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The value of animal behaviour as a bio-indicator of restoration quality

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The value of animal behaviour as a bio-indicator of restoration quality

This thesis is presented for the degree of
Doctor of Philosophy

Floyd Holmes

Edith Cowan University
School of Science
2018

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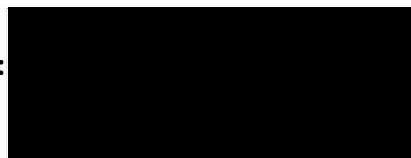
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Abstract

Woodland restoration is a complex endeavour, and restoration ecology as a scientific discipline requires constant re-assessments and adjustments if it is to improve outcomes and better provide for biodiversity. The promise of effective restoration is often used to justify destructive processes that affect many of the world's ecosystems. It is therefore imperative that those promises can be met, which comes down to restoration ecologists' and land managers' capacity to predict and facilitate desirable ecological changes in a timely and socio-economically responsible manner. As perspectives have changed, and knowledge has been gained over the past few decades there have been several fundamental shifts in how restoration is done. Efforts to 'beautify' degraded areas through the planting of fast growing non-native species is no longer thought of as responsible restoration practice. We have a better understanding of ecological thresholds, the creation of novel ecosystems and the ways ecosystems move between stable states through transitional processes. Yet many restoration projects still fail to deliver positive outcomes for certain taxonomic groups. Fauna are an important component of biodiversity, and yet ecological filters and traps remain common in restored habitats.

To date, the focus in restoration has been biased towards restoring flora, while fauna have been under-appreciated and under-utilised. This is likely due to a lack of clarity around how fauna can be used to assess restoration success. This study sought to address that issue by exploring ways fauna could be used to assess habitat quality, and evaluate whether they could fit into existing restoration management tools like a state-and-transition model. Variation in habitat quality was assessed using a number of biodiversity measures and behavioural patterns. This study used Rottnest Island in Western Australia, a mosaic landscape with a woodland restoration program that has been running for over 50 years. The Island's woodland areas support a resident population of red-capped robins *Petroica goodenovii*, which was the focal species of this study. The robins are typically ground-foraging insectivores that generally have been found to respond negatively to anthropocentric land use changes.

The suitability of the Island's robin population as an indicator for the larger avian community was assessed to determine whether management and monitoring could simply focus on improving conditions for robins. Unfortunately, robins were found to be a poor indicator of the larger avian community. Factors that were positively correlated with estimated robin

density, like woodland area and time since last fire, were negatively correlated with density of other avian species of conservation significance.

Invertebrate assembly was surveyed as a measure of food resource availability. There was a significant difference between woodland and heathland areas and to a lesser, but still significant, extent between restored woodland areas of different ages and remnant woodland. A major finding of this study was that Coleoptera were scarcely encountered in ground samples outside of remnant patches, but were among the most common orders in arboreal samples, specifically in old restoration. Given that this order is a major component of numerous insectivore diets, it is likely that this difference is influencing foraging habitat quality. This conclusion is supported by difference detected in the birds' foraging behaviour, as birds in remnants foraged predominantly on the ground, while in restored areas birds were frequently observed collecting prey items from vegetation.

Aside from changing their foraging behaviour, the birds were also found to rarely display breeding related behaviours while in restored habitat. This mimicked a significant difference in juvenile robin population density between restored and remnant patches during the breeding season. As such, it appears robins readily use restored areas for feeding resources, but remnants remain a crucial component of their functional habitat requirements, providing important breeding habitat.

Behaviour was found to be a useful tool in explaining and verifying measured differences in habitat quality, and in this case, could easily be incorporated into pre-existing fauna monitoring programs. Robins weren't found to be a suitable indicator species for the bird community, and given the small species pool on the island, management may need to consider all species of conservation significance separately.

Acknowledgements

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Preface

This manuscript has been written in the form of a cohesive single document to be presented for the degree of Doctor of Philosophy at Edith Cowan University. It is organised into five chapters, each of which contributes to an overarching narrative that explores a central idea of using animal based metrics to assess restored habitat quality. Each chapter is also written in such a way that they can stand alone, and with minor alterations, I will seek to publish them in peer review journals in the near future.

Chapters one and five are likely to be published as opinion pieces. Neither are reliant on quantitative data, and instead, both discuss the state of restoration ecology as a scientific discipline, and a land management practice. Chapter one takes a broad focus, looking at the innovation history of restoration ecology, and proposes a way to further the field through better integration of fauna into management planning. Chapter five on the other hand, focuses on the woodland management future of Rottnest Island, the specific study site where this research was conducted. This chapter includes a detailed summary of the management history of the Island, as well as recommendations for future restoration efforts. The Rottnest Island Authority (RIA) which manages the Island's restoration program is in the process of developing a woodland management plan. The results of this work will contribute to that management plan, which will shape future restoration management decisions on the Island.

The Rottnest Island woodland bird community, with a special focus on the red-capped robin *Petroica goodenovii,* is assessed in Chapter two. This work builds on previous work on the Island's avian community, and was designed in such a way as to be useful in conjunction with Birdlife Australia's ongoing Rottnest bush bird monitoring program. While the focus of this chapter is predominantly centred around the avian community on Rottnest, the chapter also explores the value and limitations of single-species/surrogate-species type studies in addressing the needs of the larger community, which is a concept that may have relevance internationally.

Chapter three contains the first formal assessment of the response of terrestrial invertebrates to woodland restoration efforts on the Island. Invertebrates were assessed as they relate to food resource availability for insectivorous birds like the red-capped robin. I intend to seek publication of this work as both an inventory of orders encountered on the

island, and as an assessment of the effects the woodland restoration program is having on invertebrate assembly.

The value of integrating animal behaviour assessment data with other forms of ecological assessments, such as biodiversity indices, habitat factor analysis and resource availability estimates, is demonstrated in Chapter four. This chapter provides a practical demonstration of the key ideas discussed in Chapter one. The key findings of Chapter four are then used to develop Chapter five's conclusions relating to how a pre-existing state-and-transition model for the Island could be improved through the inclusion of animal behaviour assessments.

As is the nature with many ecological studies, this project encountered a number of unanticipated complications. Some of these complications required fundamental changes in experimental design, while others were solved through minor adjustments to equipment or data gathering procedures. Where deemed relevant, those amendments to the experimental design are outlined in Appendices A and B.

Chapter 1: A critique on the merits of incorporating animal behaviour into restoration ecology

1.1 Introduction

Ecological restoration efforts typically focus on areas that have experienced degradation, damage or destruction, and aim to mitigate or reverse those processes and, thus, increase the resilience of biodiversity (Hilderbrand, Watts, & Randle, 2005; Wortley, Hero, & Howes, 2013). Restoration is achieved by accelerating the successional processes of plant and animal communities after a disturbance, so that biodiversity and ecological processes are returned (Hobbs & Harris, 2001; Brudvig, 2011).

An expanding human population and an economic system reliant on constant growth have resulted in many ecosystems becoming threatened. Consequently, restoration has become an increasingly important tool for protecting threatened ecosystems, and the biodiversity they contain (Hobbs & Harris, 2001; Hilderbrand *et al.*, 2005; Brudvig, 2011). Restoration is a difficult practice and, unfortunately, many restoration efforts have failed to deliver the expected results (Hobbs & Harris, 2001; Choi, 2007; Munro *et al.*, 2012; Perring *et al.*, 2015; McDonald, Jonson, & Dixon, 2016). Consequently, the science and practice of how best to plan, implement, monitor and adapt restoration efforts is still an active and rapidly developing field of study and one of central importance. This chapter first outlines the recent progress and current limitations of the scientific field of restoration ecology. I then discuss the prospect of evaluating and improving restoration outcomes by incorporating animal behaviour, an under-utilised indicator of ecological processes and measure of habitat quality, into a pre-existing management tool.

1.2 A history of innovation: How restoration ecology has improved through time

At the beginning of the twenty first century, the field of restoration ecology as a scientific discipline was no more than 20 years old (Jordan and Lubick, 2011). Problems with inappropriate goal setting, and unrealistic promises were already becoming apparent (Hobbs and Norton, 1996; Hobbs & Harris, 2001; Hilderbrand *et al.*, 2005; Hayward 2012). The

reasons for these problems can be attributed to a number of causes, such as: the extent of disturbance having breached one or more ecological thresholds; poorly defined and unrealistic targets and goals; a lack of adequate monitoring as the restoration developed; and insufficient scientific knowledge on ecological processes associated with restoration. These factors resulted in an overreliance on ad-hoc management without adequate understanding of the implications of many management decisions (Bash & Ryan, 2002; Miller & Hobbs, 2007; McDonald & Williams, 2009; Parkes *et al.*, 2012). A consequence of these decisions has been the creation of numerous hybrid and novel ecosystems containing unusual species assemblages with non-traditional interactions (Williams & Jackson, 2007; Hobbs, Higgs, & Harris, 2009).

Despite the failures of many restoration programs to meet expectations, land clearing and other activities that degrade habitat frequently use the promise of effective restoration management to gain approval for developments that would otherwise not be seen as acceptable (Hilderbrand *et al.*, 2005). The assumption that restoration can completely reverse damage and return ecosystems to some idealised harmonious state is unrealistic, and has been described as potentially harmful when used to guide conservation policy (Hobbs *et al.*, 2010; Hobbs *et al.*, 2011). Hobbs *et al.* (2011) argued that restoration is better seen as a form of ecosystem intervention that can be used within a conservation framework to adjust the trajectory an area is moving along.

Oversights in restoration projects can take a long time to become apparent, as some features don't develop naturally for decades or even centuries (Craig *et al*., 2012; Van Andel & Aronson, 2012). Given the long timescale required for restored areas to mature, and the relative infancy of restoration ecology as a scientific field, the guiding principles and conceptual models still see regular revisions (Hobbs & Harris, 2001; Hilderbrand *et al.*, 2005; Jordan & Lubick, 2012; Higgs, 2012a; Higgs, 2012b; Hobbs, Higgs, & Hall, 2013; Higgs *et al.*, 2014). Unfortunately, the destructive processes that are currently applying pressure to ecological systems across the world will not wait for the science to catch up. This means that most research is conducted at a local scale in an ad-hoc manner in conjunction with the destructive processes, or in a post-hoc manner in degraded areas, where only limited information is available about the pre-disturbance state (Hilderbrand *et al.*, 2005).

Through the study of successional processes associated with restoration, it became apparent that simply returning plant species without considering the return of structure or ecological processes was not enough (Hilderbrand *et al.*, 2005; Lindenmayer *et al.*, 2010; Hobbs *et al.*, 2011). Numerous restoration ecologists have argued that management of restoration needs to move beyond simply revegetating (McAlpine *et al.*, 2016) and assuming that animals will recolonise as the vegetation matures; this is the basis of the 'Field of Dreams' hypothesis, which states that 'if you build it they will come' (Palmer, Ambrose, & Poff, 1997). We now understand that failure to adequately return key habitat characteristics and processes, can inadvertently lead to the creation of habitat filters (Martin *et al.*, 2004; Hilderbrand *et al.*, 2005; Kanowski *et al.*, 2006; Hobbs *et al.*, 2009; Craig *et al.*, 2012). Habitat filters restrict species from recolonising an area, which can have profound ecosystem-wide consequences when those species fulfilled an important role within the system (e.g. the loss of pollinator services and changes to seed dispersal [Caves *et al.*, 2013; Ritchie *et al*., 2017]). For this reason, there has been a growing awareness of the need to improve restoration management to accommodate the faunal component of biodiversity (Schier & Needleman, 2009).

In a recent review article, McAlpine *et al.* (2016) addressed the issue of restoration projects frequently being undertaken with an overly narrow focus. The authors argued that many projects have used short-term performance monitoring strategies that assess plant establishment and diversity, rather than longer-term goals like structure, regeneration, self-sufficiency, and fauna habitat use, which are rarely assessed. Similar conclusions were reached a decade earlier in a meta-analysis, conducted by Ruiz-Jaen and Mitchell Aide (2005), on how restoration success is being measured. While the majority of studies used one or more of the following three general categories: diversity; vegetation structure; and ecological processes (Ruiz-Jaen & Mitchell Aide, 2005). Diversity measures were found to be by far the most common measure, predominantly flora richness surveys, with arthropod richness being the second most commonly used group. It was also found that most studies measured either flora or fauna, but that it was rare for restoration studies to measure both. This is likely a result of expertise being divided along taxonomic lines (Fraser *et al.*, 2017).

1.3 Understanding and explaining successional processes in restoration

Determining whether or not a restored area is developing towards a desired state can be complicated. State-and-transition models (STM) can be useful in providing intuitive depictions of restoration development that commonly relate to vegetation and habitat condition (Stringham, Krueger, & Shaver, 2003; Bestelmeyer *et al.*, 2004; Rumpff *et al.*, 2011). By defining a set of desirable and deviated states that may occur as the restored area develops, managers are able to fine tune management procedures in response to the area's regenerative progress (see Figure 1.1 for an example STM). Unfortunately most STMs are predominantly flora focused, and tend to exclude fauna from consideration (Fraser *et al.*, 2017). Grant (2006) developed one of the earliest STMs for restoration. The model described a series of processes surrounding forest restoration in retired bauxite mine pits in the Jarrah forests of Western Australia. Bauxite mining uses open cut techniques, which involve clear cutting the forest and breaking through the cap rock to expose the alumina rich bauxite. The mining process results in pits that are between 8-10 meters deep, and up to 40 hectares in size (Koch, 2007). Consequently, restoration begins with soil preparation, where the pit floor is ripped, topsoil (enriched with a seed bank and propagules) is used to fill the pits, and the edges are smoothed for proper drainage (Koch, 2007). This kind of restoration aims to accelerate the recolonisation of native flora and fauna through regrowing the vegetation in the pit. Periodic, long-term monitoring of the flora and fauna provides insight into the development of the restored area over time.

Grant's (2006) STM was derived from vegetation successional processes, and identified a set of desirable and deviated successional states and the factors that caused transitions between those states. The model proved to be an effective method for identifying a number of potential biotic and abiotic issues as they emerged, and provided management suggestions to counteract those issues. However, the model was fairly limited, as the definitions of desired and deviated states needed to be severely simplified from their original design (Grant, 2006). Consequently, a large portion of collected data were not usable. In addition, a major limitation of the model was that fauna was completely overlooked.

1.4 Fitting animals into existing models

While fauna is rarely used in restoration modelling studies, a few notable examples have effectively incorporated animal-based metrics into restoration management plans (e.g. Bosire *et al.*, 2008; Howes, Maron, & McAlpine, 2010; Fraser *et al.*, 2017). Bosire

incorporated assessments of structure, regeneration, biomass, and composition of restored mangroves, as well as biodiversity measures of various vertebrate and invertebrate taxonomic groups to gain a more complete perspective on the extent to which functionally of the restored mangrove area had been returned. Aquatic invertebrates are arguably a crucial component of biological monitoring in many aquatic systems, where biological integrity is assessed using well developed procedures (Resh, Norris, & Barbour, 1995; Fore, Karr, & Wisseman, 1996). Terrestrial invertebrates have also been shown to be a valuable assessment tool for assessing restoration success in the resource mining sector (e.g. Dunger's (1989) work on German coal mine dumps; Stannard's (1967) work on strip-mined land in Northern America; Hutson's (1980) work on reclaimed coal pits in England; and Majer's (1983) work on open cut bauxite mines in Australia. Ant monitoring was developed to assess restoration success following mining (Majer, 1983)).

Craig *et al.* (2015) investigated the relationship between Grant's (2006) five desirable and five deviated habitat states and the avian community. The study identified no discernible relationship between avian successional patterns and the states described by Grant. This was attributed to the model's design not being based on suitable ecological processes and thresholds, and the desirable and deviated states not being defined by factors that are important to the avian community (Craig *et al.*, 2015). It was concluded that altering the STM to better incorporate faunal successional patterns would improve land managers' ability to identify and address problems affecting faunal recolonisation of restored mine pits.

Howes *et al.* (2010) used a bayesian network (BN) modelling approach to assess the influence of fire (both wild and prescribed) and feral animal grazing on habitat structure, and avian assemblage. The model proved to be useful in identifying causal links between various ecological processes related to both flora and fauna, and offered guidance on how best to plan future ecosystem intervention. This study demonstrated how fauna and flora could be assessed in an integrated manner to produce meaningful data for land managers.

Fraser *et al.* (2017) developed an integrative STM that combined vegetation conditional data with avian species distribution modelling (SDM) to identify the effective restoration options for vegetation and bird species within a constrained budget. By assigning different values to each objective, based on their relative importance to either the birds or the vegetation profile, the model was able to provide land managers with recommendations for how best

to invest time and effort. The model was able to identify key variables that management should focus on, and helped manage 'trade-off' decisions between conflicting objectives, as the floral and faunal needs at times contradicted one another (Fraser *et al.*, 2017).

There are numerous ways to measure restoration development to construct a STM or other successional model. Measures like diversity, community composition, habitat structure, and various ecological processes all have their merits and drawbacks (Ruiz-Jaen & Mitchell Aide, 2005). Diversity measures can be useful but also misleading, as the pre-disturbance state is often ambiguous in degraded areas, disturbance specialists are likely to be overrepresented, and variation in detectability of different species can influence the results (Ruiz-Jaen & Mitchell Aide, 2005; Lindell, 2008). Species composition in restored areas can change dramatically as the area matures. Plant and animal vagility influences the capacity of species to locate and recolonise restored areas. Unfortunately, highly mobile species can often move through areas that provide little to no useful habitat resources, diminishing the value of presence-absence data (Lindell, 2008; Craig *et al.*, 2012). Measures of habitat structure can be useful for classifying habitat types, especially in relation to successional development. Some features change rapidly over the first few decades of development, especially in areas that have frequent fires, while other structures, such as ground logs and tree hollows can take decades to centuries to develop (Fischer & Lindenmayer, 2007; Craig *et al.*, 2012). However, habitat structure assessments can also be problematic as different animals are likely to perceive habitat barriers and features differently to us (Van Dyck, 2012).

Ecological processes provide excellent insight into how multiple ecological features have developed and interact with one another, but require an in-depth understanding of the system, and can therefore be complicated to assess (Reay & Norton, 1999). The merit of integrated approaches that use a range of different assessment measures is that by covering a wide spectrum of characteristics, we are able to gain a more comprehensive and integrated understanding of the system's development, its similarity to the reference state, and its resilience (SER, 2004). Ruiz-Jaen and Mitchell Aide (2005) argued that; while designing restoration that considers the needs of both fauna and flora is likely to be more costly and require a better understanding of ecological processes, it is more likely to result in restoration efforts meeting their biodiversity objectives.

Ecological processes are intrinsically integrative, and can provide information on resilience of restored ecosystems (Ruiz-Jaen & Mitchell Aide, 2005). Animals are crucial components of many ecological processes, yet are often overlooked in assessments of restoration success (Lindell, 2008). Examples of ecological processes facilitated by animals include: hebivory, seed dispersal, pollination, predation and parasitism (Holl, 1998; Donath, Holzel, & Otte, 2003; White *et al.,* 2004). A number of authors have argued that we need to adopt more integrative approaches to restoration assessments, and consider the ecological processes that need to be established if future projects are to succeed where projects in the past have failed (Hobbs & Norton, 1996; Neckles *et al*., 2002; SER, 2004; Lindell, 2008).

Theories such as 'Carbon Copy' (Clements, 1936) and 'Field of Dreams' (Palmer *et al.*, 1997) suggest that a disturbance or degrading activity that removes the vegetation structure previously present in the area will simply interrupt a system's ecological processes temporarily (Hilderbrand *et al.,* 2005). These theories assume that the ecological processes of an area will return along a systematic successional trajectory, back to its original state, through rebuilding the system's vegetation structure (Clements, 1936). There is little evidence to support the premise that restoration efforts achieve desired structure and functions within a shortened time span, especially without continued management as the restored area matures (Simenstad & Thom, 1996; Zedler & Callaway, 1999; Campbell, 2002; Wilkins, Keith, & Adam, 2003). Faunal species that are slow to recolonise often require further assistance through modifying restoration practices to better provide key resources and restart ecological processes (Cristescu, 2011; Craig *et al.,* 2012; Triska *et al.,* 2016). The installation of artificial nest boxes, the addition of ground logs, and the translocation of animals into a restored area, are all examples of active management that can be used to aid faunal recolonisation of restored habitats.

Habitat filters and ecological traps are clear examples of where improper restoration efforts can create new problems for species. While assessment of presence alone may indicate that the area provides viable habitat, further inquiry may demonstrate that the species found within the area are unlikely to survive or reproduce (Lindell, 2008). An ecological trap occurs when animals occupy sub-optimal areas but at low fitness, and are therefore unable to maintain a stable population without a nearby source population to subsidise following generations (Robertson & Hutto, 2006). Although they can be difficult to identify, ecological traps are thought to be common in human-modified landscapes, including restored habitats (Battin, 2004). Identifying an ecological trap can be done through measures of population

replacement, such as monitoring nest success, predation rates, and behavioural assessments (Winter *et al.*, 2006; Pidgeon, Radeloff, & Mathews, 2006). By studying parental behaviour, nestling success rate, and the reasons for nestling failure, it is possible to gain insight into the reasons why some areas have higher success rates than others. In a study on blue tits (*Cyanistes caeruleus*), the distance parents travelled to find food was found to influence fledgling success rates, which explained the variation in reproductive success between high quality deciduous woodland, and low quality coniferous woodland (Stauss, Burkhardt, & Tomiuk, 2005). The study used behavioural analysis to identify variation in habitat quality, and explain the mechanisms behind that variation.

Habitat filters are a conceptual model used to describe the presence or absence of functional habitat traits that make an area unsuitable for certain species (Cornwell & Ackerly, 2009; Craig *et al.*, 2012). Examples of potential filters include the absence of coarse woody debris (CWD) and tree hollows that are slow-developing habitat resources, whose absence from an area may exclude species that are dependent on those resources (Vesk *et al.*, 2008). While filters are generally seen as a negative or unwanted habitat characteristic, some filters can increase an ecosystem's resilience; by making it difficult for invasive species to become established (Funk *et al.,* 2008). When developing a STM that uses animal-based metrics in its assessment of desired and deviated states, unwanted filters could be used to classify areas as deviated, and through the removal of those filters, the area may transition back to the desired state.

In a study on the factors determining how successful river restoration efforts were at returning benthic invertebrate assemblages, it was determined that the presence of a potential source population of the desired species in the surrounding area was necessary (Sundermann, Stoll & Hasse, 2011). The study found that the source populations needed to exist within a 0-5 km ring around the restored area, and that source populations beyond 5km from the restored area had a relatively limited role in the recolonisation of restored areas. This study highlights the importance of understanding the vagility of faunal groups the restoration effort is intended to help, as overly large distances between source populations and restored areas can act as a filter for recolonisation.

1.5 Use of animal behaviour as an assessment tool

Ecological processes can be assessed by observing how specific species use the habitat they

occupy (Lindell, 2008). Various behavioural patterns have been shown to be indicative of habitat quality (e.g. Vaughan, Jones, & Harris, 1996; Johnson *et al.*, 2006; Stenberg & Persson, 2006), and the rate and/or outcome of certain behaviours has been linked with individual fitness, via measures based on their contribution to the next generation (Alcock, 2009). Restoration assessments that use animal behaviour to assess the availability of important habitat features, as well as evidence of ecological processes having been restored, would therefore capture considerably more information, than presence/absence data alone (Lindell, 2008).

A number of ethological papers have compared animal behaviour in restored areas to comparable reference habitats, demonstrating the potential of animal behaviour based metrics to identify important resources and processes affecting the species studied. Commonly assessed behaviours include: microhabitat selection (Gabbe, Robinson, & Brawn, 2002; Moore, 2013); Vagility (Maina & Howe, 2000); foraging strategies (Adamik & Kornan, 2004; Whelan & Jedlicka, 2007); and breeding behaviour (Berg, 2002; Berg, Lindberg, & Källebrink, 2002; Bellingham *et al.*, 2010). An example of a study technique using foraging strategy is 'giving up density' (GUD). GUD measures the level of risk animals perceive when foraging in specific patches, based on their willingness to exploit a food resource that yields depleting returns for search effort. GUD can be used to assess habitat quality through assessment of predation pressure and food availability (Brown, 1988; Jacob & Brown, 2000; Persson & Stenberg, 2006; Doherty, Davis, & van Etten, 2015). Studies like this are valuable sources of information for restoration practitioners wanting to develop STMs or other comparable models that incorporate measures of ecological processes relevant to animals and to create more integrative management plans.

For a STM to effectively represent vegetation and faunal succession, the desired and deviated states need to reflect ecological processes and thresholds relevant to both taxonomic groups (Craig *et al.*, 2015). A hypothetical schematic STM that includes abiotic, floral and faunal habitat requirements has been developed for this study, and can be seen in Figure 1.1. Stages S1-S4 represent measurable states within the desirable ecosystem parameters (this is a simplified model; actual models are likely to have far more stages). Conceptually, a restored area should move from S1 through to S4 through a combination of passively occurring successional processes, and active restoration interventions. Deviated states occur when ecosystem parameters go beyond the desired range denoted by the deviated boundaries. A well-designed STM should clearly define each developmental stage

(S1:S4), each deviated state (D1:D6), and the pathways between each stage, including management solutions for transitioning deviated states back into the desirable range (Yates & Hobbs, 1997).

The thresholds (T1:T3) are an adaptation of Grant's (2006) model for post bauxite mining mine pits, which was originally derived from Whisenant's (1999) model. To pass through the abiotic threshold, a number of management manipulations were required as the pit topography had to be manipulated. This involved return of the overburden and topsoil. Contour ripping was then used to reduce the risk of erosion, encourage water infiltration, relieve soil compaction, and encourage vegetation growth. Once these manipulations had been completed, the area passed through the abiotic threshold into the T2 stage of development. This stage involved manipulation of vegetation such that the flora threshold requirements could be met. Once the vegetation characteristics had been adequately restored, the model depicted in Figure 1.1 goes beyond Grant's (2006) model to include a threshold dedicated to fauna requirements.

This third tier (T3) involves the removal of habitat filters or factors that cause the area to act like an ecological trap. It is in this stage that animal behavioural metrics may be of greatest use in determining whether ecological processes have been restored, and quantifying the relative quality of the restored habitat for certain species. It should be noted that while this is the stage that appears to be the most logical place to assess animal behaviour, consideration of the needs of animals should be in place from the outset of the STM, as factors established during T1 may influence an area's capacity to pass through the T3 threshold. Additionally, faunal recolonisation, especially by disturbance specialists, is likely to occur early on in the restoration development, meaning faunal assessments may be of value at all stages of development.
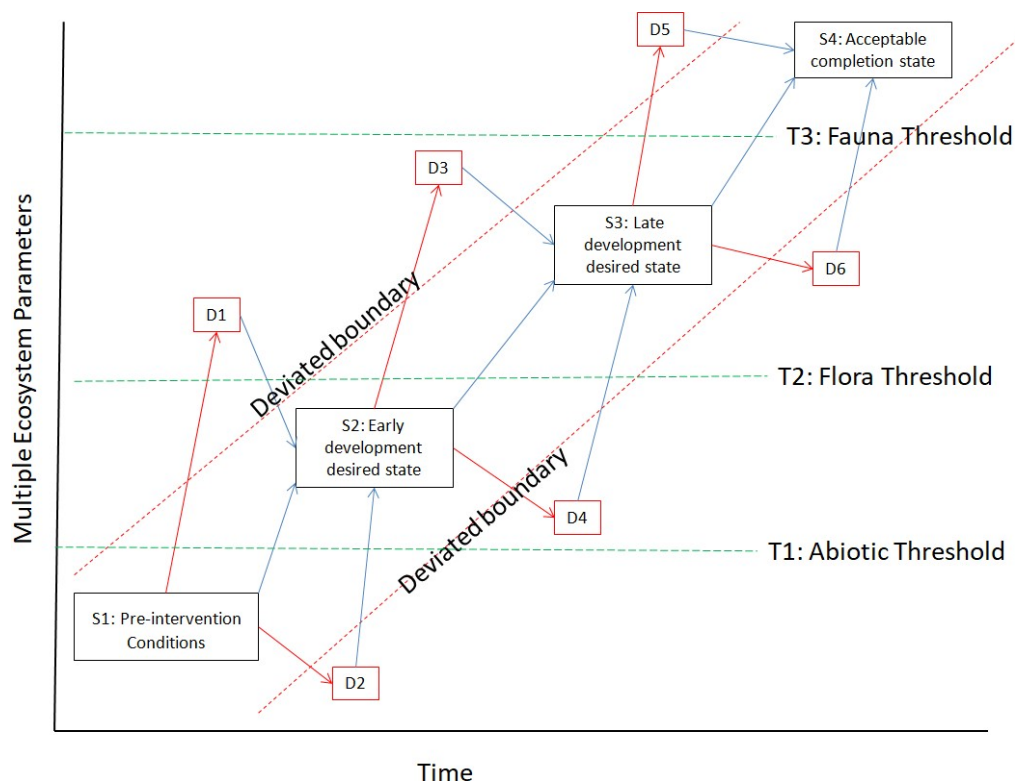
Figure 1.1: State-and-transition model depicting the progression from pre - intervention conditions through to acceptable completion state. Deviated boundaries demonstrate the limits of the desired range of ecosystem parameters for each developmental stage (defined through deviated states D1:D6). Red arrows between stages are transitional forces that can deviate area from the desired trajectory. Blue arrows signify management actions or passive developments that move area towards the desired end stage. The thresholds T1:T3 signify habitat requirements that must be met before the area can move into the next developmental phase. This model was derived from Yates and Hobbs' (1997) and Grant's (2006) STMs.

Our need to improve the science of restoration is more relevant now than ever before. Restoration ecology appears to be moving towards more integrative designs, where spatial modelling, bayesian networks and STMs are seeing increased use. It has been argued that there needs to be greater consideration of animals with regards to restoration planning and management, but there are a number of logistical challenges surrounding how best to incorporate animals. Management decisions can either facilitate or impede the recovery of important processes, which relate directly to how animals behave in an area (Lindell, 2008). While animal behaviour is currently underutilised, numerous case studies have demonstrated its value in filling in the gaps left by more traditional restoration monitoring techniques (e.g., Holl 1998; Brusati, DuBowy, & Lacher, 2001; Baguette & Van Dyck, 2007; Bennett & Hale, 2014). The inclusion of animal behaviour based metrics into restoration assessment procedures would allow us to better understand the ecological processes at work.

