The biodiversity value of revegetation

by

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Candidate's Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge, it contains no material previously published or written by another person, except where due reference is made in the text.

Nicola T Munro

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Preface

With the exception of the Introduction and Synthesis, this thesis is presented as a series of logically connected manuscripts. At the time of submission, all manuscripts were either published in peer reviewed journals, or have been peer-reviewed and are under revision. Each chapter appears as the published or reviewed paper, with the journal acknowledged at the beginning of each chapter. The content of the chapter is the same as the journal manuscript, except the labelling of tables and figures, which have been modified to be consistent with the presentation of the thesis. For example, in Chapter 2, the original Figure 1 has been changed to Figure 2.1.

The vast majority of the work has been conducted by the primary author (Nicola Munro), including the original project proposal, establishment of field sites, data collection, literature searches, statistical analyses (with guidance from others) and manuscript writing. However, all papers have co-authors to acknowledge specific contributions made by others to the different papers. Joern Fischer and David Lindenmayer contributed to all manuscripts by proof-reading and helpful discussions. Joern Fischer and Jeff Wood provided statistical guidance for all analyses except CONOCO (Chapter 3). Adam Leavesley greatly assisted in the analysis of bird composition in plantings with the use of the statistical program CONOCO. Geoff Barrett provided valuable discussions and design advice for Chapter 3.

Because each chapter was designed as a stand-alone paper for publication, there is some unavoidable repetition between chapters. There are also some minor stylistic differences between chapters, which reflect the differences between journal styles. For example, Chapter 5 was published as a research note in Ecological Management and Restoration, and therefore does not contain an abstract. Cited material is presented at the end of each chapter rather than as a single reference collection at the end of the thesis.

Abstract

The last few decades have seen a substantial increase in the establishment of revegetation plantings in production landscapes to mitigate the impacts of prior vegetation clearing. These plantings can have a variety of purposes, such as stream bank stabilisation, lowering of water tables, erosion control, and livestock protection. All plantings have the potential to also provide habitat for wildlife, but the extent to which this occurs can differ substantially depending on the attributes of the plantings. In an agricultural landscape in south-eastern Australia, I compared revegetation plantings differing in a number of key attributes. The overarching goals were to determine if: a) plantings established with high plant diversity provided greater biodiversity outcomes than plantings established with low plant diversity, and b) what other attributes of a planting provided greater biodiversity outcomes. Specifically, I identified two types of revegetation: (1) '*ecological plantings*' which were planted for ecosystem restoration purposes and were characterised by a diverse assemblage of tree, shrub and understorey species; and, (2) '*woodlot plantings*' which were planted with low plant species richness of primarily overstorey species (agro-forestry plantings). Both types of plantings were established with predominantly local, native vegetation. I investigated 27 ecological plantings and 16 woodlot plantings, both ranging in age from 2 to 26 years. In addition, I compared the plantings to 11 paddocks (cleared agricultural land) and 18 uncleared remnant sites, as the starting point and goal, respectively, of revegetation. Across a total of 72 sites, I compared the development of vegetation structure and floristic richness, the bird and mammal communities present in the sites, and the ecological function of the sites.

Vegetation structural complexity increased with age of planting, toward that of remnants, even when very few species were planted at establishment. Species richness of plants, however, did not increase with age, indicating that colonisation did not occur through time. Therefore, plantings may not provide for the conservation of non-planted flora. Ecological plantings were more similar to remnant vegetation in structure and species composition, and contained greater shrub cover, more plant lifeforms, and lower weed cover than woodlot plantings. In general, ecological plantings can achieve a similar overall structural complexity to remnant vegetation within 30 years, but will not gain a ground layer of native plants, and will not necessarily contain some structural features by this age (such as tree hollows and fallen timber).

Bird species richness in both ecological plantings and woodlot plantings, by approximately 30 years of age, was similar to that of remnants. Bird species richness was greater in ecological plantings than in woodlot plantings. Also, the species composition differed between these two

types of plantings. Ecological plantings contained a shrub-associated bird assemblage, which included some species of conservation concern, whereas woodlot plantings were dominated by generalist bird species. Remnants contained a unique bird assemblage including bark specialists and hollow users, which were not found in either of the two types of plantings. Ecological plantings appeared to provide habitat for birds sooner than woodlot plantings. Bird species richness (particularly of forest birds) responded positively to structural complexity, but not to floristic richness. Bird species richness was greater in plantings that were larger and older, and that were more similar to reference remnant sites. The quality of the site (similarity to remnant condition) appeared to be more important for bird richness than the quantity of woody vegetation cover surrounding the site.

Many native mammals, both terrestrial and arboreal, can use revegetation plantings, at least temporarily, and within only a few years of establishment. Most mammal species commonly occurring in the region were observed in both remnant and planting sites. The presence of old remnant trees in a planting significantly improved the habitat value of revegetation for the common ringtail possum (*Pseudocheirus peregrinus*) and possibly other hollow-using mammals.

As well as the development of structure and composition, the return of ecological functions is also considered important for restoration success. I explored the measurement of ecological function using a widely applied rapid assessment tool called Landscape Function Analysis. Landscape Function Analysis is based on the measurement of proxies that correlate with the functions of soil stability, water infiltration and nutrient cycling. I compared scores for soil stability, water infiltration and nutrient cycling between our woodlot plantings, ecological plantings, paddocks and remnants. However, I found that the Landscape Function Analysis was outside its optimal sensitivity range when applied to revegetation plantings. Thus, this tool may not be sufficiently sensitive enough to appropriately reflect ecological function in restoration plantings.

In summary, ecological plantings, characterised by high structural complexity and floristic diversity, can provide habitat for a distinct assemblage of birds and mammals which respond to certain features in the plantings, such as a complex shrub layer and old remnant trees. Ecological plantings may also provide habitat sooner than woodlot plantings for a range of fauna. Woodlot plantings are, however, still of some ecological value, and can provide habitat for many generalist species. In the short term (30 years), neither ecological plantings nor woodlot plantings are viable replacements for remnant vegetation, although ecological plantings appear to be on a trajectory toward a similar set of conditions to that typical of remnant sites. I conclude that ecological plantings have demonstrably higher value for plants and animals than woodlot plantings. Because of this, restorationists should give priority to the structure and composition of a planned revegetation site, rather than focussing solely on its size and location.

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Chapter 1: Introduction

1.1 Background

Clearing of woody vegetation, primarily for agriculture, has been both rapid and extensive throughout much of the world (Food and Agriculture Organisation 2006), and indications are that this will continue (Sala et al. 2000, Food and Agriculture Organisation 2006). Frequently associated with land clearing are habitat loss for flora and fauna, as well as degradation such as wind and water erosion, reduced water quality, stream bank instability, changes to ground water and salinity effects, and alterations to the chemical and nutrient make-up of the soil (Yates and Hobbs 1997). These degradation processes can be costly to manage or reverse, and can result in lower agricultural productivity, and loss of ecosystem services and a loss of biodiversity (Foley et al. 2005).

Ecosystem degradation negatively affects human health and well being, as well as the existence of other fauna (Millennium Ecosystem Assessment 2005). Habitat loss is a key threatening process driving the decline of many terrestrial species (Yates and Hobbs 1997, Fahrig 2003, Lindenmayer and Fischer 2006). Fragmentation of the remaining habitat can further exacerbate this decline (Fahrig 2003, Lindenmayer and Fischer 2006). In wealthy countries such as Australia, land clearing has been reduced as a result of legislation in recent decades (Douglass 1997). Nevertheless, many degradation processes are continuing (SoE 2006). Salinity and erosion of soils and stream banks are ongoing problems facing landholders, and many species of native flora and fauna are still declining in abundance and range (Yates and Hobbs 1997). The cessation of clearing has not been sufficient to halt many degradation processes. It is clear that many areas of agricultural land require restoration (Matson et al. 1997).

It is now well recognised that biodiversity conservation cannot rely solely on protected areas in their current extent (Pimental et al. 1992, Fischer et al. 2006). Conservation on agricultural land is also required. This may include setting areas aside specifically for conservation, or incorporating conservation practices in the production areas of a farm (Fischer et al. 2008).

To halt or reverse habitat loss and other degradation processes, restoration is required (Hobbs and Harris 2001). The Society for Ecological Restoration International defines ecological restoration as 'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed' (Society for Ecological Restoration International Science and Policy Working Group 2004). Restoration can take many different forms, and differs particularly in different regions of the world with different human histories. For example, agriculture in Europe has occurred for such a long time that pre-European conditions are no longer considered the restoration trajectory of choice. Instead, most biodiversity now occurs in landscapes managed for agriculture, but particularly under historical agricultural management (pre-industrial agriculture). Modern agricultural practices cause biodiversity decline. Restoration in much of Europe, therefore, focuses on re-establishing pre-industrial agricultural regimes, such as grazing native pastures (Lindborg 2005) and maintaining small fields and numerous hedges and ditches (Arnold 1983). The situation in Europe appears to be relatively unique, and can be considered at one end of a spectrum of restoration approaches. At the opposite end of the spectrum, in frontier colonies such as Australia and North America, the restoration trajectory is (almost) unquestioningly the pre-European condition of only a few centuries ago. The restoration goal is perceived as a wilderness untouched by humans, (despite the long presence of native peoples in both continents). In these two regions, agriculture is considered a prime driver of biodiversity loss, and therefore restoration approaches typically aim to soften agricultural land by establishing within it new patches of trees which resemble the pre-cleared vegetation.

Between the European situation at one end of a spectrum of desired restoration goals (cultural landscapes), and the Australian and North American situation at the other end (wilderness landscapes), the rest of the world may be considered to lie somewhere in between. A very long history of traditional agriculture has occurred in much of Africa, South America and the Indian Subcontinent. Areas of traditional agriculture may be considered of benefit to biodiversity in some cases (Raman 2006, Hylander and Nemomissa 2008, Ranganathan et al. 2008), although much biodiversity of conservation concern remains primarily in undisturbed forests (Daily et al. 2001, Smith-Ramirez 2004), and agricultural abandonment can lead to recovery of forest ecosystems, with positive consequences for biodiversity (Aide and Grau 2004, Aerts et al. 2008).

In Australia, agricultural expansion has resulted in the clearing of vast areas of native vegetation, typically woodlands and forests. Restoration, therefore, usually takes the form of returning trees to agricultural landscapes, in a semblance of the woodlands and forests that previously occurred there. Returning trees to such landscapes can be achieved passively (natural regeneration) or by active planting of seed or seedlings. Globally, there have been several documented cases of natural regeneration, (or secondary succession), particularly from tropical regions of Central America and elsewhere (James 1982, Finegan 1996, Zimmerman et al. 2000, Lugo and Helmer 2004, Bowen et al. 2007), and some cases of naturally regenerating Eucalypts in Australia (Arnold et al. 1999, Dorrough and Moxham 2005). However there have also been documented cases of abandoned agricultural fields remaining uncolonised by native vegetation for many decades (Duncan and Chapman 1999, Wilkins et al. 2003).

In Australian agricultural landscapes, many plantings have been established by the deliberate introduction of seeds or seedlings because natural regeneration was considered too slow or insufficient (Dorrough and Moxham 2005). In such cases, sites appear to have passed a threshold, and have entered a new stable state where structure and function are simplified (Aronson et al. 1993). The aims of revegetation are to shift the degraded ecosystem back past such a threshold, and to accelerate the restoration process (Lamb et al. 2005).

In Australia, as elsewhere, the re-establishment of woody vegetation that was previously cleared, has resulted in the form of different kinds of tree plantings, including plantation establishment (often termed 'reafforestation'), agro- or farm-forestry, and ecosystem restoration plantings (often termed 'revegetation'). These different types of plantings differ in their purpose and plant diversity, and may also differ in their value to biodiversity. Often, an individual planting may be established to meet multiple objectives. For example, a landholder may establish a planting to provide shade and wind shelter for stock, as a source of future firewood, to stabilise eroding soils, for aesthetic improvement, and to provide habitat for fauna (Brandle et al. 2004). The extent to which a planting will fulfil these objectives will depend on many factors, including some attributes of the planting itself, such as its location, size, shape and vegetation structure and composition.

The 'biodiversity value' of a revegetation planting can be loosely defined by the extent to which the site is biologically similar to the pre-cleared vegetation. Because pre-cleared conditions are usually unknown, nearby uncleared remnant vegetation can provide a reference condition with which to compare a revegetation site (Gibbons et al. 2008).

Much international research on the biodiversity value of revegetation has been very recent and examined a range of approaches to revegetation. In Europe, revegetation typically consists of hedgerows or grassy field margins; in North America and Australia, revegetation often occurs as fencerows or shelterbelts (narrow strips between fields); some large-scale plantings occur in the United States of America and China (Wenhua 2004); and commercial plantations occur throughout the world (Food and Agriculture Organisation 2006). In this thesis, I have focussed on revegetation in agricultural landscapes. I do not discuss in depth other kinds of revegetation such as naturally regenerating forests, orchards, mine site rehabilitation or restoration of non-woody vegetation.

A general consensus of international research indicates that revegetation plantings appear to contain more native biota than pre-planted conditions (fields or paddocks), but less than remnant vegetation. This mid-way position appears consistent for vegetation structural complexity (Allen 1997, Ruiz-Jaen and Aide 2005b), and plant and animal species richness and composition (Finegan 1996, Christian et al. 1998, Raman 2006, Gardner et al. 2007). A number of attributes appear to be absent from plantings, such as logs and many understorey plant species (Finegan 1996), which may affect the biodiversity found in the planting. The biodiversity value of revegetation has been very poorly studied to date.

1.2 Management implications on the biodiversity value of revegetation

Around the world, the majority of plantings are established with very few species, and most agro-forestry plantations are monocultures. One suggestion for improving the biodiversity value of plantings is to plant a mixture of tree species. This can result in some increase in the biodiversity value of the site (Leopold et al. 2001, Kanowski et al. 2005).

Another recommendation is to increase the vegetation structural complexity by increasing the plant diversity of the mid and understorey. To do this with little cost, a 'foster ecosystem hypothesis' has been proposed (Haggar et al. 1997), whereby planted trees will facilitate the natural colonisation by understorey and midstorey species. Some successes of this method have been reported from tropical plantations (Silva Júnior et al. 1995, Keenan et al. 1997, Lugo 1997, Parrotta et al. 1997a, Leopold et al. 2001), but failures have also been reported, particularly from temperate areas (Duffy and Meier 1992, Allen 1997, Wilkins et al. 2003, Aubin et al. 2008).

The objective of some plantings is to recreate, or restore, the pre-cleared vegetation by planting many native plant species of many different lifeforms. Such 'ecological' or 'restoration' plantings are ambitious and costly (Erskine 2002, Catterall et al. 2004, Lamb et al. 2005). They may, however, provide greater biodiversity value than plantation-style revegetation. Almost no research has investigated the biodiversity value of high-diversity 'ecological plantings'.

This thesis contributes information on the biodiversity value of ecological restoration plantings. To understand the gains in biodiversity achieved in an 'ecological planting', I compared these to structurally and floristically simple 'woodlot plantings'. I also identified a number of attributes of plantings that can improve the biodiversity value. This information will be of value to restoration practitioners and funding agencies when making decisions on the biodiversity gains associated with different types of plantings.

1.3 Study area

I conducted my study in Gippsland, Victoria, in south-eastern Australia (for map see Fig. 3.1). This was an ideal location for the study for several reasons. First, extensive land clearing for agriculture (approximately 150 years ago) has caused many degradation problems, which in turn have prompted many revegetation projects on most farms in the area. This provided many plantings from which to select study sites. Second, the area was one of the first in Australia to establish a Landcare network, a now national organisation which assists farmers in sustainable farm management. In Gippsland, the Landcare network encouraged, through funding and expertise, the establishment of many ecological plantings, rather than more widespread monoculture shelterbelts. The Landcare network was also particularly interested in my findings, and was most helpful in endorsing my project and providing access to landholders. Third, the area has particularly rich soils and high rainfall, so plantings grow quickly. Within ten years, a planting in Gippsland can develop a closed canopy and abundant understorey. Plantings in Gippsland achieve a structure within five to ten years that would take plantings in woodland areas of Australia several decades to achieve (NM pers. obs.).

Prior to clearing, the study area was dominated by structurally complex coastal forest (a mixture of wet and dry forest types) (Korumburra and District Historical Society Inc. 1998). It was called the 'land of the lyrebird'. The lyrebird (*Menura novaehollandiae*) is a large, predominantly ground-foraging bird, associated with wet mature forests. It is now extinct in the study area. European colonisation and associated forest clearing, as well as introduced pests, have also caused the local extinction or decline of a number of other bird and mammal species. There is some hope that revegetation works will prevent further declines in fauna species, and provide habitat for those species still remaining.

Agricultural cleared land (termed 'paddocks') which surrounded the plantings were predominantly grazed pastures. Because there were few paddock trees in these pastures, and planting and remnant vegetation was dense, this landscape conformed to a typical patch-matrix model of vegetation cover (Forman 1995), rather than the variegated model predominant in much of the woodland areas of Australia (McIntyre and Barrett 1992). This vegetation distribution has enabled simple calculations of shape, size and surrounding vegetation cover, which were used in most models.

The majority of local landholders in the study area had conducted at least some revegetation works. Most landholders hope their plantings will provide biodiversity value, particularly habitat for birds, as well as provide shelter for stock, limit erosion, increase water quality, and control salinity. Most of the 30 landholders in my study expressed a desire to understand the biodiversity value of their plantings.

For my study I selected a total of 72 sites of four site types. I differentiated between 'ecological plantings' planted for restoration purposes, with many species of trees, shrubs and understorey, and 'woodlot plantings' which were predominantly eucalypt trees only. I also selected sits of remnant vegetation and cleared agricultural land (paddocks), as reference points. I selected plantings of a range of ages and sizes. I also tried to select half my sites with old remnant trees, and half without (excluding remnants), and also half my sites in riparian locations and half away from watercourses. I was not always able to find sites that met all my criteria.

Remnant sites were the highest condition patches available in the study area, but had been subject to low levels of timber extraction and suffered some weed invasion. Remnants were not grazed. Plantings of both types were generally not grazed, although several experienced cattle incursions during the study, and one woodlot planting was gazed lightly by sheep. All paddocks were continuously grazed. For images of the sites see Figure 6.1.

1.4 Aims and Objectives

The over-arching objective of this study was to provide practical information to assist restoration practitioners and landholders, on the likely biodiversity gains of a revegetation planting, depending on particular management actions. Despite large financial investments in revegetation programs, little is known of their effectiveness in biodiversity conservation.

Specifically, I aimed to compare existing revegetation plantings that differ in structural complexity and floristic richness (broadly reflecting the level of effort to re-create the original vegetation) with remnant vegetation and cleared paddocks, to determine the degree to which a planting can provide biodiversity benefits. I also aimed to evaluate the ecological value of these site types with respect to vegetation structure and floristics, faunal richness and composition, and ecosystem function, in relation to specific management actions, such as planting around existing old remnant trees, planting along riparian areas, and the size and shape of plantings.

1.5 Thesis structure

This thesis is written as a linked collection of papers, all of which have either been published or are in revision, at the time of submission. Chapters presented here may differ slightly to the published version. Because each chapter is written as a stand-alone paper, there is, therefore, a degree of overlap between some chapters, such as the description of the study site.

At the commencement of this thesis, there were two previous reviews of the faunal colonisation of revegetation in the Australian context (Kimber et al. 1999, Ryan 1999). Both had a very limited number of studies to review (less than six), and birds were the only studied response group at the time. There has been much more research on revegetation since these previous reviews. In Chapter 2, I provide a current review of what was known of faunal use of revegetation in Australia at the commencement of the doctorate. Knowledge gaps are outlined in that review.

Chapters 3 to 6 report the results of original empirical work, which was conducted in 2005/2006. Chapter 3 focuses on several aspects of the vegetation in plantings, which forms an important foundation for research on fauna in revegetation (Chapters 4 and 5) and the ecological function of revegetation (Chapter 6).

Chapter 3 focuses on the development of vegetation structural complexity and floristic richness and composition in revegetation. This was undertaken with two primary considerations in mind. First, the vegetation at a site influences faunal colonisation and also the ecosystem function of a site. Second, provision of habitat is usually considered only in the context of fauna, whereas many native plant species also have undergone decline and fragmentation and some also require additional habitat which may be provided by revegetation plantings.

Chapter 4 outlines the findings of bird surveys in plantings, specifically the planting attributes which influence the bird richness and composition at a site. Birds are a good group to study because of their ease of observation and also they encompass a variety of guilds. Because many birds are mobile, they may colonise plantings relatively soon after establishment.

Chapter 5 presents findings on the mammals found in the revegetation plantings, and the planting attributes which influence their presence. Some mammals may be dispersal-limited, so quantifying the extent to which this impairs colonisation is important. Animals such as the common brushtail possum (*Trichosurus vulpecula*), common ringtail possum (*Pseudocheirus peregrinus*) and koala (*Phascolarctos cinereus*) are largely arboreal, and crossing open tracts of farmland may pose risks for them. The extent to which revegetation harbours introduced predators, such as the feral cat (*Felis catus*) and red fox (*Vulpes vulpes*) is also a key conservation issue.

In Chapter 6, I discuss the ecosystem function of revegetation. The return of functional capacity has been recognised as a key restoration measure of success (Society for Ecological Restoration International Science and Policy Working Group 2004), although it has been poorly defined and very rarely measured. I used a method to approximate ecosystem function which had been developed for measuring restoration success in mining areas (Tongway and Hindley 2004). I hoped this method would provide interesting insights into the functional capacity of plantings, particularly with a view to determining the trajectory of functional development, and also any differences between plantings of high plant diversity and those of low diversity. Problems were encountered, however, which enabled, instead, a critique of the method for use in revegetation plantings in agricultural landscapes.

Finally, Chapter 7 provides a synthesis of results, a general discussion, implications for restoration practice, and suggested future research directions.

Several papers were written in addition to the core ones in this thesis. A brief list of these is presented as an appendix with some summary remarks about their implications for restoration.

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Chapter 2: Faunal response to revegetation in agricultural areas of Australia – a review

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Faunal response to revegetation in agricultural areas of Australia – a review

2.1 Summary

We reviewed the literature on fauna in revegetation in Australian agricultural areas. Of 27 studies, 22 examined birds, with few studies focussing on other faunal groups (four to six studies for each remaining group) and nine examined multiple groups. Existing evidence suggests that revegetation provides habitat for many species of bird and some arboreal marsupials. Species richness of birds was greater in revegetated areas that were large, wide, structurally complex, old, and near remnant vegetation. Bats, small terrestrial mammals, reptiles and amphibians did not appear to benefit significantly from revegetation in the short term. Evidence to date suggests that revegetation is not a good replacement of remnant vegetation for many species. Key information gaps exist in the faunal response to 1. revegetation as it ages; 2. different structural complexities of revegetation; 3. revegetation that is composed of indigenous versus non-indigenous plant species; and 4. revegetation that is in riparian versus non-riparian locations. In addition little is known on the value of revegetation for declining or threatened fauna, or of the composition of fauna in revegetation. There is a need to better understand the balance between quantity of revegetation in the landscape, and the quality or complexity of revegetation at the patch scale. Based on current evidence, we recommend revegetation be conducted in patches that are large, wide, and structurally complex to maximise the benefits to fauna.

2.2 Keywords

Revegetation, habitat, restoration, plantation, structural complexity

2.3 Introduction

Throughout Australia, land clearing for agriculture has caused land degradation such as salinity and erosion (Bird et al. 1992, MDBC 1999), and the loss of native biota (Saunders 1989, Ford et al. 2001). The re-establishment of vegetation has been suggested as a potential solution to these problems (Hobbs and Saunders 1993, Hobbs 1993, Barrett 1997).

Revegetation may have several ecological benefits, for example by lowering water tables (Stirzaker et al. 2002) and providing some habitat elements for wildlife (Kimber et al. 1999, Ryan 1999).

The faunal response to revegetation in Australian agricultural areas has been reviewed by Ryan (1999) and Kimber et al. (1999). Both reports concluded (from the small number of studies then available) that revegetated sites provided habitat for a range of bird species (the only taxa studied) although the majority of these were generalist or edge species, and birds with specialised needs were not provided for by revegetation. Substantially more research has been conducted since the earlier reviews providing the impetus for this paper.

We review the use by fauna of revegetation in Australian agricultural landscapes and the effectiveness of different revegetation strategies. We define 'revegetation' as an area where native plants have been actively introduced, but we do not stipulate by what method those plants were established. Our definition of 'revegetation' includes all plantings of woody vegetation (excluding grasslands) in an area where woody vegetation previously occurred, and where the planted vegetation is native to Australia (but not necessarily locally indigenous). This includes both single species and multi-species plantings (Fig. 2.1). We exclude plantations of exotic species (e.g. *Pinus radiata*), plantations dominated by tree crops (e.g. orchards) and industrial-scale plantations, to focus the review on small-scale farm and community plantings. We define two types of revegetation based on structural complexity (Fig. 2.1): 'simple tree plantings' include windbreaks, community plantings, woodlots, and other farm plantings that are structurally simple; and 'ecological restoration plantings' which aim to re-create the vegetation communities present prior to land clearing and are usually structurally and floristically diverse. 'Structural complexity' is defined as the number of different attributes present and relative abundance of these attributes (McElhinny et al. 2005). We explain how authors have measured structural complexity where possible and appropriate.

We summarise the responses of different taxa to revegetation, and discuss the faunal response to different attributes of revegetation, such as size and shape. We conclude by outlining priorities for future research and revegetation management.

2.4 Methods

We reviewed all known scientific literature on the faunal response to revegetation in Australian agricultural areas (29 articles describing 27 studies; Table 2.1). Literature was found by searches though databases and citation lists and interviews with experts. For online searching we used the terms 'reveg*', 'restor*', 'reafforest*' or 'planting', plus terms for taxa including select individual species where appropriate (e.g. 'possum', 'koala'). Databases searched were Web of Science and Google. Anecdotal descriptions were not included. More than half the articles were published in peer reviewed journals (18 of 29 articles). Five articles were theses, resulting in one journal publication. Where multiple publications were produced from the same study (e.g. a journal publication from a thesis) we used the journal article. Four articles were reports. The remaining two articles were one booklet and one book chapter.

Different studies explored different combinations of site types. Site types examined in this review were remnant (woody) vegetation, ecological restoration planting (high plant species diversity), simple tree planting (low plant species diversity), and cleared farmland. Many studies compared plantings to reference sites such as remnants (22 studies) or cleared farmland (15 studies; Table 2.1). More studies examined simple tree plantings than ecological restoration plantings (19/27 versus 11/27), although six compared these two revegetation types.

Most studies examined birds as a response variable (22 studies). There were four to six studies for each of the following groups: arboreal marsupials, small terrestrial mammals, bats, reptiles, amphibians, invertebrates (Table 2.2). Nine studies examined multiple taxa. Most studies were conducted in woodland (17 studies), and in areas with a temperate climate (22 studies). Three studies were conducted in tropical or subtropical rainforest (Table 2.1).

Information on site attributes was not always available in the reviewed articles. Frequently missing was information on the age, size, isolation and complexity of the revegetation. Missing information has hampered this review. Several studies had limited replication: four studies had < 10 sites and only eleven studies had > 50 sites (Table 2.1).

2.5 Results

2.5.1 Birds

Typically revegetation did not support the bird richness or composition characteristic of remnant vegetation (Crome et al. 1994, Leary 1995, Green and Catterall 1998, Klomp and Grabham 2002, Hobbs et al. 2003, Kinross 2004, Kavanagh et al. 2005, Loyn et al. 2007).

Conversely, compared with open farmland, revegetation typically supported more bird species (Leary 1995, Green and Catterall 1998, Klomp and Grabham 2002, Hobbs et al. 2003, Catterall et al. 2004, Loyn et al. 2007), more woodland/forest dependent species (Loyn et al. 2007) and more declining species (Leary 1995, Kinross 2004).

