



Biodiversity in eucalypt plantings established to reduce salinity

**A report for the RIRDC / Land & Water Australia
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Joint Venture Agroforestry Program**

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Foreword

Extensive areas of trees and shrubs are being planted for land rehabilitation and wood production on previously cleared agricultural land in Australia. Environmental benefits are a major incentive for this change in land management, but data concerning the response of biodiversity to new plantings are scarce and are needed to direct restoration efforts and to underpin policy development.

This report outlines a large-scale study to guide future planting schemes for biodiversity recovery in agricultural landscapes. It documents and compares the occurrences of more than 180 species of birds, mammals, reptiles and amphibians sampled at 136 sites representing eucalypt plantings in two broad age-classes, nearby remnants of native forest and woodland, and cleared or sparsely-treed paddocks. The study design also enabled comparison of the occurrences of these species across a range of patch-sizes of both revegetation and existing remnant vegetation. Comparisons were also made of the occupancy of young plantings by birds and bats in two landscape types, which differed mainly in their proportions of retained native vegetation. The role of vegetation type and condition, and management history, in influencing the occurrences of species in plantings and remnants is discussed. The report concludes with recommendations for restoring habitat for wildlife on farms, in the context of improving conservation outcomes from eucalypt plantings that are established for multiple purposes.

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This report, a new addition to RIRDC's diverse range of over 1200 research publications, forms part of our Agroforestry and Farm Forestry R&D program, which aims to integrate sustainable and productive agroforestry within Australian farming systems.

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Acknowledgements

A study of this type and in this location has necessarily had to straddle many jurisdictions and required the assistance of many people. The study has been based primarily on privately-owned lands and consequently has involved the cooperation of numerous landowners, all of whom are listed below. Furthermore, added complexity was introduced because the study area crossed State borders and required approvals to work on lands managed by many different local government, state government and commonwealth government agencies. The reason for selecting the Albury-Wodonga region for this study was because of the availability of substantial areas of older-aged (>20 years) plantings of eucalypts (and shrubs) that were established by the Albury-Wodonga Development Corporation (AWDC) from the mid-late 1970's. These plantations represented the most significant areas of older-aged eucalypt plantings to be found anywhere west of the Great Dividing Range in NSW. Thus, they were likely to provide the best guide as to the likely gains in biodiversity that might accrue from recent efforts to establish tree and shrub plantings in agricultural landscapes that are subject to increasing salinity or other forms of land degradation. Accordingly, we wish to thank John Alker-Jones of the AWDC for his enthusiasm and support of this study and also Peter Quinn (of AWDC) who provided maps and many of the digital data layers essential for this project.

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Executive summary

Most of the original forest and woodland cover on the western slopes of NSW and the northern plains of Victoria has been cleared for agriculture (wheat, sheep and cattle), and what remains is highly fragmented and modified by a long history of grazing and cutting for fenceposts and firewood. Extensive revegetation efforts for environmental benefits including land rehabilitation, salinity reduction and habitat restoration, as well as for improved landscape amenity and wood production, have been occurring in the Albury-Wodonga region over the past three decades. These plantings of native trees and shrubs have the potential to greatly improve the quality of the habitat matrix surrounding existing remnants of forest and woodland for a wide range of fauna, and to increase the carrying capacity of the habitat within remnants. To guide future planting schemes for restoring biological diversity, we sampled 120 sites for birds, bats, arboreal marsupials, terrestrial mammals, reptiles and amphibians. These sites encompassed the range of available patch sizes, stand ages, habitat attributes, and floristic and structural conditions for revegetated areas, and remnants, in the region and we compared these to nearby paddocks. We also sampled birds and bats at another 16 planted sites to further consider the influence of proximity of the revegetated areas to remnants, and the broader landscape context. The aim of the study was to determine the extent to which eucalypt plantings could assist farmers and regional conservation planners to improve biological diversity in agricultural landscapes in south-eastern Australia and to identify the variables which may influence their effectiveness.

Eucalypt plantings were found to provide significant improvements in biodiversity compared to cleared or sparsely-treed paddocks, but the different fauna displayed a range of responses. Indeed, mixed eucalypt and shrub plantings contained similar numbers of birds and bats to remnant native forest and woodland in the region. Birds and bats made the most extensive use of plantings, particularly favouring the older (> 10 years) age class. Birds showed a strong response to patch size, with both larger eucalypt plantings and larger remnants having more species and more individuals than smaller patches of these vegetation types. Even young (<10 years old) plantings were occupied by many birds, once patch sizes were larger than 5 ha. Bats were widespread throughout all vegetation types sampled (planted areas, remnants and paddocks), although they were more common in remnant vegetation. Remnant forest and woodland were most important for arboreal mammals, nocturnal birds and reptiles, but the oldest (>20 years) plantings were also contributing to the habitat of these species. Younger plantings and cleared or sparsely-treed areas provided little habitat for these species, and no reptiles were recorded in paddocks. Ground mammals were virtually absent from the region, presumably due to the widespread and long-term impacts of grazing and the abundance of introduced predators.

Plantings of native trees and shrubs of all shapes and sizes, especially those larger than 5 ha, have an important role to play in providing habitat for many species. These plantings may also contribute significantly to increasing the effective size of remnants in agricultural landscapes. Young plantings of native vegetation are used by more species of woodland-dependent birds if they are situated in landscapes having greater amounts of remnant forest and woodland.

Recommendations arising from this study include the need to recognise the importance of existing remnants of native forest and woodland in regional conservation planning because many species were recorded most frequently, and some only, in remnant vegetation. Also, remnants usually contain old, hollow-bearing trees, providing essential roosting and nesting sites for many species. The absence of roosting and nesting hollows, as well as the often low levels of shrub cover, grass cover and logs on the ground, were important factors limiting the suitability of eucalypt plantings as habitat for many species. Site preparation for eucalypt plantings could be improved by not removing old, paddock trees and logs on the ground. In plantings established primarily for nature conservation, management should exclude grazing by domestic stock to protect against loss of shrubs and grass cover. Stock should also be excluded from some dams, or parts thereof, because these waterbodies and associated vegetation are important for frogs and many other species. Remnant vegetation should provide the focal point for habitat restoration efforts using eucalypt plantings.

Research is needed to determine whether eucalypt plantings are capable of providing breeding habitat for the wide range of species known to occupy and forage in them. Opportunities for improving the value of eucalypt plantings for wildlife need to be explored. The extent to which eucalypt plantings can augment the carrying capacity of remnants for wildlife should be determined, as well as their potential for assisting the recovery of threatened species. The rankings given to various habitat surrogates (including vegetation type and condition, patch size, connectivity and landscape context) used to underpin the estimation of biodiversity “value” of alternative management actions needs to be fully evaluated for a wide range of species. Several proposed “biodiversity toolkits” to effect land use change in agricultural areas, using both market-based and government-subsidy schemes, depend on assumptions made about “biodiversity” responses to varying levels of these habitat surrogates. Research is needed to provide confidence that the components of these toolkits are relevant and that the rankings for different restorative actions have been properly calibrated.

1. Introduction

Surprisingly little research has been done in Australia to characterise the value of eucalypt plantations and plantings of native trees and shrubs for wildlife. Currently, there are increased opportunities for planting eucalypts on private land, either as joint-venture wood production activities, or for land restoration, such as controlling salinity on agricultural lands. Documentation of the likely gains in biodiversity resulting from tree planting may positively influence the acceptance by farmers of land-use changes and may also provide significant additional incentives for investors. The ability to benchmark a range of biodiversity outcomes resulting from different plantation designs (e.g. size, age, species mix and proximity to remnant vegetation) and management practices (e.g. thinning, grazing and fire regimes) is central to the development of models predicting biodiversity values in the landscape.

Over the past two decades, many studies have documented patterns of species distribution and abundance in remnant native vegetation and have produced models to predict diversity (e.g. Loyn 1987, Pahl *et al.* 1988, Saunders *et al.* 1991, Andren 1994, Laurance 1994, Martin *et al.* 1995, MacNally and Bennett 1997, Lindenmayer *et al.* 2000). The three main factors affecting animal populations in remnant habitat are the size of the remnant, the degree of isolation and habitat quality. Habitat quality is often the most important factor and this will be different for every species. However, several key (habitat) attributes of remnants are associated with species richness in Australia and they include the extent of tree cover (Bennett and Ford 1997), the availability of old trees and associated nesting hollows (Bennett *et al.* 1994), plant species diversity (Seddon *et al.* 2003), the degree of shrub development and tree regeneration (Fisher 2001, Seddon *et al.* 2003), the amount of coarse woody debris (MacNally *et al.* 2001) and proximity to water or riparian zone vegetation (Fisher and Goldney 1997, MacNally *et al.* 2000, Soderquist and MacNally 2000). The status of these key attributes in any one broad vegetation type is now often referred to collectively as a measure of vegetation condition. In addition, it is now widely recognised that fragmentation effects are highly dependent on the ability of each species to use the matrix of vegetation surrounding each remnant (Barrett *et al.* 1994, Wiens 1994).

In contrast, very little research has been undertaken on factors influencing biodiversity in tree plantings and how this might differ from natural remnants of vegetation or other benchmark communities such as farmland (Law and Kavanagh 1998, Ryan 2000, Kavanagh *et al.* 2001, Kinross 2004). Cleared pasture is well known to support fewer native birds, mammals, and invertebrates than eucalypt forest, including regenerating forests (Green and Caterall 1998). However, there is considerable potential to increase regional biodiversity through the establishment of eucalypt tree and shrub plantings in rural landscapes. Until recently (e.g. Kavanagh *et al.* 2001, Kinross 2004), only few studies had compared the fauna of eucalypt plantations (mostly on rehabilitated mining sites) with that of nearby native forests (Woinarski 1979, Nichols and Watkins 1984, Nichols and Bamford 1985, Nichols and Burrows 1985, Collins *et al.* 1985, Curry and Nichols 1986, Kavanagh and Turner 1994) and none had compared eucalypt plantations with cleared lands used for grazing or cropping. Species richness and abundance was generally found to be lower in eucalypt plantations than in mature native forests, but plantations are unlikely to ever support the full complement of forest dependent species without supplementation of key missing resources such as tree hollows (or nest boxes) that normally occur only in old trees, hollow logs, dead wood and decorticated bark (Kavanagh 1987, Dickman 1991, Scotts 1991, Law 1996). Nonetheless, eucalypt plantations (plantings) have the potential to greatly improve the quality of the habitat matrix for fauna within rural landscapes by augmenting the important role played by forest remnants in regional conservation of biodiversity (Hobbs 1993, Saunders and Hobbs 1995).

Eucalypt plantation establishment on ex-grazing land sites west of the Great Dividing Range for wood production and/or environmental services, such as biodiversity, carbon sequestration and salinity control, is a relatively new phenomenon in New South Wales. Rising salt levels in many west-flowing streams and other “discharge” zones (lower slopes, valleys) in rural landscapes has created an urgent need to modify existing land-use practices, principally by restoring deep-rooted, long-lived plants (e.g. trees) in “recharge” zones (upper slopes). The NSW Salinity Strategy (2000) aims to slow down and

eventually reverse the level of salinity occurring in the waterways and soils of our rural landscapes. A major objective of the Strategy is to protect existing native vegetation in these landscapes and to greatly increase the number of deep-rooted plants (trees) in recharge areas so that less water leaks through to the ground water table (discharge areas). The use of market-based solutions and strategic investment have been identified as key tools in the Strategy to facilitate land-use change by means of broad-scale tree planting (re-vegetation) of recharge areas within the landscape. An essential component of both market-based solutions and government subsidy approaches is the benchmarking of biodiversity values occurring within remnant native and planted vegetation in the landscape. However, this requires a detailed knowledge of the relativity between the species assemblages occupying remnant forest and woodland, planted forests containing native trees and shrubs, and the status quo which is usually cleared or semi-cleared grazing land. Fortunately, a large-scale programme of planting eucalypts and native shrubs on ex-grazing sites was undertaken from the mid 1970's through the 1980's by the Albury Wodonga Development Corporation (AWDC) near the twin cities of Albury-Wodonga to improve the landscape character of the region and thus making it more attractive as a place for people to live. These plantations provide an ideal opportunity to investigate the likely outcomes for biodiversity that may result from current tree planting efforts in areas subject to dryland salinity, since many AWDC plantations are already 20-25 years old.

In 2001, we completed a pilot study to assess the feasibility of a larger study in the Albury-Wodonga region to investigate the biodiversity benefits of re-vegetation in agricultural landscapes, especially those that may be subject to increasing salinity (Kavanagh *et al.* 2001). That study reported the relative abundance of vertebrates (mammals, birds, reptiles and amphibians) from 21 sites in the region representing a wide range of vegetation classes including farmland, eucalypt plantations and remnant native forest and woodland. Preliminary results indicated that significant improvements in vertebrate species and numbers occur when trees are planted in agricultural landscapes. Birds and bats (both highly mobile) appeared to be the fauna groups most capable of exploiting the new habitat provided in the landscape by eucalypt plantations. Other mammal species, reptiles and amphibians appeared to be much slower to colonise eucalypt plantations, and numbers of these species were also low in the remnant forests and woodlands of the region. Plantation patch size appeared to strongly influence the numbers of birds occupying eucalypt plantations, with more species and individuals recorded in the larger plantations than in smaller ones. However, for bats, there was no evidence that plantation patch size and/or remnant size was important. The pilot study was unable to adequately assess the importance of connectivity with native forest and woodland, or the influences of elements of vegetation floristics and structure (vegetation condition) likely to affect habitat quality for vertebrates.

1.1 Aims

The broad aims of this study are to identify the major factors influencing biodiversity in the low rainfall (600-800 mm), fragmented rural landscapes that are susceptible to salinity on the western slopes of NSW, and to identify the role that plantings of native trees and shrubs may have in restoring biodiversity in agricultural landscapes. Plantation age, plantation patch size, proximity to remnant native vegetation, the numbers of tree and shrub species planted (a measure of habitat quality), and landscape context are all likely to be among the most important variables influencing biodiversity in eucalypt plantations. In this study we aim to:

- determine the value of eucalypt plantations for wildlife, relative to remnant forest and woodland, and how this is influenced by plantation design and management practices;
- investigate the relationships between vertebrate species occurrence and the age, area and plant species mix of the plantations, as well as the effect of proximity to native forest remnants;
- investigate the influence of landscape context, especially the amount of remnant vegetation remaining in the general landscape, on the occupancy of eucalypt plantations by mobile species such as birds and bats;
- identify which species benefit from the establishment of eucalypt plantations; and
- recommend how eucalypt plantations can best be used to enhance nature conservation in the landscape.

2. General methods

2.1 Study area

The “twin cities” of Albury-Wodonga are located on the NSW-Victorian State border at the upper reaches of the Murray River (Fig. 2.1). The region was first visited by Europeans in the 1820’s and settlement followed shortly afterwards. By 1900, most of the original forest and woodland in the region had been cleared, and grazing by sheep and cattle became the dominant land use (Fig. 2.2). Remnants of the original forest and woodland vegetation now occur very sparsely throughout the region, mostly as scattered river red gums near major rivers and small patches of box-ironbark-stringybark woodlands that are confined to some upper slope areas. The mid-1970’s saw the introduction of a federal government policy of “decentralisation” in which incentives were provided to encourage businesses and householders to move into the area. The Albury-Wodonga Development Corporation was established to facilitate this process, and one important strategy involved an extensive program of tree planting throughout the region to improve the amenity or character of the local landscape. Broad-scale plantings of a mix of locally indigenous species, including trees and shrubs, began in 1977 and continued through until approximately 1991.

Plantations of native tree species are scarce throughout NSW in the low rainfall areas prone to salinity, and those that do exist are generally less than 10 years old. The plantings undertaken by the Albury-Wodonga Development Corporation represent the only major exception in NSW. These plantings now cover an extensive area around the two cities of Albury and Wodonga and they provide an excellent opportunity to investigate biological diversity occurring in older-aged (20-25 year old) plantings.

The climate of the region is characterised by hot, dry summers and cold, wet winters. Mean annual rainfall for Wodonga during 1898-1984 (86 years) was 715 mm, with mean daily maximum and minimum temperatures of 31.8°C (January) and 3.1°C (July), respectively (Bureau of Meteorology). Comparative data for Albury Airport during 1973-2004 (31 years) were 737 mm, and 30.9°C and 2.6°C, respectively. Rainfall in most of south-east Australia was well below average during 2001-2003. However, the Albury-Wodonga region was one of the last in NSW to be declared drought-stricken and was among the first to receive some relieving rainfall. The main period of sampling in this study occurred during November-December 2002, just prior to two very dry summers. Another sampling period occurred in late March-early April 2004 following a slight improvement in conditions.

The Albury-Wodonga region is well covered by digital map resources. Landsat TM data (1995) were provided by the Murray-Darling Basin Commission (Fig. 2.2), and recent (2000) colour air photos (1:10,000 scale) were provided by the AWDC.

2.2 Research design

The principal aim of this study was to identify and document the role that plantings of native trees and shrubs may have in restoring biodiversity in agricultural landscapes. Accordingly, the main factor used to stratify study sites was “vegetation type”, represented by four levels: remnant forest and woodland, old (> 10 years) eucalypt plantings, young (< 10 years) eucalypt plantings and paddocks (refer Table 2.1). Plantation age was incorporated within the factor vegetation type because young plantings differed from old plantings by more than just age. Young plantings were often planted by Landcare groups, tended to have a greater component of shrub species and were often fenced to restrict grazing by domestic stock. All planted sites were at least 3 years old, and usually were closer to 7 years old in the younger age class, while sites in the older age-class were usually planted about 15-25 years prior to sampling.

Study sites were also stratified by patch area, represented by up to five levels, but not all levels were available for sampling within each vegetation type. Levels of patch area were: linear strips (< 50 m wide), small (< 5 ha), medium (5-20 ha), large (20-100 ha) and very large (> 1000 ha). Linear strips were sampled only for two vegetation types, remnants and old plantings, while the very large

size-class was only available for remnants. Very large remnants were sampled to provide an indication of the potential levels of biological diversity that may be encountered in the region. It was not possible to achieve a balanced stratification of sites with respect to proximity to native vegetation but, where possible, half of all sites in each planted forest type and patch-size class were located within 500 m of remnants (> 10 ha in size). Half of the paddock sites were selected so that sampling plots contained at least one large remnant tree. Other potentially important variables, such as grazing history, proximity to water, numbers and species of trees and shrubs present (a measure of habitat quality), and many others representing degrees of vegetation disturbance (a measure of vegetation condition), could not be controlled and so were treated as co-variates in the analysis. Our research design included a total of 12 “treatment” classes (Table 2.1), each of which was replicated ten times, providing a total of 120 sites for sampling.