The challenge will be to find robust ways to quantify animal behaviours in ways that can be incorporated into state-in-transition models that can exist alongside existing restoration assessment and management targets. To do this, there needs to be: 1) better collaboration between botanical and zoological advocates: 2) more integrative approaches that consider ecological processes and functional traits of both animals and plants: 3) an awareness of the limitations of presence/absence type data, and: 4) consideration of how behavioural data can fill knowledge gaps allowing us to better repair ecosystem functionality. The following chapters will explore ways in which animal behaviour can be used to quantify habitat quality of restored, degraded and remnant areas. The merits of these survey techniques are then compared within the context of designing a STM to improve restoration outcomes.

Chapter 2: Limitations of surrogate species models: A case study using the red-capped robin to predict woodland bird community composition on Rottnest Island.

2.1 Introduction

In Chapter one, I discussed how animals are an important component of the world's ecosystems, and argued that restoration outcomes could be improved by better incorporating the needs of animals into restoration management plans. How best to do this is still somewhat unclear. Lindell (2008) raised concerns that animals have been largely overlooked during the planning and monitoring of restoration projects, and highlighted some of the benefits of animal behaviour studies in improving restoration outcomes. While the arguments put forward by Lindell are well made, it is important to consider the limitations of animal behaviour metrics, as there are a number of challenges associated with using species-level research in restoration ecology.

Due to the difficulty in planning management around the needs of every species within a system, a number of conceptual models relating to simplifying species management have been proposed over the years that offer shortcuts for biodiversity monitoring and maintenance. Various surrogate species models such as the umbrella species (Carroll, Noss & Paquet, 2001), indicator species (Rose, 1999), keystone species (Bruinderink, 2003), and focal species (Lambeck, 1997) have been put forward. Each of these models have substantial differences, and criteria for selecting appropriate species. Population indicator species tend to be most effective when they have a rapid rate of reproduction, are resident and often restricted to the area in question, and the factors influencing their population size are well understood (Caro & O'Doherty, 1999). Focal species on the other hand are generally linked with a single threatening process like habitat fragmentation, and their response to the threatening process must be understood, and it must also mirror the response of other taxonomic groups the focal species is to act as a surrogate for (Lambeck, 1997).

Confusion surrounding the validity and interchangeability of these models has called into question the validity of surrogate species models, leading to substantial criticism, and the mis-management of numerous conservation efforts worldwide (Caro & O'Doherty, 1999;

Lindenmayer *et al.,* 2002; Wiens *et al.*, 2008; Caro, 2010; Watts *et al.*, 2010). Lindenmayer *et al.* (2002) urges caution with regard to the implementation of the focal species surrogate model, as the assumptions regarding faunal response to threatening processes need to be carefully considered, as there is considerable evidence that similar species and groups of species will respond to threatening processes in vastly different ways (e.g. Robinson *et al.*, 1992; Gascon *et al.*, 1999). In addition, it can be very difficult to identify which species is likely to be the most affected by a threatening process, which further complicates the task of identifying a suitable indicator species.

While the selection criteria and implications of each surrogate species model are subtly different, the key premise behind each of these models is the same. That premise is as follows: by managing or monitoring the needs of a chosen sub-set of species found in an area, the needs of a larger pool of other species inhabiting the area will also be met (Caro, 2010). While it can be said that natural systems are fiendishly complicated, there is an acute need for action in the wake of the biodiversity crisis currently underway. Political pressures often require rapid decisions and there is often insufficient time and funding to complete a more comprehensive management plan (Singh, Raghubansh, & Singh, 2002; Caro, 2010). Hence, the various surrogate species models offer an attractive shortcut, simplifying the task and reducing the cost of management. Unfortunately, there is considerable evidence that rejects their premise.

All animals have a number of resource requirements that are needed for that species to persist in an area (Forman, 1995). Within a single area, many species may share a subset of common resource requirements from the total resource pool available, but it is unlikely that any species will share all the resource requirements of every other species in the system (Eycott, 2007). Moreover, the specific requirements each species has in relation to each resource may be subtly different. Assessing each species' specific needs in relation to one another would be extremely complex and labour intensive, making it likely to be unfeasible (Lindenmayer *et al.*, 2002). This is the fundamental issue with models like the umbrella, indicator, keystone and focal species. Species used in these models are often large mammals and birds, but there is little evidence to support the claim that these groups will be representative of the needs of other taxonomic groups, or even other species within these groups (Roberge & Angelstam, 2004). Thus, caution needs to be taken when using any of these models, as they can only be extrapolated to a few other species within the system, if at all.

While the use of surrogate species management isn't without its criticism (e.g. Caro & O'Doherty, 1999; Lindenmayer *et al.*, 2002; Roberge & Angelstam, 2004), we have yet to develop a robust, and usable management tool that is infallible. Grouping species according to their susceptibility to particular causes of decline has logical value for land-managers aiming to prevent declines. Lambeck's (1997) focal species approach (FSA) was an evolution of the single-species models like the umbrella species model that came before it, and addressed some of the issues of its predecessors. Generic focal species (GFS) is a recent adaptation of the FSA, that uses population data modeling from a suite of different species to create a single theoretical species that has the habitat requirements, dispersal capabilities, or susceptibility to habitat alteration as all species within the chosen suite (Eycott, 2007; Watts *et al.*, 2010; Oliver *et al.*, 2012). These revisions of surrogate species design demonstrate an awareness of the limitations of past models, which will hopefully continue to be refined and improved in the future. For now, in the absence of an infallible model, the precautionary principle advises that caution be used when applying simplified principles to processes that aren't yet fully understood (Kriebel *et al.*, 2001). Currently, it appears that surrogate species models are best used in combination with other management tools, such as community level studies, and the study of ecological interactions and processes (Lindenmayer *et al.*, 2002; Eycott, 2007; Lindell, 2008; Caro, 2010; Watts *et al.*, 2010).

Within the field of restoration ecology, woodland birds have been a focal point for conservation efforts. This can be attributed to the evidence that woodland bird assemblages have severely declined across much of the tropics, sub tropics and temperate zones worldwide (Birdlife International, 2008; Mac Nally *et al.,* 2009; Watson, 2011). Local extinctions and range contractions have been well documented in Australia, Europe and North America (Ford *et al.*, 2001; Donald *et al.,* 2006; Murphy, 2003; Fuller *et al.*, 2007; Watson, 2011). The primary drivers behind these losses appear to be habitat loss and degradation (Mac Nally *et al.*, 2009). The severity of these concerns has resulted in woodland birds being recognised as a global conservation priority (BirdLife International, 2008). Restoration ecology offers an opportunity to repair lost and degraded habitats, which, if done correctly, may mitigate further declines in woodland bird communities.

Animal behaviour studies are intrinsically species focussed, typically with either a single target species, or the interaction between a few different species that are related though

interactions like predator-prey relationships or competition. While there are numerous cases of animal behaviour studies providing valuable species-level information for restoration practitioners (e.g. Lindell, 2008; Fink *et al.*, 2009; Bennett *et al.*, 2012), the results of animal behaviour studies need to be considered within the context of the larger community. Single-species studies may generate results that conflict with community level studies, which will create difficult ethical decisions for land managers.

Animal behaviour studies can provide insight into the relative habitat quality offered by different habitat patches in a number of ways. Animal behaviour studies can show the effects of patch size and shape on bird survival rates (e.g. Major, Christie & Gowing, 2001), they can identify reasons for high and low fledgling rates in different habitat types (e.g. Stauss *et al.*, 2005), and they can identify key resources, such as food and shelter, different habitat patches provide for that animal (Benton, Vickery, & Wilson, 2003).

Rottnest Island is home to a population of red-capped robins *Petroica goodenovii*, a small (7-9g), Australasian robin from the Petroicidae family, that is unrelated to the old world European or American *Turdus* robin (Boles, 1988). Red-capped robins are widely distributed across much of Australia. Their range extends from the southern most parts of the Kimberley, and Cape York (roughly 20°S) to the southern coasts of the continent, and a number of small offshore islands (Higgins *et al.*, 2001). Despite their wide distribution, red-capped robins have been identified as a declining woodland species in a number of studies (Reid, 1999; Razeng & Watson, 2012). Additionally, a number of studies have demonstrated that these robins are sensitive to habitat disturbances such as fragmentation (Radford *et al.*, 2004; Major *et al.*, 1999), and are able to change their foraging behaviour under certain conditions (Antos, Bennett & White, 2008; Recher & Davis, 2002).

The red-capped robins on Rottnest typically occupy woodland areas, which are made up of Rottnest Island teatree *Melaleuca lanceolata* and pine *Callitris presissii* (Saunders & de Rebeira, 2009). This woodland type is classified as a threatened ecological community in Western Australia, as it has suffered severe declines across the Swan Coastal Plain (Keighery *et al.*, 2003; Winn, 2008). While the robin population on the Island currently appears to be increasing, they are uncommon on the adjacent mainland Swan Coastal Plain.

The recent growth in the robin population on the Island has been attributed to the restoration efforts conducted by the Rottnest Island Authority (RIA) since 1963 (Mather, 2010). Rottnest Island has a long history of land clearing and frequent burning for agriculture and urban development (Winn, 2008). This has resulted in considerable changes to the biodiversity composition, with a number of local extinctions, numerous declines, and the invasion and colonisation of a number of species, as the availability of different habitat types shifted (Saunders & de Rebeira, 2009; Stevenson, 2011). In an effort to mediate these changes, the RIA has invested considerable resources into their woodland restoration program.

Red-capped robins have been recognized as being a population of conservation importance on Rottnest Island due to call differences from mainland populations (Baker, Baker, & Baker, 2003; Saunders & de Rebeira, 2009; Mather, 2010; Stevenson, 2011). Other bird species on the island that have been identified as significant based on their differences to mainland populations include: the western gerygone *Gerygone fusca*, the singing honeyeater *Lichenostomus virescens*, and the golden whistler *Pachycephala occidentalis* (Saunders & de Rebeira, 2009). All four of the listed species are woodland dependant, insectivorous birds. Post-European colonisation in the 1800s saw widespread clearing of woodland habitat on the island, which has since displayed very poor rates of natural regeneration (Winn, 2008). This highlights the importance of the woodland restoration program, which aims to increase the resilience of the island's biodiversity through increasing the availability of usable habitat.

Given that a number of studies of red-capped robins on the mainland have found that robins are vulnerable to habitat fragmentation (e.g. Major *et al.,* 1999) and can alter their foraging behaviour in response to habitat conditions (e.g. Antos, Bennett & White, 2008), it has been suggested that they may be a suitable indicator species (Mather, 2010), or at least act as a surrogate species for identifying ecological thresholds (Radford *et al.*, 2004). Data from Birdlife Australia's Bush Bird monitoring program (Mather, 2010) and Polson-Brown's (2012) honours thesis both revealed that robin density was highest in woodland remnants and some restored sites, but that robins were completely absent from other woodland restoration sites. This suggests that robins may be experiencing some form of habitat filter. This evidence that robins are sensitive to variation in woodland restoration design suggests that they may be a useful indicator species for the Island. This chapter will explore the relationship between the robin community on the island, and the rest of the woodland bird

community. Habitat assessments relating to structural, temporal and spatial variables are also compared with robin density estimates to identify variables that best explain variation in estimated robin density. These variables are then compared with density estimates of other species of conservation significance, to determine whether high quality robin habitat, is also likely to be high quality habitat for other priority species.

2.2 Key Questions:

1. Are robins a good indicator of avian assemblage on Rottnest Island?

2. What limitations exist when comparing single species data with community data?

3. Which habitat structure variables best predict variation in estimated robin density?

4. Is high quality robin habitat also high quality habitat for the Island's other priority woodland bird species?

2.3 Methods

2.3.1 Study site

Rottnest Island (32°1 0 S, 115°500 E) is situated approximately 20 km west of Fremantle in the south-west of Western Australia (Figure 2.1). Rottnest is classified as an 'A-Class Reserve', declared under the *Land Act 1993* and gazetted for public recreation since 1917 (RIA, 2014). Rottnest covers an area of around 1900 ha, with 200 ha of classified 'settlement' area. The island receives between 350 and 700 mm of rainfall annually, with 80% falling between May and October (Australian Bureau of Meteorology, 2017). The data for this chapter was collected from restored and remnant woodland habitat exclusively, as defined by Winn (2008).

Figure 2.1: Image of Rottnest Island, showing restored and remnant study sites. Original photo courtesy of Landgate.

There are six major terrestrial habitats on the island (Winn, 2008):

1) Coastal habitat (sandy limestone beaches and rocky cliffs);

2) Salt lakes (a sequence of vegetation zones formed through progressive decline in inundation periods);

3) Brackish swamps and freshwater pools (swamp deposits in inter-dune depressions);

4) Woodland areas that contain a combination of *Callitris preissii, Melaleuca lanceolata* and *Acacia rostellifera* in a low (<10m tall) forest to open woodland formation with little to no understory (this habitat type was once the most common habitat type on the island, but overly frequent fires and human induced disturbances have left just a few remnants covering roughly 3% of the island). Restoration efforts by the RIA have resulted in the creation of a number of restored woodland patches, the largest being almost 20 ha (based on GIS data maintained by the RIA);

5) Areas that were once woodlands, but have been inadvertently converted into low grassy heath with few shrubs, that currently covers 60% of the island; and

6) Human-developed areas on the Island, which contain a high proportion of introduced flora species (Buchanan, 1994; Playford, Leech, & Kendrick, 1977; Rippey & Rowland, 1995).

Post-European settlement saw large areas of woodland cleared and harvested for roads, railways, buildings, and agriculture (Somerville, 1949/1976). The frequency of fires increased considerably, as fire was used as a tool to clear land and hunt quokkas *Setonix brachyurus*, a small (2.5 - 5kg) marsupial native to the Island (Pen & Green, 1983; Dodd, 1994a; Dodd, 1994b). The shooting of quokkas was outlawed in 1917, however shooting allegedly continued until 1933 when firearms were prohibited on the island (Storr, 1963). From this point the quokka population increased dramatically, and by 1941 there were reports of overgrazing of crops and *Acacia rostellifera* scrub was becoming increasingly scarce (Pen & Green, 1983). Between 1919 and 1941 quokka herbivory converted a total of 800 ha of *Acacia rostellifera* scrub into grassy heath made up of *Acanthocarpus preissii* and *Austrostipa flavescens* (Storr *et al.*, 1959). Then in 1955, a massive fire burnt two thirds of the island during the height of summer (Rippey & Hobbs, 2003). In the years following the fire, Storr (1963) recorded abnormally high numbers of quokkas in burnt areas that had previously contained acacia scrubs. It is thought that increased herbivory by quokkas, coupled with the intensity of the fire, as well as two subsequent fires in 1974 and 1997 facilitated in the conversion of woodland and scrub areas to the *Acanthocarpus preissii – Austrostipa flavescens* heath, which now covers the vast majority of the island (Rippey & Hobbs, 2003).

The remaining woodland habitat is now scattered across the Island in small patches of relic remnants and restored areas (Winn, 2008). The loss of woodland habitat and creation of large heathland areas have been attributed as direct causes of the local extinction on the Island of two woodland bird species (rufous whistler *Pachycephala rufiventris* and the brush bronzewing *Phaps elegans* (Storr, 1963; Saunders & de Rebeira, 1985)*.* Additionally, it has facilitated the colonization of a number of other bird species, such as the banded lapwing *Vanellus tricolor* (Storr, 1963; Serventy & Whittell, 1976), laughing dove *Streptopelia senegalensis*, spotted dove *Streptopelia chinensis* (Storr, 1963), rainbow bee-eater *Merops ornatus* (Saunders & de Rebeira, 1985), Australian magpie (Serventy & Whittell, 1976), and Australian raven *Corvus coronoides* (Stevenson, 2011). These shifts in the avian assemblage may have reduced the capacity to which robins could serve as an indicator on the island. The species that were most sensitive to woodland habitat loss appear to have already been lost, and many of the species that are now common on the island can be characterised as disturbance specialists.

2.3.2 Habitat sampling technique

A total of 24 survey sites in Rottnest's woodland areas were selected for this study. The sites were previously used as part of a woodland bird community survey by Birdlife Australia (Mather, 2010). Many of the sites contain a mixture of both remnant and restored patches, and there has been considerable variation in the restoration techniques used in the past, meaning that a space-for-time substitution (Pickett, 1989) was not suitable. However most sites could be differentiated as either remnant (woodland containing native remnants of *Melaleuca* and/or *Callitris* but some with some infill restoration), or restoration (areas that were completely cleared or converted to heathland type habitat at one stage, but have been revegetated with *Melaleuca* and/or *Callitris* stands). A number of vegetation and habitat variables were measured at each site. The variables selected were all considered to be potentially relevant to woodland birds, and generally relate to the birds feeding ecology. Moderate-resolution imaging spectroradiometer (*MODIS*) data were used for measures involving vegetation height and density. The data were collected by Landgate SRSS, as part of its Urban Monitoring Program (2009). Other data obtained from the GIS database maintained by the RIA included restoration age, fire history, and size, shape and isolation of study sites. For data that were not available using MODIS and the GIS database, four randomly located quadrats (5 m x 5 m) were surveyed in each of the 24 sites. Data collected within quadrats included: ground substrate (measured based on percentage cover of leaf litter, bare ground and vegetation); presence of CWD (number of pieces with diameter >2 cm, length >30 cm); visibility (average distance measured using a rangefinder with four readings taken towards the 4 corners of the quadrat) and presence of horizontal branching (number of trees with horizontal branches within the quadrat).

2.3.3 Bird Surveys

In February (summer) 2015, one observer (F. Holmes) surveyed each site using the Birdlife Australia standard national bird monitoring 2 ha (100 × 200 m plot) area search method (Barrett *et al.*, 2003). During surveys, each plot was surveyed for 20 minutes and all birds heard or seen in the plot were counted. This survey method was selected, as this was the procedure used by Birdlife Australia in their Rottnest Island bush bird counts (Mather, 2010), which began in 2000, and as of 2018, is still ongoing, By selecting the same survey technique, the data collected for this study could be used by Birdlife in the future. Each site was surveyed three times, with replicate surveys occurring on non-consecutive days. All surveys were conducted within five hours of sunrise and the order in which sites were surveyed was rotated where possible to reduce any bias caused by sampling at different

times of day. As the area search involved traversing the entire plot on foot, and most birds were detected on call, differences in vegetation density was not a major factor in detectability, and distance sampling was not required since the encounter rate, including flushing of birds, is presumed to be equal in all plots. For bird taxonomy, Christidis and Boles (2008) identification guide was used.

During the robin breeding season (September - December) of 2016, a second series of surveys was completed using the same 2 ha plots at each of the 24 sites. In this second series of surveys, only red-capped robins were counted, but extra attention was given to the age and gender of all individuals encountered, as juveniles could be more reliably differentiated from adult females during this season than in summer when they have completed their post juvenile moult. This was done to determine which areas contributed most to recruitment, as well as provide insight into seasonal variation in estimated robin density and distribution on the island between breeding and non-breeding seasons. In an effort to make as many of the birds individually identifiable as possible, bird bands were fitted to as many of the robins as possible prior to the commencement of these surveys. Birds were fitted with a split-colour metal band on one leg, and a non-coloured, metal, Australian Bird and Bat Banding Scheme (ABBBS) band (n = 50). Additionally, as the site already had an active banding project, many of the birds were already fitted with ABBBS bands. For details on the bird banding procedures see Appendix A.

While no formal assessments of home range size or site vigilance was conducted for this study, anecdotal evidence suggests that the birds tend to remain loyal to a single small area. Over the three years of banding and observation of robins on the island, only two individual birds were observed at more than one of the study sites. One was an adult male robin, originally banded in a woodland remnant in July of 2015 that was observed three months later at a nearby restored woodland site approximately 400 m from the site where the bird was originally banded. That same bird was later observed at the site where it was originally banded a month later. The other record of movement between sites was between two sites, which at their closest point are approximately 20 m apart and separated by a road. Both sites are restored woodland, and are approximately the same age (50 years old). The bird was regularly seen in both sites, but this is unsurprising as the distance travelled by the bird is relatively small, and the conditions within both sites were highly similar.

2.4 Statistical Analysis

As bird surveys were conducted over two separate years during different seasons, I first assessed whether overall robin density estimates varied between the two survey periods, and assessed the correlation between robin densities across the 24 survey sites. The summer data set contained only robin density estimates, without any demographic data, but the spring dataset included demographic data relating to age and gender of birds detected. The two datasets were compared using both a Pearson correlation coefficient, and a paired samples t-test (with same sites being paired) to determine whether a significant difference between the two survey periods could be detected, and to assess the correlation between the two datasets. For the spring data, total robin density estimates, as well as adult-only and juvenile-only were included as three separate analyses. These comparisons provided information that was used for generating hypotheses for later chapters, based on variation in robin assembly between breeding and non-breeding periods.

To address the first research question, determining whether robins are a good predictor of avian assemblage on Rottnest Island, I assessed whether estimated robin density was correlated with overall avian assemblage. To do this, I assessed the Pearson correlation coefficient between total estimated robin density during the breeding season, as well as sub-categories of adult-only and juvenile-only robins, and number of robins detected during the summer survey period, against various measures of avian assemblage. Avian assemblage was measured using abundance, species richness, and species evenness (Shannon-Weiner Index). This was done using all birds detected during the community assemblage assessments (n = 22), which included migrants like the rainbow bee-eater *Merops ornatus* and non-woodland-dependant species like the silver gull *Chroicocephalus novaehollandiae*, as well as with a subset of birds that excluded those vagrants and non-woodland dependant species (n = 15). After this analysis, all avian community measures used only the woodland dependant resident birds (excluding red-capped robins).

The estimated robin density measures described above were also compared with the avian assemblage based on dietary guilds using the Pearson correlation. Each bird species was grouped into one of the following categories based on their preferred diet: carnivores, granivores, insectivores, nectarivores, frugivores, and omnivores. Preferred diet was determined based on dietary records from the *Handbook of Australian, New Zealand and Antarctic Birds* (HANZAB) (HANZAB; Marchant & Higgins, 1993; Higgins, 1999; Higgins, Peter, & Steele, 2001; Higgins and Peter 2002; Higgins, Peter, & Cowling, 2006).

A multidimensional scaling (MDS) plot was then constructed to visually represent the variation in woodland bird composition detected in restored and remnant areas, to which vectors (Pearson correlation >0.4) were applied to demonstrate the influence different bird species have on overall composition. I then performed a second independent sample t-test using the univariate community measures of avian assemblage previously used (abundance, richness, and evenness) for both the entire bird community, and the woodland dependant sub-set to see if restored and remnant sites were significantly different with respect to any of the listed measures of avian assemblage. I also compared the woodland community assemblage between restored and remnant sites with an analysis of similarity ANOSIM in PRIMER v6 (Clarke & Gorley, 2006).

To further examine whether the estimated robin density was indicative of bird community composition, I then conducted a similarity percentage (SIMPER) assessment and ANOSIM in PRIMER v6 (Clarke & Gorley, 2006). Total robin density estimate during the breeding season was used in the analysis, and sites were separated into four groups based on the density of robins detected within the 2 ha search areas (none [n = 0], low [n = 1-5], medium [n = 6-10] and high [n>10]). The maximum number of individual robins at any one site was 20, and the next highest was 13. Both the within-group similarity and between-group similarity values were calculated. I then constructed a principal component analysis (PCA), in PRIMER v6 (Clarke & Gorley, 2006), to visually represent the variation in woodland bird assembly based on the four robin density estimate categories.

Next, I performed an independent samples t-test to determine whether restored and remnant sites had significantly different numbers of robins. This was done using total estimated robin density recorded during the summer of 2015, and the total estimated robin density for spring of 2016, as well as juvenile-only and adult-only measures for the spring of 2016. Levene's test for equality of variances was used to determine whether variances could be assumed to be equal (Levene, 1960).

Multiple linear regressions with all subsets of variables were then used to test a range of habitat variables against total estimated robin density during the spring of 2016. This was done using an SPSS-specific procedure called Automatic Linear Modeling (ALM), in which a group of predictor factors (scales, ordinal variables, and dichotomous variables)

are inputted, and the combination of factors which best explain variation in response variable is found (Yang, 2013). The model was built using the Akaike Information Criterion (AICc) values to identify which subset of variables should be included to generate the best (most parsimonious) model available. The best model is identified based on lowest AICc values, with models within 2 AICc values being considered equally reliable. The estimated robin density was measured using total robin abundance detected during the spring sampling period. A total of eight predictor variables were first tested for collinearity, and some of the highly correlated variables were excluded based on logical deduction regarding redundant variables. Perimeter and area were found to be highly correlated ($R_{24} = 0.501$, $p = 0.013$), as both measures are likely to provide the same information in the model, perimeter was excluded from the model. Likewise, leaf litter and CWD were highly correlated ($R_{24} = 0.525$, $p = 0.008$), as was leaf litter and fire age ($R_{24} = 0.408$, $p = 0.048$), which is unsurprising given that fire typically removes leaf litter and CWD, both of which gradually accumulate in the absence of fire. As such, fire age was selected for the model as a proxy measure of leaf litter and CWD. Variables that were included in the model, despite being found to be correlated, included woodland area which was correlated with both distance to nearest neighbouring woodland area ($R_{24} = 0.405$, $p = 0.050$) and restoration age ($R_{24} = 0.479$, $p = 0.018$), but as these factors are correlated due to landscape management decisions by the RIA, it is was deemed unlikely that they would introduce redundancies into the model. The variables were then inputted into the model (Figure 2.2). These variables included a structural habitat variable (vegetation cover), spatial variables (woodland patch area, and distance to nearest neighbouring woodland patch), and a temporal variable (time since last fire).
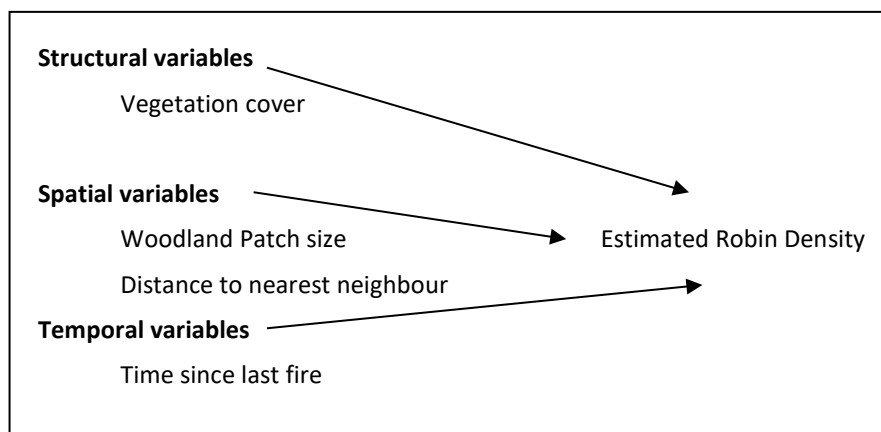
Figure 2.2: The nine variables inputted into an ALM model to test for predictors of estimated robin density.

To visually show the relationships between estimated robin density and the habitat variables

used in the ALM, a series of scatter plots were then created. The plots compare key habitat variables with the following measures of estimated robin density; adult-only robin density measured in spring (2016), juvenile-only robin density measured in spring (2016), and total robin density measured in summer (2015). Plots were also developed to show the relationship between the significant habitat variables and the density of three other woodland species (golden whistler, singing honeyeater, and western gerygone) that have been identified as priority species. These plots were generated in Microsoft Office Excel 2007 (Heldman, 2007).

2.5 Results

The estimated total robin density between the two survey periods (spring and summer) was statistically different ($t_{1, 23}$ = 3.729, p = 0.001). Robin density estimates during the two survey periods was not found to be significantly correlated (R_{24} = 0.400, p = 0.053). Adult robin density estimates in spring was statistically different from total robin density estimates in summer ($t_{1, 23}$= 2.924, p = 0.008), and were also significant correlation (R_{24} = 0.405, p = 0.050). Juvenile robin density estimates in spring were not statistically different to total robin density in summer ($t_{1, 23}$= 0.001, p = 1.000), and the two were less correlated than the measures that included adult birds in spring (R_{24} = 0.331, p = 0.114).

Robin density was not significantly correlated with any of the univariate measures of avian assemblage for the overall community, the woodland dependant subset, or abundance measures based on feeding guild (Table 2.1). Furthermore, robins were not one of the 10 most influential woodland bird species in explaining variation in assemblage using a PCA (Figure 2). Robins made up just 8.9% of the birds observed during the community counts. The three most commonly observed species were the silvereye (20%), the western gerygone (17%), and the white-browed scrubwren (14%).

Table 2.1: Pearson correlations between robin density estimates during both survey periods and various measures of avian assemblage (based on community data collected in summer).