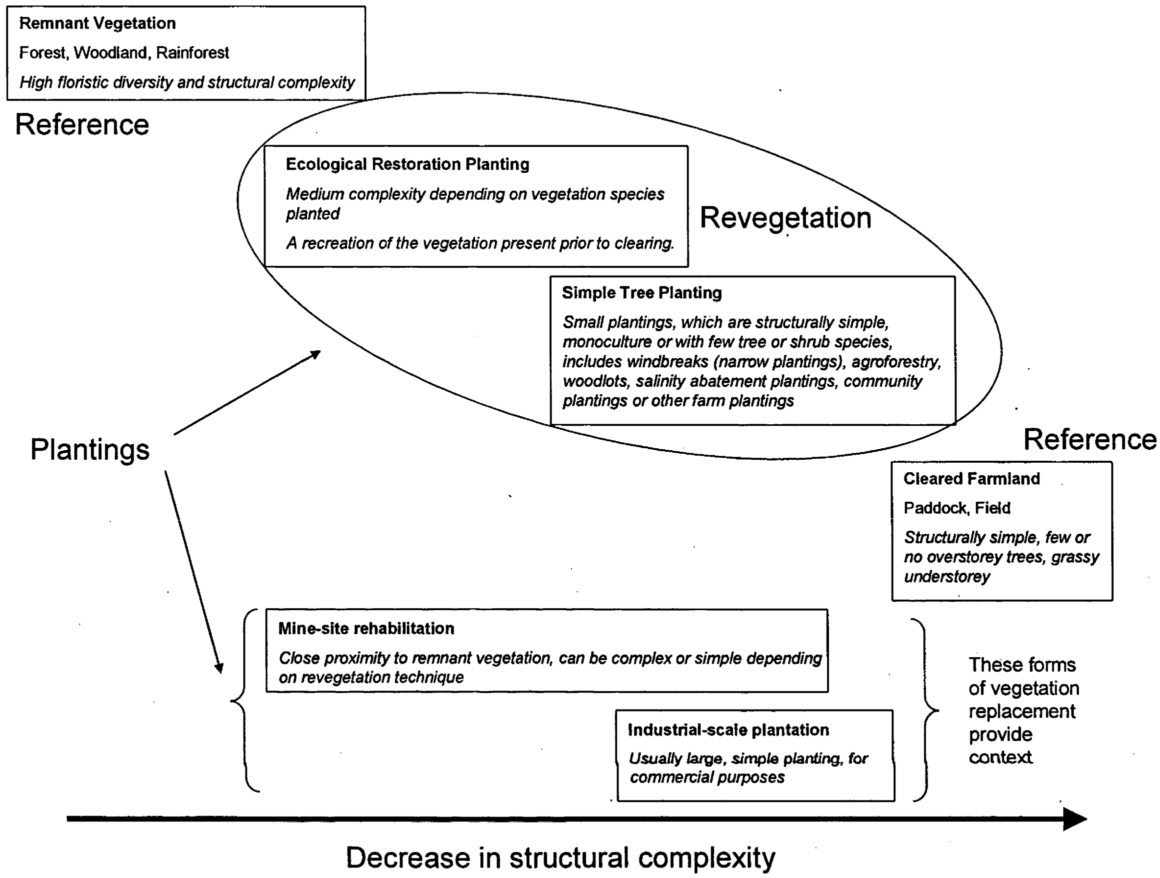

Figure 2-1 Overview of the terms used in the paper, on a scale of structural complexity

'Revegetation' includes ecological restoration plantings and simple tree plantings. These forms of revegetation are compared to reference areas of remnant vegetation and cleared farmland. Industrial-scale plantations, mine-site rehabilitation and re-growth vegetation are not of primary concern in this review. Collectively all forms of active vegetation establishment are called 'plantings'. Note that remnant vegetation sites in this review are timbered woodland, forest and rainforest.

Nichols and Nichols (2003) suggested that birds recolonising rehabilitated mine sites respond to the development of vegetation structure and diversity. A correlation between bird species richness and remnant vegetation complexity has been demonstrated in Australian ecosystems (Gilmore 1985, Hobbs et al. 2003, Rossi 2003). Revegetation does not approximate the floristic and structural diversity of remnants in the first few decades after establishment (Kanowski et al. 2003). Several studies observed that bird species richness was higher in complex revegetation than in simple revegetation (Harris 1999, Barrett 2000, Arnold 2003, Hobbs et al. 2003, Rossi 2003, Kavanagh et al. 2005). However, most of these studies did not measure complexity directly. Rossi (2003), the only author to do so, defined complexity as the number of stratas present (out of 17).

Recent revegetation guides suggest that planting local plant species should benefit local fauna (Bennett et al. 2000). This has been implicitly tested in only one study: the diversity of woodland birds was greater if local native plants were established, and conversely, exotic birds were more diverse if exotic trees were planted (Barrett 2000).

Bird abundance and species richness are relatively simple measures. Of perhaps more importance to restoration is the bird community composition in revegetation. Several studies found that the bird composition in revegetation as young as five years after mining resembled that in the surrounding forest, depending on the development of the vegetation, particularly the understorey (Nichols and Watkins 1984, Armstrong and Nichols 2000). Borsboom et al. (2002) found that largely undisturbed 40 year old simple eucalypt plantings approached the plant species richness and abundance of selectively logged old-growth forest, and also approached the bird species richness and composition of the reference forest. This latter project, however, was unable to separate the effects of plantation age and structural complexity (because complexity increased with age). Catterall et al. (2004) separated these effects and compared ecological restoration plantings (high structural complexity) with simple tree plantings (low complexity) of the same age and found that bird composition in ecological restoration plantings was closer to that in remnant forest than in simple tree plantings.

Structural complexity of revegetation, as measured by the cover or abundance of a number of vegetation attributes, increases with age (Kanowski et al. 2003, Martin et al. 2004). Possibly because of this increased complexity as well as increased time for recolonisation, bird species richness also tends to increase with revegetation age (Biddiscombe 1985, Taws et al. 2001, Borsboom et al. 2002, Martin et al. 2004). Common bird species can recolonise revegetation within two to three years (Biddiscombe et al. 1981, Taws et al. 2001, Martin et al. 2004), and many declining and uncommon birds after eight years (Taws et al. 2001). However, some bird species, such as bark-foragers, had not recolonised revegetation in northern New South Wales after 50 years (Martin et al. 2004). Woinarski (1979) noted that guilds such as granivores, nectarivores, frugivores and bark-gleaners were absent or uncommon in 25 year old simple tree plantings.

Many revegetation guides recommend maintaining remnant features such as old trees, logs and rocks (Barrett 2000, Bennett et al. 2000, Salt et al. 2004). Few studies have examined the bird responses to remnant features, although some have found increased bird diversity in plantings with retained large trees (Kavanagh and Turner 1994, Taylor et al. 1997, Barrett 2000, Grabham et al. 2002).

Only two studies investigated the response of birds to planting area, with differing results. Borsboom (2002) found no correlation between bird species richness and simple tree planting area. Kavanagh et al. (2005) found that bird species richness and abundance had a strong positive response to patch size. These studies differed in their ranges of patch sizes and complexity, with the former being small simple eucalypt plantings (1.5ha to 10.5ha), and the latter including large ecological restoration plantings (<5ha to >1000ha).

Several studies identified width of revegetation as being positively correlated with bird species richness (Taws et al. 2001, Merritt and Wallis 2004, Kavanagh et al. 2005) or richness of forest/woodland birds (Kinross 2000). The composition of birds in wider revegetation patches was no different to that in narrow revegetation patches (comparing <15m with >19m sites), although some small insectivorous species preferred wider sites to narrow (Kinross 2004).

Landscape-level attributes of revegetation have been little studied. Hobbs et al. (2003) found that adjacency to remnant vegetation increased the abundance of some birds in simple tree plantings, but overall differences between isolated plantings and those adjacent to remnant vegetation were relatively small. Kavanagh et al. (2005) compared birds in revegetation in two landscapes differing in vegetation cover – variegated and cleared – and found no difference in the total numbers of bird species in each landscape. Cunningham et al. (in press) demonstrated that bird richness was greater where the total area of both remnants and revegetation was greater. Also, the effect of plantings was greater on farms with little remnant vegetation, than on farms with more remnant vegetation (Cunningham et al. in press).

2.5.2 Arboreal marsupials

Studies of arboreal marsupials have shown that some members of this group can recolonise revegetated areas if hollows (a key resource) are present or provided (e.g. nestboxes) (Suckling and Goldstraw 1989, Irvine and Bender 1997, Smith and Agnew 2002, Kavanagh et al. 2005). While revegetation can sometimes provide habitat for arboreal marsupials, this group is typically more abundant in remnant vegetation (Green and Catterall 1998, Kavanagh et al. 2005). Cleared farmland provides almost no habitat for arboreal marsupials (Green and Catterall 1998, Kavanagh et al. 2005).

Older revegetation sites contain more arboreal marsupials than young sites (Kavanagh et al. 2005). The older areas of revegetation in that study were 20 to 25 years old, and so were unlikely to provide nesting hollows (Gibbons and Lindenmayer 2002) – hence it is unclear why these older sites contained more arboreal marsupials. Kavanagh et al. (2005) also found that arboreal marsupials were more abundant in relatively large revegetation sites (>5ha), but did not respond to planting width (where a narrow site was <50m wide).

A study by Cunningham et al. (2007) found that farms and landscapes with many revegetation plantings supported a lower abundance of arboreal marsupials. This was attributed to those farms supporting less remnant vegetation than farms and landscapes with few plantings.

2.5.3 Small native terrestrial mammals

Two of four studies examining small native terrestrial mammals had sufficient data to indicate the value of revegetation as habitat. In one study, two species were observed, and both occurred only in remnant vegetation and not in simple tree plantings (Hobbs 2003, Hobbs et al. 2003). In the other, one species was ubiquitous, and three were more abundant in remnant vegetation than simple tree plantings (Rossi 2003). Habitat complexity of plantings (as measured by the number of stratas including ground cover elements) explained most variability in native mammal richness (Rossi 2003).

2.5.4 Bats

Three studies provided results of bats in revegetation. Hobbs et al. (2003), Kavanagh et al. (2005) and Law and Chidel (2006) found greater bat foraging activity in remnant vegetation than in revegetation, and Hobbs et al. (2003) found greater species richness in remnant vegetation, whereas Kavanagh et al. (2005) did not. There also were mixed responses when bat activity in cleared farmland was compared to that in revegetation. Kavanagh et al. (2005) and Law and Chidel (2006) found no differences between revegetation of any size and cleared farmland, whereas Hobbs et al. (2003) found more bat activity in cleared farmland compared to an isolated simple tree planting, but less compared to a planting near a remnant. Law and Chidel (2006) found more bat activity in older revegetation than in younger revegetation, but Kavanagh et al. (2005) and Hobbs et al. (2003) did not.

Bats appeared to be insensitive to revegetation size and width as well as to the amount of vegetation cover in the landscape (Kavanagh et al. 2005, Law and Chidel 2006). Bat species richness and activity was negatively correlated with shrub cover, possibly because many bats experience 'structural clutter' which reduces foraging ability (Kavanagh et al. 2005).

2.5.5 Reptiles

Five of six studies examining reptiles had sufficient data to indicate responses to revegetation. Typically remnant vegetation contained more reptile species and higher abundances than revegetation, and revegetation supported more species than cleared farmland (Borsboom et al. 2002, Hobbs et al. 2003, Kavanagh et al. 2005, Kanowski et al. 2006). Kanowski et al. (2006) found mixed responses depending on the species of reptile, and whether they were rainforest-dependent, or habitat-generalists. In the south-west slopes of NSW reptile abundance and species-richness were not affected by revegetation age, width or size (Kavanagh et al. 2005). Reptiles in general (Kavanagh et al. 2005, Cunningham et al. 2007) and in one study, rainforest-specialised reptiles (Kanowski et al. 2006), appeared to be associated with complex microhabitats. Cunningham et al. (2007) found that reptiles were less abundant on farms with many revegetation plantings than on farms with little revegetation. Reptiles were, however, correlated with the amount of remnant vegetation cover on a farm (Cunningham et al. 2007).

2.5.6 Amphibians

Amphibians exhibited a mixed response to revegetation. Kavanagh et al. (2005) found that frogs were present in ponds with water regardless of vegetation type (remnant, revegetation or cleared farmland); Hobbs et al. (2003) found more frogs in remnants than in revegetation and cleared farmland, and no difference between the latter two. Frogs in western Victoria did not respond to planting width (Merritt and Wallis 2004).

2.5.7 Invertebrates

Four studies on invertebrates found more taxa in remnant vegetation than in simple tree plantings (Green and Catterall 1998, Bonham et al. 2002, Schnell et al. 2003, Cunningham et al. 2005). However, the studies found different responses of invertebrates to revegetation compared with cleared farmland. One found more ant species in six year old simple tree plantings than on cleared farmland (Schnell et al. 2003), whereas another study found no difference (Green and Catterall 1998). The latter study, plus another (Catterall et al. 2004) found highly variable responses by different invertebrate orders. Catterall et al. (2004) found that Orthoptera (grasshoppers) were much more abundant in cleared farmland than revegetation or remnants; Coleoptera (beetles) and Formicidae (ants) were reasonably abundant in all vegetation types (cleared farmland, revegetation, remnants); Amphipoda (litter hoppers) were abundant only in vegetation of high floristic diversity (remnant forest, regenerating forest and floristically-rich ecological restoration plantings), with very low numbers in cleared farmland and monoculture revegetation. Cunningham et al. (2005) found the species richness of Coleoptera (beetles), Lepidoptera (moths) and Hymenoptera (ants, bees and wasps) did not differ between simple tree plantings, remnant vegetation and cleared farmland, but the community composition differed between site types for Coleoptera and Lepidoptera. They also found no differences in community composition of these insect groups between edge and interior habitats, or between isolated plantings and those adjacent to remnant vegetation. Bonham et al. (2002) found no difference in the number of native species of invertebrate with age of revegetation.

Majer and Nichols (1998) found that the composition of ants in an ecological restoration planting of a mined site approached that in a remnant forest sooner than that in a simple tree planting. Ant richness increased in both revegetation plots over a 14 year period, and the composition approached that of remnant forests in both revegetation types (Majer and Nichols 1998).

Table 2-1 Studies of faunal response to revegetation in agricultural landscapes.
A tick indicates the vegetation types researched in each study/article.

Authors	No. of sites	Taxa	Paddock	Plantation	Regrowth	Ecological planting	Remnant	Vegetation type	Climatic zone	Comments
Woinarski (1979)	2	Birds		✓			✓	Forest	Temperate	Unreplicated, observational
Biddiscombe (1985)	3	Birds		✓				Woodland	Temperate	A longitudinal study for 7 years, descriptive
Crome *et al.* (1994)	64	Birds, arboreal marsupials, terrestrial mammals		✓	✓		✓	Rainforest	Tropical	Sites were on a single farm, poorly replicated
Leary (1995)	15	Birds	✓	✓			✓	Woodland	Temperate	Honours thesis
Green and Catterall (1998)	40	Birds, arboreal marsupials, reptiles, amphibians, invertebrates	✓		✓		✓	Forest	Subtropical	Site types were clustered
Harris (1999)	25	Birds		✓	✓	✓	✓	Woodland	Temperate	Honours thesis
Kinross (2000)	84	Birds	✓	✓			✓	Woodland	Temperate	PhD thesis
Fisher (2001)	6	Birds			✓			Woodland	Temperate	Descriptive
Taws *et al.* (2001)	132	Birds	✓			✓	✓	Woodland	Temperate	
Bonham *et al.* (2002)	92	Invertebrates		✓			✓	Forest	Temperate	Both pine and Eucalypt plantations
Borsboom *et al.* (2002)	18	Birds, terrestrial mammals, bats, reptiles, amphibians	✓	✓	✓			Forest	Subtropical	
Grabham *et al.* (2002)	5	Birds		✓				Woodland	Temperate	
Klomp and Grabham (2002)	12	Birds	✓	✓			✓	Woodland	Temperate	Only 3 replicates in study
Arnold (2003)	27	Birds		✓			✓	Woodland	Temperate	
Hobbs *et al.* (2003)	28	Birds, small and large terrestrial mammals, bats, reptiles, amphibians	✓	✓			✓	Forest	Temperate	

Authors	No. of sites	Taxa	Paddock	Plantation	Regrowth	Ecological planting	Remnant	Vegetation type	Climatic zone	Comments
Rossi (2003)	54	Birds, terrestrial mammals	✓	✓			✓	Forest	Temperate	Masters thesis, both pine and eucalypt plantations
Schnell *et al.* (2003)	15	Invertebrates (ants)	✓	✓			✓	Woodland	Temperate	The 'remnant' vegetation is old regrowth
Martin *et al.* (2004)	12	Birds	✓	✓		✓	✓	Woodland	Temperate	Sites were not spatially independent
Bond (2004)	20	Birds				✓	✓	Woodland	Temperate	Honours thesis
Catterall *et al.* (2004)	104	Birds, reptiles, invertebrates	✓	✓	✓	✓	✓	Rainforest	Tropical and subtropical	
Kinross (2004)	84	Birds	✓	✓			✓	Woodland	Temperate	Paper resulting from thesis
Merritt and Wallis (2004)	10	Birds, amphibians				✓		Woodland	Temperate	
Cunningham *et al.* (2005)	27	Invertebrates	✓	✓			✓	Forest	Temperate	
Kanowski *et al.* (2005)				✓		✓	✓			Review restricted to rainforests, with some additional data
Kavanagh *et al.* (2005)	120	Birds, arboreal marsupials, bats, reptiles, amphibians	✓	✓		✓	✓	Woodland	Temperate	
Kanowski *et al.* (2006)	104	Reptiles	✓	✓	✓	✓	✓	Rainforest	Tropical and subtropical	
Law and Chidel (2006)	120	Bats	✓			✓	✓	Woodland	Temperate	
Cunningham *et al.* (2007)	184	Arboreal marsupials, reptiles			✓	✓	✓	Woodland	Temperate	
Loyn *et al.* (2007)	105	Birds	✓	✓			✓	Forest	Temperate	
Total articles = 29			16	21	8	11	23			
Total studies = 27			15	19	8	11	22			

2.6 Revegetation attributes affecting fauna use

2.6.1 Patch size

In fragmented landscapes, patch size of remnants tends to have a positive effect on birds (Loyn 1987, Lindenmayer et al. 2002, Seddon et al. 2003), arboreal marsupials (Pahl et al. 1988) and reptiles (Mac Nally and Brown 2001). The effect of patch size has been poorly researched in revegetation studies. Larger revegetation patches may benefit some faunal groups such as birds and bats (see sections above), whereas the effect of patch size on other faunal groups is largely unknown.

2.6.2 Width of revegetation

Bird species richness is generally higher in relatively wide plantings (see above), whereas frogs, bats, arboreal marsupials and reptiles appear to show no consistent response to revegetation width.

2.6.3 Age of revegetation

Birds and arboreal marsupials appear to increase in richness and abundance with increased revegetation age, but bats, reptiles and invertebrates do not. We found no studies with data on the response of small terrestrial mammals and amphibians to revegetation age. Most revegetation plantings examined in this review were young (mostly <30 years). Some key resources such as large logs, dead trees, tree hollows, or ground cover complexity may take longer than this to develop (McElhinny et al. 2006), whereas others may be independent of revegetation age (e.g. water availability, rocks).

Faunal composition also may change in revegetation over time. Young revegetated mine sites in southwest WA contained competitive colonising species or generalist species of mammal, bird and ant; then as the vegetation matured, a new suite of species took advantage of the changes in structure at the site (Majer and Nichols 1998, Nichols and Nichols 2003). In Queensland, bird guilds in simple tree plantings became more like those in selectively-logged forest over time (Borsboom et al. 2002).

Table 2-2 The number of studies found of particular faunal groups.
The groupings used in this graph are those referred to throughout the review. Several studies researched more than one faunal group (see Table 2.1).

Faunal Group	Number of studies
Birds	22
Arboreal marsupials	4
Terrestrial marsupials	4
Bats	4
Reptiles	6
Amphibians	5
Invertebrates	4

2.6.4 Structural complexity and floristic diversity

Structurally complex revegetation typically supports more fauna species and a different faunal composition than structurally simpler revegetation. Some attributes of complexity are particularly important to some faunal groups. For example, amphibians and reptiles respond predominantly to complexity in the ground layer, and small terrestrial mammals respond to complexity in the mid- and understorey layer (McElhinny et al. 2006). Similarly, the presence of old trees in a eucalypt plantation can significantly increase bird diversity and abundance (Grabham et al. 2002)

Vegetation that is floristically diverse may contain more fauna species than monocultures, even if vegetation structure is similar (Barrett 2000, Kanowski et al. 2005). Plantings established for ecological restoration generally exhibit greater floristic and structural diversity than simple tree plantings, and typically support higher faunal diversity (Catterall et al. 2004, Kanowski et al. 2005, Kavanagh et al. 2005).

2.6.5 Adjacency to remnant vegetation

Adjacency to remnant vegetation can increase the use of revegetation by birds (Hobbs 2003, Hobbs et al. 2003). Less mobile species such as mammals are less likely to inhabit planted vegetation than highly mobile animals such as birds (Hobbs 2003, Hobbs et al. 2003). White et al. (2004) found that plantings close to remnants had higher numbers of rainforest plants dispersed by birds, small mammals and wind, than distant sites, indicating that adjacency may benefit plants as well as animals.

2.6.6 Vegetation cover in the landscape

The amount of overstorey vegetation cover in the landscape has been identified as a key variable determining the presence of birds at revegetated sites (Barrett 2000, Kavanagh et al. 2005). Birds, arboreal marsupials and reptiles are also more likely to inhabit revegetation when remnant cover is high (Kavanagh et al. 2005, Cunningham et al. 2007, Cunningham et al. in press).

2.7 Comparisons with mine site rehabilitation

Revegetated mine sites provide an interesting parallel to revegetated areas in agricultural landscapes. However, the contextual position of revegetated mine sites, which are usually surrounded by remnant vegetation, is very different to revegetation in agricultural areas where issues of isolation and vegetation cover occur. Revegetated mine sites can therefore provide important information on the faunal use in the absence of isolation, landscape cover or gap-crossing issues.

Revegetated mine sites show successional trends in bird species, beginning with generalist taxa (Nichols and Nichols 2003). Recolonisation of revegetated mine sites appears to be rapid: birds may recolonise within six years (Nichols and Watkins 1984); reptile species richness may resemble that of low quality remnant vegetation after four to six years (Nichols and Bamford 1985); many invertebrate orders had similar species richness to surrounding unmined forest within seven years (Nichols et al. 1989); native small mammals recolonised sand mined forests within eight years (Fox and Fox 1984); and many birds were breeding in revegetated sites within ten years (Curry and Nichols 1986). Birds that did not breed in the revegetated sites had requirements for features not yet available in the sites, such as tree hollows (Curry and Nichols 1986). The presence of lizards in rehabilitated sand mining sites was predominantly explained by vegetation complexity (Twigg and Fox 1991). Bauxite mine sites in Western Australia have seen an evolving rehabilitation method (Collins et al. 1985, Armstrong and Nichols 2000). Older rehabilitation sites contained very little understorey vegetation, while more recent sites contained an understorey plant species richness and diversity comparable to unmined forests (Collins et al. 1985). The older sites contained very low bird species richness and densities, whereas the recent sites with understorey support bird species richness and densities similar to those in unmined forest (Nichols and Watkins 1984, Collins et al. 1985, Armstrong and Nichols 2000). Bird species composition was similar in the recently rehabilitated areas to that of forests (Collins et al. 1985). Similarly, ant species richness and composition was positively associated with plant species richness and diversity and age of the planting (Majer et al. 1984, Majer and Nichols 1998). These studies have emphasised the benefits of developing an understorey in the plantings (where an understorey originally occurred) (Armstrong and Nichols 2000).

2.8 Landscape-scale role of revegetation

Lindenmayer et al. (2002) suggested that remnant vegetation fragments of all sizes and shapes have significant conservation value, both as habitat and as stepping stones through the landscape (Fischer and Lindenmayer 2002). This notion may extend to revegetation, despite the lower faunal use compared to remnants. Revegetation may also help buffer adjacent remnants from climatic extremes and other degrading processes, and may stabilize key ecological processes in agricultural landscapes (e.g. by reducing water tables) (Hobbs 1993, Bennett et al. 2000, Kanowski et al. 2005). At a landscape scale, there may be negative consequences for fauna if remnant vegetation is replaced with revegetation (Cunningham et al. 2007), and positive consequences if revegetation is situated on already cleared farmland.

2.9 Progress to date

Much new research has been completed since previous reviews on revegetation in agricultural landscapes in Australia (Kimber et al. 1999, Ryan 1999) (Table 2.1). However, many knowledge gaps remain. Much research has focussed on the value of revegetation for birds, but there is a paucity of information on other faunal groups and on threatened and declining taxa. Most research has focussed on simple measures of species richness and abundance but faunal composition would provide valuable information on the benefits of revegetation to fauna.

Establishment of ecological restoration plantings is a relatively new practice. It is logical to study both ecological restoration plantings (as an example of the best revegetation currently conducted) and simple tree plantings (as the most common form of revegetation). Differences between these forms of revegetation can provide insights into the conservation capacity of revegetation under both a best-case scenario and the current scenario of mostly simple tree plantings.

The value of revegetation to fauna is rarely put into a landscape context. This context is important because patch-scale research provides information on the local faunal richness (alpha diversity), but it is the landscape faunal richness (beta diversity) that is often of greatest conservation concern (Tscharntke et al. 2005).

Most studies have not examined underlying processes involved in faunal use of revegetation. We found only one study which explored this issue - use of revegetation by birds for breeding (Bond 2004). To date no research has been conducted on processes such as competition or predation in revegetation.

The faunal response to revegetation studied to date is mostly short-term because revegetation has become common only in recent decades. As revegetation ages, and incorporates more features such as logs and leaf litter, its value to wildlife may increase. Ongoing studies will be required to assess the long-term benefits of revegetation.

2.10 Recommendations

Many research projects are written as reports or unpublished theses that are not widely available. To maximise accessibility of findings to other researchers we advocate publication in peer-reviewed journals. There is also a need for scientists to more clearly explain site attributes of revegetation – in particular age, size, isolation, and structural complexity and floristic diversity. Much of this basic information was unavailable in the reviewed articles. Clear and consistent information can provide future opportunities for systematic reviews or meta-analyses.

We suggest further research should target the following areas:

- long-term trends and successional changes in revegetation including the development of key structural features and their effect on fauna;
- comparisons of different types of revegetation including analyses of potential trade-offs between quantity and quality of revegetation at the landscape scale;
- the value of planting indigenous plant species for fauna;
- the faunal composition changes in revegetation over time and with different site attributes;

- the response by terrestrial mammals to revegetation;
- the resource needs of reptiles, amphibians and bats which could be provided by revegetation;
- the conservation value of revegetation for declining or threatened fauna;
- the value to wildlife of revegetation in riparian compared to non-riparian areas; and
- the interaction of remnant vegetation and revegetation.

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Chapter 3: Revegetation in agricultural areas: the development of structural complexity and floristic diversity

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Revegetation in agricultural areas: the development of structural complexity and floristic diversity

3.1 Abstract

Revegetation plantings have been established to ameliorate the negative effects of clearing remnant vegetation and to provide new habitat for fauna. We assessed the vegetation development of revegetation established on agricultural land in Gippsland, south-eastern Australia. We compared: (1) woodlot plantings (overstorey eucalypts only), and (2) ecological plantings (many species of local trees, shrubs and understorey), with remnants and paddocks for development of vegetation structural complexity and colonising plant species. We also assessed structural complexity and plant species composition in response to several site parameters. Structural complexity increased with age of planting, toward that of remnants, even when very few species were planted at establishment. Richness of all plants and native plants, however, did not increase with age. Native ground cover plants were not included at establishment in either planting type, and their richness also did not increase with age of planting. This indicated that colonization did not occur through time, which does not support the 'foster ecosystem hypothesis'. Weed species richness was unrelated to native plant richness, which does not support the 'diversity-resistance hypothesis'. Weed cover increased with age of planting in woodlot plantings, but decreased with age in ecological plantings. Richness of all plants and native plants in plantings did not increase with planting size or with the presence of old remnant trees, and was greater in gullies and where vegetation cover in the landscape was greater. Structural complexity was unaffected by planting size, but was positively correlated with floristic richness. Ecological plantings had higher condition scores, greater shrub cover, more plant lifeforms and fewer weeds than woodlot plantings indicating a possible greater benefit as habitat for wildlife. We conclude that ecological plantings can achieve similar overall structural complexity as remnant vegetation within 30 to 40 years, but will not gain a native ground layer, and will not necessarily contain some important structural features by this age. Ecological plantings may provide habitat for the conservation of fauna (through the development of structural complexity) but they may not provide for the conservation of non-planted flora (given the absence of re-colonising smaller lifeforms).