Table 2.1 The 12 treatment classes (and abbreviations) sampled in this study

<i>Plantings older than 10 years</i>	
Large Plantations (20-100 ha)	POL
Medium-sized plantations (5-20 ha)	POM
Small plantations (<5 ha)	POS
Narrow plantation strips (<50m)	POLi
<i>Plantings younger than 10 years</i>	
Young medium-sized plantations (5-20 ha)	PYM
Young small plantations (<5 ha)	PYS
<i>Remnant forest and woodland</i>	
Very large remnants (> 1000 ha)	RVL
Large remnants (>20-100 ha)	RL
Medium-sized remnants (5-20 ha)	RM
Small remnants (<5 ha)	RS
Narrow remnant strips (<50m)	RLi
<i>Paddocks</i>	
Paddocks with and without remnant trees	P or PAD

All 120 sites were located approximately between Holbrook in southern NSW and Wangaratta in northern Victoria (Fig. 2.1). Within this landscape, almost all planted sites were located within 5 km of a forest remnant > 10 ha in size.

The importance of landscape context was investigated further by selecting an additional 16 study sites in the same region that were more than 5 km from remnants of this size. These sites provided the opportunity to investigate thresholds in the distance of eucalypt plantings from remnant vegetation, and for the proportion of remnant vegetation in the landscape, in terms of the use of these plantings by highly mobile species such as birds and bats. Eight new planted sites were selected within the young medium size-class (5-20 ha) and eight new planted sites were selected within the young small size-class (< 5 ha). Vertebrate species sampled at these 16 new “distant” sites were compared to results obtained from concurrent sampling at the existing 20 young planted sites in the medium and small size-classes.

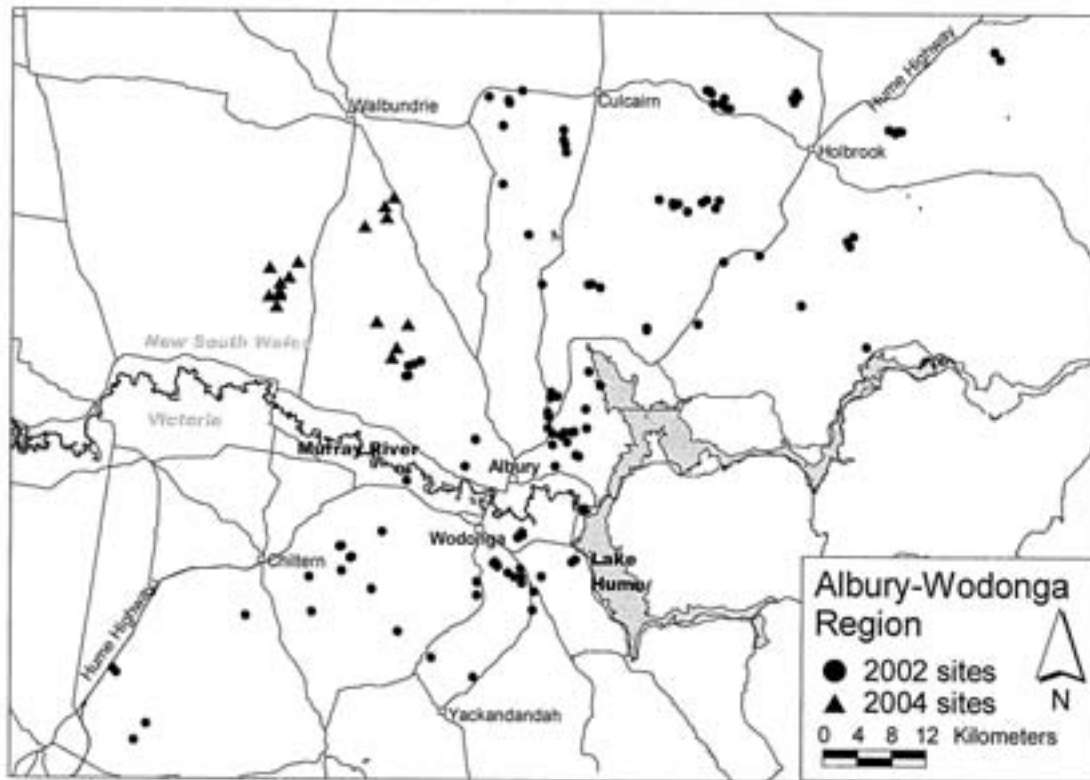


Figure 2.1 Location of the 120 sites sampled in 2002-2004 (circles) and the 16 sites sampled in 2004 (triangles) in the Albury-Wodonga region.

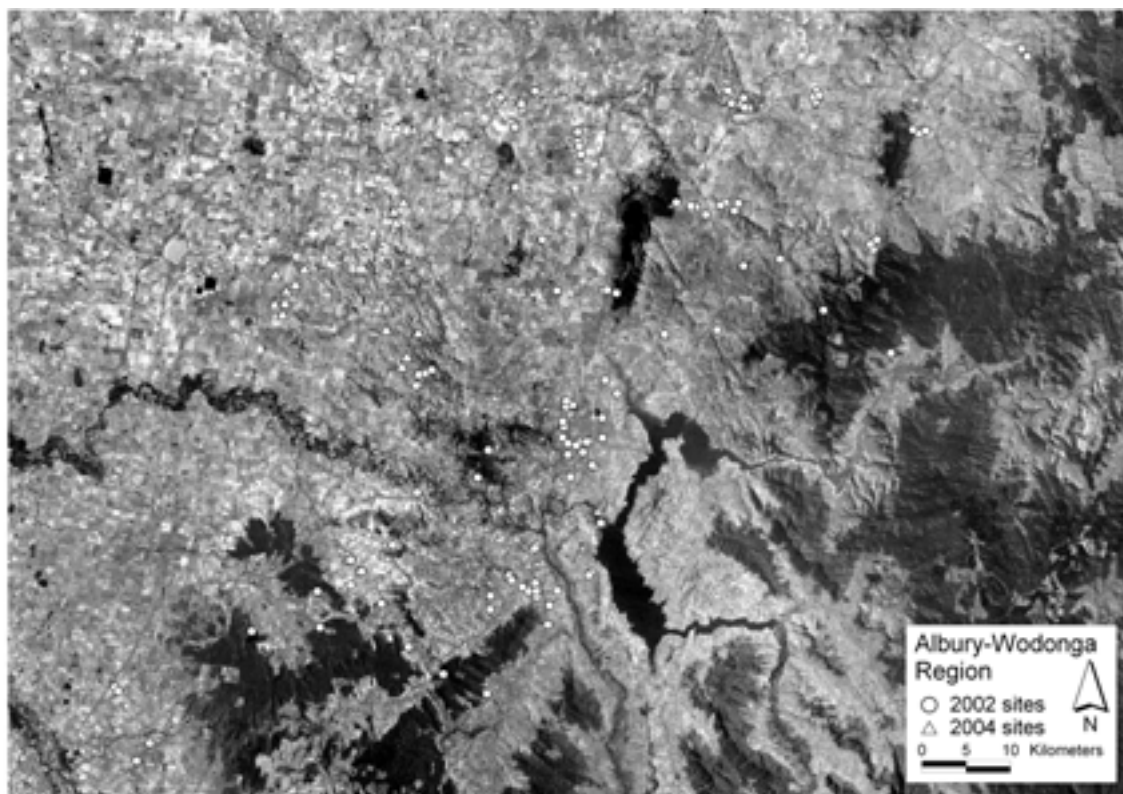


Figure 2.2 Location of the 136 study sites in relation to broad vegetation cover (Landsat TM pseudo-colour image 1995). Dark red/brown colour indicates remnant vegetation.

2.3 Sampling methods

Field assessments for all vertebrate fauna groups at 120 sites began in early November 2002 and, for most groups, were completed by late December 2002. This included systematic sampling for birds, arboreal mammals, terrestrial mammals, bats, reptiles and frogs. Assessments were also completed during this period for attributes of animal habitat and vegetation floristics, structure and condition at each sampling site.

Survey effort for each fauna group was held constant irrespective of the patch size of eucalypt plantings, native forest and woodland remnants and open paddocks. In this way, survey effort was not confounded with the effects of patch size on fauna species richness, a problem found in many other studies.

Details of the sampling methods employed for each fauna group are provided in later sections. In general, birds were sampled in each study area on two fixed-area (0.785 ha) circular plots over a standard period of time. Mammals were sampled at either one point (bats, using bat call detectors) or along a 200 m transect line (arboreal mammals, using spotlights; terrestrial mammals, using hair tubes). Elliott trapping and cage trapping for mammals was not employed in this study because of the low capture rate encountered during our earlier pilot study (Kavanagh *et al.* 2001) using these methods. Nocturnal birds were sampled using listening and call-playback methods during spotlight counts for arboreal marsupials. Reptiles and frogs were sampled in traps, including pitfall traps and wooden coverboards, by observation along the 200 m transects, and frogs were also sampled by searches of special wetland features (e.g. dams, stream banks).

Sampling for birds and bats on the 16 new sites, and the previously sampled 20 young planted sites, occurred during late March-early April 2004.

2.4 Analysis

This report is concerned mainly with the broad differences existing between the fauna assemblages of eucalypt plantings, forest and woodland remnants, and cleared paddocks typical of the agricultural landscape in the Albury-Wodonga region. Accordingly, most emphasis is placed on the differences between these vegetation categories in terms of their total species richness of vertebrates and in terms of the total numbers of individual animals. Bird assemblages are further categorised as woodland-dependent and non woodland-dependent species. Associations based on bird species with similar patterns of occurrence on the study sites are presented, along with associations of study sites in terms of their bird species assemblages. For the most abundant species groups, birds and bats, relations are also explored between species occurrences and the habitat variables measured or assessed at each sampling site. Bat species abundances are also analysed for differences between the major vegetation categories sampled in this study. The data for species groups other than birds and bats were generally not sufficient for statistical analyses. Full details of the analytical methods used are presented in the methods section in each of the following chapters.

3. Vegetation condition and connectivity

3.1 Introduction

One of the key questions relating to the use of planted vegetation by wildlife is the extent to which vegetation condition, especially structure, resembles that occurring within remnants of similar size and also larger, less disturbed remnants. Restoration of structure in plantings at present is poorly understood. Studies on the development of forest structure in rainforest planted on cleared land have found that different types of plantings restore some elements of structural complexity to cleared land, although often vegetation structure remains relatively simple (Kanowski *et al.* 2003). Greater resemblance in structural attributes was found between older plantings and regrowth than with intact rainforest. Similarly, a study of young restoration plantings (< 9 years) on the Cumberland Plain in Sydney found some convergence in the structure of planted areas with remnant vegetation, but vegetation composition could not be distinguished from pasture (Wilkins *et al.* 2003). Clearly, different types of reforestation are likely to vary in the structural complexity that they provide and hence in their value as fauna habitat (Kanowski *et al.* 2003).

In this chapter we explore the differences in habitat between vegetation classes (eucalypt plantings, remnant forest and woodland, and paddocks), focusing on various aspects of vegetation condition and physical structure as these are likely to be of fundamental importance to a range of fauna groups. Landscape connectivity is also likely to be a key attribute, which will influence recolonisation and site occupancy to different extents. Mobile fauna, like birds and bats, will presumably have greater dispersal capabilities than ground-dwelling fauna, such as frogs and reptiles. As our research design could not balance sites fully with respect to connectivity, it is important to explore how this attribute varied across the different vegetation classes sampled.

3.2 Methods

Habitat variables

A range of variables was recorded at each site to describe aspects of its vegetation and physical structure (e.g. log and rock cover), topography, grazing history and landscape connectivity (Table 3.1). Each attribute was assessed on one plot 20 m x 50 m (0.1 ha) per site. All sites were assessed by one observer (M.H.) to minimise observer bias. All cover and height variables were visually estimated. Shannon-Weaver diversity indices were calculated using counts of identified stems in our plots.

Landscape attributes of each site were calculated using Arcview GIS software (ESRI). First, each site was defined as a discrete patch of vegetation so that we could calculate patch area and perimeter using a combination of aerial photography (1:10 000), supplied by AWDC and Holbrook Landcare, and Landsat imagery (25m resolution - Murray-Darling Basin Commission). Next, a data-layer of woody vegetation in the study region was defined by mapping all remnants and planted areas, with these two vegetation classes identified and separated where possible. Native plantings were easily separated from pine plantations. Scattered trees in paddocks were not mapped. Spatial-layers of Tree Canopy Density (DLWC, NSW) and Tree Density (DSE, Victoria) also aided this process. The woody vegetation layer was used to estimate connectivity for each of our sites by calculating the area of woody vegetation (excluding pine plantations) in circular buffers with radii of 0.5 km and 5 km. The extent of fragmentation surrounding each site was also indexed as the amount of perimeter or edge within each buffer.

Data are first presented as a comparison of means across vegetation classes or as presence/absence at a site expressed as a proportion of each vegetation class (e.g. % of sites where mistletoe was recorded). A multivariate analysis was then undertaken to explore the extent to which sites could be separated on the basis of structural attributes at each site. These included ground cover, litter cover, litter depth, log cover, shrub cover, shrub height, overstorey cover, overstorey height, basal area, stem

density and large stem density (> 60 cm dbh). Sites were ordinated by hybrid non-metric multidimensional scaling (Belbin 1995), after first range-standardising variables.

Table 3.1 List of habitat variables recorded on 120 study sites in the Albury-Wodonga region.

Target Variable	Methodology
Topographic position	Nine point scale; 1=Peak to 9=Flat
Overstorey height (m)	Mean height of vegetation > 5m above ground
Overstorey cover (%)	Visual estimate of projective cover of overstorey vegetation
Overstorey stem density (stems ha ⁻¹)	Density of all overstorey stems (>5cm dbh) recorded
Large stem density (stems ha ⁻¹)	Density of large overstorey stems (60cm dbh)
Hollows (stems ha ⁻¹)	Density of stems with hollows
Shrub height (m)	Mean height of vegetation < 5m above ground (includes shrubs, understorey trees and young eucalypts)
Shrub cover (%)	Visual estimate of projective cover of understorey vegetation
Ground cover (%)	Visual estimate of projective cover of ground vegetation (includes herbaceous, small woody plants and grasses)
Log cover (%)	Visual estimate of the proportion of ground covered by logs
Rock cover (%)	Visual estimate of the proportion of ground covered by rocks
Litter cover (%)	Visual estimate of the proportion of ground covered by leaves and other dead plant material
Litter depth (cm)	Maximum depth of litter cover over the ground
Bare earth (%)	Visual estimate of ground that does not have some form of cover
Mistletoe	Presence/absence of occurrence within plot
Grazing	Presence/absence of current or recent stock within plot
Water	Presence/absence of surface water (creek or farm dam) within plot

3.3 Results

Vegetation complexity

Woody stems

The structure of the woody vegetation varied considerably across the vegetation classes (Fig 3.1). Stem density was greatest in young plantings, peaking at an average of 700 stems ha⁻¹ in young, medium-sized plantings. Older plantings had markedly lower stem density, falling consistently in the range of 100-200 stems ha⁻¹. This density of vegetation matched that found in remnant vegetation, except for very large remnants, which supported a higher density of about 600 stems ha⁻¹. When only large stems (> 60 cm dbh) were considered, that is remnant trees, a different pattern was found. Not surprisingly, there were more large stems in remnant forest and woodland than in planted areas, where just an occasional tree had been retained on site. Within remnant vegetation, very large remnants had the lowest number of large stems, while the greatest density occurred in linear remnants. The distribution of large stems essentially explains the pattern of basal area. Basal area was low in plantings and higher in remnant vegetation, but with very large remnants having a similar basal area to plantings. The density of hollows also matched closely the density of large stems, indicating that hollows were visible in most of these stems.

Overstorey height increased with age of planting and from plantings to remnant vegetation. Plantings were usually 10-12 m tall, while remnant vegetation was on average 18-20 m tall. The lowest overstorey height occurred in young, small plantings, because mostly planted trees in this age-class were shorter than our definition of overstorey (> 5 m high), with exceptions provided by the very occasional presence of individual old remnant trees. The spread of overstorey cover was reasonably even across vegetation classes, excepting paddocks and young, small plantings. Overstorey cover was virtually absent from these latter two classes, except for that provided by single paddock trees. For the remaining classes, overstorey cover varied, on average, between 12 and 25 %.

Shrub height was very consistent across all vegetation classes, ranging from 3-4 m, except for paddocks, where a shrub layer was absent, and in small remnants, where shrubs were about one m tall. Shrub cover was very low in remnants, averaging < 5 % cover. Very large remnants were an exception, as some of the highest shrub cover values were recorded in this class (average = 17 %). Old plantings had about twice the shrub cover as remnants. There was a clear decrease in shrub cover among plantings from young plantings to old plantings. However, this difference was due largely to the low height of vegetation in the young plantings being allocated to the shrub cover height class (< 5 m).

Shannon-Weaver indices of diversity also varied considerably across vegetation classes. Diversity for all tree and shrub species combined was greatest in plantings and lowest in remnants, with very large remnants again being an exception in having higher diversity, equivalent to that of plantings. When diversity was calculated separately for eucalypts and non-eucalypts, an almost identical pattern was evident for eucalypt diversity. Non-eucalypt diversity was different: again plantings had the greatest diversity, but diversity was substantially lower in old, large plantings. Diversity was similarly low in all remnants, including very large remnants, indicating that remnants supported few non-eucalypt trees or shrubs.

Ground Layer Complexity

Vegetative ground cover was typically no more than a few centimetres high, with cover ranging from only 22-47 % across all vegetation classes, except for young, small plantings where there was, on average, 67 % cover. The generally low amount of vegetative cover was not compensated for by increased levels of rock or log cover. Rock cover was always low, while log cover was notably highest in very large remnants. Woody debris in the form of leaf litter was minimal at both paddocks and small remnants, being only 1-2 cm deep. Litter depth ranged from an average of 4-7 cm at other sites, with little obvious difference between sites. Very low vegetative cover is likely to have resulted from the interaction between drought conditions during the survey and the widespread grazing by stock (cattle or sheep). Indeed, often much of the ground was bare during sampling in 2002.

Although grazing was widespread throughout the study area, distinct differences between vegetation classes were notable. Young plantings were less likely to be grazed as they were usually fenced. Indeed, small plantings in this age class were always fenced. Among older plantings, 40-90% of sites were grazed, although linear plantings of this age-class were often fenced (80 %). Aside from paddocks, which were all grazed, grazing was most frequent in remnants (50-100 % of sites). Very large remnants were again an exception, as only about 10 % of sites showed recent evidence of grazing.

Mistletoe

Mistletoe was present in all vegetation classes in the study area. Its occurrence ranged from about 10 % of sites in young-medium plantings, old-linear plantings, small remnants and medium-sized remnants up to about 40 % of sites in old-medium and old-large plantings.

Topography

By far the majority of sites were located on lower topography (41 %), with the next most common topography sampled being flat (25 %). No peaks were sampled and just 12 % of sites were located on upper slopes or ridges.