Avian assemblage	Summer Total Robin Density		Spring Total Robin Density		Spring Adult Robin Density		Spring Juvenile Robin Density	
Total community	R_{24}	P	R_{24}	P	R_{24}	P	R_{24}	P
Total Abundance	0.195	0.362	0.031	0.885	0.104	0.63	-0.052	0.81
Total Richness	0.046	0.832	0.222	0.297	0.191	0.37	0.22	0.302
Total Evenness	0.062	0.775	0.297	0.159	0.2783	0.197	0.276	0.192
Woodland community								
Woodland Abundance	0.117	0.585	0.055	0.797	0.14	0.513	-0.045	0.833
Woodland Richness	-0.055	0.8	0.327	0.119	0.299	0.155	0.305	0.147
Woodland Evenness	-0.062	0.774	0.389	0.06	0.381	0.066	0.335	0.109
Dietary guilds								
Carnivores	0.097	0.651	-0.168	0.433	-0.16	0.455	-0.15	0.485
Granivores	-0.168	0.433	0.057	0.792	0.146	0.497	-0.048	0.822
Insectivores	0.385	0.063	0.274	0.195	0.332	0.113	0.168	0.432
Nectarivores	0.101	0.64	-0.156	0.467	-0.17	0.428	-0.116	0.589
Frugivores	-0.168	0.433	-0.082	0.704	-0.146	0.497	0.079	0.714
Omnivores	0.297	0.159	0.156	0.467	0.214	0.316	0.069	0.75

The overall abundance of birds found in the restored areas was higher than the abundance of birds found in remnants ($t_{1, 23}$= 1.772, p = 0.090). This is a well-documented phenomenon that occurs when disturbance specialist species respond to restoration efforts. This can be seen in figure 2.3, where species such as the Australian raven and silvereye are responsible for most of the differences in assemblage assembly. This is unsurprising, as restored areas often have higher abundances of disturbance specialists than remnants. While the ANOSIM revealed a significant difference in community composition between restored and remnant areas (R = 0.202, P= 0.015), no significant difference in the univariate measures of avian assemblage were detected (Table 2.2).
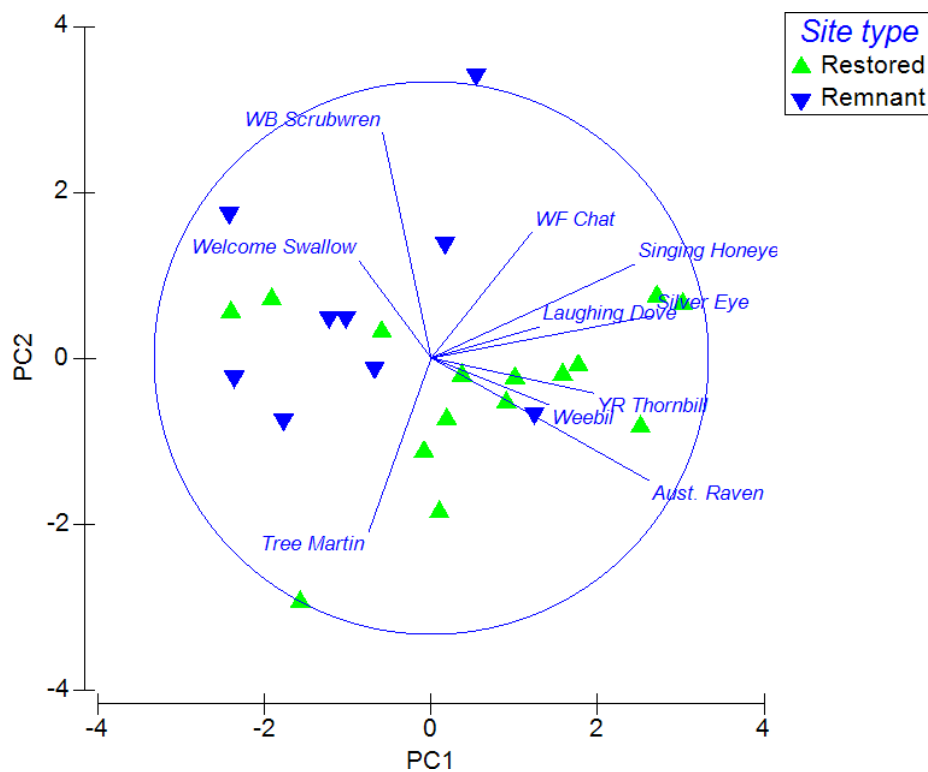
Figure 2.3: PCA displaying variation in avian assemblage between sites, with vectors displaying the 10 most influential bird species. No vector is displayed for robins, as their correlation with overall assembly was too weak.

Table 2.2: Assessment of similarity between various univariate measures of avian assemblage found in restored and remnant woodland areas on Rottnest Island. Samples that reject Levene's test for equality of variance (P<0.05) used a corrected t -test value for unequal variances.

	Levene's test		t-test	
Community measure	F(22)	P	t(22)	P
Total Abundance	0.154	0.698	1.611	0.121
Total Richness	0.049	0.826	1.168	0.255
Total Evenness	0.364	0.552	-1.432	0.166
Woodland community				
Woodland Abundance	0.218	0.645	1.633	0.117
Woodland Richness	0.262	0.614	1.455	0.160
Woodland Evenness	0.104	0.750	-1.436	0.165
Dietary guilds				
Carnivores (df =9)	**5.166**	**0.033**	-1.029	0.331
Granivores	0.095	0.761	0.153	0.880
Insectivores	0.145	0.707	-.325	0.748
Nectarivores (df=14)	**7.091**	**0.014**	1.468	0.168
Frugivores	0.095	0.761	-0.153	0.880
Omnivores	3.088	0.093	1.347	0.192

Between-group-similarity (based on density of robins, similarity based on avian community

assembly) was lowest between areas with high robin density and areas with no robins, and the highest similarity was found between areas with high and low numbers of robins. Overall, the between-group-similarity varied by less than 10% between the various combinations of robin density categories. This can be seen in PCA ordination (Figure 2.4), which demonstrates the overall similarity between the four categories. The ANOSIM found a significant, difference in bird assemblage between the robin density classes (R= 0.109, N = 11, P= 0.049), with significant differences detected between sites with no robins and sites with high density estimates of robins, and between low and medium density of robins (Table 2.4).

Table 2.3: Results of SIMPER analysis of woodland bird community relative to robin density categories (high [> 5 robins per ha], medium [3 -5 robins per ha], low [1-2 robins per ha], and none [0 robins per ha]).

Within Group Similarity	
Robin presence	Similarity (%)
None	57.97
Low	69.71
Medium	71.08
High	73.85
Between Group Similarity (%)	
None & High	62.35
None & Low	63.98
None & Medium	65.3
Low & Medium	68.08
Medium and High	68.39
Low & High	72.26

Table 2.4: Results of ANOSIM for woodland bird community assemblage relative to robin density categories (high [> 5 robins per ha], medium [3 -5 robins per ha], low [1 -2 robins per ha], and none [0 robins per ha]).

Groups being compared	R	P	N (Possible)	N (Actual)	N (Observed)
High & None	**0.256**	**0.024**	**462**	**462**	**11**
High & Low	-0.065	0.808	1716	999	807
High & Medium	0.217	0.058	462	462	27
None & Low	0.084	0.188	792	792	149
None & Medium	0.045	0.325	462	462	150
Low and Medium	**0.163**	**0.045**	**1716**	**999**	**44**
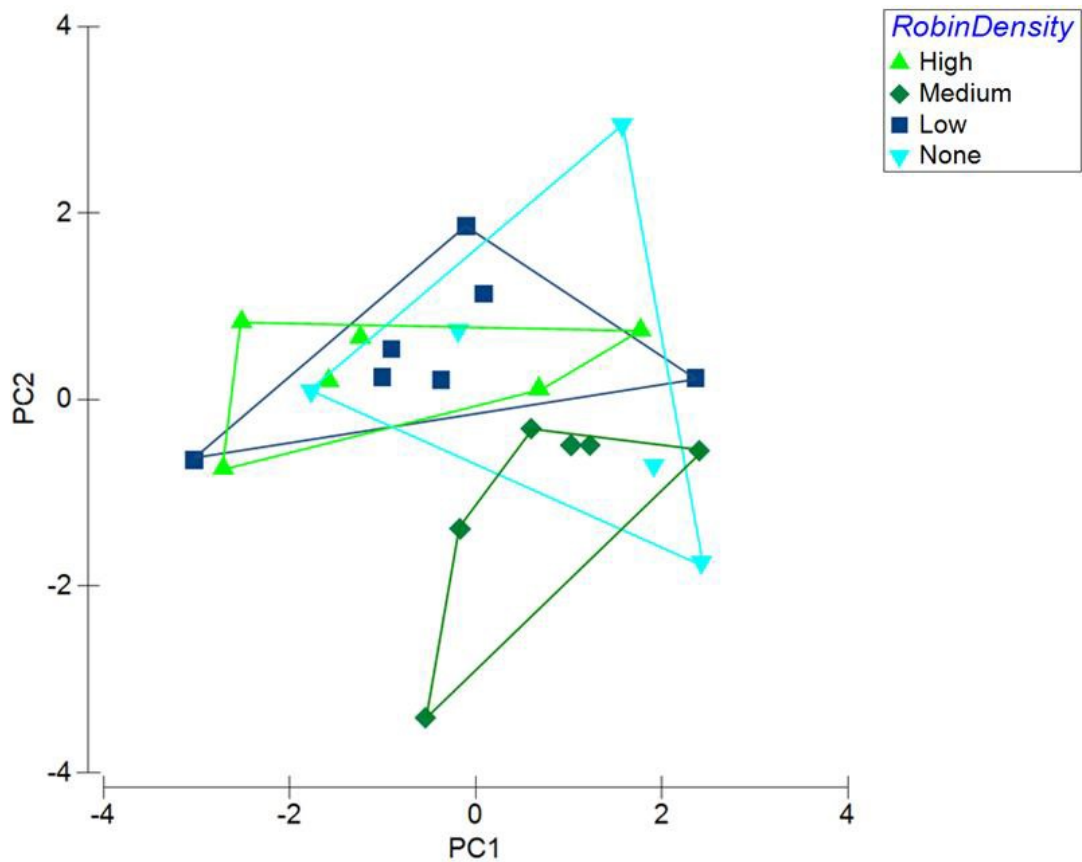
Figure 2.4: PCA based on avian community composition displaying relative size and overlap of four habitat categories defined by robin density estimates (high [> 5 robins per ha], medium [3-5 robins per ha], low [1-2 robins per ha], and none [0 robins per ha]).

Robin density was not found to vary between restored and remnant areas. Levene's test for equality of variances was not violated for the 2015 summer survey data ($F_{1,22}$ = 0.140, p = 0.712), and the independent samples t-test found no significant difference in robin density between restored and remnant sites (t_{22} = 0.662, p = 0.515). The spring 2016 data also did not violate Levene's test for equality of variances for total robins ($F_{1,22}$ = 0.151, p = 0.702), adult-only ($F_{1,22}$ = 1.497, p = 0.235), or juvenile-only ($F_{1,22}$ = 0.495, p = 0.489). Interestingly, for this survey, juvenile robin density was found to be significantly higher in remnant areas than restored areas (t_{22} = 2.402, p = 0.026). Total robin, and adults-only were not found to significantly differ between the two site types (total-robins [t_{22} = 2.069, p = 0.051], adult robins [t_{22} = 1.385, p = 0.180]), however total robin density was very close to being significant. The differences between robin density in restored and remnant areas can be seen in Figure 2.5.
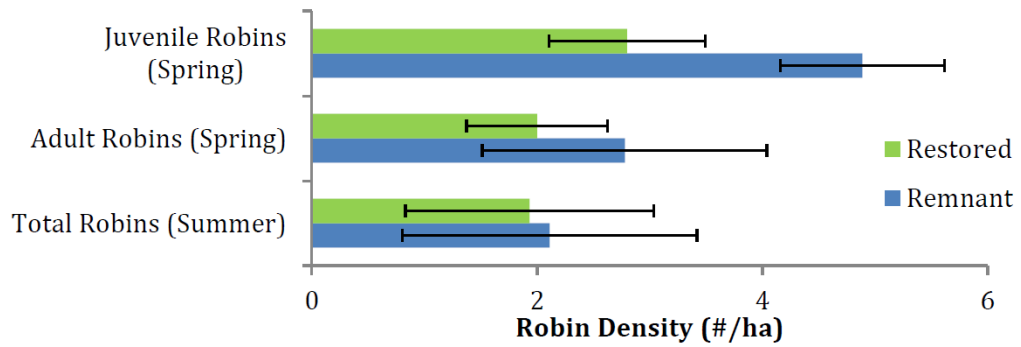
Figure 2.5: Comparison of robin density in restored and remnant areas.

Table 2.5: List of factors included in the best model (lowest AICc) that had a significant ($p < 0.05$) effect on robin density.

Factors	Coefficient	P-value	Importance
Woodland patch size	0.001	0.001	0.459
Time since fire	0.179	0.001	0.541

The results of the ALM showed that of the nine inputted variables, a group of three were significantly related to robin density. These were; woodland patch size, time since last fire, and site type (restored/remnant). In other words, the combination of woodland area (spatial), time since fire (temporal), and restored/remnant (site type) best explained variation in robin density (see Table 2.5).

The best model (lowest AICc value with no other similar models within 2 AICc values of this model) was found when just woodland area and time since last fire were included. The accuracy of the final model was high ($R^2 = 48.2\%$) with an AICc value of 66.499. Time since last fire was of greater importance in the model.. Scatter plots (Figure 2.6) revealed the relative contribution of adults and juveniles to this result, as well as suggest that these habitat factors may have varying levels of influence outside of the breeding season.

The general trends of woodland area, perimeter and fire age in relation to robin density (summer) can be seen in Figure 2.6. All three relationships were found to be positively correlated for adults and juveniles in spring, and all robins in summer. Juveniles appear to be the most strongly correlated group with woodland area, which may be an indication that juvenile survival is linked with patch size. Long unburnt areas also don't necessarily appear to have high numbers of robins, but recently burnt areas rarely contained high numbers of robins.
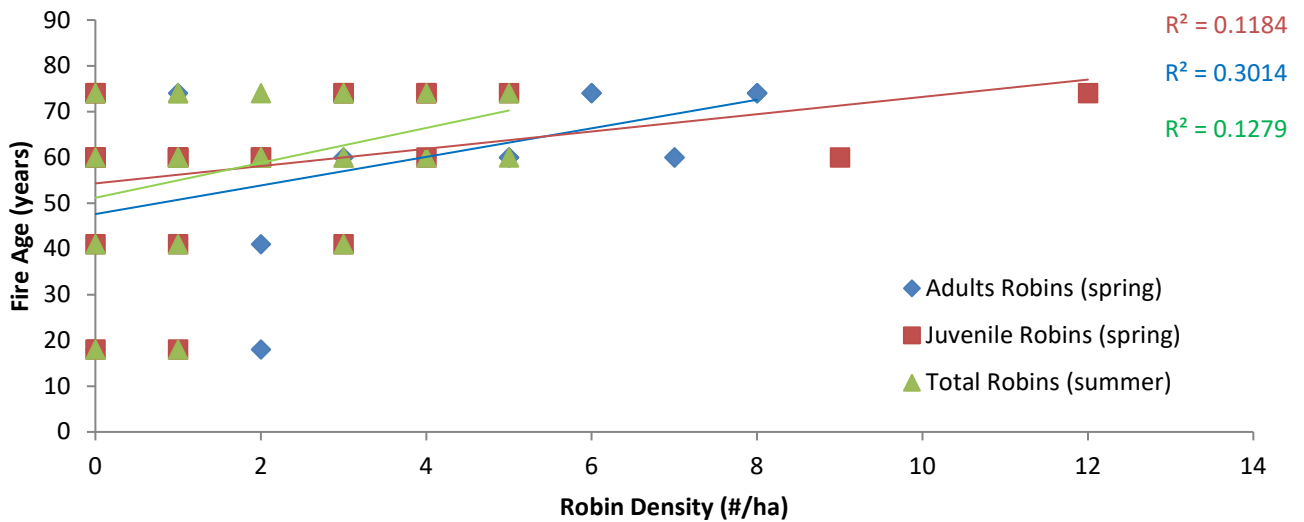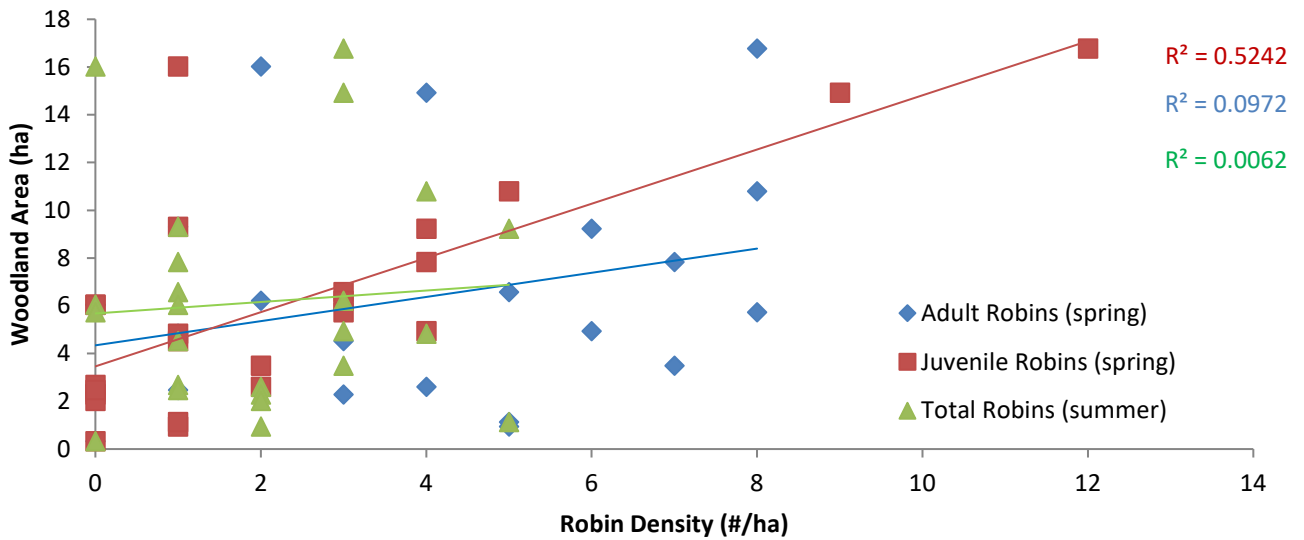
Figure 2.6: Scatter plots that visually display relationships between the habitat variables used in the ALM model and measures of robin density in breeding and non-breeding seasons.

Unlike the general trends between the different measures of robin density, some of the other woodland bird species of conservation significance were found to display negative relationships (Figure 2.7). Red-capped robins, golden whistlers and western gerygones all respond similarly to variation in woodland area and perimeter, but singing honeyeaters had a negative relationship with both. Singing honeyeaters appeared to have no correlation with time since last fire, unlike the robins and whistlers who both had a positive relationship. Gerygones on the other hand had a negative relationship, indicating that they appear to occur at higher densities in areas that had been more recently burnt than those that had not been recently burnt.
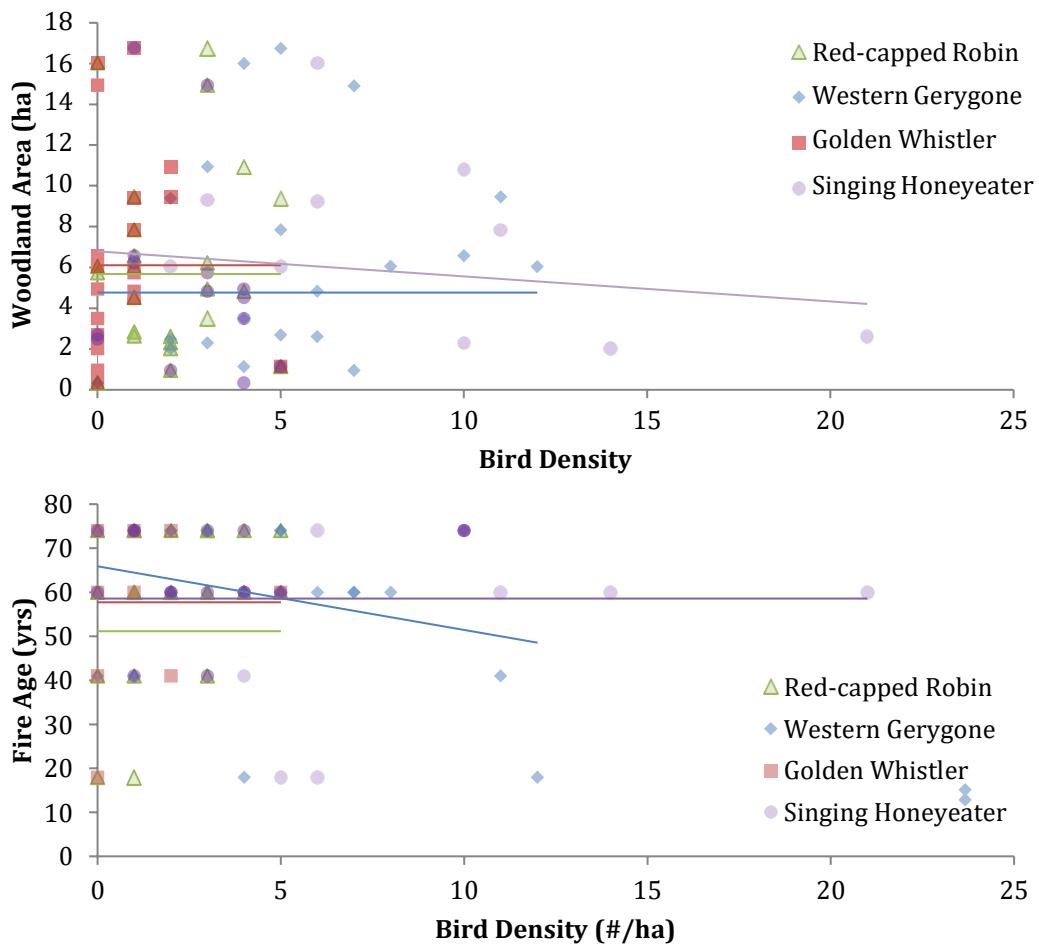
Figure 2.7: Scatter plots that visually display relationships between the habitat variables used in the ALM model and measures of the robin, whistler, gerygone, and honeyeater species found on the island. Each of which has been identified as being of conservation significance (Saunders & de Rebeira 2009).

2.6 Discussion

An expectation of a good indicator species is that through observation and analysis of abundance of the species, an understanding of the broader differences in community types, habitat conditions, or environmental changes can be gained (McGeoch, 1998; Niemi & McDonald, 2004; De Cáceres, Legendre, & Moretti, 2010). The distribution of robins on the island was not found to be indicative of the overall avian assemblage, even when the focus was narrowed to only species typically associated with woodland habitat. Tested bird community measures included diversity, abundance, and composition, and the absence of correlation was irrespective of feeding guild. As the distribution of robins across the island wasn't found to significantly relate to any of the listed measures of avian community types, it would appear that robins are an unsuitable indicator of avian community types in this system.

These results indicate that robins do not meet the assumption of the focal species surrogate model. As explained by Lindenmayer *et al.* (2002), there is considerable evidence that similar species and groups of species often respond to threatening processes in vastly different ways (e.g. Robinson *et al*., 1992; Gascon *et al.*, 1999). Hence caution should be taken with regard to the implementation of the focal species models. This does not necessarily de-value enquiry into the mechanisms behind the distribution of species like the robins from areas such as Rottnest, but highlights the reality that taking shortcuts that exclude species through the use of surrogate species models may lead to ecologically damaging errors in management.

Robins appear to occur in greater densities in large patches of woodland areas, in areas that are long un-burnt, and in remnant woodland rather than restored woodland. CWD density, leaf litter cover, and vegetation density are all structural habitat variables that are likely to increase in the absence of fire (Maron & Kennedy, 2007) and be higher in remnants than restored areas. Yet these variables were not found to have a significant effect on robin density. Further inquiry into microhabitat selection may be required, if we are to gain insight into the mechanisms behind variation in robin density on the island. This topic is explored further in Chapter four.

A study into the influence of habitat size and shape on the age-structure and density of a red-capped robin community revealed similar findings (see Major *et al.*, 1999). Major *et al.* (1999) found that robin density was significantly higher in large, non-linear remnants than in small, linear remnants. They also found that age structure and delayed plumage maturation was also linked with patch size, highlighting the complexity of metapopulation interactions between patches of varying sizes. The Major *et al.* (1999) study concluded with a warning to land managers to avoid over-reliance on narrow, linear wildlife corridors, and emphasised the importance of large areas of native vegetation. Cunningham *et al.* (2008) also found that red-capped robins responded positively to large elliptical or block shaped plantings in farmland areas. The concern with overly narrow habitat patches is the risk of adverse edge effects, such as predation. This study found that perimeter was positively correlated with robin density, however it should be noted that perimeter and patch size were also correlated.

The Australian raven *Corvus coronoides*, an opportunistic, disturbance specialist species that has successfully colonised and become common on Rottnest Island, is a predator of eggs

and nestling bush birds (Stevenson, 2011). In a study conducted on Rottnest Island, artificial nests received a 20% predation rate, indicating a high capacity for potential impact on bush birds. Camera traps at genuine robin nests, as well as raven stomach contents analysis, which found feathers and bird bones, have been used to verify the assertion that ravens predate bird nests on Rottnest (Appendix B; Stevenson, 2011). The majority of the raven population tends to be centralised around the urban areas of the island, away from the restored and remnant woodland habitats where robins and other bush birds are commonly found. In addition, stomach contents analysis revealed that the plant material and invertebrates made up the majority of the ravens' diets (Stevenson, 2011). As such, Stevenson (2011) concluded that management of the raven population was recommended as a precautionary approach, but that the positive effects that woodland restoration efforts were having on bush bird recruitment appeared to outweigh the loss of eggs and nestlings to ravens. This offers an explanation for higher densities of robins occurring in larger woodland patches. Larger patches are less exposed to edge effects; meaning robins may suffer less from raven predation. Given that this study found that area had a much stronger correlation with robin density, especially with juvenile birds, and other studies have found robins occur at higher densities in areas with high area to perimeter ratios, it is likely that patch size is of greater importance than edge length.

Red-capped robins are not generally thought to be particularly fire susceptible, and have been described as 'favoured' in early seral stages when habitat understories are open (Woinarski & Recher, 1997). This is likely due to the transformative process fire has on woodland habitats. Fire can open up the canopy and thin the understory vegetation, creating desirable foraging habitat for ground pouncing insectivorous birds (Recher, Davis, & Calver, 2002). This study found that robin density was typically higher in long un-burnt areas, which is interesting given the history of fire and fragmentation on the island. On Rottnest, fire has been shown to convert woodland habitats into scrub or heathland habitats with no canopy and a dense understory (Rippey & Hobbs, 2003). For more information on how fire transforms woodland habitats on Rottnest, see Chapter five. Fire and fragmentation have both had considerable influence on the avian assemblage on the Island since European settlement (Winn, 2008). Prior to European settlement in 1831, the majority of Rottnest Island was covered in large areas of *Callitris preissii - Melaluca lanceolata* woodland, a now threatened ecological community in Western Australia (Keighey *et al.,* 2003; Winn, 2008). At this time, fires are thought to have been extremely infrequent, as the Island's separation from the mainland protected it from bushfires and Aboriginal burning regimes (Marchant &

Abbott, 1981). There is evidence of occasional fires caused by lightning strikes, which would have been fierce and widespread due to the accumulation of dead wood over long periods between fires (McArthur, 1996). It is presumed that these fires would have destroyed a considerable portion of the Island's fire-sensitive plants, including *Callitris* and *Melaleuca,* neither of which is well adapted to survive fire (Boland *et al.*, 1984). Both species are easily killed when exposed to fire, and rely on regeneration from seed, which necessitates long intervals between fires, to give the seedlings time to mature and set seed (Storr, 1963; Wykes & McArthur, 1995; Marchant, 1997).

Efforts to mitigate the negative effects of fire, woodland habitat loss and fragmentation, have been in operation on the Island for over 50 years, with the first woodland restoration beginning in 1963 (Winn, 2008). The goals of the woodland restoration program on Rottnest are to: prevent local extinction; extend woodland habitat to protect Island wildlife; and to enhance the natural recreation amenity of the island (RIA, 2014). The RIA recognises robins as a priority species on the island, and as such, have expressed intent to ensure that restored woodland patches provide valuable habitat for the species (RIA, 2014). This study found that during summer there was no significant difference in robin density between restored and remnant areas. The spring sampling period, on the other hand, which included demographic data collection with respect to age and gender where possible, found a significant difference in juvenile robin density between restored and remnant areas. While the reasons for this difference remain unclear, I hypothesise that restored areas may offer lower quality breeding habitat than remnant areas, and restored areas may provide less protection from predators than remnants. These hypotheses are explored at length in Chapter four.

Given that the distance between restored and remnant patches varies, and is at some sites very small (<30 m), and robins are easily able to traverse such distances, it could be reasoned that many of the robins on the Island can feasibly move between restored and remnant patches. This means juvenile robin occurrence isn't definitive evidence of robins breeding in an area. Further research into animal behaviour would be required to determine whether robins are breeding in both restored and remnant areas, and whether the two habitat types offer different resources for the birds. This is explored further in Chapters three and four, in which food resources and bird behaviour are studied.

As stated previously, the goal of the RIA restoration program is to extend woodland habitat such that it protects island wildlife. Both fire and fragmentation have created conditions that can be favorable for invasive predators or competitors (Maron & Kennedy, 2007). Prescribed burns used in the past to reduce fuel loads have had considerable impact on biodiversity, in ways that are currently poorly understood. This is also true of much of the forestry industry, where prescribed burns are seen as necessary (Granström, 2001). Edge effects, a result of habitat fragmentation, are also well known to influence fauna in a number of ways (Watson, Whittaker & Freudenberger, 2005). Alteration of habitat characteristics, creation of habitat suitable for disturbance specialists, increased predation, competition, and parasitism are all well documented impacts of edge effects (Forman, 1995; Chace *et al.*, 2003; Batary & Baldi, 2004; Maron & Kennedy, 2007).

Management decisions aimed at improving habitat for robins should ideally coincide with improvements to the larger community. The avian assemblage overall doesn't appear to be more abundant, diverse or rich in larger patches of woodland habitat. Fire age doesn't appear to share the positive relationship with the overall community composition, and some species appear to prefer more recently burnt areas (e.g. the western gerygone). Restored and remnant areas had no significant differences between any of the univariate measures of avian assemblage, but were found to differ when composition was assessed. Given the severity of the changes Rottnest Island's woodland habitats have experienced, the communities now found in those remnant patches are likely to be fairly resilient to fragmentation and isolation. The system is now in a novel state, meaning pre-conceived interactions and behaviours may not apply. Further inquiry into direct interactions may provide insight into how the system is operating, and what could be done to improve those interactions for species found on the island. In Chapter three I explore how the invertebrate community, specifically as it relates to food availability may provide useful insight into the distribution of robins on the Island. There has been a considerable number of cases where invertebrates have proven to be a suitable taxonomic group to act as surrogate measures of ecosystem functionality (e.g. Stannard, 1967; Majer, 1983; Dunger, 1989; Resh, Norris, & Barbour, 1995; Fore, Karr, & Wisseman, 1996). Given that every species of woodland bird on the Island is at least partially insectivorous, it can be argued that the invertebrate population on the Island plays a vital role in sustaining the avian community.