3.2 Keywords

Foster Ecosystem Hypothesis, Diversity-resistance Hypothesis, revegetation, vegetation structural complexity, seral stages, restoration, floristic richness

3.3 Introduction

Clearing of native vegetation can cause serious land degradation problems such as salinity (Commission 1999, Stirzaker et al. 2002) and soil erosion (Bird et al. 1992). It can also lead to a loss of native wildlife (Saunders et al. 1991, Yates and Hobbs 1997, Ford et al. 2001). Revegetation has been conducted in many agricultural landscapes with the aim of reducing or reversing these problems (Greenwood et al. 1992, Hobbs and Saunders 1993, Department of Natural Resources and Environment 2002, Barrett et al. 2008) and has recently been incorporated into government policies for biodiversity conservation, such as offsetting clearing and subsidies for vegetation enhancement. However, in the first few decades after establishment, revegetation is unlikely to perform all the functional, structural and compositional roles of remnant vegetation (Hobbs 1993, Munro et al. 2007).

Although revegetation has been hailed as a potential solution to biodiversity loss (Hobbs 1993), previous reviews suggest that revegetation is not suitable as habitat for many wildlife species (Kimber et al. 1999, Cunningham et al. 2007, Munro et al. 2007, Cunningham et al. 2008). The potentially limited habitat value of revegetation has been attributed, in part, to low structural and floristic complexity of revegetation (Nichols and Watkins 1984, Arnold 2003, Hobbs et al. 2003), because both structure and floristic composition are important for a wide range of fauna (Barrett 2000, Borsboom et al. 2002, McElhinny et al. 2006). For example, complex vegetation structure provides resources for nesting, perching and shelter (Recher 2004, McElhinny et al. 2006), microhabitats for seedling establishment (Parrotta et al. 1997a, White et al. 2004) and physical structures influencing radiation and the flow of wind and water through a system (Bird et al. 1992, Vought et al. 1995). Floristic diversity may provide diverse food and shelter resources for fauna (Arnold 1988, Mac Nally 1990).

The 'foster ecosystem hypothesis' has been proposed (Haggar et al. 1997) whereby understorey plant species will colonise plantings, potentially stimulating native forest regeneration (Silva Júnior et al. 1995, Loumeto and Huttel 1997, Lugo 1997, Parrotta et al. 1997b, White et al. 2004). Planted trees are assumed to provide a 'nurse-tree' function, whereby the development of overstorey structure will provide the abiotic environment (leaf litter, moisture, shade) for the establishment of understorey species, provided propagules can migrate there (Lugo 1997, Parrotta et al. 1997b, Harvey 2000, White et al. 2004). In addition, it has been suggested that the overstorey and midstorey structure could also provide perches and habitat for fauna, which can carry propagules to the site (e.g. in faeces or fur) (Parrotta et al. 1997b). Very few studies have been conducted to test the 'foster ecosystem' effect in restoration plantings (Wilkins et al. 2003).

The diversity-resistance hypothesis (Elton 1958) suggests that high native plant diversity will resist invasion of exotic plant species. However, many studies have found that weed species can be high when native richness is high, due to, among other things, a consistent response by both exotic and native plants to environmental heterogeneity or resource richness (Lonsdale 1999, Fridley et al. 2007). We could find no studies on weed cover or richness in revegetation plantings, despite its implications for management and possibly for fauna recolonisation.

Depending in part on the applicability of the foster ecosystem hypothesis and the diversity-resistance hypothesis, structural complexity and floristic diversity in revegetation may ultimately resemble that of nearby remnants or may form different but complementary vegetation components in the landscape (Barrett et al. 2008, Cunningham et al. 2008). The development of structural complexity and floristics may also influence faunal recolonisation of a planting (Munro et al. 2007). However, there is currently an absence of information on how revegetation structure and floristic diversity develops, particularly under different vegetation establishment techniques. This information is important for understanding the biodiversity value of revegetation and for identifying establishment techniques which will maximise the benefits of revegetation for fauna.

Here we outline the development of the structural complexity and floristics of revegetated areas within an agricultural region of south-eastern Australia. We compare revegetation sites differing in floristic diversity at establishment. We distinguish between ecological plantings and woodlot plantings where (1) ecological plantings were established for ecosystem restoration purposes, and (2) woodlot plantings were those established for other purposes (e.g. shelterbelts, woodlots, farm forestry), and were typically planted with only overstorey species. At establishment, the plant species richness of 'ecological plantings' was usually higher than that of 'woodlot plantings', and the diversity of lifeforms also was typically greater (e.g. shrubs and other smaller lifeforms were included). We hypothesised that (1) structural complexity and floristic richness would increase with age of the planting in both types of plantings (a test of the foster ecosystem hypothesis); (2) structural complexity and floristic diversity of both kinds of plantings would, over time, become similar to that of remnant vegetation, but at different rates and to different extents (the trajectory of plantings); (3) plant composition in ecological plantings would be more like that in remnants, than woodlot plantings; (4) ecological plantings would have fewer weed species than woodlot plantings (a test of the diversity-resistance hypothesis); and (5) ecological plantings would have a higher condition score, and be more similar to that in remnants than woodlot plantings.

3.4 Methods

3.4.1 Study area

We conducted our study in West Gippsland, Victoria, in south-eastern Australia. The region is temperate with fertile deep soils and relatively high rainfall (approximately 1000 mm per year). Approximately half the study area was in the Strzelecki Ranges, and half on the Gippsland Plain (Fig. 3.1). The area was heavily cleared for agriculture in the late 19^{th} century, with approximately 12 % percent woody vegetation, and only 6 % of non-coastal woody vegetation remaining (DSE 2003). Prior to clearing, the vegetation consisted of lowland forest on the ridges, damp and wet forest in the gullies, and shrubby forest in the foothills. The forest vegetation was structurally complex throughout the region with a dense and tangled understorey (Korumburra and District Historical Society Inc. 1998). Most of the remaining vegetation communities are classified as vulnerable or endangered (DSE 2003).

The development of slope and tunnel erosion in recent decades prompted many landholders to revegetate areas of cleared land. Some revegetation in our study area was co-ordinated by experienced natural resource managers (Bass Coast Landcare, pers. comm.). We termed these revegetation works '*ecological plantings*'. They were characterised by a planted species richness of approximately 20 tree, shrub and understorey species each. Ground cover forbs and grasses were seldom planted. '*Woodlot plantings*' also were common in the study area but these were typically planted with low plant species richness (one to five species), of primarily overstorey species only.

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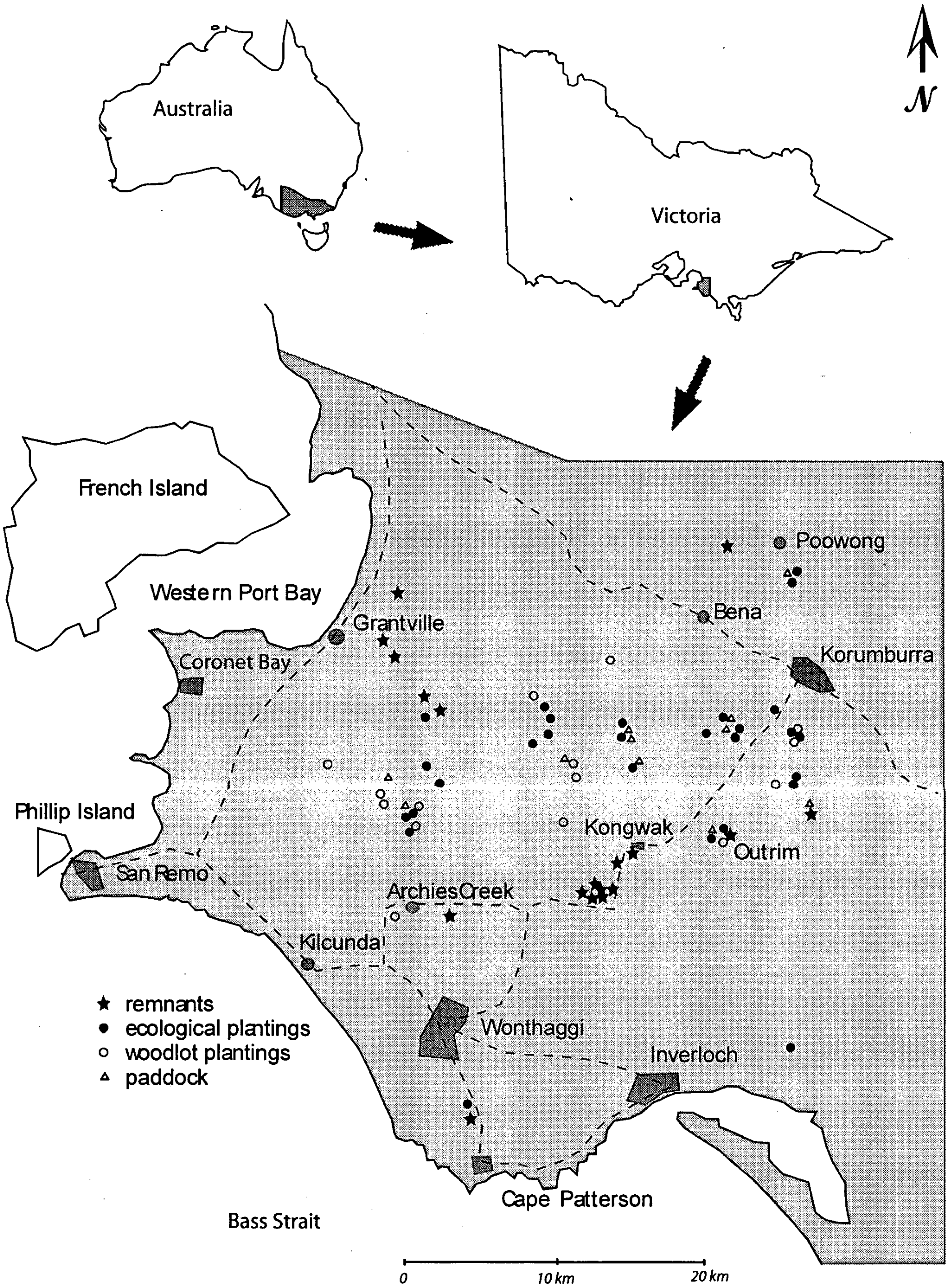


Figure 3-1 Map of the study area.
Sites are marked by a symbol (see the key), major roads are marked by dotted lines and towns are shown by darker patches with town names.

3.4.2 Site and patch selection

We selected sites within four vegetation types: ecological plantings (26 replicates), woodlot plantings (16), remnants (18) and paddocks (cleared agricultural land; 11) (Figs. 3.1, 3.2). Half the sites of each type were in riparian locations (the banks of a watercourse), and half were not; approximately half the plantings and paddocks contained old remnant trees and half did not. Patches were of eight different Ecological Vegetation Classes (EVC) (Department of Sustainability and Environment 2003b) – Lowland Forest (24 sites), Damp Forest (14), Wet Forest (13), Shrubby Foothill Forest (10), Swamp Scrub (7), Swampy Riparian Woodland (2), Riparian Forest (1) and Damp Sands Herb-rich Woodland (1). Site types were spread among the different EVCs. We did not distinguish between different EVCs in our analyses. Remnant sites (which were used as reference sites) were degraded to different extents. Almost all had been selectively logged in the 19th century and possibly grazed in the past, and although some were relatively pristine, others were infested by weeds. Despite these imperfections, the remnants we chose constituted the most pristine remnants available.

We selected patches by using maps and aerial photographs, on-ground searching and by consulting a database of local Landcare projects. We selected one site (100 x 40 m in area) within each patch, to be representative of the patch, and it was located centrally in the patch. We conducted all vegetation assessments at the site level. We obtained information about patch size and vegetation cover in the landscape from a Geographical Information System (ArcMap v.9.1, ESRI 2005).

Remnants and both planting types were of similar size range and mean with an overall mean of 5.9 ha and range of 0.07 to 27.5 ha. We also included two large remnants of 528 and 43 ha and one large woodlot planting (96.5 ha), to examine the effect of large area on structure and floristics. The age of plantings ranged from 2 to 26 years for both woodlot and ecological plantings. We made an attempt to balance the ages between these vegetation types. However, on average, the woodlot plantings were older (mean of woodlot plantings was 13.6 years; mean of ecological plantings was 8.1 years, P=0.006). This was due to a recent trend toward establishing predominantly ecological plantings for non-commercial purposes.

Figure 3-2 Photographs of site types.
a) paddock, b) woodlot planting, c) ecological planting, d) remnant.

Woodlot plantings were originally planted with a mean of 4.7 species (range 1 to 15), and ecological plantings were planted with a mean of 20 species (range 8-39). The extra species included in the ecological plantings were primarily shrubs and large tussocks, with virtually no grasses or forbs.

3.4.3 Vegetation assessment

In spring of 2005 and 2006, we collected data on floristics and vegetation structure of each site (Appendix 3.1 and 3.2). We identified all plant species at each site and each species was assigned a 'lifeform' as per the Victorian Vegetation Quality Assessment, such as epiphyte, small shrub, large tufted graminoid, etc. (Department of Sustainability and Environment 2004). We noted species while conducting a 2 hr structural assessment (see below), plus each site was searched for an additional half hour for plant species. We visually estimated a percent cover of vegetation for each species at each site, plus a total cover for each of four broad strata: overstorey, midstorey, understorey and ground cover. We defined each species as native, exotic or weed, where a weed was defined as an environmental or noxious weed listed in at least one relevant weed identification guide. We defined a native species as one that occurred in the study area before European settlement, with inclusion in local native plant identification books.

We calculated structural complexity in two ways: (1) Principal Components Analysis (PCA) of site structural variables, and (2) a Habitat Complexity Score (Catling and Burt 1995). We also calculated a measure of vegetation condition using the Habitat Hectares Condition Score (Parkes et al. 2003).

First, we conducted a Principal Components Analysis (PCA) on selected variables to derive a data-driven quantitative measure of structural complexity (methods described in detail below).

Second, we calculated a Habitat Complexity Score (HCS4). This was a modified version of that used by Tasker and Bradstock (2006) and Coops and Catling (1997). We allocated a score of 0 to 4 to eight pre-determined vegetation layers, according to the percent cover of that layer. Scores were as follows:

Score:	0	1	2	3	4
% cover:	<1%	1-5%	6-30%	31-70%	>70%

The eight vegetation layers were: tree canopy, small trees (>2 m), tall shrubs (>2 m), medium shrubs (1-2 m), small shrubs (<1 m), ground cover, logs and leaf litter. Understorey was emphasised in this score as per Tasker and Bradstock (2006). We included measurements of log and leaf litter cover as per Coops and Catling (1997), Freudenberger (2001) and Oliver and Parker (2006), since these features were deemed to be an important structural element in undisturbed remnant forest.

Finally, we calculated the Habitat Hectares Condition Score, developed in Victoria, Australia by Parkes et al. (2003), for each site. This score assesses the cover and quality of seven site components against a benchmark for that vegetation type. We used only the condition score in the current analyses (i.e. the landscape component, worth 25 of 100 points was omitted). For each site we determined the Ecological Vegetation Class (EVC) from current EVC distribution maps (Department of Natural Resources and Environment 2002). For planted sites, we determined the EVC from maps of pre-1770 EVC distributions which were developed by the state of Victoria (Department of Natural Resources and Environment 2002). Each site was compared to its relevant EVC benchmark. The seven components in the assessment were: number and health of large trees, canopy cover and health, understorey composition, cover of weeds, recruitment of perennial plants, organic litter cover, and log cover. We compared the cover or abundance of each of these components to the relevant benchmark, and scores were allocated accordingly. We summed these scores to give the Habitat Hectares Condition Score. A high score indicated a strong similarity in the quantified features to pristine reference sites of the same EVC (from which the benchmarks were derived). See Parkes *et al.* 2004 for details.

We calculated the amount of native vegetation cover in a circle of radius 2.25 km (area of 15.9 km^2) for each site using GIS, vegetation cover data and aerial images. Our calculations of vegetation cover included remnant vegetation of dense to medium density (i.e. scattered trees were excluded) plus revegetation. We excluded exotic Cypress (*Cupressus* spp.) and pine (*Pinus* spp.) patches and house gardens.

3.4.4 Analyses

To visualize floristic differences between site types, we used non-metric multidimensional scaling (NMDS) using the Sorensen dissimilarity measure (Legendre and Legendre 1998). The presence/absence of all (359) plant species was used for this analysis. We included vectors from the (0,0) point of the NMDS plot to the centroid of each lifeform, to indicate which lifeforms contributed to the plant composition of site types.

To determine structural differences between site types, we performed a PCA on twenty selected structural variables (Appendix 3.1). We used hierarchical cluster analysis (Legendre and Legendre 1998) to assist with variable reduction, by highlighting correlated and composite variables. We selected variables so they equally represented overstorey, midstorey and understorey strata. We standardized all variables before conducting the PCA analysis. The results of the initial PCA were strongly influenced by the presence of paddock sites (not shown). Hence, we repeated the PCA without paddock sites.

We used multiple linear regression to identify key patch attributes related to structure, condition and floristic richness. We used an iterative, manual, forwards and backwards variable selection process (using the computer package 'R'). Our response variables included floristic data (total plant richness, native plant richness, native groundcover richness, weed cover and number of lifeforms), structural data (the first two components of the PCA and the Habitat

Complexity Score (HCS4)) and condition data (the Habitat Hectares Condition Score). Linear regression modeling was conducted for these structural, condition and floristic richness data, as a function of selected site variables (the explanatory variables): age of planting, size of patch, riparian/non-riparian location, topographic position of patch, the presence of old trees, surrounding vegetation cover, and patch type. For all models we assigned the age of remnants and paddocks to the mean age of both types of plantings to remove problems associated with a null score. For example, paddocks and remnants do not have an age since planted, and an arbitrary age cannot be allocated because this would unduly influence the analysis. Similarly for patch size, we assigned paddocks the mean area of the remaining three site types. Age and area were natural log-transformed prior to analysis so they were approximately normally distributed.

To explore the strength of the relationship between vegetation structure and floristics, we modeled the HCS4 score and plant richness at a site using linear regression. The relationship between structure and condition were analyzed the same way.

3.5 Results

3.5.1 Site descriptions

We found that site types differed significantly in a number of variables (at significance of $P<0.05$). Notably, remnants had significantly more strata, higher projected foliage cover of shrubs, higher understorey cover, more large trees, dead trees, logs, leaf litter, tussocks, and much lower weed cover than plantings or paddocks. Ecological plantings differed from woodlot plantings by having a higher projected foliage cover of shrubs and midstorey cover, lower overstorey cover and ground cover, fewer large trees and dead trees, and a lower number of trees per hectare. Ecological plantings and remnants had the highest total number of species (mean of 39.2 for ecological plantings, and 34.7 for remnants, cf. 24.4 for woodlot plantings) and the highest number of native species (mean of 24.9 for ecological plantings, and 27.6 for remnants, cf. 10.2 for woodlot plantings). Remnants had the highest number of native ground cover species (mean of 8.9).

Table 3-1 Results of linear regression models of weed cover and number of lifeforms as a function of patch attributes.

Response	Patch variables in model	Parameter Estimates	SE	P-value
Number of lifeforms	Intercept (incl. ecological plantings)	13.862	0.731	<0.001 ***
	Type – paddock	-4.547	0.673	<0.001 ***
	Type – remnant	0.483	0.561	0.392
	Type – woodlot plantings	-1.196	0.489	0.017 *
	PFC	0.026	0.012	0.035 *
	Old trees (absent)	-1.176	0.483	0.018 *
Weed cover	Intercept (incl. ecological plantings)	104.164	18.551	<0.001 ***
	Type – paddock	44.072	7.595	<0.001 ***
	Type – remnant	-20.387	7.865	0.012 *
	Type – woodlot plantings	-50.213	24.441	0.044 **
	Old trees (absent) [a]	8.569	6.704	0.206
	Riparian	13.236	5.321	0.015 *
	Log (age)	-28.050	8.142	0.001 **
	Type (woodlot planting) by log (age)	35.753	10.903	0.002 **

a. old trees were included in this model because of the influence of remnants, which have old trees and which have low weed cover

3.5.2 Floristic composition

We identified a total of 359 plant species, of which 80 were weeds. The majority of overstorey, midstorey and understorey plants in ecological plantings were native (86%) and were planted (82%). The remainder were primarily colonised weed species, as well as a few remnant species such as old trees which occurred at a site prior to planting. In woodlot plantings, 60% of overstorey, midstorey and understorey plants were native, with several species planted that were non-native. Only 40 % of the larger (excluding ground cover) plant species were planted.

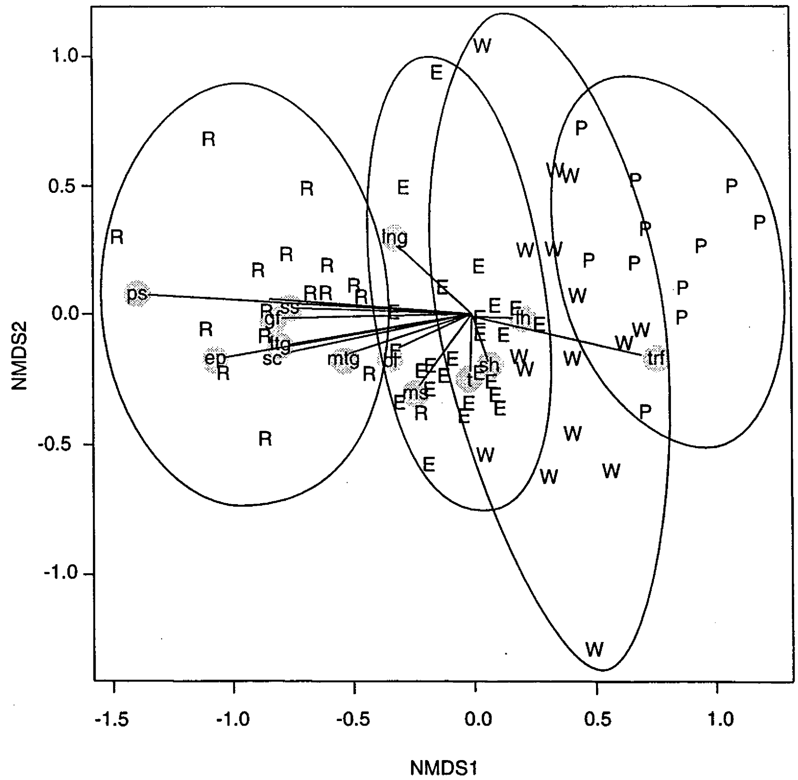


Figure 3-3 Plot of NMDS of presence/absence of all plant species found, showing only sites and lifeforms.
The centroid of each lifeform is located at the end of each vector, and is located within a grey circle. Lifeform codes are as follows: bl – bryophyte, ep – epiphyte, gf – ground fern, lh – large herb, lng – large non-tufted graminoid, ltg – large tufted graminoid, ms – medium shrub, mtg – medium tufted graminoid, ot- overstorey tree, ps – prostrate shrub, sc – scrambler/climber, sh – small herb, ss – small shrub, t – subcanopy tree, trf - treefern. Sites are plotted and are indicated by the letter R - remnants, E - ecological planting, W - woodlot planting and P - paddock. Large circles surround clusters of sites of the same type. Site types differ in species composition (R=0.6815, stress of 18.8%, P<0.001) and in diversity of lifeforms (P<0.001).

Site types differed in total species composition (P<0.001, Fig. 3.3) and native species composition (P<0.001, not shown). Remnants were characterised by particular lifeforms: prostrate shrubs, epiphytes, creepers, small herbs, tufted graminoids and ground ferns (bracken, *Pteridium esculentum*). Ecological plantings were characterised by medium-sized shrubs and sub-canopy trees, woodlot plantings were characterised by large herbs, and paddocks were characterised by tree ferns. Species composition of remnants was distinct from plantings and paddocks, whereas that of ecological plantings and woodlot plantings showed considerable overlap (Fig. 3.3). We also found overlap between species composition of woodlot plantings and paddocks, but none between ecological plantings and paddocks. The results of an NMDS of native species composition (not shown) did not differ substantially from that of all vegetation species. Vegetation composition did not differ between riparian and non-riparian sites (P=0.125, not shown).

We found significant differences in the number of lifeforms between site types ($P<0.001$, Table 3.1) with most in remnants, and significantly more in ecological plantings than woodlot plantings. The number of lifeforms at a site was related to the projected foliage cover of the site ($P=0.035$) and the presence of old remnant trees ($P= 0.018$; Table 3.1).

3.5.3 Floristic richness

The total number of species and the number of native species at a site did not differ significantly between ecological plantings and woodlot plantings (Table 3.2). There were more plant species in riparian than non-riparian locations, and the richness of native plants was highest in gullies. The majority of the higher plant species richness in riparian areas was due to the cover of weeds which was greater in riparian than non-riparian locations (Table 3.1). Total plant richness and native plant richness was higher where the vegetation cover in the landscape was higher (Table 3.2).

The number of native ground cover species did not differ between ecological plantings and woodlot plantings, and was only slightly higher than in paddocks (Table 3.2). Most native ground cover species were found in remnants. Richness of all plants, native plants and native ground cover plants was not related to age of the plantings, or the presence of old trees. Richness of all plants and native plants was not related to patch area. Native ground cover richness was weakly related to patch area, but not to vegetation cover in the landscape, or to the number of species planted initially (Table 3.2).

Weed species richness was unrelated to native plant richness ($P=0.2$). Weed cover was high in paddocks and both types of plantings, and low in remnants (Table 3.1). Many weeds were pasture grasses. Weed cover was related to the age of the plantings, but in different ways between the two types of plantings. As woodlot plantings increased in age, weed cover increased. However, as ecological plantings aged, weed cover decreased (Table 3.1, Fig. 3.4a). Weed cover was also greater in riparian than upland areas (Table 3.1).

Table 3-2 Results of linear regression models of floristic richness as a function of patch attributes.

Response	Patch variables in model	Parameter Estimates	SE	P-value
Total number of species	Intercept (incl. ecological plantings)	4.372	4.178	0.299
	Type – paddocks	-15.596	2.664	<0.001 ***
	Type – remnants	1.799	2.800	0.523
	Type – woodlot planting	2.138	3.044	0.485
	Number of species planted	1.249	0.152	<0.001 ***
	Landscape vegetation cover	0.402	0.108	<0.001 ***
	Riparian – riparian	4.090	1.631	0.015 *
	Weed cover	0.098	0.034	0.006 **
Number of native species	Intercept (incl. ecological plantings)	0.841	2.776	0.763
	Type – paddocks	-13.185	1.947	<0.001 ***
	Type – remnants	2.047	2.257	0.368
	Type – woodlot plantings	1.283	2.411	0.597
	Number of species planted	1.042	0.125	<0.001 ***
	Landscape vegetation cover	0.283	0.081	<0.001 ***
	Location – gully	3.615	1.661	0.034 *
	Location – ridge	-3.994	3.146	0.209
	Location – slope	1.574	1.633	0.339
	Log (area)	0.182	0.823	0.826
	Type (remnant) by log (area) [a]	2.258	1.017	0.030 *
	Type (woodlot planting) by log (area)	-0.770	1.046	0.465
Number of native ground cover species	Intercept (incl. ecological plantings)	3.491	0.609	<0.001 ***
	Type – paddocks	-2.009	1.062	0.063
	Type – remnants	3.320	0.940	<0.001 ***
	Type – woodlot plantings	-1.522	0.931	0.107
	Shape[b] – round	1.370	0.817	0.098
	Log (area)	0.613	0.248	0.016 *

a. although area by type was significant for remnants, further analysis indicated this was influenced by two very large remnants. Otherwise area was not considered a predictor of the number of native species.

b. Shape remained in this model because paddocks, which had a category of 'no shape' strongly influenced the results – there were more native ground cover species in patches that had a shape (remnants and plantings) than paddocks.

