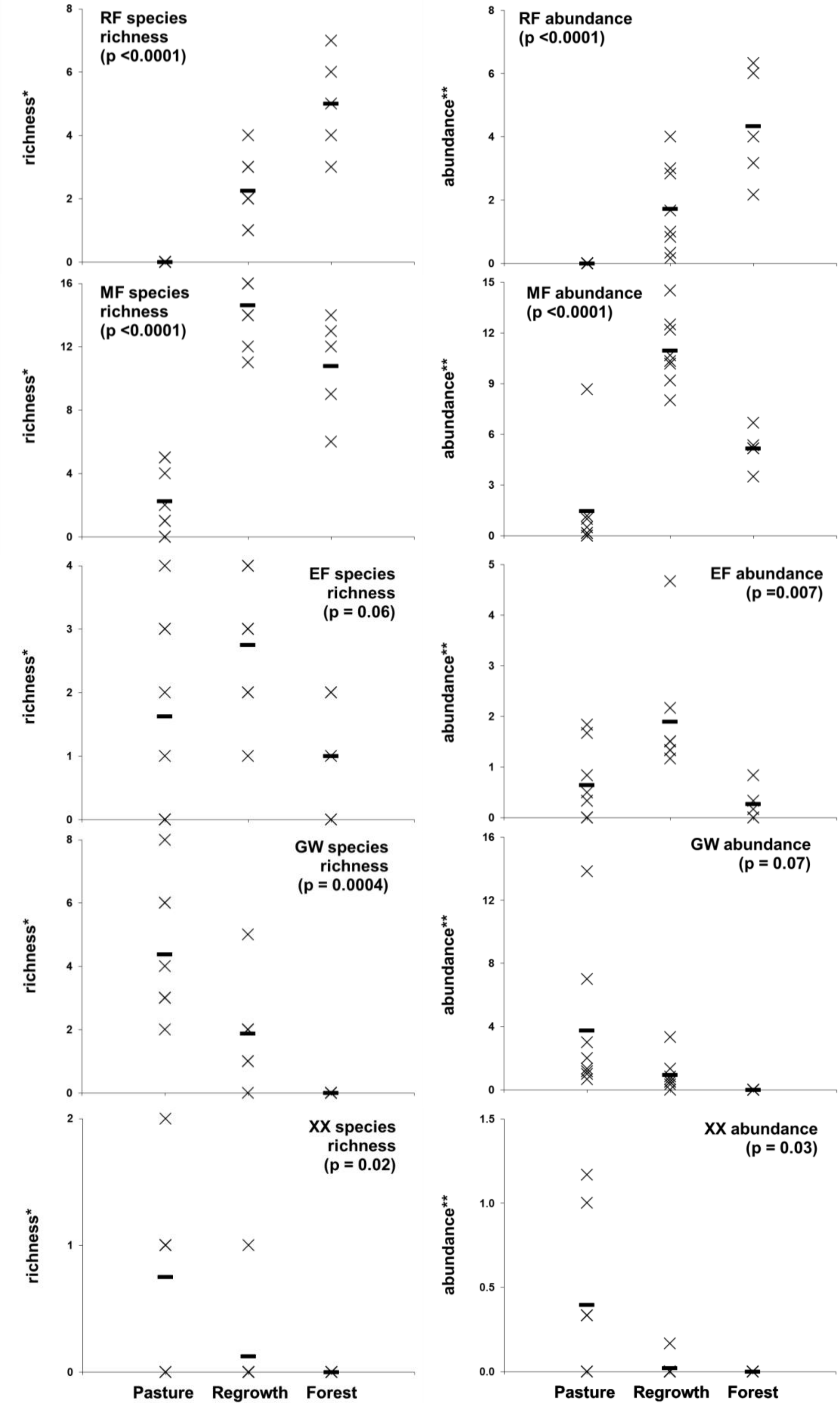

Figure 4.1: Species richness and abundance of bird functional habitat groups in pasture, regrowth and forest (N = 8, 8, 5 in P, R, F respectively). Crosses show site values; horizontal bars show habitat means; p values from ANOVAs. Species richness is total no. of species recorded across six repeat 30-minute surveys; abundance is average no. of individuals per survey; both within 0.3 ha.

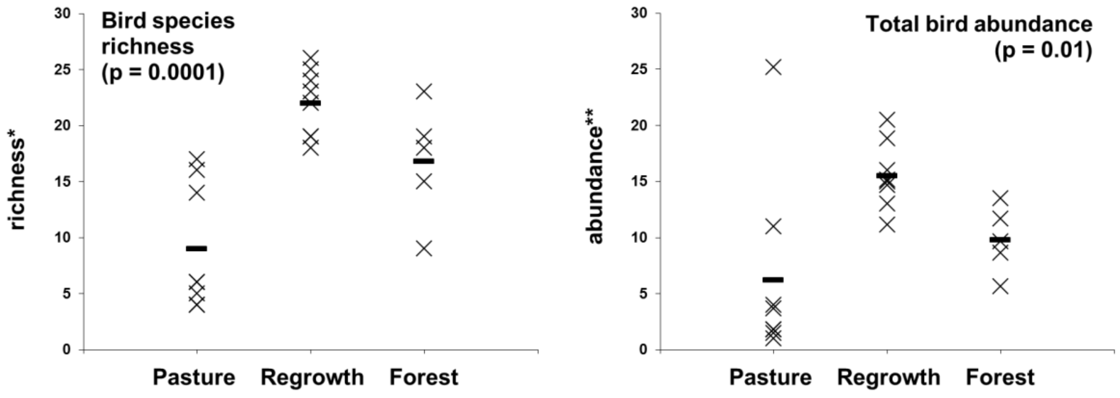

Figure 4.2: Total bird species richness and abundance in pasture, regrowth and forest (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and ANOVA p values are displayed. Species richness is the total no. of species recorded across six repeat 30-minute surveys; abundance is the average no. of individuals per survey; both within 0.3 ha.

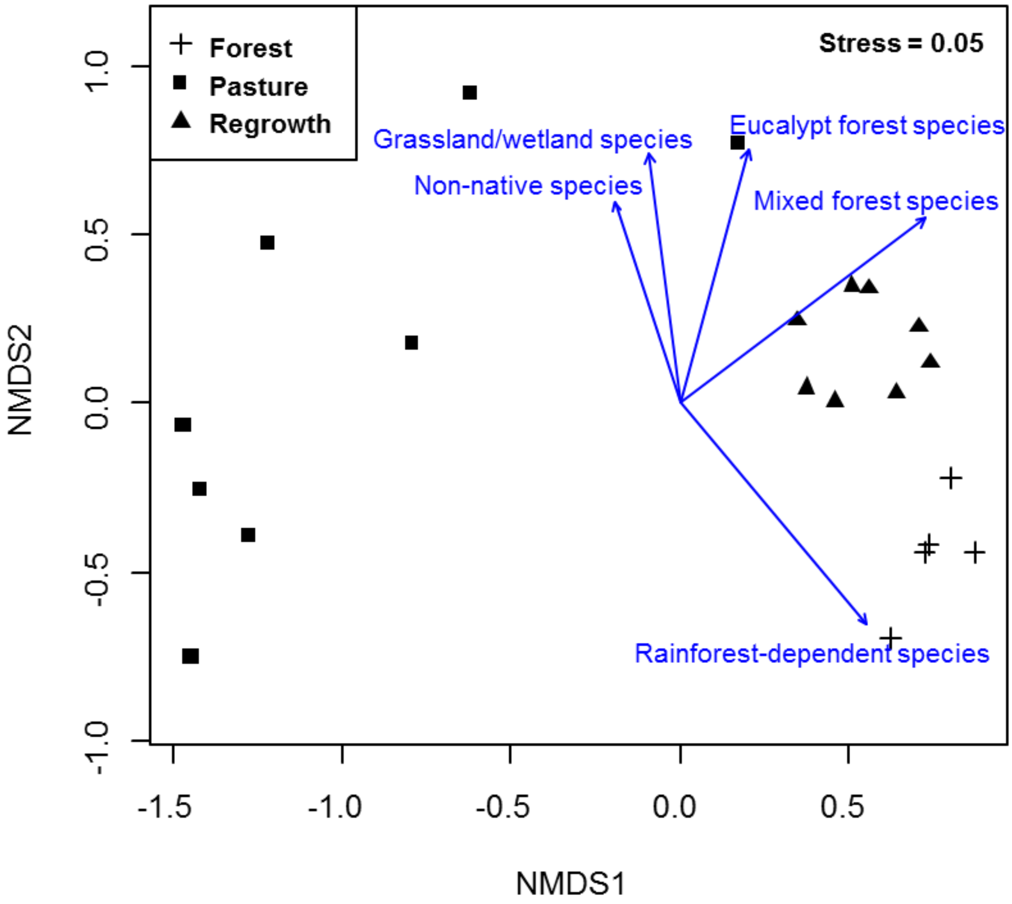

Figure 4.3: MDS ordination of 21 sites (N = 8, 8, 5 for P, R, F respectively) based on Bray-Curtis similarity of abundance of the five bird functional habitat groups. Arrows show biplot vectors for all intrinsic variables whose site specific values were significantly associated ($p<0.05$) with sites’ locations in the plot. Abundance is the average number of individuals recorded per 30-minute survey (across 0.3 ha) from the six repeat surveys.

Eleven of the 16 bird species tested differed significantly among at least two of the three habitats, with the only exceptions being the sulphur-crested cockatoo, eastern spinebill, willie wagtail, Australian magpie and welcome swallow (Table 4.2). The Australasian pipit and common starling had achieved fast or full recovery, with regrowth abundances significantly lower than pasture and both species absent from forest (Fig. 4.4). The brown gerygone and eastern whipbird had achieved intermediate recovery, with regrowth abundances greater than pasture but still lower than in forest (Fig. 4.4). The large-billed scrubwren and logrunner, both rainforest-dependent species, had achieved slow recovery, with low and zero abundance values respectively in regrowth (Fig. 4.4). The remaining five species had shown a “deflected” recovery pattern: white-browed scrubwren, brown thornbill, Lewin’s honeyeater, golden whistler and grey fantail abundances were all greatest in regrowth, with forest values greater than pasture for all species (Fig. 4.5).

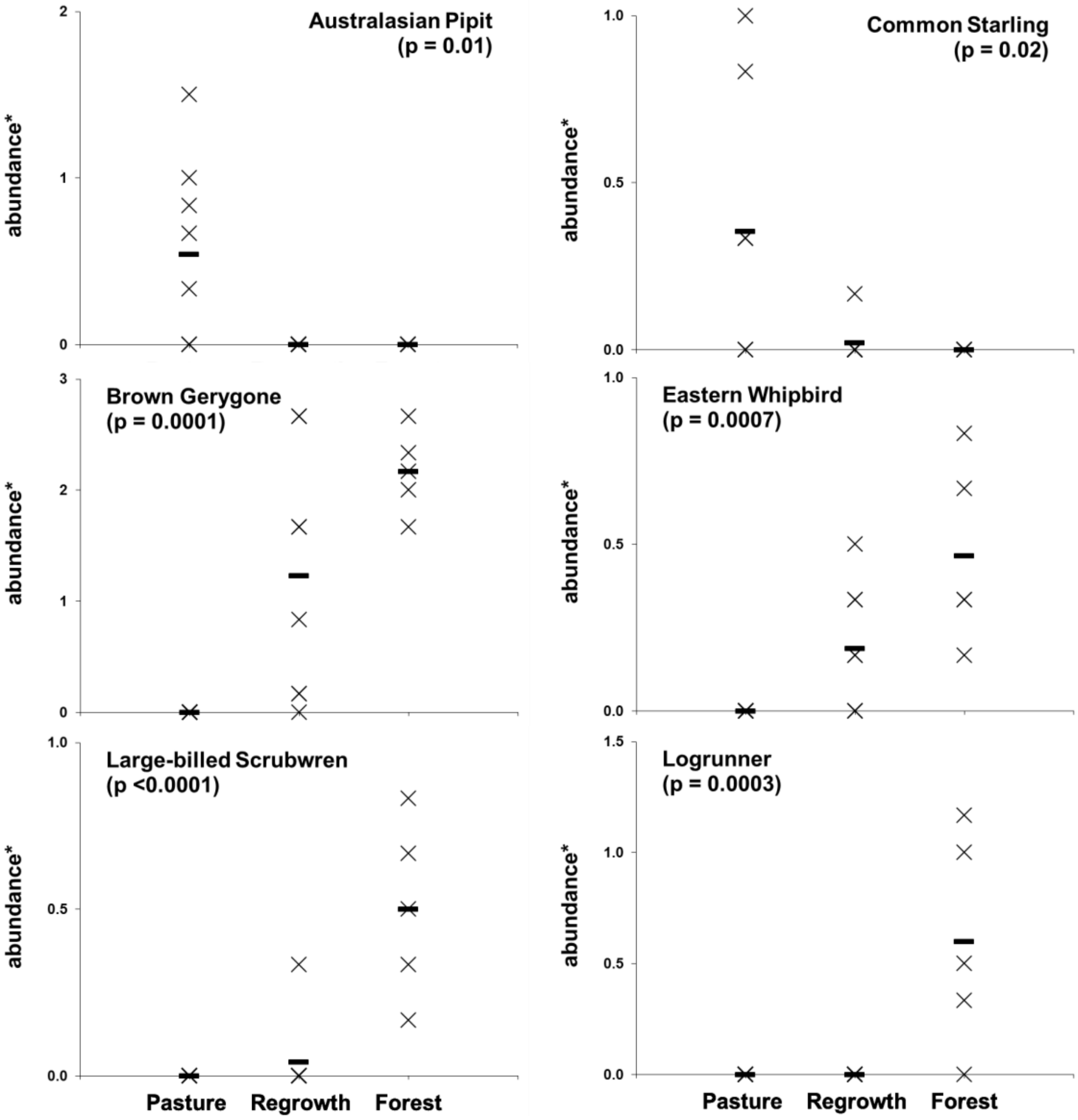

Figure 4.4: Bird species abundance in pasture, regrowth and forest (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and p values are from ANOVAs. Abundance is the average number of individuals recorded per 30-minute survey (across 0.3 ha) from the six repeat surveys.

Table 4.2: Abundance1 values (mean, SE) in pasture (P), regrowth (R) and forest (F) of bird species recorded in four or more of the 13 combined pasture and forest sites, and the results of ANOVAs among habitat types; significant differences ($p < 0.05$) are bolded. In cases where the ANOVA $p < 0.10$, the relative recovery distances (RRD) and recovery rates are also shown; N = 8, 8, 5 sites in P, R, F respectively.

Family	**Species**	**Habitat**[2]	**P mean (SE)**	**R mean (SE)**	**F mean (SE)**	**ANOVA *p value***	**RRD**[3]	**Rate**[3]
Cacatuidae	*Cacatua galerita* (sulphur-crested cockatoo)	MF	0.04(0.01)	0.25(0.05)	0.23(0.09)	0.40	-	-
Acanthizidae	*Sericornis frontalis* (white-browed scrubwren)	MF	0.00	1.31(0.03)	0.27(0.03)	**<0.0001**	485%	D
	Sericornis magnirostra (large-billed scrubwren)	RF	0.00	0.04(0.01)	0.5(0.05)	**<0.0001**	8%	S
	Gerygone mouki (brown gerygone)	RF	0.00	1.23(0.14)	2.17(0.07)	**0.0001**	57%	I
	Acanthiza pusilla (brown thornbill)	MF	0.00	1.40(0.07)	1.03(0.08)	**<0.0001**	135%	D
Meliphagidae	*Meliphaga lewinii* (Lewin's honeyeater)	MF	0.10(0.02)	0.85(0.06)	0.67(0.05)	**0.001**	131%	D
	Acanthorhynchus tenuirostris (eastern spinebill)	MF	0.10(0.03)	0.40(0.04)	0.20(0.04)	0.13	-	-
Orthonychidae	*Orthonyx temminckii* (logrunner)	RF	0.0	0.0	0.6(0.1)	**0.0003**	0%	S
Eupetidae	*Psophodes olivaceus* (eastern whipbird)	MF	0.00	0.19(0.02)	0.47(0.05)	**0.0007**	40%	I
Pachycephalidae	*Pachycephala pectoralis* (golden whistler)	MF	0.00	0.85(0.05)	0.63(0.06)	**<0.0001**	135%	D
Rhipiduridae	*Rhipidura albiscapa* (grey fantail)	MF	0.02(0.01)	1.21(0.03)	0.70(0.05)	**<0.0001**	175%	D
	Rhipidura leucophrys (willie wagtail)	GW	0.27(0.05)	0.00	0.00	0.11	-	-
Artamidae	*Cracticus tibicen* (Australian magpie)	GW	0.46(0.07)	0.35(0.06)	0.00	0.22	-	-
Motacillidae	*Anthus novaeseelandiae* (Australasian pipit)	GW	0.54(0.07)	0.00	0.00	**0.01**	100%	F
Hirundinidae	*Hirundo neoxena* (welcome swallow)	GW	1.02(0.21)	0.19(0.05)	0.00	0.18	-	-
Sturnidae	*Sturnus vulgaris* (common starling)	XX	0.35(0.05)	0.02(0.01)	0.00	**0.02**	94%	F

[1] Abundance is the average number of individuals recorded per 30-minute survey (across 0.3 ha) from the six repeat surveys.

[2] RF = rainforest-dependent species, MF = mixed forest species, GW = grassland/wetland species and XX = non-native species.

[3] RRD = 100 X (R mean – P mean)/(F mean – P mean); rate = F fast, I intermediate, S slow, respectively RRD 71-130%, 30-70%, -29-29%; D deflected, RRD <= -30% or >130%.