When robin density was compared with that of the other three avian species identified by Saunders and de Rebeira (2009) as being of conservation significance, patch size and

perimeter influenced all but the honeyeater in a similar manner. Time since fire was only positively correlated with the robins and whistlers, who are unlikely to use the scrub or heathland habitat that is likely to replace woodland habitat after a fire. The honeyeaters don't appear to be influenced by fire, and gerygones appear to favour recently burnt areas, highlighting the differences in management requirements between similar species.

Compared with vertebrates as biodiversity indicators, it has been argued that invertebrates may better reflect trends in species richness and community composition (Gerlach, Samways & Pryke, 2013). This is largely due to their greater diversity and abundance (Kremen, WIlliams, & Thorp, 1993; Bisvac & Majer, 1999). Invertebrates are sensitive to local conditions, their mobility enables them to move in response to changes, a short gestation time means their population size can fluctuate quickly, and it is relatively simple and cheap to collect large samples of them (Samways & Sharratt, 2010; Gerlach, Samways & Pryke, 2013). In a study of North American butterfly distributions using presence-absence data for select species, 82% of the combined distributions of birds and butterflies could be described using a general linear model (Fleishman *et al.,* 2005). The study demonstrated the value invertebrate species can have in predicting the distribution and diversity of other species that have similar dispersal mechanisms, even those at higher taxonomic levels. The invertebrate community of Rottnest is explored in detail in Chapter three.

The results of this chapter demonstrate that in this case, management based on one species will not serve the needs of the whole community. The particular sensitivities of robins to fire and habitat patch size are important considerations for land managers, but their needs may need to be weighed up against the needs of other species whose needs conflict with those of the robins; such as the honeyeater that appears to prefer smaller patches to large ones, and the gerygone that prefers recently burnt sites to long un-burnt sites. The differences in density estimates between summer and autumn, and lack of clear patterns with regards to avian assemblage should serve as warnings for land managers and ecologists with regards to data gathering and analysis. Time of year, and methods of data collection will influence the results, and presence/absence data alone may be insufficient to determine whether two habitat types are of equal quality, especially between different seasons. These ideas will be explored at greater length in the following chapters.

Chapter 3: Using food resource availability as a measure of habitat quality: Case study of the invertebrate assemblage on Rottnest Island.

3.1 Introduction

There are concerns about declines in woodland bird assemblages across many regions worldwide. There are numerous cases of once widespread species of woodland birds becoming restricted and scarce, and local extinctions have become increasingly common (Ford *et al.*, 2001; Murphy, 2003; Donald *et al.,* 2006; Fuller *et al.*, 2007; Watson, 2011). Habitat loss and degradation, often through fragmentation, are generally attributed as the main driving forces behind these declines (Mac Nally *et al.*, 2009). The severity of these concerns has resulted in woodland birds being recognised as a global conservation priority (BirdLife, 2008).

Ecological restoration efforts typically aim to mitigate or reverse habitat degradation, and thus, increase the resilience of biodiversity (Wortley *et al.*, 2013). Unfortunately, many restoration efforts have failed to achieve that aim (Choi, 2007). This can be attributed to a number of causes, such as: 1) the level of disturbance being beyond an ecological threshold, meaning complete reversal is impossible; 2) the long timescale required for ecological processes to develop; 3) the relative infancy of the discipline of study, 4) poorly defined targets set out at the beginning of the restoration effort, 5) a lack of adequate monitoring the restoration developed, and 6) the failure to apply scientifically backed research to restoration planning, in favour of ad-hoc management planning (Bash & Ryan, 2002; Miller & Hobbs, 2007; McDonald & Williams, 2009; Parkes *et al.*, 2012). All of these factors have contributed to the creation of numerous hybrid and novel ecosystems containing unusual species assemblages with non-traditional interactions and behaviours (Hobbs *et al.*, 2009; Williams & Jackson, 2007). Often these changes are irreversible, meaning land-managers are then restricted to finding novel ways of mitigating the effects of habitat degradation.

Given the rate and extent of vegetation clearing and other ecologically damaging processes affecting woodland habitats in recent years, it is not surprising that numerous woodland-dependant biotas have declined (Recher, 1999). Interestingly, it has been noted that some

woodland species are more likely to be in decline than others (Antos & Bennett, 2006). Traits common among declining bird species include: small size; sedentary nature; ground-foraging; and insectivory (Reid, 1999; Ford *et al.*, 2001; Ford, 2011). If we are to mitigate or remove the cause of these declines, we need to understand why these groups of birds are declining.

Habitat fragmentation is one of the most prolific areas of research within the field of conservation biology, and has been since the field began (Harrison & Bruna, 1999). This is likely due, in part, to the extent to which fragmentation has occurred to natural habitats globally, and the impact that fragmentation has on biota. The widespread fragmentation of natural systems to accommodate our ever increasing need for agricultural land, natural resources, and residential areas has put tremendous pressure on the world's ecosystems (Hobbs & Harris, 2001; Hilderbrand *et al.*, 2005; Brudvig, 2011). Ecologists and land managers, who use conservation and restoration practices to mitigate the pressure being applied to affected areas, are therefore interested in the effects of fragmentation on ecological systems.

Insectivores are among the worst affected woodland birds, which may indicate that declines are linked in some way to changes in invertebrate assembly (Mühlner *et al.*, 2010; Watson, 2011). In addition to being an important food resource within most ecosystems, invertebrates contribute overwhelmingly to the overall biodiversity of those systems (Anderson & Smith, 2004). Given this, it stands to reason that biodiversity monitoring programs aimed at studying the integrity of an ecosystem could not be considered adequate without assessment of invertebrates (Taylor & Doran, 2001).

Changes in nutrient availability due to changes in land use practices, such as increased agricultural activity, can have profound effects on the invertebrate assemblage (Mac Nally *el at.*, 2009). This can have flow-on effects that influence other taxa at different stages of the food web (Schaub *et al.*, 2010; Watson, 2011). In this sense, monitoring of invertebrates provides insight into changes in underlying ecosystem productivity and functionality, allowing invertebrates to act as bioindicators for specific aspects of the system, such as; food resource availability, soil condition, and functionality of pollinator services (Paoletti, Thomson, & Hoffmann, 2007).

The value of invertebrates as an assessment tool comes from their great abundance, diversity, functional importance, sensitivity to disturbances, and ease with which they can

be sampled (Recher, Majer, & Ganesh, 1996; Brown, 1997; McGeoch 1998). This makes invertebrates as a taxonomic or functional group, a potential indicator of ecosystem conditions and functionality (Gerlach *et al.*, 2013). In the context of woodland insectivorous birds living in fragmented habitats, invertebrates can be seen as having functional importance as a food resource, and may be sensitive to habitat disturbances themselves. Gaining a better understanding of the ways in which terrestrial invertebrates respond to landscape disturbances and restoration, may improve our understanding of how changes in food resource availability may be contributing to insectivore declines.

Aquatic invertebrates are a cornerstone of biological monitoring in aquatic systems, where biological integrity is assessed using well developed procedures (Resh, Norris, & Barbour, 1995; Fore, Karr, & Wisseman, 1996). Terrestrial invertebrates, on the other hand, are commonly overlooked as an important topic of study in the research agenda of restoration practitioners. This is especially true of the mining sector (Majer, Brennan, & Moir, 2007). Notable exceptions include: Dunger's (1989) work on German coal mine dumps; Stannard's (1967) work on strip-mined land in Northern America; Hutson's (1980) work on reclaimed coal pits in England; and Majer's (1983) work on open cut bauxite mines in Australia. Ant monitoring was developed to assess restoration success following mining (Majer, 1983). Majer's work on ants was one of the earliest uses of insects as a bioindicator in land management anywhere in the world (Anderson & Majer, 2004). Since then, ant monitoring has become more widely adopted in the mining sector, as part of best-practice environmental management (Andersen, 1997; Anderson & Smith, 2004). Ant monitoring is also a useful tool for conservation assessments (Underwood & Fisher, 2006), as well as assessing the impacts of grazing in rangelands (Landsberg, Morton, & James, 1999). Despite this, monitoring of other terrestrial invertebrates remains relatively rare, and appears to be a grossly underutilised resource for ecosystem quality and health assessments.

Habitat complexity can be a key driver of invertebrate assembly, which can in turn influence insectivorous bird communities. Areas with diverse and complex habitats are likely to contain the microhabitat requirements of more taxonomic groups than habitats that are simple and uniform (Heck, 1977; Taniguchi, Nakano, & Tokeshi, 2003; Hendrickx *et al*., 2007). This can be an issue in restored areas that are likely to be highly uniform, especially in the early years of development. Habitat diversity and heterogeneity tend to increase with stand age and time since last major disturbance (McClain & Barry, 2010). Habitat diversity and heterogeneity are therefore desirable traits that reward niche differentiation, which is

when different species with different resource requirements face less competitive pressure than species utilising the same resources (Peterson & Holt, 2003).

When a habitat is altered, species with specialised microhabitat requirements may be maladapted to surviving in the newly altered habitat (Julliard *et al.,* 2006). Important substrates such as fallen logs, leaf litter, and understory vegetation can all be altered through disturbances, such as timber harvesting, altered fire regimes, weed invasion, trampling and herbivorous grazing (Braunack & Walker, 1985; Cousin, 2004). This may alter the viability of the area for invertebrates that live in that substrate, and thus, alter the viability of species that rely upon those invertebrates.

Aside from small-scale structural factors associated with what makes up viable habitat, which can be described as micro-habitat characteristics, there are also landscape level factors that influence community assembly, and can be described as macro-habitat characteristics. There is a considerable body of literature on the interactions between isolated patches of habitat and the landscape within which they are situated. MacArthur and Wilson's Theory of Island Biogeography (1967), Clements' theory of Successional Dynamics (1916; 1936), and the Metapopulation Concept, described by Levins (1969) are all well established theoretical models used to describe the ecological processes associated with community assembly across isolated patches of habitat.

These models have been promoted as a theoretical basis for the design of nature reserves for decades (Terborgh, 1974; Diamond, 1975; Lovejoy & Oren, 1981). The models predict how species richness in fragments will change over time based on various species-area relationships in insular communities (Connor & McCoy, 1979; Ricklefs & Lovette, 1999; Laurance, 2008). These models can be useful tools for land managers wanting to predict how species will respond to restoration efforts in fragmented landscapes. They provide the basis for arguments in favour of increased connectivity, and larger patch size of isolated fragments (Donald & Evans, 2006; Morrison, Marcot, & Mannan*,* 2012).

In the context of Rottnest Island, a heavily disturbed landscape, made up of a mosaic of remnant and altered habitats, it is likely that the Island's recent disturbance history has resulted in changes to the terrestrial invertebrate assemblage. Isolated patches of remnant and restored woodland areas are scattered within a matrix of low scrub and heathland habitat. The fragmentation and creation of scrub and heathland habitats over the past few

hundred years are a result of historical agricultural practices, the construction of roads and a settlement on the eastern side of the island, an increase in fire frequency, and increased herbivorous grazing by the resident quokka *Setonix brachyurus* population. For a more detailed summary of the history of disturbances on Rottnest, see Chapter two. The history of the Rottnest Island woodland restoration program, as well as an explanation of the different habitat types/states and the processes, through which transitions between states occur, is discussed in greater length in Chapter five. In short, restoration efforts over the past 50 years have aimed to reverse the conversion of the native *Callitris preissii - Melaleuca lanceolata* woodland to the closed scrub *Acacia rostellifera* and grassy heath *Acanthocarpus preissii – Austrostipa flavescens*. There a number of factors preventing natural regeneration of the woodland habitat, and without intervention, this threatened ecological system could be lost (Winn, 2008). Management protocols and procedures have changed considerably over the years, which have led to the creation of a number of woodland patches of varying sizes, ages, and levels of isolation.

One of the goals of the Rottnest Island Authorities' (RIA) woodland restoration program is provide wildlife habitat and increase native biodiversity richness (RIA, 2014). Yet very little is known about the terrestrial invertebrate community found on the island, and to date, there has been no formal assessment of the response of invertebrates to restoration efforts on the island. The invertebrate communities are an important component of biodiversity in their own right. But they also perform a number of ecosystem services such as pollination, seed dispersal and organic matter decomposition. They are also an obligate food resource for many species. On Rottnest the woodland avian community is made up of numerous insectivorous species. Hence, this study's first aim was to gain a better understanding of how the invertebrate community in woodland areas compares with the invertebrate community found in heathland areas.

Given that the original reason for this study's focus on invertebrates was as a food resource for a specific insectivorous woodland bird, the red- capped robin *Petroica goodenovii,* who typically captures prey from either the ground or off foliage (Recher *et al.*, 2002; Antos, Bennett, & White, 2008), the invertebrate community was assessed at strata levels that aligned with the birds' feeding habits. The second aim of the study was to determine whether restoration efforts are successfully restoring the terrestrial invertebrate community found in remnant woodland areas, or whether they were creating a novel amalgam of the communities found in heathland and remnant woodland areas. The

recovery rate for the invertebrate assemblage will depend on a number of successional trajectories related to vegetation development, resource availability, dispersal capacity of species, and matrix permeability.

An intention of restoration is to accelerate some of these processes. To gauge the rate of these successional trajectories, I also assessed how the invertebrate assemblage varied between two different age groups of restoration, and compared these to un-restored heath and the remnant woodland reference states. The developmental stages selected were young restoration that was 7-12 years old and old restoration that was 30-50 years old. By identifying the habitat variables that influence assembly, this study aimed to provide land managers with a list of habitat related factors that are most influential to ground dwelling and arboreal invertebrates found in woodland areas. Food is a key driver of animal behaviour, which in turn dictates population distribution. In Chapter two robins were found to be a poor indicator of avian assembly on Rottnest. Given that Rottnest Island's bird community is largely made up of insectivorous species, the invertebrate community is of critical importance to many of the birds that occupy the Island.

The extent to which food availability is affecting birds differently in restored and remnant areas, may be affecting the usefulness of birds as indicators of restoration quality. The relationship between food resource availability, habitat conditions, and robin distribution and behaviour is explored in greater depth in Chapter four. Future projects aimed at refining the restoration management program to improve invertebrate biodiversity and/or abundance, or exploring the use of invertebrates as an indicator species may find this information useful. Finally, as fragmentation and fire have both dramatically altered the landscape of Rottnest Island, this study aimed to identify the extent to which isolation, patch size, and time since last fire influenced woodland invertebrate assembly.

3.2 Hypotheses

1   Assemblages of invertebrates will differ between macrohabitats (heathland, young restoration, old restoration and remnants).

1a Heathland sites will be more different to the three woodland site types (young restoration, old restoration and remnants) than the woodland sites will be to each other.

1b. Restored sites of a similar age will be more similar to one another than they are to heathland or remnant sites.

1c   Old restored sites will be more similar to remnants than young restoration.

2.   Invertebrate assembly at the two assessed strata levels (ground and arboreal) will be strongly influenced my small scale vegetation and habitat variables. .

3.   Invertebrate assemblage will be positively correlated with patch size and proximity to other woodland patches. . Ground dwelling invertebrates will be affected more than arboreal ones, as they are generally less mobile.

4.   Time since last fire will influence invertebrate assemblage, as many invertebrate orders require a build up of leaf litter and dead wood which are both removed by fire, and slow to regenerate.

3.3 Field methods and design

3.3.1 Experimental design

Where possible, prey availability data were collected concurrently with bird foraging observational data (discussed in Chapter four). Unfortunately, as both data collection activities were very time consuming, invertebrate data collection was separated into two discrete surveys, with each survey designed to answer specific research hypotheses. The first survey was conducted in the spring of 2015, concurrently with the bird foraging data, and involved all 24 mature woodland sites discussed in Chapters two and four, the results of which were used to address hypotheses 2, 3 and 4.

The second survey was conducted 12 months later (to minimise seasonal variation), and involved 12 sites, three from each of the following four categories: heathland, young restoration (7-12 years old), old restoration (30-50 years old), and remnant. As discussed in the previous chapter, prior to European settlement, the majority of the island was made up of woodland-type habitat (Winn, 2008). Currently, approximately a third of the island is made up of the heathland type habitat. All restored and remnant woodland sites used were dominated by the overstorey species *Melaleuca lanceolata*. Restored areas have been planted at different densities, and different times of year, however records of the procedures used are limited. Further information on the history of restoration on the

island can be seen in Chapter five. All restored sites used in this study were heathland habitats prior to restoration. This data were used to address hypotheses 1a, 1b and 1c.

While the data collected during the two trapping periods was kept separate for analysis, six sites were surveyed during both trapping periods. These six were all developed woodland areas, and made up the remnants and old restoration sites from the 2016 survey. The distribution of sites used in the 2015 and 2016 surveys can be seen in Figure 3.1. Sites with overlapping symbols were surveyed in both 2015 and 2016.
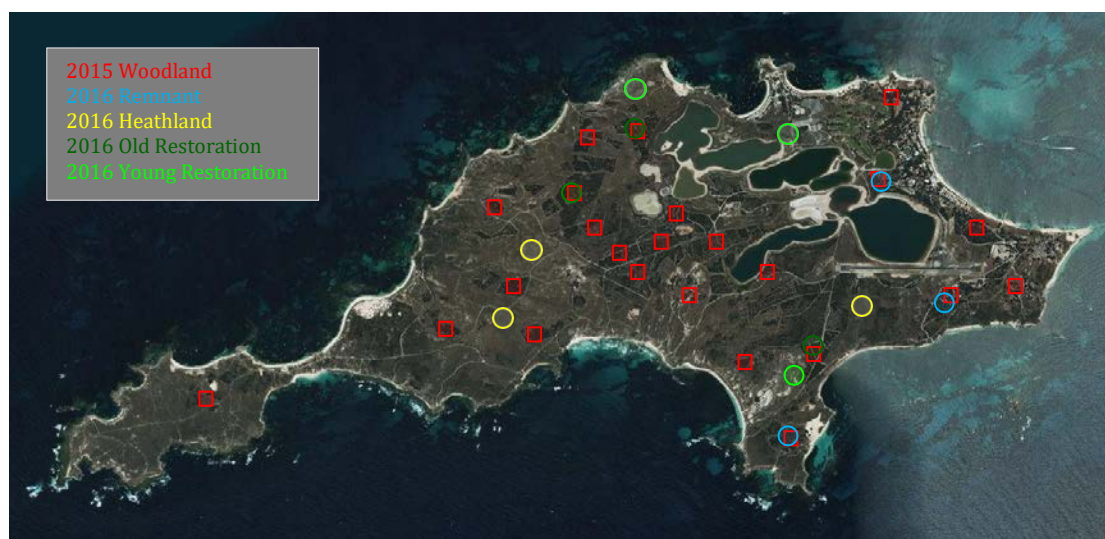
Figure 3.1: Image of Rottnest Island, with points mapping study sites used in the 2015 and 2016 surveys. Original photo courtesy of Google Earth.

As insectivores often display high specificity for preferred foraging substrates, with varying capacities for foraging plasticity, accurately assessing food resource availability can be challenging (Parrish, 2000; Watson, 2011). Red-capped robins typically forage on the ground, while species like the golden whistlers *Pachycephala pectoralis* typically forage on leaves and branches (Ford, Noske, & Bridges, 1986; Major *et al.*, 1999; Higgins *et al.*, 2001; Higgins & Peter, 2002; Higgins *et al.*, 2006). This demonstrates how insectivores are likely to respond to different invertebrate assemblages at different strata levels. To address this variability, two trapping techniques were used to survey invertebrates at two separate strata levels (arboreal and ground dwelling). Samples were collected using both pitfall trapping and beat sampling (described below). At each site, a total of 10 pitfall samples and 10 beat samples were collected.

3.3.2 Pitfall trapping

Pitfall trapping is one of the most commonly used sampling techniques in biodiversity inventories, and is suitable for collecting invertebrates that move along the ground, but may also capture flying insects (Woodcock, 2005; Bulbert & Ginn, 2007; Richter & Groom, 2013). Pitfall cups (5 cm diameter, 5 cm depth) were sunk into the ground so that the rim was flush with the surface of the ground. Inside each cup, we added a pebble, a small amount of water (approximately 2 cm depth) and a drop of liquid detergent to reduce the surface tension (Majer *et al.*, 2007). A plastic plate was suspended, using skewers, above each trap approximately 10 cm above the ground to form a roof to reduce debris falling into the trap, reduce evaporation during the middle of the day, and also to prevent predation of collected samples (Woodcock, 2005). The premise behind this trapping technique is that invertebrates that are active on the ground may fall into the trap and will then be unable to escape.

The location of the traps was recorded using a GPS, and marked with flagging tape. Two trapping grids, with five traps per grid were set up at each site. The trapping grid was designed in a quincunx pattern, as used by the Australian Museum (Bulbert *et al.*, 2007), with four traps making up the corners of a square, with sides 20 m in length, and a fifth trap placed in the centre of the square. The centres of the two trapping grids were random points within the two hectares that encompass the study sites, and were at least 50m apart. All traps were open for a total of four consecutive nights. Each time the traps were checked, the cups were emptied and all specimens were stored in 80% ethanol before they were sorted.

3.3.3 Beat Sampling

This is a widely used technique for collecting flying invertebrates, and invertebrates that live on plants. It can be used to sample any part of the plant including branches, leaves, flower heads and dead wood. It is used to catch insects "on the wing", but it is often more effective when used to catch them at rest as described by Bulbert *et al.* (2007). At each of the 10 sampling points (closest tree or shrub to paired pit fall trap), invertebrates were collected by beating the tree with a broom handle to dislodge invertebrates, which then fall into a collection tray. The vegetation was hit exactly 20 times, while holding the collection tray (W 50 cm x L 65 cm x H 140 cm) underneath the part being hit. Invertebrates were then transferred from the tray to a labelled vial containing 80% ethanol using a pooter, as recommended by Bulbert *et al.* (2007). A single researcher (F. Holmes) collected all samples

in order to standardise sampling effort, and samples were not collected on days of inclement weather, or in windy conditions.

3.3.3 Storage and Sorting

Invertebrates were sorted and identified to order using an identification key supplied by the Australian Museum (Bulbert & Ginn, 2007). This was done so that diversity and abundance measures could be recorded for each sample and major taxonomic group. For invertebrate orders that had a large variation in size between specimens captured, sub-groups large (>1 cm) and small (<1 cm) specimens were used to separate those orders. All samples were stored in vials containing 80% ethanol.

As the focus of this study was to assess the variation in food availability for insectivorous birds, it was necessary to reduce the overall sample size to only invertebrates likely to feature in the birds' diets. This was informed by research into insectivorous birds' diets, conducted by Razeng and Watson (2012). The study involved a comprehensive literature search of insectivore feeding records and stomach contents analyses. They listed the taxonomic groups of insects found in the diets of a number of insectivorous woodland bird species, including the red-capped robin. Based on the data presented by Razeng and Watson (2012), nine of the 24 orders of invertebrates captured in this study were included in the analysis.. The orders included in the study made up 53% of the total invertebrates caught in pitfall traps, and 56% of beat samples.

3.3.4 Habitat sampling technique

Vegetation surveys were conducted in November 2015 at each of the 24 sites. At each site, four 10 m x 10 m quadrats, with centre points aligned with the centre points of the quincunx pitfall arrays. Data collected from within each quadrat included: leaf litter cover; number of pieces of CWD (Length > 30 cm, Width > 10 cm); and vegetation cover (0-1 m strata only). Additional data were also extracted from GIS data maintained by the Rottnest Island Authority (RIA). Data collected in this way included: boundaries of each habitat type, vegetation height and percentage vegetation cover.

3.4 Data analysis

Annual variation in capture rates of invertebrates was assessed using a paired sample t-test using the data collected in 2015 and 2016 for the remnant and old restored sites. This was done using the average abundance (total number of specimens), richness (number of

invertebrate orders), and diversity (Shannon-Weiner Index) for each site, with sites being paired between years. This was done for ground dwelling (pit fall) and arboreal (beat) samples separately, as were all other analyses.

Analysis of variance (ANOVA) was used to test for differences in abundance, richness and diversity between each of the four habitat types (heath, young restoration, old restoration, and remnant). Where significant differences ($P \leq 0.05$) occurred, a Tukey honest significant difference (HSD) post-hoc analysis was used to reveal the nature of the difference. This allowed me to test whether heathland sites contained distinctly different invertebrate assemblages to the three kinds of woodland sites (young restored, old restored, and remnant) sites. I was also able to test whether restoration efforts appear to be successfully transitioning areas from the heathland site state to a state similar to the remnant woodland state. Finally, I was able to assess the timescale required for the invertebrate community to transition from the heathland state to one that better resembles the one found in remnant woodland areas.

To assess differences in invertebrate taxonomic composition between the four habitat types, a Bray-Curtis resemblance matrix of sites was generated using number of individuals captured, standardised using a square-root transformation. Gross differences between sites within each habitat type were then compared using analysis of dissimilarity (ANOSIM). The similarity percentage analysis (SIMPER) procedure was then used to identify orders that contributed most to the similarity between sites within habitat types, as well as the similarity between the four habitat types (a 70% cut-off value was used). The ANOSIM and SIMPER procedures were conducted in the software package PRIMER v6 (Zhou & Zhang, 2003). The compositional similarity of the 12 sites was also visually displayed using principle coordinate analysis (PCO). This was also done in PRIMER v6.

To assess the influence of habitat complexity, patch size, isolation and fire history on invertebrate assembly, habitat data were compared with invertebrate assembly data using distance-based linear modelling (DistLM). This was done using a number of structural habitat measures, as well as digitized geographical data collected and maintained by the RIA on their geographical information system (GIS). The habitat complexity measures were derived from field measurements at each of the 24 sites surveyed in 2015, and included leaf litter cover, vegetation ground cover, CWD density, visibility, and vegetation height. The data extracted from the GIS for analysis included isolation, patch size, and time since last fire. The variation in the invertebrate assemblage at each of the 24 sites surveyed in 2015 was

measured using abundance, richness, and diversity. The DistLM used all subsets of variables, with best models chosen from these subsets as the ones with the lowest AICc value, and those within two AICc value(s) of this best model. Once again, the arboreal and ground samples were analysed separately. A MDS was then generated to visually show the variation between sites in terms of composition, with vectors to show the habitat factors most influential in explaining the variation.

3.5 Results

No significant difference was detected between the 2015 and 2016 samples, demonstrating that annual variation between the two years wasn't large, and that sampling effort was sufficient to generate repeatable results (Table 3.1).

Table 3.1: Paired t -test for six woodland sites that were surveyed in both the spring of 2015 and 2016.

	t	df	Sig. (2-tailed)
Arboreal Abundance	0.974	5	0.375
Arboreal Richness	0.808	5	0.456
Arboreal Diversity	0.809	5	0.455
Ground Abundance	1.480	5	0.199
Ground Richness	1.387	5	0.224
Ground Diversity	1.978	5	0.105

Assessment of the differences in abundance, richness and diversity across the four habitat types, revealed two significant results. These significant differences were between ground diversity and arboreal abundance (Table 3.2). Post-hoc analyses revealed that only old restoration and heathland were significantly different for arboreal abundance ($F_{1,3}$ = 68.33, p = 0.016). This indicates that the abundance, richness and diversity across the four habitat types were overall highly similar.

Table 3.2: ANOVA results displaying variation in univariate measures of invertebrate assemblage (abundance, richness, and diversity) between the four habitat types (heathland, young restoration, old restoration, and remnant woodland).

Measure of assemblage	F	Sig.
Ground Abundance	1.057	0.419
Ground Richness	1.565	0.272
Ground Diversity*	**4.682**	**0.036**
Arboreal Abundance*	**5.99**	**0.019**
Arboreal Richness	2.9	0.102
Arboreal Diversity	0.444	0.728
	* denotes significant result	

Based on Bray-Curtis similarity indices, there was a significant difference in invertebrate community composition between the different habitat types (using ANOSIM), with none of the 5000 random permutations exceeding the global R statistic for arboreal samples (R = 0.157, $p < 0.001$), and only 18 of the 5000 permutations exceeding the R statistic for ground samples (R = 0.07, p = 0.004). Pairwise comparisons between site types revealed that the only non-significant differences ($p > 0.05$) in arboreal samples were between remnants and old restoration, and between young and old restoration. Ground samples found no significant differences between young restoration and remnants, young restoration and heath, or heath and old restoration (Table 3.3). All other combinations for both arboreal and ground samples were significant.

Table 3.3: Pairwise comparisons between habitat types (ANOSIM) displaying significance of differences in composition.

		Remnant		Young Restoration		Old Restoration	
		R	P	R	P	R	P
Arboreal	Young Restoration	**0.131**	**0.006**				
	Old Restoration	0.012	0.311	0.022	0.218		
	Heath	**0.133**	**0.007**	**0.302**	**0.001**	**0.271**	**0.001**
Ground	Young Restoration	0.009	0.318				
	Old Restoration	**0.166**	**0.001**	0.022	0.23		
	Heath	**0.176**	**0.001**	0.031	0.169	0.029	0.182

Large (>1 cm) spiders (Areneae) accounted for the majority of the similarity between heathland sites for both the arboreal (66%) and ground (53%) samples. Small (<1 cm) spiders also made up the remaining (47%) in ground samples for heathland sites. Restored site similarity was mostly explained by beetles (Coleoptera), true bugs (Hemiptera) and spiders.

Remnant similarity was mostly due to consistencies in small spiders, beetles, and flies (Diptera) (Table 3.4. Dissimilarity between habitat types was primarily a function of differences in abundance between orders, such as beetles and spiders, which occurred in all habitat types (Table 3.4).

Table 3.4: Percentage contributions of orders to similarities within hab itat types and pairwise similarities between habitat types based on Bray –Curtis similarity indices (derived from SIMPER analysis in PRIMER v6).