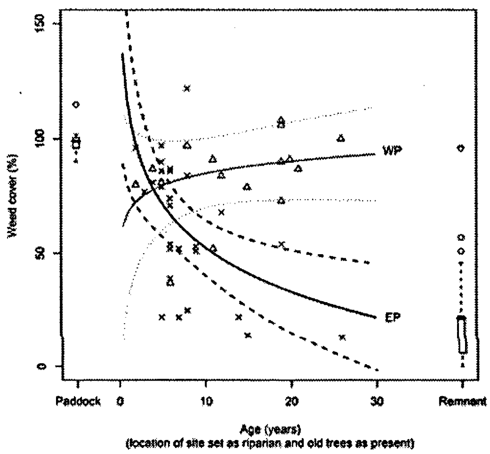
3.5.4 Structural complexity

We used two different methods to estimate the structural complexity of the vegetation, and both were broadly consistent. Age of planting significantly influenced both measures of structural complexity.

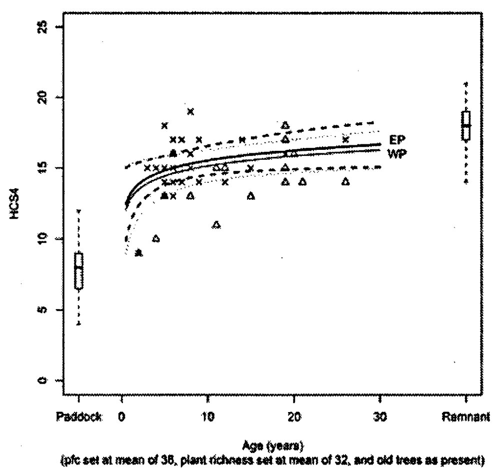
Results from the first two components of the PCA are displayed in Table 3.3. The PCA was repeated without paddocks to remove the influence of this very different vegetation type. However, this resulted in little change in the variables influencing the components.

Without paddock sites, the first principal component described a gradient from sites with complex vegetation and old trees to sites with a high level of ground cover (particularly forb cover; Table 3.3). This separated remnant sites from planted sites (Fig. 3.5). The second principal component described a gradient of sites from non-shrubby to shrubby, which separated the two planting types (Fig. 3.5).

a)



b)



c)

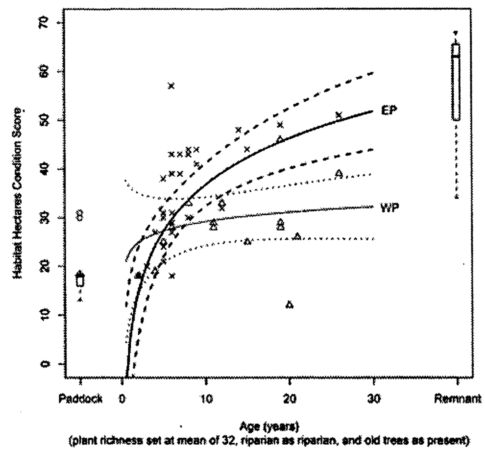


Figure 3-4 Site variables of weed cover, HCS4 and Habitat Hectares Condition Score by age of planting and site type.
Predicted relationships and their 95% confidence intervals are shown. Also displayed are actual data. Crosses and the bold and dashed lines are ecological plantings (EP), triangles and light and dotted lines are woodlot plantings (WP). The values for paddocks and remnants are indicated by boxplots. (a) Weed cover. It was possible to have greater than 100% weed cover, since cover was assessed in each of four main strata and summed. The 'location of the site' and 'presence of old trees' were significant variables in the model, and for the predicted relationship were set as 'riparian location' and 'old trees as present' respectively. (b) HCS4 score. The 'projected foliage cover (pfc)', 'plant richness' and 'presence of old trees' were significant variables in the model, and for the predicted data were set as indicated below. (c) The Habitat Hectares condition score. The 'plant richness', 'riparian location' of the site and 'presence of old trees' were significant variables in the model, and for the predicted data were set as indicated.

Table 3-3 Output of Principal Components Analysis without paddock sites, showing the first two components, the variance they explain and the most dominant positive and negative variables with their loadings.

Principal component	Variance explained (%)	Variables	Loading
1. Overstorey diversity vs. ground cover	35.8	Log (largest dbh trees)	-0.331
		Standard deviation of dbh	-0.316
		Number of strata	-0.315
		Forb cover	0.156
		Ground cover	0.094
2. Non-shrubby vs. shrubby	12.5	Ground cover	-0.388
		Largest height of trees	-0.238
		Log (midstorey cover)	0.436
		Projected Foliage Cover of shrubs	0.380
		Log (number of small shrubs per ha)	0.351

A linear regression model on the first component (PC1) (without paddocks) indicated PC1 was significantly related to type of patch, age, riparian/non-riparian location and the presence of old trees (Table 3.4). PC1 was not different between ecological plantings and woodlot plantings, but remnants contained more overstorey diversity and more large trees than either planting type. Overstorey diversity and size of largest trees increased with age of planting, and was higher in riparian compared to non-riparian locations. Logically, the presence of old remnant trees in a planting increased the size of largest trees.

The second component (PC2) was significantly related to type of patch. Ecological plantings were significantly more shrubby than woodlot plantings (as per the original selection criteria), and did not differ from remnants (Table 3.4, Fig. 3.5).

Using a linear regression model of the HCS4 score as a function of patch attributes, we identified patch type, age, plant species richness, projected foliage cover and the presence of old trees as significant variables influencing structure (Table 3.4). The HCS4 score did not differ between ecological plantings and woodlot plantings in the model, but it was significantly higher in remnants, and lower in paddocks. The HCS4 score increased rapidly with age of planting from that of paddocks, toward that of remnants (Fig. 3.4b), and was greater where old trees were present in a planting. Structural complexity was also strongly related to the number of plant species present (correlation co-efficient of the natural log of total species richness with HCS4 = 0.62, not shown).

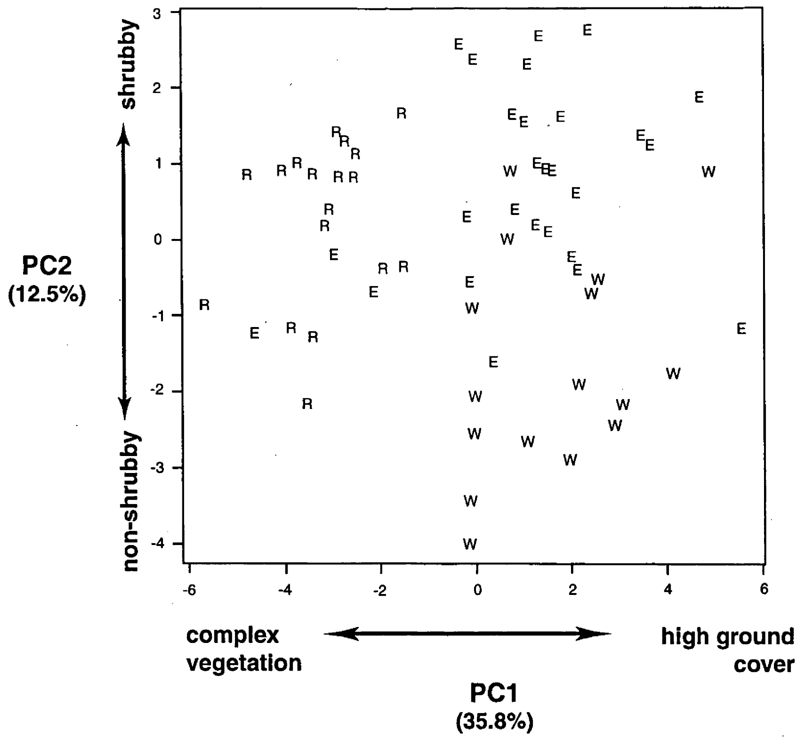


Figure 3-5 Graphical display of the results of the Principal Components Analysis of structure (paddock sites excluded).
The first two components are displayed in ordination space. The letters W, E and R refer to woodlot plantings, ecological plantings, and remnants respectively. The variables most influencing the components are indicated with the variability explained

3.5.5 Vegetation Condition

When we used a linear regression model of the Habitat Hectares Condition Score as a function of patch attributes, we found that patch type, age, plant species richness, riparian location and the presence of old trees were significant variables influencing condition (Table 3.4). The Habitat Hectares Condition Score was greater in ecological plantings than woodlot plantings and the Score increased with age in ecological plantings more than it increased with age in woodlot plantings (Fig. 3.4c). This score in ecological plantings increased with age towards that of remnants, but in woodlot plantings it increased more slowly, falling short of that in remnants (Fig. 3.4c). The Habitat Hectares Condition Score was greater where plant richness was greater, in riparian areas, and where old remnant trees were present. Condition was strongly correlated with structural complexity (correlation co-efficient of the natural log of the Habitat Hectares Condition Score with HCS4 = 0.78).

Table 3-4 Linear regression models
of the first two components of the Principal Components Analysis (without paddock sites)of structural elements, the HCS4 score and the Habitat Hectares Condition Score. "Type" has three levels in PC1 and PC2 (ecological planting, woodlot planting and remnant) and four levels for HCS4 and Habitat Hectares (including paddocks), "age" is log-transformed and is a continuous variable, "riparian/non-riparian" and "presence of old trees" are binary variables. PFC is projected foliage cover.

Response	Patch variables in model	Parameter Estimates	SE	P-value
PC1	Intercept (incl. ecological plantings)	5.080	0.643	<0.001 ***
	Type – remnants	-2.985	0.528	<0.001 ***
	Type – woodlot plantings	0.487	0.493	0.328
	Log (age)	-1.515	0.330	<0.001 ***
	Riparian/non-rip. location - riparian	-0.778	0.342	0.027 *
	Old trees - present	-1.893	0.545	0.001 **
	Type (woodlot plantings) by old trees	2.022	0.863	0.023 *
PC2	Intercept (incl. ecological plantings)	0.767	0.243	0.003 **
	Type – remnants	-0.470	0.385	0.228
	Type – woodlot plantings	-2.395	0.399	<0.001 ***
HCS4 – all sites	Intercept (incl. ecological plantings)	9.262	1.295	<0.001 ***
	Type – remnants	1.885	0.581	0.002 **
	Type – paddocks	-5.282	0.830	<0.001 ***
	Type – woodlot plantings	-0.384	0.624	0.540
	Log (age)	1.029	0.443	0.023 *
	Plant species richness	0.083	0.020	<0.001 ***
	PFC	0.035	0.013	0.010 **
	Old trees - absent	-1.072	0.500	0.036 *
Habitat Hectares Condition Score	Intercept (incl. ecological plantings)	-1.897	7.231	0.794
	Type – remnants	12.356	2.605	<0.001 ***
	Type – paddocks	-11.958	3.031	<0.001 ***
	Type – woodlot plantings	17.692	8.518	0.042 *
	Log (age)	13.679	2.681	<0.001 ***
	Plant species richness	0.396	0.091	<0.001 ***
	Riparian/non-rip. location - riparian	-5.522	1.808	0.003 **
	Old trees – absent	-2.789	2.207	0.211
	Woodlot planting by log (age)	-10.948	3.646	0.004 **

3.6 Discussion

We found that revegetation plantings in Victoria were intermediate in species composition, structure and condition between remnant forest and cleared agricultural land. In plantings up to 26 years of age, structural complexity but not floristics had converged with that of reference forest remnants. Ecological plantings and woodlot plantings differed particularly in their weed cover and condition. To revisit our hypotheses set out in the introduction, we found that (1) structural complexity increased with age of planting, but species richness did not; (2) structural complexity of plantings became like that of remnants but species richness did not; (3) ecological plantings and woodlot plantings contained similar vegetation composition and both were distinct from remnants; (4) ecological plantings had lower weed cover than woodlot plantings; and (5) ecological plantings had a condition more similar to remnants than did woodlot plantings.

3.6.1 The foster ecosystem hypothesis

The establishment of native floristic richness of the ground cover (forbs, grasses, many tussocks and prostrate shrubs) is of particular importance in revegetation plantings. These smaller lifeforms are seldom planted, and there has been the suggestion that these species will return if overstorey structure is adequate – the 'foster ecosystem hypothesis' (Haggar et al. 1997, Loumeto and Huttel 1997, Lugo 1997, White et al. 2004). Our study showed no increase in native ground cover species with age of the plantings, and richness remained substantially below that of remnants. Thus, we found no support for the foster ecosystem hypothesis.

The presence of remnant overstorey trees also was not significantly related to the number of native ground cover species. The ground cover of many plantings was dominated either by exotic pasture grasses, or by leaf litter. The lack of native species recruitment in plantings has been reported elsewhere (Wilkins et al. 2003, Norman et al. 2006), but Keenan et al. (1997) found species richness increased with age of plantings in Queensland, primarily due to recolonising tree species. Many previous studies on the 'foster ecosystem hypothesis' investigated colonising woody species only, rather than ground cover grasses and forbs (Silva Júnior et al. 1995, Haggar et al. 1997, Lugo 1997), and some authors have acknowledged that many of the colonising species were exotic (Haggar et al. 1997, Wilkins et al. 2003).

3.6.2 The development of structural complexity

Structural complexity increased with age of planting, no matter how many species were planted at establishment. Statistical modeling showed that HCS4 score of both types of plantings was similar to that in remnants after approximately 30 years. An increase in structural complexity of planted vegetation with age has been noted elsewhere (Kanowski et al. 2003). This may be particularly important for recolonisation by fauna since several studies have highlighted strong correlations between structural complexity and the occurrence of fauna (Arnold 1988, Catling et al. 1998, Cueto and Casenave 1999, McElhinny 2002). In some cases structural complexity is considered to be more important than floristic richness for fauna colonisation (Erdelen 1984, Gilmore 1985, Garden et al. 2007).

The increase in structural complexity in plantings with age, but the lack of increase in plant species richness has been found in other studies (Finegan 1996, Wilkins et al. 2003, Marín-Spiotta et al. 2007). This pattern may indicate that plantings could provide habitat features for fauna in the form of structural complexity, but are unlikely to provide sites for the conservation of plants such as forbs and native grasses, unless these are specifically planted or seed is introduced. Many ground cover plants in this study were found exclusively in remnant vegetation, possibly indicating local ecosystem processes or health.

3.6.3 Vegetation composition

In our study both planting types contained a plant assemblage intermediate between that of remnants and of paddocks. Vegetation in our study region may take much longer than 26 years (our oldest planting) to achieve a similar composition to remnants, or may not be on a trajectory toward remnants given the lack of recruitment of native species, discussed above. Reay and Norton (1999) found vegetation composition in 35 year old plantings was similar to a reference remnant site in New Zealand forest, but both DeWalt et al. (2003) and Marín-Spiotta et al (2007) found that vegetation composition of 70 and 80 year old secondary forest in Central America was different from that in remnants, and the development of composition was slower than structural features in the trajectory toward remnants. In our study, the key lifeforms in remnants that distinguished the composition from plantings, were lifeforms not normally planted, such as prostrate shrubs, creepers, epiphytes and ferns.

3.6.4 Weeds and the diversity-resistance hypothesis

We found that weed species richness was unrelated to native plant richness; a result contrary to predictions from the diversity-resistance hypothesis (Elton 1958). This result also differed from many findings of high weed richness where native richness was high (Espinosa-Garcia et al. 2004, Gilbert and Lechowicz 2005, Fridley et al. 2007). Weed cover is possibly more ecologically meaningful than weed richness, and in our study it was low in remnants and high in paddocks due to the cover of pasture grasses and forbs in the latter, with intermediate weed cover in plantings. Weed cover was higher in riparian than non-riparian sites, which has also been noted in US grasslands (Stohlgren et al. 1998).

Our results for weed cover in revegetation plantings were surprising. Weed cover in woodlot plantings increased with age of the planting, while it declined with age in ecological plantings. It is possible that the greater shrub cover in ecological plantings shaded out weeds, thus reducing the habitat available for (typically) sun-loving exotic plants. Panetta and Groves (1990) noted that weeds in revegetation plantings could be reduced by maximising vegetation cover. A caveat, however, is a possible correlation between landholders who may have ecological knowledge sufficient to establish ecological plantings and also control weeds effectively. This would negate a direct causal relationship between shrub cover and resistance to weed invasion. Weed invasion of plantings is generally considered a difficult management problem (Tucker and Murphy 1997, Schirmer and Field 2001), occasionally preventing uptake of revegetation programs by landholders (Bass Coast Landcare, *pers. comm.*). Our data suggest it is possible that the establishment of shrubs in plantings at the outset may provide resistance to weed invasion, thereby aiding future weed control efforts.

3.6.5 The development of condition

Statistical modeling of the Habitat Hectares Condition Score indicated that ecological plantings could achieve a level of condition similar to that in remnants after approximately 30 years. However, in woodlot plantings, condition remained below that of remnants and did not approach that of remnants. Although the Habitat Hectares Condition Score was designed for remnant vegetation (Parkes et al. 2003), we feel that it is reasonably appropriate to apply it to plantings, because plantings still fall within the range of condition values for which the score was designed. However, condition scores are likely to be low in plantings because of low levels of maturity (recruitment was not evident for all species, and large trees were few in number). We believe this is the first recorded use of Habitat Hectares, or of any similar measure of condition, for plantings in Australia.

3.6.6 Revegetation in riparian areas

Riparian areas contained greater floristic richness than non-riparian locations, though this may be due primarily to weed species. Riparian areas were lower in condition score. Species composition and structure were not different between riparian and upland sites. Previous research near our study area found that remnant riparian areas supported a distinct and more structurally complex vegetation than non-riparian areas (Soderquist and Mac Nally 2000, Palmer and Bennett 2006). Possibly responding to vegetation differences, Australian fauna have been found to be more abundant and species-rich in riparian areas than surrounding hillsides (Suckling and Heislers 1978, Bentley and Catterall 1997, Mac Nally et al. 2000, Soderquist and Mac Nally 2000, Woinarski et al. 2000, Catterall et al. 2001, Palmer and Bennett 2006) although this is not a universal rule (Sabo et al. 2005). Riparian areas have been targeted for restoration because of potentially higher biodiversity richness, as well as abiotic reasons such as water quality and bank stabilisation (Naiman et al. 1993, Fisher and Goldney 1997, Staton and O'Sullivan 2006).

It is likely that the structure of plantings in our study had not reached maximum complexity, therefore differences between riparian and non-riparian areas may not be distinguishable. For example, it may be many years before the trees in revegetation plantings contribute large logs to the ground or to streams, yet high log loads have been identified as key features of remnant riparian forest (Palmer and Bennett 2006).

3.6.7 Non-significant variables

The size and shape of a patch was not significantly related to the structural complexity, condition or floristics at the site within the patch. The projected foliage cover, topography of the site, the number of species planted at establishment and the vegetation cover in the surrounding landscape also did not affect most measures of structural complexity or floristic diversity. It is possible that plantings of up to 26 years of age are not sufficiently mature to develop significant species density-area relationships, and to respond to landscape features such as topography and vegetation cover. More long-term data, or data on older plantings, if available, may improve our understanding of the long-term biodiversity value of plantings.

3.7 Conclusion

As predicted, we found that ecological plantings had higher condition scores, greater plant diversity, lower weed cover, and greater midstorey cover than woodlot plantings. Ecological plantings were similar to remnants in species richness, number of lifeforms and approached remnants over time in condition score and weed cover. Both ecological plantings and woodlot plantings approached remnants in structural complexity. Ecological plantings were more similar to remnants in species composition than woodlot plantings. Because of these features, ecological plantings may provide greater habitat value for wildlife, as well as provide sites for the ongoing existence of those shrub species that were planted.

Despite these positive trends, in the short term, even when best-practice revegetation techniques are used, plantings are unlikely to be a viable replacement of remnant vegetation in temperate forest communities. After 30 years, woodlot and ecological plantings did not attain the ground layer floristic diversity of remnants, and appear unlikely to do so without deliberate introduction of additional species.

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3.10 Appendices

Appendix 3.1. List of variables measured at each site. The second column lists the variables used in the PCA of structural complexity, with transformations necessary to achieve approximately normal distributions.

Structure	Variables used in PCA
% cover tree canopy	
% cover small trees (>2m)	
% cover tall shrubs (>2m) incl. tree ferns	
% cover medium shrubs (1-2m)	Log (midstorey cover)
% cover small shrubs (<1m)	Log (understorey cover)
% cover ground cover	Log (ground cover)
% cover logs	
% cover leaf litter	
Number of tees per ha (>60cm DBH)	Log (trees per ha)
Number of dead trees per ha (>60cm DBH*)	Log (dead trees)
Number of big shrubs per ha (>2m height)	Log (big shrubs)
Number of small shrubs per ha (0.5-2m height)	Log (small shrubs)
Length of logs per ha (>10cm diameter)	Log (logs)
Number of tree hollows per ha	Log (hollows)
Average height of three largest trees (m)	Largest height
Average DBH of three largest trees (m)	Log (largest DBH)
Standard deviation of tree height (m)	StdDev (height)
Standard deviation of tree DBH (m)	StdDev (DBH)
Grass cover (an average of cover scores)	Grass
Forbs cover (an average of cover scores)	Forbs
Tussocks cover (an average of cover scores)	Tussocks
Leaf litter cover (an average of cover scores)	Litter
Projected foliage cover of trees	PFC trees
Projected foliage cover of shrubs	PFC shrubs
Number of strata	Strata

Floristics
Overstorey richness, native species richness and cover
Midstorey richness, native species richness and cover
Understorey richness, native species richness and cover
Ground cover richness, native species richness and cover
Total species richness and native species richness
Weed cover

** DBH - diameter at breast height*

Appendix 3.2. Table of means for selected site characteristics. Values in cells are the mean ± the standard error of the mean and the range.

Site attribute	Paddocks	Woodlot planting	Ecological planting	Remnant
Number of sites	11	16	27	18
Age (years)	n/a	13.6 ± 1.8	8.1 ± 1.0	n/a
		2 to 26	2 to 26	
Area (ha)	n/a	7.9 ± 6.0	2.3 ± 0.5	10.0 ± 2.4
		0.1 to 10.9, 96.5	0.07 to 10.6	0.1 to 27.5, 43, 528
Total number of spp.	19.9 ± 1.6	24.4 ± 2.1	39.2 ± 2.1	34.7 ± 2.6
	10 to 29	12 to 43	17 to 68	17 to 52
Number of native spp.	5.5 ± 1.0	10.2 ± 1.3	24.9 ± 1.7	27.6 ± 2.2
	1 to 11	4 to 24	9 to 48	13 to 47
Number of native GC spp.	1.9 ± 0.3	3.1 ± 0.6	4.1 ± 0.4	8.9 ± 1.2
	1 to 4	1 to 9	0 to 7	3 to 21
Number of planted species	n/a	4.7 ± 1.1	20.0 ± 1.5	n/a
		1 to 15	8 to 39	
% weed cover [a]	98.8 ± 2.0	83.9 ± 4.6	61.9 ± 5.6	27.4 ± 5.7
	90 to 115	37 to 108	13 to 122	0 to 96
HCS4[b]	7.7 ± 0.7	13.9 ± 0.6	15.2 ± 0.4	17.7 ± 0.5
	4 to 12	9 to 18	9 to 19	14 to 21
PFC[c] trees	11.8 ± 5.4	39.1 ± 3.7	39.3 ± 3.7	43.9 ± 2.8
	0 to 60	2 to 62	0 to 66	13 to 62
PFC[c] shrubs	0.4 ± 0.2	5.4 ± 2.5	14.9 ± 2.5	35.2 ± 5.2
	0 to 2	0 to 33	0 to 44	0 to 76
Number of large trees	6.8 ± 4.3	15.4 ± 5.7	9.7 ± 4.5	79.1 ± 12.5
(dbh>60cm)	0 to 48	0 to 75	0 to 110	10 to 165
Trees per ha	4.6 ± 2.2	232.9 ± 58.8	95.5 ± 20.1	231.8 ± 19.8
	0 to 22	0 to 771	0 to 440	110 to 420

[a] % weed cover is the sum of weed cover in 4 strata layers, hence it can be greater than 100%

[b] HCS4 is the Habitat Complexity Score – for description see text

[c] PFC is the projected foliage cover

Chapter 4: Bird response to revegetation plantings of different structure and floristics – are restoration plantings restoring bird communities?

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Bird response to revegetation plantings of different structure and floristics - are restoration plantings restoring bird communities?

4.1 Abstract

Revegetation plantings have been established throughout the world to mitigate the effects of clearing, including loss of faunal habitat. Revegetation plantings can differ substantially in structural complexity and plant diversity, with potentially differing habitat qualities for fauna. We studied bird occurrence in revegetation of different complexity and floristics in southern Australia. We assessed bird species richness and composition in remnant forest and cleared agricultural land as reference points, and in two types of plantings differing in structure and floristics – (1) 'woodlot plantings' composed of native trees only, and (2) 'ecological plantings' composed of many species of local trees, shrubs and understorey. By approximately 20 years of age, both types of plantings had a similar bird species richness to that in remnants. Bird species richness was greater in ecological plantings than woodlot plantings. Species composition also differed. Ecological plantings contained a shrub-associated bird assemblage, whereas woodlot plantings were dominated by generalist bird species. Remnants contained a unique bird assemblage, which were not found in either of the two types of plantings, suggesting that plantings are not a viable replacement of remnant vegetation over this time period. Bird species richness responded positively to structural complexity, but not to floristic richness. Bird species richness was greater in plantings that were older, in riparian locations, and where weed cover was lower. We conclude that plantings in general can provide habitat for many species of birds, and that structurally complex ecological plantings in particular will provide unique and valuable additional habitat for birds.

4.2 Keywords

Bird communities; countryside biogeography; floristic diversity; restoration; revegetation; vegetation structure.

4.3 Introduction

Around the globe, forest and woodland vegetation has been cleared for agricultural expansion, causing land degradation (Bird et al. 1992, Eberbach 2003, Foley et al. 2005) and a loss of wildlife (Saunders et al. 1991, Yates and Hobbs 1997, Fahrig 2003, Teyssèdre and Couvet 2007). To ameliorate these problems, vegetation restoration, including revegetation, has been conducted on previously cleared land (King and Keeland 1999, Turner and Ward 2002, Rey Benayas et al. 2009). However, the value of revegetation as habitat for wildlife is often poorly understood (Kimber et al. 1999, Munro et al. 2007).

In Australia, vegetation clearing for agriculture has been recent (about 150 years), extensive and intense (Yates and Hobbs 1997). The last three decades have seen a dramatic increase in revegetation schemes and incentives (Wilson et al. 1995, Herbohn et al. 2000), such as for farm forestry, and plantings for salinity abatement, erosion control and habitat provision for wildlife. The plantings resulting from these schemes can differ substantially in structure and composition, with some focussing on tree production, and others attempting to re-create the properties of remnant vegetation. The comparative biodiversity benefits of these different types of revegetation plantings are unknown.

We focus on birds because they are an easily studied taxon with diverse habitat needs (Recher 2004). They are relatively mobile, and hence recolonisation of plantings may be rapid. Birds have also undergone dramatic changes in populations in the agricultural areas of Australia, with some species now of conservation concern, and a small number of generalists having increased in abundance (Recher 1999, Ford et al. 2001).