Landscape context and attributes of sites

Calculated patch areas for study sites were consistent with the size categories used to stratify sites in our experimental design (Fig 3.1). The amount of wooded vegetation (remnants and plantings) within 5 km of a site was reasonably similar for all vegetation classes, being about 1000 ha or 13 % of the buffer (7854 ha). Very large remnants were an exception as these sites were surrounded by almost 3500 ha of wooded vegetation (45 % of the landscape). At a more local scale, the amount of woody vegetation within 0.5 km of a site was variable across vegetation classes. Study sites located in very large remnants were obviously surrounded by the greatest areas of wooded vegetation because remnants of this size filled all of the 0.5 km buffer (78.5 ha). Sites located in most other vegetation and patch-size classes were surrounded by about 10 ha (13 % of the buffer). Exceptions were old medium-sized and old large plantings (20 and 30 ha, respectively) and large remnants (40 ha) which were often embedded in areas having more vegetation nearby. Generally, however, this landscape has little of its original woody vegetation remaining, and most of that which does remain has been highly fragmented. The extent of local fragmentation (0.5 km buffer) was reflected by more edges surrounding old medium-sized and large plantings. These were surrounded by more than twice the edge of the majority of other sites. Small remnants and very large remnants were surrounded by the least amount of edge.

Multivariate analysis

The results of a multivariate analysis of 11 site structural variables are summarised by the ordination plot shown in Fig 3.2. Multidimensional scaling of sites resulted in three distinct groupings. The most separated group comprised seven paddock sites, including two with remnant trees. Habitat plots for these latter two sites did not encompass the location of the paddock tree and hence from a plot point of view they were paddocks without trees. Note that fauna sampling at these two sites targeted the paddock tree. The three remaining paddock sites with trees fell in a second grouping that contained all remnant sites. Only two planted sites fell within this remnant group; both were old, linear plantings. The remaining planted sites were spread as a cloud of sites surrounding the remnants, but distinct from paddocks. There was some separation of old plantings from young plantings in this ordination, with younger plantings being located in the top left of the plot, while older plantings were located to the right (top and bottom). The degree of overlap between these groups suggests that age of planting is not easily separated by structural attributes.

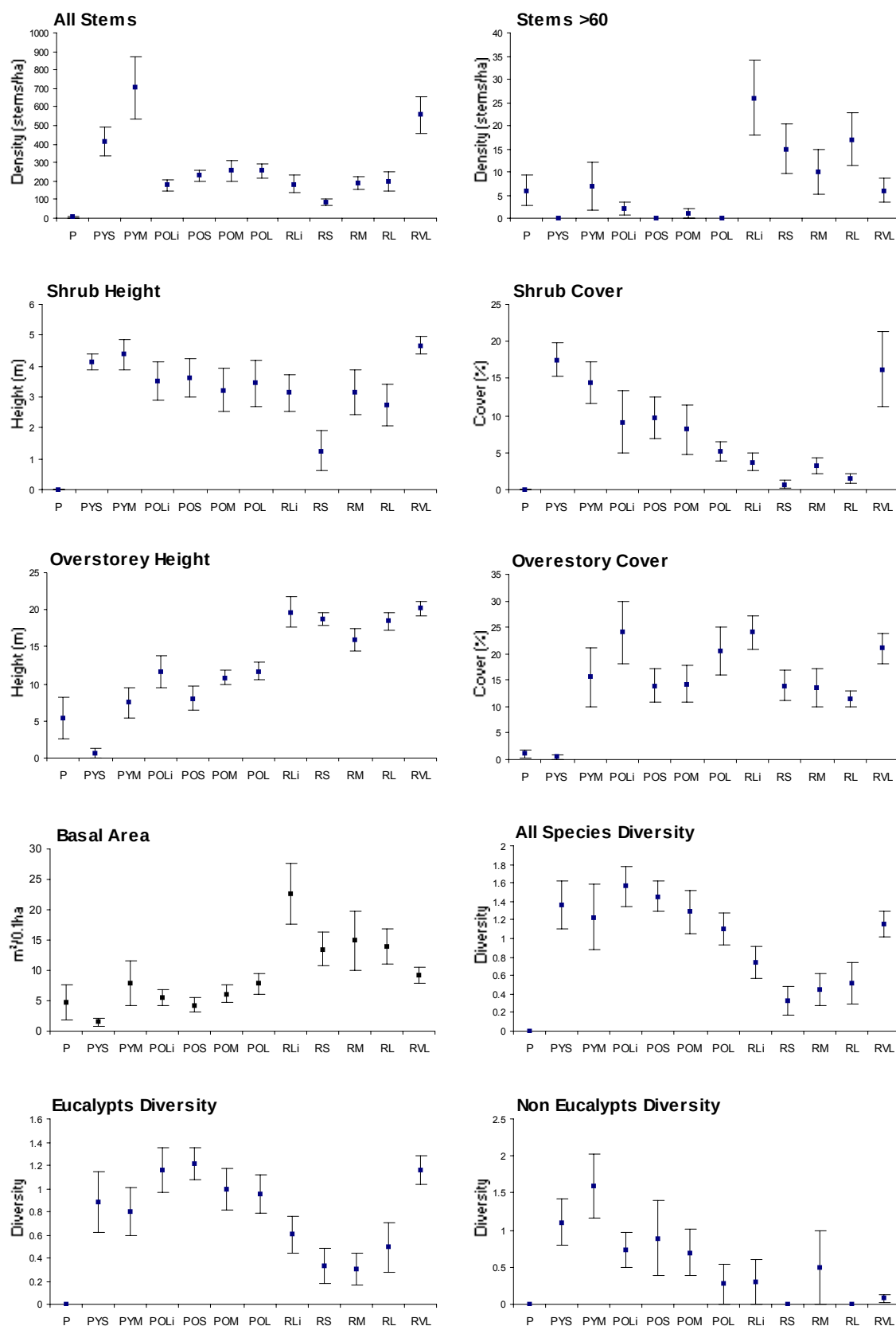


Fig 3.1 Plots of habitat and landscape attributes, and presence of grazing, for 12 vegetation classes in the study area (n=120 sites). See Table 2.1 for a description of codes. Values are either means $\pm$ SE or proportions if error bars are absent.

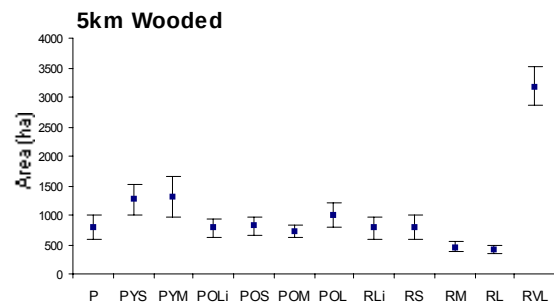
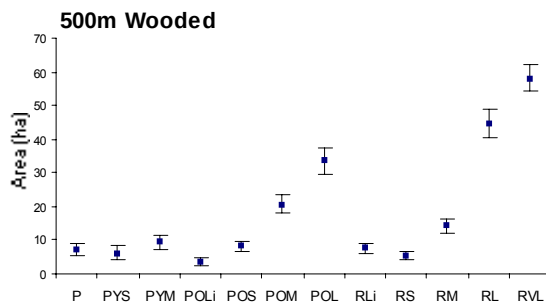
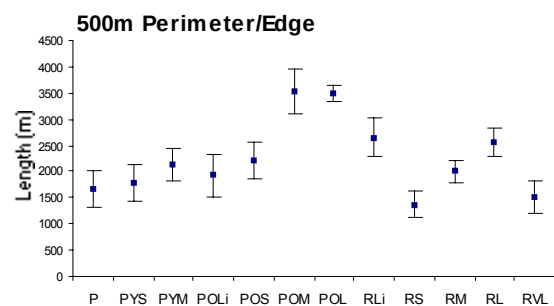
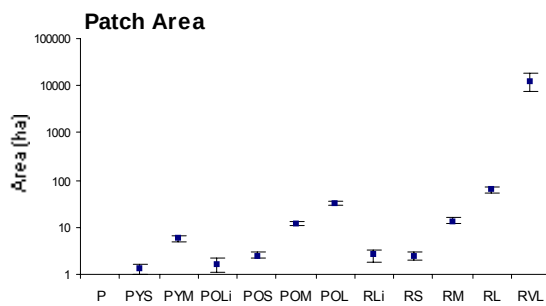
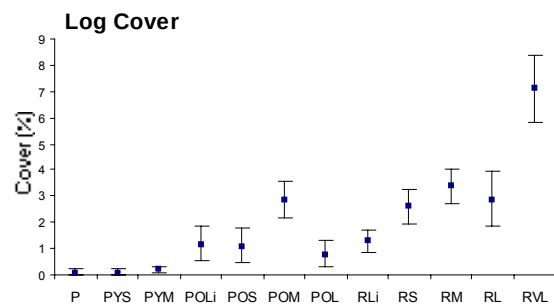
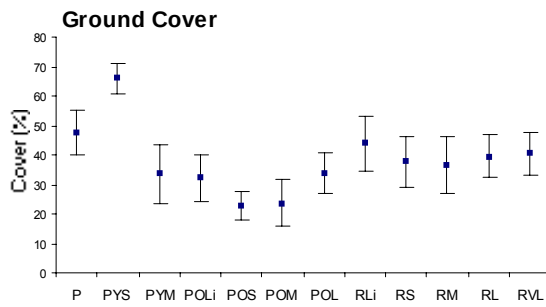
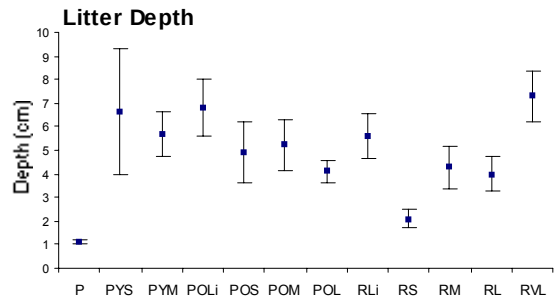
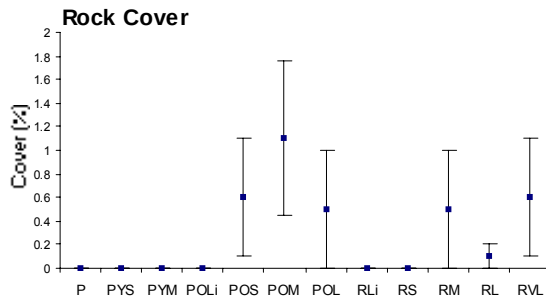
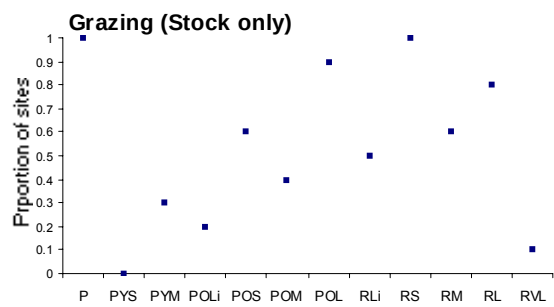
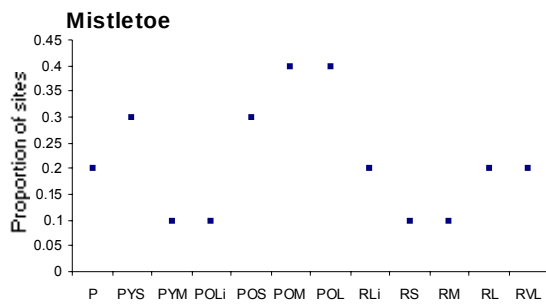


Fig 3.1 Continued

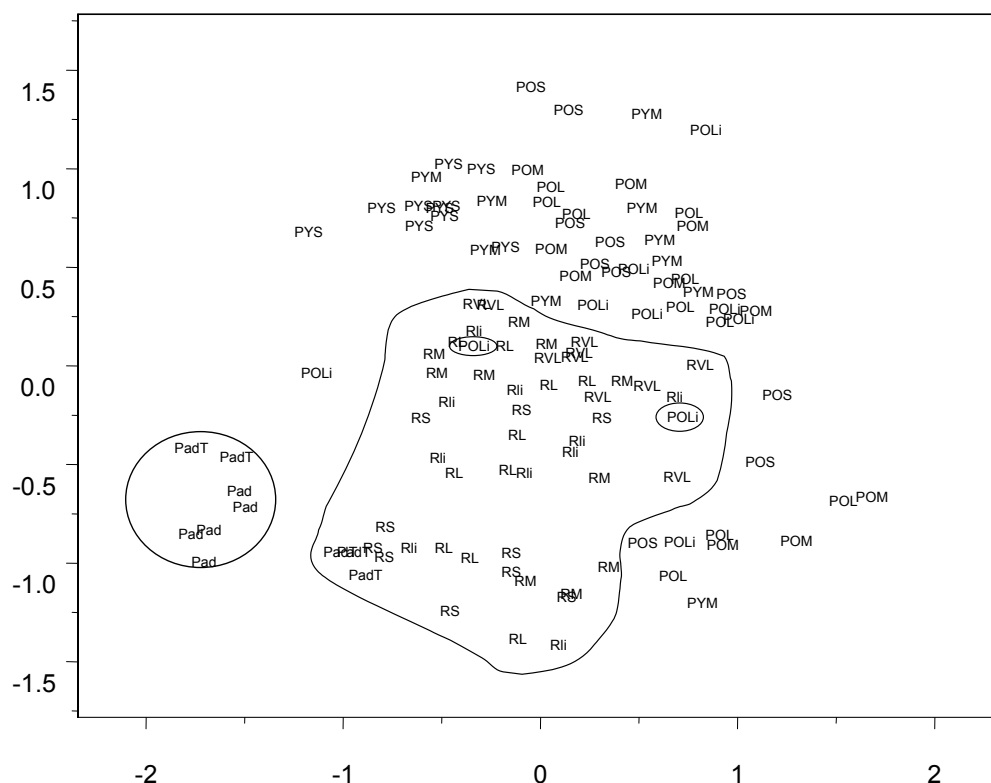


Fig 3.2 Two-dimensional ordination of paddock, planting and remnant sites (n=120) by 11 structural attributes (stress = 0.19) produced by multidimensional scaling.

3.4 Discussion

The primary finding from multidimensional scaling of our 120 sites was that paddocks, eucalypt plantings and remnants differ in terms of vegetation structure. Not surprisingly, plantings provided structural complexity well beyond that found in paddocks. The distinction between eucalypt plantings and remnants was blurred in that planted sites tended to surround the remnants grouping and indeed two old linear plantings fell within the remnants group. There was also a partial separation of older and younger plantings, indicating continuing structural changes to the vegetation as the plantings become older.

Univariate comparisons of structural attributes across vegetation classes indicated that the main distinguishing features can be summarised as follows. Young plantings had high stem density, high shrub (including young eucalypt) cover and ground cover as well as low log cover and overstorey cover. Old plantings had moderate overstorey cover. Remnants had the highest density of large stems and basal area but, except for very large remnants, remnants had lower tree species diversity than eucalypt plantings. While old eucalypt plantings had few structural attributes that distinguished them from both remnants and younger plantings, they were clearly lacking in large stems and log cover, which were a feature of remnants. It is also apparent that vegetation was quite patchy in many old plantings, with open patches being quite common. This is reflected by the decrease in stem density from 400-600 stems ha⁻¹ in young plantings to about 200 stems ha⁻¹ in older plantings. Vegetation patchiness probably resulted from mortality during past episodes of drought and drought-related grazing.

Very large remnants (> 1000 ha) differed in a number of ways to the general pattern found for remnants of smaller patch-size classes. Very large remnants usually had high shrub cover and log cover as well as low density of large stems. The low number of large stems in very large remnants

may be due to past logging in some areas (e.g. Woomargama and Benambra), but could also be due to differences in site-quality (i.e. productive capacity) between areas previously cleared and now planted, compared to those areas which have never been cleared. Support for the latter interpretation comes from the linear remnants, i.e. areas immediately adjacent to cleared areas. Linear remnants had the highest density of large stems, averaging 26 ha⁻¹, and may well have occurred on some of the most productive sites in the region. Other studies have also highlighted the importance of linear remnants, such as roadside strips, in providing large trees with an abundance of hollows. For example, van der Ree and Bennett (2001) found that the density of large trees (> 70 cm dbh) was 20 stems⁻¹ along roadsides of the northern plains region of Victoria.

The high ground vegetative cover in young plantings was related to the fact that they were usually fenced and therefore ungrazed. Indeed, small plantings in this age class were always fenced. However, about 30 % of young, medium-sized plantings were grazed, highlighting the increased costs of fencing larger areas and maintaining them ungrazed. Most of the older plantings and many of the remnants had reduced levels of ground cover and shrub cover due to grazing by domestic stock.

The amount of vegetation (forest, woodland and plantation) within buffers surrounding a site is considered to be one of the more reliable measures of landscape connectivity and isolation (Tischendorf *et al.* 2003). Connectivity of vegetation was similar across our vegetation classes, except for very large remnants. Also, the majority of sites were embedded in landscapes with similar levels of fragmentation, the exceptions being small remnants and very large remnants, which were surrounded on average by relatively less edge and therefore more fragmentation. It was also apparent that most of the landscape studied typically retained only a small percentage of its original vegetation (13 % when remnants and planted areas are included). This is similar to other studies of highly fragmented systems (Major *et al.* 2001, Radford and Bennett 2004) and is considered to be below the threshold to maintain the viability of a number of declining species (Radford and Bennett 2004).

4. Diurnal birds

4.1 Introduction

Relationships between bird species diversity and habitat fragmentation are better known for birds than for any other fauna group. Most of the early biogeographical studies, and the guiding principles developed from them, were based on the responses of bird species to habitat disturbance and fragmentation (e.g. MacArthur and Wilson 1967, Whitcomb *et al.* 1981, Harris 1984). While these and other studies emphasised the conservation value to fauna of remnants of native vegetation, it is now becoming increasingly well recognised that the quality of the surrounding matrix of vegetation has a major influence on species conservation at the landscape scale (e.g. Barrett *et al.* 1994, Wiens 1994). In Australia, eucalypt plantations (plantings) have the potential to significantly increase the quality of the vegetation matrix in predominantly agricultural landscapes, and thus to significantly improve the conservation value of existing native remnants.

The role of eucalypt plantations in providing habitat for forest and woodland fauna is best known for birds, but only few studies have been published (e.g. Woinarski 1979, Kavanagh and Turner 1994). These studies showed that bird species diversity was greatest in natural forest and in plantations where there were some remnant old trees. This was due to the broader range of specialised foraging and nesting resources available to, and essential for, some birds in older-aged forests. Nonetheless, eucalypt plantations without old trees provided suitable habitat for many bird species.

Very few studies have been reported on the biodiversity value of eucalypt plantings established on ex-pastoral sites in Australia (Kimber *et al.* 1999, Ryan 2000, Kavanagh *et al.* 2001, Law and Chidel 2002, Kinross 2004). In this Chapter, we report the relative abundance and species richness of birds occurring in large and small-sized eucalypt plantations near Albury-Wodonga, and compare these data with the bird assemblages occurring in nearby remnant native forest and woodlands and in paddocks used for grazing by domestic stock that are typical of the sites on which eucalypt plantations (plantings) were established in the region.