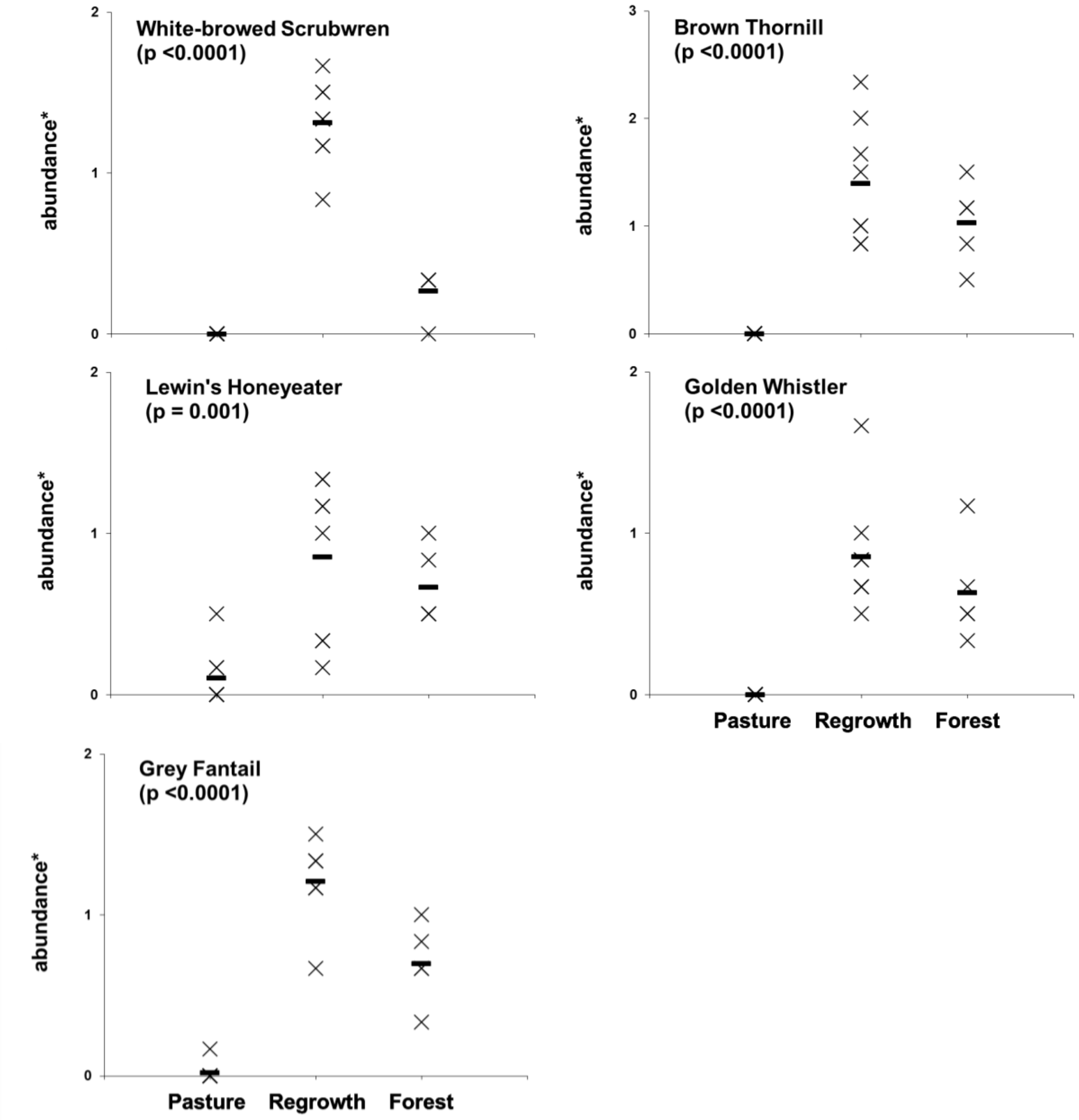

Figure 4.5: Bird species abundance in regrowth (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and p values are from ANOVAs. Abundance is the average number of individuals recorded per 30-minute survey (across 0.3 ha) from the six repeat surveys.

The community composition of bird species differed significantly among the three habitats (ANOSIM global R = 0.73, p = 0.0002; pasture vs forest R = 1, p = 0.002; regrowth vs pasture R = 0.89, p = 0.0002; regrowth vs forest R = 0.80, p = 0.0006), with ordination (Fig. 4.6) showing a pattern in which regrowth was clearly distinct from pasture, and more similar to forest, with pasture having greatest among-site variability. Of the 47 bird species analysed, 17 were significantly (p<0.05) associated with the ordination pattern. Individual rainforest-dependent species were primarily associated with forest, mixed forest species associated with regrowth and forest, and non-native and grassland/wetland species associated with specific pasture sites. This is consistent with the ANOVA results, where rainforest-dependent species were among the slowest to recover, grassland/wetland species and non-natives were mostly fast recoverers and mixed forest species largely had deflected recovery patterns, where their abundances in regrowth exceeded both pasture and forest values.

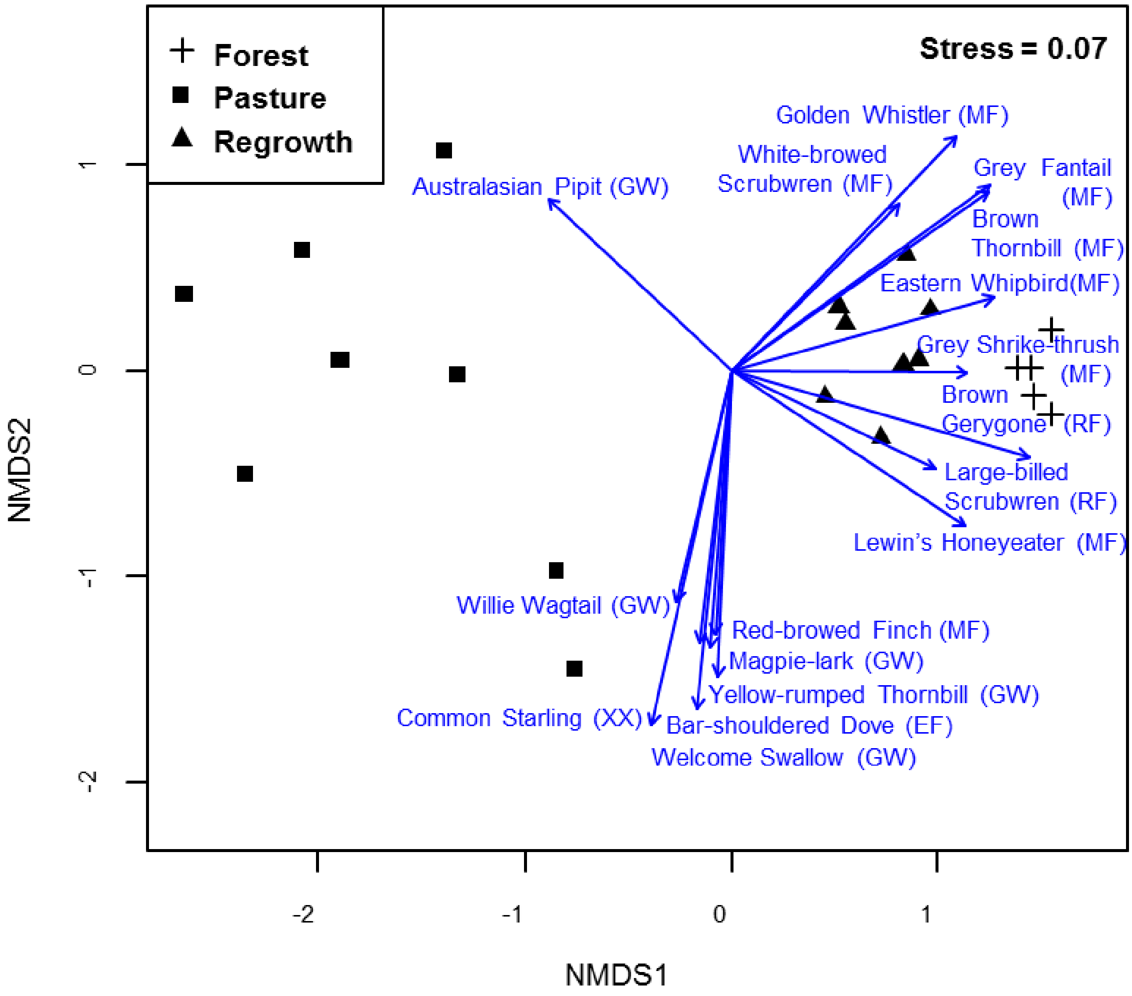

Figure 4.6: MDS ordination of 21 sites (N = 8, 8, 5 for P, R, F respectively) based on Bray-Curtis similarity of the abundance of 47 bird species with non-zero values at ≥2 sites. Arrows show biplot vectors for all intrinsic variables whose site specific values were significantly associated ($p < 0.05$) with sites' locations in the plot. Habitat functional group also shown for each species (RF = rainforest-dependent; MF = mixed forest; EF = eucalypt forest; GW = grassland/wetland; and XX = non-native species). Abundance is the average number of individuals recorded per 30-minute survey (across 0.3 ha) from the six repeat surveys.

5. POTENTIAL FOR REGENERATION

5.1 INTRODUCTION

Within tropical and subtropical rainforests vertebrate frugivores (particularly birds) are the primary seed-dispersers (Wilson et al., 1989; Howe & Smallwood, 1982) for potentially hundreds of fleshy-fruited plant species (Dennis & Westcott, 2006; Moran et al., 2004). In Australian rainforests it has been estimated that approximately 70% of rainforest plant species may rely on frugivore seed-dispersal (Wilson et al., 1989), with large frugivores identified as significant contributors to dispersal (Fritz & Purvis, 2010). Conversely, the provision of multiple food resources (particularly fleshy-fruits) in rainforests may be a key factor supporting high faunal diversity (Catterall et al, 2004; Kanowski et al, 2003), while fleshy fruits are comparatively lacking in pasture.

Native frugivore diversity and abundance have been shown to decrease along a gradient ranging from secondary forest to pasture (Gardner et al., 2009).The absence, or decreased abundance, of frugivorous seed-dispersers can result in: reductions in seed dispersal of fruiting trees (Holbrook & Loiselle, 2009), particularly large-seeded plants (Menke et al., 2012; Moran et al., 2009); reductions in seedling recruitment (Moran et al., 2004; Cordeiro & Howe, 2003); decreased plant diversity (Terborgh et al., 2008); and the inhibition of rainforest regeneration (Lehouck et al. 2009; Cordeiro & Howe 2003). Understanding the dynamics of, and capacity for, successful rainforest regeneration in the tropics and subtropics requires knowledge of the usage of regrowth areas by seed-dispersing birds (Neilan et al., 2006).

The present chapter addresses this study's final key research question: what is the potential for rainforest regeneration to occur in areas of post-pasture regrowth vegetation that are dominated by small-leaved privet?

It does so through quantitative analysis of previously-described site-based data on densities of tree and shrub species and abundances of bird species, together with additional data on seed-dispersal modes of trees and shrubs, seed-dispersal capacities of frugivorous birds, and the site-specific amounts of available fleshy fruit and grazing by domestic livestock. Analyses are structured to reveal the extent to which the areas of regrowth possess characteristics associated with the facilitation of rainforest regeneration.

5.2 METHODS

5.2.1 Data measurements

Tree and shrub dispersal modes and densities

For each site, the densities and floristic composition of woody-stemmed plants (trees and shrubs) within different stem size classes (> 1 m tall and <2.5cm dbh, 2.5-10 cm dbh and >10 cm dbh size classes) were obtained from the data described in Chapter 3. Additionally, all tree and shrub species were grouped into four seed-dispersal classes, as previously used by Shoo et al. (2016): wind dispersed; frugivore dispersed with diaspore size <10 mm; frugivore dispersed with diaspore size ≥10 mm; and "other" (see Appendix III).

Wind dispersed species comprised plants for which wind was considered to be the primary vector of seed dispersal. Frugivore-dispersed species comprised plants for which fruit-eating birds (and bats) were considered to be the primary dispersal vectors, due to the presence of some form of nutritional reward external to the seed, combined with an absence of flotation-related structures (e.g. wings). Diaspore size was defined as the fruit diameter, or seed diameter for arillate fruits and fruits easily broken apart, and measured as the smallest dimension of the seed or fruit. Other dispersal comprised plants for which wind or frugivores were not the primary vectors for seed dispersal, this included dispersal facilitated by water, gravity or explosive dispersal mechanisms.

The species-specific dispersal categories were derived from a corrected version of those presented within Kanowski et al. (2010), which included a large proportion of the trees and shrubs identified in the present study. For 29 additional species recorded only in the present study, relevant information about the nature and morphology of their fruits was obtained from descriptions and identification guides relating to rainforest flora of the Australian subtropics (Harden et al., 2006; Floyd, 2008; Harden et al., 2014); the last-named source also provided information about their consumption by frugivorous birds. This was further supplemented by information extracted from the Queensland Herbarium database and various online sources.

Abundance, species, and functional composition of frugivorous birds

For seed-dispersing birds, site-specific species' abundances were obtained from the data described in Chapter 4. Frugivorous species were distinguished from non-frugivores using the criteria and accompanying list of subtropical frugivorous species presented by Moran et al. (2004). Accordingly, fruit-eating species that typically crush seeds and which therefore would disperse few viable seeds (and are functionally granivores rather than frugivores) were not considered to be "seed dispersers"; these mainly comprised some pigeons and the Australian brush turkey, which both have muscular seed-grinding gizzards, and all parrots. Additionally, there were three species in the present study that are here considered to be seed-dispersing frugivores but which were not recorded or classified by Moran et al. (2004): their "seed-disperser" status was recognised by first screening all recorded species for primary diet, as reported in a range of reference sources, and then seeking further information sources for the smaller remaining number of potential frugivores.

The final list of seed-dispersing bird species recorded in the present study was then divided into two subcategories of functional seed-dispersal potential, based on information from Moran et al. (2004) and Moran and Catterall (2014): Seed-disperser A and B. "Seed-disperser A", considered to have greatest potential as seed-dispersers, is defined as having a gape size >1 cm (as an indicator of the potential size of fruit which could be ingested) AND a diet that includes fruit as more than a minor component AND reported to consume >50 fleshy-fruited plant species. "Seed-disperser B" comprises species that meet some but not all of the criteria of the "A" category, having either smaller (<1 cm) gapes OR a diet that includes fruit as a minor or partial component OR reported to consume ≤50 fleshy-fruited plant species. The remaining species were classified as "Unlikely dispersers", defined as eating little or no fruit OR regularly eating fruit but known to crush seeds.

Analyses of seed-dispersing birds used the abundance measurements (average number of individuals across six 30-minute searches of 0.3 ha) of each species classified in the Seed-disperser A or B categories, together with their assigned categories of seed-dispersal potential, and also their previously-defined habitat preference categories (rainforest RF, mixed forest MF, eucalypt forest EF, grassland/wetland GW and non-native XX, see Section 4.2.2).

Fruiting levels

Levels of fruiting were recorded on six occasions at each site, corresponding with the timing of bird surveys (see Section 4.2.1: two surveys conducted in winter from 19/08/2015-27/08/2015, two in spring from 12/10/2015-21/10/2015 and two in summer from 10/12/2015-20/12/2015). Two measures of fruiting were recorded: number of trees bearing fruit; and average number of fruits per fruiting tree. Number of fruiting trees was scored using: no trees in fruit = 0; 1-2 trees in fruit = 1, 3-10 trees = 2; 11-100 trees = 3; and >100 trees = 4. Average number of fruits per fruiting tree, recorded as a rough visual estimate, was scored using: no fruits or fruiting trees = 0; ≤10 fruits per fruiting tree = 1; 11-50 fruits = 2; 51-100 fruits = 3; and >100 fruits = 4. These two measures were summed to give an overall level of fruiting score (0-8) for each visit, with scores then averaged across the six surveys to obtain a site-level value.

Grazing pressure

Grazing pressure was measured on a single occasion corresponding with the vegetation structure survey (see Section 3.2.1: surveys conducted from 14/04/2015-26/06/2015). Four different measures of grazing pressure were recorded: presence or absence of cow tracks; abundance of cow pats; evidence of browsing; and abundance of cows. Cow tracks were measured as a presence/absence score: absent = 0; and present = 1. Cow pats were scored using: no cow pats (faeces) = 0; 1-10 cow pats = 1; and >10 cow pats = 2. Evidence of browsing was scoring using: no evidence = 0; minor evidence (signs of browsing on <5 trees on edges of plot) = 1; moderate evidence (browsing on 5-10 trees at edges and within the plot) = 3; and major evidence (browsing on >10 trees within the plot) = 4. Abundance of cows was scored using: no cows = 0; 1-10 cows = 1; and >10 cows = 3. The four measures were summed to obtain an overall grazing pressure site-level value (0-10).

5.2.2 Data analyses

Tree and shrub species and dispersal modes

To provide an initial indication of floristic recovery in regrowth towards a rainforest-like state, the ten most abundant tree species that were recorded as stems >1 m tall across the five remnant rainforest sites were identified from the floristic data (Table 3.4, Section 3.3.2), and their dispersal modes (four possibilities – see Methods) and frequencies in regrowth, as particular stem size classes or as seedlings, were assessed.

A second indication of development of species composition in regrowth was obtained by identifying all tree and shrub species occurring across the eight regrowth sites as stems >1 m tall, and using the stem densities (log stems/ha) to construct comparable rank-abundance histograms for each of three stem size classes (<2.5 cm dbh, 2.5-10 cm dbh, >10 cm dbh), reflecting different stem ages (years since first recruitment).

To assess the extent to which recruitment of tree and shrub species into regrowth potentially depended on dispersal by birds, the percentages of species belonging to each of the four seed dispersal classes were calculated: for all stems >1 m tall in each of forest and regrowth; and in regrowth also for separate stem dbh classes <2.5 cm, 2.5-10 cm, and >10 cm dbh and for seedlings.

Abundance and composition of seed-dispersing birds

To assess the extent to which regrowth sites were visited by seed-dispersing birds, comparisons among forest, regrowth and pasture (N = 8, 8, 5 respectively) were again conducted, using ANOVAs and RRD values (see Section 3.2.4), for: species richness and total abundance within the “Seed-disperser A” and “Seed-disperser B” functional groups; and abundances of all individual species within these groups that were present in at four or more of the 21 sites.

To assess the among-site and among-habitat differences in species composition of seed-dispersers, multivariate MDS ordination and ANOSIM analyses were again undertaken (see Section 3.2.4, but in this case using Bray-Curtis rather than Euclidean inter-site dissimilarities), using all site-specific abundances of individual seed-disperser species that were present in at least two of the 21 sites. Biplot vectors were added to the ordination to identify species associated with differences among sites.