Two important variables, structural complexity and floristic richness, have long been known to be important predictors of bird species diversity (MacArthur and MacArthur 1961, Recher 1969). However, little is known about the extent to which these variables influence the value of newly created revegetation to birds (Nichols and Watkins 1984, Twedt et al. 2002, Arnold 2003). Previous studies have shown that in planted windbreaks in Japan, bird diversity was closely related to foliage height diversity (Hino 1985); wildlife species richness in American shelterbelts was correlated with diversity and complexity of vegetation (Johnson and Beck 1988); and European hedgerows that were large, dense and composed of many plant species had the highest species richness of birds (Osborne 1984, MacDonald and Johnson 1995, Hinsley and Bellamy 2000). To date, only one Australian study has compared the bird richness between structurally and floristically diverse plantings in tropical rainforests with agroforestry plantings of one or a few species, and found a greater species richness in the former (Kanowski et al. 2005).

In this study, we compared bird species richness and composition in remnant vegetation (as a reference goal), paddocks (cleared agricultural land, as the starting point), and revegetation plantings of varying structural complexity and floristic richness. We assessed the effect of structure and floristics on bird species richness, and richness of forest-dependent birds and open-country birds. We predicted that: (1) bird species richness would be greater in revegetation with greater structural complexity and floristic richness; and (2) the bird assemblage would be different between sites types (remnant, paddock, revegetation), with the bird species composition in floristically diverse plantings predicted to be more similar to that of remnants than species-poor plantings. In addition, we investigated the effects of resources such as shrubs, logs and old trees (McElhinny et al. 2006), riparian or non-riparian location (Palmer and Bennett 2006), patch size (Mac Nally and Watson 1997, Kavanagh et al. 2007) and the amount of surrounding vegetation cover in the landscape (Radford and Bennett 2007, Haslem and Bennett 2008). Although these variables were not the core focus of our study, they were shown to be important in previous work, and therefore needed to be accounted for.

4.4 Methods

4.4.1 Study area

Our study was conducted in West Gippsland, Victoria, southern Australia (Figure 1). Prior to clearing in the late 19th century, the region was dominated by structurally complex forest with overstorey trees to 30 m, dense midstorey vegetation in the gullies, and a dense understorey (Korumburra and District Historical Society Inc. 1998). Remnant woody vegetation now is approximately 12% cover with only 6% of non-coastal woody vegetation remaining (Department of Sustainability and Environment 2003a). All sites in the study were from non-coastal locations.

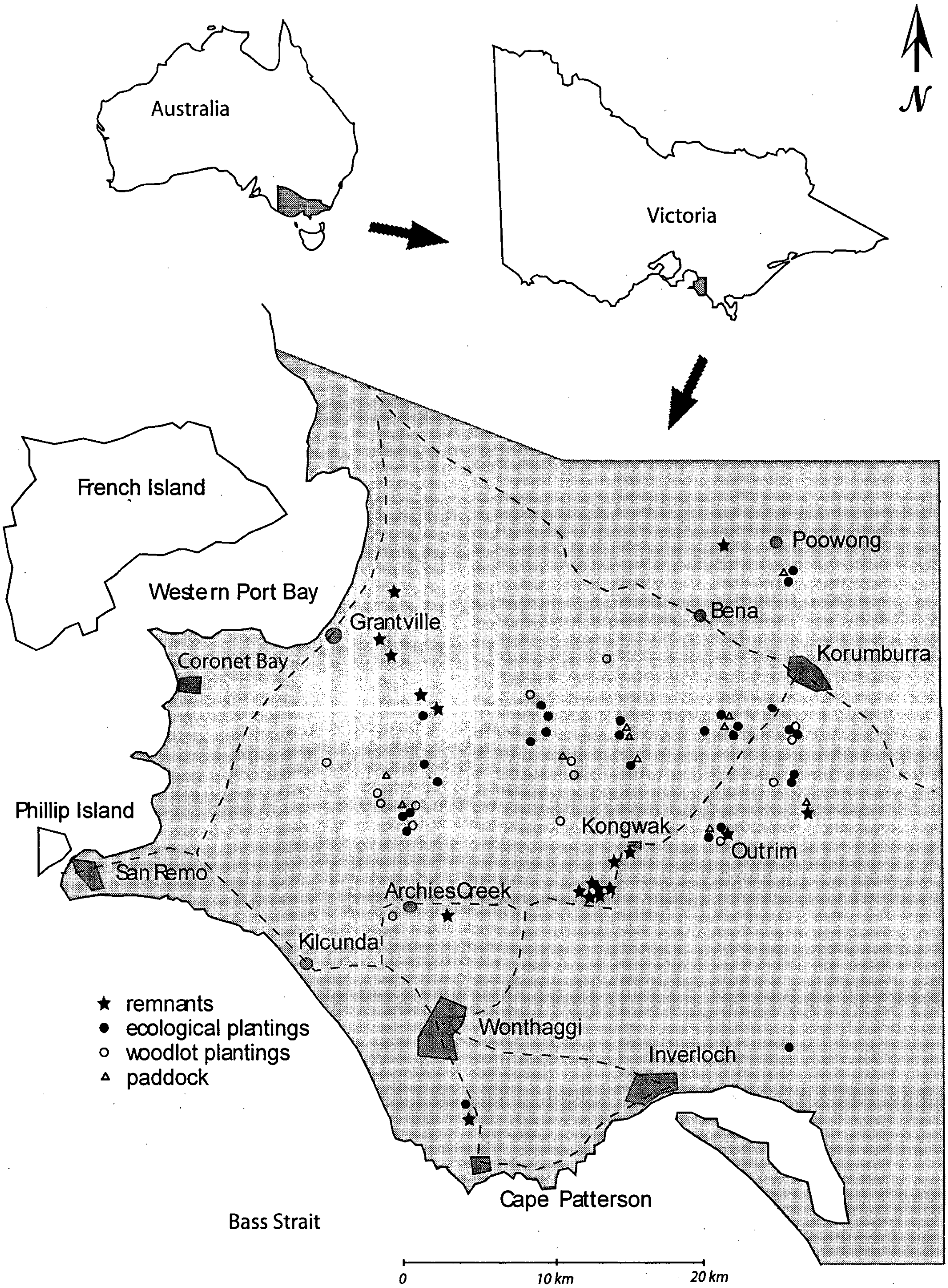

Figure 4-1 Map of study area.
The study area was in Gippsland, southern Victoria, Australia. Terrestrial land is indicated in the main map by pale grey, townships are in dark grey, dotted lines are major roads. Sites are indicated by different symbols (see key).

Various types of land degradation, such as erosion and biodiversity loss, have prompted many landholders to revegetate areas of previously cleared land. We identified two types of revegetation: (1) '*ecological plantings*' were planted for ecosystem restoration purposes and were characterised by a diverse assemblage of tree, shrub and understorey species; and, (2) '*woodlot plantings*' were typically planted with low plant species richness of primarily overstorey species (agro-forestry plantings). Both types of plantings were established with predominantly local, native vegetation, and most were planted for multiple reasons such as for shelter for stock, erosion control and prevention, water clarity, habitat provision and aesthetics. The vegetation composition and structure of these planting types are described in more detail in Munro et al. (2009). Briefly, from our previous research, structural complexity increased in both planting types with age, toward that of remnants, but plant species richness did not. Ecological plantings were more similar to remnants than woodlot plantings in plant composition, the number of strata and vegetation lifeforms, and in overall ecosystem condition (Munro et al. 2009).

4.4.2 Site and patch selection

We selected patches of four types using maps and colour aerial photographs, on-ground searching and consultation with a local Landcare Network: ecological plantings (27 patches), woodlot plantings (16), remnants (18) and paddocks (cleared agricultural land; 11). All paddocks were mostly or completely cleared grazed agricultural land. Remnant patches (which were used as reference sites) were degraded to different extents. Almost all had been selectively logged in the 19th century and possibly grazed in the past. Although some were in near-natural condition, others were infested by weeds (invasive exotic plants). Despite these imperfections, the remnants we chose constituted the best reference patches available. Within each selected patch was a single 100 m x 40 m site, located approximately centrally, and selected to be representative of the patch. We included some patches smaller than this site size, in which the site was the entire patch.

Both planting types ranged in size from 0.07 to 10.9 ha with one large woodlot planting of 96.5 ha. Most remnants ranged in size from 0.1 to 27.5 ha although there were two large remnants of 43 and 528 ha. The age of plantings ranged from 2 to 26 years for both woodlot and ecological plantings. Woodlot plantings were, on average, older than ecological plantings (mean of woodlot plantings was 13.6 ± 1.8 (s.e.) years; mean of ecological plantings was 8.1 ±

1.0 years, t_{41}=2.90, P=0.006). Half the sites within each type were in riparian locations, and half were at least 100 m from the nearest watercourse; approximately half the plantings of both types and half of the paddocks contained old remnant trees.

Table 4-1 List of explanatory variables included initially in GLM modelling, and CANOCO analyses, including units and transformations used prior to analysis, where applicable.

Variable	Description	Range
Used in all analyses		
Type	4 categories: paddock, woodlot planting, ecological planting, remnant	-
Age	Years since planting, at 2006 (natural log transformed)	2 – 26
Area	In hectares, natural log transformed	0.07 – 96.5
Shape	Long or round	-
Riparian	Riparian or non-riparian	-
Large trees	Number of large trees (>60 cm dbh) per ha	0 – 165
Small shrubs	Number of small shrubs (<1 m height) per ha	0 – 5550
Woody species richness	Plant richness of all woody species, plus large graminoids such as *Dianella, Gahnia, Lepidosperma*.	0 – 47
Weed cover[a]	% cover of exotic invasive plants from 4 vegetation layers, summed	0 – 122
Log length	The total length of logs (m) per site, where a log was >10 cm in diameter	0 – 635
Landscape vegetation cover	% area of woody vegetation (remnants, plantings, house gardens, conifers) in 2.25 km radius surrounding the site	2 – 57
Habitat Hectares Condition Score	Sensu Parkes et al. (2003). The similarity to reference remnant sites. The maximum score attainable was 75	12 – 68
HCS4	The sum of scores for 8 different strata layers: tree canopy, small trees (>2 m), tall shrubs (>2 m), medium shrubs (1-2 m), small shrubs (<1 m), ground cover, logs and leaf litter	4 – 21
Also used in the CANOCO analyses		
Understorey cover[a]	% cover of vegetation between 0.5 m and 1 m (used instead of number of small shrubs/ha)	0 – 99
Midstorey cover[a]	% cover of vegetation between 1 m and 3 m	0 – 83
Overstorey cover[a]	% cover of vegetation greater than 3 m	0 – 100
Total species	The number of plant species found (used instead of woody species richness)	12 – 62
Species planted	The number of species planted at establishment	1 – 39
Height of largest trees	The average height of the 3 largest trees in the site (m)	0 - 40
Dead trees	The number of dead trees per ha	0 - 140
pfc	Projected foliage cover (%) of overstorey, average of 5 plots	0 - 66

[a] *% cover was estimated visually*

4.4.3 Measurement of structure and floristics

In addition to the above stratifying variables, we recorded the density of large trees, height of largest trees, vegetation cover in different layers, vegetation species richness, patch shape (long or round), the amount of surrounding vegetation cover, the number of structural strata (see below), plus a score of vegetation condition, the Habitat Hectares Condition Score (Table 1). The Habitat Hectares Score is a measure of how similar the vegetation is to an essentially unmodified reference site of the same vegetation type (Parkes et al. 2003, Munro et al. 2009). The condition component of the Habitat Hectares Score was derived from measures of seven different site-based attributes. The seven attributes were: number and health of large trees, canopy cover and health, understorey composition, cover of weeds, recruitment of perennial plants, organic litter cover, and log cover. The measure of each attribute was compared to a benchmark value. High scores indicated greatest similarity to the benchmark. The scores were then summed to derive a final condition score for each site (for details, see Munro et al., 2009). Notably, the reference conditions used in this score are determined by the state conservation agency (Victorian Department of Sustainability and Environment) and are very pristine. That is, the reference sites used to calculate the Habitat Hectares Score were different from the remnant sites used in our study. We used only the condition component of the Habitat Hectares Score (i.e. we did not include its landscape component; see Parkes et al., 2003).

We also calculated a Habitat Complexity Score (HCS4), modified from that of Tasker and Bradstock (2006) and Coops and Catling (1997). We allocated a score of 0 to 4 to eight pre-determined vegetation layers, according to the percent cover of that layer. The eight vegetation layers were: tree canopy, small trees (>2 m), tall shrubs (>2 m), medium shrubs (1-2 m), small shrubs (<1 m), ground cover, logs and leaf litter (for details, see Munro et al., 2009). We determined the size of the patch containing each site, and the amount of woody vegetation cover within a radius of 2.25 km using colour aerial photographs (taken in 2004, 15 cm resolution) and GIS software (ArcMap v.9.1, ESRI 2005). Our choice of a 2.25 km radius was informed by the approximate scale at which we believed birds may respond to woody vegetation cover (e.g. Villard et al. 1999, Pearman 2002, Radford et al. 2005). All variables are defined in Table 1, with the means of key variables summarised in Table 2. Further information on the collection of vegetation data is detailed in Munro et al. (2009).

4.4.4 Bird count methods

We detected birds by the point-count method (Bibby et al. 1992) with a single observer (NTM), at three plots spaced at 50 m intervals within each site. Birds were counted for 5 minutes at each plot within a 20 m radius. Birds were detected by both sight and sound. Over-estimating numbers of the same bird species was avoided by only counting multiple birds of the same species if they were seen or vocalised at the same time. We counted birds between sunrise and 1000 h on fine days (without rain or strong wind), in spring of 2005 and 2006, with two visits per site per year. We pooled data over the three plots at a given site to give a species list per visit per year (a total of four visits). We included birds in analyses only if three or more individuals were detected during the entire study. We assigned birds to two habitat guilds based on published classifications (Bennett and Ford 1997, Radford and Bennett 2005): forest-dependent birds and open-country birds.

Table 4-2 Table of means of several site variables.
Presented are some design variables, variables that are significantly different between site types, and bird richness data. Values in cells are the mean ± the standard error of the mean, and the range.

Site attribute	Paddocks	Woodlot planting	Ecological planting	Remnant
Number of sites	11	16	27	16
Age (years)	n/a	13.6 ± 1.8 2 to 26	8.1 ± 1.0 2 to 26	n/a
Area (ha)	n/a	7.9 ± 6.0 0.1 to 10.9, 96.5	2.3 ± 0.5 0.07 to 10.6	10.0 ± 2.4 0.1 to 27.5, 43, 528
Number of plant species	19.9 ± 1.6 10 to 29	24.4 ± 2.1 12 to 43	39.2 ± 2.1 17 to 68	34.7 ± 2.6 17 to 52
Number of woody plant species	5.29 ± 1.2 0 to 12	11.59 ± 1.7 2 to 26	24.00 ± 1.6 12 to 47	20.89 ± 1.4 10 to 34
Number of species planted	n/a	4.7 ± 1.1 1 to 15	20.0 ± 1.5 8 to 39	n/a
Habitat Hectares Condition Score [a]	19.2 ± 1.8 13 to 31	27.9 ± 2.0 12 to 46	35.6 ± 2.1 18 to 57	51.9 ± 2.1 34 to 68
Landscape vegetation cover within 2.25 km	4.3 ±0.8 2 to 10	5.6 ±0.6 2 to 11.5	4.5 ±0.6 2 to 18	17.6 ±3.2 2 to 57
Number of bird species (seen 3 or more times)	9.6 ± 1.2 4 to 18	15.4 ± 1.29 7 to 24	16.2 ± 0.91 2 to 29	21.8 ± 0.9 14 to 27
Number of forest bird species	± 1.1 0 to 12	9.9 ± 1.1 2 to 16	11.4 ± 0.7 1 to 20	16.2 ± 1.2 5 to 23
Number of open-country bird species	6.8 ± 0.6 4 to 12	5.6 ± 0.5 1 to 9	4.9 ± 0.3 1 to 9	5.6 ± 0.5 2 to 9

[a] The maximum Habitat Hectares Condition Score attainable is 75.

4.4.5 Analyses

Bird counts provided two measures – species richness at a site and the detection rate of birds at a site. Species richness was the sum of different bird species detected across all plots, visits and years. We performed analyses on the species richness of three groups of birds: all birds, forest-dependent birds and open-country birds. The detection rate of birds was the number of times a bird species was present at a site out of four visits. We used the detection rate to characterise bird composition using correspondence analyses.

We fitted generalised linear models, with a quasi-Poisson distribution to account for overdispersion, and a logarithmic link function, to bird species richness data as a function of selected site attributes (Table 1). Variables were initially included in the models based on their known or suspected impact on bird species richness, and were primarily variables of management concern. Models were first developed using all four site types, with the variables initially included in the models as follows: site type, natural log (age), natural log (area), woody species richness, the number of large trees (>60cm dbh) per ha, riparian or non-riparian location, landscape vegetation cover and weed cover (Table 1). We used an information-theoretic approach to select regression models. All possible subsets of the explanatory variables were considered and the model with the lowest Akaike Information Criterion (Akaike 1974) was selected. Some of our explanatory variables were correlated (Tables 3 and 4) and therefore other models fitted the data almost as well as the one selected. Multicollinearity is often an unavoidable problem in ecological datasets. Our model selection protocol focussed on finding parsimonious models, and produced ecologically credible results. For this reason, we feel that the degree of multicollinearity in our data, while not ideal, did not undermine the validity of our findings.

Table 4-3 Correlation Matrix of explanatory variables in the linear models with all four site types included. (* significant at 0.05, ** significant at 0.01. *** significant at <0.001).

	Age	Area	Large trees	Landscape veg. cover	Weed cover
Area	0.260 *				
Large trees	0.719 ***	0.400 *			
Landscape veg. cover	0.614 ***	0.675 ***	0.574 ***		
Weed cover	-0.647 ***	-0.239 *	-0.493 ***	-0.424 ***	
Woody spp.	0.265 *	0.224	0.269 *	0.237 *	-0.402 ***

To remove problems of unrepresented data (e.g. paddocks and remnants did not have an 'age', and paddocks had no 'area' of woody vegetation), the age of paddocks and remnants was assigned the mean natural log (age) of the plantings, the area of paddocks was assigned the mean natural log (area) of plantings and remnants, and the number of species planted for paddocks and remnants was assigned the mean number of species planted in the plantings. This effectively eliminated the effect of site types where a particular variable was undefined.

Second, to explore the specific features of plantings related to bird species richness, we fitted generalised linear models to species richness as a function of several site attributes with only data from ecological plantings and woodlot plantings. We excluded site type in these analyses. Variables initially included in these models were: age, area, woody species richness, number of small shrubs per ha, length of logs, number of large trees per ha, riparian or non-riparian location, landscape vegetation cover, HCS4, weed cover and shape. The same information-theoretic approach outlined above was used to select a final model.

Table 4-4 Correlation matrix of explanatory variables in the linear models of plantings only. (* significant at 0.05, ** significant at 0.01. *** significant at <0.001).

	HCS4	Age	Area	Large trees	Landscape veg. cover	Weed cover	Woody spp.	Lengtl logs
Age	0.301							
Area	-0.076	0.261						
Large trees	0.367 *	0.612 ***	0.171					
Landscape veg. cover	0.170	0.330 *	0.382 *	0.057				
Weed cover	-0.143	-0.001	0.063	-0.017	0.108			
Woody spp.	0.549 ***	-0.054	0.023	0.194	-0.063	-0.304 *		
Length logs	0.113	0.230	0.217	0.559 ***	-0.012	-0.020	0.141	
Small shrubs	-0.146	-0.172	-0.087	-0.095	-0.118	-0.108	0.256	-0.095

To investigate the comparative effects of landscape-scale vegetation cover versus local-scale vegetation condition, we fitted a generalised linear model to bird species richness as a function of landscape woody vegetation cover (the amount of woody vegetation cover within a 2.25 km radius of the site) and condition (as measured by the Habitat Hectares Condition Score).

Finally, we tested the relationship between bird species composition and vegetation structure using direct gradient analysis conducted in CANOCO 4.53 (ter Braak and Šmilauer 2002). We conducted a preliminary detrended correspondence analysis (DCA) on the bird detection rate data to determine the appropriate response model. A DCA length of gradient of the first axis of <3 indicated that redundancy analysis (RDA) was appropriate and we used it for all subsequent analyses. RDA assumes a linear response to the underlying environmental variables. For these analyses we used a default value of zero for the age of paddocks, and we assigned an age of 30 years to all remnants (a little above our oldest planting). We followed the procedures as outlined in Lepš and Šmilauer (2005) and ter Braak and Šmilauer (2002) and tested the significance of the relationship of each habitat variable (Table 1) to the bird data using a Monte Carlo permutation test (999 permutations), with only significant variables included in the final analysis ($P < 0.05$). Rare species were down-weighted following the procedure recommended by ter Braak and Šmilauer (2002). We tested the differences in bird assemblages between site types by pair-wise comparisons using a Monte Carlo permutation test (999 permutations) on each possible pair of site types (ter Braak and Šmilauer 2002). We repeated the RDA as above, but including only the two types of plantings.

4.5 Results

We observed 75 bird species, of which 55 were observed at least three times throughout the entire study (19 open-country species, 36 forest-dependent species).

4.5.1 Bird species richness in all site types

Total bird species richness was significantly greater in remnants, and significantly lower in paddocks, but was not different between ecological plantings and woodlot plantings (Table 5). Total bird species richness increased with site age (Figure 2; Table 5). Bird richness continued to increase after approximately 20 years of age (the maximum age in our analysis was 26 years) and was higher in riparian than non-riparian areas (Table 5).

Forest bird species richness was significantly higher in ecological plantings than in woodlot plantings, and forest bird richness was significantly lower in paddocks, but was not different between remnants and plantings (Table 5). Forest bird species richness increased with age and patch size (Table 5). Open-country bird richness was significantly higher in paddock and remnant sites than in either type of planting.

4.5.2 Bird species richness in plantings only

We refitted our models with data only from the plantings, omitting site type as an explanatory variable. Richness of all birds and forest birds increased with higher HCS4 scores, and was greater in larger patches and where weed cover was lower (Table 6). Total bird richness was also greater in older patches. Open-country birds responded positively to age of planting and there was greater richness in riparian areas (Table 6).

Table 4-5 Bird species richness in all four site types.
Results from generalised linear models of species richness of all birds, forest birds and open-country birds as a function of four site types and several environmental variables. Presented are the models with the lowest AIC, so some variables are retained which are not significant at P=0.05.

Response	Variables in model	Parameter estimates	Standard Error	P-value
Total bird species richness	Intercept (incl. ecological plantings)	2.306	0.161	<0.001
	Type - paddocks	-0.487	0.111	<0.001
	Type – remnants	0.173	0.078	0.030
	Type – woodlot plantings	-0.103	0.088	0.251
	Log (age)	0.250	0.063	<0.001
	Log (area)	0.033	0.018	0.070
	Location - riparian	0.166	0.059	0.006
	Weed cover	-0.002	0.001	0.071
Forest bird species richness	Intercept (incl. ecological plantings)	1.872	0.216	<0.001
	Type - paddocks	-1.427	0.230	<0.001
	Type – remnants	0.096	0.143	0.504
	Type – woodlot plantings	-0.261	0.131	0.050
	Log (age)	0.221	0.103	0.035
	Log (area)	0.063	0.029	0.033
	Number of large trees per ha	0.002	0.001	0.130
	Location - riparian	0.139	0.095	0.148
Open-country birds	Intercept (incl. ecological plantings)	0.754	0.190	<0.001
	Type - paddocks	0.304	0.103	0.005
	Type – remnants	0.376	0.126	0.004
	Type – woodlot plantings	-0.013	0.108	0.902
	Log (age)	0.380	0.090	<0.001
	Log (area)	-0.044	0.027	0.107
	Number of large trees per ha	-0.003	0.001	0.039
	Location – riparian	0.137	0.077	0.077

4.5.3 'Quality' versus 'quantity' of vegetation

We investigated how birds responded to landscape-scale quantity of vegetation (measured by the area of vegetation within 2.25 km of the site) and local-scale 'quality' of vegetation (measured by the Habitat Hectares Condition Score). The range of these scores is presented in Table 2. Richness of all birds and forest birds was significantly positively correlated with the condition score, but not with vegetation cover in the landscape (Table 7). Open-country birds did not respond to either measure. The interaction term 'quality' by 'quantity' was not significant for all three bird groupings.

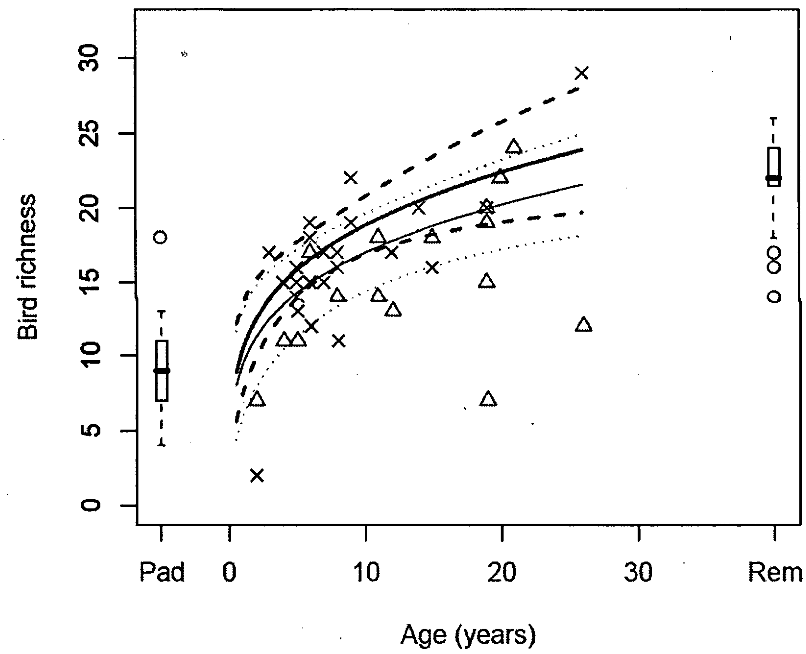

Figure 4-2 Total bird species richness by age.
Lines are the plotted generalised linear model. The solid lines are the fitted model, dotted lines are 95% confidence intervals. Dark lines are ecological plantings, pale lines are woodlot plantings. The data points are also plotted (crosses are ecological plantings, triangles are woodlot plantings). Boxplots are displayed of the original data, for remnants and paddocks. There was significantly higher bird species richness in ecological plantings than in woodlot plantings.

4.5.4 Bird composition

We found a significant difference in bird composition between site types (Fig. 3). Pair-wise comparisons indicated that each site type was significantly different from every other type (Table 8). When we modelled just the two types of plantings, bird composition in ecological plantings differed significantly from that in woodlot plantings (P=0.004, F-ratio=2.518, Figs. 4 and 5).

Table 4-6 Bird species richness in plantings only.
Results from generalised linear models of richness of all birds, forest birds and open-country birds as a function of several environmental variables. Only data for ecological plantings and woodlot plantings were included in the model, and site type was excluded. N=42 sites. Presented are the models with the lowest AIC, so some variables are retained which are not significant at P=0.05.

Response	Variables in model	Parameter estimates	SE	P-value
Total bird species richness	Intercept	1.830	0.273	<0.001
	Age	0.012	0.005	0.023
	Area	0.004	0.002	0.054
	Location – riparian	0.134	0.069	0.061
	Weed cover	-0.003	0.001	0.005
	HCS4	0.063	0.018	0.001
Forest bird species richness	Intercept	0.898	0.361	0.017
	Area	0.006	0.002	0.020
	Weed cover	-0.004	0.001	0.013
	HCS4	0.114	0.022	<0.001
Open-country birds	Intercept	1.234	0.089	<0.001
	Age	0.026	0.006	<0.001
	Location - riparian	0.184	0.086	0.039

The environmental variables that were significantly related to bird species composition when all four site types were included were the same as when paddock sites were removed. They were: the number of dead trees/ha, understorey cover, the number of native plant species, the average largest tree height, the projected foliage cover, age, and the number of species planted. Most of these variables were positively associated with remnant vegetation, whereas the number of species planted and the overstorey projected foliage cover were positively associated with ecological plantings (Fig. 3). When we included just the two planting types, the environmental variables related to bird species composition were: understorey cover, the average largest tree height, age, the number of species planted, the number of large trees (>60 cm dbh) per ha, and weed (invasive exotic plant) cover (Fig. 4). Variables positively associated with ecological plantings were understorey cover and the number of species planted (the majority of which were shrubs). Variables positively associated with woodlot plantings were age, the number of large trees and weed cover.