Order	Heath	Young Rest	Old Rest	Rem	Heath & Yrest	Heath & ORest	Heath & Rem	Yrest & Orest	Yrest & Rem	Orest & Rem
Arboreal Invertebrates										
Lg.Araneae	66.01						9.66			
Orthoptera	18.35									
Hemiptera		37.33			28.74	19.58	14.64	23.85	26.28	20.96
Sm.Araneae		30.18	31.46	23.34	19.81	19.09	16.93	17.54	17.52	17.91
Sm.Coleoptera		22.25	43.44	41.72	20.24	34.94	26.96	28.78	22.5	29.81
Diptera				12.96	13.49		10.38		12.06	9.92
Total	84.36	89.76	74.9	78.02	82.28	73.61	78.57	70.17	78.36	78.6
Ground Invertebrates										
Lg.Araneae	52.64		36.16		22.34	26.02		18.01		13.64
Sm.Araneae	47.36	56.08	42.4	31.03	29.99	28.02	21.27	25.02	21.48	19.21
Lg.Coleoptera		21.56		29.46	17.48		30.32	12.54	34.03	25.13
Sm.Coleoptera				38.87	10.64		24.08		23.15	17.25
Orthoptera						9.65				
Diptera						12.38		14.47		
Total	100	77.64	78.56	99.36	80.45	76.07	75.67	70.04	78.66	75.23

The arboreal community composition in young and old restored sites appears to be clumped closer together than the communities found in heathland or remnant sites. This pattern is less clear in ground samples; however, remnants appear to be the most clumped of the four habitat types (Figure 3.2).
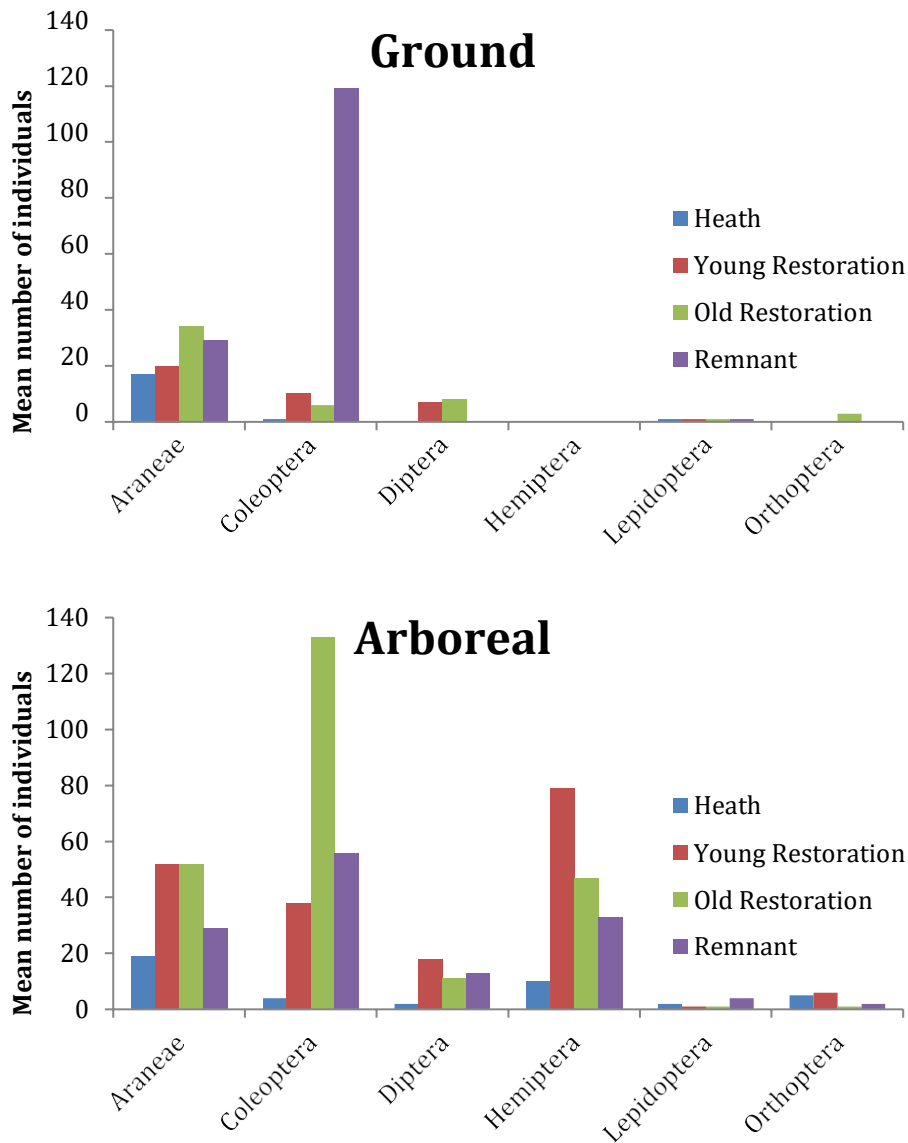
Figure 3.2: Relative abundance across the four habitat types of six orders Razeng and Watson (2012) identified as commonly occurring in avian insectivore diets.

The relative abundance of orders identified by Razeng and Watson (2012) as being most common in insectivore diets can be seen in Figure 3.2. The majority of ground samples were made up of beetles (Coleoptera) and spiders (Araneae). A total of 136 Coleoptera were captured from ground samples across the four habitat types, 119 (87%) of those captures were from remnant sites, 16 (12%) from the two restoration ages, and just 1 (1%) from heathland sites. Araneae were more evenly distributed, with the highest capture rate of 34 (34%) from old restoration, and the lowest 17 (17%) from heath. Arboreal samples had more substantial contributions from a wider range of orders. A total of 231 Coleoptera across the four habitat types were captured. The majority 133 (58%) were collected from old restoration, 56 (24%) from remnants, 38 (16%) from young restoration, and just 4 (2%) from heathland sites. Araneae were tied for highest abundance between old and young

restoration with 52 (34%), and the lowest abundance was at heath sites with just 19 (13%) individuals caught. Additionally, 169 Hemiptera were captured in the arboreal samples, an order that was completely absent from ground samples. The highest abundance 79 (47%) of Hemiptera were collected from young restoration, while the lowest 10 (6%) were from heathland. The remaining four orders made up just 9% of the ground samples collected and 6% of the arboreal samples.

The composition of invertebrates collected in each habitat type at the two strata levels can be seen in Figure 3.3. Restored sites appear to be far more closely grouped than either the heath or remnant sites for arboreal samples, but remnants are far more closely grouped for ground samples than any other site type. Beetles (Coleoptera), true bugs (Hemiptera) and spiders (Araneae) are both shown in vectors as being influential in explaining variation between the different site types.
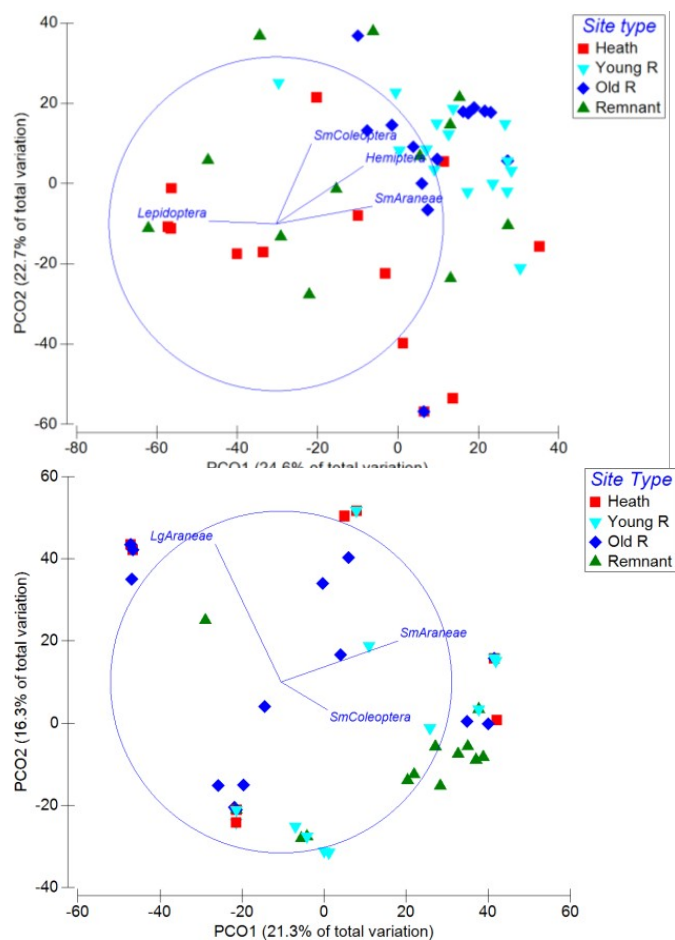
Figure 3.3: PCO displaying effect of site type on arboreal (top) and ground (bottom) invertebrate assemblage, with vectors displaying the most influential invertebrate.

Isolation was found to be the most influential factor in the DistLM for ground invertebrate assemblage. Fire age, leaf litter, and understory cover appear to be proximal factors that strengthen the model. Variation in the arboreal assemblage was best described by fire age, with patch size as the next most commonly occurring factor in each model. CWD and leaf litter were the next two most influential factors (Table 3.5). It should also be noted, that fire age was weakly correlated with leaf litter ($F = 0.408$, $P = 0.048$) and understory cover ($F = 0.541$, $P = 0.006$), but not CWD ($F = 0.474$, $P = 0.072$), so caution needs to be taken when interpreting these results. Based on the AICc values, there is little difference between the listed models, suggesting that factors 2, 3 and 4 in the respective models are only proximal factors.

Table 3.5: Results of distance -based linear modelling (DistLM) for invertebrate community composition based on structural, temporal, and spatial habitat variables. Only the best models (based on AICc values) are included.

Substrate	Model #	Factor 1	Factor 2	Factor 3	Factor 4	AICc	R^2
Ground	1	Isolation				154.31	0.150
	2	Isolation	Fire Age			155.2	0.210
	3	Isolation	Leaf Litter	Understory Vegetation		156.44	0.260
Arboreal	1	Fire Age				144.1	0.145
	2	Fire Age	Size	CWD		144.86	0.299
	3	Fire Age	Size	CWD	Leaf Litter	145.87	0.361

3.6 Discussion

The community assemblage did not differ significantly between the two successive years studied. While the comparison was only between two consecutive years, this result is encouraging as significant deviation from one year to the next may indicate that either the variability from one year to the next is large enough to make this study redundant for predicting future variability, or that insufficient sampling was done. Either way, significant differences would have diminished the value of these findings.

Comparisons between the four habitat types at the two strata levels based on overall abundance, diversity and richness, revealed significant differences in ground diversity and

arboreal abundance. Ground abundance and richness, as well as arboreal richness and diversity weren't found to significantly differ. Ground diversity was highest in remnant areas, while arboreal abundance was highest in old restoration. A well-documented limitation of univariate biodiversity indices in disturbed habitats is that disturbance specialists may have replaced more specialised species, resulting in similar overall biodiversity, but a change in composition (Devictor, Julliar & Jiguet, 2008). Thus, caution should be taken when comparing variation in biodiversity indices. A possible explanation for the differences in ground diversity is that relic ground dwelling species may still persist in remnant areas, but are yet to recolonise restored areas. The similarity between remnant ground samples was best explained by the presence of spiders Araneae and beetles Coleoptera of the two different size classes. Spiders contributed the most to within group similarity in all four site types at the ground level, while large beetles were only identified as important contributors to two habitat types (remnants and young restoration), and small beetles were only significant in remnants. This may be an indication that the beetle community is slow to recolonise restored areas, either due to dispersal barriers, or reduced suitability in restored habitats.

Beetles can have a number of important trophic roles as herbivores, carnivores, omnivores, and scavengers (Davies & Margules, 1998; Lassau *et al.*, 2005; Schaffers *et al.*, 2008; Vandewalle *et al.*, 2010), pollinators, seed dispersers and decomposers (Grimbacher *et al.*, 2007; Nichols *et al.*, 2008; Gibb & Cunningham, 2010; Vandewalle *et al.*, 2010). They are also the most important prey type for many insectivorous birds (Poulin, Lefebvre, & McNeil, 1994; Wilson *et al.,* 1999; Buchanan *et al.,* 2006; Razeng & Watson, 2012). Razeng and Watson (2012) assessed the dietary records of 26 declining woodland birds in South Australia, 13 of which were ground-foraging insectivorous passerines including the red-capped robin.

Beetles were the dominant prey group in nine of the 13 ground-foraging insectivores. Poulin *et al.* (1994) also found beetles to be the most commonly consumed prey group for land birds in Venezuela. Buchanan *et al.* (2006) and Wilson *et al.* (1999) also found beetles to be of disproportionate importance to birds in both the United Kingdom and northern Europe. This demonstrates the global importance of beetles as a food source. Unfortunately, there is evidence that current habitat restoration practices may not be adequately providing important environmental variables, such as native vegetation structure and soil condition, which limits beetles' capacity to recolonise restored habitats (Jellinek, Parris, & Driscoll,

2013). There is evidence that beetle communities can struggle to persist in heavily cleared landscapes (Hopp *et al.,* 2010), and changes in habitat conditions and arrangements can influence richness, abundance and diversity. This study's finding that remnants are most similar based on their beetle communities, while heath sites had less consistent beetle communities is likely a reflection of a reduction in habitat viability for beetles as areas have been converted from woodland to heathland habitats.

The ground-dwelling invertebrate communities in restored areas also don't appear to be moving towards a state that resembles the remnant state. Remnants and heathland sites were unsurprisingly found to be significantly different with regards to ground invertebrates, but unlike in the arboreal samples, old restored sites were also found to have significantly different ground invertebrate compositions to remnants. Given that the old restored areas were 30-50 years old, this demonstrates that either development is too slow to have been detected in this study, or the systems are developing along a divergent pathway from the remnant habitat state. In Chapter four a number of habitat characteristics, such as understory vegetation cover, vegetation height, leaf litter and presence of CWD are compared between restored and remnant areas. These variables were selected for their relationship to bird microhabitat selection, but studies have also found that factors like habitat complexity, CWD, and vegetation height can relate to invertebrate richness and diversity (e.g. Longcore, 2003; Higgins *et al.*, 2014). Two habitat variables that were found to be significantly different between Rottnest's remnant and restored areas are the presence of CWD, and vegetation height. CWD is an ecologically important resource for numerous ground dwelling invertebrates (Braccia & Batzer, 2001). Unfortunately, CWD is slow developing, meaning without active introduction, it may be absent from restored areas for a long time (Jonsson 2000; Craig *et al.*, 2012).

The results of the arboreal data are more in line with expectations than the ground data. Remnants were found to be significantly different from heathland sites, but mature restored sites were found to resemble remnants. Young restoration was found to be significantly different to heathland, demonstrating that a change in the invertebrate community can be detected after 7-12 years of development, but it wasn't until the restored woodland matured that it became statistically similar to the remnant state. As predicted, the restored habitats were also found to be similar to one another. The true bug (Hemiptera) contributed to within group similarity in young restoration only, which may suggest that they are a group best suited to young restored sites, which may help distinguish young restoration from

other habitat types. The short gestation period, high reproductive potential, high population size, and responsiveness to microhabitat changes of invertebrates make them an ideal taxon to track year-to-year changes in site conditions (Longcore, 2003; Higgins *et al.*, 2014). Large spiders and grasshoppers were the two most significant groups of invertebrates for explaining between site similarities in heathland arboreal samples. Both were found in Razeng and Watson's (2012) study to be consumed by as many as 11 of the 13 ground-foraging insectivorous passerines. It should however be noted that these were less frequently consumed than the more important beetles, ants, butterflies and moths. Restored sites, on the other hand, were dominated by beetles (Coleoptera), true bugs (Hemiptera) and spiders. The invertebrate community was also found to be much more abundant in restored areas than any other assessed site type. This is a promising result for the success of restoration efforts, given the importance of invertebrates as a food source and performers of important tasks like pollination, which can be problematic in restored habitats (Liu *et al*., 2010; Cordingley, 2012; Jellinek, Parris, & Driscoll, 2013).

Patch isolation and the permeability of the surrounding matrix have been shown to influence assembly in a number of ways. Much like vertebrates and plants, invertebrates have been shown to be more prone to extinction on smaller and more isolated fragments (Fishcher & Lindenmayer, 2007; Boscolo & Metzger, 2011). This can be attributed to variation in colonisation and extinction rates, as well as increased selection pressure in smaller patches (Hanski & Ovaskainen, 2000). Oliver *et al.* (2006) found that paddock trees in grazed native pastures contained distinctly different invertebrate communities to the surrounding agricultural landscape. They also found that those community level differences were reflected in differences in soil and leaf litter variables that followed gradients away from the paddock trees. The study demonstrated how the provision of the necessary resources can alter the invertebrate assembly, and may provide a "stepping stone" for animal movement across the landscape (Manning *et al.*, 2009; Nadkarni & Haber, 2009). Restoration efforts on Rottnest have attempted to convert heathland areas back to woodland areas through the use of fire and mechanical slashing to reduce vegetation competition (see Chapter five for details), the introduction of fences to exclude quokkas that would otherwise graze on seedlings and the planting of woodland species seedlings. These steps are an effort to decrease the isolation of existing woodland patches, and extend the available woodland habitat on the Island. Given that patch size and isolation were identified in the linear model as important factors, it is clear that habitat fragmentation has influenced the invertebrate community at both measured strata levels.

As mentioned previously, wild fires and controlled burns have changed the vegetation profile of the Island in the past. Controlled burns were used to clear unwanted vegetation from restoration sites at an early stage of site preparation until 1986 (Winn, 2008). The impact fire has on invertebrate assembly is poorly understood, and no information is currently available on the impact fire has on the terrestrial invertebrate communities on Rottnest. York (1999) found that Australian dry eucalypt forests that had been subjected to frequent low-intensity fires, commonly had 41-82% lower abundances of spiders, ticks and mites, pseudoscorpions, woodlice, springtails, bugs, beetles, ants and insect larvae in leaf litter than adjacent, unburnt areas. This was attributed to reduced leaf litter, simplified habitat structures, and less available moisture. Anderson (1991) documented differences in the profiles of ant functional groups in mine site restoration that had been exposed to different fire regimes in Northern Australia. Higgins *et al.* (2014) found that stand-replacing wildfires in Colorado USA resulted in higher abundances of the major invertebrate taxonomic groups, with the exception of spiders, after five years than in comparable unburnt areas. This study found that fire age had a significant correlation with both ground and arboreal composition. The initial experimental design wasn't set up to look at fire age, and so the effects of fire on the invertebrate community couldn't be readily analysed without additional data gathering, which was beyond the scope of this study. Further research into how the invertebrate community responds to fire would be of considerable value, given that Rippey and Hobbs' (2003) state and transition model describes the ways fire can transition areas from one stable habitat type to another.

Overall, the species composition found in restored areas does appear to better resemble those found in remnants than in heathland sites for both ground and arboreal invertebrates. The biodiversity of ground-dwelling invertebrates was similar in all three woodland sites, but much lower in heathland sites. This demonstrates that ground biodiversity does increase in response to woodland restoration efforts, and much like in the Oliver *et al.* (2006) study, the provision of necessary resources appears to be facilitating recolonisation events. Remnants and old restored sites tended to contain a greater plant diversity and structural complexity (as discussed in Chapter four), which may explain the lower rates of internal similarity in their invertebrate community. The compositional differences that were detected in this study weren't fully explored as it was beyond the scope of this study. Further study into the distribution of specialist and generalist invertebrate taxons would be valuable in the future.

The focus of this study was on invertebrates as a food source (explored in greater detail in Chapter four), and as such, the invertebrates were only coarsely sorted to order, and little attention was given to the invertebrates' biology.

It is clear that the invertebrate community has been influenced by the fragmentation and changes in fire regimes of the past. The woodland restoration program appears to have influenced the invertebrate assembly on the Island, as the communities found in restored areas no longer resemble those found in heathland areas. Arboreal invertebrates appear to be responding better to restoration efforts than ground dwelling invertebrates. Future work into the impact of fire on the woodland invertebrate community, and the distribution of beetles in restored and remnant areas are recommended, as both may have management implications in the future. Overall, there is no clear difference in the quality of food resources for birds between restored and remnant areas. Beetles appear to be more diverse and abundant in remnants, but overall invertebrate abundance on vegetation is higher in restored areas. Chapter four explores how robin behaviour varies between restored and remnant areas, and whether the variation in invertebrate assembly is reflected in the birds' behaviour.

Chapter 4: Habitat quality measured using animal behaviour and microhabitat selection

4.1 Introduction

In chapter one, I outlined the progress of restoration ecology as a scientific discipline, and discussed the potential benefits of incorporating animal behaviour metrics into assessments of restoration development. Restoration practitioners often organise management goals using conceptual frameworks such as state-and-transition models (STM). A STM can be used to characterise the various stages of development from the pre-intervention state, to the desired end state where monitoring and management are no longer required. STMs are also useful for identifying possible deviated states that may arise during development, and offer management strategies to return the successional trajectory to the desired path (Stringham *et al.*, 2003; Bestelmeyer *et al.*, 2004; Rumpff *et al.*, 2011). The model requires appropriate completion goals that will lead to the recovery of an area to a state that adequately resembles an appropriate local indigenous reference ecosystem (McDonald *et al.*, 2016). The intensity of restoration effort required will depend on a number of factors, such as the severity of the degradation, the resilience and regenerative capacity of the area, and any socio-economic factors associated the restoration effort (McDonald *et al.*, 2016). Currently, most STMs are centred around abiotic and flora based recovery, with little to no consideration of faunal recovery (Craig *et al.*, 2015).

Successful restoration efforts are dependent on correctly predicting the successional trajectories initiated by the restoration effort, and adequately considering the processes that need to be restored (Hilderbrand *et al.*, 2005). Unfortunately, ad-hoc style adaptive management is still often necessary, as scientific knowledge about how best to restore ecosystem functions, and accelerate successional trajectories is often limited (McDonald *et al.*, 2016). This can be problematic, as errors made at the outset of the restoration effort can be difficult, and expensive, to correct retrospectively (Perring *et al.*, 2015). Improving our understanding of how best to implement and monitor restoration development, will reduce the need for ad-hoc management. This will result in better and more efficient restoration outcomes.

Many restoration assessments in the past have been criticised for oversimplifying and overlooking important components. It has been argued that an overemphasis has been placed on flora, while fauna has seen inadequate attention (Halle & Fattorini, 2004; Craig *et al.*, 2015). Where fauna are considered, assessments often only involve indices of animal biodiversity, specifically species diversity and richness (Lindell, 2008). This kind of data can be problematic as it may be inadequate for answering important questions. If the goal of a restoration effort is to provide habitat capable of supporting a stable and self-sustaining population of a specific species, then there is an assumption that the species should be able to maintain a net reproductive rate equal to or greater than one. Unfortunately, presence/absence data alone is insufficient for determining whether an adequate reproductive success rate and population replacement rate has been achieved (Aldridge & Boyce, 2007; Lindell, 2008).

Current trends in research into the management of restored ecosystems suggest that increased use of integrated approaches may be beneficial. Numerous studies have shown the benefits of focussing on the interactions between flora and fauna, when determining how best to monitor and manage ecosystem development (e.g. Kaiser- Bunbury, Traveset & Hansen, 2010; Daws & Koch, 2015; Schleuning, Fründ, & García, 2015). This literature supports the argument that biodiversity should be seen as secondary to the interactions between organisms when dealing with ecosystem management. Within an ecological system, numerous animals may depend on plants for food and shelter, while plants depend on animals to facilitate processes like pollination and seed dispersal (Lindell, 2008). A disruption in these processes could destabilise the equilibrium in the system.

For restoration efforts aimed at restoring important biological interactions, there is a need to understand how species interact with their habitat. Habitat, which can be viewed from either a structural or functional perspective, can be difficult to measure as it is inherently subjective, and the degree of functionality can be difficult to assess. Structural habitat (e.g. vegetation or land cover types) is easier to measure, as it generally relates to how humans perceive habitat, and can be measured using variables like vegetation height and density (Van Dyck, 2012). Functional habitat measures are more complex, as they are an attempt to quantify resource-based habitat distribution relative to an animal's movement (Breedlove *et al*., 2004). These resources may be either consumables or conditions, and their availability is dependent on the animal's perception of the world around them (Van Dyck, 2012). For fragmented landscapes and highly mobile species, an animal's functional habitat

may be distributed across several structural habitat types. This can complicate researchers' attempts to quantify the relative quality of patches of a specific structural habitat type.

When selecting habitat to use, animals are likely to select areas that provide high intrinsic value, meaning high resource densities, protection from predators and parasites, and any other factors likely to enhance survivorship and offspring production (Muller *et al.*, 1997). Given that animals may not see some habitat units in the same way as humans perceive them (e.g. disturbance history or land cover type), the Umwelt-concept from ethology may be a useful approach to understanding how animals perceive the habitat around them (Van Dyck, 2012). Animals are likely to view the range of available resources in a mosaic landscape made up of both restored and remnant areas in a different way to humans. By taking a resource-based approach to habitat assessments, that considers the distribution of resources (consumables and conditions), we may be able to gain greater insight into how animals perceive their own environment (Van Dyck, 2012).

Understanding habitat quality is a complex, but important task for ecologists and restoration practitioners (Johnson, 2007). Functional habitat, by definition, can only really be considered at the species level. Thus, functional habitat quality is inevitably highly subjective (Van Dyck, 2012). In addition, there are a number of complicating factors, such as reproduction, survival, and abundance not necessarily being positively correlated with one another (Van Horne, 1983). Intraspecific and interspecific interactions (e.g. competition, predator-prey relationships, conspecific attraction) can influence species occurrence across a landscape, and potentially push animals into sub-optimal habitat (Bock & Jones, 2004; Campomizzi *et al.*, 2008). Finally, different species have varying capacities to alter their behaviour and habitat selection, which can further complicate researchers' attempts to determine the relative quality of the animal's habitat (e.g. Bock & Jones, 2004; Nielson*, et al.*, 2013; Bennett, 2013).

Despite all of these complicating factors, there have been numerous cases of behavioural patterns providing an effective indication of habitat quality (Lindell, 2008). Vaughan *et al.* (1996) used the feeding rates of bats upstream and downstream of 19 sewage outputs to determine whether an impact of poor water quality could be detected. The feeding rate of both species was found to be lower at downstream sites than upstream sites, demonstrating a conservation issue that is directly influencing the local wildlife. Johnson (2000) used observational foraging data from three species of warblers, alongside arthropod sampling

using a 'branch clipping' technique to verify that the sampling technique correctly recorded a representative estimate of prey availability. Without detailed foraging behaviour data and stomach contents data, it would be very difficult to correctly sample invertebrate prey availability. The study's assessment of food availability closely matched the observed foraging rates on different plant species, demonstrating the accuracy with which the sampling technique could be used to sample prey availability for foliage-gleaning species.

Resource selection functions (RSF) can be a useful way to identify how animals select habitat, and which habitat variables should be assessed to measure habitat quality (Johnson, 2000; Chetkiewicz & Boyce, 2009; Fattebert *et al.*, 2015). An animal's behaviour in a heterogeneous environment is shaped by its experiences and expectations. By observing the microhabitats an animal chooses to occupy, the risks it is willing to take, or the point at which it will abandon a resource (e.g. giving up densities [GUD]) we can begin to understand the factors determining the quality of a specific habitat (Jacob & Brown, 2000; Persson & Stenberg, 2006).

Foraging technique and prey attack rates have been shown to directly relate to food availability, which is a crucial habitat component (Carter & Dixon, 1982; Vaughan *et al.*, 1996; Morrison *et al.*, 2010). In a study on the variation of breeding success of blue tits (*Parus caeruleus*) in high-quality deciduous woodland, compared with low-quality coniferous woodland, behavioural analysis was used to identify the mechanism behind the variation in habitat quality (Stauss *et al.*, 2005). The researchers measured both the breeding success, and parental feeding behaviour. The birds in high quality habitat were found to travel smaller distances than those in low quality areas. The difference in distance travelled by the birds was reflected in a significant difference in the amount of food provided to nestling birds (Stauss *et al.*, 2005). The study clearly demonstrates how supplementary behavioural animal data can provide a clear rational for differences in the quality of two different habitats (Lindell *et al.*, 2008).

Foraging rate has been shown to have a positive relationship with food availability in fish and birds (Repasky 1996; Delestrade 1999; Shepherd & Boates 1999; Marchand *et al.*, 2002; Wellenreuther & Connell, 2002; Kilgo 2005). Foraging rate has also been shown to directly influence fledgling growth (Naef- Daenzer, Naef-Daenzer & Nager, 2000; Wilkin, King & Sheldon, 2009) and in some cases, breeding success (Stauss *et al.*, 2005). It should be noted, however, that some studies have shown that adjustments in parental behaviour in areas

with poorer food resource availability can yield equal biomass per hour per chick, and no discernible differences in breeding success (Naef-Daenzer *et al.*, 2000; Tremblay *et al.*, 2005; Wilkin *et al.*, 2009). Despite the somewhat conflicting evidence provided by these studies, it is clear that food resource quality and animal behaviour are interlinked, and it is unwise to make assumptions about the relationships between food resource availability, foraging behaviour, parental attentiveness and breeding success in one system, based on the results of studies in other systems with other species.

This chapter involves a study of the microhabitat selection and behavioural patterns of an insectivorous woodland bird species in restored and remnant areas on Rottnest Island. The red-capped robin (*Petroica goodenovii*) is a species previously identified as being highly sensitive to woodland condition and recognised as a declining woodland bird (Razeng and Watson, 2012). While the population on Rottnest appears to be stable, they have been described as being of conservation significance based on their isolation from the mainland population (Baker *et al.*, 2003; Saunders & de Rebeira, 2009; Mather, 2010; Stevenson, 2011). Red-capped robins are one of the priority species identified by the Rottnest Island Authority (RIA), and a goal of its woodland restoration effort is to create viable habitat for this species (Baker *et al*., 2003; Mather, 2010; RIA, 2014).