Levels of fruiting and livestock grazing pressure

To assess the potential of regrowth to provide significant fruit resources for seed dispersers, and the potential suppression of seedling recruitment by livestock, the site-specific indices for fruiting level and for grazing pressure were separately compared among pasture, regrowth and forest (N = 8, 8, 5 respectively), using single-factor ANOVAs and the calculation of RRD as described previously (see Section 3.2.4).

5.3 RESULTS

5.3.1 Abundance and dispersal modes of trees and shrubs in regrowth

Only two of the ten most abundant tree and shrub species in forest were recorded in regrowth as stems >1 m tall: coachwood (*Ceratopetalum apetalum*) and hoop pine (*Araucaria cunninghamii*). However, five of the ten species were present in the seedling layer of one or more regrowth sites (Table 5.1). In total, seedlings of these forest-associated species were recorded at four different regrowth sites.

Table 5.1: Species of forest-associated trees and shrubs recorded in regrowth sites; number of regrowth sites with stems >1 m tall, and size classes represented, and number of sites with seedlings. Dispersal mode of each species is also displayed: (F<10) = frugivore-dispersed with diaspore <10 mm; (F≥10) = frugivore-dispersed with diaspore ≥10 mm; and (W) = wind-dispersed.

Family	Species	Dispersal mode	dbh classes of stems >1m (No. of R sites)	Seedlings (No. R sites)
Araucariaceae	*Araucaria cunninghamii*	Wind	>10cm (1)	Y (1)
Cunoniaceae	*Callicoma serratifolia*	F <10	-	Y (1)
	Ceratopetalum apetalum	Wind	<2.5cm and 2.5-10cm (1)	Y (1)
Ericaceae	*Trochocarpa laurina*	F ≥10	-	Y (1)
Lauraceae	*Cryptocarya glaucescens*	F ≥10	-	Y (3)

In total, 15 tree and shrub species were recorded as stems >1 m tall across all regrowth sites. Small-leaved privet (*Ligustrum sinense*) was clearly numerically dominant across all size classes (Fig. 5.1). Common hawthorn (*Crataegus monogyna*) was the only other non-native species recorded and was represented across all size classes. For all size classes the white tea-tree (*Kunzea ericoides*) was the second most abundant species, however after this the order of rank abundance varied among stem size classes. Some species, such as Sydney blue gum (*Eucalyptus saligna*), were present only in the largest stem size class, while other species, such as rice flower (*Ozothamnus diosmifolius*), had greater abundance in the small to medium size classes.

There was no overall tendency for native trees and shrubs to be present at higher densities in the form of smaller individuals (i.e., mainly younger and more recently recruited) than for larger stem diameters (Fig. 5.1). However, 28 tree and shrub seedlings were recorded in the regrowth sites (23 being native, of which 9 were also represented as stems > 1 m tall), suggesting some potential for increased representation of native forest species among younger recruits. Of the 28 tree and shrub species recorded as seedlings in regrowth, 11 species were also present in the forest seedling layer and 18 were represented as stems >1 m tall (see Appendix VII).

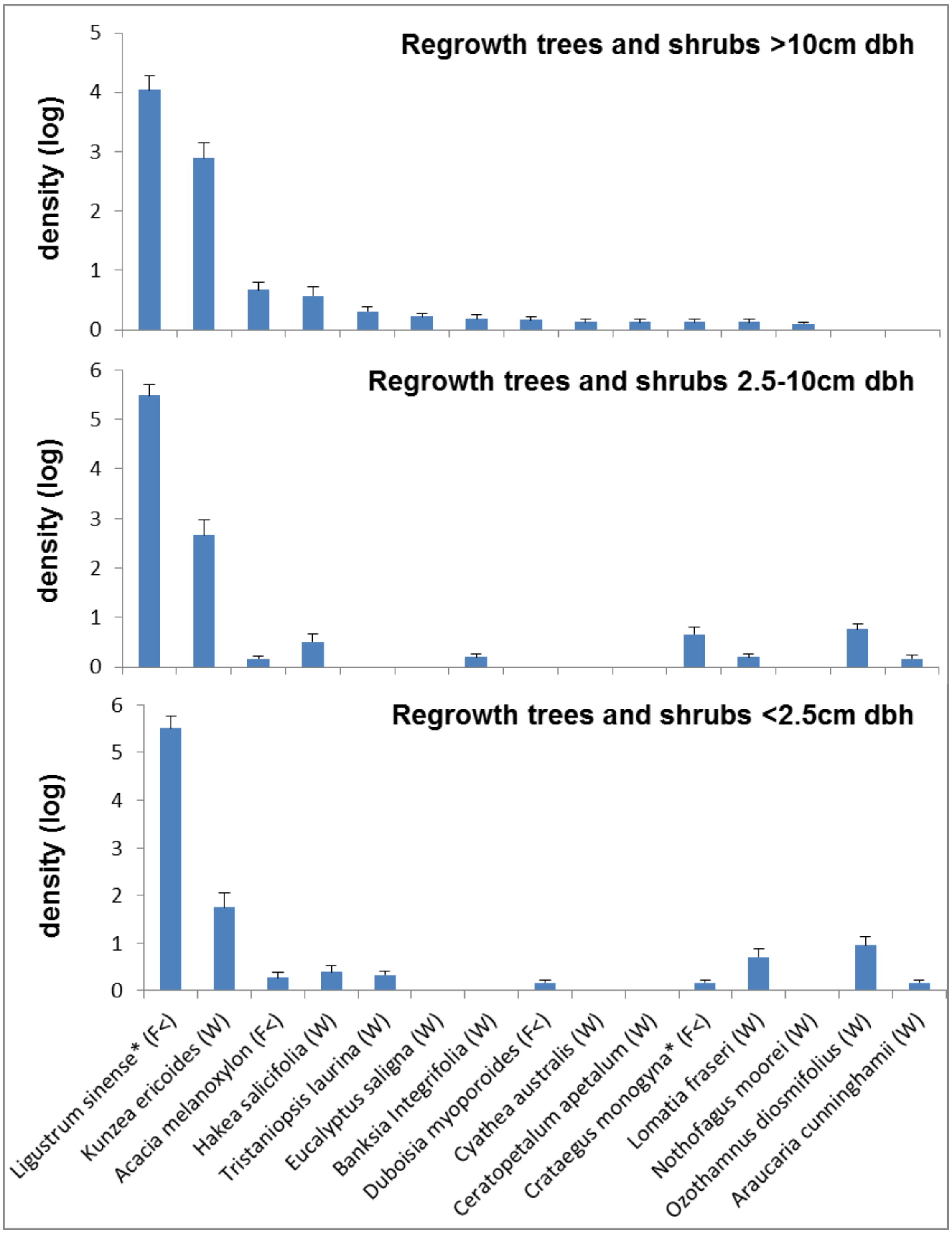

Figure 5.1: Density (log stems/ha) of the 15 tree and shrub species recorded in regrowth as stems >1 m tall, shown as a breakdown by size class. Species arranged in relation to their abundance (from highest to lowest) in the largest stem size class (>10 cm dbh). Non-native species are identified with an asterisk and the dispersal mode of each species is also displayed: (F<) = frugivore-dispersed with diaspore <10 mm and (W) = wind-dispersed.

Among the 15 tree and shrub species recruited as stems >1 m tall in regrowth sites, 73% were wind-dispersed, whereas frugivore-dispersed species accounted for only 27% of stems; a similar pattern was also seen within each of the three stem diameter classes (Fig. 5.1, Table 5.2). However, this pattern was reversed in forest, where 69% of species represented as stems >1 m tall were frugivore-dispersed and wind dispersal accounted for only 27% (Table 5.2). Furthermore, frugivore-dispersed species with large diaspores (≥10 mm) were notably absent among stems >1 m tall in regrowth, but accounted for 28% in forest (Table 5.2). However, among seedlings in regrowth the pattern was more similar to that seen in forest, with 69% of species being frugivore-dispersed (including some with large diaspores) and only 28% being wind-dispersed (Table 5.2). Additionally, frugivore-dispersal tended to be more common within the smaller (younger) than larger (older) stem diameter classes of regrowth trees and shrubs (Table 5.2).

Table 5.2: Dispersal mode of tree and shrub species within forest and regrowth: total number of species within each habitat and percent of species within each dispersal class (wind dispersed; frugivore dispersed with diaspore <10 mm; frugivore dispersed with diaspore ≥10 mm; other dispersal mode) are shown. Forest values are for all stems >1 m tall; regrowth values are shown for all stems >1 m tall, individual stem size classes and seedlings.

Habitat and stem size class	**% Frug (<10mm)**	**% Frug (≥10mm)**	**% Wind**	**% Other**	**Total No. spp.**
Forest: all stems >1 m tall	41	28	27	4	71
Regrowth: all stems >1 m tall	27	0	73	0	15
Regrowth: stems >10 cm dbh	31	0	69	0	13
Regrowth: stems 2.5-10 cm dbh	33	0	67	0	9
Regrowth: stems <2.5 cm dbh	40	0	60	0	10
Regrowth: seedlings	57	14	29	0	28

5.3.2 Level of fruiting

Level of fruiting had a “deflected” recovery (>130%) pattern in regrowth (RRD = 725%, Rate = D), with regrowth values (mean = 2.9, SE = 0.2) much greater than forest values (mean = 0.4, SE = 0.1), while there was no fruiting in pasture (mean = 0.0) (Fig. 5.2).

Within forest sites, a number of different fleshy-fruited tree and shrub species were recorded in-fruit, however these were not identified to the species level and fruiting was only recorded at three sites (60%) during the summer surveys (10/12/2015-

20/12/2015). Within regrowth sites, levels of fruiting were predominantly associated with the presence of small-leaved privet and often involved dense fruit set, however there was also evidence of a small amount of fruiting of white tea-tree at a single site. Fruiting was recorded in regrowth during both winter (19/08/2015-27/08/2015) and summer (10/12/2015-20/12/2015) surveys and 100% of sites had evidence of fruiting (Appendix VIII).

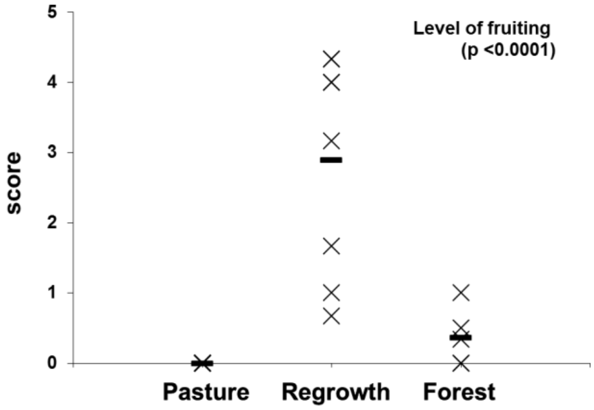

Figure 5.2: Level of fruiting in pasture (P), regrowth (R) and forest (F) habitat (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and the ANOVA p value is displayed. Level of fruiting measured as the combined score given for number of trees bearing fruit and average number of fruits per fruiting tree, averaged across the six surveys (see Section 5.2.1).

5.3.3 Use of regrowth by seed-dispersing birds

Of the 70 bird species recorded from all 21 sites (see Chapter 4, Section 4.3), 20 species were classified as potential seed dispersers: six species were classified as "Seed-disperser A" (best potential seed-dispersers) and 14 species as "Seed-disperser B" (other potential seed dispersers). Additionally, two further seed-disperser species were recorded as "patch-only" observations; see Appendix IX for the list of species and their attributes.

In total, eight of the 11 seed-dispersing bird variables tested differed significantly ($p<0.05$) among the three habitats (Table 5.3). Seven of these displayed a "deflected" recovery pattern in which regrowth contained the most dispersers: species richness and abundance of Seed-disperser A and B categories, and abundance of the Lewin's honeyeater (*Meliphaga lewinii*), satin bowerbird (*Ptilonorhynchus violaceus*) and silvereye (*Zosterops lateralis*). "Seed-disperser A" group and component species had greater values in forest than pasture, while "seed-disperser B" group and most component species had greater values in pasture than forest (Table 5.3; Fig. 5.3).

Four of the six species classified as “Seed-disperser A” were excluded from species-level analyses as they were recorded at less than four sites: figbird (*Sphecotheres vieilloti*), green catbird (*Ailuroedus crassirostris*), olive-backed oriole (*Oriolus sagittatus*) and pied currawong (*Strepera graculina*).

The three habitats differed significantly in the species composition of seed-dispersing birds (global R = 0.65, p = 0.0002; pasture vs forest R = 0.80, p = 0.0004; regrowth vs pasture R = 0.64, p = 0.0004; regrowth vs forest R = 0.59, p = 0.0008). Ordination revealed that regrowth sites were distinct from both forest and pasture (Fig. 5.4). Two of the 12 variables included in analyses were significantly (p <0.05) associated with this pattern, with the Lewin’s honeyeater associated with regrowth and the common starling associated with pasture sites.

Table 5.3: Species richness and abundance values (mean, SE) of seed-dispersing birds in pasture (P), regrowth (R) and forest (F); presented for “Seed-disperser A” and “Seed-disperser B” groups, and individual species from these groups with non-zero values at four or more sites (N = 8, 8, 5 sites in P, R, F respectively). Functional habitat group of each species, number of sites in which the species/group was recorded and the results of ANOVAs among habitat types are shown; significant differences (p <0.05) are bolded. In cases where ANOVA p<0.10, relative recovery distances (RRD) and recovery rates are shown.

Variable[1]	**FHG**[2]	**No. sites**	**P mean (SE)**	**R mean (SE)**	**F mean (SE)**	***p* *value***	**RRD**[3]	**Rate**[3]
Species richness								
Seed-disperser A	-	16	0.4 (0.1)	2.0 (0.1)	1.8 (0.2)	**0.001**	143%	D
Seed-disperser B	-	18	2.6 (0.1)	3.8 (0.2)	0.6 (0.2)	**0.004**	190%	D
Abundance								
Seed-disperser A	-	16	0.10 (0.02)	1.71 (0.20)	0.97 (0.09)	**0.02**	185%	D
Lewin’s honeyeater	MF	16	0.10 (0.02)	0.85 (0.06)	0.67 (0.05)	**0.001**	132%	D
Satin bowerbird	MF	5	0.00	0.31 (0.04)	0.00	**0.01**	-	D
Seed-disperser B	-	18	0.73 (0.09)	1.73 (0.14)	0.10 (0.03)	**0.03**	-159%	D
Yellow-faced honeyeater	EF	6	0.04 (0.01)	0.21 (0.04)	0.00	0.20	-	-
Australian magpie	GW	12	0.5 (0.1)	0.4 (0.1)	0.0	0.22	-	-
Torresian crow	MF	8	0.23 (0.05)	0.23 (0.05)	0.00	0.47	-	-
Silvereye	MF	9	0.02 (0.01)	1.06 (0.04)	0.00	**0.02**	-52%	D
Common starling	XX	6	0.35 (0.05)	0.02 (0.01)	0.00	**0.02**	94%	F

[1] Species richness is the total no. of species recorded across six repeat 30-minute surveys; abundance is the average no. of individuals per survey; both within 0.3 ha.
[2] MF = mixed forest species, EF = eucalypt forest species, GW = grassland/wetland species and XX = non-native species.
[3] RRD = 100 X (R mean – P mean)/(F mean – P mean); rate = F fast (RRD 71-130%), D deflected (RRD ≤ -30% or >130% or F and P both zero but R values are larger).

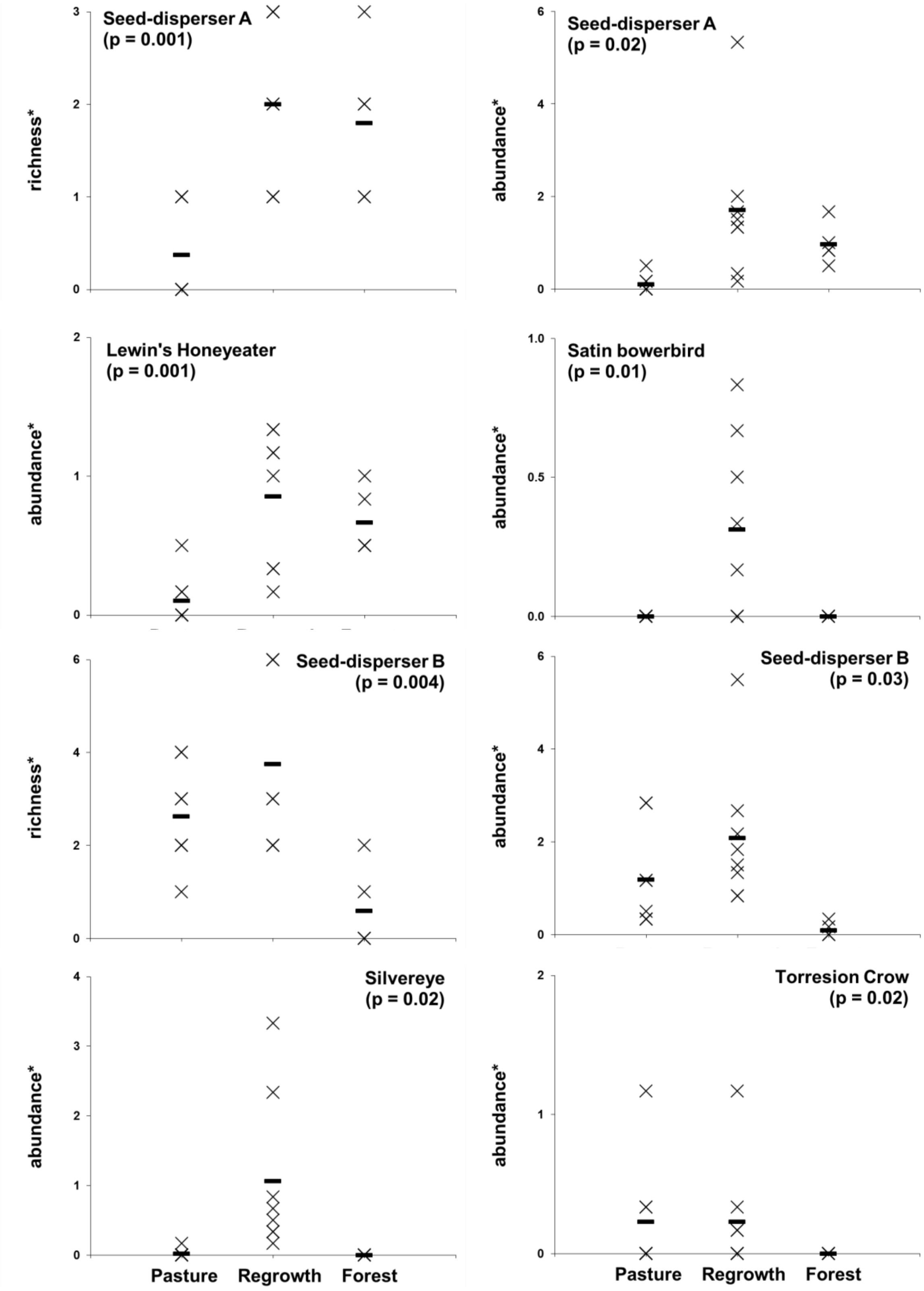

Figure 5.3: Species richness and abundance of Seed-disperser A and B functional groups and abundance of individual species from these groups with non-zero values at four or more of the 13 combined pasture and forest sites. Crosses show site values; horizontal bars show habitat means; p values are from ANOVAs. Species richness is the total no. of species recorded across six repeat 30-minute surveys; abundance is the average no. of individuals per survey; both within 0.3 ha. (N = 8, 8, 5 in P, R, F respectively).