Woodlot plantings were dominated by generalists and birds associated with partly cleared landscapes, such as the Magpie Lark (mlark), Red Wattlebird (rwb), Little Raven (lrav), Australian Magpie (amag) and Eastern Rosella (eros) (Fig. 5, see Appendix for scientific names of birds). Ecological plantings contained shrub-using species, such as the Superb Fairy Wren (sfw), Crested Shrike-tit (cstit), Brown Thornbill (btb), Golden Whistler (gw) and Eastern Yellow Robin (eyr) (Fig. 5). Ecological plantings between 4 and 8 years of age contained a similar bird composition to woodlot plantings of 11 to 15 years. Remnants included species dependent on resources supplied by mature trees such as the White-throated Treecreeper, Varied Sitella (which are bark specialists), Mistletoebird (a mistletoe specialist), Blue-winged Parrot, Laughing Kookaburra, Eastern Rosella (hollow users), and others such as Rufous Whistler, Eastern Yellow Robin and Fan-tailed Cuckoo (not shown). Weed cover in the plantings appears to be associated with a particular composition of birds, including the European Goldfinch and Common Myna (egf and cmyn respectively in Fig. 5), both of which are introduced species.

Table 4-7 Habitat quality versus quantity.
Results from generalised linear models of species richness of all birds, forest birds and open-country birds as a function of the Habitat Hectares Condition Score (quality) and vegetation cover in a 2.25km radius of the site (quantity). Only data for ecological plantings and woodlot plantings are included in the model, and site type is excluded as a variable.

Response	Variables in model	Parameter estimates	SE	P-value
Total bird species richness	(Intercept)	2.125	0.160	<0.001
	Habitat Hectares Condition Score	0.018	0.004	<0.001
	Vegetation cover in landscape	0.009	0.025	0.716
	HH Condition by landscape cover	-0.0002	0.0004	0.674
Forest bird species richness	(Intercept)	1.331	0.255	<0.001
	Habitat Hectares Condition Score	0.027	0.006	<0.001
	Vegetation cover in landscape	0.017	0.030	0.661
	HH Condition by landscape cover	-0.0003	0.0007	0.633
Open-country birds	(Intercept)	1.331	0.255	<0.001
	Habitat Hectares Condition Score	0.027	0.006	<0.001
	Vegetation cover in landscape	0.017	0.039	0.661
	HH Condition by landscape cover	-0.0003	0.0007	0.663

4.5.5 Conservation benefits of plantings

Of the 55 bird species used in analyses, six were considered of conservation concern in the study region (Radford and Bennett 2005). Of these, three occurred only in remnants (blue-winged parrot *Neophema chrysostoma*, varied sittella *Daphoenositta chryoptera* and white-throated treecreeper *Cormobates leucophaeus*), two were rarely detected and only in remnants and ecological plantings (brown-headed honeyeater *Melithreptus brevirostris* and leaden flycatcher *Myiagra rubecula*), and one occurred rarely, and only in ecological plantings (crested shrike-tit *Falcunculus frontatus*). No birds of conservation concern were detected in woodlot plantings or paddocks.

4.6 Discussion

Throughout Australia in agricultural landscapes, many forest and woodland birds are thought to be declining (Recher 1999, Ford et al. 2001, Barrett et al. 2007). Revegetation plantings have been established, in part, to provide habitat for these birds. We found that revegetation plantings can contribute substantially as habitat for birds, but their habitat value differed depending on their structure. Plantings established for restoration purposes (with structurally complex vegetation and high plant species diversity) contained higher forest–dependent bird species richness than plantings of low complexity and floristics, and attained a similar richness to that in remnants after approximately 30 years. Bird composition also differed significantly between the planting types, with a generalist bird assemblage in woodlot plantings, and a bird assemblage that was associated with the shrub layer in ecological plantings.

Bird species richness responded positively to two measures of structural complexity (HCS4 and Habitat Hectares Condition Score). Age of the planting was also a positive predictor of bird species richness, as has been noted elsewhere (Martin et al. 2004, Kavanagh et al. 2007). Structural complexity in plantings increases with age (Munro et al. 2009), and particular features such as mature bark, leaf litter and logs appear. A positive response of birds to structural complexity has been noted elsewhere in remnant vegetation (Willson 1974, Roth 1976, Freemark and Merriam 1986, Cueto and Casenave 1999), and in a limited number of studies of revegetation in Australia (Rossi 2003) and overseas (Hinsley and Bellamy 2000, Twedt et al. 2002, Rotenberg 2007).

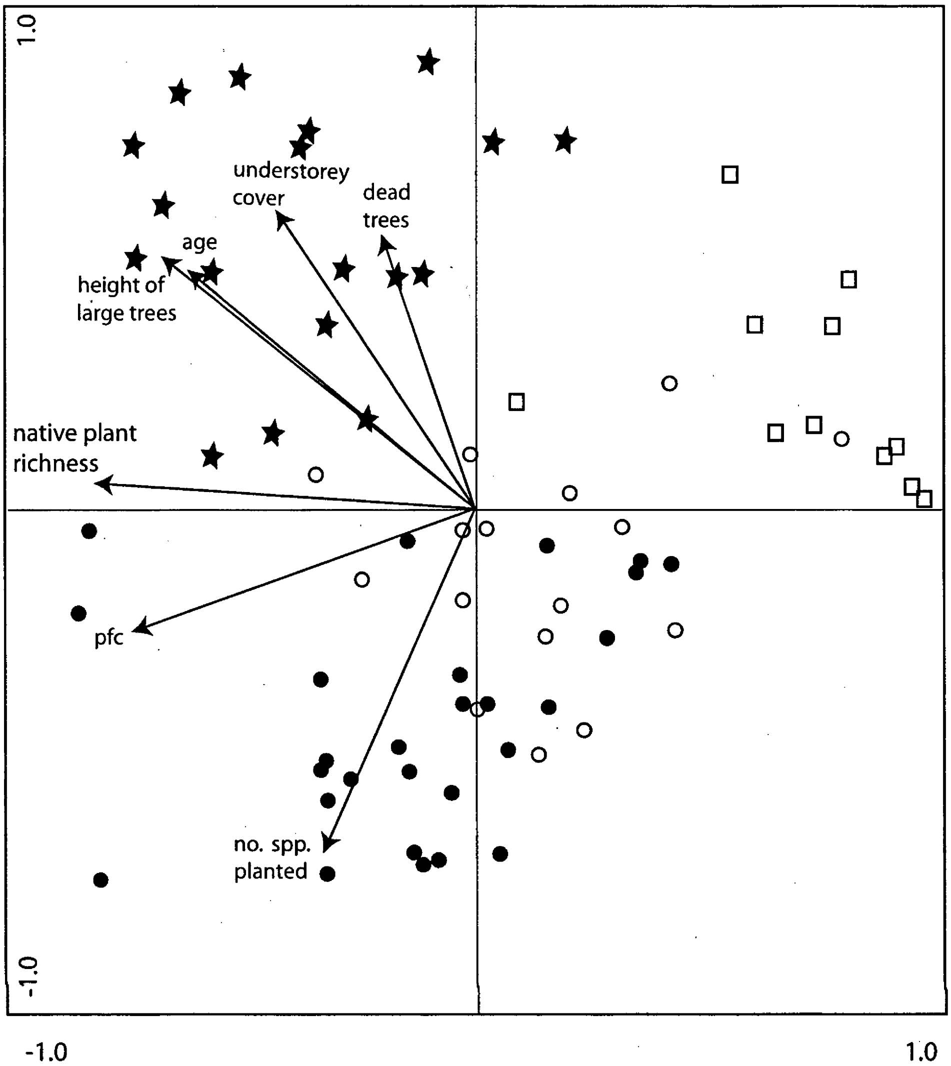

Figure 4-3 Canonical distribution of sites with significant environmental variables based on bird composition at sites, from the Redundancy Analysis.

Site types are: remnants (stars), ecological plantings (filled circles), woodlot plantings (open circles), paddocks (open triangles). Environmental variables that were significantly related to differences in species composition are displayed as vectors. Of note is that ecological plantings close to '*a.*' are old and particularly floristically rich and structurally diverse; those close to '*b.*' are very young plantings; the woodlot planting by '*c.*' was a very young plantation; the paddock site by '*d.*' contained many large old trees; and the remnant by '*e.*' was a degraded roadside.

Species richness of all birds and forest birds was lower where weed cover was higher. We found no other studies that examined the cover of weeds and its effect on bird species richness, although Barrett (2000) found that exotic birds were proportionally more abundant in exotic trees, while native birds were more abundant in native trees. Similarly, we found that two common exotic birds, the European Goldfinch and Common Myna, were associated with high weed cover.

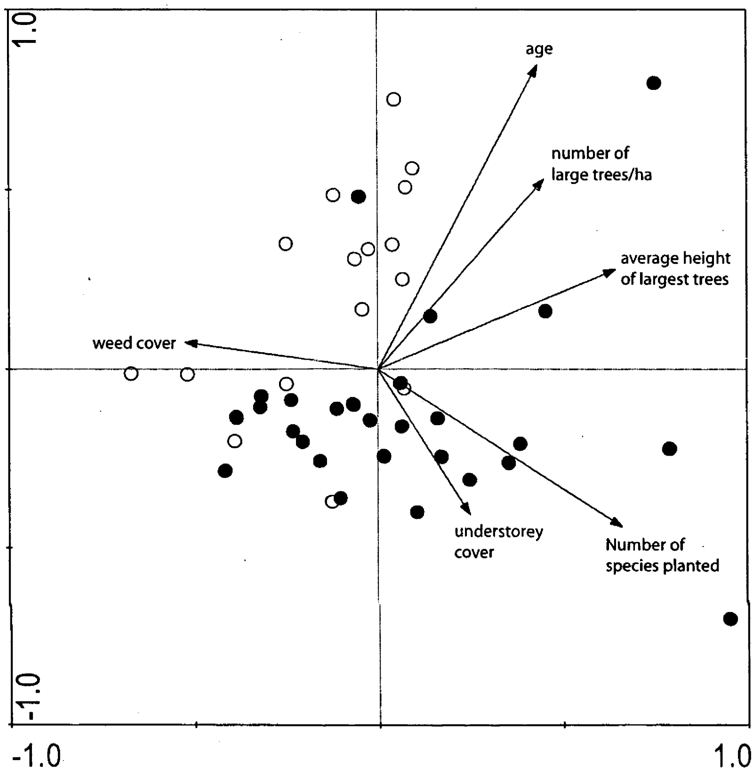

Figure 4-4 Redundancy analysis of bird composition in plantings only, displaying sites and environmental variables.
Open circles are woodlot plantings, filled circles are ecological plantings. Environmental variables that were significantly related to differences in species composition are displayed as vectors.

Table 4-8 Pairwise comparisons of site types from the redundancy analysis of bird composition.
All four site types are included in the analysis. Values are the F-ratio, with significance values indicated (*** is significant at P<0.001, ** is P=0.005).

Site type	Remnant	Ecological planting	Woodlot planting	Paddock
Remnant	-	4.383 ***	4.776 ***	11.920 ***
Ecological planting		-	2.529 **	10.504 ***
Woodlot planting			-	5.669 ***
Paddock				-

We predicted that bird richness would be greater in riparian locations, in larger patches, and where vegetation cover in the landscape was greater. Bird richness was indeed greater in riparian locations, and in larger patches, but was not higher where landscape cover was higher. In remnant vegetation, riparian locations can be more structurally complex than upland locations (Soderquist and Mac Nally 2000, Palmer and Bennett 2006), which may, in turn, increase bird richness. Greater bird richness in riparian areas than upland areas has been observed (Bentley and Catterall 1997, Fisher and Goldney 1997, Mac Nally et al. 2000, Woinarski et al. 2000, Palmer and Bennett 2006) although this is not necessarily a universal finding (Sabo et al. 2005). The positive relationship with patch size was not surprising because many local birds are area-sensitive (Department of Sustainability and Environment 2003a), and previous studies have found a positive correlation between bird richness and vegetation patch area for both remnants (McIntyre 1995, Kavanagh et al. 2007) and revegetation plantings (Kavanagh et al. 2007).

In a simple model of 'quality' versus quantity of vegetation, birds responded positively to 'quality' (condition compared to reference sites), but not to the amount of woody vegetation cover in the landscape (quantity). Selwood et al. (2009) found a similar importance of site-based features on breeding in birds, but little effect of landscape context. We appreciate there are significant limitations to our simple comparison of content versus context. Birds are likely to respond variably to the scale of vegetation cover in the landscape, connectivity was not measured explicitly, and the measure of 'quality' may not be closely related to some bird species' habitat requirements. In addition, the amount of vegetation cover in the landscape was generally low (average of 8 %; ¾ of sites had <10 %, and only 2 of 72 sites had greater than 30 % cover in 2.25 km radius around the site). Although our findings should be treated with some caution, they do suggest the value of establishing high quality plantings in our study region, where vegetation cover is generally low.

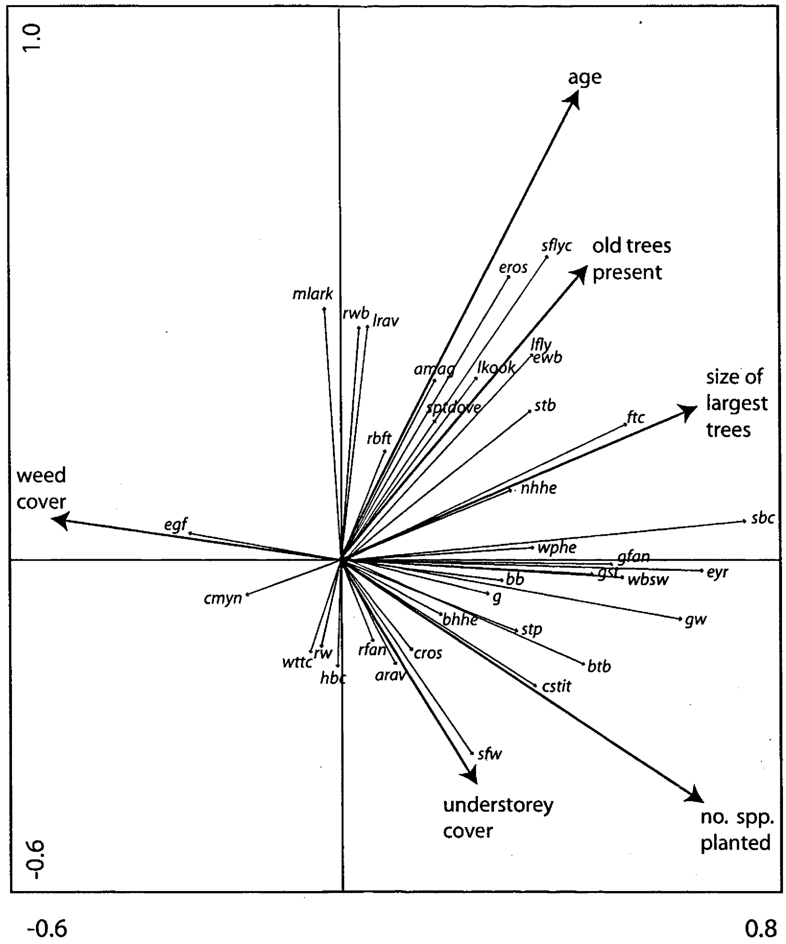

Figure 4-5 Redundancy analysis of bird composition in plantings only, displaying bird species and the environmental variables, that were significantly related to compositional differences between site types. Bird species with vector lengths less than 0.1 are not shown. Codes to bird names are given in the Appendix.

Arguably, the species composition of birds is of greater ecological relevance than species richness *per se*. The bird composition in all four site types was distinct, with significant differences between ecological plantings and woodlot plantings. Contrary to expectations, bird composition in ecological plantings was not more similar to remnants, but instead contained a collection of birds that responded to the number of species planted at a site, including many birds that use shrubby habitats. Remnant patches contained birds that preferred large old trees and dead trees, such as bark specialists and hollow users. The absence of bark foragers and forest specialists in revegetation has been previously noted (Woinarski 1979, Loyn et al. 2007, Rotenberg 2007). While some remnants and ecological plantings supported species of conservation concern, woodlot plantings were dominated by generalist birds (Christian et al. 1998, Kinross 2004) suggesting a lower biodiversity value for this planting type.

The composition of birds in plantings became more similar to that in remnants with age of planting. This is consistent with previous work in mine site revegetation (Nichols and Grant 2007). Notably, the composition of birds in ecological plantings was similar to that in woodlot plantings of twice the age. This indicates that ecological plantings may be ecologically valuable to the bird community sooner than woodlot plantings. This is a critical consideration in revegetation planning, considering the risks to biodiversity posed by substantial time lags in habitat provision (Vesk and Mac Nally 2006, Vesk et al. 2008).

4.7 Conclusion

Revegetation plantings can provide habitat for some bird species, with a succession of species occupying revegetation as it ages and acquires characteristics similar to remnant vegetation. We showed that bird species richness was higher in plantings of greater structural complexity, but did not respond to floristic richness. Birds species richness was higher in plantings that were more similar to reference remnant vegetation, in riparian locations, in larger patches, and where weed cover was lower. Ecological plantings contained a distinct bird community that included many shrub specialists, including some of conservation concern. In contrast, plantings low in structural complexity and floristic richness supported a generalist bird community, and did not cater for birds of conservation concern. Although bird species richness increased with age of planting and was similar to that in remnants after approximately 30 years, the distinct composition of birds in remnants suggested that plantings will not be a viable replacement of remnant vegetation in the medium term. Of key importance is that structurally complex plantings contributed better bird habitat, and in a shorter period of time, than structurally simple woodlot plantings.

4.8 Implications for Practice

- To maximise habitat value of plantings for birds, we suggest practitioners attempt to replicate the structural complexity of remnant vegetation as much as possible. Such plantings may be more valuable to the bird community sooner than simple tree plantings, which reduces time lags in habitat resources.
- Woodlot plantings still provide a degree of habitat provision for many birds and should be valued in landscapes. Birds would benefit if woodlot plantings were enhanced with shrubs.

- Planting in riparian areas, and planting as large an area as possible will increase bird species richness in plantings. Focussing planting efforts around existing old remnant trees also may help to increase bird species richness.
- Long-term success of restoration plantings as habitat for birds probably cannot be accurately assessed in plantings less than 20 to 30 years of age.

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4.11 Appendix

Full list of detected birds in the study, including the bird codes used in Figure 5, and the total number of detections during the study.

Bird code	Common name	Scientific name	Habitat guild	Total number of detections
amag	Australian Magpie	Gymnorhina tibicen	open country	172
arav	Australian Raven	Corvus coronoides	forest	9
bgth	Bassian Ground Thrush	Zoothera lunulata	forest	1
bb	Blackbird *	Turdus merula	forest	135
bche	Black-chinned Honeyeater	Melithreptus gularis	forest	1
bduck	Black Duck	Anas superciliosa	open country	2
bwp	Blue-winged Parrot	Neophema chrysostoma	forest	5
bf	Brown Falcon	Falco berigora	open country	1
bgos	Brown Goshawk	Accipiter fasciatus	forest	1
bhhe	Brown-headed Honeyeater	Melipthreptus brevirostris	forest	3
brcuckoo	Brush Cuckoo	Cuculus variolosus	forest	1
brwb	Brush Wattlebird	Anthochaera chrysoptera	forest	5
brtb	Buff-rumped Thornbill	Acanthiza reguloides	forest	1
btb	Brown Thornbill	Acanthiza pusilla	forest	153
csh	Collared Sparrow Hawk	Accipiter cirrhocephalus	forest	2
cstar	Common Starling *	Sturnus vulgaris	open country	129
cmyn	Common Myna *	Acridotheres tristis	open country	8
cstit	Crested Shriketit	Falcunculus frontatus	forest	3
cros	Crimson Rosella	Platycercus elegans	forest	45
eros	Eastern Rosella	Platycercus eximius	open country	47
esb	Eastern Spinebill	Acanthorhynchus tenuirostris	forest	4
ewb	Eastern Whipbird	Psophodes olivaceus	forest	5
eyr	Eastern Yellow Robin	Microeca leucophaea	forest	79
egf	European Goldfinch *	Carduelis carduelis	open country	36
ftc	Fan-tailed Cuckoo	Cuculus flabelliformis	forest	29
g	Galah	Cacutua roseicapilla	open country	8
gib	Glossy Ibis	Plegadis falcinellus	open country	1
gbut	Grey Butcherbird	Cracticus torquatus	forest	24
gfan	Grey Fantail	Rhipidura fuliginosa	forest	204
gst	Grey Shrike-thrush	Colluricincla harmonica	forest	153
gw	Golden Whistler	Pachycephala pectoralis	forest	120
hbc	Horsfield's Bronze Cuckoo	Chrysococcyx basalis	forest	22
lkook	Laughing Kookaburra	Dacelo novaeguineae	forest	34
lfly	Leaden Flycatcher	Myiagra rubecula	forest	3
lhe	Lewins Honeyeater	Meliphaga lewinii	forest	4
lc	Little Corella	Cacatua sanguinea	open country	3
lrav	Little Raven	Corvus mellori	open country	56
mlark	Magpie Lark	Grallina cyanoleuca	open country	35
mlap	Masked Lapwing	Vanellus miles	open country	2
mtb	Mistletoe Bird	Dicaeum hirundinaceum	forest	2
nhhe	New Holland Honeyeater	Phylidonyris novaehollandiae	forest	17

Bird code	Common name	Scientific name	Habitat guild	Total number of detections
nmin	Noisy Miner	Manorina melanocephala	open country	18
pcw	Pied Currawong	Strepera graculina	open country	1
rbft	Red-browed Finch	Neochmia temporalis	forest	8
rwb	Red Wattlebird	Anthochaera carunculata	forest	63
rfly	Restless Flycatcher	Myiagra inquieta	forest	1
rpip	Richard's Pipit	Anthus novaeseelandiae	open country	9
rft	Rufous Fantail	Rhipidura rufifrons	forest	6
rw	Rufous Whistler	Pachycephala rufiventris	forest	35
sking	Sacred Kingfisher	Todiramphus sanctus	forest	2
sflyc	Satin Flycatcher	Myiagra cyanoleuca	forest	9
sbc	Shining Bronze Cuckoo	Chrysococcyx lucidus	forest	36
seye	Silvereye	Zosterops lateralis	forest	23
spp	Spotted Pardalote	Pardalotus punctatus	forest	47
sptdove	Spotted Turtle-dove *	Streptopelia chinensis	open country	19
stp	Striated Pardalote	Pardalotus striatus	forest	11
stb	Striated Thornbill	Acanthiza lineata	forest	118
snib	Straw-necked Ibis	Threskiornis spinicollis	open country	3
sfw	Superb Fairy-wren	Malurus cyaneus	forest	175
tm	Tree Martin	Cecropis nigricans	forest	1
vsit	Varied Sitella	Daphoenositta chrysoptera	forest	3
wee	Weebill	Smicornis brevirostris	forest	1
wels	Welcome Swallow	Hirundo neoxena	open country	15
wib	White Ibis	Threskiornis molucca	open country	2
wbsw	White-browed Scrubwren	Sericornis frontalis	forest	129
wehe	White-eared Honeyeater	Lichenostomus leucotis	forest	56
wfh	White-faced Heron	Egretta novaehollandiae	open country	5
wnhe	White-naped Honeyeater	Melipthreptus lunatus	forest	13
wphe	White-plumed Honeyeater	Lichenostomus penicillatus	forest	19
wttc	White-throated Treecreeper	Climacteris leucophaea	forest	28
wag	Willie Wagtail	Rhipidura leucophrys	open country	11
wduck	Wood Duck or maned duck	Chenonetta jubata	open country	1
yfhe	Yellow-faced Honeyeater	Lichenostomus chrysops	forest	70
yrtb	Yellow-rumped Thornbill	Acanthiza chrysorrhoa	open country	2
ytbc	Yellow-tailed Black Cockatoo	Calyptorhynchus funereus	forest	1

Chapter 5: The effect of structural complexity on large mammal occurrence in revegetation

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The effect of structural complexity on large mammal occurrence in revegetation

5.1 Keywords

Floristic diversity; mammal occurrence; restoration; revegetation; vegetation structure.

5.2 Introduction

Worldwide, revegetation plantings have been established on cleared land to mitigate problems associated with land clearing and degradation (Eberbach 2003). However, information is limited on the extent to which plantings can provide habitat for fauna, and in what time frame (reviewed by Munro *et al.* 2007).

Revegetation plantings can differ substantially in their structural complexity and floristic richness, largely due to the mixture of plant species at establishment. In this study, we assessed the habitat value of plantings differing in structure and floristics for medium-sized and large mammals and we compared this to the habitat value of remnant vegetation and paddocks as the goal and starting point for vegetation restoration, respectively. We also investigated the effects of planting size, planting age, the presence of old remnant trees, and the amount of vegetation cover surrounding the site on mammal occurrence. Mammals are an important taxon in the context of revegetation plantings because they (1) have suffered severe declines in Australia (Short and Smith 1994), and (2) can have highly specific habitat requirements, e.g. for tree hollows, logs and understorey cover (McElhinny et al. 2006), which may be met in a planting only with careful prior planning.

5.3 Methods

Our study was conducted in West Gippsland, Victoria. Prior to clearing in the late 19th century, the region was dominated by structurally complex forest. Non-coastal woody vegetation cover now is only approximately 6 % (Department of Sustainability and Environment 2003a).

Numerous revegetation plantings have been established in the area. We identified two types of revegetation, reflecting broad differences in structural complexity and floristic richness (detailed in Munro et al. 2009): (1) '*ecological plantings*' were planted for ecosystem restoration purposes and were characterised by a diverse assemblage of tree, shrub and understorey species; and (2) '*woodlot plantings*' were typically planted with few species of primarily overstorey trees. We use these site types as surrogates for structural complexity and floristic richness.

We selected 72 patches of four vegetation types: ecological plantings (27 patches), woodlot plantings (16), remnants (18) and paddocks (11). Within each patch we established a 100 m transect, located centrally and selected to be representative of the patch.

Patches covered a wide range of sizes and ages. Both planting types ranged in size from 0.07 to 10.9 ha with one additional large woodlot planting of 96.5 ha. Remnants ranged in size from 0.1 to 27.5 ha with two additional large remnants of 43 and 528 ha. The age of plantings ranged from 2 to 26 years for both types. However, woodlot plantings were, on average, older than ecological plantings (mean of woodlot plantings was 13.6 ± 1.8 (s.e.) years, mean of ecological plantings was 8.1 ± 1.0 years, t_{41}=2.90, P=0.006). Approximately half the plantings and paddocks contained old remnant trees and half did not. The forest vegetation cover within a radius of 2.25 km around each transect was calculated using GIS to be between 2 and 57 %.

We counted mammals by two methods. First, we conducted 20 minute spotlight searches along each transect, between one hour after sunset and 2 am, twice, in February 2006. All mammals sighted within 50 m of the transect (visually estimated distance), and within the patch, were recorded. Second, we recorded incidental sightings of mammals, and wombat holes in use as evidence of presence. Incidental observations were restricted to the same transects, and were taken during other surveys (vegetation and bird surveys; Munro et al. submitted). We consider that the observation effort was similar between sites. Due to lack of data and many zero counts, we combined our spotlight and incidental observation data, and converted the data to presence/absence data for each species.