Red-capped robins are typically described as ground-foraging insectivorous birds (Razeng & Watson, 2012), that typically forage using the 'pounce' technique, as described by Holmes and Recher (1986). This method involves flying from an elevated perch down onto the ground to capture a prey item. As such, ground dwelling invertebrates are an important food resource for this species. The results of the invertebrate surveys described in Chapter three suggest that the diversity of ground dwelling invertebrates was higher in remnant areas than restored areas. Additionally the abundance of beetles (*Coleoptera*), an order of invertebrates that numerous studies have found to make up the majority of many insectivorous birds' diets (Poulin *et al.,* 1994; Wilson *et al.,* 1999; Buchanan *et al.,* 2006; Razeng & Watson, 2012) was found to be considerably higher in remnants than restored areas. Of the four habitat types (heathland, young restoration [7-12 years], old restoration [30-50 years], remnant), each of which had an equal trapping effort, 87% of the beetles captured on the ground were from remnant sites, while 58% of the beetles captured in arboreal samples were in old restoration, and a further 16% were from young restoration.

Additionally, the overall abundance of invertebrates was found to be significantly higher in

restored areas than remnants or heathland sites. This may indicate that restored areas have superior arboreal foraging microhabitat conditions, but inferior ground foraging habitat compared to remnants. By assessing the foraging behaviour of the birds, specifically in relation to the substrate from which they capture prey, this chapter attempts to determine whether the results of Chapter three reflect differences in resource availability from the robins' perspective and influence their foraging behaviour between restored and remnant areas.

Aside from assessing the quality of restoration in terms of foraging habitat, the quality of breeding habitat is also of critical importance. The capacity of a restored area to positively contribute to annual recruitment is a necessary target for restoration aiming to extend the birds' viable habitat. Stevenson's (2011) study on the impact of ravens on Rottnest Island bush birds concluded that the restoration efforts are alleviating the pressure being applied to bush birds through nest predation. Habitat fragmentation has been shown to increase nest predation rates, while increasing habitat complexity, specifically relating to foliage height diversity has been shown to reduce nest predation rates (Marzluff & Ewing, 2001). Red-capped robins have been shown to be strongly affected at the population level by variation in habitat size and shape (see Major *et al.*, 1999), and nest predation is the main cause of nest failures for red-capped robins (Dowling, Antos, & Sahlman 2003; Dowling, 2003). In Chapter two, juvenile robin density was found to be significantly higher in remnants than restored sites. This suggests that recruitment rates may differ between the two site types, or that birds are actively moving from restored areas to remnants after the birds fledge.

While ongoing monitoring of fauna in restored areas is not uncommon, the information gathered is generally restricted to presence absence and abundance type data. This chapter explores the possible benefits of going beyond those more basic forms of assessment, and assessing whether the birds behave differently between the different habitat types with respect to breeding-related behaviours, it may be possible to determine why there is a difference in juvenile recruitment rates between remnants and restored areas. This may have management implications if restoration practitioners want to improve restoration outcomes and remove any factors that are reducing the breeding habitat quality of restored sites. If the issue is simply related to edge effects and nest predation, then designing restoration sites so they act as buffer zones that increase the

area to perimeter ratio may improve the breeding habitat quality for the birds. A number of studies have shown that the creation of corridors can be detrimental to the birds, and have argued that infilling, or surrounding existing fragments is more beneficial to woodland birds (Major *et al.*, 1999; Fischer & Lindenmayer, 2007). As such, this study looked at the breeding behaviour and recruitment rate of birds in restored and remnant areas as a way of assessing the quality of breeding habitat provided by the two site types.

4.2 Hypotheses:

1. Differences in robin feeding and breeding related behaviours will reflect differences in microhabitat characteristics related to those behaviours.

2. Variation in foraging microhabitat selection will reflect variation in prey availability as defined by the results of Chapter three. Robins will collect prey from foliage more frequently in restored sites than remnants, and from the ground more frequently in remnants than restored sites.

3. Differences in breeding behaviour will reflect the apparent difference in breeding habitat quality between restored and remnant areas (Chapter two). Remnants have higher population densities of juvenile robins, and so are likely to be superior breeding habitat

4.3 Field methods

4.3.1 Site selection

A total of 11 sites were selected for this study based on the frequency with which robins had been sighted at each site during the Birdlife bush bird counts (Mather, 2010), and from personal observations (Figure 4.1). This was done as the logistics of moving between sites was a limiting constraint, and it was considered a priority to maximize the sample size as much as possible. Of the 11 sites selected for this study, four were remnants and seven were restored sites. The western most remnant site was excluded from this study as it is heavily degraded, and no robins have been recorded at that site in the past two decades of bird surveys. The northern most remnant site was also excluded as it is very small, located in the middle of a camping ground, and because it has been revegetated with non-native eucalypt trees. The eastern most site was excluded from this study as very few robins (2) had previously been seen in that area. The site is also part of the settlement area known as the Kingstown Barracks. The restored sites chosen for this study were all located on the eastern

half of the island, as these areas typically have higher densities of robins, and were made up of a range of different sizes. All restored sites were at least 20 years old.

Figure 4.1: Map of Rottnest Island. The red spots indicate sites used in this component of the study. Original image courtesy of Landgate.

4.3.2 Bird banding

A total of 50 birds (23 adult male, 15 adult female, and 12 unsexed [juvenile]) were banded for this study. For details on banding techniques used, see Appendix A. Birds were fitted with split colour metal bands making them individually identifiable with binoculars, as well as a numbered Australian Bird and Bat Banding Scheme (ABBBS) metal band. As the site had a pre-existing banding project set up, some birds observed in this study had ABBBS bands already fitted, but didn't have the split colour metal bands.

4.3.3 Bird behavioural observations

Each site was visited at least six times on non-consecutive days during October to December 2015. There is evidence that variation in activity levels in the first 4-5 hours after sunrise is minor, but that activity levels drop during the middle of the day, before rising again to near the morning levels at dusk (Verner & Ritter, 1986; Bibby, 2000). To compensate for decreased activity levels, surveys were conducted across a broad part of the day. Surveys were conducted between approximately 6.00am to 11.00am, and then from 2.00pm until 5.00pm.

In addition to collecting up to five-minutes of observational data from as many banded birds as possible, I also collected five-minutes of data from any unbanded birds sighted, provided I was certain only five minutes had been collected on that bird. This meant that up to five minutes of data could be collected for unbanded adult males, unbanded females and

unbanded juveniles. In one case, where a banded bird was seen with an unbanded mate (that also carried a leg abnormality), it was possible to individually identify the unbanded individual through its association with the banded bird, and its leg abnormality.

The method used for collecting bird behaviour data was derived from a study that looked at differences in the feeding ecology of male and female Raso Larks *Alauda razae* in Cape Verde (Donald *et al.*, 2007), which is very similar to the focal animal sampling methods described by Altmann (1974). In both studies, birds were located by walking random transects and looking and listening for the birds. Once a bird was located, its band ID, age and gender were noted, and a timer was started. Over the next five minutes, the frequency and time the bird spent performing a number of different activities, and using different microhabitats was recorded on a voice recording phone application, and later timed and tallied using a timer.

Activities recorded in this way included: first forage technique used [as described by Holmes and Recher (1986)], any subsequent foraging attempts, substrate prey was taken from, description of the prey item taken, time spent calling, preening, nest building, territorial displays (aggression between two males), courting (male and female interactions often involving feeding and calling to one another), assisted feeding (adults feeding juveniles, and males feeding females), description of habitat being used (restoration or remnant), description of whether the bird was near the edge or middle of said habitat type and a description of the nearest adjacent habitat type, approximate perch height use (to the nearest metre), number of perch changes, number of long flights (>10 m).

4.3.4 Vegetation surveys

At each site, surveys were conducted at four 100 m^2 (10 m x 10 m) quadrats in random locations at least 20 m from the edge of the woodland habitat, and 20 m away from each other. The variables measured relate to microhabitat characteristics the birds are likely to rely on while foraging, or macrohabitat characteristics associated with patch size, isolation and vegetation density, which can influence predation pressures related to breeding. In each quadrat, the percentage of ground cover (leaf litter, bare ground, or vegetation) was estimated; the presence of horizontal branching at three separate strata levels was recorded (0-1 m, 1-2 m and >2 m); the number of pieces of coarse woody debris (CWD) (length > 30 cm, width > 5 cm) were counted; four visibility measures were taken using a rangefinder, with each measurement being aligned with a compass cardinal point.

Additional data were also extracted from GIS data maintained by the RIA. Data collected in this way included: woodland boundary length, woodland patch size, mean vegetation height and percentage tree canopy cover.

4.3.5 Prey availability

Invertebrate prey availability data used in Chapter three was collected concurrently with the behaviour data collected for this chapter. As such, the invertebrate community in restored and remnant sites could be compared. For details on how invertebrates were collected and sorted, see Chapter three.

4.3.6 Robin population density and recruitment

Approximately 12 months after the behavioural surveys were completed, a second bird survey was conducted in which the population density of robins was estimated, and demographic data relating to the age structure of birds found at each site was counted. This survey is described in Chapter two.

4.4 Data analysis

The difference in availability of various microhabitat characteristics in restored and remnant areas was first compared using an analysis of similarity (ANOSIM) in which several microhabitat variables were compared together, and then those same variables were tested individually using analysis of variance (ANOVA) to determine whether any pair-wise differences were present. The microhabitat variables tested were availability of horizontal branches at height classes (0-1 m, 1-2 m, and >2 m), ground substrate (leaf litter, bare ground and vegetation), average vegetation height, and visibility.

To determine whether robins behave differently in the two habitat types (restored and remnant woodland), robin behaviour was first compared using an ANOVA. This test compared the rate that all observed robins in each habitat type performed various behaviours. The behaviours assessed included: foraging, calling, courting, parental feeding, and preening/resting, as these were behaviours observed at least five times each.

To further examine robin behaviour specifically related to foraging, the birds' prey attack method was compared between restored and remnant areas using an ANOVA. The different

prey attack methods were grouped into four foraging behaviour categories based on the microhabitat characteristics involved in the manoeuvre. These categories were as follows: 1. Pounce (requires perches, open understory, visible prey on the ground); 2. Glean and probe (requires prey that can be captured over short distances, possibly by digging into bark or leaves); 3. Hawk, hover & snatch (requires open area that allows the bird to locate and capture prey while on the wing); 4. Fed by a parent (typically the prey item would have been captured via one of the other three categories). A bar chart with standard error bars was then used to visually display the relative frequency with which the behaviours were performed.

The frequency with which the birds in used different substrates in restored and remnant areas to source prey items was then compared using an ANOVA. The substrates birds were recorded foraging from were grouped into the categories: leaf litter, bare ground, vegetation and air. The relative frequency with which each foraging substrate was used was then visually represented with a bar chart and standard error bars. The relationship between foraging behaviour and prey availability was assessed using a Pearson correlation. The rate at which each foraging technique was used at each site was compared with the abundance (total number of invertebrates caught) and diversity (number of orders encountered) for both arboreal and ground dwelling invertebrates.

Finally, the relationships between time spent performing behaviours related to territoriality and reproduction (calling, courting, territorial displays and nest building) were compared with the adult and juvenile robin density estimates using a Pearson correlation.

4.5 Results

Overall restored and remnant patches weren't found to significantly differ when all microhabitat measures were analysed simultaneously (R = -0.061, P = 0.724). When each microhabitat was analysed individually, vegetation height and CWD were found to significantly differ between the two habitat types (Table 4.1). All other measures were not found to significantly differ between site types. Remnants had an average vegetation height of 4.89 m (sd = 2.74), while restored sites had an average height of 4.16 m (sd = 0.94 m). Remnants also had a CWD density of 6.05 pieces / 100 m^2 (sd = 1.53 pieces / 100 m^2), while restored sites had just 4.50 pieces / 100 m^2 (sd = 2.30 pieces / 100 m^2).

Table 4.1: Results of an ANOVA testing differences in microhabitat characteristics between restored and remnant sites. Significant differences are shown in bold with asterisks.

Microhabitat characteristic	df	F	Sig.
Understory veg	10	.471	.500
Visibility	10	1.003	.327
Veg height*	10	**5.381**	**.030**
Leaf litter cover	10	.010	.920
Coarse woody debris*	10	**9.402**	**.006**
Horizontal branching 0-1m	10	.458	.505
Horizontal branching 1-2m	10	.920	.348
Horizontal branching >2m	10	.505	.485

There were no significant differences detected between restored and remnant sites for time spent foraging, calling, assisted feeding, or preening/resting (Table 4.2), however assisted feeding was very close to significant. There was a difference in time spent courting, and on territorial displays, and nest building, with the majority of all three occurring in remnant areas. This suggests that while the birds occupy and use both habitat types, differences in the ways they use the two habitat types may exist, and those differences may reflect differences in habitat quality, as they relate to reproductive activities

Table 4.2: Results of an ANOVA testing differences in bird behaviour between restored and remnant sites. Significant differences are shown in bold and with an asterisk.

Behaviour	df	F	P
Foraging	10	1.065	0.329
Calling	10	3.733	0.085
Courting*	10	**8.851**	**0.016**
Territorial display*	10	**7.04**	**0.026**
Nest building*	10	**5.723**	**0.04**
Assisted feeding	10	4.795	0.056
Preening/resting	10	0.013	0.913

Further enquiry into foraging behaviour revealed that pounce foraging was the most commonly observed technique used (Figure 4.2), making up 68% of all observed foraging. Neither pounce foraging nor parental feeding were found to be significantly different between restored and remnant habitat (Table 4.3). Glean/probe and the hawk/hover/snatch categories were both found to be significantly different between restored and remnant sites, and both were found to be higher in restored than remnant habitats.
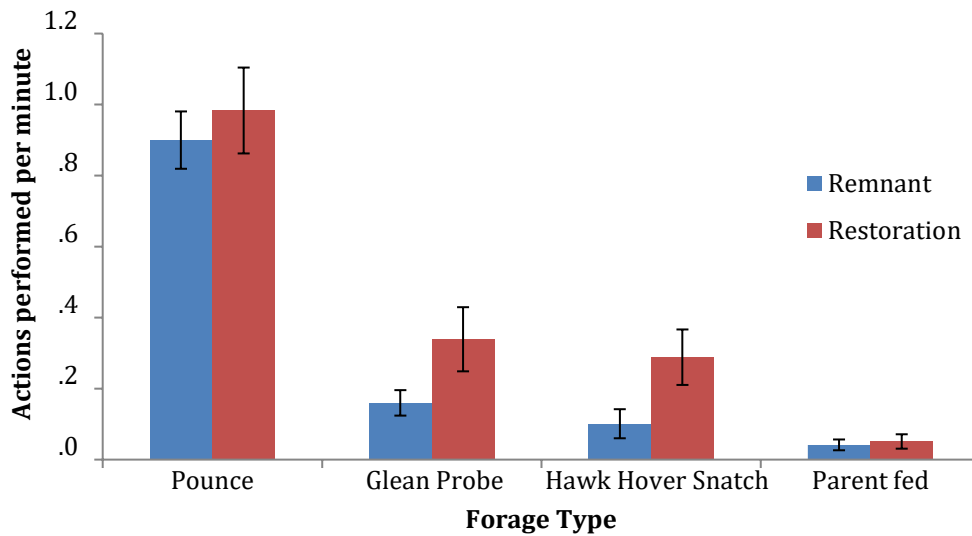
Figure 4.2: Relative frequency with which each foraging technique was performed in restored and remnant areas by red-capped robins.

Leaf litter was the most commonly used ground substrate from which prey items were captured in both restored and remnant areas (Figure 4.3). Vegetation was more commonly used in restored areas than remnants. No significant difference was found between the frequency with which items were captured on the ground (including both leaf litter and bare ground), or the air (Table 4.4). Vegetation was the only substrate in which a significant difference was detected, with more frequent use occurring in restored areas (Table 4.4). This difference in substrate use is reflected in the difference in foraging technique used, as the majority of glean/probe and hawk/hover/snatch foraging occurred in restored areas, and these techniques are commonly used when feeding on vegetation.

Table 4.3: Results of an ANOVA testing differences in foraging strategy use between restored and remnant sites. Significant differences are shown in bold and with an asterisk.

Foraging Strategy	df	F	Sig.
Pounce	10	0.358	0.55
Glean/Probe*	**10**	**4.265**	**0.04**
Hawk/Hover/Snatch*	**10**	**5.297**	**0.022**
Parent	10	0.142	0.707
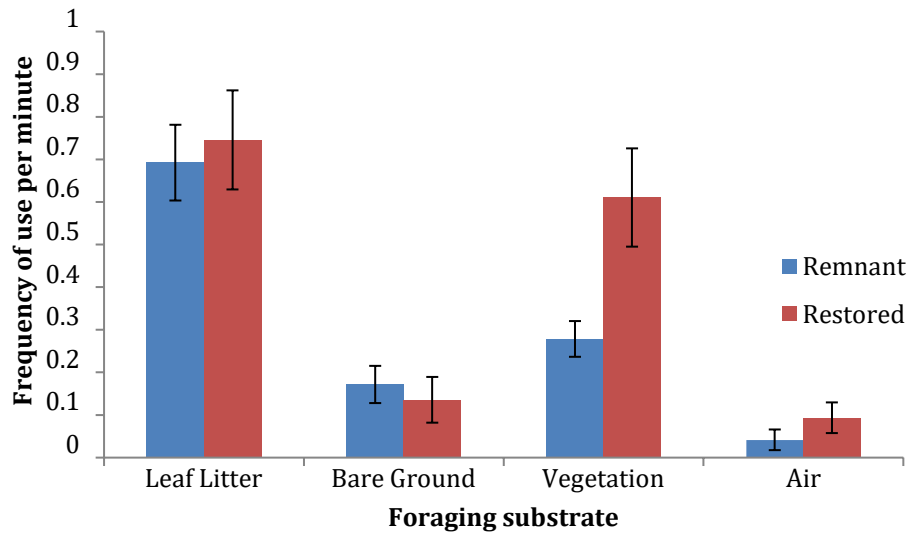
Figure 4.3: Relative frequency with which each substrate was used to collect prey items in restored and remnant areas by red-capped robins.

Table 4.4: Results of an ANOVA testing differences in frequency of foraging microhabitat substrate used by red-capped robins between restored and remnant areas. Significant differences are shown in bold and with an asterisk.

Foraging Strategy	df	F	Sig.
Leaf litter	10	0.138	0.711
Vegetation*	**10**	**9.332**	**0.002**
Bare ground	10	0.274	0.601
Air	10	1.532	0.217

No significant relationships were found between prey availability and foraging frequency or behaviour (Table 4.5). This suggests that prey availability isn't driving robin habitat selection, and robin foraging behaviour is not indicative of prey availability. It should however be noted that while the p values were not found to be significant ($P < 0.05$), the R values were quite high (e.g. pounce foraging and ground abundance, and parental feeding and arboreal diversity), which may be a result of the relatively small sample size (n = 11). A larger sample size may have yielded a significant relationship between foraging behaviour and food availability.

Table 4.5: Pearson correlations between frequency of foraging technique used by robins and invertebrate abundance and diversity.

Foraging Technique	df	Ground Abundance		Diversity	
		R	P	R	P
Pounce	10	0.425	0.193	0.189	0.578
Glean	10	0.203	0.550	0.239	0.480
Hawk	10	0.053	0.877	0.046	0.892
Parent fed	10	0.077	0.822	0.221	0.515
Foraging Technique	df	Arboreal Abundance		Diversity	
		R	P	R	P
Pounce	10	0.325	0.330	0.297	0.375
Glean	10	0.083	0.807	0.397	0.227
Hawk	10	0.213	0.529	0.159	0.641
Parent fed	10	0.121	0.723	0.459	0.156

The amount of time birds spent calling and courting were both found to be positively linked with higher population densities of juvenile robins (Table 4.6). Courting behaviour was also found to be linked with adult robin population density, but not juveniles. This can be seen in Figure 4.4.

Table 4.6: Pearson correlations between frequency of territorial and breeding related behaviours and population density estimates for juvenile and adult robins. Significant differences are shown in bold and with an asterisk.

Foraging Technique	df	Adult		Juvenile	
		R	P	R	P
Calling*	10	0.585	0.059	**0.678**	**0.022**
Courting*	**10**	**0.678**	**0.022**	**0.661**	**0.027**
Territorial displays	10	0.348	0.295	0.374	0.257
Nest building	10	0.403	0.220	0.471	0.144
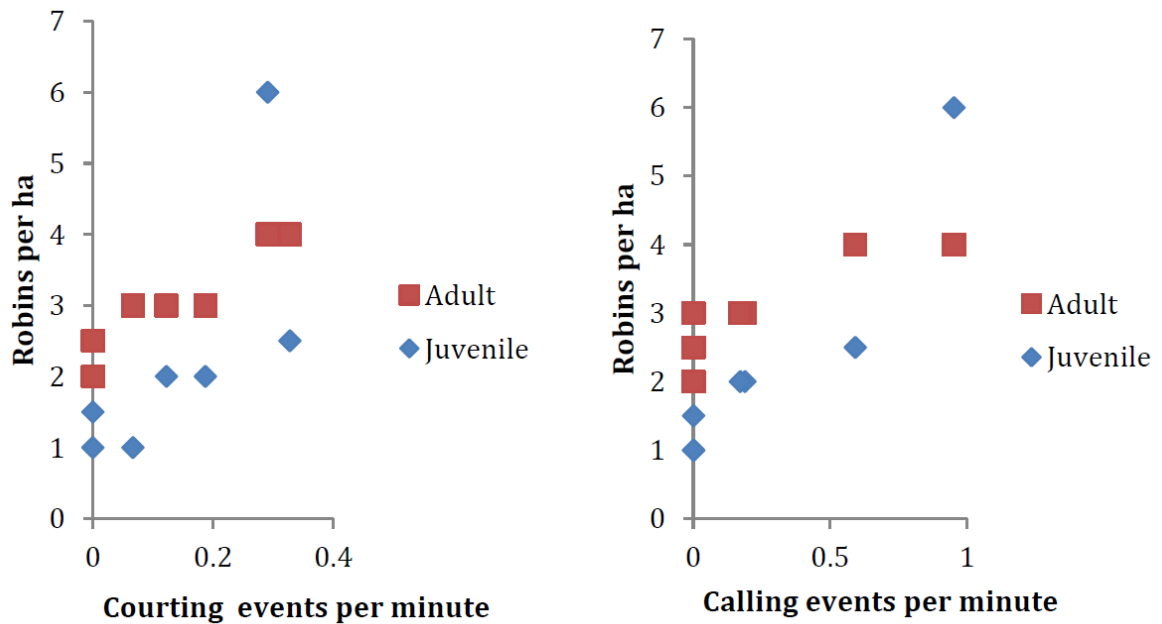
Figure 4.4: Adult and juvenile population density relative to observed courting and calling behaviour frequency.

4.6 Discussion

Vegetation height was found to be higher, and CWD was found to be more abundant in remnants than restored areas.. Numerous studies have shown that the presence of CWD can influence invertebrate (Braccia & Batzer, 2001; Longcore, 2003; Higgins *et al.*, 2014), bird (Greenberg & Lanham, 2001; Ford, 2011) and reptile (Mac Nally *et al.*, 2001; Kanowski *et al.*, 2006) assemblages. Given that CWD is slow to develop, and can easily be lost in fires, this can be somewhat problematic for restoration practitioners aiming to accelerate successional processes in an area. Empirical studies have also shown how bird species richness increases with vertical height of vegetation (e.g. MacArthur & MacArthur, 1961; Lindenmayer *et al.*, 2008; Kutt & Martin, 2010). This is thought to be a result of increased niche availability, as taller vegetation offers additional strata for foraging, nesting, and shelter (Barton *et al.*, 2014). Vegetation height was higher, and CWD density was greater in remnants compared with restored sites. This is unsurprising, as both are features that are likely to increase with age, but may be an indication that restored sites are less complex and offer lower quality habitat to wildlife. Chapter two found that overall robin density was no different between restored and remnant areas, but juvenile density was higher in remnants than restored areas.

No differences were detected between the overall feeding rates, or with calling, assisted feeding or preening/resting. Courting, territorial displays, and nest building activities were all found to be more frequent in remnants than restored areas, all three of which relate to reproduction. This suggests that while robins readily use both restored and remnant areas for some aspects of their functional habitat requirements, remnants appear to be preferred for reproductive activities. This difference in behaviour is consistent with the juvenile robin population density discussed in Chapter two.

Closer examination of the birds' foraging behaviour revealed that pounce foraging was the preferred foraging technique employed by robins in both habitats. This was an unsurprising result, as it is well documented that red-capped robins are a predominantly ground-pouncing insectivore (Recher, Davis, & Calver, 2002; Higgins & Peter, 2002; Razeng & Watson, 2015). In a study assessing the foraging behaviour of five species of ground-pouncing birds across a number of West Australian woodlands, red-capped robins were found to pounce forage more frequently than all other foraging maneuvers combined at all but one study location (Recher *et al.*, 2002). Only at Yellowdine, a salmon gum, gimlet and morrel (*Eucalyptus salmonophloia, E. salubris, E. longicornis*) woodland 400 km east of Perth, were robins found to use the hawk foraging technique as frequently as the pounce technique (Recher *et al.*, 2002). This demonstrates that while robins typically use the pounce forage technique, they are capable of changing their behaviour in certain habitats. Rottnest Island robins appear to conform to the more typical foraging technique seen across the majority of the mainland.

Glean and probe foraging frequency, which typically involves capturing prey from either the ground or from bark was found to be significantly more commonly used in restored areas than remnants. Hawk, hover, and snatch foraging, which involve capturing prey while on the wing, from either the air or foliage were also found to occur more frequently in restored areas. In Chapter three, restored areas were found to have higher abundances of invertebrates likely to appear in robins' diets on foliage and leaves than in remnants, which supports the arguement that arboreal prey availability is better in restored areas. It is well documented that feeding frequency is positively linked with prey availability and higher quality food resource areas (Repasky, 1996; Delestrade, 1999; Shepherd & Boates, 1999; Marchand *et al.*, 2002; Wellenreuther & Connell, 2002; Kilgo, 2005). While leaf litter was the most commonly used substrate to collect prey items in both restored and remnant areas, robins use of vegetation in restored sites was a close second, and was considerably higher

than in remnants. It appears that robins are altering their foraging style in restored areas to capitalize on the improved prey availability on foliage. This adaptation to the altered habitat demonstrates the birds' capacity to change their behaviour in response to altered conditions, reflecting the findings of Recher *et al.* (2002), who found that red-capped robins predominantly pounce foraged, but would change their behaviour under certain conditions. It appears that restored areas provide valuable foraging habitat for birds that are able to exploit prey found on foliage.

Comparisons between prey attack method and invertebrate abundance or diversity measures revealed no significant differences. In an experimental study on the effects of food availability on flocking behaviour and foraging efficiency of the alpine chough (*Pyrrhocorax graculus*), reduction in food availability resulted in a reduction in mean flock size, a reduction in the proportion of birds that had access to food, and a reduction in mean pecking rate (Delestrade, 1999). This study found that foraging rate appears to be relatively equal between the two site types, which may be an indication that prey availability, is similar or adequate between the two site types. Differences in habitat structure have been shown to have a greater influence on foraging habitat quality than invertebrate abundance due to structure drastically influencing detectability (Holmes & Schultz, 1988; Butler & Gillings, 2004). Given that the overall foraging rate appears to be similar between the two habitat types, there is little cause for concern regarding the quality of foraging habitat, despite the variation in foraging technique used.

In a study on hooded warbler (*Wilsonia citrine*) attack rates in Bottomland Hardwood forests, foraging frequency was found to be positively associated with arthropod abundance (Kilgo, 2005). The study also found that attack rates among adult birds foraging for fledgling birds did not vary with invertebrate abundance (which was linked to distance from timber harvest gaps). Shepherd and Boates (1999) found that semipalmated sandpipers' (*Calidris pusilla*) foraging efficiency dropped by 68.5% in areas that were disturbed by the introduction of baitworm (*Glycera dibranchiate*) harvesting. Core sampling for invertebrates in the sediment revealed that dug sediments contained reduced prey density; however, reduction in prey availability due to the obstruction of visual and tactile prey cues may also be a contributing factor (Shepherd & Boates, 1999). While it was not always possible to accurately record the success rate for robin attacks, because the birds would sometimes be too far away to clearly see the prey item, it was possible to compare the habitat variables the birds used. This study found no significant difference in availability of horizontal perches,

or overall visibility between remnants and restored woodland areas. This suggests that structurally, the restored areas appear to be relatively similar to the remnant areas, with the exceptions of vegetation height and the presence of CWD, which can have a positive influence on invertebrate abundance (Mac Nally *et al.*, 2001). While overall abundance and diversity of invertebrates between restored and remnant sites weren't significantly different, significant differences in the invertebrate assemblage were detected between restored and remnant areas for both ground dwelling and arboreal invertebrates in Chapter three. This demonstrates that some differences in invertebrate assembly do exist, and further enquiry into distribution taxons at a finer scale is recommended, especially in relation to the beetle order *Coleoptera*, which is an important taxon for insectivorous birds (Razeng & Watson, 2012), and was found to be much more common in leaf litter in remnant areas than leaf litter in restored areas (see Chapter three).

Behaviours relating to reproduction were found to differ significantly between restored and remnant areas. Courting displays, territorial aggression and nest building activities were all more commonly observed in remnant areas, suggesting that these habitats are superior in some way for the birds. Additionally, areas with more courting and calling behaviours were found to yield higher densities of juvenile robins. This may be an indication that remnant areas offer superior/preferred-breeding habitat, and contribute more to annual recruitment. This is a somewhat concerning result, as it may be an indication of a source-sink dynamic between remnants and restored areas. A source-sink dynamic occurs when a mobile species routinely moves from areas where recruitment is good (source) into areas where recruitment is poor (sink) resulting in the species' occupancy of the sink area being dependent on supplemented migration from the source area (Pulliam & Danielson, 1991). One of the worst forms of source-sink dynamics is the 'ecological trap', in which species prefer low quality habitat over higher quality habitat (Dwernychuck & Boag, 1972; Battin, 2004). By definition, an ecological trap is a habitat that is low in quality for reproduction and survival, is not capable for sustaining a population, and is preferred over other available, high-quality habitat (Donovan & Thomson, 2001). Given that the apparent lower rate of recruitment of robins in restored areas was found to coincide with lower rates of breeding related behaviours, it would appear that the birds are less likely to select restored sites as breeding habitat. This suggests that an ecological trap is unlikely to be in effect, and birds are simply utilising the available woodland habitats for different resource requirements. Given the small separation distance of patches of remnants and restored areas compared with the movement capacity of the birds, it is unsurprising that they are able to move

between the different habitat types in accordance with optimal resource availability.