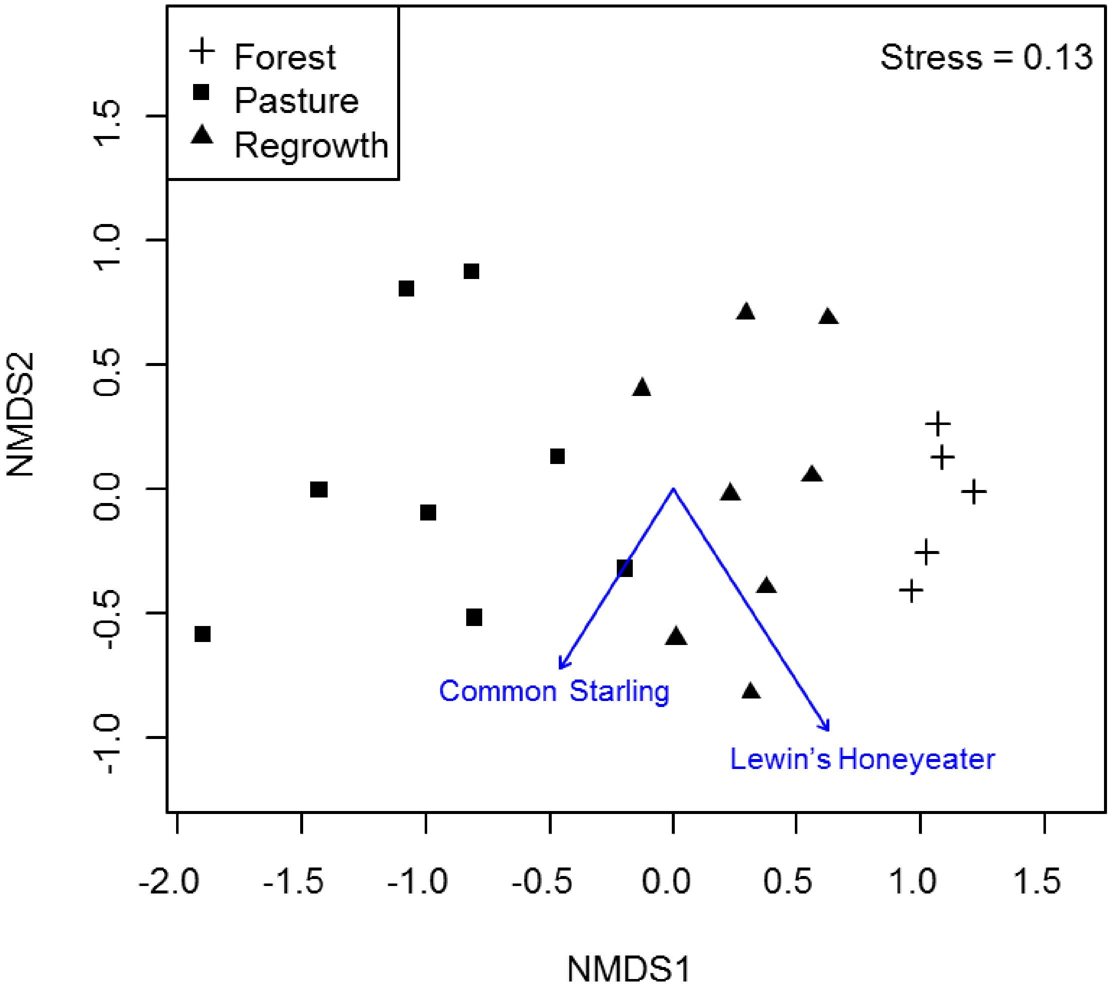

Figure 5.4: MDS ordination of 21 sites (N = 8, 8, 5 for P, R, F respectively) based on Bray-Curtis similarity of the abundance of 12 seed-dispersing bird species with non-zero values at ≥2 sites. Arrows show biplot vectors for all intrinsic variables whose site specific values were significantly associated ($p < 0.05$) with sites' locations in the plot. Abundance is the average number of individuals recorded per 30-minute survey (across 0.3 ha) from the six repeat surveys.

5.3.4 Grazing pressure

Grazing pressure had shown slow to no recovery (-29%-29%) in regrowth (RRD = 3%, Rate = S), with regrowth values (mean = 3.8, SE = 0.4) similar to pasture values (mean = 3.9, SE = 0.4), while there was no grazing pressure in forest (mean = 0.0) (Fig. 5.5). There was considerable variation in grazing pressure among both regrowth and pasture sites, with values in each habitat ranging from no pressure to high pressure and grazing metrics varying from site to site (Appendix X).

While two of the regrowth sites with low grazing pressure had the highest number of seedlings recorded, and two sites with high pressure had the lowest numbers of seedlings, a clear pattern did not occur across all sites (Appendix X). It should be noted here that grazing in some regrowth sites was undertaken on a rotational basis, while grazing pressure surveys were undertaken on a single occasion per site.

Figure 5.5: Grazing pressure in pasture (P), regrowth (R) and forest (F) habitat (N = 8, 8, 5 in P, R, F respectively). Crosses show site values, horizontal bars show habitat means and the ANOVA p value is displayed. Grazing pressure measured as the combined scores give for presence or absence of cow tracks, abundance of cow pats (faeces), evidence of browsing and abundance of cows (see Section 5.2.1).

6. DISCUSSION

6.1 ECOLOGICAL VALUES OF POST-PASTURE REGROWTH DOMINATED BY SMALL-LEAVED PRIVET

This study demonstrated that privet-dominated regrowth in riparian areas of the eastern Dorrigo Plateau had recovered many of the ecological values that were associated with areas in which old-growth rainforest habitats had been conserved, but which were absent from areas of pasture. This was most strongly the case for vegetation structure and bird species richness, abundance and community composition. However, the regrowth has recovered much less of the floristic diversity and community composition characteristic of rainforest. Nevertheless, many ecological factors that could facilitate rainforest regeneration were recorded within regrowth sites.

Below I discuss separately the findings for bird communities, vegetation structure, and floristic composition and diversity.

Bird communities

In this study regrowth sites had achieved fast to full recovery of bird communities, with overall community composition showing greater similarity to forest than pasture habitat and bird species richness and abundance greatest in regrowth sites. This finding is reflected in a number of Australian and global studies that show bird abundance in regrowth sites can rapidly reach parity with, or overshoot, values of reference forest sites (Catterall et al., 2012; Renner et al., 2006; Waltert et al., 2005; Blake & Loiselle, 2001; Arnold et al., 1999). For example, Catterall et al. (2012) found that total bird species richness and abundance within biodiversity plantings in oldfields of the Australian Wet Tropics had achieved similar levels to forest sites five years after planting.

However, the high bird abundance in regrowth may have been due to an influx of generalist species, particularly those that can dominate edge and matrix habitats (Catterall et al., 1997). This study found that the total species richness and abundance of mixed forest bird species, which can utilise a variety of open and closed forests and woodlands, had overshot forest values, with the white-browed scrubwren, brown thornbill, Lewin’s honeyeater and eastern spinebill all showing “deflected” recovery in regrowth. Habitat generalists such as the Lewin’s honeyeater respond positively to decreasing forest cover (Moran & Catterall, 2014) and are more likely to be present in cleared areas and regrowth (Catterall et al., 1998). The patterns of bird abundance recorded in this study are consistent with previous studies, whereby increases in total abundance may be offset by declines in species richness and abundance of forest-specialist birds (Catterall et al., 2012; Catterall et al., 1997), which can decrease significantly as surrounding forest cover decreases (Moran & Catterall, 2014).

In this study, the species richness and abundance of rainforest-dependent birds had only achieved intermediate recovery in regrowth. Of the three rainforest-dependent birds analysed at the species level; the brown gerygone had achieved intermediate recovery, the large-billed scrubwren had only achieved slow recovery and the logrunner was not recorded in regrowth sites. Scientific literature indicates that large-billed scrubwrens rely on the protection and foraging resources provided in the dense understorey of rainforest while logrunners feed upon insects inhabiting the thick leaf litter (Catterall et al, 2004). The loss of the specialised rainforest structural features that can attract rainforest-dependent bird species (Laliberte et al., 2010; Lindenmayer, 2009) may have contributed to the reduced abundances of rainforest-dependent birds recorded in regrowth. However, given the limited number of forest sites in this study (N = 5), and the sparse distribution of some rainforest-dependent bird species, caution needs to be applied when drawing any inferences about the relative abundances of these species in different habitats. Conversely, this study found that grassland/wetland bird species and non-native species had decreased in abundance in regrowth. As grassland/ wetland birds favour open habitat, the decreased abundance of these species, which were recorded in proximal pasture habitat, is indicative of an increase in the overall structural complexity of regrowth vegetation.

Vegetation structure

Vegetation structure within regrowth sites of the eastern Dorrigo Plateau had achieved intermediate recovery, with overall vegetation structure composition falling roughly mid-way between pasture and forest sites. Some structural attributes, such as canopy cover, densities of small (<20 cm dbh) and medium (20-50 cm dbh) tree and shrub stems, tree basal area and grass cover, demonstrated fast or full recovery in regrowth. These results are consistent with unassisted regrowth sites in the Australian Wet Tropics which had achieved canopy cover levels and tree and shrub stem densities comparable to forest reference values within 40 years (Shoo et al., 2016).

Conversely, canopy height, coarse woody debris and density of large (>50 cm dbh) tree and shrub stems were all significantly lower in regrowth than forest. Life-form diversity was also comparatively depauperate in regrowth, with an absence of epiphytes, small palms and strap-leaf herbs, low abundance of tree ferns and epiphyte ferns and low site representation of robust vines. These structural attributes are all associated with intact rainforest (Kanowski et al., 2005) and the under-representation of these attributes is consistent with previous studies which show that young regrowth forest can have less complexity, fewer specialised life-forms, less large woody debris and more open canopy than mature forest (Kanowski et al, 2003). Many of these attributes are also associated with the provision of habitat for rainforest-specialist fauna (Kanowski et al., 2010) and the scarcity of structures such as robust vines may have been one of the factors contributing to the limited recovery of rainforest-dependent birds in regrowth.

Floristic composition and diversity

Regrowth sites in this study had achieved slow to no recovery of floristics, with overall community composition of trees and shrubs in regrowth more similar to pasture than forest. Native tree and shrub species richness and density were greater in regrowth than pasture but still considerably lower than in forest; while non-native species richness and abundance, across all size classes, were greatest in regrowth. Furthermore, eight of the ten most abundant families, and all ten of the most abundant species, in forest had achieved little to no recovery in regrowth, with three families not recorded at all and one family and five species recorded only in the seedling layer. These results are consistent with findings from other Australian studies which indicate that important tree species associated with intact rainforest are often missing from regrowth (Goosem et al., 2016; Shoo et al., 2016). Similarly, a study in restoration sites on abandoned farmland in Costa Rica reported a notable lack of large-seeded mature forest tree species in soil seed banks of both unassisted regrowth sites and biodiversity plantings in the early stages of development (Reid et al., 2015).

In this study, epiphytes were not recorded within regrowth and recovery of native vines was also slow, though the abundance of non-native vines was high. These findings are again consistent with other Australian studies, which indicate many rainforest resources are slow to mature (Goosem et al., 2016; Catterall et al., 2004) and the recovery of vine and epiphyte compositions characteristic of intact rainforest may require timeframes exceeding six decades (Shoo et al., 2016). Conversely, fern species richness had recovered well in regrowth and species richness and abundance of native grasses and herbs had overshot recovery, with values in regrowth significantly greater than forest and pasture values. Previous research confirms that ferns, herbs and grasses can form dense ground cover in regenerating post-pasture forest; however these often decline as regrowth matures (Catterall, 2016).

6.2 FACTORS THAT MAY FACILITATE OR INHIBIT RAINFOREST REGENERATION IN PRIVET-DOMINATED REGROWTH

Facilitation of rainforest regeneration

Many of the factors than can facilitate rainforest regeneration were present in areas of privet-dominated regrowth along the eastern Dorrigo Plateau. The abundance of grasses in regrowth had significantly decreased from pasture-like levels. The dense grass cover typical of pasture habitat, whether native or non-native, can supress the recruitment of tree seedlings (Catterall, 2016). However, as pioneer tree and shrub species colonise oldfields they can supress grass cover (Shoo & Catterall, 2013; Holl, 2007) and potentially create space for native tree and shrub seedlings to germinate (Elgar et al., 2014). Conversely, canopy cover in regrowth had increased significantly, achieving comparable levels to forest reference sites. Increases in canopy cover can help to ameliorate weather extremes and establish microclimate conditions conducive to the growth of rainforest plants (Catterall et al., 2014).

This study also recorded a variety of fauna-attracting features in regrowth, such as high densities of small and medium stemmed trees and high levels of fruiting. Previous studies have shown that the availability of perches in the landscape can lead to increases in the abundance of forest-associated birds (Lindenmayer, 2009). The increase in the availability of perches, provided by the recovery of small and medium tree stem densities, may explain the intermediate recovery of rainforest-associated birds in regrowth. Furthermore, the high levels of fruiting recorded in regrowth, which were largely attributed to privet, represent abundant food resources for frugivorous species. The availability of fruit resources can be a key driver of seed-fall patterns (Garcia et al., 2010; Herrera & Garcia, 2010), and the food and habitat resources provided by non-native plants can attract frugivorous birds which may facilitate native rainforest plant seed dispersal (Elgar et al., 2014; Kanowski et al., 2008; Neilan et al., 2006; Zahawi & Augspurger, 2006). The dense fruit-set of small-leaved privet is likely to attract frugivorous seed-dispersing birds, and this study noted bird foraging behaviour within privet-dominated regrowth on several occasions. Therefore, the high species richness and abundance of seed-dispersing birds in regrowth may have been a function of the availability of abundant fruit resources.

There was also evidence to suggest that the initial stages of rainforest regeneration were underway in the post-pasture regrowth of eastern Dorrigo Plateau. Despite forest tree and shrub species being under-represented as stems >1 m tall, five of the ten most abundant forest species were recorded in the regrowth seedling layer. Three of these species, *Callicoma serratifolia*, *Trochocarpa laurina* and *Cryptocarya glaucescens*, rely on animal-mediated seed dispersal, and the latter two species have large (≥10 mm) diaspores. Furthermore, 20 of the total 28 seedling species recorded in regrowth sites are frugivore-dispersed (four with diaspores ≥10 mm). These seedlings included three species from the Lauraceae family and one species from the Sapindaceae family, which are known to be common components in the diets of major and mixed-diet frugivores respectively (Moran & Catterall, 2010). Seedling abundance was not analysed in this study as data comprised presence records, however observations indicated that seedlings of native rainforest trees were only present in low numbers in regrowth sites.

This study demonstrated that rainforest tree and shrub species were being recruited into the seedling layer of regrowth sites, and seed-dispersing birds appeared to be facilitating the dispersal of plants with large-diaspores, an important component of rainforest tree and shrub assemblages (Menke et al., 2012; Moran et al., 2009; Kitamura et al., 2002). However, despite the high percentage of frugivore-dispersed native plant species in the seedling layer, and greater proportions of these in the smaller stem classes, the majority of these species did not appear to be surviving long enough to recruit into the larger stem classes. Therefore, given the apparent potential for regeneration in regrowth, it would appear that other factors were inhibiting the recovery of floristic diversity and composition characteristic of intact rainforest.

Inhibition of rainforest regeneration

Factors that potentially could inhibit the regeneration of rainforest trees and shrubs within the privet-dominated regrowth include ecological processes that would either: (1) restrict the dispersal of seeds from remnant rainforest into the regrowth; or (2) reduce the likelihood of germination of dispersed seeds; or (3) reduce the survival or growth of germinated seedlings. This study did not directly investigate many of the processes involved in either seed dispersal or the germination of seeds and survival of seedlings, but it did provide data relevant to some of these possibilities, as discussed below.

Restrictions to seed dispersal could occur even when some seed-dispersing birds are abundant in regrowth (as found in the present study). This would be the case if the rainforest-dependent frugivorous bird species that are most effective at dispersing large rainforest seeds are under-represented in regrowth. Moran and Catterall (2014; see also Moran et al., 2009) identified four subtropical Australian bird species that were both important dispersers of rainforest seeds (including larger seeds) and mostly restricted to intact rainforest: the wompoo, superb and rose-crowned fruit-doves and the green catbird. In the present study, these species were rare or absent (respectively recorded at 0, 0, 0 and 2 of the five forest sites and absent from regrowth and pasture, with the rose-crowned fruit-dove heard off-transect at one forest site). Since the Dorrigo region is at high elevation, and relatively close to the southern geographical range limit of these species, it is possible that they are relatively less common than in other parts of the subtropics. For example, Howe et al. (1981) noted that the rose-crowned fruit-dove was rare in the extensive conserved rainforest area of Dorrigo National Park, and absent from small isolated remnant patches in the region. These species' overall scarcity, at least during the time of this study, has prevented any analysis of their relative abundances in regrowth versus rainforest. Nevertheless, such a regional scarcity would limit the dispersal capacity of many rainforest tree and shrub species.