Three species had sufficient counts to enable individual statistical modelling of their presence (Common Ringtail Possum, Common Wombat and Red Fox; see Fig. 5.1 for scientific names). The explanatory variables initially considered were site type, age, area, the presence or absence of old remnant trees, and surrounding vegetation cover. We fitted generalised linear models, with a binomial error distribution, to presence/absence data as a function of the above explanatory variables. We used an iterative, manual, forwards and backwards variable selection process. We log-transformed age and area. To remove problems of unrepresented data (e.g. paddocks and remnants did not have an 'age'), we assigned the age of paddocks and remnants the mean natural log(age) of the plantings, and the area of paddocks was assigned the mean natural log(area) of plantings and remnants. This effectively eliminated the effect of site type where a particular variable was undefined.

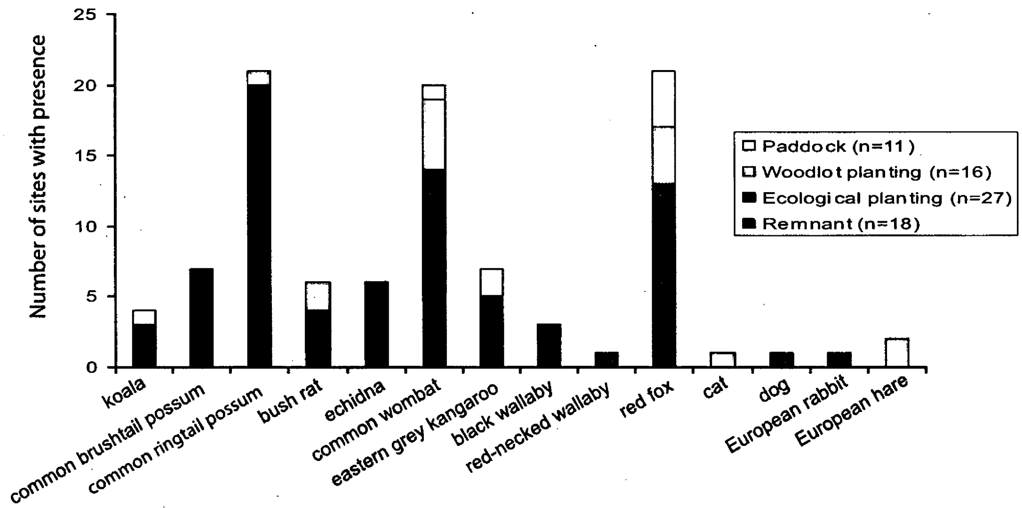

Figure 5-1 The number of sites with a presence of each species observed, shown by site type.
The scientific name of each mammal from left to right is: Phascolarctos cinereus, Trichosurus vulpecula, Pseudocheirus peregrinus, Rattus fuscipes, Tachyglossus aculeatus, Vombatus ursinus, Macropus giganteus, Wallabia bicolour, M. rufogriseus, Vulpes vulpes*, Felis catus*, Canis lupus familiaris*, Oryctolagus cuniculus*, Lepus europaeus*, (* denotes an introduced species).

We further analysed data for the Common Ringtail Possum by combining site type with the presence or absence of old trees, thereby creating seven new categories (paddocks with and without old trees, woodlot plantings with and without old trees, ecological plantings with and without old trees, and remnants). We removed three categories where no individuals were observed, and modelled Ringtail Possum presence at the remaining four site categories.

5.4 Results

We observed five introduced species, three native arboreal marsupials, and six native terrestrial species (Fig. 5.1). Most species had very low detection rates. Remnants and ecological plantings contained most species of native mammals (Fig. 5.1). In paddocks, by contrast, we observed only two species (Common Wombat and Red Fox). The Common Brushtail Possum and Black Wallaby occurred only in remnants and ecological plantings containing old remnant trees.

There were enough sites with a presence of the Common Ringtail Possum, Common Wombat and Red Fox to enable individual modelling. Models for the Common Wombat and Red Fox did not contain any significant explanatory variables at $P<0.1$.

Presence of the Common Ringtail Possum was significantly related to site type ($P<0.001$) and the presence of old remnant trees ($P=0.004$). Patch size, age and vegetation cover in the landscape were not significantly related to Ringtail Possum occurrence. We found no Ringtail Possums in paddock sites, and none in woodlot plantings without old remnant trees. After we removed the site categories with no Ringtail Possums, modelling showed there were significant differences between 'site type - presence/absence of old tree' categories for the overall model ($P=0.001$). Examination of pair-wise comparisons showed that the Ringtail Possum was more likely to occur in ecological plantings with old trees than in ecological plantings without old trees ($P=0.022$), and more likely to occur in remnants than in ecological plantings without old trees ($P<0.001$; Fig. 5.2). Although we found a weak trend for a lower probability of occurrence of the Ringtail Possum in woodlot plantings with old trees compared to ecological plantings with old trees, this was not statistically significant ($P=0.32$). There was no significant difference between ecological plantings with old trees and remnants ($P=0.39$), or between woodlot plantings with old trees and remnants ($P=0.11$).

We observed the Common Wombat and the Bush Rat in revegetation plantings as young as four years (no old trees), and the three arboreal marsupial species in plantings as young as eight years (some with old trees, some without). Several mammal species (Common Ringtail Possum, Bush Rat, Koala and Eastern Grey Kangaroo) occurred in revegetation patches that were at least six kilometres from substantial patches of remnant vegetation. The intervening land contained small, isolated patches of roadside vegetation, very few paddock trees, and several house gardens.

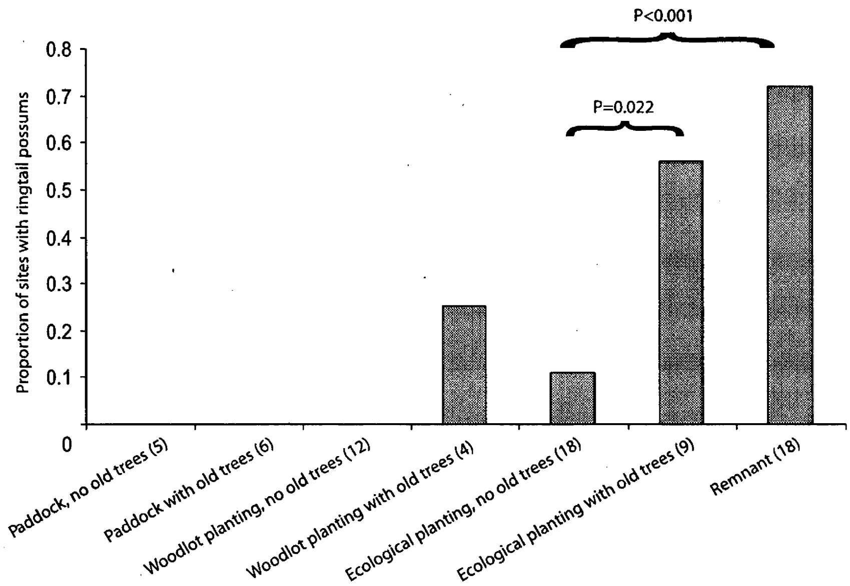

Figure 5-2 The proportion of sites with a presence of ringtail possums. Significant differences ($P < 0.05$) were found only between ecological plantings without old trees and ecological plantings with old trees, and between ecological plantings without old trees and remnants.

5.5 Discussion

Revegetation was used by three arboreal and six terrestrial native mammal species. All native species used both remnant vegetation and revegetation. Ecological plantings characterised by greater structural complexity and floristic richness appeared to be preferred by several native mammals (Fig. 5.1). Although data were insufficient for formal analyses, these trends suggest that the habitat value of plantings may be enhanced by including a large diversity of plants at establishment.

Previous studies have shown arboreal marsupials are uncommon in revegetation (Kavanagh et al. 2005, Cunningham et al. 2007) or regrowth forest (Kutt 1995), compared to remnant vegetation, but none of these studies considered the presence of old remnant trees. Most arboreal mammals require old trees with hollows for nesting (Gibbons and Lindenmayer 2002). We found that remnant trees were a valuable resource especially for the Common Ringtail Possum. Detection rates were also broadly similar in ecological plantings with old trees (3/9 sites) and remnants (4/18 sites) for the Common Brushtail Possum (Fig. 5.1).

Key ecological resources such as tree hollows and large logs can take decades to establish in revegetation, causing time-lags in habitat suitability for many species. This is of particular concern where plantings are used as offsets for vegetation clearing (Cunningham et al. 2007, Vesk et al. 2007). Our findings suggest that young plantings will have greater habitat value for several mammal species if old trees with hollows are incorporated into their design.

We conclude that many native mammals can use revegetation plantings, at least temporarily, and within only a few years of establishment. Use by mammals may be increased if plantings contain diverse plant species and old remnant trees.

5.6 Acknowledgements

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Chapter 6: Is the Landscape Function Analysis a useful tool for measuring the success of restoration plantings?

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Is the Landscape Function Analysis a useful tool for measuring the success of restoration plantings?

6.1 Abstract

Landscape Function Analysis (LFA) (sensu Tongway and Hindley 2004) is a tool that can be used to rapidly assess the functional attributes of soil stability, water infiltration and nutrient cycling. We compared these attributes, as determined by the LFA, between revegetation plantings, remnants and cleared agricultural land (paddocks). In an agricultural landscape in south-eastern Australia, we differentiated between 'woodlot plantings' (planted with overstorey eucalypts only) and 'ecological plantings' (planted with many indigenous species of trees and shrubs). Remnants and paddocks formed reference sites indicating the goal and starting point of restoration, respectively. Sites of remnant vegetation scored highest for all three functional attributes, whereas paddocks had high scores for soil stability, but low scores for infiltration and nutrient cycling. Contrary to our expectations, soil stability, infiltration and nutrient cycling did not differ between ecological plantings and woodlot plantings. Although LFA provided an overview of some key functional differences between site types, it may be too coarse as a tool to measure restoration success. Specifically, the three functions considered by the LFA were strongly influenced by a single variable relating to perennial vegetation cover, but were essentially unaffected by more subtle differences between site types. We also caution that Landscape Function Analysis derives surrogates of very basic functional attributes, which may not be sufficiently sensitive to accurately reflect more complex ecological functions such as habitat provision for wildlife.

6.2 Keywords

Species diversity – ecosystem function relationship, Landscape Function Analysis, revegetation, restoration, ecosystem function.

6.3 Introduction

Quantifying the ecosystem function of landscapes or sites is a major challenge, especially in a restoration context (Bengtsson 1998). A critical measure of successful restoration is that a restored site is functioning adequately (Society for Ecological Restoration International Science and Policy Working Group 2004). However, despite the general recognition of the importance of restoring ecosystem function, practical methods for determining restoration success are often lacking (Bengtsson 1998).

Landscape Function Analysis is a rapid assessment technique that has been used in rehabilitation and restoration sites throughout the world. Initially developed by Tongway and Hindley (2004) to monitor the functional state of rehabilitating mine sites in the Australian rangelands, use of the method has since expanded to monitor rangeland condition under other threats such as grazing (McIntyre and Tongway 2005), as well as other land uses such as orchards and tree crops, and other vegetation types such as forests (Koch and Hobbs 2007), woodlands, grasslands (Rezaei et al. 2006) and savannah (Tongway and Hindley 2004). The method has not been used in revegetation of previously cleared agricultural land, with the exception of case study on 14 metres of a single revegetation patch in south-eastern New South Wales, Australia (Leguédois et al. 2008).

To mitigate the negative effects of past land clearing, vegetation has been deliberately replanted in many parts of the world (particularly the 'New World'), often as numerous small patches on private and public land (Rey Benayas et al. 2009). In Australia, these patches are called 'revegetation'. The ecological success of revegetation can be judged on three key attributes – its structure, its composition and its function (Noss 1990, Ruiz-Jaen and Aide 2005a). 'Structure' can be defined as the horizontal heterogeneity and vertical complexity of the vegetation, 'composition' as the community of plants, animals and other taxa, and 'function' as the status of biotic and abiotic processes (Bengtsson 1998). Vegetation structural complexity and the composition of flora and fauna in revegetation plantings have been reasonably well studied (Marín-Spiotta et al. 2007, Munro et al. 2007). The ecological function of revegetation, however, has been the topic of almost no research, despite recognition of its importance (Armstrong 1993, Pimentel and Kounang 1998).

Ecological function has been defined in many ways. At the most basic level, function relates to the flows of water and nutrients through a site (Bengtsson 1998, Srivastava and Vellend 2005). At a higher level, and often at a larger spatial scale, there are more complex ecological functions such as (for revegetation plantings) lowering of water tables to prevent or reduce salinity (Hatton and Nulsen 1999), reducing wind and water erosion (Bird et al. 1992), improving stream-bank stabilisation and water quality (Vought et al. 1995), and providing habitat and connectivity for plants and animals (Rosenberg et al. 1997, Munro et al. 2007). Other measures of ecological function include the productivity or rate of biomass accumulation of a site (Henry et al. 2001, Erskine et al. 2006, Foster et al. 2007), species interactions across functional traits and trophic levels (Palmer et al. 1997, Walker et al. 1999), pollination processes (Balvanera et al. 2005), and carbon storage (Balvanera et al. 2005).

We trialled the use of the Landscape Function Analysis (LFA) to assess the development of ecosystem function in revegetation, particularly in relation to the basic ecological functions of soil stability, water infiltration and nutrient cycling. LFA has been field tested by Tongway and Hindley (Tongway and Hindley 2003), who found that LFA-derived indices of function were highly correlated with detailed measures of those same functions for mine sites in a variety of vegetation types (Tongway and Hindley 2003).

We compared two types of revegetation plantings differing in their plant species diversity and similarity to remnant vegetation. We were particularly interested in whether the LFA method could detect differences in the functions of soil stability, water infiltration and nutrient cycling between these planting types. In addition to these two types of plantings, we also assessed remnant vegetation and grazed pastures (paddocks) as reference points. We expected that paddocks, which were cleared approximately 150 ago, sown with exotic pasture grasses, fertilised and grazed for at least 100 years, would have low measures of function according to the LFA. We also predicted that revegetation plantings that were more similar to remnants, and with greater plant diversity, would score higher for measures of function than low-diversity plantings, but still fall short of high condition remnant vegetation.

6.4 Methods

6.4.1 Study area

We conducted our study in West Gippsland, south-eastern Australia. Prior to clearing in the mid-19th century for agriculture, the region was dominated by structurally complex forest (Korumburra and District Historical Society Inc. 1998). Current non-coastal remnant woody vegetation is approximately 6% cover (Department of Sustainability and Environment 2003a), and the area is dominated by dairy farming. All sites in our study were in non-coastal areas.

The study area has a Mediterranean climate, with predominantly winter rainfall. Rainfall is approximately one metre per year, and the area is characterised by a slow overland flow of water, with many small waterways. The topography of the area is low, steep hills above a flat coastal plain, and the soils are young and fertile by Australian standards. Erosion is relatively common on cleared land in the forms of tunnel erosion and land slips.

Various types of land degradation, including erosion and biodiversity loss, have prompted landholders in the study area to revegetate areas of previously cleared land. We identified two types of revegetation: (1) '*ecological plantings*' which were planted for ecosystem restoration purposes and were characterised by a diverse assemblage of tree, shrub and understorey species; and, (2) '*woodlot plantings*' which were typically planted with low plant species richness of primarily overstorey species. Both types of plantings were established with predominantly local, native vegetation with an overstorey of *Eucalyptus* trees. Ecological plantings were more similar to remnants in a number of attributes, than woodlot plantings. We describe the vegetation composition and structure of these planting types in detail in Munro et al. (2009).

6.4.2 Site selection

We selected patches of four vegetation types using maps and aerial photographs, on-ground searching, and by consultation with a local Landcare Network (Fig. 6.1). We selected a single site (100 m transect) within each patch, located approximately centrally and selected to be representative of the patch. Our study was comprised of 72 sites: ecological plantings (27 patches), woodlot plantings (16), remnants of native forest (18), and paddocks (11).

We selected patches with a range of sizes and ages. Both planting types ranged in size from 0.07 to 10.9 ha with one additional large woodlot planting of 96.5 ha. Most remnants ranged in size from 0.1 to 27.5 ha with two additional large remnants of 43 and 528 ha. The age of plantings ranged from 2 to 26 years for both woodlot and ecological plantings. Woodlot plantings were, on average, older than ecological plantings (mean of woodlot plantings was 13.6 ± 1.8 (s.e.) years; mean of ecological plantings was 8.1 ± 1.0 years, t_{41}=2.90, P=0.006). Half the sites within each type were in riparian locations, and half were not; approximately half the plantings and paddocks contained old remnant trees and half did not.

a) b)

c) d)

Figure 6-1 Photographs of site types in our study.
a) paddock, b) woodlot planting, c) ecological planting, and d) remnant.

Remnant sites were the highest condition patches available in our study area, but had been subject to low levels of timber extraction and suffer some weed invasion. Remnants were not grazed. Plantings of both types were generally not grazed, although several experienced cattle incursions during the study, and one woodlot planting was grazed by sheep. Several plantings were established on sites of previous erosion (creek bank erosion, landslips and tunnel erosion), or on steep slopes which may be susceptible to erosion. All paddocks were continuously grazed.

For each site, we recorded patch size, age, patch shape (long or round), the presence/absence of remnant old trees, the area of surrounding tree cover within 2.25 km, slope (flat, hilly, steep), and riparian/non-riparian location.

Table 6-1 Description of LFA indices recorded per plot.

Indicators	Purpose of indicator	Method of measurement	Scoring method	Indices that include this indicator
Rainsplash protection	The protection of soil surface from raindrops, which may cause erosion	The cover to 0.5 m height of perennial vegetation, wood and rocks.	5 classes (1 to 5) indicating 1% or less cover to >50% cover	Stability
Perennial vegetation cover	Contribution of below-ground biomass of perennial vegetation to nutrient cycling	'Basal cover' of perennial grass or density of canopy cover of trees and shrubs	4 classes (1 to 4) indicating 1% or less cover to >20% cover	Infiltration Nutrient cycling
Leaf litter	Plant litter accumulation is related to effectiveness of decomposition/incorporation processes	Amount, origin and degree of decomposition of plant litter, including annual and ephemeral plants	3 components: 10 classes (1 to 10) indicating cover and thickness, plus local origin (scoring 1.5) or transported (1), plus degree of decomposition (4 levels) – nil (1), slight (1.33), moderate (1.66), extensive (2)	Stability Infiltration Nutrient cycling
Cryptogam cover	Indicator of soil surface stability and nutrient cycling	Cover of cryptogams on soil surface	5 classes (0 to 4) where 0 is no stable crust present, 1 is 1% or less, to 4 of >50% cover	Stability Nutrient cycling
Crust brokenness	Indicator of soil stability. A broken crust can be more susceptible to erosion	Level of crust brokenness	5 classes (0 to 4) where 0 is no crust present, 1 is crust extensively broken, to 4 of crust intact	Stability
Soil erosion	The presence and severity of erosion can indicate instability	Type and severity of recent soil erosion	4 classes (1 to 4) of severity (severe to insignificant)	Stability
Deposited material	The presence of deposited material indicates erosion upslope	Amount of material present	4 classes (1 to 4) of amount (extensive to none)	Stability
Soil surface roughness	Rough surfaces have the capacity to capture and retain mobile resources such as water, soil, organic matter	Depth of depressions	5 classes (1 to 5) indicating smooth to very rough (deep depressions)	Infiltration Nutrient cycling
Resistance to disturbance	Resistance to erosion by wind, water or trampling	The ease with which soil can be penetrated by object (finger or knife)	5 classes (1 to 5): loose sand (10), easily broken (6.6), moderately hard (3.3), very hard (1), non-brittle (6.6)	Stability Infiltration
Slake test	Stability of natural soil fragments to rapid wetting. Stable soil fragments maintain cohesion when wet	Dry fragment in water to observe intactness	5 classes (0 to 4), where 0 is not applicable, 1 is very unstable to 4 of very stable.	Stability Infiltration
Soil texture	Permeability of the soil	Moist bolus test of texture of surface soil	4 classes (1 to 4) from very slow infiltration rate (clay) to high infiltration rate (sand)	Infiltration

6.4.3 Measurement of structure and floristics

We measured several structural and floristic attributes at each site: total plant species richness, overstorey tree species richness, shrub cover, and cover of ground layer, grass, leaf litter and logs (Munro et al. 2009).

6.4.4 Measurement of function

We used Landscape Function Analysis developed by Tongway and Hindley (2004) to quantify resource regulation at each site. Landscape Function Analysis involves collection of eleven soil surface indices at multiple plots within a given site (Table 6.1). Our selection of data plots differed slightly from that of Tongway and Hindley (2004) because the structure of the ground surface and vegetation was considered relatively homogeneous, with little or no patchiness. Hence, a random stratified design was used for selection of plot location. Each site consisted of a 100 m transect, along which we recorded vegetation characteristics. At 25 and 75 m, we established two cross-transects which followed the direction of water flow downhill. Each cross-transect was 20 m long, with 10 m either side of the main transect. We divided each cross-transect into four 5 m segments. We placed a 1 m sampling plot randomly within each 5 m segment. Thus, we had eight replicate plots within each site. At each sampling plot, we assessed the eleven indices of the LFA (Table 6.1). These were averaged over the eight replicates to obtain a single set of eleven different indices for each site. Each index examined the activity of a surface process. We created composite indices from these eleven indices at each plot for soil stability, infiltration and nutrient cycling. Table 6.2 gives a description of how the composite indices were created, and summarises the rationale for these indices (for details see Tongway and Hindley (2004)).

6.4.5 Statistical Analyses

We compared the eleven independent indices of the LFA between site types using analysis of variance. We also conducted regression analyses on the three composite indices of the LFA. We examined these composite indices as a function of the measured site variables (size, age, plant species richness etc. as listed above). We used forward stepwise linear regression analysis to build models. We forced the models to retain site type, and selected the model with the lowest value for the Akaike Information Criterion (Akaike 1974) indicating the most parsimonious model.

Table 6-2 Computation of the composite indices.

Composite indices	Individual indices that make up composite index	How computed	Interpretation
Soil stability	Rainsplash protection Litter cover (class score only) Cryptogam cover Crust brokenness Erosion Deposited materials Resistance to disturbance Slake test	Sum of scores. If all individual indices are present, this ranges from 8 to 40	The ability of the soil to withstand erosive forces and to reform after disturbance
Infiltration	Perennial vegetation cover Litter cover (class score x origin score x decomposition score) Surface roughness Resistance to disturbance Slake test Soil texture	Sum of scores. If all individual indices are present, this ranges from 6 to 57	How the soil partitions rainfall into soil-water (available for plants) and runoff water which is lost from the system, sometimes transporting materials with it
Nutrient cycling	Perennial vegetation cover Litter cover (class score x origin score x decomposition score) Cryptogam cover Surface roughness	Sum of scores. If all individual indices are present, this ranges from 4 to 43	How efficiently organic matter is cycled back into the soil

6.5 Results

We summarise the results of analysis of variance for the individual indices by site type in Table 6.3. Three of the eleven indices were highly significantly different ($P<0.001$) between our site types. These were rainsplash protection, perennial vegetation cover, and leaf litter decomposition. Cryptogam cover, resistance to disturbance, slake test and soil texture also were significantly different between site types (Table 6.3). The generally high values for crust brokenness, soil erosion and deposition, and for the slake test indicated that the soils under all four site types were fairly stable.

Table 6-3 Means and associated standard errors of the individual components of the LFA,
with P-value from analysis of variance comparing site types. The F-ratio is presented. Note there are 12 indices listed below, rather than 11, because leaf litter cover and leaf litter incorporation are two components of the one index.

LFA components	Paddock	Woodlot planting	Ecological planting	Remnant	$F_{3,68}$	P-value
Rainsplash protection	3.9 ± 0.2	2.2 ± 0.3	1.9 ± 0.2	3.1 ± 0.2	15.31	<0.001
Perennial vegetation cover	1.4 ± 0.2	3.2 ± 0.2	3.4 ± 0.1	3.7 ± 0.1	35.89	<0.001
Leaf litter cover	5.2 ± 0.2	5.2 ± 0.3	5.0 ± 0.2	5.7 ± 0.1	1.94	0.132
Leaf litter incorporation	1.14 ± 0.04	1.48 ± 0.05	1.42 ± 0.04	1.63 ± 0.03	18.09	<0.001
Cryptogam cover	1.1 ± 0.1	1.6 ± 0.2	1.7 ± 0.1	1.9 ± 0.2	3.80	0.015
Crust brokenness	3.2 ± 0.2	2.9 ± 0.1	2.8 ± 0.1	3.0 ± 0.2	1.02	0.389
Soil erosion severity	3.6 ± 0.2	3.7 ± 0.1	3.6 ± 0.1	4.0 ± 0.0	2.19	0.097
Deposited material	3.7 ± 0.1	3.6 ± 0.1	3.5 ± 0.1	3.8 ± 0.1	2.01	0.120
Soil surface roughness	3.8 ± 0.1	3.6 ± 0.1	3.6 ± 0.1	3.8 ± 0.1	1.33	0.272
Resistance	1.8 ± 0.5	2.3 ± 0.2	2.7 ± 0.2	3.1 ± 0.2	3.53	0.019
Slake test	3.9 ± 0.1	4.0 ± 0	4.0 ± 0	4.0 ± 0	3.63	0.017
Texture	2.6 ± 0.1	2.5 ± 0.1	2.5 ± 0.1	3.0 ± 0.1	3.62	0.017

6.5.1 Stability

The stability index was significantly higher in remnants and paddocks than in both planting types (Table 6.4, Fig. 6.2). We found no difference in stability between ecological plantings and woodlot plantings. Stability was higher where leaf litter cover was lower. However, this result appeared to have been influenced by paddock sites where stability was generally high but litter cover was low. Stability did not increase with age of planting (Table 6.4).

When examining the individual indices that comprised the composite index of stability, it appeared that differences in stability between site types were strongly influenced by rainsplash protection (cover less than 0.5 m in height, data not shown). To test this, we included a single value for rainsplash protection (the mean of 2.6) for all sites and re-ran the model of stability as a function of site type. We found that stability was no longer significantly different between sites, indicating a dominant influence of rainsplash protection on differences in the index of stability in our models.

6.5.2 Infiltration

We found no difference in infiltration between woodlot plantings and ecological plantings (Table 6.4, Fig. 6.2). However, there was significantly less infiltration in paddocks, and significantly more in remnants (Table 6.4). We found greatest infiltration where grass cover was low, and on flat land. The presence of grass cover in the model appeared to be strongly influenced by paddocks, which had full grass cover, and low infiltration. As with stability, we examined the individual indices that comprised the composite index of infiltration, and found that the differences in infiltration between site types appeared to be dominated by the sub-index of perennial vegetation cover.

6.5.3 Nutrient cycling

Nutrient cycling between ecological plantings and woodlot plantings did not differ significantly (Table 6.4). We found significantly lower nutrient cycling in paddocks, and significantly higher nutrient cycling in remnants, compared to plantings (Table 6.4, Fig. 6.2). Nutrient cycling also was greater where grass cover was lower, and on flat land. We found a non-significant trend toward greater nutrient cycling in older sites ($P= 0.06$). Nutrient cycling in both planting types approached that of remnants within 10 years (data not shown). Similar to infiltration, the difference in nutrient cycling between site types was substantially influenced by the sub-index of perennial vegetation cover.

We found that the size and shape of patches, presence of old remnant trees, surrounding vegetation cover, riparian or non-riparian location, vegetation richness, shrub cover and understorey cover were not significant variables in any of the models we developed.

Table 6-4 Final regression models of composite indices of the LFA.
Note that in the models the natural logarithm of age for remnants and paddocks was standardised to the mean natural log of age of the plantings. Models were developed using the Akaike information Criterion to find the most parsimonious model. Therefore, some explanatory variables are included which have a P-value greater than 0.05.