A component of this study that was originally planned to be a major component, but was removed due to technical difficulties, was the observation of nest success rates and parental attentiveness. Camera traps were to be set up at nests to observe the incubation and parental care of birds until they fledged. For details on the rationale and associated procedures, see Appendix B. Unfortunately, fewer nests were located than originally anticipated, and of the nests located, only two were suitable for camera trap installation. As such, I was not confident that a representative sample of nests had been located, or that sufficient data had been gathered to draw meaningful results. Of the seven nests that were found over 16 days of active searching by one to four experienced bird watchers. Six of the seven nests were located within 10 m of roads. Five of the seven nests were located in restored areas. Six were in *Melaleuca* and one was in *Callitris.* Due to the placement of the nests, only two of the seven were suitable for camera installation. The other nests were checked every 3-5 days until the juvenile birds fledged or the nest failed. Four out of the five nests without cameras failed, presumably due to predation from ravens. The remaining two fledged two birds each. Of the nests with cameras, one was predated by a raven within 24 hours of the camera being deployed, and the other successfully fledged two birds. The nest with the camera that succeeded appeared to have been the second successful breeding attempt of the season, as the male was observed feeding juveniles while the female was sitting on the nest. The parents of the failed nest with the camera had a second breeding attempt, but the second nest was placed in a tree adjacent to a raven's *Corvus coronoides* nest, and also failed. It is likely that the nests I was able to locate were less discretely placed than many of the nests in the areas searched, and based on the ratio of fledglings to adult birds observed towards the end of the breeding season, it is clear that there were many successful breeding attempts, especially in remnants where only one nest was found.

Unfortunately, the habitat variables assessed in this study didn't yield a clear indication for the mechanism behind the difference in breeding habitat quality. Vegetation height may be a determining factor in nest site selection, but further study is required to verify this. Further enquiry into how robins select breeding habitat is required if we are to gain an understanding of why remnants offer superior breeding habitat. Within woodland patchiness, branching density and tree species composition may be habitat factors worthy of further investigation in relation to breeding habitat quality. Food resources don't appear to

be a driving factor as no difference in foraging behaviour of birds feeding fledgling birds was detected. Locating and monitoring nests proved to be more challenging than originally anticipated, and the large raven population on the island makes the use of camera traps inadvisable as it may increase the probability of predation of eggs or fledgling birds (Stevenson, 2011; *personal obs*).

This study was able to detect differences in behaviours associated with feeding and breeding habitat selection. The mechanisms behind the differences in feeding behaviour can be linked to variation in invertebrate abundance on foliage in restored sites. Overall, it doesn't appear that food resources are a limiting factor for robins on Rottnest; however the study did demonstrate the birds' capacity to adapt to variation in prey availability, which may be relevant for breeding success as birds' capacity to feed their young can relate to breeding success. Breeding-associated behaviours occurred more frequently in remnants than restored areas, which is supported by evidence that fledgling robin density towards the end of the breeding season is higher in remnant areas than restored areas. The mechanisms behind breeding habitat selection remain unclear, but may relate to vegetation height, within site patchiness or foliage density. The benefits and drawbacks of various animal behaviour study techniques used in this study will be addressed in the next chapter.

Chapter 5: Challenges and merits of including behavioural measures into restoration monitoring programs: A case study on Rottnest Island

5.1 Introduction

The previous chapters of this thesis explored several ways animal surveys can be used to assess habitat quality. Some of the assessment strategies involved measures of resource availability, structural habitat conditions and site history, while others looked at differences in animal community composition and differences in animal behaviour between sites. As discussed in Chapter one, animals are often an under-utilised and under-appreciated component of biodiversity in restoration efforts (Ruiz-Jaen & Mitchell Aide, 2005; Craig *et al.*, 2015; McAlpine *et al.*, 2016). If this weakness in restoration management is to be addressed, there needs to be consideration of how best to integrate fauna assessments into pre-existing management and monitoring protocols. Unfortunately, animal behaviour surveys can be highly labour intensive, and can yield ambiguous results. Furthermore, surrogate species models are still a somewhat contentious topic, meaning any inference drawn from the behaviour of one species may not have broad applications for the system as a whole, or even other species within that system. Ultimately, the assessments of restoration success need to be broadly applicable, reliable, and provide meaningful results.

Attempts to over-simplify ecosystems, and the use of broad-brush approaches like the 'field of dreams' or 'umbrella species' concepts, have been widely criticised by the scientific community (Palmer *et al.*, 1997; Caro & O'Doherty, 1999; Lindenmayer *et al.*, 2002; Hilderbrand *et al.*, 2005). As such, the way forward appears to be using integrated models that have multiple goals along separate trajectories, and an adaptive management framework (Choi, 2007; Lindell, 2008; Fraser *et al.*, 2017). This may present a challenging task for restoration managers who are often required to operate within ecologically, economically and socially accepted frameworks (Choi, 2007). Restoration managers generally require, or prefer, a straight-forward administrative plan so that the logistics can be suitably managed. A useful tool that may provide a framework for this type of management is the state-and-transition model (STM) (Stringham *et al.*, 2003). STMs are a management tool that can be used to synthesize and communicate information about alternative states and transitional processes inherent in specific systems, and have seen

widespread use in restoration management (Stringham *et al.,* 2003; Bestelmeyer, Goolsby, & Archer, 2011). Hobbs *et al*. (2009) posited that restoration as a field of study is in its infancy and that it has yet to achieve internal consistency, generality, and proven applicability of concepts in the field. They highlighted the practical limitations in identifying alternative states, transitions, thresholds and filters, as well as inconsistencies in the terminology used to describe these, and associated processes, within the literature (Hobbs *et al.*, 2009). Despite this, until a more unified conceptual base to ecosystem restoration has been developed and proven to be robust for applied ecological restoration, the STM approach appears to be a useful management strategy for restoring degraded areas, such as the woodland habitats of Rottnest Island.

This chapter explores the benefits and limitations of gathering and using various kinds of fauna-centric data to evaluate habitat quality. While restoration ecology can be studied as both a theoretical and an applied science (Hobbs & Norton, 1996; Choi, 2007), for people working to improve habitats through ecosystem intervention management and monitoring, its value comes from the applied aspect of the field. Hence, the data that restoration managers collect needs to be useful for predicting or explaining ecological processes acting in the system, while also complying with socioeconomic responsibilities. The value of data gathered will therefore be evaluated based on the ease with which it can be collected, the insight it can provide, and its applicability within a STM or other comparable frameworks. While the data gathered throughout this thesis was collected from Rottnest Island, and used primarily to assess habitat quality on the island for a particular woodland bird species, the overarching methodology may provide restoration practitioners in other regions with useful information on the merits and drawbacks of using various fauna assessment tools to measure habitat quality. In addition, as the Island has an ongoing woodland restoration program, which will soon be re-evaluated by the RIA, this work will provide valuable insight into faunal distribution patterns, habitat assessment procedures and future priorities for restoration work on the Island.

5.2 Case study - Rottnest Island woodland restoration program

As described in Chapter two, Rottnest Island (32°1 0 S, 115°500 E) is situated approximately 20 km west of Fremantle in the south-west of Western Australia. The island is classified as an 'A-Class Reserve', declared under the *Land Act 1993* and gazetted for public recreation since 1917 (RIA, 2014). Rottnest covers an area of around 1900 ha, and has been separated from the mainland for approximately 6000 years (Playford, 1983). The Rottnest Island pine

(*Callitris preissii*) and Rottnest Island tea tree (*Melaleuca lanceolata*) woodland is listed as a 'vulnerable' threatened ecological community under state legislation (RIA, 2014). The island is currently free from mammalian predators, unlike the mainland where feral cat *Felis catus* and fox *Vulpes vulpes* populations have considerable impacts on small vertebrate fauna (Risbey *et al.*, 2000; Doherty *et al.*, 2015). The only mammalian fauna currently found on the island are the quokka *Setonix brachyurus,* a herbivorous marsupial, the house mouse *Mus musculus*, an introduced species found in high numbers in the settlement area, and the white-striped mastiff bat *Tadarida australis*, a native species found across much of Australia (Stevenson, 2011). The bird community on the island is much smaller than that of the adjacent mainland, with just 60 species making up the island's avifauna, seven of which are isolated on the island with no populations on the immediately adjacent mainland and three species that are sufficiently different to constitute conservation management units (Saunders & de Rebeira, 2009).

The island is one of 545 islands that are larger than 20 ha off the coast of Western Australia, but has a number of environmental and social characteristics that make it unique (Saunders & de Rebeira, 2009). These include the deep, saline, inland waters that make up approximately 10% of the island's area, and provide important habitat for a number of wading birds that migrate from the Northern Hemisphere to the island during the austral summer (Saunders & de Rebeira, 1985). The terrestrial vegetation has been severely altered over the past 150 years since European settlement (Pen & Green, 1983; Rippey *et al.*, 2003). The once dominant *Melaleuca lanceolata* and *Callitris preissii* woodland have been severely damaged by fires and anthropogenic activities, which have transformed the majority of the Island's terrestrial vegetation into sclerophyllous grassy heath, dominated by the prickly low shrub *Acanthocarpus preissii* and the grass *Austrostipa flavescens* (Rippey *et al.,* 2003). Between the 1920s and the 1980s, the island's woodland areas were reduced from around 66% to 8%, and is now estimated to be around 4% (Pen & Green, 1983; RIA, 2014). Prolific quokka grazing and high tree density in mature woodland stands have restricted the capacity of the woodland tree species to naturally regenerate (Storr, 1963; Rippey & Hobbs, 2003; Winn, 2008). A state and transition model (STM) for Rottnest Island, developed by Rippey and Hobbs (2003), can be seen in Figure 5.1. Rippey and Hobbs' (2003) model describes the processes through which *Melaleuca lanceolata* and *Callitris preissii* woodland transitions into (and from) *Acacia rostellifera* woodland, or *Acanthocarpus preissii – Austrostipa flavescens* heath. Heavy grazing of seedlings by quokkas prevents regeneration, meaning grassy heath dominates. Vegetation is especially prone to this transition after a

fire, as regenerating seedlings can be heavily consumed when quokkas are abundant.

Frequent fires can further accelerate this, as the seed bank may not have had time to replenish between fires, meaning no new seedlings are available. Reduction in frequency of fires and intensity of grazing are required for woodland regeneration.
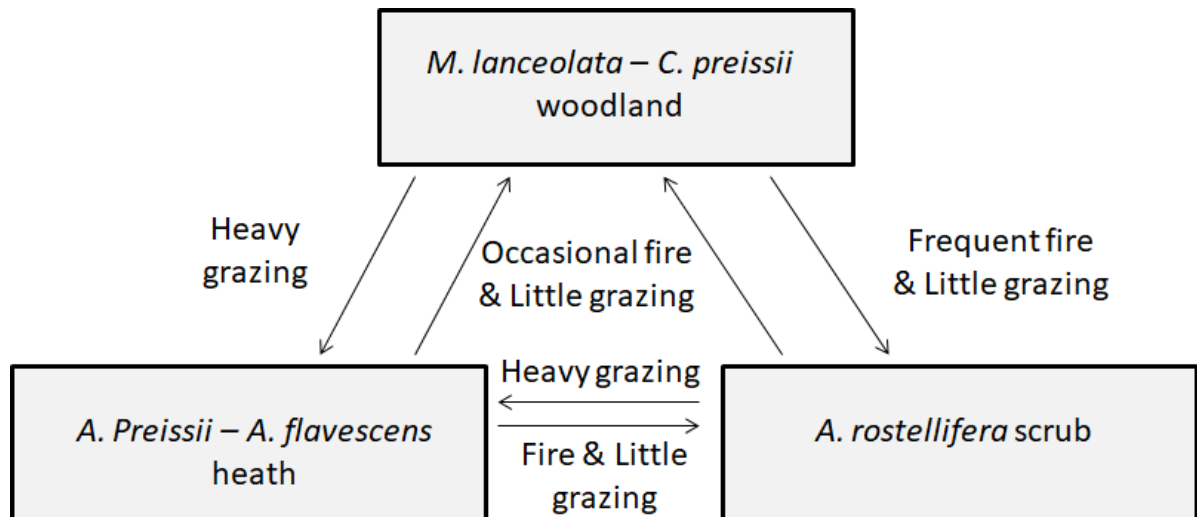
Figure 5.1: State and transition model of Rottnest Island vegetation responses to fire and grazing by quokkas. Model developed by Rippey and Hobbs (2003).

In an effort to reverse the loss of woodland habitat on the Island, the RIA began a woodland restoration program in 1963, resulting in widespread woodland restoration patches across the Island (Winn, 2008). The goals and targets of the woodland restoration program have gone through several revisions over the years, and as such, a range of different restoration strategies have been employed over the past 54 years, with varying degrees of success (Winn, 2008). Since 1963 there have been several improvements made to the vegetation management plans relating to woodland restoration. The initial woodland restoration strategy had no stated target, and merely aimed to restore lost vegetation and 'beautify' the island, often with the use of non-native plantings. Since then, specific restoration targets have been introduced, the first being brought in after 11 years of woodland restoration, which was to restore 20% of the island with woodland habitat (Rottnest Island Management Planning Group, 1962). This target was later changed in 1998 to the new target of restoring all suitable areas of the eastern two-thirds of the island to a woodland state by 2018 (Rottnest Island Management Planning Group, 2004). In addition to changing the target, the species composition being planted was changed in 1998, 35 years after the program began, from the mixture of endemic and non-endemic tree species to the current combination of the two endemic tree species, *Melaleuca lanceolata* and *Callitris preissii.*

There are a number of substantial differences in the practices used in the woodland restoration program over the years, partly due to changes in perspectives held by individuals driving the woodland restoration program, and partly due to re-evaluation of management techniques after reports of failed attempts to reach restoration targets (Winn, 2008). Species composition in some years included as many as seventeen species, fourteen of which were non-endemic (Rottnest Island Management Planning Group, 1962). This was during a time when restoration efforts were "to beautify the Island by planting trees" (Sten, 1959). As such, many fast-growing non-endemic species were planted in straight lines, and no records exist of the propagation method used prior to 1982. In addition, selection criteria used to select new woodland restoration sites weren't recorded until 1975, and it wasn't until 1986 that site selection moved from being based on distance above sea level to selection being based on historical evidence, an aerial photograph from 1941, showing sites that were once woodland (Rottnest Island Management Planning Group, 1985).

Site preparation involved controlled burning from 1963 until 1985, but was later replaced with a mechanical slasher that served the same purpose of reducing vegetation competition (Winn, 2008). Fences have been used since 1963 to exclude quokkas that would otherwise graze on seedlings, and in some sites tree guards have also been used. Finally, planting design has changed from the initial symmetrical lines spaced 5-6 m between seedlings to random clusters that were at first 3.5 m between seedlings, but that spacing has since been further reduced to 1-1.5 m between seedlings (Winn, 2008).

The first 30 years of restoration management and practices have been described as unsuccessful in restoring *Melaleuca lanceolata* and *Callitris preissii* woodland with a similar composition and structure to that of naturally regenerated *Melaleuca lanceolata* and *Callitris preissii* woodland (White & Edmiston, 1974; Rottnest Island Management Planning Group 1985; Winn, 2008). The most recent two decades have seen considerable improvement in restoration outcomes, which can largely be attributed to improved documentation of woodland restoration activities, better evaluations of success of woodland restoration management and practice allowing for adaptive management to occur, and better integration between science, management and practice in developing sensible ecological and economic targets and goals for management (Winn, 2008). The progression of Rottnest's woodland restoration programs over the years can be seen in Table 5.1. From 1963 to 2017 when this thesis was written, several changes have been made as a result of adaptive management, and in accordance with changes in perspectives

in relation to responsible restoration practices.

Table 5.1: Summary of woodland restoration management and practices since the program began in 1963.

Woodland restoration program 1963-2017

Time period	1963-1974	1975-1994	1995-2005	2005-2017
Area planted	100 ha, 6.8% Island area.	78 ha, 5.3% Island area.	59.3 ha, 4.1% Island area.	65.3 ha, 4.5% Island area.
Stated Target	None.	Restore 291.8 ha, 20% Island area. No time limit.	Restore 291.8 ha, 20% Island area. No time limit.	Restore a relative abundance of woodland throughout eastern two-thirds of Island, 972.7 ha by 2018.
Site selection criteria	Not stated.	Based on island topography until 1985, then changed to historical records of areas previously containing woodland habitat.	Based on historical records of areas previously containing woodland habitat (aerial photograph from 1941).	Based on historical records of areas previously containing woodland habitat (aerial photograph from 1941).
Site Preparation strategy	Controlled burn. Fence erection.	Controlled burn until 1986, then mechanical slashing. Fence erection.	Mechanical slashing. Fence erection.	Mechanical slashing. Fence erection and use of tree guards.
Planting design	Symmetrical rows.	Random cluster.	Random cluster.	Random cluster.
Plant spacing	5-6 m between seedlings.	Initially 3.5 m, decreasing through the years to 1-1.5 m between seedlings.	1-1.5 m between seedlings.	1-1.5 m between seedlings.
Direct assessments of the impact restoration has on terrestrial fauna	None.	None.	Birdlife Australia began conducting bush bird surveys in 2000, but in the years before 2009 there were issues with record keeping, site selection, and collection bias (Mather, 2010).	Birdlife Australia bush bird counts. Polson-Brown's (2012) study on robins and whistlers. And this study on birds and invertebrates.

The changes in targets, monitoring programs and management procedures are unsurprising given the time scale of the program, and the infancy of restoration ecology as a scientific discipline. Restoration ecology as a scientific discipline has changed dramatically over the past half century (Jordan & Lubick, 2011). Initially, there was heavy reliance on the premise that the balance of nature would naturally return, and restoration efforts could accelerate that process (Palmer *et al*., 1997); a concept that has since been largely discredited (Simenstad & Thom, 1996; Zedler & Callaway, 1999; Campbell, 2002; Wilkins *et al.*, 2003). By the beginning of the 21st century, there was a growing awareness that disturbed systems may never return to their pre-disturbance state, and that historical records may now be of only limited value for restoration practitioners (Higgs *et al.*, 2014). Historical records could serve as a desirable reference habitat, and help explain how processes interact, but ecosystem novelty may persist indefinitely (Hobbs *et al.*, 2006). In 2008, Winn developed a STM for woodland restoration on Rottnest Island, with a focus on supporting restoration of the threatened *Melaleuca lanceolata* and *Callitris preissii* woodland (Figure 5.2) (Winn, 2008). The model was developed to improve restoration outcomes in a damaged system that is highly novel (as discussed previously). Winn's model built on the work of Elizabeth Rippey and Richard Hobbs, who identified three stable vegetation states and the transitional forces that move areas between those three stable states (Rippey & Hobbs, 2003). The model also used the restoration history of the Island (Figure 5.1) to identify and remove weakness in the restoration program. While Winn (2008) acknowledged that the woodland community should provide three specific woodland services (woodland conservation, wildlife protection, and recreation amenities), the wildlife perspective was overlooked in her model. This was due to limited availability of information at the time. As a result, some questionable assumptions were made in the model, specifically, the capacity of each vegetation state to provide wildlife protection services.
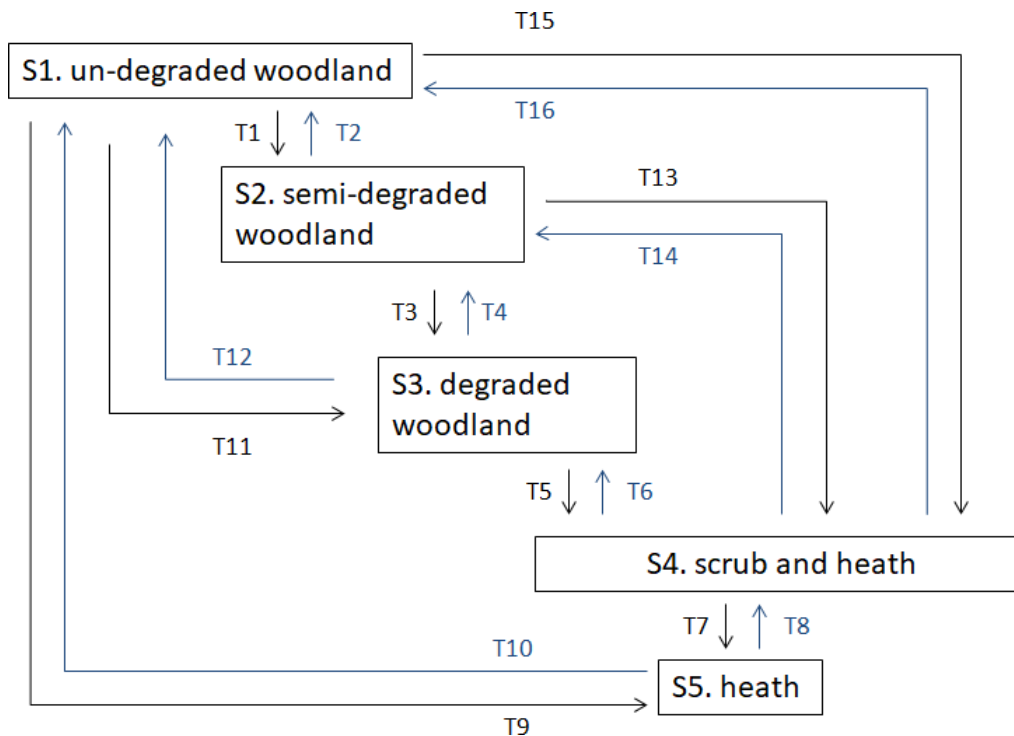
Figure 5.2 Winn' s (2008) state and transition model for woodland restoration on Rottnest Island. For descriptions of states and transitions, see Table 5.2 and 5.3 respectively.

Table 5.2 Description of Winn' s (2008) state and transitional stable woodland states.

Vegetation states:

S1   Un-degraded woodland

Contains only *Melaleuca lanceolata* and *Callitris preissii* species, forming a structure similar to that of the reference habitat (naturally regenerated woodland stands that remain on the Island). Little to no understory. The woodland is fully providing the intended service(s).

S2   Semi-degraded woodland

Woodland with decreased abundance of *Melaleuca lanceolata* and *Callitris preissii* trees. *Acacia rostellifera* and/or *Acanthocarpus preissii – Austrostipa flavescens* heath communities occupy some of the gaps in the woodland canopy. Some evidence of *Melaleuca lanceolata* and *Callitris preissii* regeneration. Woodland stands are not fully providing the intended woodland service(s).

S3   Degraded woodland

Senescent *Melaleuca lanceolata* and *Callitris preissii* woodland with no evidence of woodland regeneration. The area is largely occupied by *Acacia rostellifera* and/or *Acanthocarpus preissii – Austrostipa flavescens* heath communities. Other tree species may be present. Woodland does not provide any woodland services.

S4   Scrub and heath

Mixture of *Acacia rostellifera* closed scrub and *Acanthocarpus preissii – Austrostipa flavescens* heath communities.

S5   Heath

Acanthocarpus preissii – Austrostipa flavescens heath community.

Table 5.3 Description of the transitions between Winn' s (2008) state and transitional stable woodland states.

Transitions between states:

T1 Decreased abundance of *Melaleuca lanceolata* and *Callitris preissii* associated with: (1) natural senescence and limited site maintenance e.g. supplemented plantings; and/or (2) historical restoration practices of planting introduced tree species. Light and short-term quokka grazing limiting the amount of woodland regeneration.

T2 & T12 Removal of quokka grazing by erecting fences. Improving quantity and quality of *Melaleuca lanceolata* and *Callitris preissii* woodland by: (1) planting seedlings using the appropriate planting design required for the provision of the intended woodland service(s); or (2) promoting natural regeneration through the use of fire (low intensity burn) or selectively thinning the canopy. Mid-recovery intervention may be required to manipulate the woodland structure so it complies with that of the reference system for the intended woodland service(s). Ongoing site maintenance is essential. Introduced tree species should be removed if present.

T3 & T11 *Melaleuca lanceolata* and *Callitris preissii* are reaching the end of their life span (~110 years). No site maintenance. Heavy and long-term quokka grazing preventing woodland regeneration.

T4, T6 & T14 Removal of quokka grazing by erecting fences. Improving the quality and quantity of *Melaleuca lanceolata* and *Callitris preissii* woodland by: (1) planting seedlings using the appropriate planting design required for the provision of the intended woodland service(s); or (2) promoting natural regeneration through the use of fire (low intensity burn) or selectively thinning the canopy. Introduced species are not removed. No-mid-recovery intervention or site maintenance undertaken.

T5 *Melaleuca lanceolata* and *Callitris preissii* have reached the end of their lifespan (~110 years). No site maintenance has been undertaken. Heavy and long-term quokka grazing preventing woodland regeneration.

T7 Acacia rostellifera has reached the end of its life span and regeneration is prevented due to heavy and long-term quokka grazing.

T8 Disturbance, such as fire, to stimulate germination of Acacia rostellifera seed stored in the soil. Disturbance must occur within 60 years since *Acacia rostellifera* occupied the site due to the seed survival rate of *Acacia rostellifera* in the soil.

T9, T13 & T15 The combination of wildfire, no fences erected immediately following the fire to prevent quokka grazing on woodland regeneration, and presence of heavy and long-term grazing. No woodland restoration practice implemented.

T10 & T16 Removal of quokka grazing by erecting fences. Slashing existing vegetation and planting *Melaleuca lanceolata* and *Callitris preissii* seedlings using the appropriate planting design required for the provision of the intended woodland service(s). Alternatively direct seeding and brushing techniques regeneration methods can be trialled. Site maintenance and mid-recovery interventions is undertaken when required.

There is a notable lack of direct assessments of the impact woodland restoration is having on the terrestrial fauna on Rottnest Island (Winn, 2008) outside of the Birdlife Australia bush bird surveys that have been running since 2000 (Mather, 2010), and an honours thesis on the value of restored and remnant habitat for robins and whistlers on the Island (Polson-Brown, 2012). This was a reason for the omission of wildlife considerations from Winn's (2008) state and transition model. The RIA has stated that it intended to develop and launch a woodland management plan by 2017 to "enhance the long-term resilience of the woodland community, while continuing to provide important fauna habitat" (RIA, 2014). The plan is now likely to be completed in 2018 (*pers. comms*.). The implication being that restoration is, and will continue to provide, important habitat for woodland fauna.

While there have been limited fauna studies that directly assess the impact the restoration program is having on the island's fauna, a number of fauna studies have been conducted on the island that relate to interactions between different faunal species (Stevenson, 2011), limiting habitat resources (Poole *et al.,* 2015), and seasonal fluctuations in resource availability (Phillips, Chambers, & Bencini, 2017). These studies are useful in providing insight into how fauna on the island have adapted to their habitat(s), as well as identifying potential threats to the resilience of those species and their continued existence on the Island. But for the woodland restoration program to accurately gauge the impact restoration efforts are having on the Island's fauna, formal assessments of faunal responses to restoration are required. How best to measure faunal responses to restoration is a topic that I will now discuss, with reference to my previous three chapters, each of which explored one or more fauna assessment strategies.

5.3 Animal biodiversity measures

Indices of animal biodiversity (richness, diversity and abundance) can provide valuable insight for restoration monitoring, as they can be used to track recolonisation, and help identify species affected by habitat filters (Craig *et al.*, 2012). However, they can be somewhat misleading, as disturbed sites often have higher species richness measures due to increased numbers of disturbance specialists, while having lost specialist species that respond negatively to the disturbance (Devictor & Robert, 2009). Fortunately, there is evidence that this issue can be mitigated by using compositional assessments that separate species into specialist and generalist, or other functional groups (Devictor & Robert, 2009). Chapter two of this thesis, looked at the woodland bird community using both compositional and biodiversity measures. In Chapter three I studied the invertebrate communities found in heathland, remnant woodland, and old and young restored woodland areas using both compositional and biodiversity analyses.

A major limitation of biodiversity estimates is that an animal's presence in an area does not necessarily mean the site contributes positively to individual reproductive success and population replacement (Aldridge & Boyce, 2007; Lindell, 2008). This can be seen in ecological traps (Robertson & Hutto, 2006), which have been found to be common in human modified landscapes (Battin, 2004). An ecological trap occurs when animals select inferior habitat over superior habitat leading to a drop in fitness (Lindell, 2008).

Biodiversity measures alone also offer no explanation for why a species may be scarcer or absent in certain areas. The absence of a species from an area may be an indication of a habitat filter (Craig *et al.*, 2012), but the mechanism behind the species' absence would require a more targeted assessment. This makes biodiversity measures a useful tool for identifying when a problem exists, provided they are able to pick up variation in both generalist and specialist species, but ineffective for identifying negative habitat features such as filters. Conversely, behavioural studies can be used to compare habitat quality between sites, identify reasons for differences in habitat quality, identify critical resources that determine habitat quality, and explain the mechanisms through which species contribute to ecosystem functions (Lindell, 2008). Behavioural studies do however, require a more comprehensive understanding of the species being studied, as experimental design often must be tailored made for the species in question. These types of studies are also generally more labour intensive than a biodiversity assessment, and, as this study found, can often produce somewhat ambiguous results.

Chapter two focussed on the biodiversity of woodland birds across a number of isolated fragments of woodland habitat that had a range of different disturbance histories. The data were compared with presence-absence data for a single Australasian robin species that is resident in some, but not all, of the study sites. No differences were detected between restored and remnant habitats with respect to bird diversity, richness or composition. This contradicts an element of Winn's (2008) STM, which assumed that semi-degraded and degraded woodland sites would not provide the woodland service 'wildlife protection' as well as the un-degraded woodland areas. Heath and scrub habitats were not assessed in this study, but robins and a number of other woodland dependent species are known to be absent from those habitats (Saunders & de Rebeira, 2009). Further investigation, described below, was necessary to measure the relative quality of woodland areas across different stable states.