Restricted dispersal of seeds from rainforest to regrowth would also occur if the seed-dispersing birds that were common in regrowth did not frequently move between the regrowth and areas of remnant rainforest. Of the seven seed-disperser species recorded in this study at four or more study sites, only the Lewin's honeyeater was recorded in both forest and regrowth, being very common in both habitats. Lewin's honeyeaters consume a wide variety of rainforest fruit species (Moran & Catterall, 2014). They could therefore facilitate dispersal of rainforest seeds into regrowth, and it is likely that this species contributed significantly to the observed recruitment. However, such seed dispersal would only occur if individual birds moved sufficiently regularly between intact forest and regrowth. This study's regrowth sites were located between approximately 35-700 m from the nearest substantial remnant rainforest patch (>5 ha), and Lewin's honeyeaters typically tend to be sedentary rather than mobile (Birdlife

Australia, 2017), and so would be most likely to disperse seeds into regrowth located adjacent or close to remnant rainforest, rather than more distant sites.

Germination of rainforest seeds after they have been imported by birds into the Dorrigo privet regrowth may potentially be limited by the shade cast by the privet canopy. However, investigation of this possibility was beyond the scope of the present study. More generally, germination of the seeds of rainforest trees, especially pioneer species, may be limited in shady conditions, but may be triggered by sudden light, such as occurs following tree-fall in the forest (Chazdon, 2008). Similarly, in camphor laurel regrowth in the Big Scrub, killing the overstorey camphor trees resulted in a flush of germination of rainforest pioneer species from the seed bank which had accumulated as a result of visits by seed-dispersing birds (Kanowski et al., 2008; Woodford, 2000). This study found that seeds of some rainforest tree species were being dispersed into regrowth, and successfully germinating, however the absence of data on soil-stored seeds and seedling abundance prohibited analyses of dispersal and germination rates. Therefore, consideration of potential limitations to seed dispersal and germination are of relevance to the privet-dominated regrowth of the eastern Dorrigo Plateau.

Either high mortality or limited growth of recently germinated tree and shrub seedlings may occur as a consequence of grazing pressure, from either domestic livestock or wild herbivorous mammals (Catterall, 2016). Even grazing impacts of low-intensity may help to maintain a non-forest state, preventing regrowth from establishing in the first place (Gunaratne et al., 2010), and more generally, herbivory-induced reductions in plant growth and reproduction can favour the dominance of one or two plant species, even when grazing pressure is low (Myers & Bazely, 2003; Crawley, 1983). On the Dorrigo Plateau, the present study focused on previously-established woody regrowth, so it seems that at some time in the past the privet seedlings must have escaped from grazing pressure for long enough to establish the high observed stem densities. However, this study found a high current level of livestock grazing in the privet regrowth sites, similar to that in pasture (and being zero in forest). Evidence of grazing included not only cowpats and tracks, but also visible signs of browsing the foliage of well-established trees, indicating that the small-leaved privet is palatable to livestock. It is probable that any newly recruited seedlings of rainforest trees and shrubs also faced a high risk of being grazed by livestock, and this may have been a strong contributor to the low numbers of recorded native stems above 1 m in height in the regrowth sites, compared with their observed better representation as seedlings. Given the possibility that rainforest tree and shrub seedlings were being recruited into regrowth but failing to grow into larger stem classes, the potential impacts of high grazing pressure on seedling survival and development are worthy of further investigation in the study region.

Additionally, high mortality or limited growth of recently germinated rainforest tree and shrub seedlings could result from competitive suppression of rainforest seedlings by the privet overstorey. High privet stem densities suggest that germinated seedlings could face strong below-ground competition for water or nutrients, and the dense privet canopy reduces the available light. Elgar et al. (2014) discussed the potential for similar processes, suggestive of competitive interactions, in relation to regrowth in the Australian tropics dominated by the non-native wild tobacco (*Solanum mauritianum*), but that study did not test for evidence of their importance. Conversely, the development of a shady understorey in regrowth patches dominated by non-native trees could also aid the establishment of some native seedlings by creating suitable micro-environments for species whose seedlings normally grow in moist, sheltered locations on the forest floor (Catterall, 2014). Further experimental investigations would be needed to reveal whether competition from small-leaved privet in these regrowth patches is limiting the survival or growth of rainforest tree seedlings on the eastern Dorrigo Plateau.

6.3 MANAGEMENT IMPLICATIONS

Regrowth dominated by non-native small-leaved privet on the eastern Dorrigo Plateau has developed a complex habitat structure and resources that support a high diversity and abundance of bird species. In addition to the provision of habitat for bird species, regrowth may support recovering populations of a number of other faunal groups. For example, forest-associated reptiles, beetles and mites may benefit from the habitat provided by the increasing canopy cover and tree basal area, while the latter two taxa may also utilise the refugia created by increasing levels of leaf litter and woody debris (Kanowski et al., 2010), albeit coarse woody debris was still comparatively lacking. This regrowth also has the potential to stabilise stream banks, reduce erosion in catchments, improve soil quality and contribute significantly to the sequestration of atmospheric carbon (Dwyer et al., 2010; Feldpausch et al., 2004; Silver et al., 2000). If small-leaved privet were removed from the regrowth, a large part of the physical structure and many of the resources would also disappear, with the consequence that many of these ecological values and ecosystem functions would be lost; with native seed-dispersing birds likely to be particularly negatively impacted by reduced availability of fruit.

Environmental managers and field ecologists are increasingly debating the merits of retaining heavily modified landscapes, including those dominated by non-native species, and managing these areas as “novel ecosystems” (Truitt et al., 2015; Hobbs et al., 2009; 2006; Choi et al., 2008; Hobbs & Cramer, 2008; Seastedt et al., 2008). Novel ecosystems comprise persistent compositions, or relative abundances, of species that are occurring for the first time, as a result of anthropogenic activities, and are atypical to the area in which they are occurring (Hobbs et al., 2006). Novel ecosystems can arise as a result of land clearance, conversion of land from one use to another and the introduction of new species to areas outside their natural distribution (Truitt et al., 2015; Hobbs et al., 2006). Post-pasture regrowth dominated by small-leaved privet has many characteristics of a novel ecosystem: high abundances of a non-native species occurring in a mixed community together with native species in areas where the original vegetation has been cleared and the land has been converted to another use. In such situations, efforts to remove non-native pioneer species worldwide have often proved extremely labour-intensive and a burden on resources, and success has often been limited (Hobbs et al., 2006; Ewel & Putz, 2004).

In general terms, land managers face a dilemma when deciding how to treat a novel ecosystem of woody post-pasture regrowth dominated by a non-native tree or shrub. The options that need to be considered are: attempted removal of the non-native species, tolerating it, or adopting some kind of management aimed at inducing or accelerating a transition towards greater dominance by native species (Catterall, 2016). Such decisions also require consideration of a range of socio-political factors, such as whether the community preference is to maintain the current ecosystem, restore historic conditions or create something else (Truitt et al., 2015). A key consideration to help facilitate informed discussion is whether or not the ecosystem contains ecological values worth retaining or if the creation of an alternative future state would provide greater environmental benefits (Choi et al., 2008).

Forest regrowth in Puerto Rico provides one example of an ecologically valuable novel ecosystem. The rainforests of Puerto Rico were heavily fragmented following conversion to agricultural land in the early 20th century, however following the abandonment of agricultural activities unassisted regrowth began to establish widely and, by the late 20th century, came to represent a high proportion of total forest cover (Lugo & Helmer, 2004). The development of these emerging forests was largely facilitated by non-native pioneer tree species, with two of the eight most abundant species of non-natives origin: Malabar plum (*Syzygium jambos*), a fleshy-fruited tree from Southeast Asia commonly planted as an ornamental; and African tuliptree (*Spathodea campanulata*) a flowering tree native to the African tropics (Lugo & Helmer, 2004). Following 60-80 years of growth, the floristic diversity of these forests was still comparatively simple, comprising a novel mix of native and non-native plant species and fewer endemics than intact forest, however species turn-over rates were high and native plant species richness had increased over time as the dominance of non-native species decreased (Lugo & Helmer, 2004). Overall, these forests were found to possess ecological value as catalysts for native tree recruitment and as major contributors to total forest cover in Puerto Rico (Lugo & Helmer, 2004). There are clear parallels between the novel forests of Puerto Rico and privet-dominated regrowth of eastern Dorrigo: in both scenarios a non-native fleshy-fruited plant of Asian origin is a key pioneer and while recovery of vegetation structure is progressing native floristic recovery is lagging. Furthermore, in both ecosystems, knowledge gaps exist in relation to the possible factors that may be inhibiting more complete rainforest regeneration.

Once the ecological factors that are facilitating or inhibiting the process of secondary succession in a novel ecosystem of woody post-pasture regrowth are understood, it becomes more feasible to accelerate natural regeneration by implementing management measures to mitigate the inhibitory factors (Elgar et al., 2014), while also taking care to retain both the facilitative factors and ecological values such as wildlife habitat (Gosper et al., 2005; Zavaleta et al., 2001). Management interventions aimed at accelerating reforestation may be necessary (Erskine et al., 2007) even when non-native pioneer species, such as wild tobacco and camphor laurel, possess characteristics that can make them beneficial to rainforest regeneration (Catterall, 2016). For example, in other parts of subtropical Australia, management interventions to selectively remove non-native camphor laurel trees have increased native recruitment while maintaining habitat available to native fauna species (Kanowski et al., 2008; Parkes et al 2012).

An important factor underlying the success of these management interventions is the presence of sufficient native rainforest tree species as both seedlings and soil-stored seeds (Paul et al., 2012; Elgar et al., 2014), whose germination or growth can then be stimulated by the increased light penetration following canopy removal. In situations where consumption of seedlings by grazing livestock is limiting recruitment of native trees, livestock exclusion is an important candidate for management action. For example, previous Australian studies have shown that recruitment of understorey shrubs into post-pasture landscapes (Onans & Parsons, 1980), and native tree and shrub species diversity in Jarrah woodlands (Petit & Froend, 2001), has increased following livestock removal.

On the eastern Dorrigo Plateau, informal conversations with landholders revealed that management responses to the presence of small-leaved privet had been varied; some land managers had participated in control activities while others had allowed the species to persist on their properties. However, prior to this study, the potential ecological values of privet-dominated regrowth in the region had not been assessed. This study indicates that greater conservation benefits may be derived from the retention of small-leaved privet than would likely be achieved from its removal. Nevertheless, it is also clear that the recruitment of native rainforest trees (especially beyond the seedling stage) is limited, and this could be attributed to a number of factors, related to seed dispersal, grazing pressure, and competition. Gaining a clearer understanding of the importance and operation of these factors will be critical to the effective management of post-pasture regrowth. This could be achieved through a combination of experimental management interventions and monitoring. For example, trial habitat manipulations could consist of selective removal of small-leaved privet, with and without the removal of livestock grazing, together with research to assess the outcomes.

Privet-dominated regrowth on the eastern Dorrigo Plateau is clearly contributing to biodiversity in the region and facilitating the initial recruitment of native rainforest tree species into oldfields (at least into the seedling layer). If restoration practitioners are willing to trade-off slower recovery, privet-dominated regrowth could be utilised to contribute to reforestation over large spatial scales, especially if recovery is accelerated through the identification and mitigation of factors currently inhibiting rainforest regeneration.

APPENDIX I

Table I: Description of the 21 study sites comprising their habitat type, location, elevation, slope, waterway type and distance, and surrounding land-use type and distance. Latitude, longitude and elevation were recorded at the 0 m mark of transect 1 at each site. Waterway type and surrounding land-use were recorded as a general description based on observation.

Site name	Habitat type	Latitude	Longitude	Elevation (m)	Slope[1]	Waterway type	Water distance[2]	Surrounding land use	Land-use distance[3]	Distance to forest[4]
P1	Pasture	S30˚13.24.1”	E152˚53.40.0”	549	2	creek	5	rail track	1	150 m
P2	Pasture	S30˚16.208”	E152˚51.823”	517	2	creek	5	road	2	525 m
P3	Pasture	S30˚16.255”	E152˚51.886”	535	2	creek	5	road	1	450 m
P4	Pasture	S30˚16.485”	E152˚51.919”	529	1	dam	3	road	1	690 m
P5	Pasture	S30˚17’22.2”	E152˚53’46.7”	551	3	creek	5	road	1	270 m
P6	Pasture	S30˚18.238”	E152˚52.969”	556	2	creek	4	road	1	300 m
P7	Pasture	S30˚18’11.2”	E152˚52’57.70”	545	2	creek	5	road	1	390 m
P8	Pasture	S30˚17.45.25”	E152˚52.49.16”	554	2	creek	4	road	1	640 m
R1	Regrowth	S30˚13.317”	E152˚53.545”	527	1	creek	2	pasture	4	185 m
R2	Regrowth	S30˚16.127”	E152˚51.811”	520	1	creek	1	pasture	1	450 m
R3	Regrowth	S30˚16.288”	E152˚51.762”	523	1	creek	1	pasture	1	650 m
R4	Regrowth	S30˚16.436”	E152˚51.891”	514	1	creek	1	pasture	1	700 m
R5	Regrowth	S30˚17’00.7”	E152˚53’40.1”	534	2	creek	1	forest	2	35 m
R6	Regrowth	S30˚18.380”	E152˚52.966”	543	1	creek	2	pasture	1	90 m
R7	Regrowth	S30˚18’12.9”	E152˚53’00.7”	532	2	creek	1	house	1	400 m
R8	Regrowth	S30˚17.40.8”	E152˚52.59.9”	542	1	creek	1	pasture	1	660 m
F1	Forest	S30˚15.59.1”	E152˚54.07.1”	634	2	creek	1	forest	1	0 m
F2	Forest	S30˚16.17.5”	E152˚54.13.7”	650	4	dry gully	1	forest	1	0 m
F3	Forest	S30˚12.48.0”	E152˚51.56.4”	567	3	dry gully	2	forest	1	0 m
F4	Forest	S30˚13.43.4”	E152˚53.59.6”	561	3	dry gully	1	forest	1	0 m
F5	Forest	S30˚13.39.8”	E152˚55.15.6”	599	3	creek	2	road	2	0 m

[1] Slope: description recorded in four classes based on visual observation: 1 = flat; 2 = gentle; 3 = moderate; 4 = steep

[2] Waterway distance: recorded as approximate distance (in classes) to nearest waterway: 1 = <10m; 2 = 10-50m; 3 = 51-100m; 4 = 101-200m; 5 = >200m

[3] Distance to surrounding land-use: recorded as approximate distance (in classes) to nearest alternative land-use: 1 = <10m; 2 = 10-50m; 3 = 51-100m; 4 = 101-200m; 5 = >200m.

Figure I: Examples of the three habitat classes used in the study: (a) pasture, site P3 with dense, managed grass cover; (b) pasture, site P6 with riparian regrowth visible in the middle ground and forested hills in the far distance; (c) pasture, site P7 with transect midline and 1m x 1m vegetation survey quadrat shown; (d) regrowth, site R2 with dense small-leaved privet in the understorey; (e) regrowth, site R3 with a more open understorey and visible cow tracks; (f) forest, site F1 with an open understorey, tall, closed canopy and leaf litter and large woody debris all visible; and (g) forest, site F3 with special life forms including tree ferns, small palms and vines.
(Photo (f) courtesy of Dr Bill McDonald).

APPENDIX II

Table II: Vegetation structure attributes included in analyses; method of measurement, type of analyses undertaken and reason for inclusion or exclusion from analyses are given for each variable. Site frequencies are given for pasture (P; N = 8), regrowth (R; N = 8) and forest (F; N = 5) separately, for combined pasture and forest sites (N = 13) and for all study sites (N = 21).

Variable	**Measurement**	**Site frequency:**					**ANOVA and RRD**[1]	**Ordination**[2] **/ANOSIM**	**Reasons for inclusion/exclusion**
		F	**P**	**F&P**	**R**	**Total**			
Ground									
Grass cover	% cover	2	8	10	8	18	Y	Y	Attributes such as grass and herb cover associated with pasture habitat, while other ground cover attributes and increasing woody debris associated with forest habitat.
Fern cover	% cover	4	0	4	5	9	Y	Y	
Vine cover	% cover	5	0	5	4	9	Y	Y	
Herb cover	% cover	4	6	10	8	18	Y	Y	
Moss cover	% cover	4	0	4	3	7	Y	Y	
Litter cover	% cover	5	3	8	8	16	Y	Y	
Bare ground	% cover	5	7	12	8	20	Y	Y	
Other cover	% cover	5	2	7	7	14	Y	Y	
Fine woody debris	number/site	5	1	6	8	14	Y	Y	
Coarse woody debris	number/site	4	0	4	6	10	Y	Y	
Special life forms									
Slender vines	% frequency	5	0	5	5	10	Y	Y	Presence of special life forms, and increasing life form richness, associated with forest structure. (Life forms present at ≥4 combined P and F sites included in all analyses; life forms present at ≥2 study sites included in ordinations and ANOSIMs, life forms present at <2 sites not included in any analyses).
Small palms	% frequency	5	0	5	0	5	Y	Y	
Tree ferns	% frequency	5	0	5	1	6	Y	Y	
Ground ferns	% frequency	5	0	5	8	13	Y	Y	
Epiphyte ferns	% frequency	5	0	5	1	6	Y	Y	
Strap-leaf herbs	% frequency	5	0	5	0	5	Y	Y	
Life form richness	life forms/site	5	0	5	8	13	Y	Y	
Strangler figs	% frequency	2	0	2	0	2	N	Y	
Hemi-epiphytes	% frequency	3	0	3	0	3	N	Y	
Robust vines	% frequency	2	0	2	0	2	N	Y	
Vine tangles	% frequency	3	0	3	2	5	N	Y	
Individual scramblers	% frequency	1	0	1	4	5	N	Y	
Other epiphytes	% frequency	2	0	2	0	2	N	Y	
Thicket-form scrambler	% frequency	0	0	0	1	1	N	N	
Cordylines	% frequency	0	0	0	1	1	N	N	
Canopy and trees									
Canopy height	metres (m)	5	0	5	8	13	Y	Y	High canopy cover, canopy height and tree density associated with forest structure. Tree and stag basal areas are indicators for above ground biomass and potential fauna habitat respectively.
Canopy cover	% cover	5	0	5	8	13	Y	Y	
Stems <20cm dbh*	stems/ha	5	1	6	8	14	Y	Y	
Stems 20-50cm dbh*	stems/ha	5	0	5	8	13	Y	Y	
Stems >50cm dbh*	stems/ha	5	0	5	6	11	Y	Y	
Tree basal area	m^2/ha	5	1	6	8	14	Y	Y	
Stag basal area	m^2/ha	4	0	4	1	5	Y	Y	

[1] ANOVAs conducted on variables recorded at ≥4 of the 13 combined pasture and forest sites. Where ANOVA $p<0.10$, relative recovery distance (RRD) was calculated using: RRD = 100 x (R mean – P mean) / (F mean – P mean) = %.