4.2 Methods

Sampling Technique

Systematic (fixed-time, fixed-area) counts were made for birds at two sampling points in each study site (n= 120 sites). A point-based ten-minute count was employed as the basic sampling unit for diurnal birds in this study. Two counts were made at each point, on different days and by different observers, during November-December 2002. Further sampling was undertaken at a sub-set of these sites and also at a range of new sites (n= 16) in late March-early April 2004 (see Chapter 8). The method required a single observer to stand quietly and record the total number of individuals of each species that occupied the surrounding area during the sample period. Identifying bird species by their calls is a crucial component of this method and helps reduce site bias that would result from a purely visual assessment. The data were recorded in four distance categories around each point: 0 to 10 m, 10 to 20 m, 20 to 30 m and 30 to 50 m. Individuals more than 50 m away were recorded as ‘in the sampled habitat’ or ‘in a different habitat’. Each bird recorded was placed in the distance category closest to the observer that it reached during the sampling period. Birds flying through the area were not recorded on the plots unless they were regarded as making some use of that habitat, e.g. foraging, nesting, sheltering or social interactions.

Analysis

Differences between “treatments” (vegetation type, patch-size and age-class categories) were assessed using pre-planned contrasts and poisson regression with a log link function (implemented in SAS using Proc Genmod; SAS Version 8.2). Dependent variables included the total numbers of species and of individuals recorded during the four 10 minute counts, and within a 50m radius of the two sampling plots, at each site. Analyses were also made on the components of these two variables that were designated as either “woodland-dependent” or “non woodland-dependent” birds

(see Table 4.1). Tests for the poisson distribution of these variables were undertaken by visual inspection of simple plots of the count data. Plots were also made of the mean counts (with standard errors) for species and individuals by treatment type.

Bird assemblages (counts of each species) at each sampling site were grouped using the two-way table function available in the pattern analysis program PATN (Belbin 1995). Data used in the analysis were mean counts for each species from the four 10 minute counts within the two 50 m radius sampling plots at each site. Species that were recorded at fewer than four of the 120 sites were excluded from the analysis. The data were $\log_{(10)}$ transformed to reduce the influence of high values for some species. Association between objects (ASO) was achieved using the Bray and Curtis measure for sampling sites and the Two-step measure for species (Belbin 1995). Agglomerative Hierarchical Fusion (FUSE) used Flexible UPGMA with a beta value of -0.1 for both sites and bird species. Bird species groups (11) and site groups (6) as presented in the two-way table were determined using dendrograms (DEND) that were visually assessed to specify the number of groups for Group Definitions (GDEF). A multi-dimensional scaling was also performed in PATN using the semi-strong hybrid (SSH) method. Modifications to standard default settings included the selection of 2 dimensions rather than three, the selection of the interval method instead of the ratio method, and 999 iterations were performed instead of the default 10 iterations.

Canonical correspondence analysis (CANOCO version 3.12; Ter Braak 1991) was used to investigate species relations with habitat variables. Fourteen key habitat variables, representing vegetation condition and resource availability, landscape context and management history, were selected after inspection of a matrix of correlations among a much larger set of habitat variables measured or assessed at each site. As for the PATN analyses, only species recorded on at least four of the 120 sites were used in the analysis, with the remainder of those species recorded within 50 m radius of the sampling plots designated as either “uncommon woodland-dependent” (UCWD) or “uncommon open country” (UCOC) species, providing a total of 74 “species” for analysis.

4.3 Results

A total of 138 bird species were recorded during this study (Table 4.1), but an additional eight species (mainly waterbirds and aviary escapees) were recorded during our earlier pilot study in the region (Kavanagh *et al.* 2001). These species have been classified as either “woodland dependent” (74 species) or “non-woodland dependent” (64 species) based on our own knowledge of species distributions and species ecology, but also using guidance from similar recent classifications of the same bird species in NSW (Reid 2000) and Victoria (Bennett and Ford 1997) (Table 4.1).

Species richness

One-hundred and ten species were recorded within 50 m radius at sampling plots within the main group of 120 sites during formal census counts (Table 4.2). These data show that the very large (>1000 ha) remnants (RVL) of forest and woodland in the region had the greatest mean number of bird species recorded and, not surprisingly, paddocks (PAD) had the least (Fig. 4.1). However, similar bird species counts were recorded within the smaller patches of remnant forest and woodland (RM+RS) and in older-aged eucalypt plantings (POM+POS) comparable in size (Table 4.3), including even the medium size-category of young eucalypt and shrub plantings (PYM) (Fig. 4.1). Linear strips of remnant vegetation (RLi) appeared to have more bird species than linear strips of older-aged eucalypt plantings (POLi), but these differences were not significant statistically (Fig. 4.1, Table 4.3). Paddocks with remnant trees (PadT) had many more bird species than paddocks without remnant trees (Pad). Increasing patch size in both remnant vegetation and in eucalypt plantings, both older-aged and younger, had a clear positive effect on the mean number of bird species recorded (Fig. 4.1, Table 4.3).

Table 4.1 Classification of the 138 bird species recorded in this study as either woodland-dependent or non woodland-dependent by Kavanagh and Stanton (K&S), compared to similar recent classifications for the same bird species in western NSW and Victoria by Reid (2000) and Bennett and Ford (1997).

Woodland dependant species					Non-woodland dependant species				
Common Name	Species	K&S	Reid	B&F	Common Name	Species	K&S	Reid	B&F
Brown Goshawk	<i>Accipiter fasciatus</i>	1	1	0	Stubble Quail	<i>Coturnix pectoralis</i>	0	0	0
Collared Sparrowhawk	<i>Accipiter cirrhocephalus</i>	1	1	0	Brown Quail	<i>Coturnix ypsilophora</i>	0	0	0
Painted Button-quail	<i>Turnix varia</i>	1	1	1	Australian Wood Duck	<i>Chenonetta jubata</i>	0	na	na
Bush Stone-curlew	<i>Burhinus grallarius</i>	1	1	1	Pacific Black Duck	<i>Anas superciliosa</i>	0	na	na
Common Bronzewing	<i>Phaps chalcoptera</i>	1	1	1	Hoary-headed Grebe	<i>Poliocephalus poliocephalus</i>	0	na	na
Peaceful Dove	<i>Geopelia striata</i>	1	1	1	Little Pied Cormorant	<i>Phalacrocorax melanoleucos</i>	0	na	na
Gang-gang Cockatoo	<i>Callocephalon fimbriatum</i>	1	0	0	White-faced Heron	<i>Egretta novaehollandiae</i>	0	na	na
Little Lorikeet	<i>Glossopsitta pusilla</i>	1	1	1	Nankeen Night Heron	<i>Nycticorax caledonicus</i>	0	na	na
Australian King Parrot	<i>Alisterus scapularis</i>	1	0	0	Australian White Ibis	<i>Threskiornis molucca</i>	0	na	na
Crimson/Yellow Rosella	<i>Platycercus elegans</i>	1	0	1	Straw-necked Ibis	<i>Threskiornis spinicollis</i>	0	na	na
Turquoise Parrot	<i>Neophema pulchella</i>	1	1	1	Black-shouldered Kite	<i>Elanus axillaris</i>	0	0	0
Brush Cuckoo	<i>Cacomantis variolosus</i>	1	0	0	Whistling Kite	<i>Haliastur sphenurus</i>	0	1	0
Fan-tailed Cuckoo	<i>Cacomantis flabelliformis</i>	1	1	1	Wedge-tailed Eagle	<i>Aquila audax</i>	0	0	0
Horsfield's Bronze-Cuckoo	<i>Chrysococcyx basalis</i>	1	1	1	Little Eagle	<i>Hieraetus morphnoides</i>	0	1	0
Shining Bronze-Cuckoo	<i>Chrysococcyx lucidus</i>	1	1	1	Brown Falcon	<i>Falco berigora</i>	0	0	0
Southern Boobook	<i>Ninox novaeseelandiae</i>	1	1	1	Australian Hobby	<i>Falco longipennis</i>	0	0	0
Barking Owl	<i>Ninox connivens</i>	1	1	1	Peregrine Falcon	<i>Falco peregrinus</i>	0	1	0
Australian Owllet-nightjar	<i>Aegotheles cristatus</i>	1	1	1	Nankeen Kestrel	<i>Falco cenchroides</i>	0	0	0
Azure Kingfisher	<i>Alcedo azurea</i>	1	na	1	Dusky Moorhen	<i>Gallinula tenebrosa</i>	0	na	na
Laughing Kookaburra	<i>Dacelo novaeguineae</i>	1	1	0	Masked Lapwing	<i>Vanellus miles</i>	0	na	0
Sacred Kingfisher	<i>Todiramphus sanctus</i>	1	1	1	Spotted Turtle-Dove	<i>Streptopelia chinensis</i>	0	0	0
Dollarbird	<i>Eurystomus orientalis</i>	1	1	1	Crested Pigeon	<i>Ocyphaps lophotes</i>	0	0	0
White-throated Treecreeper	<i>Cormobates leucophaeus</i>	1	1	1	Galah	<i>Cacatua roseicapilla</i>	0	0	0
Brown Treecreeper	<i>Climacteris picumnus</i>	1	1	1	Little Corella	<i>Cacatua sanguinea</i>	0	0	0
Spotted Pardalote	<i>Pardalotus punctatus</i>	1	1	1	Sulphur-crested Cockatoo	<i>Cacatua galerita</i>	0	0	0
Striated Pardalote	<i>Pardalotus striatus</i>	1	1	0	Cockatiel	<i>Nymphicus hollandicus</i>	0	0	0
White-browed Scrubwren	<i>Sericornis frontalis</i>	1	1	1	Eastern Rosella	<i>Platycercus eximius</i>	0	1	0
Speckled Warbler	<i>Chthonicola sagittata</i>	1	1	1	Australian Ringneck	<i>Barnardius zonarius</i>	0	0	0
Weebill	<i>Smicromis brevirostris</i>	1	1	1	Red-rumped Parrot	<i>Psephotus haematonotus</i>	0	1	0
Western Gerygone	<i>Gerygone fusca</i>	1	1	1	Budgerigar	<i>Melopsittacus undulatus</i>	0	0	0
White-throated Gerygone	<i>Gerygone olivacea</i>	1	1	1	Pallid Cuckoo	<i>Cuculus pallidus</i>	0	0	0
Brown Thornbill	<i>Acanthiza pusilla</i>	1	1	1	Barn Owl	<i>Tyto alba</i>	0	0	0
Buff-rumped Thornbill	<i>Acanthiza reguloides</i>	1	1	1	Tawny Frogmouth	<i>Podargus strigoides</i>	0	1	0
Yellow Thornbill	<i>Acanthiza nana</i>	1	1	1	Rainbow Bee-eater	<i>Merops ornatus</i>	0	0	0
Striated Thornbill	<i>Acanthiza lineata</i>	1	1	1	Superb Fairy-wren	<i>Malurus cyaneus</i>	0	1	0
Red Wattlebird	<i>Anthochaera carunculata</i>	1	1	1	Chestnut-rumped Heathwren	<i>Hylacola pyrrhopygia</i>	0	0	na
Noisy Friarbird	<i>Philemon corniculatus</i>	1	1	1	Yellow-rumped Thornbill	<i>Acanthiza chrysorrhoa</i>	0	1	0
Little Friarbird	<i>Philemon citreogularis</i>	1	1	1	Black Honeyeater	<i>Certhionyx niger</i>	0	0	0
Blue-faced Honeyeater	<i>Entomyzon cyanotis</i>	1	1	1	White-fronted Chat	<i>Epthianura albifrons</i>	0	0	0
Noisy Miner	<i>Manorina melanocephala</i>	1	1	0	Restless Flycatcher	<i>Myiagra inquieta</i>	0	1	0
Yellow-faced Honeyeater	<i>Lichenostomus chrysops</i>	1	1	1	Maggpie-lark	<i>Grallina cyanoleuca</i>	0	0	0
Fuscous Honeyeater	<i>Lichenostomus fuscus</i>	1	1	1	Willie Wagtail	<i>Rhipidura leucophrys</i>	0	0	0
White-plumed Honeyeater	<i>Lichenostomus penicillatus</i>	1	1	1	Black-faced Cuckoo-shrike	<i>Coracina novaehollandiae</i>	0	1	0
Black-chinned Honeyeater	<i>Melithreptus gularis</i>	1	1	1	Masked Woodswallow	<i>Artamus personatus</i>	0	0	0
Brown-headed Honeyeater	<i>Melithreptus brevirostris</i>	1	1	1	White-browed Woodswallow	<i>Artamus superciliosus</i>	0	1	0
White-naped Honeyeater	<i>Melithreptus lunatus</i>	1	1	1	Pied Butcherbird	<i>Cracticus nigrogularis</i>	0	0	0
Eastern Spinebill	<i>Acanthorhynchus tenuirostris</i>	1	1	1	Australian Magpie	<i>Gymnorhina tibicen</i>	0	1	0
Jacky Winter	<i>Microeca fascians</i>	1	1	1	Pied Currawong	<i>Strepera graculina</i>	0	0	0
Scarlet Robin	<i>Petroica multicolor</i>	1	1	0	Australian Raven	<i>Corvus coronoides</i>	0	1	0
Red-capped Robin	<i>Petroica goodenovii</i>	1	1	1	Little Raven	<i>Corvus mellori</i>	0	0	0
Flame Robin	<i>Petroica phoenicea</i>	1	0	0	Singing Bushlark	<i>Mirafra javanica</i>	0	0	0
Hooded Robin	<i>Melanodryas cucullata</i>	1	1	1	Richard's Pipit	<i>Anthus novaeseelandiae</i>	0	0	0
Eastern Yellow Robin	<i>Eopsaltria australis</i>	1	1	1	House Sparrow	<i>Passer domesticus</i>	0	0	0
Grey-crowned Babbler	<i>Pomatostomus temporalis</i>	1	1	1	Eurasian Tree Sparrow	<i>Passer montanus</i>	0	0	0
White-browed Babbler	<i>Pomatostomus superciliosus</i>	1	1	1	Double-barred Finch	<i>Taeniopygia bichenovii</i>	0	1	0
Varied Sittella	<i>Daphoenositta chrysoptera</i>	1	1	1	(Eurasian) Goldfinch	<i>Carduelis carduelis</i>	0	0	0
Crested Shrike-tit	<i>Falcunculus frontatus</i>	1	1	1	Welcome Swallow	<i>Hirundo neoxena</i>	0	0	0
Golden Whistler	<i>Pachycephala pectoralis</i>	1	1	1	Fairy Martin	<i>Hirundo ariel</i>	0	na	0
Rufous Whistler	<i>Pachycephala rufiventris</i>	1	1	1	Red-whiskered Bulbul	<i>Pycnonotus jocosus</i>	0	na	na
Grey Shrike-thrush	<i>Colluricincla harmonica</i>	1	1	1	Clamorous Reed-Warbler	<i>Acrocephalus stentoreus</i>	0	na	0
Leaden Flycatcher	<i>Myiagra rubecula</i>	1	1	1	Rufous Songlark	<i>Cincloramphus mathewsi</i>	0	0	0
Grey Fantail	<i>Rhipidura fuliginosa</i>	1	1	1	Brown Songlark	<i>Cincloramphus cruralis</i>	0	0	0
White-bellied Cuckoo-shrike	<i>Coracina papuensis</i>	1	1	1	Common Blackbird	<i>Turdus merula</i>	0	0	0
White-winged Triller	<i>Lalage sueurii</i>	1	1	1	Common Starling	<i>Sturnus vulgaris</i>	0	0	0
Olive-backed Oriole	<i>Oriolus sagittatus</i>	1	1	1					
Dusky Woodswallow	<i>Artamus cyanopterus</i>	1	1	1					
Grey Butcherbird	<i>Cracticus torquatus</i>	1	1	0					
White-winged Chough	<i>Corcorax melanorhamphos</i>	1	1	1					
Satin Bowerbird	<i>Ptilonorhynchus violaceus</i>	1	na	0					
Red-browed Finch	<i>Neochmia temporalis</i>	1	1	0					
Diamond Firetail	<i>Stagonopleura guttata</i>	1	1	1					
Mistletoebird	<i>Dicaeum hirundinaceum</i>	1	1	1					
Tree Martin	<i>Hirundo nigricans</i>	1	na	0					
Silvereye	<i>Zosterops lateralis</i>	1	1	0					

Table 4.2 Numbers of birds recorded within 50m radius of sampling plots within each treatment category during formal census counts. The total number of sites (n=120) where species were recorded within 50m radius is also indicated.