Response	Adjusted R-squared	Variables in model	Parameter estimates	Standard Error	P-value
Stability	0.219	Intercept (including ecological plantings)	68.56	1.60	<0.001
		Type – paddock	6.26	2.55	0.017
		Type – remnants	8.23	2.11	<0.001
		Type – woodlot plantings	1.79	2.11	0.401
		Litter cover (%)	-0.08	0.03	0.022
Infiltration	0.466	Intercept (including ecological plantings, flat topography)	57.66	2.79	<0.001
		Type – paddock	-6.29	2.16	0.005
		Type – remnants	6.67	1.96	0.001
		Type – woodlot plantings	0.24	1.79	0.895
		Grass cover (%)	-0.08	0.03	0.008
		Site topography – gentle slopes	-4.48	2.18	0.043
		Site topography – moderate slopes	-6.55	2.03	0.002
		Site topography – steep slopes	-6.31	2.68	0.022
Nutrient Cycling	0.509	Intercept (including ecological plantings)	49.80	5.40	<0.001
		Type – paddock	-8.78	2.69	0.002
		Type – remnants	8.23	2.42	0.001
		Type – woodlot plantings	-0.51	2.36	0.828
		Ln (age)	3.47	1.82	0.061
		Grass cover (%)	-0.08	0.03	0.024
		Site topography – gentle slopes	-4.97	2.71	0.072
		Site topography – moderate slopes	-6.51	2.56	0.013
		Site topography – steep slopes	-5.65	3.30	0.092

6.6 Discussion

Contrary to expectations, LFA did not distinguish between woodlot plantings and ecological plantings in the ecosystem functions measured. We expected the soil stability in paddocks to be lower than that of the plantings, but our data revealed the opposite was the case.

If we were to use the LFA as a measure of success of revegetation, we would suggest that revegetation was generally functioning better than paddocks, and not as well as remnants, but the lack of age as a significant variable in the models suggested that plantings were not on a trajectory toward that of remnants (within the 26 year time-frame of our study).

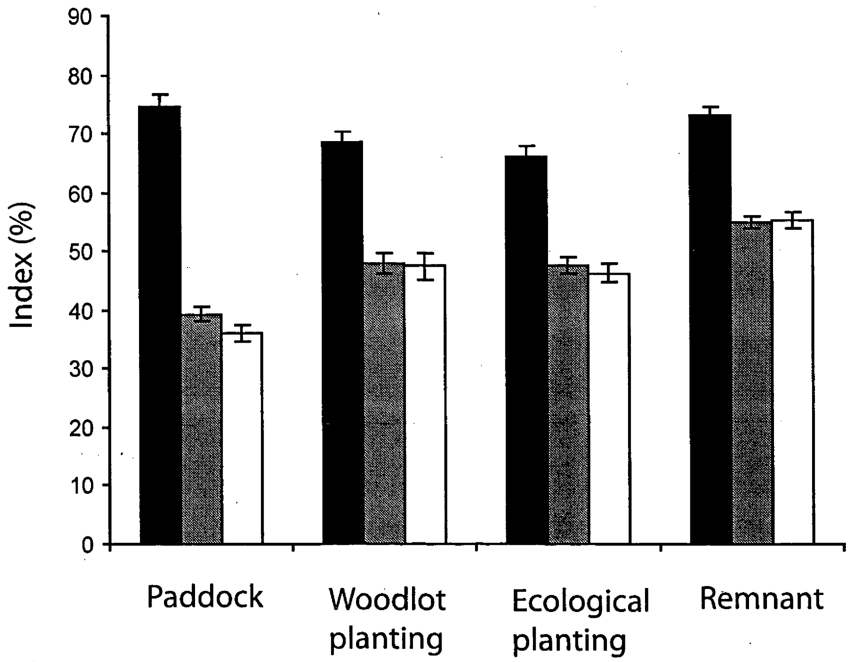

Figure 6-2 The LFA scores (converted to percentages) of soil stability, water infiltration and nutrient cycling in our four site types.
Soil stability (black bars), water infiltration (grey bars), nutrient cycling (white bars) with standard errors indicated. For significance values between site types, see Table 6.4.

Landscape Function Analysis has been used extensively to monitor the rehabilitation of mine sites (Tongway and Hindley 2004). In mine site rehabilitation, the starting reference condition is usually highly disturbed soils devoid of vegetation. The LFA scores for these sites have been shown to increase rapidly from a very low level to one similar to reference remnant sites within a few years (Tongway and Hindley 2004). In our study, paddocks were assumed to be the most degraded sites, with revegetation plantings on a trajectory toward the condition of remnants. However, it appears that paddocks in our study were not of a sufficiently degraded state for our study to benefit from the information provided by the LFA (see Fig. 6.3 on the S curve of increasing function with time). This is despite almost complete clearing of the original forest, 150 years of intensive agriculture, establishment of exotic pasture grasses and continuous grazing by cattle. McIntyre and Tongway (2005) also found that the stability index of the LFA did not decline significantly with grazing until a heavily grazed state was achieved.

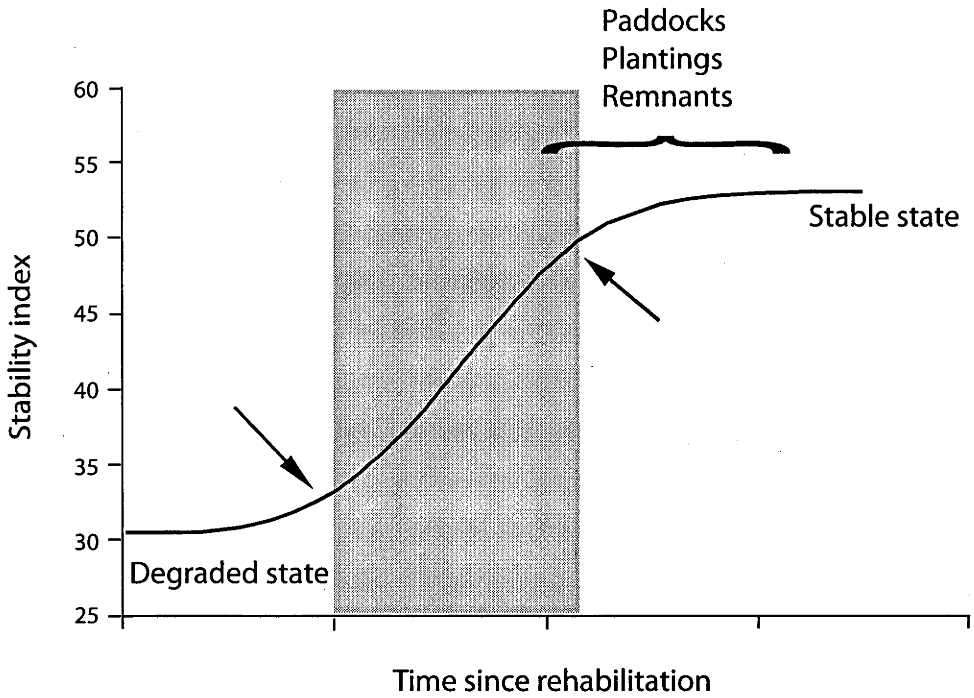

Figure 6-3 Diagram of LFA function curve, using soil stability as an example. The function curve (sensu Tongway and Hindley, 2004) indicates an increase in function from a degraded state to a stable state over time since rehabilitation. The arrows indicate thresholds or points of change from the degraded and stable states. The grey section is the optimal range of states for the LFA methodology. We suggest that our sites occupy the bracketed area, where the indices of stability, infiltration and nutrient cycling are high for all site types, and where LFA is not a very sensitive method.

The differences in the three composite indices of the LFA between our site types were each dominated by a single sub-index – rainsplash protection (low vegetation cover) in the case of stability, and perennial vegetation cover in the case of both infiltration and nutrient cycling. These two subindices reflect an obvious vegetation difference between paddocks, plantings and remnants. There was, however, no difference between ecological and woodlot plantings, despite the presence and cover of understorey and midstorey being part of the original selection criteria for distinguishing ecological and woodlot plantings. We suggest that either: (1) the soil structure and processes such as water infiltration and nutrient cycling were not significantly different between site types, possibly reflecting a similarity in function across our study region, (2) the LFA method is too coarse to detect differences in these functions in our context, or (3) the functions of soil stability, infiltration and nutrient cycling may take much longer than 26 years to develop a detectable difference between planting types, or to be on a trajectory with remnants. In the case of the last point, it may take centuries, rather than a few decades, for such functions to fully develop.

The LFA index of soil stability was high for paddocks, indicating a stable state. However, the region suffers from extensive erosion (tunnel erosion, landslips and gully erosion) on the cleared land. We therefore suggest that soil stability most likely is, in fact, lower in paddocks than in remnants (where erosion is rare). In our study, the LFA method did not reveal a discernable difference in soil stability between paddocks and remnants, which is contrary to the extensive problems with erosion on cleared land in the region. Several plantings in our study were established on sites of previous erosion. However, erosion and its counterpart, soil deposition, were not significantly different between site types, suggesting the previous erosion was not detected by the LFA method.

One of the reasons for conducting the current study was the concern that woodlot plantings, in particular, would not provide the soil stability initially intended by establishing a planting. The ground cover of woodlot plantings tends to be a mixture of bare ground, leaf litter and some patches of grass, which could facilitate rapid overland flow of water, resulting in erosion, rather than preventing it. According to our results, ecological plantings were not better at preventing erosion than woodlot plantings. Although soil stability in both planting types was lower than in paddocks and remnants, possibly suggesting that the exposed ground may facilitate erosion, the difference in soil stability appeared to be strongly associated with the amount of vegetation within 50 cm of the ground. Therefore, the lower soil stability score in plantings appeared to be related to less understorey cover, rather than measuring erosion *per se*.

Ecosystem function is an important attribute, along with vegetation composition and structure, that needs to be established in revegetation plantings, for those plantings to be on a trajectory of self-sustainability, and to meet the criterion of 'success' (Ruiz-Jaen and Aide 2005a). However, we consider that at this stage, 'function' is often poorly defined or understood, and is difficult to measure. LFA is one of the few methods currently available for measuring ecological function. Although this method is used widely by the mining industry throughout world (Tongway and Hindley 2004), and is growing in popularity, we caution its uncritical adoption in systems outside those for which it was intended. We conclude that use of the LFA as a measure of restoration success in agricultural plantings such as ours is likely to give prematurely optimistic values.

The woodlot and ecological plantings in our study differed significantly in plant diversity, shrub cover, and the number of vegetation strata (Munro et al. 2009). These vegetation differences between ecological and woodlot plantings may affect more complex ecological functions such as habitat provision for wildlife, rather than the basic ecological functions of soil stability, water infiltration or nutrient cycling, as measured by the LFA.

6.7 Implications for Practice

- The Landscape Function Analysis is increasingly being used as a tool to measure the success of restoration. We caution that the LFA may provide measures of ecosystem function that are insufficiently sensitive to reflect true levels of ecosystem function in revegetation plantings. It may, therefore, give a premature reflection of 'success'.

6.8 Acknowledgements

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Chapter 7: Synthesis

This thesis has provided new information on the biodiversity value of revegetation. Specifically, the thesis explored the difference in biodiversity value between plantings established specifically for restoration purposes, and other plantings not specifically targeting biodiversity. In this chapter, I present a synthesis of my findings, as well as a short discussion on what my findings mean for revegetation, some implications for practice, and suggested directions for future research.

7.1 Key findings

My review of fauna in revegetation in Australia (Chapter 2) demonstrated that many species of fauna can recolonise revegetation plantings, and that certain attributes of plantings can influence faunal presence. Of the 27 studies included in the review, the majority focussed on very few site attributes or landscape features that may influence faunal colonisation. Most of these attributes related to size, width and/or isolation of plantings. By contrast, few studies measured the level of structural complexity or 'quality' of the revegetation sites. This thesis attempted to fill this (and other) knowledge gaps, by considering planting attributes that may be of importance to fauna.

'Ecological plantings', by my definition, were established with more species of native vegetation than 'woodlot plantings'. However, in Chapter 3, I showed that neither planting type could be relied upon to provide habitat for non-planted vegetation, because colonisation of new plant species through time was largely absent. Contrary to my expectations, I found no support for the 'foster ecosystem hypothesis' (Haggar et al. 1997), whereby overstorey vegetation facilitated the colonisation of mid- and understorey species. Because of the greater diversity of plants at establishment, the condition of ecological plantings was more similar to that of remnant vegetation than that of woodlot plantings (Chapter 3). In ecological plantings, both vegetation condition and weed cover became more similar to the condition in remnants with age, whereas woodlot plantings were not on a similar trajectory, achieving lower condition, and actually increasing in weed cover with age. Therefore, I suggest that ecological plantings may aid the conservation of plants that were deliberately planted by providing habitat for these species. The greater plant diversity, and development of vegetation structural complexity could be expected to provide greater habitat opportunities for fauna by increasing the number of niches at the site, and approaching a 'natural' condition found in remnant forests. I tested this hypothesis on two groups of animals, birds and mammals (Chapters 4 and 5).

In Chapter 4, I showed that ecological plantings can provide habitat for a distinct, shrub-associated assemblage of birds. This, in turn, may contribute to the regional or beta-diversity of birds. Woodlot plantings contained a lower species richness of birds than ecological plantings, and were dominated by generalist species (Chapter 4). In addition, it appears that ecological plantings can provide habitat for birds sooner after establishment than woodlot plantings. Despite this, several bird species and guilds, such as bark- and hollow-specialists, were rare or absent from both types of plantings, suggesting that plantings will not be viable replacements of remnant vegetation in the short term (30 years).

In Chapter 5, I showed that several species of mammal inhabited both ecological and woodlot plantings. Notably, even dispersal-limited arboreal marsupials were able to cross the paddocks to colonise plantings, often within only a few years. Some mammals appeared to respond to certain features in the plantings, such as a complex shrub layer and old remnant trees.

In Chapter 6, I discussed the use of the Landscape Function Analysis for measuring indices of soil stability, water infiltration and nutrient cycling in the context of revegetation. Landscape

Function Analysis (sensu Tongway and Hindley 2004) was originally designed for monitoring the rehabilitation of mine sites, although its application has been extended to other ecosystems and threatening processes, including revegetation plantings (Leguédois et al. 2008). On applying this procedure to revegetation plantings, I found that the condition at my sites was outside the optimal range of sensitivity of the Landscape Function Analysis (Chapter 6).

In summary, ecological plantings offer a range of environmental benefits, and appear to have greater biodiversity value than woodlot plantings (especially for plants and birds), but could not be considered of equal value to remnant vegetation. Woodlot plantings will offer habitat for a subset of predominantly generalist species of plants, birds and mammals.

7.2 General Discussion

In many situations, research on plantings aims to determine if animals do or do not recolonise a site, given the site's particular characteristics (e.g. Twedt et al. 2002, Hamel 2003, Paquet et al. 2006, Loyn et al. 2007). However, trajectories of change, and critical attributes of plantings that facilitate desired biodiversity, would likely provide more information with which to guide future revegetation (Anand and Desrochers 2004). Trajectories of change can give critical information on the likely outcome of the planting, and the time in which to expect particular changes (Vesk and Mac Nally 2006). The desired trajectory of a planting would likely be toward a remnant condition. The time taken to achieve similarity to remnants in structure and composition, is critical for understanding time-lags in provision of resources for biodiversity (Vesk et al. 2007). This thesis focussed particularly on the trajectories of change, using a space for time substitution.

There are many attributes of plantings which can be manipulated by management actions to alter their structure and composition. At the time of planning, decisions are made on the location, size, shape and plant diversity of the planting. Heterogeneity can be created actively by planting clumps of species, or leaving strategic gaps in vegetation, or this can occur naturally with direct seeding of sites. The presence of logs could increase biodiversity in the planting, given their importance in many remnant ecosystems (Mac Nally et al. 2001). Logs can be left if already present, manually added, or planned for (e.g. short-lived trees could be added that will provide logs after a few years). These examples suggest that relatively minor management actions could provide large biodiversity benefits. This project sought to identify which management actions could create biodiversity gains in plantings.

In particular, throughout the thesis, I have considered plantings differing in 'quality'. For most analyses, I categorised the plantings into 'ecological plantings' (considered high quality) and 'woodlot plantings' (low quality). The initial quality of a planting was assessed subjectively at the time of selection, based on whether or not shrubs and understorey were included in the planting mix at establishment. Ecological plantings are usually the more costly to establish, due primarily to their greater variety of plant species (up to 39 in my study). Conservation and funding agencies, as well as landholders wish to know if the extra cost of creating ecological plantings is worth it. In this thesis, I investigated the differences in biological gains between ecological and woodlot plantings, in terms of additional plant, bird and mammal species (Chapters 3, 4 and 5). I provided information only on the biological gains, rather than a cost-benefit analysis of whether those gains were worth the cost.

In addition to considering plantings differing in 'quality', I also considered other planting attributes, such as the presence of existing old remnant trees in the planting, the riparian or non-riparian location, the surrounding woody vegetation cover and the size of the planting. Each of these attributes, or landscape features, can be manipulated by planning decisions, and can lead to measureable changes to the biodiversity value of the planting. Of importance to land managers and restorationists is information about the relative biodiversity value each attribute contributes, so that trade-offs and synergies can be incorporated into decision making (Maron and Cockfield 2008).

7.3 Implications for practice

On the basis of the findings of this thesis, I recommend the following practical actions, which are particularly pertinent to the Gippsland region:

Recommendation	Justification	Reference
Plant a large diversity of local species of vegetation, including understorey plants, to mimic as closely as possible the pre-cleared vegetation, or reference remnant vegetation	Plantings with many local plant species provides habitat for local shrub-associated birds, may provide better habitat for arboreal marsupials, and provides 'habitat' for the planted species.	Chapters 2, 3, 4 and 5
Do not rely on natural colonisation of understorey plants species, but manually establish these where desired	Natural colonisation of understorey plants may take a long time, or may not occur at all.	Chapter 2

Woodlot plantings should be valued in our landscapes. Birds, in particular, would benefit if woodlot plantings were enhanced with shrubs	Despite their simple structure, woodlot plantings provide a degree of habitat for a range of birds and mammals	Chapter 4
Make plantings as large as possible	Large planting size increases bird species richness	Chapter 4
Plant around existing old remnant trees	An old tree incorporated into a planting can increase bird richness	Chapters 2 and 4
Give consideration to the 'quality' of a planting, by attempting to match the structural complexity and vegetation condition of remnants	High 'quality' plantings have greater value to biodiversity	Chapters 3, 4 and 5
Control exotic weeds in plantings, to reduce invasion by exotic birds	Exotic birds are associated with exotic plants	Chapter 4
Conduct long-term monitoring on the vegetation development, plant species succession and faunal colonisation of plantings	Long-term monitoring is essential to determining the 'success' of a planting, since vegetation development and species succession may take a very long time, and 'success' is not guaranteed	Chapter 3
Time lags should be considered in vegetation clearing policies that use plantings to offset losses from clearing elsewhere	Revegetation plantings are a valuable addition to other conservation attributes in our agricultural landscapes, but they take time to develop and be colonised	Entire thesis

7.4 Future research directions

Several key gaps remain in our understanding of the biodiversity value of revegetation. I suggest further research should target the following areas:

- Long-term trends on the development of vegetation structure and floristics, including key structural features such as old trees and logs, and their effect on fauna. This will provide information on what biodiversity gains can be expected over time, and with certain management actions.

- Analyses of the trade-offs between quantity and quality of revegetation plantings at the landscape scale, to assist managers in prioritising limited funds for maximum biodiversity gains.
- The value of local indigenous plant species for native fauna. Local or exotic plants may have a profound influence on the fauna, and understorey flora, in the planting.
- The response to revegetation by less studied taxa such as reptiles, small terrestrial mammals, amphibians and bats, since the needs of these taxa may differ from those commonly studied.
- The value of revegetation to declining or threatened fauna, since these are the species of conservation concern.
- The assessment of ecosystem function of revegetation, including the development of reliable methods of measuring function. Function has been perceived as a critical aspect in restoration 'success', but understanding and measuring function in revegetation remains poor.
- Further work on the particular features of plantings that maximise indigenous biodiversity, and how those features are best incorporated or encouraged into a planting.

Implementing the findings of this thesis, and further addressing the research directions outlined above, will hopefully lead to increased biodiversity outcomes, in a shorter time-frame, for revegetation on agricultural lands.

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Appendix 1: Additional work

This Appendix presents a overview of five additional papers written during the doctorate which were not directly related to the thesis.

1. Fire and aquatic invertebrates

Munro, N. T., K.-J. Kovac, D. Niejalke, and R. B. Cunningham (2009). The effect of a single burn event on the aquatic invertebrates in artesian springs. Austral Ecology **in press.**

Abstract

Fire can often occur in aquatic ecosystems, which may affect aquatic invertebrates. Despite the importance of aquatic invertebrates to ecosystem function, the effect of fire in these environments has been little studied. We studied the effects of fire on aquatic invertebrates in artesian springs in the arid zone of South Australia. Artesian springs are a unique and threatened ecosystem, containing several rare and endemic species. Evidence suggests these wetlands were routinely burnt by indigenous Aboriginal people before European settlement over 100 years ago. Recently, burning has been suggested as a re-instated management tool to control the dominant reed *Phragmites australis*. A reduction in the cover of the reed may benefit the threatened flora and fauna through enhancement of water flow. Three artesian springs were burnt and aquatic invertebrates sampled from the burnt and three unburnt springs. A single fire in late winter completely burnt the dominant vegetation, followed by recovery of *Phragmites* over the following two years. A single fire event did not deplete populations of endemic aquatic invertebrates in artesian springs, but probably did not substantially benefit these populations either. Isopods, amphipods, ostracods and 3 species of hydrobiid snail survived the fire event, and most had increased in number one month post fire but then returned to pre-burnt numbers by one year post fire. Morphospecies richness of all identified invertebrates increased over time in all springs, but did not differ appreciably between burnt and unburnt springs. If burning artesian springs is to be adopted as a management tool to suppress the growth of *Phragmites australis*, we conclude that the endemic aquatic invertebrates will survive a single burn event, without negative effect to their populations.

2. Seedling survival under native and exotic browsing

Munro, N. T., K. E. Moseby, and J. L. Read (In minor revision). The effects of browsing by feral and re-introduced native herbivores on seedling survivorship in the Australian Rangelands. The Rangeland Journal.

Abstract

Browsing by introduced cattle and rabbits can limit the recruitment of some arid zone tree and shrub species. In a study conducted at the Arid Recovery Reserve, Roxby Downs, South Australia, we aimed to quantify initial recruitment changes in shrubs after the removal of cattle and rabbits, and the re-introduction of locally extinct fauna. The presence and abundance of seedlings were measured at groves of seven native perennial shrubs over six years under four browsing treatments: 1. 'reserve-reintroductions' (re-introduced stick-nest rats, burrowing bettongs and greater bilbies), 2. 'reserve-no browsers', 3. 'pastoral-stocked' (rabbits and cattle) and 4. 'pastoral-destocked' (rabbits). Recruitment of mulga (*Acacia aneura*), silver cassia (*Senna artemnisioides petiolaris*) and sandhill wattle (*Acacia ligulata*) was significantly greater in the two browsing regimes inside the Reserve compared to the two pastoral regimes. The number of recruits of these three species declined at 'pastoral-destocked' and 'pastoral-stocked' sites but increased at 'reserve-reintroductions' and 'reserve-no browsers' sites from 2001 to 2006. Narrow-leaf hopbush (*Dodonaea viscosa*) showed a trend towards increased recruitment at sites in both browsing regimes inside the Reserve, and decreased recruitment at sites in both pastoral regimes. Native plum (*Santalum lanceolatum*), native apricot (*Pittosporum phyllarioides*) and bullock bush (*Alectryon oleofolius*) exhibited no significant difference in recruitment between the four browsing regimes, within the study timeframe. These results suggest that excluding rabbits and stock may benefit the germination and survival of mulga, silver cassia and sandhill wattle. To date, re-introduced native herbivores at low numbers have not been found to negatively affect the recruitment or growth rate of the seven perennial plant species studied.

3. Single-species and ecosystem-orientated research

Lindenmayer, D. B., J. Fischer, A. Felton, R. Montague-Drake, A. D. Manning, D. Simberloff, K. Youngentob, D. Saunders, S. P. Blomberg, D. Wilson, A. M. Felton, C. Blackmore, A. Lowe, S. Bond, N. Munro, and C. P. Elliot (2007). The complementarity of single-species and ecosystem-orientated research in conservation research. Oikos **116:**1220-1226.

Abstract

There has been much debate about the relative merits of single-species vs ecosystem-oriented research for conservation. This debate has become increasingly important in recent times as resource managers and policy makers in some jurisdictions focus on ecosystem-level problems. We highlight the potential strengths and limitations of both kinds of research, discuss their complementarity and highlight problems that may arise where competition occurs between the two kinds of research. While a combination of approaches is ideal, a scarcity of funding, time, and expertise means it is impossible to study and manage each species, ecological process, or ecological pattern separately. Making decisions about priorities for the kinds of research, priorities for the kinds of conservation management, and associated allocation of scarce funds is a non-trivial task. We argue for an approach whereby limited resources for conservation research are targeted at projects most likely to close important knowledge gaps, while also promoting ongoing synergies between single-species and ecosystem-oriented research.

4. Climate change publications

Felton, A., J. Fischer, D. B. Lindenmayer, R. Montague-Drake, A. Lowe, D. Saunders, A. M. Felton, W. Steffen, N. T. Munro, K. Youngentob, J. Gillen, P. Gibbons, J. E. Bruzgul, I. Fazey, S. J. Bond, C. P. Elliott, B. C. T. Macdonald, L. L. Porfirio, M. Westgate, and M. Worthy (2009). Climate change, conservation and mangement: an assessment of the peer-reviewed scientific journal literature. Biodiversity and Conservation **18:**2243-2253.

Abstract

Recent reviews of the conservation literature indicate that significant biases exist in the published literature regarding the regions, ecosystems ad species that have been examined by

unlikely to be achieved contemporarily in the absence of a) this knowledge, and b) previous species and processes.

Paper 3 (single-species and ecosystem research) and Paper 4 (climate change literature) suggest directions for future research which could redress some imbalances in current internationally-targeted publications. Balancing research output will reduce bias and hopefully provide a more accurate framework within which we can recommend actions be taken.

Paper 5 (current threats and climate change) presents the important message that we know sufficient to act on many threats. Papers 1 and 2 demonstrate that there are always some uncertainties involved, but with research applied at different scales or targets (Paper 3) and balanced from a global perspective (Paper 4), we can accumulate sufficient knowledge to recommend actions to be taken with a reasonable degree of certainty. We must carefully acknowledge the uncertainties that exist, and consider the implications of not-acting compared to acting.

Wind Harps
by Fay White

When the wind and rain were new on Earth,
And the Casuarinas came to birth,
No human here to see them stand,
In the forests of Gondwanaland.
No human ear, no human eye,
As they spread their jointed fingers high,
To catch the breeze that makes them hum,
In the rippling wind their needles thrum.

And they grow in male and female kind,
In the spring the male fronds flowers find,
But the wind will fly their pollen home,
To the females heavy with new-flowered cones,
Where copper-winged butterflies dance with grace,
And possum and kangaroo find their place,
And the seasons turn from heat to cold,
And time moves slow in days of old.

CHORUS
Wind harps of the Wimmera,
Allocasuarina luehmannii,
Brother belong through Dja Dja Wurrung country,
Sigh as the breeze blows by.

Wind harps of the Western Plains,
Keen where the west winds blow,
Songs of old Gondwanaland,
And the days they used to know.

We came, we looked, we did not see,
Our eyes in love with bright grass green,
It was scraggy, scruffy scrub for miles,
While neat and tidy was more our style.
With axe and saw, to build and burn,
We cleared and took and did not learn,
Of the life that flew and crept and roamed,
In their Casuarina woodland home.

Will the golden sun moth cease to fly?
Will we loose the bush stone curlews cry?
Will the gold jewel beetle fold its wings,
And the little cicadas cease to sing,
Will the red-tailed black cockatoo disappear?
And the tiny forest bats go from here,
Will the sweet quondong grow back and thrive,
Will the Casuarina woodlands yet survive?

May the old tree lonely in the field,
Have a chance to yet set seed and yield,
May we see a change across the plain,
With woodlands winding round the grain,
We will fence out stock, and knock down weeds,
To create a place for the falling seeds,
Then the seeds will spring from autumn rain,
And the Casuarina woodlands come again.