Collecting data on the entire community was no more difficult and required no more equipment than surveying the robin population alone, and required minimal equipment and little specialised training. A pair of binoculars and a familiarity with the birds found on the island was all that was required. The biodiversity data were useful in identifying which sites had the highest and lowest diversity and abundance of different species. When comparing biodiversity measures with habitat variables using various multivariate analyses, links between variations in habitat conditions and avian assemblage were found. What couldn't

be specifically demonstrated from this study alone, was how factors like patch size, time since last fire, habitat type (restored/remnant), and perimeter were linked with robin population density, or the assembly of the avian assemblage as a whole. Additionally, factors like perimeter and patch size were both correlated with bird abundance, but also strongly correlated with one another. It is unclear from this data alone whether these robins prefer larger areas because they offer more edges or because they have larger centres away from the edges, or a combination of both. Fortunately, numerous studies have been conducted on these topics, and so we have an understanding of how faunal assemblages are influenced by factors like patch size, and shape, time since last fire, and disturbance history (e.g. Bender, Contreras, & Fahrig, 1998; Davis, 2004; Anderson *et al.*, 2005). This allows researchers to infer causal links between detected variation in biodiversity measures and habitat characteristics. This is an important integration be between life history type studies and faunal response to restoration efforts.

Meta-analyses of factors that involve large-scale phenomena, like the effect of patch size or fire on avian assembly, provide valuable insight on general relationships (Gurevitch *et al.*, 1992; Bender, Contreras, & Fahrig, 1998). These types of studies are useful resources for predicting how species will respond to relevant habitat factors, as they are able to evaluate the results of multiple data sets that would be difficult to collect within a single study, due to time, money and effort constraints (Bender, Contreras, & Fahrig, 1998). A limitation of meta-analyses in areas like Rottnest Island, compared with more targeted single species studies, is that general processes may act differently in novel systems with unusual species compositions, interactions and functions (Hobbs *et al.*, 2009). For an area like Rottnest Island, that has a number of historical and geological characteristics that differ from those found on the neighbouring mainland, it could be argued that the system is highly novel. The absence of mammalian predators since the last of the feral cat population was eradicated in 2002, after 40 years of attempts to remove the species due to concerns regarding their impact on native fauna (Algar, Angus, & Onus, 2011), is one example of the Island's novelty compared with mainland reference sites. The widespread conversion of the once dominant *Melaleuca lanceolata* and *Callitris preissii* woodland habitat type to the now dominant heathland habitat type with scattered patches of restored woodland containing numerous non-endemic plant species is an example of the Island's novelty arising from its history (Winn, 2008).

Saunders and de Rebeira (1985) reported three avian extinctions on Rottnest Island, and

seven migrations that were directly attributed to human influence, mostly in relation to woodland habitat loss. Storr (1985) and Brooker *et al.* (1995) reported the suspected extinction of two species of skink, and declines in several others, again due to human activities. By contrast, the quokka population appears to have surpassed the carrying capacity for the Island, as a result of supplemented food and water resources in the settlement, and protection from hunting for almost 100 years (RIA, 2004; Winn, 2008).

Meta-analyses may be useful in explaining large-scale phenomena, especially those relating to biodiversity relationships in fragmented landscapes. However, it is important to remember that systems like this one are novel, and processes may operate in unconventional ways. As such, more detailed studies of specific interactions may yield meaningful results regarding how species are using the available habitat on Rottnest, and how species interact with one another. Overall, biodiversity measures are relatively simple to collect and can be used to sample a wide range of taxonomic groups. When used in conjunction with existing literature on large-scale phenomena, and detailed studies on interactions that are potentially novel, they can form a strong assessment tool for restoration practitioners.

5.4 Surrogate species models

Surrogate species models are generally based around the assumption that by managing or monitoring the needs of a chosen sub-set of species, the needs of a larger pool of other species will also be met (Caro, 2010). While there has been controversy in the past over the legitimacy of surrogate species models being able to account for the needs of the entire species pool, the general consensus appears to be that models that include numerous species (focal species approach [FSA] and generic focal species [GFS]) are preferable to single species models (umbrella & keystone) (Caro & O'Doherty, 1999; Lindenmayer *et al.*, 2002; Wiens *et al.*, 2008; Caro, 2010; Watts *et al.*, 2010). This is because it is unlikely that a single species' habitat requirements will reflect the needs of all other species found in that area. For cases where suitable focal species can be identified, behavioural assessments may be relevant to help identify key resources and interactions affecting those species.

This study assessed the suitability of red-capped robins to act as an indicator of avian assemblage in Chapter two. Total avian assemblage, as well as woodland dependant avian assemblage, and woodland assemblages split by dietary guilds were all assessed separately, and my findings were that robin population density was a poor indicator for all three

measures of avian assemblage (richness, diversity and composition), (see Chapter two). This may be due to a number of factors, such as the high abundance of generalist species, like the silvereye *Zosterops lateralis* and the white-browed scrubwren *Sericornis frontalis,* in the dataset who are unlikely to be dependent on any characteristic differences between restored and remnant patches. Alternatively, it may be due to the absence of other specialist woodland birds, that are either naturally absent from the Island, or have already become extinct, like the rufous whistler *Pachycephala rufivenmtris* and the brush bronzewing *Phaps elegans*, both extinctions having been directly attributed to the loss of woodland habitat (Saunders & de Rebeira, 2009). This result differs from a number of mainland comparable studies that found red-capped robins to be a suitable indicator of the degree and extent of pervasive mining disturbances (Read, Parkhurst, & Delean, 2015), and indicators of the impacts of logging activities on avian communities (Kavanag *et al.*, 2004). While the red-capped robin on Rottnest is a species of conservation significance, due to differences with the mainland population (Baker *et al.*, 2003; Saunders & de Rebeira, 2009; Mather, 2010), management focussed solely on creating ideal habitat for robins would be inappropriate given these findings. Assessments should instead consider the wider community of woodland birds, including the western gerygone *Gerygone fusca*, the singing honeyeater *Lichenostomus virescens* and the golden whistler *Pachycephala occidentalis*, all of which have also been listed as being of conservation significance due to their differences from mainland populations (Saunders & de Rebeira, 2009).

5.5 Microhabitat use and resources

For animals that are easy to detect, like most birds, biodiversity measures can be collected quickly and cheaply with minimal equipment. Other taxonomic groups aren't as simple, and may require targeted trapping efforts or the use of equipment such as motion sensitive cameras or other recording devices to detect their presence in an area. Unfortunately, these systems can be expensive and specialised, meaning the value of the data acquired may not always justify the investment cost of purchasing said equipment. In addition, an animal's presence in an area isn't necessarily evidence that the habitat is of sufficient quality for the species to sustain itself without supplemented migration from neighbouring areas (Battin, 2004; Robertson & Hutto, 2006; Lindell, 2008). Resource availability assessments on the other hand, may provide a more meaningful way to measure habitat quality, especially for species that are otherwise difficult to observe. Food resources are obviously an important habitat component, and as the focus of this thesis was on an insectivorous bird, the invertebrate community was selected as a resource worthy of investigation. In chapter three

the invertebrate community was assessed in restored, remnant and heathland type sites at the two strata where the birds frequently forage. This study found that restored woodland areas contain invertebrate communities more similar to remnants than heathland sites, and that restored areas get progressively more similar to remnants as they age. The study assessed the rate with which the invertebrate community responds to restoration efforts, and demonstrated links between factors like patch size and time since last fire, and variation in the invertebrate community at the two measured strata levels.

Through the identification of important habitat resources, and assessment of the availability of those resources, it is possible to systematically gauge the quality of a study site relative to a reference site. Identifying important habitat resources and microhabitat conditions can be difficult without an understanding of how fauna use the available habitat. Numerous studies have demonstrated the value of integrating fauna behaviour into restoration assessments as a way of identifying filters (e.g. Craig *et al.*, 2015), and separating high and low-quality habitat (e.g. Lindell, 2008). This study focused on behaviours relevant to feeding and breeding, and identified a number of differences in the ways robins use restored and remnant areas.

In Chapter four, through evaluation of how animals use available resources in the two habitat types (restored and remnant), I was able to identify differences in the way robins use those habitat types, and infer information about their relative quality in relation to feeding and breeding resource availability. Robins were found to forage equally frequently in both habitat types, but employed different foraging techniques. While pounce foraging onto the ground was the most commonly observed foraging behaviour in both habitat types, robins in restored areas were far more likely to glean or probe invertebrates off vegetation than in remnants. This difference in behaviour was reflected in the differences in the invertebrate assemblage measured in Chapter three, with birds changing from an almost exclusive ground foraging technique used in remnants, to a heavy reliance on arboreal invertebrates in restored areas. This demonstrates the capacity of the birds to adapt to superior/inferior habitat conditions. Differences in observed breeding related behaviours (courting, territorial displays, and nest building) were also observed between restored and remnant areas, however the resource requirements relevant to these behaviours are less clear.
Further study would be required to identify why robins appear to prefer remnants over restored areas for breeding related activities.

Woodland birds as a group have been severely affected by habitat loss and degradation, often through fragmentation (Mac Nally *et al.*, 2009). Insectivores as a sub-set of woodland birds, appear to be among the worst affected which may indicate that declines are linked in some way to changes in invertebrate assembly (Mühlner *et al.*, 2010; Watson, 2011). Aside from being an important food resource within most ecosystems, invertebrates contribute overwhelmingly to the overall biodiversity (Anderson & Smith, 2004). Hence, it has been argued that biodiversity monitoring programs aimed at studying the integrity of an ecosystem should not be considered adequate without assessment of invertebrates (Taylor & Doran, 2001).

In Chapter three of this study, a number of differences in invertebrate assemblages between heathland and woodland areas were identified. Differences were also found between the three different types of woodland assessed (young restoration, old restoration, and remnants). These differences were at both the ground and arboreal level, which are likely to influence insectivores differently, as ground foraging birds like the red-capped robins are likely to utilise different insects to the canopy gleaning golden whistlers *Pachycephala occidentalis*. The arboreal and ground communities were found to respond to different habitat variables that related to isolation, fire history, patch size and ground cover. This highlighted the inter-relatedness of microhabitat and resource availability, and demonstrated the value in assessing resources and habitat components like the invertebrate community at multiple strata levels.

Heathland sites were found to be more different to any of the woodland sites, demonstrating that restoration efforts are successfully transitioning the invertebrate community to one that more closely resembles that of the remnant woodland habitat type. Restored sites of similar ages were more similar to each other than restored sites of different ages, which is either a result of common successional development or a result of the changes in restoration procedures over the past 50 years or a combination of the two. Time since last fire and patch size also influenced invertebrate assemblage, highlighting that those are two factors worthy of further study in the future. These results are meaningful in that they provide evidence that the differences in plant species and vegetation structure between the different habitat types are reflected in the invertebrate community. This is an indication of how restoration efforts have influenced the invertebrate community, which in turn, is likely to influence fauna species dependant on invertebrates. This is a positive result for restoration managers, and future woodland management plans should consider

continued monitoring of the terrestrial invertebrate community. The data gathering process was moderately labour intensive, as it required targeted trapping, collection, sorting and identification of all invertebrates, but most of the materials involved were cheaply sourced, and species identification down to order is relatively simple with field identification keys.

Microhabitat use by birds was assessed in Chapter four using behavioural analysis. This was far less straight forward than the invertebrate surveys in Chapter three, but offered insight into how the birds use and perceive restored and remnant woodland areas. Data collection methods were derived from the Donald *et al.* (2007) study on raso larks *Alauda razae* in Cape Verde. I focussed predominantly on behaviours that related to feeding and reproduction, and the microhabitat characteristics and resources that related to those requirements. These features were identified as 'ultimate' factors determining the success of breeding and thus determining fitness. Foraging rates were found to be higher in restored areas than remnants, which is likely an indication that greater food abundance is available in restored areas (Shepherd & Boates, 1999; Kilgo, 2005). This mirrors the results of Chapter three, which found that arboreal invertebrate abundance was higher in restored areas.

Breeding related behaviours on the other hand were rarely observed in restored areas, suggesting that the birds use the restored areas for feeding, but are still reliant on remnants for breeding related activities. Determining why the birds are predominantly breeding in remnants was beyond the scope of this study, but would be a valuable area to investigate in the future. These results may be indicative of a source-sink dynamic that may exist between remnants and restored areas, meaning that restored areas may not be supporting viable populations, and may be reliant on remnants to supplement annual recruitment.

The behavioural component of this study was challenging, as data collection was highly time consuming, required a rigorous collection methodology, and is subject to observational bias, meaning that comparisons between studies may be difficult. For details on behavioural data collection procedures, see Appendix C. Bird banding was also required so that observations were not biased due to individual behavioural patterns of a select few birds. This was a time consuming, and labour-intensive component of the study that required specialty qualifications, equipment, and approvals. In this case there were additional complications due to a moratorium having been placed on colour banding this species, which was eventually lifted through negotiations with the ABBBS and the use of custom made bands.

For details on the bird banding process and acquisition of specialty bands for this project see Appendix A.

When conducting behavioural assessment surveys, the selection criteria used to collect data have a fundamental effect on the detectability of patterns and overall outcomes. An example of this from this study can be seen in the overall foraging rate between restored and remnant areas not being found to significantly differ, while the frequency with which some foraging methods were used in the two habitat types was significantly different. These differences were found to relate to resource availability, but would not have been detected had prey attack method not been recorded. This demonstrates a challenge for researchers, as designing an experimental procedure to accurately capture significant behavioural differences requires a comprehensive *a priori* understanding of the animal's behaviour, which may not always be available. Additionally, this study was conducted on a commonly found, easily observed species with a relatively restricted home range. Studies of this kind may be highly challenging on a species that is harder to locate and observe for extended periods of time and over large areas.

A research component that was omitted from the study was the use of camera traps to monitor nest visitation and breeding success. The section was omitted due to low numbers of nests, a lack of confidence in how representative the located nests were for all nests used by the birds, and the placement of nests being ill-suited to camera placement requirements (for full details, see Appendix B). The financial cost of acquiring the cameras (which was over 50% of the total equipment expenses for the project), the effort that went into searching for nests, as well as the challenge of transporting the cameras around the island on a bicycle did not justify the results that the cameras generated. As such, camera traps would not be advised for future work on species like robins, especially in areas like Rottnest where large populations of ravens exist, as there is a risk that cameras may increase the detectability of nests for predators.

5.6 Proposed addition to pre-existing state and transition model

Winn's (2008) model describes a logical set of stable habitat states, with a set of transitional processes that can move an area from one state to another. The model does however omit the inclusion of faunal requirements, as Winn said it was beyond the scope of her study, and relatively little faunal information was available at the time that the model was developed.

Based on the findings of this study, I propose a four-tiered classification system be added to Winn's model that relates to the stages of woodland degradation (Figure 5.3). For an area to transition from one tier to another, evidence of either a key process or an animals' presence in the area is required. This will need further testing and refining, as the criterion listed here are largely retrospective. Additionally, the criteria discussed here will specifically relate to taxonomic groups assessed over the course of this study, and work on other woodland taxonomic groups may be incorporated into this model in the future. The feasibility and logistics of this type of assessment will need to be discussed with the RIA. Financial limitations in the past have resulted in many of the monitoring programs having been conducted by volunteer groups, which is likely to continue in the future. As such, it is likely that monitoring programs will need to be designed in such a way that data can be adequately gathered by semi-trained volunteers, and between-year surveys can be compared.

Tier	Definition
Tier 3	Woodland is classified as un-degraded. Avian community richness, diversity, abundance and composition are similar to that of the reference state. Invertebrate community richness, diversity, abundance and composition are similar to that of the reference state. Evidence of successful feeding and breeding (e.g. presence of juvenile birds in spring) of woodland dependant species of conservation significance.
Tier 2	Woodland is classified as semi-degraded. Avian community richness, diversity, abundance and composition are approaching a similar state that resembles the reference state. Invertebrate community richness, diversity, abundance and composition are approaching a similar state that resembles the reference state. Area provides some resource requirements, but species still appear to be reliant on neighbouring habitats for key resource requirements (e.g. breeding).
Tier 1	Woodland is classified as degraded. A number of woodland dependant avian species (e.g. robins, whistler and gerygone) are absent or rarely observed during surveys. Invertebrate community composition is significantly different to the reference state, may resemble the heathland state.
Tier 0	Area is scrub or heathland. Most if not all woodland species are absent.

Figure 5.3: Proposed addition of a four tier system to Winn's (2008) state and transition model to better account for faunal use of woodland areas at various stages of degradation.

5.7 Recommendations and future work

While a robust STM that includes assessment of faunal responses to restoration activities would be desirable, logistically there are a lot of hurdles before that will be achieved. The parameters set up by Winn (2008), which list degraded and semi-degraded woodlands as not providing 'wildlife protection' woodland services was not found to be true for robins, as they readily use degraded areas for feeding resources. The proposed tier system outlined in this study will require further refining based on logistic constraints.

The use of camera traps to monitor breeding success is not advised, based on logistical constraints associated with cost of the units, difficulty locating nests, inaccessibility of some nests to the placement of cameras and the potential increased nest predation rates due to camera placement. Observation of fledgling density towards the end of the breeding season is a far simpler way to measure breeding success. Foraging behaviour can be easily observed and can be done while performing bird surveys.

Assessments of the invertebrate community for this study were focussed predominantly on invertebrates as a food resource; however, there is a notable gap in the literature surrounding the terrestrial invertebrate assembly on the island. This would be a valuable area for future work. The invertebrate community was found to differ significantly between heathland, and the various woodland states. Further enquiry into whether differences in the invertebrate community can be detected between more of the stable states outlined in Winn's (2008) model was not assessed, but would be valuable information.

Special focus should be given to the distribution of Coleoptera on the island, as they are an important component of the woodland ecosystem, and appear to be absent from heathland sites, and while arboreal Coleoptera appear to readily recolonise restored areas, ground dwelling Coleoptera appear to be slower to return.

Given the apparent limitations of surrogate species models, a straight-forward STM that adequately incorporates all woodland dependant animal taxa through detailed study of a few key species appears to be unfeasible; especially given the limitations of presence/absence data being used to assess habitat quality, and the complexity of accounting for novelty within restored habitats. Behavioural studies on the other hand, have the capacity to provide valuable insight into the mechanisms behind specific interactions. Behavioural studies do however require a high level of understanding of the

animal's Umwelt (Van Dyck, 2012), and potentially labour intensive investigation of how the animal interacts with its habitat. As such, detailed behavioural studies of species of interest, derived from life history data, observational data, and, if possible, experimental manipulative studies, coupled with abundance, presence/absence, and resource-focussed studies appear to be the most comprehensive way forward. For Rottnest Island, future work is recommended to identify how robins are selecting breeding habitat, as this study found that robins typically prefer remnant areas over restored areas for reproductive activities.

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6. Appendix A: Bird Banding

6.1 Introduction

To ensure independence of observations, as well as track birds' survival and success in reproducing, individuals were marked so they would be identifiable *in-situ*. This was done using coloured bands, which were attached to the legs of birds. As a bird banding project was already active on the Island prior to the commencement of this project, many of the birds on the Island were already fitted with Australian Bird and Bat Banding Scheme (ABBBS) bands. ABBBS bands make birds individually identifiable in the hand, as each ABBBS band is printed with a unique prefix/number, but are generally not suitable for identification in the field, as the numbers are too small to be reliably read through binoculars. As such, a colour marking permit was added to the banding project, which would permit the addition of a second band which contained a unique colour combination.

Due to past issues with coloured plastic bands causing problems for very thin legged passerines, like the red-capped robin and the purple-crowned fairy wren, there was a moratorium on colour banding red-capped robins at the outset of this project. After discussion with the banding office and several different banding groups from around the world, the moratorium was lifted for this project, provided specific conditions were met. Those conditions were as follows; 1. Only one band per leg was to be fitted to the birds; 2. No plastic bands were to be used; and 3. The birds would be monitored for the next three years to assess whether the issues with plastic bands had been mitigated through the change in material. As such, I used split colour metal bands, each of which contained a unique combination of two out of nine available colours. One leg was fitted with an ABBBS band, while the other was fitted with the coloured band. No adverse effects of the bands were detected in any of the birds with the split coloured metal bands that have been regularly sited in the same areas consistently over the past three years.

6.2 Trapping technique

The trapping technique used in this study is derived from the trapping technique used by Major *et al.* (1999) in a study that looked at the differences in population density of red-capped robins in different shaped habitat remnants. The major change to the technique used is the reduction in spacing from 100 m intervals to 50 m, which is due to the reduced

total area that needed to be covered, and the observed high population density of robins on the Island. A grid, marked at 25 m intervals, was established in each of the woodland sites. Song play-back was conducted at alternate points on the grid such that a playback point was situated every 50 m. In narrow, linear sites, no grid was established: song playback was conducted from points at 50 m intervals along the mid-line of the strip (Major *et al.*, 1999). Up to four 15 second bursts of target species' song, interrupted by 15 second listening periods, was played at each point from a hand-held MP3 player. The song played was a recording of the robin call from Rottnest Island recorded by F. Holmes. Playback was terminated at each point immediately upon detecting one of the target species in the area. If a bird responded to the playback, mist-nets were erected near the playback point.

6.3 Data gathered while banding

While banding the birds, a series of biometric information (Table 6.1) was collected from all birds caught. This data provided demographic and physiological information that was submitted to the ABBBS.

Table 6.1. Biometric data collected from captured birds. All data collected using the techniques described in The Australian bird bander's manual (1989)

Variable	How measured
General information	
species	morphology
age	feather condition, plumage, soft parts
gender	plumage
Morphological	
wing length	wing ruler (1mm)
Head-bill length	callipers (0.01mm)
fat score	ordinal chart (1-8) based on (Kaiser, 1993)
weight	scales (0.1g)
wing moult	stage of moult in the primaries

7. Appendix B: Nest monitoring

7.1 Introduction

A component of this project that was cut from the final manuscript was a study on the breeding success of robins using motion sensitive cameras set-up at robin nests. The section was omitted from the study due to logistical issues associated with locating nests, the majority of located nests being situated in places where a camera couldn't readily be fitted, and a lack of confidence in the data gathered from monitored nests. Below is a rationale for why nest monitoring was to be included, followed by an account of the issues encountered, and the results that were generated.

7.2 Nest monitoring study

Australian passerines are generally characterized as having smaller clutches, longer breeding seasons, more broods and extended parental care, when compared with Northern Hemisphere passerines (Woinarski & Bulman, 1985; Rowley & Russell, 1991). These adaptations have been attributed to seasonal climatic conditions, year-round food availability, higher rates of nest predation and reduced adult mortality (Robinson & Rotenberry, 1991; Martin *et al.*, 2000). Studies on nestling survival rates demonstrate considerable variation, between very similar species (Dowling, 2003). An example of this is a study on scarlet robins *Petroica boodang* which found that only 8% of eggs laid produced fledglings (Robinson & Rotenberry, 1991), whereas another study on a similar species the grey-headed robin *Heteromyias albispecutlaris* reports 39% of eggs produced fledglings (Frith & Frith, 2000). Breeding success of a single species has also been shown to vary substantially both temporally and spatially (Powell & Frasch, 2000; Armstrong *et al.*, 2000). Currently, there is very little information available relating to how behaviour changes temporally or spatially and how behaviour relates to breeding success.

Having a good understanding of the breeding ecology and behaviour of a population is essential to evolutionary biologists and conservation managers (Dowling, 2003). Breeding ecology can provide insight into selection pressures and help in the development of realistic and effective modeling of risk assessments and population viability analyses. By studying how birds behave during the breeding period, and monitoring their success rate, I attempted to identify characteristic behaviours associated with breeding success.

Through assessment of the survivorship and number of successful broods produced by each mating pair of birds, I hoped to identify behaviours linked with successful reproduction. Additionally, as this project's primary goal was to explore methods for assessing habitat quality for fauna, successful recruitment rates would have been used as the baseline measure of an areas' value for conserving that species.

7.3 Hypotheses:

1. Parents in remnant patches will return to the nests more frequently than those in restored patches.
2. Nest predation rate will be higher in restored areas than remnants
3. Recruitment rates will consequently be higher in remnant sites than restored sites.

7.4 Field methods and design

Using the birds located for chapter 4, females were followed continuously for up to 20 minute intervals, by between one and four experienced bird watchers, to determine whether they have begun nesting (Dowling, 2003). Once a nest was located, a camera was to be set up in such a way that each time a bird returned to the nest a photo would be taken.

Unfortunately, locating nests proved to be more challenging than originally anticipated, and despite considerable efforts over a total of 16 full days of searching, only seven nests were located. Additionally, of the seven nests located, six were within 10 m of a road, and five of the seven were in restored areas. Given the small sample size, the skew towards areas where nests are likely to be more readily detectable, and the degree to which this conflicts with observed distributions of juvenile robins (majority having been detected in remnants), it was decided that this dataset was unlikely to be representative of nesting and breeding activities.

An additional complication was the placement of cameras at nests. Many of the nests were in areas where no camera could be fitted, either due to the camera's physical size being too large and heavy to be adequately supported by the surrounding vegetation, or because the nest was situated in such a place that the addition of a camera would likely draw the attention of nearby nesting ravens who are a known nest predator of the robins. As such only two cameras were fitted above nests, and the remaining five nests were monitored through routine checks every three to five days until the birds fledged or the nest failed.

7.5 Data analysis

Nesting attempts were defined as any nest that was built, regardless of whether eggs were eventually laid (Dowling, 2003). Total nesting success was to be calculated using all nests that were found to be active, regardless of whether they were found during nest-building or incubation. Parametric tests were to be used to analyse the data after the dependant variables were tested for normality (Dowling, 2003). All tests were to be two-tailed, with a significance criterion of $P < 0.05$. I intended to test whether the frequency with which the parents returned to the nest was correlated with food availability, as well as with the nestling survival rate. Additionally, if nest helpers were present then I would have incorporated them into the analysis to see if they had any effect on survival rate.

To test the effects male and female parents had on the survival of nestling birds, I intended to conduct a two-factor MANOVA with parent gender and frequency with which the birds returned to the nest as the factor independent variables, and successful fledging as the independent variable (Dowling, 2003). I also intended to compare the frequency with which birds returned to the nest in restored and remnant patches as the two factors in a t-test.

As a suitable dataset could not be collected in the available time, none of these analyses were conducted as the results would likely be misleading if used to predict community level patterns.

7.6 Qualitative results of monitored nests.

While only two nests were fitted with cameras during this study, the resulting images were somewhat noteworthy. The other five nests resulted in just two successfully fledged birds from a single nest, all others were either abandoned or predated.

The first nest that was monitored by a camera was attended to by two birds, both of which were colour banded. The female was observed sitting on eggs (two), and once the birds had hatched, she fed the birds until they fledged. The male was seen feeding the female while she was sitting on the eggs, and fed the chicks from the monitored nest. Interestingly, while

setting up the camera, the male was observed feeding recently fledged birds, possibly indicating that this was not their first brood of the season.

The second nest was maintained by two birds, one of which was colour banded, and the other carried an abnormality on his foot making him individually identifiable. These birds have been residents in that location since 2014 and were last seen in late 2017 in the exact same tree. The camera was set at the nest during construction while there were no eggs. The nest was visited by a raven just two hours after the camera had been placed, but the nest wasn't damaged. The following morning, however, a raven is observed taking eggs and destroying the nest. It is unclear whether it was the same raven.

Both nests were within 10 m of the road in restored areas. The predated nest was located near a raven nest, and subsequent breeding attempts by that pair of robins were observed without the use of cameras, as it was considered likely to increase their risk of nest predation.

8. Appendix C – Behavioural assessments

The following is a list of monitored behaviours and definitions used to classify observed actions. It should be noted that many of these behaviours are potentially subjective without adequate clarity on behavioural definitions and so caution should be taken when comparing this type of data with other studies. As such, I have defined the behaviours included in this study in the following table. Additionally, observer bias may influence the results, reducing the reliability of this type of data in meta-analyses. For this study, to minimise influence of observer bias, all behavioural data was collected by F. Holmes.

1. Perching (the bird is sitting, possibly scanning the area but is relatively stationary)
2. Flying (horizontal flight from one perch to another)
3. Preening
4. Calling
    a. No audible or visual response from other birds (announcing presence)
    b. Audible or visual response from another bird of the same gender (aggression)
    c. Audible or visual response from another bird of the opposite gender (courting)
5. Other territorial/courting displays
    a. Fighting/chasing a member of the same species, same gender
    b. Courting a member of the same species, opposite gender
    c. Nest building
    d. Brooding
    e. Feeding mate/juvenile
6. Foraging technique used, as described by Holmes and Recher (1986).
    a. *Probe or prise:* a bird inserts its beak part-way into a substrate to remove a food item. Prising involves lifting up or flaking off parts of a substrate.
    b. *Pounce:* a bird uses a deliberate jump (often to a surface above the bird or the ground) or short run to capture food.

c. *Glean:* a standing or hopping bird pecks at food on a nearby substrate. This includes reaching where a bird stretches up, down or away from its perch.

d. *Hang-glean:* a bird hangs upside-down in a stationary posture while taking prey from a substrate.

e. *Hover:* A flying bird hovers in the air for a brief period while picking a food item from a substrate.

f. *Snatch:* a bird makes a short flight or jump to capture prey from a nearby substrate. It does not land and usually returns to a different perch.

g. *Hawk:* a bird takes flight to capture a flying insect in mid-air.

h. *Nectar:* a bird probes a blossom to take nectar.

i. *Seeds:* birds take seeds from seed heads or capsules.

The sampling procedures used for this study were derived from the Donald *et al.* (2007) study on differences in the feeding ecology of male and female Raso Larks *Alauda razae* in Cape Verde. The sampling procedure was originally trialled on red-capped robins and golden whistlers, but it was determined that the sampling procedure was too short a period to collect meaningful data on the whistlers. Many of these trial surveys captured 5 minutes of the whistler sitting and calling, with no other behaviours observed. It is likely that with a longer time interval, it would be possible to capture a wider range of behaviours. Given that the data collection procedures were already highly labour intensive, and other bird species that were thought to be potential candidates for this study (singing honeyeaters and western gerygones) were difficult to observe and follow for extended periods of time, it was decided that it would be better to narrow the focus to just the red-capped robins.