Species	Paddock	Young plantation		Old plantation			Remnant forest and woodland						Total inds.	Total sites
		Small < 5ha	Medium 5-20 ha	linear < 50m	Small < 5ha	Medium 5-20 ha	large 20-100 ha	linear < 50m	small < 5ha	medium 5-20 ha	large 20-100 ha	v. large > 1000 ha		
Australian Magpie	37	36	17	39	30	31	38	19	19	28	34	20	348	90
Willie Wagtail	0	15	16	28	50	39	33	21	29	40	21	16	308	85
Eastern Rosella	14	16	21	33	32	41	31	35	45	39	46	18	371	84
White-plumed Honeyeater	1	32	65	55	123	122	98	62	92	138	105	29	922	81
Striated Pardalote	7	7	9	15	14	8	12	47	39	21	33	12	224	73
Red-rumped Parrot	17	26	15	7	63	40	9	12	42	73	22	10	336	62
Superb Fairy Wren	1	44	52	59	51	31	56	24	5	23	19	31	396	59
Galah	27	14	28	19	13	4	6	34	65	72	35	6	323	56
Grey Shrike-thrush	0	3	3	9	17	17	16	7	3	9	16	25	125	56
Rufous Whistler	0	5	39	2	14	32	51	4	0	2	16	33	198	54
White-browed Woodswallow	15	11	42	5	80	115	97	32	102	72	68	9	648	53
Black-faced Cuckoo-shrike	0	0	6	2	9	3	2	13	7	11	16	18	87	45
Grey Fantail	0	5	75	4	11	20	31	2	0	1	10	37	196	40
Noisy Miner	15	6	2	33	14	25	44	37	46	42	46	0	310	39
Welcome Swallow	14	10	2	4	7	11	2	1	18	7	3	2	81	38
Australian Raven	2	0	1	7	3	10	9	12	10	16	3	6	79	37
Magpie-lark	2	7	3	3	6	4	8	4	14	5	10	1	67	34
Common Starling	15	22	5	8	8	12	1	17	42	24	4	0	158	32
Crimson/Yellow Rosella	1	20	7	8	19	5	12	3	7	10	4	11	107	31
Mistletoebird	3	2	12	0	1	3	9	5	2	4	4	18	63	31
Crested Shrike-tit	0	0	3	0	7	6	9	3	3	6	10	11	58	28
Red Wattlebird	0	0	3	8	4	12	29	1	2	2	3	1	65	27
White-throated Gerygone	0	1	9	0	3	0	7	4	1	2	4	15	46	27
Brown Treecreeper	0	1	0	0	0	0	0	11	11	23	28	27	101	26
Sacred Kingfisher	0	0	5	0	1	9	4	11	7	2	17	22	78	26
Laughing Kookaburra	0	1	2	5	2	3	3	4	3	4	5	4	36	26
Restless Flycatcher	0	0	2	0	5	1	10	3	7	8	6	8	50	25
White-winged Chough	0	8	16	6	6	17	29	40	14	12	5	22	175	24
Crested Pigeon	3	0	2	6	3	4	1	5	10	6	3	0	43	23
Little Friarbird	0	0	6	7	2	0	1	5	1	7	9	3	41	23
Yellow Thornbill	0	4	32	12	30	19	45	16	0	0	9	0	167	20
Weebill	1	1	2	0	10	29	34	4	0	2	8	8	99	20
Spotted Pardalote	0	0	4	5	3	7	10	0	0	1	0	18	48	20
Red-browed Finch	0	24	35	0	9	0	5	11	0	10	0	6	100	19
Peaceful Dove	0	1	2	1	5	13	15	0	1	2	7	5	52	19
White-throated Treecreeper	1	0	0	0	3	6	0	0	0	1	7	20	38	19
Dusky Woodswallow	0	0	0	0	2	0	3	3	3	17	13	32	73	17
Brown-headed Honeyeater	0	1	2	18	7	3	3	3	0	0	0	22	59	16
Jacky Winter	1	0	0	0	6	2	0	0	1	3	14	14	41	16
Varied Sittella	0	0	0	0	0	12	8	2	2	0	2	22	48	15
Rufous Songlark	0	3	1	0	0	4	3	9	0	10	9	3	42	15
Yellow-rumped Thornbill	2	4	6	4	10	2	2	2	4	2	2	0	40	15
Striated Thornbill	0	0	33	9	0	33	34	4	0	0	4	13	130	14
Silvereye	3	0	62	4	0	4	18	0	0	0	0	5	96	14
Eastern Yellow Robin	0	0	2	0	0	2	0	0	0	0	4	21	29	14
Common Blackbird	0	0	9	0	1	1	4	1	0	4	0	0	20	13
Tree Martin	7	3	1	0	1	0	25	0	39	5	0	0	81	12
Yellow-faced Honeyeater	2	0	25	6	0	4	1	0	0	0	0	21	59	12
Rainbow Bee-eater	0	0	6	0	0	1	1	0	5	4	4	0	21	12
Fuscous Honeyeater	0	0	1	0	0	0	6	0	0	0	18	116	141	11
Buff-rumped Thornbill	0	0	2	0	0	12	19	0	0	2	0	15	50	11
Brown Thornbill	0	0	20	0	0	0	4	2	0	0	4	30	60	10
Noisy Friarbird	0	15	4	1	0	2	2	0	0	3	0	3	30	10
Goldfinch	0	4	15	2	2	0	13	0	0	2	0	0	38	9
Sulphur-crested Cockatoo	0	2	10	4	0	0	0	0	7	3	1	1	28	9
White-winged Triller	1	0	0	0	3	0	7	0	3	0	2	2	18	9
Common Bronzewing	0	0	4	1	1	0	0	0	0	0	4	4	14	9
Dollarbird	0	0	1	0	0	0	0	3	3	1	1	0	9	8
House sparrow	1	2	0	33	1	4	0	0	0	0	0	0	41	7
Olive-backed Oriole	0	0	0	0	0	0	0	1	0	0	1	18	20	7
Masked Woodswallow	1	0	0	0	1	2	1	0	4	0	2	0	11	7
Horsfields Bronze Cuckoo	0	0	1	0	2	3	0	0	4	0	0	0	10	7
Speckled Warbler	0	0	6	0	0	0	7	0	0	0	0	3	16	6
Brown Songlark	0	3	1	0	2	0	2	0	0	0	0	0	8	5
Red-capped Robin	0	0	1	0	1	2	4	0	0	0	0	0	8	5
Pied Currawong	0	0	1	0	0	1	1	0	0	0	0	3	6	5
Fairy Martin	33	0	0	0	0	3	0	0	14	8	0	0	58	4
White-browed Scrubwren	0	0	2	0	0	8	0	3	0	0	4	0	17	4
White-bellied Cuckoo-shrike	0	0	0	0	1	0	1	0	0	0	0	5	7	4
Brown Goshawk	0	0	2	0	0	0	1	0	0	1	0	0	4	4

Species	Paddock	Young plantation		Old plantation			Remnant forest and woodland						Total inds.	Total sites
		Small < 5ha	Medium 5-20 ha	linear < 50m	Small < 5ha	Medium 5-20 ha	large 20-100 ha	linear < 50m	small < 5ha	medium 5-20 ha	large 20-100 ha	v. large > 1000 ha		
Shining Bronze Cuckoo	0	0	0	1	0	1	1	0	0	0	0	1	4	4
Leaden Flycatcher	0	0	0	0	0	1	1	0	0	0	1	1	4	4
Little Lorikeet	0	0	0	0	6	2	0	0	0	0	0	3	11	3
Cockatiel	4	0	5	0	0	0	0	0	0	0	0	0	9	3
Pallid Cuckoo	0	0	0	0	0	0	3	0	3	0	0	0	6	3
Australian Hobby	0	0	0	0	0	0	0	0	5	0	0	0	5	3
Golden Whistler	0	0	5	0	0	0	0	0	0	0	0	0	5	3
Western Gerygone	0	0	1	0	0	0	1	0	0	0	0	1	3	3
Black Honeyeater	0	0	1	0	14	0	0	0	0	0	0	0	15	2
White-naped Honeyeater	0	0	0	0	2	0	0	0	0	0	0	3	5	2
Australian White Ibis	0	0	0	0	0	0	1	2	0	0	0	0	3	2
Blue-faced Honeyeater	0	0	0	1	0	0	1	0	0	0	0	0	2	2
Nankeen Kestrel	1	0	0	0	0	0	0	0	0	1	0	0	2	2
Pacific Black Duck	0	0	0	0	0	0	0	1	1	0	0	0	2	2
Australian Owllet-nightjar	0	0	0	0	0	0	0	0	0	0	0	2	2	2
Little Raven	0	0	0	0	0	0	0	0	1	0	1	0	2	2
Grey-crowned Babbler	0	0	0	0	0	0	0	0	7	0	0	0	7	1
Diamond Firetail	0	0	6	0	0	0	0	0	0	0	0	0	6	1
Double-barred Finch	0	0	0	0	0	0	6	0	0	0	0	0	6	1
White-browed Babbler	0	0	0	0	0	0	0	0	0	0	0	4	4	1
Clamorous Reed-warbler	0	0	0	0	0	0	0	3	0	0	0	0	3	1
Little Cormorant	0	0	2	0	0	0	0	0	0	0	0	0	2	1
Australian Wood Duck	0	0	0	0	0	0	0	0	0	0	2	0	2	1
Black-chinned Honeyeater	0	0	0	0	0	0	0	0	0	0	0	2	2	1
Eastern Spinebill	0	0	0	0	0	0	0	0	0	0	0	2	2	1
Masked Lapwing	0	0	0	0	0	0	0	0	2	0	0	0	2	1
Painted Button-quail	0	0	0	0	0	0	0	0	0	0	0	2	2	1
Wedge-tailed Eagle	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Brown Falcon	0	0	1	0	0	0	0	0	0	0	0	0	1	1
Collared Sparrowhawk	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Little Eagle	0	0	0	0	0	1	0	0	0	0	0	0	1	1
Fan-tailed Cuckoo	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Pied Butcherbird	1	0	0	0	0	0	0	0	0	0	0	0	1	1
Spotted Turtle dove	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Brush Cuckoo	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Brown Quail	0	0	0	0	0	0	0	1	0	0	0	0	1	1
Chestnut-rumped Heathwren	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Eurasian Tree Sparrow	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Nankeen Night Heron	0	0	1	0	0	0	0	0	0	0	0	0	1	1
Peregrine Falcon	0	0	0	0	0	0	1	0	0	0	0	0	1	1
Grey Butcherbird	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Richard's Pipit	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Straw-necked Ibis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Whistling Kite	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hooded Robin	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Turquoise Parrot	0	0	0	0	0	0	0	0	0	0	0	0	0	0
White-faced Heron	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Black-shouldered Kite	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gang-gang Cockatoo	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Red-whiskered Bulbul	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Scarlet Robin	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Singing Bushlark	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tawny Frogmouth	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Azure Kingfisher	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Budgerigar	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Total species	30	35	67	39	53	56	66	48	47	50	57	66	110	
Total inds.	233	359	783	474	721	809	952	549	755	791	732	850	8008	

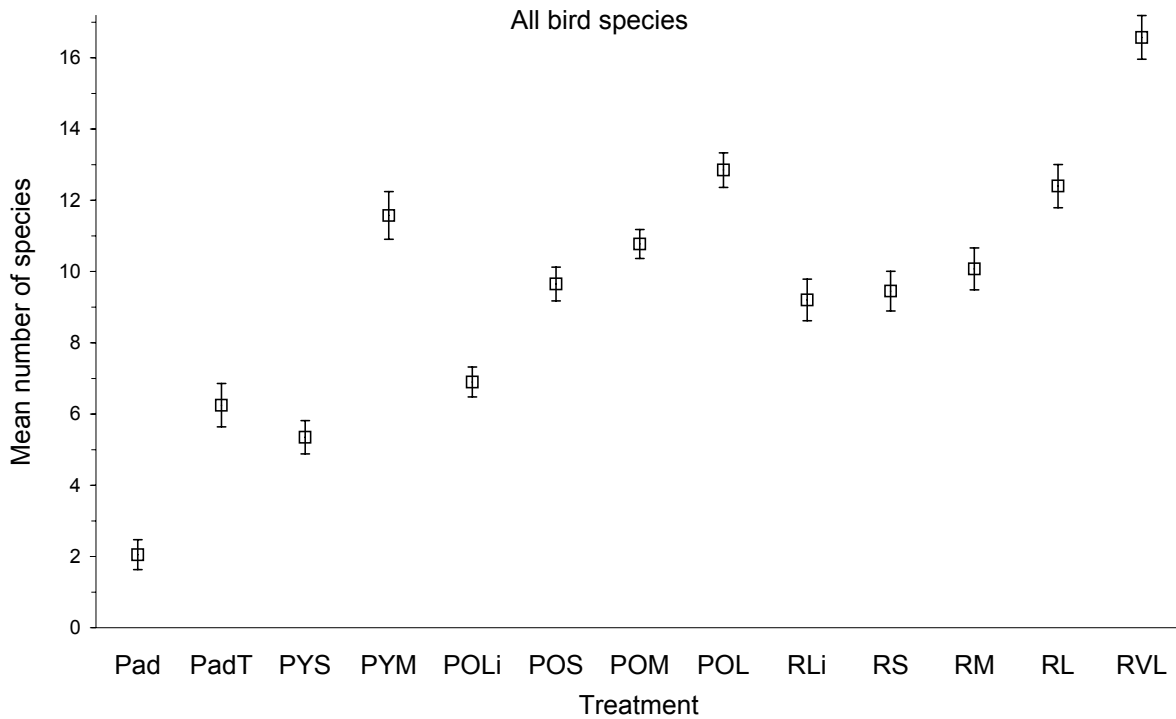


Fig. 4.1 Mean numbers of bird species (with standard errors) counted in 10 minute, 50 m radius sampling plots (n=40) in each treatment category. See text for treatment codes. The data for “paddocks” have been split to illustrate differences between paddock sites with (PadT) and without (Pad) remnant trees.

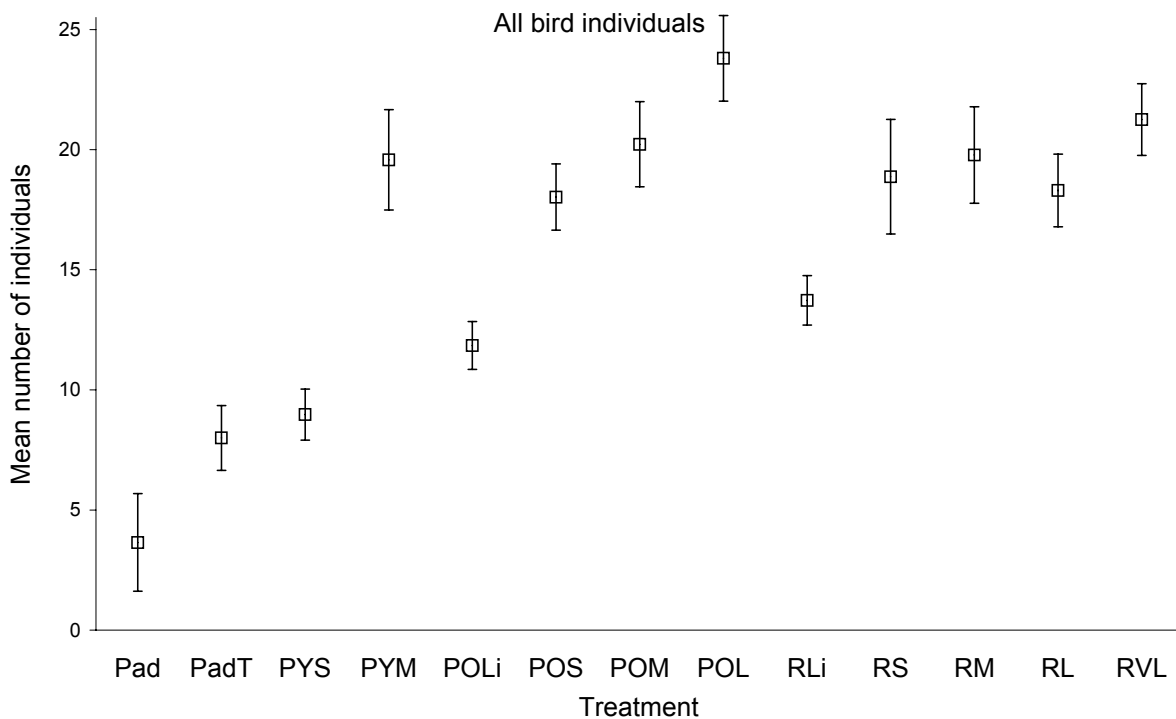


Fig. 4.2 Mean numbers of individual birds (with standard errors) counted in 10 minute, 50 m radius sampling plots (n=40) in each treatment category. See text for treatment codes. The data for “paddocks” have been split to illustrate differences between paddock sites with (PadT) and without (Pad) remnant trees.

Table 4.3 Results of pre-planned statistical contrasts between treatments for total numbers of species and individuals. Data are Chi-square values and their significance levels (at df. 1). ***, $P < 0.01$; **, $P < 0.05$. WD, woodland-dependent. Codes for treatments indicated in text.

Contrast	Variable					
	Non WD species	WD species	Total species	Non WD birds	WD birds	Total birds
PAD vs POM	1.4	31.6***	24.3***	13.7***	110.0***	109.5***
PAD vs PYM	1.1	18.6***	13.2***	0.3	14.9***	5.5**
PAD vs RM	2.9	19.5***	19.0***	6.8***	20.5***	26.2***
RM+RS vs PYM+PYS	11.3***	9.6***	19.6***	38.5***	5.5**	33.2***
RM+RS vs POM+POS	3.1	11.8***	1.2	1.8	48.3***	16.6***
POM+POS vs PYM+PYS	2.5	42.1***	30.5***	23.8***	83.4***	95.9***
POL+POM+POS +POLi vs RL+RM+RS+Rli	12.6***	64.9***	75.6***	0.3	117.8***	87.4***
POLi vs Rli	0.8	2.4	0.1	5.8**	2.1	0.8
PYM vs PYS	1.8	72.2***	47.8***	0.1	129.2***	76.7***
POM vs POS	0.8	8.0***	7.3***	0.8	26.6***	11.6***
RM vs RS	0.3	25.6***	9.6***	27.4***	9.9***	2.3

Bird abundance

Roughly similar numbers of birds were recorded in remnants (very large, large, medium and small), older eucalypt plantings (large, medium and small) and young (medium-sized) eucalypt plantings (Fig. 4.2), although significant differences occurred within vegetation types due to patch size (Table 4.3). Both linear remnants and linear older plantings had fewer, but similar bird counts. Small patches of young plantings (PYS) contained fewer birds, similar to paddocks with remnant trees, while paddocks without remnant trees had the least number of birds recorded (Fig. 4.2, Table 4.3).

Many species were recorded more frequently in some treatments than others. Species attaining their greatest abundance in remnant forest and woodland compared to other treatment types included Brown Treecreeper, Fuscous Honeyeater, Dusky Woodswallow, White-throated Treecreeper and Sacred Kingfisher (Table 4.2). While the Noisy Miner and Common Starling occurred commonly in each vegetation type, they also were most common in remnants, especially the smaller patches but, interestingly, neither species was recorded in very large remnants. Species displaying their greatest abundance in older plantings included Yellow Thornbill, Weebill and Striated Thornbill (Table 4.2). Species most common in young plantings included Silvereye and Red-browed Finch. The Fairy Martin was patchy in its occurrence, but this species attained its greatest abundance at one paddock site.

Woodland-dependent vs non woodland-dependent birds

Bird assemblages also differed markedly between “treatments”, with patch size appearing to have the greatest influence (Fig. 4.3 a,c). The largest patches of either remnant forest and woodland, or old and young eucalypt plantings, had the most woodland-dependent birds and bird species. As expected, the very largest remnants had the greatest mean number of woodland dependent species (Fig. 4.3 a). Paddocks without remnant trees had no woodland-dependent species. Most treatments

had similar numbers of non woodland-dependent species, including the remnants (Fig. 4.3 c). Surprisingly, linear strips of old eucalypt plantings and remnants had fewer non woodland-dependent birds than larger size-classes of these two treatments (Fig. 4.3 d). Among older plantings and remnants, there was an inverse relationship between patch size and the numbers of non woodland-dependent species recorded, especially for remnants (Fig. 4.3 d, Table 4.3).

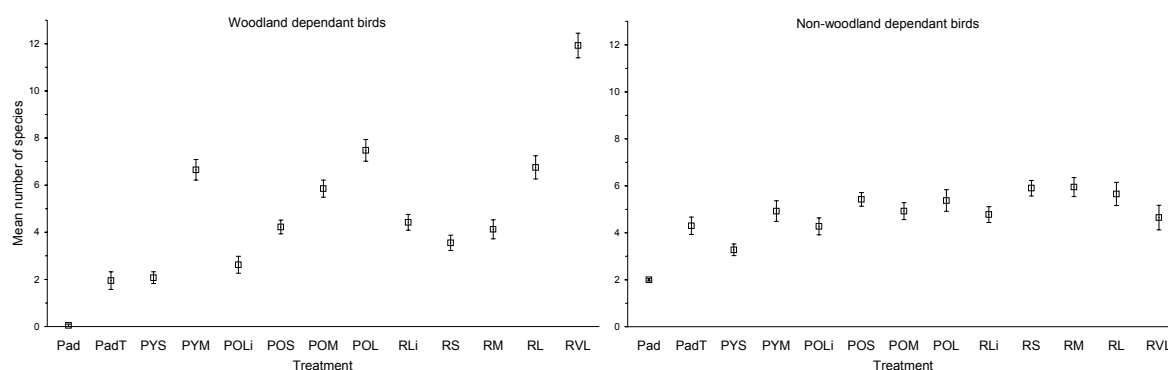


Fig. 4.3 a,b Mean numbers of woodland-dependent and non woodland-dependent bird species (with standard errors) counted in 10 minute, 50 m radius sampling plots (n=40) in each treatment category. See text for treatment codes. The data for “paddocks” have been split to illustrate differences between paddock sites with (PadT) and without (Pad) remnant trees.