[2] All variables recorded at ≥2 of the 21 total study sites were included in Multi-dimensional scaling (MDS) ordinations and ANOSIM analyses, undertaken in R software with the additional Vegan Package.

* Pooled stem variables log-transformed prior to analyses.

APPENDIX III

Table III: Full list of tree and shrub families recorded at study sites, the species recorded within each family and the dispersal mode of each species. Site frequencies are given for pasture (P; N = 8), regrowth (R; N = 8) and forest (F; N = 5) separately, and for all study sites together (N = 21). Types of statistical analyses undertaken and reasons for inclusion or exclusion from analyses are given for each species. All species level analyses included trees and shrub stems >1m in height (all dbh classes); all family level analyses included trees and shrub stems >1m in height (>10cm dbh).

Family	**Species**	**Disp. Mode^1**	**Site frequency:**				**ANOVA and RRD^2**	**Ordination^3 /ANOSIM**	**Reason for inclusion/exclusion**
			P	**R^s**	**F**	**Total**			
APOCYNACEAE	*Alyxia ruscifolia*	F<10	0	0	2	2	N	Y*	Family present at ≥2 sites
	Tabernaemontana pandacaqui	F<10	0	2^s	4	6	N	Y***	Species present at ≥2 sites
ARALIACEAE	*Polyscias sambucifolia*	F<10	0	2^s	0	2	N	Y**	Species present at ≥2 sites
ARAUCARIACEAE	*Araucaria cunninghamii*	Wind	0	1	5	6	Y***	Y***	Top 10 forest species/family
ARECACEAE	*Archontophoenix cunninghamiana*	F≥10	0	0	1	1	N	N	Present at <2 sites
	Linospadix monostachya	F<10	0	0	5	5	N	N	Seedlings/stems <10cm dbh
	Livistona australis	F≥10	1	0	0	1	N	N	Present at <2 sites
ASTERACEAE	*Ozothamnus diosmifolius*	Wind	0	5	0	5	N	Y***	Family present at ≥2 sites
CELASTRACEAE	*Denhamia celastroides*	F<10	0	0	5	5	Y*	Y***	Top 10 forest family
CUNONIACEAE	*Caldcluvia paniculosa*	Wind	0	0	1	1	Y*	Y*	Top 10 forest family
	Callicoma serratifolia	F<10	0	1^s	5	6	Y***	Y***	Top 10 forest species/family
	Ceratopetalum apetalum	Wind	0	2	5	7	Y***	Y***	Top 10 forest species/family
	Geissois benthamii	Wind	0	0	1	1	Y*	Y*	Top 10 forest family
	Karrabina benthamiana	Wind	0	0	1	1	Y*	Y*	Top 10 forest family
	Schizomera ovata	F≥10	0	2^s	4	6	Y*	Y***	Top 10 forest family
CUPRESSACEAE	*Callitris macleayana*	Wind	0	0	1	1	N	N	Present at <2 sites
CYATHEACEAE	*Cyathea australis*	Wind	0	1	1	2	N	Y***	Species present at ≥2 sites
	Cyathea leichhardtiana	Wind	0	0	4	4	N	Y***	Species present at ≥2 sites
DRACAENACEAE	*Cordyline rubra*	F<10	0	0	2	2	N	N	Seedlings/stems <10cm dbh
	Cordyline stricta	F≥10	0	0	1	1	N	N	Present at <2 sites
EBENACEAE	*Diospyros pentamera*	F≥10	0	0	1	1	N	N	Present at <2 sites

Family	Species	Disp. Mode^1	Site frequency:				ANOVA and RRD^2	Ordination^3 /ANOSIM	Reason for inclusion/exclusion
			P	R^s	F	Total			
ELAEOCARPACEAE	*Elaeocarpus reticulatus*	F<10	0	1^s	3	4	N	Y***	Species present at ≥2 sites
	Sloanea australis	F<10	0	0	1	1	N	Y*	Family present at ≥2 sites
	Sloanea woollsii	F<10	0	0	4	4	N	Y***	Species present at ≥2 sites
ERICACEAE	*Leucopogon sp1.*	F<10	0	1^s	0	1	N	N	Present at <2 sites
	Trochocarpa laurina	F≥10	0	2^s	5	6	Y**	Y***	Top 10 forest species/family
EUPHORBIACEAE	*Austrobuxus swainii*	F≥10	0	0	3	3	N	Y***	Species present at ≥2 sites
	Baloghia inophylla	F≥10	0	0	1	1	N	Y*	Family present at ≥2 sites
	Excoecaria ovalis	F<10	0	0	1	1	N	Y*	Family present at ≥2 sites
EUPOMATIACEAE	*Eupomatia laurina*	F<10	0	0	1	1	N	N	Present at <2 sites
GROSSULARIACEAE	*Anopterus macleayanus*	Wind	0	0	2	2	Y*	Y***	Top 10 forest family
	Polyosma cunninghamii	F≥10	0	0	4	4	Y*	Y***	Top 10 forest family
LAMIACEAE	*Clerodendrum floribundum*	F<10	0	0	1	1	N	N	Present at <2 sites
LAURACEAE	*Cinnamomum oliveri*	F≥10	0	1^s	5	6	Y*	Y***	Top 10 forest family
	Cryptocarya dorrigoensis	F≥10	0	0	3	3	Y*	Y***	Top 10 forest family
	Cryptocarya erythroxylon	F≥10	0	0	1	1	Y*	Y*	Top 10 forest family
	Cryptocarya glaucescens	F≥10	0	3^s	5	8	Y***	Y***	Top 10 forest species/family
	Cryptocarya meissneriana	F≥10	0	0	5	5	Y***	Y***	Top 10 forest species/family
LAURACEAE	*Crypotcarya rigida*	F≥10	0	0	2	2	Y*	Y***	Top 10 forest family
	Cryptocarya sp1	F≥10	0	0	1	1	Y*	Y*	Top 10 forest family
	Endiandra crassiflora	F<10	0	0	2	2	Y*	Y*	Top 10 forest family
	Endiandra discolor	F≥10	0	0	4	4	Y*	Y***	Top 10 forest family
	Endiandra introrsa	F<10	0	0	2	2	Y*	Y***	Top 10 forest family
	Endiandra muelleri	F≥10	0	0	3	3	Y*	Y***	Top 10 forest family
LAURACEAE	*Litsea reticulata*	F≥10	0	0	3	3	Y*	Y***	Top 10 forest family
	Neolitsea dealbata	F<10	0	1^s	5	6	Y*	Y***	Top 10 forest family
MELIACEAE	*Synoum glandulosum*	F<10	0	0	4	4	N	Y***	Species present at ≥2 sites
MIMOSACEAE	*Acacia melanoxylon*	F<10	0	6	5	11	N	Y***	Species present at ≥2 sites
	Archidendron grandiflorum	Other	0	0	1	1	N	Y*	Family present at ≥2 sites

Family	Species	Disp. Mode[1]	Site frequency:				ANOVA and RRD[2]	Ordination[3] /ANOSIM	Reason for inclusion/exclusion
			P	R^{s}	F	Total			
MONIMIACEAE	*Doryphora sassafras*	Wind	0	1^{s}	5	6	Y*	Y***	Top 10 forest family
	Wilkiea huegeliana	F≥10	0	0	4	4	Y*	Y***	Top 10 forest family
MYRTACEAE	*Acmena smithii*	F<10	0	0	4	4	Y***	Y***	Top 10 forest species/family
	Archirhodomyrtus beckleri	F<10	0	0	4	4	Y*	Y***	Top 10 forest family
	Backhousia citriodora	Wind	0	0	1	1	Y*	Y*	Top 10 forest family
	Backhousia myrtifolia	Wind	0	0	1	1	Y*	Y*	Top 10 forest family
	Eucalyptus grandis	Wind	0	0	1	1	Y*	Y*	Top 10 forest family
	Eucalyptus saligna	Wind	0	2	1	3	Y*	Y***	Top 10 forest family
	Gossia acmenoides	F<10	0	0	1	1	Y*	Y*	Top 10 forest family
	Kunzea ericoides	Wind	0	6	0	6	Y*	Y***	Top 10 forest family
	Lophostemon grandiflorus	F<10	0	0	2	2	Y*	Y*	Top 10 forest family
	Pilidiostigma glabrum	F≥10	0	0	5	5	Y*	Y***	Top 10 forest family
MYRTACEAE	*Syzygium oleosum*	F≥10	0	0	2	2	Y*	Y***	Top 10 forest family
	Tristaniopsis laurina	Wind	0	4	3	7	Y*	Y***	Top 10 forest family
NOTHOFAGACEAE	*Nothofagus moorei*	Wind	0	1	0	1	N	N	Present at <2 sites
OLEACEAE	*Ligustrum lucidum*	F<10	0	1^{s}	0	1	N	Y*	Family present at ≥2 sites
	Ligustrum sinense	F<10	3	8	0	11	N	Y***	Species present at ≥2 sites
PITTOSPORACEAE	*Pittosporum multiflorum*	F<10	0	0	4	4	N	Y***	Species present at ≥2 sites
	Pittosporum revolutum	F<10	0	0	3	3	N	Y***	Species present at ≥2 sites
	Pittosporum undulatum	F<10	0	0	1	1	N	Y*	Family present at ≥2 sites
PRIMULACEAE	*Myrsine howittiana*	F<10	0	0	2	2	N	Y***	Species present at ≥2 sites
PROTEACEAE	*Banksia Integrifolia*	Wind	0	1	0	1	Y*	Y*	Top 10 forest family
	Hakea salicifolia	Wind	0	2	1	3	Y*	Y***	Top 10 forest family
	Lomatia fraseri	Wind	0	2	0	2	Y*	Y***	Top 10 forest family
	Orites excelsa	Wind	0	0	5	5	Y***	Y***	Top 10 forest species/family
	Persoonia media	F<10	0	2^{s}	3	5	Y*	Y***	Top 10 forest family
	Stenocarpus salignus	Wind	0	0	3	3	Y*	Y***	Top 10 forest family
	Triunia youngiana	F≥10	0	0	5	5	Y*	Y***	Top 10 forest family

Family	Species	Disp. Mode[1]	Site frequency:				ANOVA and RRD[2]	Ordination[3] /ANOSIM	Reason for inclusion/exclusion
			P	R[s]	F	Total			
QUINTINIACEAE	*Quintinia verdonii*	Wind	0	0	3	3	N	Y***	Species present at ≥2 sites
RHAMNACEAE	*Alphitonia excelsa*	F<10	0	1[s]	1	2	N	Y***	Species present at ≥2 sites
ROSACEAE	*Crataegus monogyna*	F<10	0	2	0	2	N	Y***	Species present at ≥2 sites
	Rubus anglocandicans	F<10	0	2[s]	0	2	N	Y*	Family present at ≥2 sites
RUBIACEAE	*Atractocarpus benthamianus*	F<10	0	0	3	3	Y**	Y***	Top 10 forest species/family
	Unidentified genus	Other	0	1[s]	1	1	N	Y*	Family present at ≥2 sites
RUTACEAE	*Acradenia euodiiformis*	Other	0	0	2	2	N	Y*	Family present at ≥2 sites
	Acronychia pubescens	F≥10	0	0	2	2	N	Y***	Species present at ≥2 sites
	Melicope hayesii	F<10	0	0	2	2	N	Y***	Species present at ≥2 sites
SAPINDACEAE	*Guioa semiglauca*	F<10	0	1[s]	2	3	Y*	Y***	Top 10 forest family
	Sarcopteryx stipata	F<10	0	0	5	5	Y***	Y***	Top 10 forest species/family
SAPOTACEAE	*Planchonella australis*	F≥10	0	0	1	1	N	N	Present at <2 sites
SOLANACEAE	*Duboisia myoporoides*	F<10	0	3	1	4	N	Y***	Species present at ≥2 sites
	Solanum inaequilaterum	F<10	0	0	1	1	N	Y*	Family present at ≥2 sites
	Solanum mauritianum	F<10	0	3[s]	0	3	N	Y*	Family present at ≥2 sites
STERCULIACEAE	*Argyrodendron actinophyllum*	Wind	0	0	2	2	N	Y***	Species present at ≥2 sites
WINTERACEAE	*Tasmannia insipida*	F<10	0	0	5	5	Y*	Y***	Top 10 forest family

[1] Seed dispersal modes are classified as follows: F<10 = frugivore dispersed with a diaspore size <10 mm; F≥10 = frugivore dispersed with a diaspore ≥10 mm; Wind = wind dispersed; and Oher = other method of seed dispersal. Three species were categorised as ‘other’ because their seed dispersal mode could not be determined: *Archidendron grandiflorum* (Mimosaceae) has a nonvolant seed with an elaiosome; *Acradenia euodiiformis* (Rutaceae) has a small diaspore (<10 mm); and the third species (in the Rubiaceae family) could not be identified to the genus level.

[2] ANOVAs conducted on all variables recorded at ≥4 of the 13 combined pasture and forest sites. Where ANOVA p<0.10 relative recovery distance (RRD) was also calculated using: RRD = 100 x (R mean – P mean) / (F mean – P mean) = %.

[3] All variables recorded at ≥2 of the 21 total study sites were included in Multi-dimensional scaling (MDS) ordination and ANOSIM analyses, undertaken in R software with the additional Vegan Package.

[S] Species only present in the seedling layer of regrowth sites

* Analysed at the family level only ** Analysed at the species level only *** Analysed at both the family level and the species level

APPENDIX IV

Table IV: Full list of species recorded within other life forms: epiphytes, ferns, herbs, grasses and vines. Site frequencies are given for pasture (P; N = 8), regrowth (R; N = 8) and forest (F; N = 5) separately, and for all study sites together (N = 21). Species level analyses were not undertaken but all species were included in life-form level analyses.

Life-form	**Family**	**Species**	**Site frequency:**			
			P	**R**	**F**	**Total**
Epiphyte	ASPLENIACEAE	*Asplenium australasicum*	0	0	4	4
		Asplenium polyodon	0	0	2	2
	DAVALLIACEAE	*Davallia pyxidata*	0	0	1	1
	EPIDENDROIDEAE	*Bulbophyllum sp1*	0	0	1	1
	ORCHIDACEAE	*Plectorrhiza tridentata*	0	0	1	1
		Dendrobium tetragonum	0	0	2	2
	POLYPODIACEAE	*Platycerium bifurcatum*	0	0	5	5
		Platycerium superbum	0	0	1	1
	TRIMENIACEAE	*Sarcochilus falcatus*	0	0	2	2
Fern	BLECHNACEAE	*Blechnum nudum*	0	2	1	3
		Blechnum patersonii	0	2	0	2
		Blechnum wattsii	0	0	4	4
		Doodia aspera	0	2	1	3
		Blechnum cartilagineum	0	2	4	6
	DAVALLIACEAE	*Davallia solida*	0	0	2	2
	DENNSTAEDTIACEAE	*Pteridium esculentum*	0	5	0	5
		Hypolepis sp1	1	1	0	2
	DICKSONIACEAE	*Calochlaena dubia*	0	1	0	1
	DRYOPTERIDACEAE	*Polystichum setiferum*	0	2	0	2
		Lastreopsis microsora	0	0	1	1
		Lastreopsis decomposita	0	0	1	1
	GLEICHENIACEAE	*Gleichenia dicarpa*	0	2	1	3
		Sticherus flabellatus	0	1	1	2
	LINDSAEACEAE	*Lindsaea microphylla*	0	1	0	1
	OSMUNDACEAE	*Todea barbara*	0	1	0	1
	POLYPODIACEAE	*Microsorum scandens*	0	0	2	2
	PSILOTACEAE	*Tmesipteris truncata*	0	0	1	1
		Psilotum nudum	0	1	1	2
	PTERIDACEAE	*Adiantum silvaticum*	0	1	4	5
	THELYPTERIDACEAE	*Christella dentata*	0	3	0	3
Grass	CYPERACEAE	*Carex breviculmis*	0	6	1	7
		Cyperus eragrostis	0	3	0	3
		Scleria mackaviensis	0	1	0	1
		Gahnia sieberiana	0	1	0	1
		Gahnia aspera	0	1	2	3
		Exocarya sclerioides	0	0	4	4

Life-form	**Family**	**Species**	**Site frequency:**			
			P	**R**	**F**	**Total**
Grass	POACEAE	*Andropogon virginicus*	1	3	0	4
		Axonopus affinis	5	5	0	10
		Dactylis glomerata	1	0	0	1
		Entolasia marginata	0	5	0	5
		Imperata brevifolia	0	2	0	2
		Lolium perenne	1	0	0	1
		Microlaena stipoides	0	6	0	6
		Oplismenus imbecillis	0	7	0	7
		Paspalum urvillei	3	2	0	5
		Poa annua	2	0	0	2
		Setaria australiensis	3	1	0	4
		Sporobolus africanus	2	1	0	3
		Pennisetum clandestinum	7	2	0	9
		Paspalum dilatatum	3	0	0	3
		Microstegium nudum	0	1	0	1
		Panicum sp1	1	0	0	1
		Unidentified species	1	1	0	2
Herb	ALSTROEMERIACEAE	*Drymophila moorei*	0	0	4	4
	APIACEAE	*Centella asiatica*	0	5	0	5
		Hydrocotyle acutiloba	2	3	0	5
	ARACEAE	*Gymnostachys anceps*	0	0	2	2
	ASTERACEAE	*Coronidium rupicola*	0	5	0	5
		Bidens pilosa	0	2	0	2
		Crassocephalum crepidioides	0	3	0	3
		Hypochaeris radicata	2	6	0	8
		Conyza canadensis	0	2	0	2
		Calotis sp1	0	1	0	1
		Senecio madagascariensis	3	0	0	3
		Ambrosia artemisiifolia	0	1	0	1
		Taraxacum officinale	2	0	0	2
		Lactuca serriola	1	0	0	1
		Unidentified species	2	2	0	4
	BRASSICACEAE	*Lepidium sp1*	1	0	0	1
	CAMPANULACEAE	*Lobelia purpurascens*	0	1	0	1
	COMMELINACEAE	*Tradescantia fluminensis*	0	1	0	1
	CONVOLVULACEAE	*Dichondra repens*	1	0	0	1
	FABACEAE	*Trifolium repens*	7	2	0	9
	GERANIACEAE	*Geranium solanderi*	0	3	0	3
	HALORAGACEAE	*Gonocarpus tetragynus*	0	1	0	1
	HEMEROCALLIDACEAE	*Dianella caerulea*	0	0	3	3