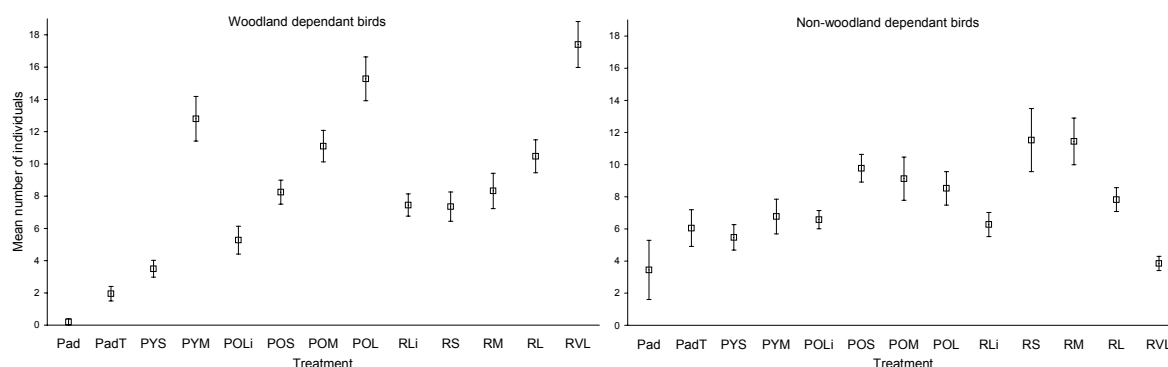


Fig. 4.3 c,d. Mean numbers of woodland-dependent and non woodland-dependent individual birds (with standard errors) counted in 10 minute, 50 m radius sampling plots (n=40) in each treatment category. See text for treatment codes. The data for “paddocks” have been split to illustrate differences between paddock sites with (PadT) and without (Pad) remnant trees.

Species groups

Four main species groupings were recognised, using the PATN dendrogram, based on the occurrences of each bird species on the 120 sampling sites. Broadly, group 1 (Australian Magpie to Yellow-rumped Thornbill) consists of a suite of common, open-country inhabiting, mainly non woodland-dependent species (Table 4.4). Five species in this group, Striated Pardalote, White-plumed Honeyeater, White-winged Chough, Noisy Miner and Tree Martin, have been classified as woodland-dependent but these species are widespread in degraded woodlands (ones where the understorey is lacking) in agricultural landscapes. Group 2 (Black-faced Cuckoo-shrike to Noisy Friar Bird) consists of common, mainly woodland-dependent species, the only exceptions being three

species, Black-faced Cuckoo-shrike, Rufous Songlark and Restless Flycatcher, two of which were regarded as woodland-dependent by Reid (2000).

The third main species grouping is an amalgam of four sub-groups (from Brown Songlark to House Sparrow) consisting of uncommon, mainly non woodland-dependent species, but also including three uncommon woodland-dependent species (White-browed Scrubwren, Dollarbird and Horsfield's Bronze-Cuckoo) (Table 4.4). The fourth main group is an amalgam of five sub-groups (from Brown-headed Honeyeater to Shining Bronze-Cuckoo), nearly all of which (excepting only Pied Currawong) are regarded as woodland-dependent species. Indeed, most of the species in this group are also common forest birds. Three of the sub-groups were comprised of only one or two species, all of which were uncommon in the study area (Table 4.4).

Site groups

Six site groupings were recognised based on the composition of their bird assemblages (Table 4.4). The majority of sites (111) were placed into three large groups, comprising 21, 62 and 28 sites, respectively, for groups 1, 2 and 3. All three main site groups were heterogenous with respect to treatment, comprising representatives from each of the forest / woodland remnant and eucalypt plantation vegetation types and also included most patch size-classes and plantation age-classes (Table 4.4). For example, remnants comprised 43%, 37% and 64%, respectively, of sites in groups 1, 2 and 3. The remaining three site groups (4, 5 and 6) were very small, comprising only 2, 5 and 2 sites, respectively. All nine of these sites were paddocks, mostly without any remnant trees, or young, small (<5ha), planted areas. Inspection of the dendrogram suggested that site groups 4, 5 and 6 could easily be merged into one group based on the similarity of their bird species assemblages. A similar result was obtained from the semi-strong hybrid multi-dimensional scaling analysis implemented in PATN, suggesting that the main "outliers" from the bulk of the sites sampled in this study were the cleared paddocks but also included some of the young, small planted areas.

Species relations with habitat variables

Seventy-two species were recorded within 50 m of sampling plots during formal census counts on four or more of the main group of 120 sites. The remaining 38 species (of 110 species) were grouped as either “uncommon woodland-dependent” (UCWD) or “uncommon open country” (UCOC) species, providing a total of 74 “species” for analysis. Fourteen key habitat variables representing vegetation condition and resource availability (log cover, number of trees with large hollows, overstorey cover, number of stems less than 60 cm diameter, shrub cover, overstorey species diversity, presence of mistletoe, amount of bare earth), landscape context (amount of remnant forest and woodland within 500m radius, amount of remnant forest and woodland within 5 kilometre radius, amount of eucalypt and shrub plantings within 500m radius, local topography) and management history (presence of stock grazing, vegetation age class) were measured or assessed at each site to explain the patterns of species occurrence in relation to their physical and biotic environment.

The variables most responsible for structuring bird assemblages in this study were the landscape context variables, including the amount of remnant vegetation within 5 km, the amount of remnant vegetation within 500 m and the amount of eucalypt plantings within 500 m, together with the site-specific variables of vegetation age class and the amount of log cover (Fig. 4.4). Other important variables (which were also inversely-related) included the amount of shrub cover (and number of stems less than 60 cm diameter) and the presence of grazing by sheep, cattle or horses. Woodland-dependent species (indicated in bold type) were shown to be associated more strongly with ungrazed sites occurring in landscapes having a higher proportion of remnant vegetation than non woodland-dependent species (Fig. 4.4). Species strongly associated with the presence of older forest patches embedded in landscapes having a higher proportion of remnant forest and woodland include Olive-backed Oriole, White-bellied Cuckoo-shrike, Fuscous Honeyeater, Eastern Yellow Robin, White-throated Treecreeper, Brown Treecreeper, Jacky Winter, Dusky Woodswallow, Brown Thornbill and Buff-rumped Thornbill. Conversely, species associated with disturbed or more typically agricultural landscapes include Welcome Swallow, Common Starling, Masked Woodswallow, Crested Pigeon, Red-rumped Parrot, Galah and Fairy Martin. Interestingly, several woodland-dependent species are also included in this group including Dollarbird, Tree Martin, Noisy Miner, Horsfields Bronze Cuckoo, Striated Pardalote and White-plumed Honeyeater, none of which are disadvantaged by the absence of an understorey layer (Fig. 4.4). Bird species characteristic of landscapes having a higher proportion of planted forest include Red-capped Robin, Brown Goshawk, Silvereye, Yellow Thornbill, Speckled Warbler and Striated Thornbill, all of which are regarded as woodland-dependent species (Fig. 4.4). Other species associated with planted landscapes include European Goldfinch and Common Blackbird.

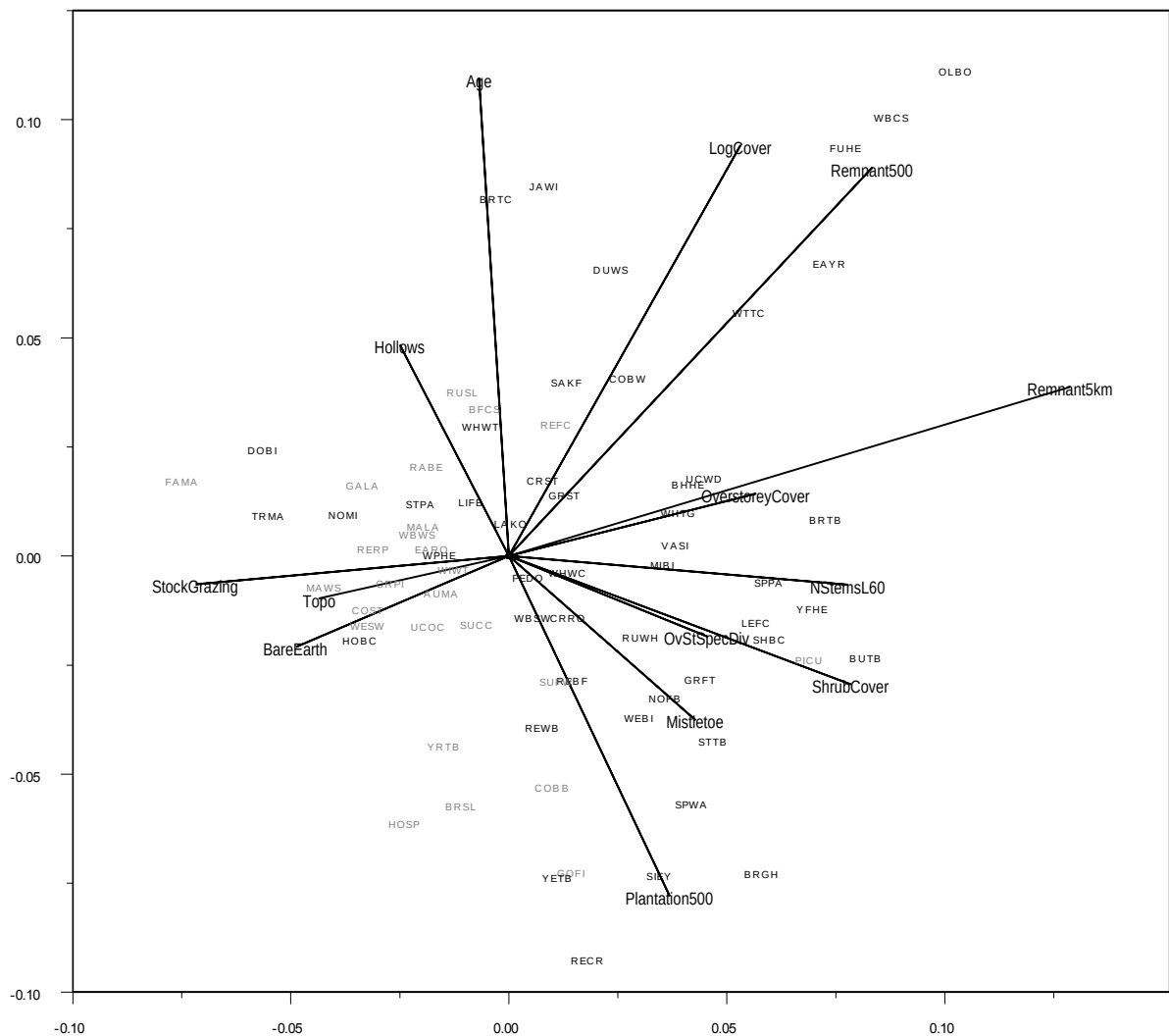


Fig. 4.4 Canonical correspondence analysis of the relations between 74 bird species or species groups and 14 habitat or landscape variables assessed at 120 sites in the Albury-Wodonga region. The length of the rays represents the variance explained by each variable and the distance along a ray subtended perpendicularly by each species represents the degree of association with that variable.

4.4 Discussion

Birds that are characteristic of native forests and woodlands (e.g. woodland-dependent species) were surprisingly widespread throughout the eucalypt plantings and the remnants sampled, indicating that eucalypt plantations have the potential to make a significant contribution to conserving bird species in agriculturally-dominated landscapes. This result appears to be a substantially new and welcome finding (see also Kinross 2004), and may be attributable to the much larger plantation patch sizes and the generally older plantation age-classes sampled in this study compared to the few other previously reported studies (Ryan 2000). This is good news because it shows that eucalypt and shrub plantings can contribute significantly to habitat restoration for birds within predominantly cleared landscapes. Landscape context, in particular the amount of remnant forest and woodland nearby, is also likely to be an important variable explaining these results, compared to other studies, and this issue is explored in more detail in Chapter 8.

Several species, including the Brown Treecreeper, Sacred Kingfisher and Dusky Woodswallow, were clearly more common in remnants. Significantly, the Brown Treecreeper is listed as a vulnerable species under the NSW Threatened Species Conservation Act (1995). However, eucalypt plantings clearly appeared to provide the best habitat for a range of other species, including the Yellow Thornbill and Silvereye. Indeed, several species listed as vulnerable, including the Speckled Warbler and Regent Honeyeater (observed nearby), were mainly recorded in eucalypt plantations. While there is now strong evidence of the use of eucalypt plantings by many bird species as foraging habitat, the extent to which these plantings can provide nest sites for these birds is unknown and in need of further investigation.

The pre-planned statistical contrasts clearly provided a powerful test of the differences between specified vegetation categories (treatments). For example, the PATN analyses reported a reasonably high level of heterogeneity for vegetation classes among the site groupings, and also plots of the treatment means (with standard errors) suggested that there may be more overlap (fewer differences) between some treatments. Consequently, there was some concern that control over the family-wise Type 1 error rate may be required in these contrasts to reduce the probability of falsely rejecting at least one null hypothesis. However, Quinn and Keough (2002) advise that this should not be necessary provided that the number of independent contrasts does not exceed $df_{\text{groups}}(p-1)$; that is eleven in this case, which was the number employed.

Plantation characteristics useful for birds

Planting area (patch size) was found to be an important variable influencing bird species richness, a result also found in our earlier pilot study in the region (Kavanagh *et al.* 2001), but otherwise apparently novel. Kinross (2004) reported that wide (>19 m) planted windbreaks were closer in bird species composition to remnant woodland than narrow (≤ 15 m) planted windbreaks. The greater numbers of bird species recorded in larger (5-20 ha, and above) plantings compared to smaller (<5 ha) plantings was consistent whether the plantings were young (< 10 years) or old (> 10 years), and the main differences were due to the greater numbers of woodland-dependent species in the larger sized plantings. A similar result was obtained for bird species richness in larger compared to smaller patches of remnant vegetation, a finding that has also been reported in several recent Australian studies (Loyn 1985, 1987, Bennett and Ford 1997, Major *et al.* 2001, Seddon *et al.* 2003).

Plantation age was also found to be an important variable influencing bird species richness. On average, older plantings had more bird species and individuals than younger (<10 years) plantings. Interestingly, young plantings greater than 5 ha in size which, in this study, also tended to have greater numbers of shrub species and greater stem densities than older plantings, also had high numbers of birds. No other Australian studies appear to have specifically contrasted plantation age in agricultural landscapes as a variable influencing habitat occupancy by birds.

Habitat connectivity, or distance from remnant forest and woodland, is also likely to be an important variable influencing bird species richness. While the data have not been analysed explicitly in relation to distance from remnants, it is clear from the CCA biplot (Fig. 4.4) that the amount of remnant vegetation occurring within a 500 m radius of planted areas had a major role in structuring the

composition of bird species assemblages. In Chapter 8, the numbers of woodland-dependent birds in planted areas was correlated with the total amount of vegetation occurring within a 5 km radius “landscape” around each planted area.

Log cover and shrub cover were inversely correlated with grazing history. This gradient had a powerful influence on the composition of the bird species assemblage occurring at all sites. In particular, grazing history was inversely related to the numbers of woodland-dependent species recorded. The importance of coarse woody debris (log cover) and shrub cover in explaining the richness of bird species in rural landscapes has also been shown by Mac Nally *et al.* (2001) and Seddon *et al.* (2003). Numbers of old hollow-bearing trees, an essential habitat attribute for many woodland-dependent species was, paradoxically, associated with sites having a history of grazing (e.g. Bennett *et al.* 1994). The relative abundance of several woodland-dependent species at heavily-grazed sites may be explained in part by the presence of tree hollows but also by an association with the canopy rather than understorey vegetation. Eucalypt plantings established on previously grazed sites that include some retained hollow-bearing paddock trees have the potential to provide suitable habitat and essential cover for a much wider range of bird species.

The plant species composition of most of the eucalypt plantings sampled in this study was richer than might normally be expected, especially compared to plantations established primarily for wood production, but also compared to most of the smaller remnants. Only few wood production plantations were sampled in this study but, like most of the small, grazed remnants, these contained the least number of bird species. While not analysed explicitly, the existence of multiple species of native trees and shrubs in most eucalypt plantings sampled is thought to be a major contributor to bird species richness.

The location of plantings in relation to riparian zones is also likely to have a significant influence on bird species composition. Riparian zones, including permanent streams and defined gullies, are known to provide much better habitat for birds than the surrounding drier forests on ridges and slopes (Fisher and Goldney 1997, MacNally *et al.* 2000).

Response of declining species

Bennett and Ford (1997) and Reid (2000) classified bird species of north-western Victoria and south western NSW, respectively, into woodland dependent and non-woodland dependent species (see Table 4.1), and then used RAOU Bird Atlas data collected nearly 20 years earlier to assess changes in species conservation status. Reid (2000) categorised 21 of 198 landbird species as “decliners”, 17 of which were recorded in this study (Table 4.5). We, and Bennett and Ford (1997), classified three of these species, the Whistling Kite, Restless Flycatcher and White-browed Woodswallow as non woodland-dependent because all three are capable of utilising a broad range of wooded vegetation types. The Whistling Kite was rarely recorded in our study, and never on the sampling plots, but both the Restless Flycatcher and the White-browed Woodswallow were recorded in remnants and in eucalypt plantings (Table 4.5). Indeed, eucalypt plantings appeared to provide useful habitat for at least 10 of the “declining” species recorded in this study, notably so for three species, the Speckled Warbler, Red-capped Robin and Rufous Whistler. This result was also reported recently by Kinross (2004). Remnant forest and woodland appears to be most limiting for two species, the Brown Treecreeper and Dusky Woodswallow.

Table 4.5 Mean counts for bird species categorised by Reid (2000) as “decliners” in the sheep-wheat belt of the south-west slopes region of New South Wales. Data are from Table 4.2 and based on the results of standardised fixed time, fixed area sampling counts (does not include incidental observations).

Species	Remnants (n=50 sites)	Old plantings (n=40 sites)	Young plantings (n=20 sites)	Paddocks (n=10 sites)
Painted Button-quail	0	0	0	0
Brown Treecreeper	2.0	0	0.1	0
Speckled Warbler	0.1	0.2	0.3	0
Jacky Winter	0.6	0.2	0	0.1
Red-capped Robin	0	0.2	0.1	0
Hooded Robin	0	0	0	0
Eastern Yellow Robin	0.5	0.1	0.1	0
Grey-crowned Babbler	0.1	0	0	0
White-browed Babbler	0.1	0	0	0
Varied Sittella	0.6	0.5	0	0
Crested Shrike-tit	0.7	0.6	0.2	0
Rufous Whistler	1.1	2.5	2.2	0
Dusky Woodswallow	1.4	0.1	0	0
Diamond Firetail	0	0	0.3	0
Whistling Kite	0	0	0	0
Restless Flycatcher	0.6	0.4	0.1	0
White-browed Woodswallow	5.7	7.4	2.7	1.5

5. Bats

5.1 Introduction

Bats are a particularly important component of the fauna in fragmented, rural landscapes (Lumsden *et al.* 1995, Law *et al.* 1999, Law *et al.* 2000). On the south-west slopes of NSW 10 taxa of bats have been recorded (Law *et al.* 1999). Studies of bats in these landscapes have revealed that some species commute from tree-roosts across cleared land to forage in small remnants, even isolated paddock trees (Lumsden *et al.* 1995, Law *et al.* 1999, Law *et al.* 2000, Lumsden *et al.* 2002). Because flight aids movement across the landscape (Lumsden *et al.* 2002) it potentially facilitates the encounter and use of patches of revegetation. The ability to fly also provides an interesting comparison to studies on birds, which being conspicuous, are probably the most typical subjects of studies in fragmented landscapes (Blake and Karr 1987, Loyn 1987, Barrett *et al.* 1994, Major *et al.* 2001, Seddon *et al.* 2003). No studies have been published on the use of revegetation by Australian bats (Ryan 2000).

In this chapter we report on activity levels of microchiropteran bats recorded by ultrasonic detectors across the 12 vegetation classes. We predicted that patch size would have little influence on bat activity due to the widespread distribution of many bat species in rural landscapes (Law *et al.* 1999).

5.2 Methods

Bat sampling

Bat activity was sampled in late spring (November/early December) 2002 using Anabat detectors (Titley Electronics). Each site was sampled remotely at a single location for one full night. While a single night of sampling per site is normally considered modest (Hayes 1997), we placed emphasis on sampling spatial variation by having a high level of site replication (120 sites in total). We also found that activity levels were often very high, sometimes with more than 1000 calls recorded in a night (see Results). We attempted to address the effects of nightly variation in activity by only sampling in warm conditions and avoiding rain and the full moon. We also completed all sampling within an intensive field period of one month using eight units to simultaneously sample a mixed set of vegetation classes each night.

Detectors were positioned away from the edge of a patch, except for linear strips, which were all edges. The microphone was angled up at 45° from the ground and faced into vegetation openings or gaps to reduce the influence of call attenuation from vegetation (Parsons 1996, Patriquin *et al.* 2003), but always avoided tracks that may serve as bat flyways (Law and Chidel 2002). Detectors sampled each site for a full night by recording files to a lap-top computer via a zero-crossing interface (ZCAIM), with each pass being stored as a single file. A pass follows the definition of Law *et al.* (1988), consisting of a minimum of three pulses, with pulses not separated from another pulse by more than five seconds. Bat activity in a site was expressed for each species as the number of passes per night.

Automated identification of calls to species

All files were processed by software (Anascheme; Matt Gibson, Ballarat University, unpubl.) designed to automate the process of call identification. Anascheme reads Anabat files and models individual pulses using regression analysis (Gibson and Lumsden 2003). The regression model allows the extraction of a range of parameters that can be used to develop an identification key. The identification key used in this study was based on one developed for the nearby northern plains region of Victoria (Gibson and Lumsden 2003, L. Lumsden unpubl. data).

Small modifications to the key were made to address slight differences in locally collected reference calls, which possibly result from geographic variation in calls (Law *et al.* 2002). Two species that probably occur in the area were not added to the key because they were considered to be uncommon and to have a very restricted local distribution. One was the cave-roosting *Miniopterus*

schreibersii, which occurs just to the east of the study area (Law *et al.* 1999). Manual identification of a sample of passes from sites near rocky topography and possible caves (e.g. Benambra and Tabletop) revealed this species constituted 0.7 % of passes (n=451). The other is the low frequency form of *Vespadelus regulus*. This species displays marked geographic variation in call frequency (Law *et al.* 2002). The study area falls within an overlap zone for two frequency groups, but the low frequency form has only been confirmed from large stands of forest in hilly country at Woomargama (Law *et al.* 1998) and at Tabletop Reserve (C. Grabham unpubl. data). Thus identifications from these two areas in our study may have incorporated some passes of the low frequency form of *V. regulus* into *Vespadelus darlingtoni*, which overlaps in frequency. Two species of *Nyctophilus* occur in the study area (*N. geoffroyi* and *N. gouldi*; Law *et al.* 1998), but it is not possible to distinguish them by call, even by the manual method, so calls from these two species were lumped as *Nyctophilus* spp.

The following options were set within Anascheme. Identifications were made only when a minimum of 50 % of pulses within a pass was identified to the same species. As part of this process, we excluded pulses classed as unknown from contributing to the total number of passes, because they are usually poor quality that are typically ignored when a pass is identified manually. Only passes with a minimum of three pulses were identified.

We ran a series of tests using Anascheme to ensure that this new method of identifying bat calls was reliable. The accuracy of identifications was tested using a reference call library, which consisted of 109 local calls, supplemented with 27 calls collected from other regions for species whose calls appear unaffected by geographic variation (*Tadarida australis*, *Nyctophilus* spp., *Chalinolobus morio*). The key is conservative in nature to avoid false identifications (L. Lumsden pers. comm.), and we found only 0.7 % of the reference calls were incorrectly identified, while 63 % were correctly identified (Law and Chidel, unpubl. data). Further testing on a sample (1039 calls) of remotely recorded calls revealed that manual identification based on both quantitative (e.g. characteristic frequency) and qualitative (e.g. shape) features could identify 74 % to species level compared to automated identification by Anascheme, which could identify 60 %.

Statistical analysis

Given that bat activity was recorded for just one night per site, we tested for the influence of nightly fluctuations in temperature and humidity on mean nightly total activity. Hourly weather records were collated for the study period from Albury airport (Bureau of Meteorology). Mean temperature and humidity was calculated for each night of sampling from one hour before sunset to sunrise. Means were then compared to the corresponding mean nightly total activity across sites using scatter-plots and Pearson correlation coefficients (n=17 nights).

Analysis of variance was used to test for differences across vegetation classes for total bat activity ($\log_{10}(x+1)$ transformed) and species richness. Five pre-planned contrasts were specified for what were considered to be the main effects. These were old plantings vs paddocks, young plantings vs paddocks, old plantings vs remnants, young plantings vs remnants and old plantings vs young plantings. Different patch-size classes within these contrasts were averaged, but only size categories that were present in both groups were included. Post-analysis, plots of all means were inspected and major differences between means were tested with specific post-hoc comparisons. The α -level (0.05) for post-hoc comparisons was adjusted by the number of comparisons in order to keep experiment-wise error rate at α . Activity levels of individual species were usually skewed with many zeroes. Thus the non-parametric Kruskal-Wallis test was used to analyse individual species data. Paddocks, either with or without a tree, were pooled for all analyses.

Differences in species composition across vegetation classes was investigated using non-metric multi-dimensional scaling (NMDS) by ordination of sites, based on the activity levels of species recorded in those sites. Activity was $\log_{10}(x+1)$ transformed to increase the contribution from rare species (Clarke 1993). The similarity matrix required for the NMDS was calculated using the Euclidian distance, with the saved matrix subsequently used to perform the NMDS in the program Statistica.

To predict which characteristics of revegetation were beneficial to bats we used generalised linear models (GLM) for both species richness and total activity against revegetation attributes. To minimize inter-correlations between landscape and habitat variables, we reduced the final list of variables for analysis after checking for collinearity with a correlation matrix. The GLM used a best sub-sets procedure where we selected the best simple model, as determined by the Akaike Information Criterion (AIC), that included at most four predictors. The best simple model was then evaluated by re-running a GLM with only the selected variables. The significance of each explanatory variable was tested with the Wald statistic. A normal error distribution was used for bat activity after $\log_{10}$ transformation and for species richness, both with a log-link function. Model fit was indicated by plots of predicted versus observed values, as well as normal probability plots and scaled deviance divided by the degrees of freedom, a goodness of fit statistic.

5.3 Results

Across our 120 study sites we recorded for 1080 hours, which resulted in 40,423 computer-recorded files. Anascheme recognized 17,913 of these as bat passes with more than two pulses. The remainder had fewer than three pulses, consisted of noise not produced by bats, such as stridulating insects, or were empty files generated by a fault in an early version of Anabat (CF read) software. Anascheme identified 56 % of recognized passes to species level, with 11 species recorded in total (Table 5.1). A further 7 % of pulses were identified to species groupings.

Table 5.1 Species recorded at 120 study sites, the number of passes and the percentage of sites at which they were recorded.

Common Name	Scientific Name	Number of passes	% of sites
Little Forest Bat	<i>Vespadelus vulturnus</i>	4012	89
Southern Freetail-bat (long penis form)	<i>Mormopterus planiceps lp</i>	1444	85
Long-eared Bat	<i>Nyctophilus</i> spp.	993	86
Large Forest Bat	<i>Vespadelus darlingtoni</i>	951	64
White-striped Freetail-bat	<i>Tadarida australis</i>	905	68
Gould's Wattled Bat	<i>Chalinolbus gouldii</i>	710	83
Southern Freetail-bat (Eastern form)	<i>Mormopterus</i> sp2	371	49
Chocolate Wattled Bat	<i>Chalinolobus morio</i>	336	48
Southern Forest Bat (High frequency form)	<i>Vespadelus regulus</i>	236	45
Inland Broad-nosed Bat	<i>Scotorepens balstoni</i>	43	19
Yellow-bellied Sheath-tail-bat	<i>Saccolaimus flaviventris</i>	5	2

Total activity

Mean nightly bat activity was poorly correlated with both mean nightly temperature ($r=0.25$) and humidity ($r=-0.35$). The weak influence of these variables on bat activity was probably due to the consistent warm conditions throughout the study. Mean nightly temperature varied between 14.7 and 27.7 °C, while nightly humidity varied between 27.0 and 76.7 %.

Total activity per night was high across all vegetation classes varying from an average of 50 passes in young, small plantings to 302 passes in small remnants (Fig. 5.1). The greatest activity occurred at

a large remnant (No Man's Land), where 1253 calls were recorded in a night. One paddock tree site was considered to be an outlier, because 1035 passes were recorded, presumably because the tree supported a day-roost, with socialising at night resulted in high activity. When this site was excluded from analyses, total activity varied significantly across the vegetation classes ($F_{11, 107}=2.6$, $P<0.01$; Fig 5.1). Pre-planned contrasts found that bat activity was greater in remnant vegetation than in both old ($t=-3.3$, $P=0.001$) and young plantings ($t=-3.2$, $P=0.002$). Total activity in old and young plantings was similar to paddocks (old: $t=0.6$, $P=0.5$; young: $t=0.2$, $P=0.8$) and there was no difference between the two age-classes of plantings ($t=-0.3$, $P=0.7$). Among old plantings, the large size class supported the greatest activity (mean = 137 passes per night), and although this was twice that of paddocks, post-hoc comparisons (α adjusted to 0.02) revealed that the difference was not significant ($t=1.9$, $P=0.06$). Within each age class, there was also support for a trend of greater bat activity in larger sized plantings, but again there was considerable variation between sites resulting in no significant difference (old: $t=1.9$, $P=0.07$; young: $t=2.0$, $P=0.05$). There was significantly more activity in remnant vegetation than in paddocks ($t=2.6$, $P=0.01$), although it is noteworthy that bat activity in small remnants (< 5 ha) was similar to very large remnants (> 1000 ha).

Species richness

Species richness was high at most sites, varying from an average of 5 species in paddocks to 8 in large remnants (Fig 5.2). Pre-planned contrasts found that old plantings had fewer species than remnants of an equivalent size ($t=-2.8$, $P=0.007$), but there was no difference between young plantings and remnants ($t=-1.5$, $P=0.15$). Neither old nor young plantings supported significantly more species than paddocks (old: $t=1.4$, $P=0.2$; young: $t=1.3$, $P=0.2$). Amongst plantings, the largest size in each age class (old-large and young-medium) supported the most species. But species richness in old, large plantings was not different from paddocks as determined by post-hoc comparisons ($t=1.9$, $P=0.06$ - α adjusted to 0.02). Conversely, old large plantings supported similar numbers of species as large remnants ($t=-1.4$, $P=0.14$). Remnants supported more species than paddocks ($t=2.9$, $P<0.01$).

Species composition

A two-dimensional plot of sites, ordinated by bat species composition, showed no distinct groupings based on their vegetation class (stress=0.19; Fig 5.3). This indicates that there was extensive overlap in species composition across vegetation classes. Paddock sites formed the tightest cluster in multi-dimensional space, indicating that species composition found in this vegetation class was less variable than in other classes.

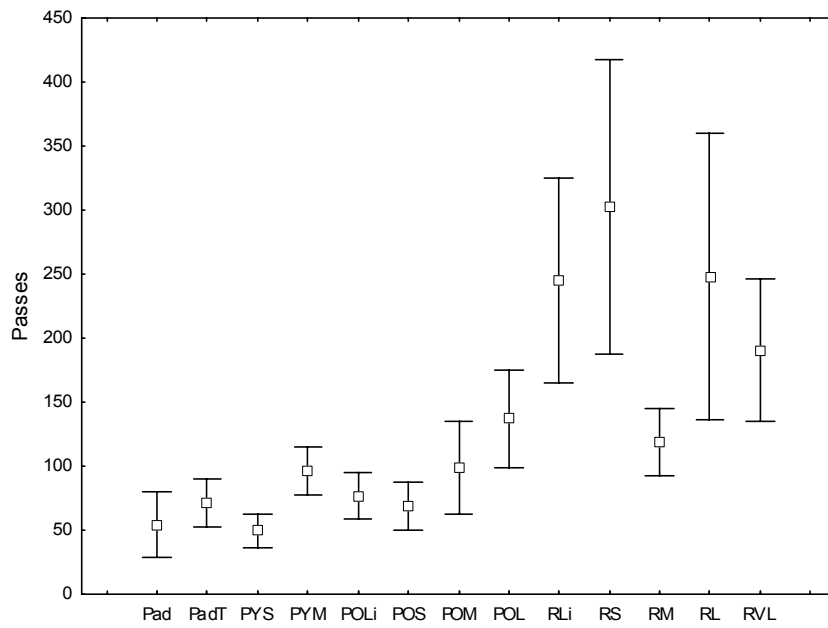


Fig 5.1 Total bat activity (passes per night) across 12 categories of vegetation (n=10 sites per category). Note that paddocks were split in this plot into those with trees (n=4; one outlier omitted) and those without (n=5). Means $\pm$ standard errors are shown.

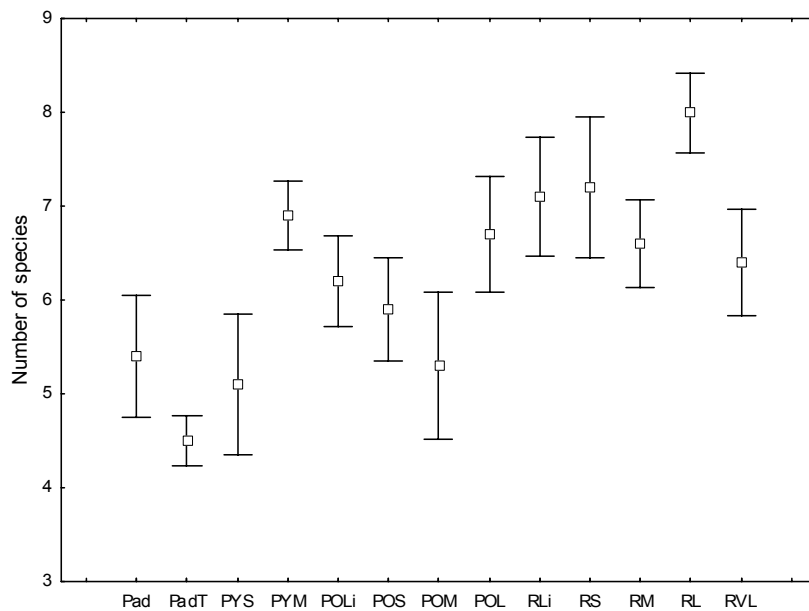


Fig 5.2 Species richness across 12 categories of vegetation (n=10 sites per category, except paddocks as above). Means $\pm$ standard errors are shown.

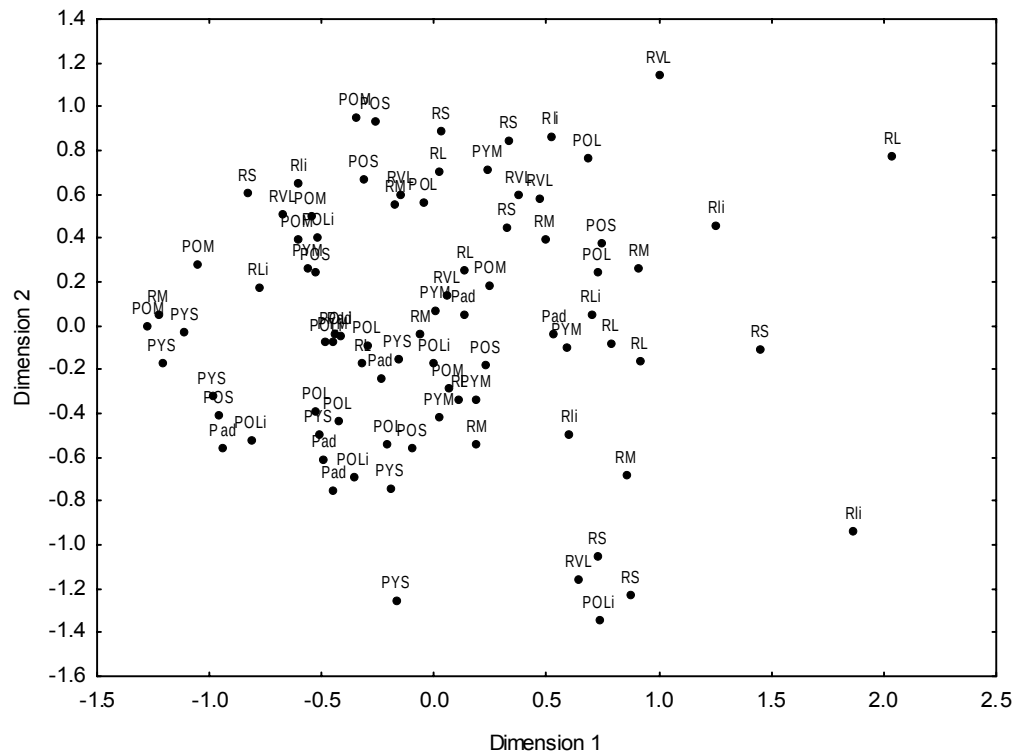


Fig 5.3 Two dimensional ordination of sites sampled in the study area by their bat species composition using non-metric multi-dimensional scaling (stress=0.19). Data were log-transformed. See Table 2.1 for site labels.