Life-form	**Family**	**Species**	**Site frequency:**			
			P	**R**	**F**	**Total**
Herb	JUNCACEAE	*Juncus usitatus*	2	1	0	3
	LAMIACEAE	*Ajuga australis*	1	0	0	1
		Lamium amplexicaule	0	1	0	1
		Prunella vulgaris	1	5	0	6
	LOBELIACEAE	*Lobelia trigonocaulis*	0	2	0	2
	OXALIDACEAE	*Oxalis sp1*	0	1	0	1
	PLANTAGINACEAE	*Digitalis purpurea*	0	1	0	1
		Plantago lanceolata	2	4	0	6
		Plantago sp1	2	0	0	2
	POLYGONACEAE	*Persicaria microcephala*	1	0	0	1
		Rumex obtusifolius	1	1	0	2
		Rumex brownii	0	1	0	1
	RANUNCULACEAE	*Ranunculus sp1*	0	3	0	3
	ROSACEAE	*Fragaria vesca*	0	1	0	1
	RUBIACEAE	*Galian leptogonium*	0	1	0	1
	SOLANACEAE	*Solanum americanum*	0	2	0	2
		Physalis sp1	0	1	0	1
	VERBENACEAE	*Verbena rigida*	1	2	0	3
	VIOLACEAE	*Viola hederacea*	0	3	0	3
	XANTHORRHOEACEAE	*Dianella tasmanica*	0	1	0	1
		Lomandra longifolia	0	2	0	2
		Lomandra spicata	0	0	5	5
	UNKNOWN	*Unidentified species*	0	2	0	2
Vine	APOCYNACEAE	*Parsonsia straminea*	0	1	3	4
		Melodinus australis	0	0	5	5
		Parsonsia induplicata	0	0	1	1
	ARACEAE	*Pothos longipes*	0	0	1	1
	ARALIACEAE	*Cephalaralia cephalobotrys*	0	0	1	1
	ARECACEAE	*Calamus muelleri*	0	0	3	3
		Calamus australis	0	0	2	2
	CAPRIFOLIACEAE	*Lonicera japonica*	0	8	0	8
	DILLENIACEAE	*Hibbertia scandens*	0	0	3	3
	FABACEAE	*Austrosteenisia glabristyla*	0	0	1	1
	FLACOURTIACEAE	*Berberidopsis beckleri*	0	0	1	1
	MONIMIACEAE	*Palmeria racemosa*	0	0	3	3
		Palmeria scandens	0	0	2	2
	MORACEAE	*Trophis scandens subsp. scandens*	0	0	1	1
	MYRSINACEAE	*Embelia australiana*	0	1	4	5
	PETERMANNIACEAE	*Petermannia cirrosa*	0	0	5	5
	RIPOGONACEAE	*Ripogonum elseyanum*	0	0	4	4

Life-form	**Family**	**Species**	**Site frequency:**			
			P	**R**	**F**	**Total**
Vines	RIPOGONACEAE	*Ripogonum fawcettianum*	0	0	4	4
	ROSACEAE	*Rubus moluccanus*	0	5	0	5
		Rubus rosifolius	0	2	0	2
		Rubus nebulosus	0	0	2	2
	RUBIACEAE	*Morinda jasminoides*	0	1	3	4
	SMILACACEAE	*Ripogonum discolor*	0	0	5	5
		Smilax glyciphylla	0	0	4	4
	SMILACACEAE	*Geitonoplesium cymosum*	0	0	2	2
		Eustrephus latifolius	0	0	1	1
		Smilax australis	0	0	1	1
	TRIMENIACEAE	*Trimenia moorei*	0	0	2	2
	VITACEAE	*Cissus sterculiifolia*	0	0	4	4
		Cissus hypoglauca	0	0	5	5

APPENDIX V

Table V: Full list of bird species recorded “on site” during this study: showing family, functional habitat group (FHG), site frequency, number of individuals recorded and mean abundance in pasture (P), regrowth (R) and forest (F). Site frequencies are shown for the three habitat types separately (N= 8, 8, 5 in P, R, F respectively), for combined pasture and forest sites (N = 13) and for all study sites (N = 21). Mean abundance is the average no. of individuals per 30 minute survey within the 0.3 ha survey area (“on site”).

Family	**Species**	**Common name**	**FHG**[1]	**Site frequency:**					**No. birds**	**P mean**	**R mean**	**F mean**
				P	**R**	**F**	***F&P***	***Total***				
Megapodiidae	*Alectura lathami*	Australian brush-turkey	RF	0	1	0	*0*	*1*	1	0.00	0.02	0.00
Anatidae	*Chenonetta jubata*	Australian wood duck	GW	0	2	0	*0*	*2*	3	0.00	0.06	0.00
	Anas superciliosa	Pacific black duck	GW	0	2	0	*0*	*2*	8	0.0	0.1	0.0
Ardeidae	*Egretta novaehollandiae*	White-faced heron	GW	1	0	0	*1*	*1*	1	0.02	0.00	0.00
	Ardea pacifica	White-necked heron	GW	0	1	0	*0*	*1*	1	0.00	0.02	0.00
Rallidae	*Porphyrio porphyrio*	Purple swamphen	GW	1	0	0	*1*	*1*	1	0.02	0.00	0.00
Scolopacidae	*Gallinago hardwickii*	Latham's snipe	GW	1	0	0	*1*	*1*	1	0.02	0.00	0.00
Charadriidae	*Vanellus miles*	Masked lapwing	GW	2	2	0	*2*	*4*	17	0.2	0.1	0.0
Columbidae	*Columba leucomela*	White-headed pigeon	RF	0	1	0	*0*	*1*	1	0.00	0.02	0.00
	Macropygia amboinensis	Brown cuckoo-dove	RF	0	3	1	*1*	*4*	10	0.00	0.17	0.07
	Ocyphaps lophotes	Crested pigeon	GW	1	0	0	*1*	*1*	7	0.15	0.00	0.00
	Geopelia humeralis	Bar-shouldered dove	EF	3	4	0	*3*	*7*	25	0.31	0.21	0.00
Cacatuidae	*Cacatua galerita*	Sulphur-crested cockatoo	MF	6	3	2	*8*	*11*	41	0.04	0.25	0.23
Psittacidae	*Trichoglossus haematodus*	Rainbow lorikeet	MF	0	0	1	*1*	*1*	1	0.00	0.00	0.03
	Alisterus scapularis	Australian king-parrot	MF	1	3	1	*2*	*5*	13	0.02	0.17	0.03
	Platycercus elegans	Crimson rosella	MF	2	3	1	*3*	*6*	15	0.1	0.2	0.0
	Platycercus eximius	Eastern rosella	EF	3	0	0	*3*	*3*	6	0.1	0.0	0.0
Cuculidae	*Chrysococcyx lucidus*	Shining bronze-cuckoo	MF	0	1	2	*2*	*3*	4	0.0	0.1	0.1
Aegothelidae	*Aegotheles cristatus*	Australian owlet-nightjar	EF	0	0	1	*1*	*1*	1	0	0	0.03
Halcyonidae	*Dacelo novaeguineae*	Laughing kookaburra	MF	1	1	0	*1*	*2*	5	0.02	0.08	0.00
Climacteridae	*Cormobates leucophaea*	White-throated treecreeper	RF	0	0	3	*3*	*3*	7	0.0	0.0	0.2
Maluridae	*Malurus cyaneus*	Superb fairy-wren	EF	3	8	0	*3*	*11*	60	0.1	1.1	0.0

Family	Species	Common name	FHG[1]	Site frequency:					No. birds	P mean	R mean	F mean
				P	**R**	**F**	***F&P***	***Total***				
Pardalotidae	*Pardalotus punctatus*	Spotted pardalote	EF	0	0	1	*1*	*1*	1	0.00	0.00	0.03
Acanthizidae	*Sericornis citreogularis*	Yellow-throated scrubwren	RF	0	4	3	*3*	*7*	24	0.0	0.2	0.5
	Sericornis frontalis	White-browed scrubwren	MF	0	8	4	*4*	*12*	71	0.0	1.3	0.3
	Sericornis magnirostra	Large-billed scrubwren	RF	0	1	5	*5*	*6*	17	0.00	0.04	0.50
	Gerygone mouki	Brown gerygone	RF	0	7	5	*5*	*12*	124	0.0	1.2	2.2
	Acanthiza pusilla	Brown thornbill	MF	0	8	5	*5*	*13*	98	0.0	1.4	1.0
	Acanthiza chrysorrhoa	Yellow-rumped thornbill	GW	2	2	0	*2*	*4*	23	0.4	0.1	0.0
	Acanthiza nana	Yellow thornbill	EF	0	0	1	*1*	*1*	1	0.00	0.00	0.03
Meliphagidae	*Anthochaera chrysoptera*	Little wattlebird	EF	0	1	0	*0*	*1*	1	0.00	0.02	0.00
	Meliphaga lewinii	Lewin's honeyeater	MF	3	8	5	*8*	*16*	66	0.1	0.9	0.7
	Lichenostomus chrysops	Yellow-faced honeyeater	EF	2	4	0	*2*	*4*	6	0.04	0.21	0.00
	Melithreptus lunatus	White-naped honeyeater	EF	0	0	1	*1*	*1*	4	0.0	0.0	0.1
	Acanthorhynchus tenuirostris	Eastern spinebill	MF	3	8	3	*6*	*14*	31	0.1	0.4	0.2
	Myzomela sanguinolenta	Scarlet honeyeater	MF	0	1	2	*2*	*3*	3	0.0	0.02	0.07
Petroicidae	*Petroica rosea*	Rose robin	MF	0	0	3	*3*	*3*	5	0.0	0.0	0.2
	Eopsaltria australis	Eastern yellow robin	MF	0	8	3	*3*	*11*	35	0.0	0.7	0.1
Orthonychidae	*Orthonyx temminckii*	Logrunner	RF	0	0	4	*4*	*4*	18	0.0	0.0	0.6
Eupetidae	*Psophodes olivaceus*	Eastern whipbird	MF	0	4	5	*5*	*9*	21	0.0	0.2	0.5
Pachycephalidae	*Pachycephala pectoralis*	Golden whistler	MF	0	8	5	*5*	*13*	60	0.0	0.9	0.6
	Pachycephala rufiventris	Rufous whistler	EF	0	3	0	*0*	*3*	12	0.0	0.3	0.0
	Colluricincla harmonica	Grey shrike-thrush	MF	0	6	3	*3*	*9*	17	0.0	0.2	0.2
Monarchidae	*Monarcha melanopsis*	Black-faced monarch	MF	0	3	2	*2*	*5*	10	0.00	0.15	0.10
	Symposiarchus trivirgatus	Spectacled monarch	RF	0	1	1	*1*	*2*	3	0.00	0.04	0.03
	Myiagra rubecula	Leaden flycatcher	MF	0	1	0	*0*	*1*	1	0.00	0.02	0.00
	Grallina cyanoleuca	Magpie-lark	GW	3	0	0	*3*	*3*	27	0.6	0.0	0.0
Rhipiduridae	*Rhipidura rufifrons*	Rufous fantail	MF	0	6	2	*2*	*8*	17	0.0	0.3	0.1
	Rhipidura albiscapa	Grey fantail	MF	1	8	5	*6*	*14*	82	0.02	1.21	0.70
	Rhipidura leucophrys	Willie wagtail	GW	6	0	0	*6*	*6*	13	0.3	0.0	0.0

Family	Species	Common name	FHG[1]	Site frequency:					No. birds	P mean	R mean	F mean
				P	**R**	**F**	***F&P***	***Total***				
Campephagidae	*Coracina novaehollandiae*	Black-faced cuckoo-shrike	EF	0	1	1	*1*	*2*	2	0.00	0.02	0.03
	Lalage leucomela	Varied triller	MF	0	1	0	*0*	*1*	1	0.00	0.02	0.00
Oriolidae	*Oriolus sagittatus*	Olive-backed oriole	MF	0	1	0	*0*	*1*	1	0.00	0.02	0.00
	Sphecotheres vieilloti	Figbird	RF	0	0	1	*1*	*1*	5	0.0	0.0	0.2
Artamidae	*Cracticus torquatus*	Grey butcherbird	EF	1	1	0	*1*	*2*	2	0.02	0.06	0.00
	Cracticus nigrogularis	Pied butcherbird	EF	1	0	0	*1*	*1*	1	0.02	0.00	0.00
	Cracticus tibicen	Australian magpie	GW	7	5	0	*7*	*12*	52	0.46	0.35	0
	Strepera graculina	Pied currawong	MF	2	2	1	*3*	*5*	28	0.00	0.52	0.03
Corvidae	*Corvus tasmanicus*	Forest raven	MF	2	1	0	*2*	*3*	13	0.00	0.02	0.00
	Corvus orru	Torresian crow	MF	6	6	0	*6*	*12*	40	0.2	0.2	0.0
Ptilonorhynchidae	*Ailuroedus crassirostris*	Green catbird	RF	0	0	2	*2*	*2*	3	0.0	0.0	0.1
	Ptilonorhynchus violaceus	Satin bowerbird	MF	0	5	0	*0*	*5*	15	0.0	0.3	0.0
Motacillidae	*Anthus novaeseelandiae*	Australasian pipit	GW	5	0	0	*5*	*5*	26	0.5	0.0	0.0
Estrildidae	*Neochmia temporalis*	Red-browed finch	MF	2	5	0	*2*	*7*	53	0.8	0.3	0.0
Nectariniidae	*Dicaeum hirundinaceum*	Mistletoebird	MF	1	1	0	*1*	*2*	3	0.00	0.04	0.00
Hirundinidae	*Hirundo neoxena*	Welcome swallow	GW	6	2	0	*6*	*8*	61	1.0	0.2	0.0
	Petrochelidon nigricans	Tree martin	GW	1	0	0	*1*	*1*	3	0.1	0.0	0.0
Timaliidae	*Zosterops lateralis*	Silvereye	MF	1	8	0	*1*	*9*	55	0.02	1.06	0.00
Sturnidae	*Sturnus vulgaris*	Common starling	XX	5	1	0	*5*	*6*	18	0.35	0.02	0.00
	Sturnus tristis	Common mynah	XX	1	0	0	*1*	*1*	2	0.04	0.00	0.00

[1] Functional habitat groups were defined as follows:

RF = Rainforest-dependent: species are largely confined to, or dependent on, rainforest; MF = Mixed Forest: species occur mainly in a wider range of forested habitats spanning both rainforest and the more open-canopied eucalypt forests and woodlands; EF = Eucalypt Forest: species are typically found in eucalypt forest or woodland, and only occasionally occur in denser forest (including rainforest), or less wooded habitats; GW = Grassland/ Wetland: species occur mainly in grassland, wetland or water, although they may also occur within lightly-timbered open habitat, or be dependent on dense swampy vegetation; includes aerial feeding species; and XX = Non-native: species are introduced species which have established free-living populations since European settlement.

APPENDIX VI

Table VI: Full list of bird species recorded “on patch”, as “flyovers” or where identification to the species level was not possible during this study: showing family, functional habitat group (FHG), record type, site frequency and number of individuals recorded. Site frequencies are shown for the three habitat types separately (N= 8, 8, 5 in P, R, F respectively), for combined pasture and forest sites (N = 13) and for all study sites (N = 21). “On patch” is all individuals recorded within a 20 m buffer outside the 0.3 ha survey area. “Flyovers” are defined as individuals flying over a site at more than 10 m above the average canopy height for that habitat type.

Family	**Species**	**Common name**	**FHG**[1]	**Record type**	**Site frequency:**					**No. birds**
					P	**R**	**F**	***F&P***	***Total***	
Phasianidae	*Coturnix pectoralis*	Stubble quail	GW	patch	1	0	0	*1*	*1*	1
Anatidae	*Chenonetta jubata*	Australian wood duck	GW	patch	0	2	0	*0*	*2*	5
	Anas superciliosa	Pacific black duck	GW	patch/flyover	3	1	0	*3*	*4*	8
Phalacrocoracidae	*Microcarbo melanoleucos*	Little pied cormorant	GW	flyover	1	0	0	*1*	*1*	1
Ardeidae	*Egretta novaehollandiae*	White-faced heron	GW	patch	0	2	0	*0*	*2*	2
	Ardea pacifica	White-necked heron	GW	patch	2	0	0	*2*	*2*	3
Threskiornithidae	*Threskiornis spinicollis*	Straw-necked ibis	GW	patch	2	0	0	*2*	*2*	3
Rallidae	*Porphyrio porphyrio*	Purple swamphen	GW	patch	2	1	0	*2*	*3*	4
Charadriidae	*Vanellus miles*	Masked lapwing	GW	patch	4	4	0	*4*	*8*	12
Columbidae	*Macropygia amboinensis*	Brown cuckoo-dove	RF	patch	0	1	4	*4*	*5*	7
	Geopelia humeralis	Bar-shouldered dove	EF	patch	4	4	0	*4*	*8*	10
	Leucosarcia picata	Wonga pigeon	MF	patch	0	1	3	*3*	*4*	4
	Ptilinopus regina	Rose-crowned fruit-dove	RF	patch	0	0	1	*1*	*1*	1
Cacatuidae	*Calyptorhynchus funereus*	Yellow-tailed black-cockatoo	MF	patch	0	1	1	*1*	*2*	2
	Cacatua galerita	Sulphur-crested cockatoo	MF	patch/flyover	5	2	2	*7*	*9*	34
Psittacidae	*Trichoglossus haematodus*	Rainbow lorikeet	MF	patch	1	2	2	*3*	*5*	5
	Alisterus scapularis	Australian king-parrot	MF	patch/flyover	1	3	1	*2*	*5*	9
	Platycercus elegans	Crimson rosella	MF	patch/flyover	2	2	2	*4*	*6*	7
	Platycercus eximius	Eastern rosella	EF	patch	1	1	0	*1*	*2*	2
Cuculidae	*Cacomantis flabelliformis*	Fan-tailed cuckoo	MF	patch	0	3	2	*2*	*5*	7
	Chrysococcyx lucidus	Shining bronze-cuckoo	MF	patch	0	3	3	*3*	*6*	8

Family	**Species**	**Common name**	**FHG**[1]	**Record type**	**Site frequency:**					**No. birds**
					P	**R**	**F**	***F&P***	***Total***	
Cuculidae	*Scythrops novaehollandiae*	Channel-billed cuckoo	EF	patch	0	0	1	*1*	*1*	1
Alcedinidae	*Ceyx azureus*	Azure kingfisher	MF	patch	0	1	0	*0*	*1*	2
Halcyonidae	*Dacelo novaeguineae*	Laughing kookaburra	MF	patch	1	2	0	*1*	*3*	4
	Todiramphus sanctus	Sacred kingfisher	EF	patch	1	0	0	*1*	*1*	1
Pittidae	*Pitta versicolor*	Noisy pitta	RF	patch	0	0	3	*3*	*3*	4
Climacteridae	*Cormobates leucophaea*	White-throated treecreeper	RF	patch	0	1	3	*3*	*4*	9
Maluridae	*Malurus cyaneus*	Superb fairy-wren	EF	patch	2	5	0	*2*	*7*	10
Acanthizidae	*Sericornis citreogularis*	Yellow-throated scrubwren	RF	patch	0	2	1	*1*	*3*	3
	Sericornis frontalis	White-browed scrubwren	MF	patch	1	7	2	*3*	*10*	14
	Sericornis magnirostra	Large-billed scrubwren	RF	patch	0	1	2	*2*	*3*	4
	Gerygone mouki	Brown gerygone	RF	patch	0	2	3	*3*	*5*	6
	Acanthiza pusilla	Brown thornbill	MF	patch	2	7	2	*4*	*11*	11
	Acanthiza chrysorrhoa	Yellow-rumped thornbill	GW	patch	3	0	0	*3*	*3*	3
Meliphagidae	*Meliphaga lewinii*	Lewin's honeyeater	MF	patch	0	6	5	*5*	*11*	27
	Lichenostomus chrysops	Yellow-faced honeyeater	EF	patch	0	3	1	*1*	*4*	5
	Acanthorhynchus tenuirostris	Eastern spinebill	MF	patch	0	6	2	*2*	*8*	11
	Myzomela sanguinolenta	Scarlet honeyeater	MF	patch	0	0	2	*2*	*2*	2
Petroicidae	*Petroica rosea*	Rose robin	MF	patch	0	0	1	*1*	*1*	2
	Tregellasia capito	Pale-yellow robin	RF	patch	0	0	1	*1*	*1*	1
	Eopsaltria australis	Eastern yellow robin	MF	patch	0	3	3	*3*	*6*	13
Orthonychidae	*Orthonyx temminckii*	Logrunner	RF	patch	0	0	4	*4*	*4*	5
Eupetidae	*Psophodes olivaceus*	Eastern whipbird	MF	patch	0	6	4	*4*	*10*	28
Pachycephalidae	*Pachycephala pectoralis*	Golden whistler	MF	patch	0	6	5	*5*	*11*	21
	Pachycephala rufiventris	Rufous whistler	EF	patch	0	2	0	*0*	*2*	3
	Colluricincla harmonica	Grey shrike-thrush	MF	patch	0	4	4	*4*	*8*	16
Monarchidae	*Monarcha melanopsis*	Black-faced monarch	MF	patch	0	5	2	*2*	*7*	9
	Symposiarchus trivirgatus	Spectacled monarch	RF	patch	0	1	1	*1*	*2*	2
	Grallina cyanoleuca	Magpie-lark	GW	patch	6	0	0	*6*	*6*	6

Family	**Species**	**Common name**	**FHG**[1]	**Record type**	**Site frequency:**					**No. birds**
					P	**R**	**F**	***F&P***	***Total***	
Rhipiduridae	*Rhipidura rufifrons*	Rufous fantail	MF	patch	0	2	1	*1*	*3*	3
	Rhipidura albiscapa	Grey fantail	MF	patch	2	6	3	*5*	*11*	16
	Rhipidura leucophrys	Willie wagtail	GW	patch	1	1	0	*1*	*2*	3
Campephagidae	*Coracina novaehollandiae*	Black-faced cuckoo-shrike	EF	patch	3	0	1	*3*	*4*	4
Artamidae	*Cracticus torquatus*	Grey butcherbird	EF	patch	0	1	1	*1*	*2*	5
	Cracticus nigrogularis	Pied butcherbird	EF	patch	0	2	0	*0*	*2*	2
	Cracticus tibicen	Australian magpie	GW	patch/flyover	7	3	0	*7*	*10*	28
	Strepera graculina	Pied currawong	MF	patch/flyover	1	4	4	*5*	*9*	13
Corvidae	*Corvus tasmanicus*	Forest raven	MF	patch/flyover	2	0	1	*3*	*3*	11
	Corvus orru	Torresian crow	MF	patch/flyover	8	6	1	*9*	*31*	40
Ptilonorhynchidae	*Ailuroedus crassirostris*	Green catbird	RF	patch	0	0	3	*3*	*3*	3
	Ptilonorhynchus violaceus	Satin bowerbird	MF	patch	2	3	2	*4*	*7*	11
Motacillidae	*Anthus novaeseelandiae*	Australasian pipit	GW	patch	3	0	0	*3*	*3*	6
Estrildidae	*Neochmia temporalis*	Red-browed finch	MF	patch	1	1	0	*1*	*2*	2
Nectariniidae	*Dicaeum hirundinaceum*	Mistletoebird	MF	patch/flyover	1	0	0	*1*	*1*	1
Hirundinidae	*Hirundo neoxena*	Welcome swallow	GW	patch/flyover	4	1	0	*4*	*5*	8
Cisticolidae	*Cisticola exilis*	Golden-headed cisticola	GW	patch	3	0	0	*3*	*3*	3
Timaliidae	*Zosterops lateralis*	Silvereye	MF	patch	0	1	0	*0*	*1*	1
Sturnidae	*Sturnus vulgaris*	Common starling	XX	patch	6	0	0	*6*	*6*	12
	Sturnus tristis	Common mynah	XX	patch	1	1	0	*1*	*2*	3
-	-	Unknown species 1	UNK	unidentified	0	0	1	*1*	*1*	1
	-	Unknown species 2	UNK	unidentified	0	0	1	*1*	*1*	1
	-	Unknown species 3	UNK	unidentified	0	1	0	*0*	*1*	1

[1] Functional habitat groups were defined as follows: RF = Rainforest-dependent: species are largely confined to, or dependent on, rainforest; MF = Mixed Forest: species occur mainly in a wider range of forested habitats spanning both rainforest and the more open-canopied eucalypt forests and woodlands; EF = Eucalypt Forest: species are typically found in eucalypt forest or woodland, and only occasionally occur in denser forest (including rainforest), or less wooded habitats; GW = Grassland/ Wetland: species occur mainly in grassland, wetland or water, although they may also occur within lightly-timbered open habitat, or be dependent on dense swampy vegetation; includes aerial feeding species; XX = Non-native: species are introduced species which have established free-living populations since European settlement; and UNK = habitat functional group unknown.

APPENDIX VII

Table VII: All tree and shrub species recorded in regrowth, the number of regrowth (R) and forest (F) sites in which they were recorded as seedlings and as stems >1 m tall (all dbh classes); N = 8, 5 in R, F respectively.

Species[1]	Dispersal mode[2]	No. R sites seedlings	No. R sites stems >1m	No. F sites seedlings	No. F sites stems >1m
Acacia melanoxylon	F<10	5	3	0	5
Alphitonia excelsa	F<10	1	0	0	1
Araucaria cunninghamii	Wind	1	1	2	5
Callicoma serratifolia	F<10	1	0	0	5
Ceratopetalum apetalum	Wind	1	1	3	5
Cinnamomum oliveri	F≥10	1	0	3	4
*Crataegus monogyna**	F<10	1	2	0	0
Cryptocarya glaucescens	F≥10	3	0	2	4
Doryphora sassafras	Wind	1	0	1	5
Duboisia myoporoides	F<10	3	2	0	1
Elaeocarpus reticulatus	F<10	1	0	2	3
Guioa semiglauca	F<10	1	0	2	1
Hakea salicifolia	Wind	1	2	0	1
Kunzea ericoides	Wind	2	6	0	0
Leucopogon sp.	F<10	1	0	0	0
*Ligustrum lucidum**	F<10	1	0	0	0
*Ligustrum sinense**	F<10	5	8	0	0
Lomatia fraseri	Wind	1	2	0	0
Neolitsea dealbata	F<10	1	0	2	5
Ozothamnus diosmifolius	Wind	1	4	0	0
Persoonia media	F<10	2	0	0	3
Polyscias sambucifolia	F<10	1	0	0	0
Schizomera ovata	F≥10	2	0	2	4
Tabernaemontana pandacaqui	F<10	1	0	1	3
Tristaniopsis laurina	Wind	1	4	0	3
Trochocarpa laurina	F≥10	1	0	3	5
*Rubus anglocandicans**	F<10	1	0	0	0
*Solanum mauritianum**	F<10	1	0	0	0

[1] Species with an asterisk are non-native species.

[2] Dispersal modes: wind = wind-dispersed; F<10 = frugivore dispersed with diaspore size <10 mm; and F≥10 = frugivore dispersed with diaspore size ≥10 mm.

APPENDIX VIII

Table VIII: Level of fruiting values (mean, SE) in pasture (P), regrowth (R) and forest (F) across the six survey periods, based on the combined scores for 'number of trees fruiting'[1] and 'average number of fruits per fruiting tree'[2].

Level of fruiting survey	P mean (SE)	R mean (SE)	F mean (SE)
Survey 1 (winter)	0.0	3.8 (0.2)	0.0
Survey 2 (winter)	0.0	3.8 (0.2)	0.0
Survey 3 (spring)	0.0	0.0	0.0
Survey 4 (spring)	0.0	0.0	0.0
Survey 5 (summer)	0.0	4.9 (0.5)	0.6 (0.3)
Survey 6 (summer)	0.0	5.0 (0.5)	1.6 (0.3)

[1] Number of fruiting trees: 0 = no trees in fruit; 1 = 1-2 trees in fruit; 2 = 3-10 trees; 3 = 11-100 trees; 4 = >100 trees.

[2] Average number of fruits per fruiting tree: 0 = no fruits or fruiting trees; 1 = ≤10 fruits per fruiting tree; 2 = 11-50 fruits; 3 = 51-100 fruits; 4 = >100 fruits.

Figure VIII: Small-leaved privet (*Ligustrum sinense)* in fruit (photo courtesy of Dr Bill McDonald).

APPENDIX IX

Table IX: Bird species classified as seed-dispersers for this study: showing family, record type, functional habitat group (FHG), gape size class, frugivory level, number of native fleshy-fruited plant species consumed and functional seed-dispersal group (FSG).

Family	**Species**	**Common name**	**Record type**[1]	**FHG**[2]	**GC**[3]	**FL**[4]	**No spp.**[5]	**FSG**[6]
Columbidae	*Ptilinopus regina*	Rose-crowned fruit-dove	P	RF	M	Ma	>70	A
Cuculidae	*Scythrops novaehollandiae*	Channel-billed cuckoo	P	EF	L	Ma	<20	B
Meliphagidae	*Anthochaera chrysoptera*	Little wattlebird	S	EF	S	Mi	<20	B
	Lichenostomus chrysops	Yellow-faced honeyeater	S	EF	nn	nn	nn	B
	Meliphaga lewinii	Lewin's honeyeater	S	MF	M	Mx	>70	A
	Myzomela sanguinolenta	Scarlet honeyeater	S	MF	S	Mi	nn	B
Campephagidae	*Coracina novaehollandiae*	Black-faced cuckoo-shrike	S	EF	L	Mi	<20	B
	Lalage leucomela	Varied triller	S	MF	S	Mx	<20	B
Oriolidae	*Oriolus sagittatus*	Olive-backed oriole	S	MF	L	Mx	nn	A
	Sphecotheres vieilloti	Figbird	S	RF	L	Ma	>70	A
Artamidae	*Strepera graculina*	Pied currawong	S	MF	L	Mx	51-70	A
	Cracticus nigrogularis	Pied butcherbird	S	EF	nn	nn	nn	B
	Cracticus tibicen	Australian magpie	S	GW	L	Mi	nn	B
	Cracticus torquatus	Grey butcherbird	S	EF	L	Mi	<20	B
Corvidae	*Corvus orru*	Torresian crow	S	MF	L	Mi	<20	B
	Corvus tasmanicus	Forest raven	S	MF	nn	Mi	nn	B
Ptilonorhynchidae	*Ptilonorhynchus violaceus*	Satin bowerbird	S	MF	L	Mx	>70	A
	Ailuroedus crassirostris	Green catbird	S	RF	L	Mx	>70	A
Nectariniidae	*Dicaeum hirundinaceum*	Mistletoebird	S	MF	S	Ma	<20	B
Timaliidae	*Zosterops lateralis*	Silvereye	S	MF	S	Mi	21-50	B
Sturnidae	*Sturnus tristis*	Common mynah	S	XX	S	Mx	nn	B
	Sturnus vulgaris	Common starling	S	XX	S	Mx	nn	B

[1] Record type: S = on site (recorded within the 0.3 ha survey area); P = on patch (recorded within a 20 m buffer surrounding the 0.3 ha survey area).

[2] Functional habitat groups (FHG) were defined as follows: RF = Rainforest-dependent: species are largely confined to, or dependent on, rainforest; MF = Mixed Forest: species occur mainly in a wider range of forested habitats spanning both rainforest and the more open-canopied eucalypt forests and woodlands; EF = Eucalypt Forest: species are typically found in eucalypt forest or woodland, and only occasionally occur in denser forest (including rainforest), or less wooded habitats; GW = Grassland/ Wetland: species occur mainly in grassland, wetland or water, although they may also occur within lightly-timbered open habitat, or be dependent on dense swampy vegetation; includes aerial feeding species; and XX = Non-native: species are introduced species which have established free-living populations since European settlement.

[3] Gape classes (GC) were defined based on Moran et al. (2004) as follows: S = small (<1.0 cm); M = medium (1.0–1.5 cm); and L = large (>1.5 cm).

[4] Frugivory levels were defined based on Moran et al. (2004) as follows: Ma = Major fruit-dominated diet; Mx = mixed diet including fruit; Mi = Fruit a minor dietary component.

[5] Number of native flesh-fruited plant species (No. spp.) estimated to be in the bird's diet was based on Moran and Catterall (2014) and classified as follows: ≤20 plants; 21-50 plants; 51-70 plants; and >70 plants.

[6] Functional seed-dispersal groups (FSG) used in this study were defined as follows: A = "Seed-disperser A", considered to have greatest potential as seed-dispersers, defined as having a gape size >1 cm (as an indicator of the potential size of fruit which could be ingested) AND a diet that includes fruit as more than a minor component AND reported to consume >50 fleshy-fruited plant species; B = "Seed-disperser B" comprises species that meet some but not all of the criteria of the "A" category, having either smaller (<1 cm) gapes OR a diet that includes fruit as a minor or partial component OR reported to consume ≤50 fleshy-fruited plant species. The remaining species were classified as "Unlikely dispersers", defined as eating little or no fruit OR regularly eating fruit but known to crush seeds, and are not included in this table.

APPENDIX X

Table X.1: Values (mean, SE) of individual grazing measurements in pasture (P) and regrowth (R) and forest (F).

Variable	P mean (SE)	R mean (SE)	F mean (SE)
Cow tracks[1]	0.6 (0.1)	0.8 (0.1)	0.0
Cow pats (faeces)[2]	1.0 (0.1)	1.2 (0.1)	0.0
Cow browsing[3]	0.6 (0.1)	1.5 (0.2)	0.0
Cows seen[4]	1.6 (0.2)	0.3 (0.1)	0.0
Total	**3.9 (0.4)**	**3.8 (0.4)**	**0.0**

[1] Cow tracks: recorded as present (1) or absent (0)

[2] Cow pats: recorded in three classes: 0 = no pats; 1 = 1-10 pats; 2 = >10 pats

[3] Cow browsing: recorded in four classes: 0 = no evidence; 1 = minor evidence (<5 trees along edges); 3 = moderate evidence (5-10 trees at edges and within plot); 4 = major evidence (>10 trees within plot)

[4] Cows seen: recorded in three classes: 0 = no cows; 1 = 1-10 cows; 3 = >10 cows

Table X.2: Total grazing pressure, and individual scores for each separate measure, in each regrowth site (N=8). Total number of seedlings, and number of non-native seedlings also shown for each site.

Variable	R1	R2	R3	R4	R5	R6	R7	R8
Cow tracks[1]	0	1	1	1	0	1	1	1
Cow pats[2]	1	0	2	2	0	1	2	2
Cow browsing[3]	0	0	3	3	0	1	4	1
Cows seen[4]	0	0	1	1	0	0	0	0
Total grazing pressure	**1**	**1**	**7**	**7**	**0**	**3**	**7**	**4**
No. seedlings recorded	12	3	2	1	6	12	5	2
No. NN seedlings	3	1	1	1	0	1	0	1

[1] Cow tracks: recorded as present (1) or absent (0)

[2] Cow pats: recorded in three classes: 0 = no pats; 1 = 1-10 pats; 2 = >10 pats

[3] Cow browsing: recorded in four classes: 0 = no evidence; 1 = minor evidence (<5 trees along edges); 3 = moderate evidence (5-10 trees at edges and within plot); 4 = major evidence (>10 trees within plot)

[4] Cows seen: recorded in three classes: 0 = no cows; 1 = 1-10 cows; 3 = >10 cows

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