Relationships with revegetation characteristics

The suite of habitat and landscape variables we measured was reduced by removing those that were inter-correlated, which resulted in a sub-set of 11 variables for model-building (Table 5.2). The best models for both species richness and total activity, as determined by the lowest AIC score, included just two predictors. Species richness was negatively associated with shrub cover (Wald Stat. = 9.99; $P < 0.01$) and planting age (Wald Stat.=2.60, $P=0.11$). Total activity was also negatively associated with shrub cover (Wald Stat.=11.56; $P < 0.001$) and positively to the amount of wooded vegetation within a 0.5 km buffer of the sites (Wald Stat.=3.78; $P=0.05$). However, neither model fitted the data well, as evidenced by the spread of points in plots of observed versus predicted values. This suggests that the models indicate general trends, but not with high predictive accuracy.

Activity of individual species

Over all sites, very high activity levels (> 1000 passes) were recorded for *V. vulturnus* and *Mormopterus planiceps* sp. Moderate activity (500-1000 passes) was recorded for *Nyctophilus* spp., *V. darlingtoni*, *T. australis* and *C. gouldii*. Low activity (0-500 passes) was recorded for *Mormopterus planiceps* sp., *C. morio*, *V. regulus* HF, *S. balstoni* and *S. flaviventris*. Bat species differ in call intensities and so comparisons of activity levels between species must be interpreted cautiously. The differences highlighted above only very broadly indicate difference in abundance.

Table 5.2 Characteristics of revegetation sites used for modelling bat species richness and total activity, after checking for collinearity.

Variable	Description
Stem density (stems ha ⁻¹)	Number of stems (including dead) > 5 m high per ha
Stems > 60 cm dbh (stems ha ⁻¹)	Number of stems (including dead) > 60 cm dbh per ha
Overstorey height (m)	Estimated height of overstorey vegetation
Ground cover (%)	Estimated percent cover of ground vegetation
Shrub cover (%)	Estimated percent cover of shrub vegetation (< 5 m)
Tree diversity	Shannon-Weaver Diversity Index of eucalypt and non-eucalypt species
Tree age (years)	Oldest age of trees planted on site
Topography	Scaled from 1 (peak) to 9 (flat)
Wooded veg – 0.5 km	Total area of wooded vegetation within 0.5 km buffer radius
Wooded veg – 5 km	Total area of wooded vegetation within 5 km buffer radius
Log Area (ha)	Area of the patch transformed by log ₁₀

Most bat species were patchily, but widely, distributed across the vegetation classes (Fig 5.4, 5.5). *Chalinolobus morio* was slightly more active in remnant vegetation than plantations, and it was absent from all paddocks, except at the paddock tree (29 passes) which was excluded from analysis because of its apparent use as a roost ($H=21.1$, $P<0.05$). *Vespadelus vulturnus* occurred in all classes, but was least active in treeless paddocks, young plantings, old small and strip plantings ($H=35.2$, $P<0.001$). Its greatest activity was at a large remnant (“No Man’s Land”), where over 500 passes were recorded. *Vespadelus regulus* HF was not recorded in young, small plantings and was uncommon in paddocks ($H=31.8$, $P<0.01$). Its activity was greatest in large remnants, but this was primarily due to high activity (56 passes) in one large remnant of river red gum *Eucalyptus camaldulensis* forest, which was within 1.4 km of a large roost in a cabin ceiling (B. Law and M. Chidel, unpubl. data). None of the remaining eight species in the study showed significant differences across vegetation classes. *S. flaviventris* was not analysed because only five passes were recorded.

Canonical Correspondence Analysis was unable to reveal a close relationship between the activity patterns of individual species and either site or landscape variables (Fig 5.6). All species were plotted close to the centre of the plot, indicating that the recorded environmental variables had little influence over species activity levels. Environmental variables omitted from the biplot because of their relative unimportance (i.e. plotted as short vectors) in describing the environments sampled included patch area, amount of woody vegetation within 0.5 km of the site, topography, tree species diversity and riparian sites.

Further exploration of species’ distribution patterns at a univariate level revealed that four species, *V. darlingtoni*, *V. regulus* HF, *C. morio* and *M. planiceps* sp, had a significant association with river red gum forests, which occurred at 17 sites (Fig 5.7). The remaining six species were equally distributed between sites with river red gum and those where it was absent (Fig 5.7).

Fig 5.4 Box plots of bat species' activity where a significant difference between vegetation categories was found. Medians, 25-75 % quartiles and non-outlier extremes are shown.

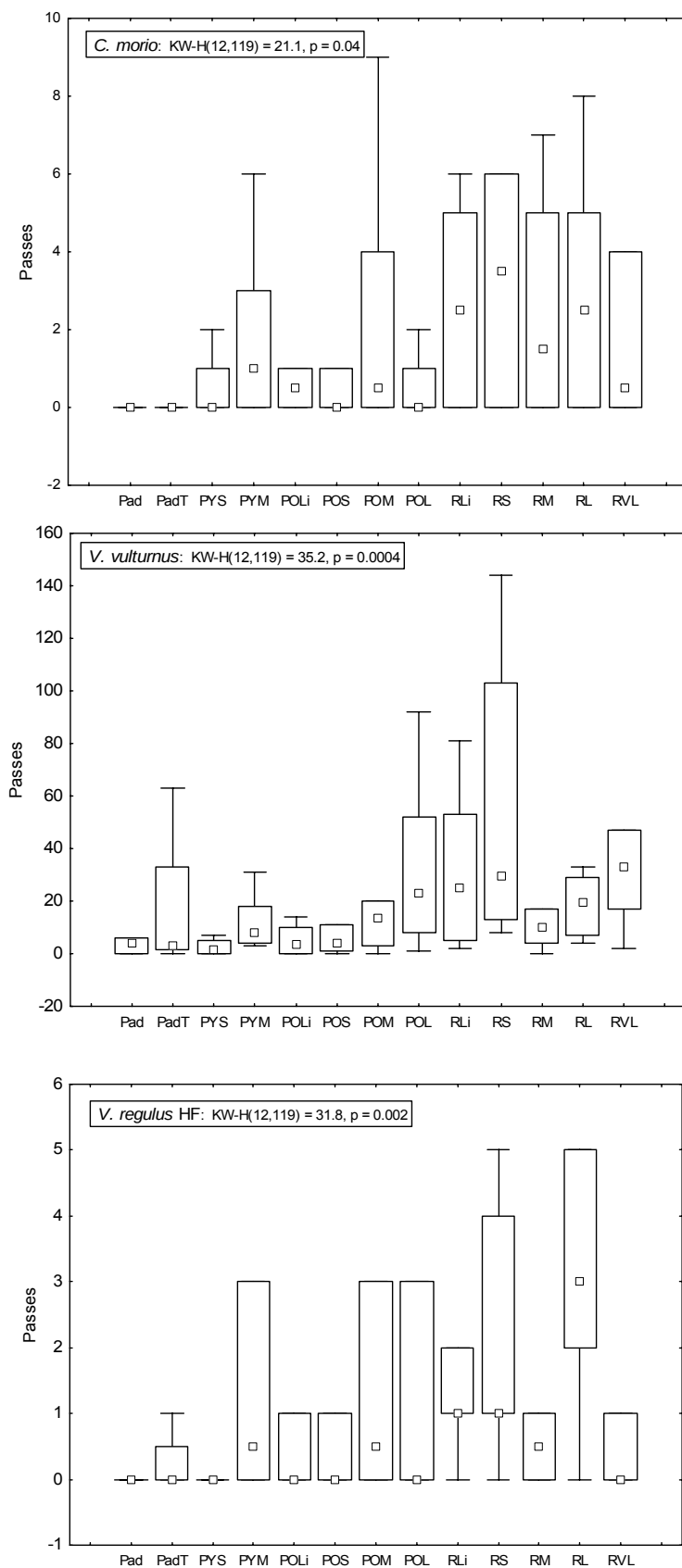
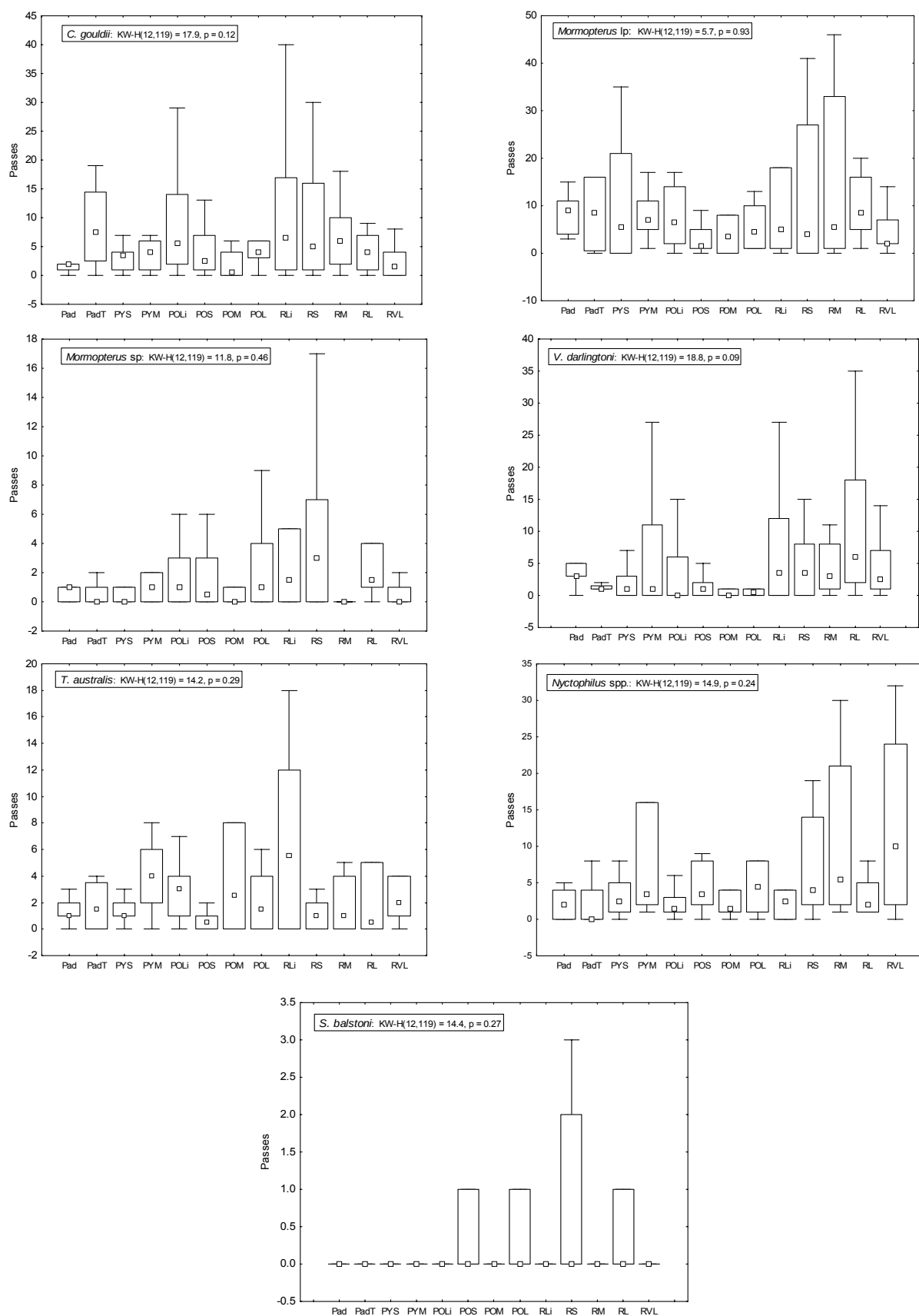


Fig 5.5 Box plots of bat species' activity where no significant difference between vegetation categories was found. Medians, 25-75 % quartiles and non-outlier extremes are shown.



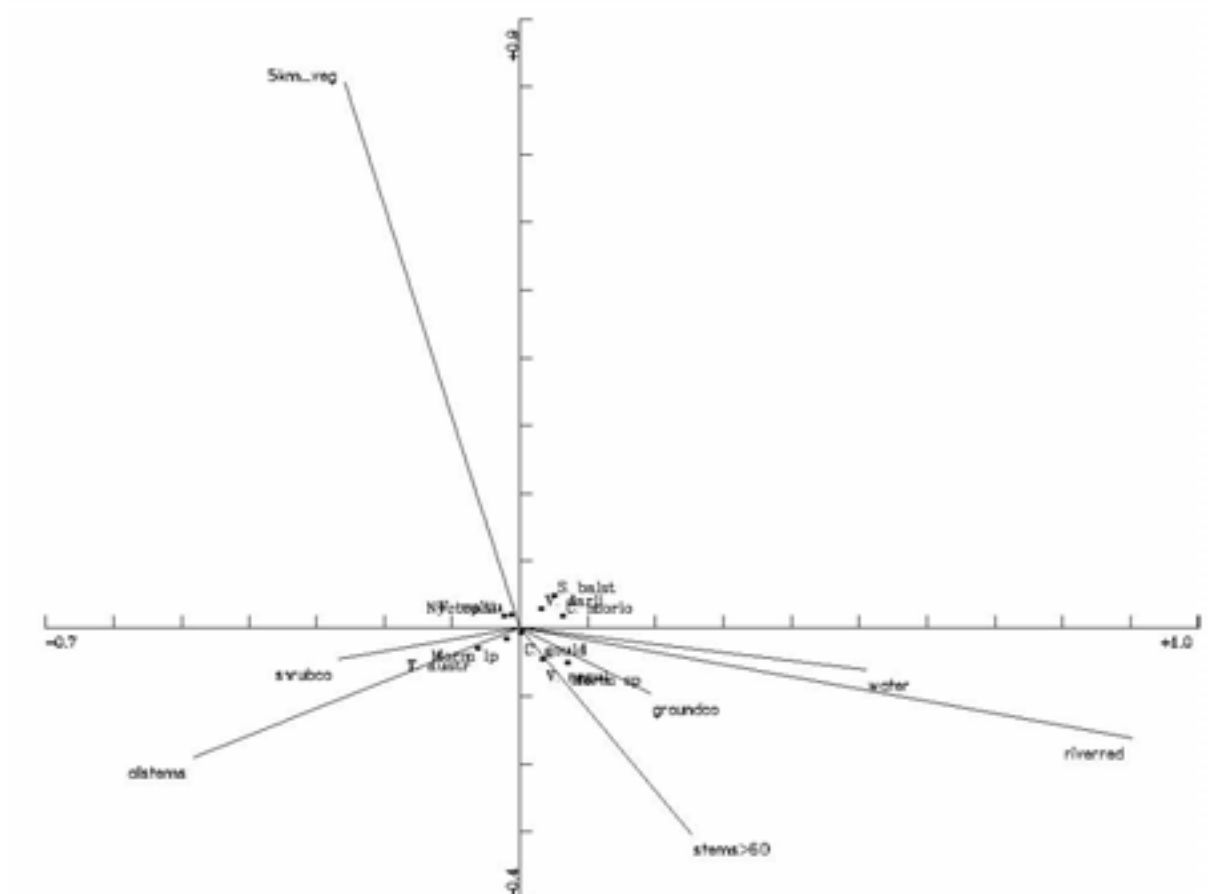


Fig 5.6 Biplot of environmental variables and individual bat species activity (log transformed).

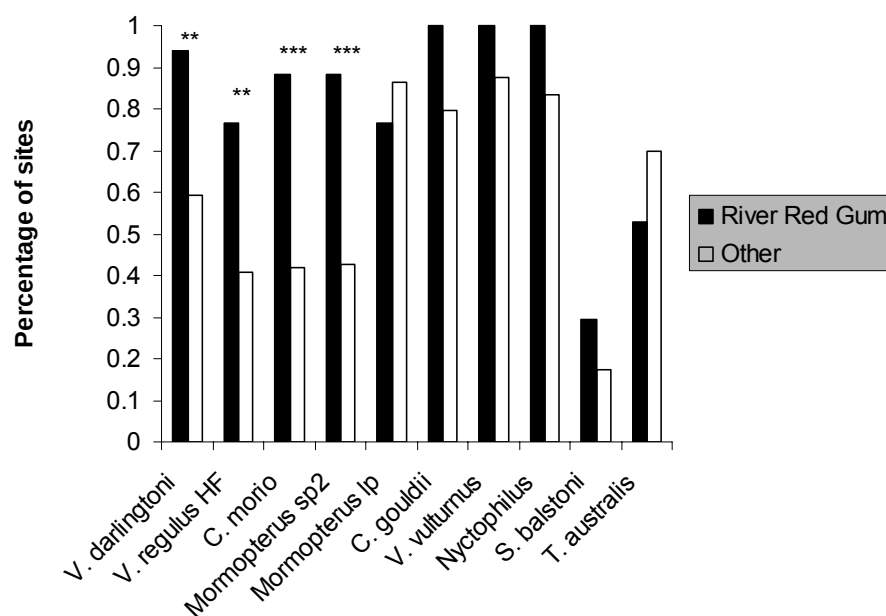


Fig 5.7 Proportional occurrence of bat species at sites with river red gum forest *Eucalyptus camaldulensis* (n=17) and all other sites (n=103). Significant associations with *E. camaldulensis* were tested with χ^2 tests (** $P < 0.01$ and *** $p < 0.001$).

Additional observations that are of interest include:

- *C. gouldii* – activity was broadly similar across the landscape, including regular use of paddock trees and linear strips and somewhat less use of very large remnants;
- *M. planiceps* lp - three sites with highest activity included a small remnant and two linear remnants;
- *M. planiceps* sp - two sites with highest activity were in riparian river red gum *Eucalyptus camaldulensis* forest (one large remnant on the Murray River and one linear remnant along Billabong Creek);
- *Nyctophilus* spp. - one young medium-sized planting (Tasmanian blue gum) had very high activity (171 passes), although a similar planting nearby did not;
- *S. balstoni* – very low activity, with the greatest being eight passes at a small remnant in the semi-urban outskirts of the town Wodonga;
- *T. australis* – unusually high activity (389 passes) at a very large remnant (Tabletop Reserve) and a large remnant (No Man's land), possibly indicating roosts nearby;
- *V. darlingtoni* – unusually high activity (348 passes) at one small remnant (Wyabalena), possibly in the vicinity of a roost.

5.4 Discussion

We found that bats were broadly distributed across the entire landscape of the study area and that all species used the various kinds of revegetation assessed. Although revegetation was used regularly by bats (on average 87 passes per night), the level of use was no greater than the use of paddocks and was just under a third of that in remnants. *C. morio* was the only exception as it was not found in paddocks, but plantings were used, although to a lesser extent than patches of remnant vegetation. We also found that species richness was greatest in remnants and lower in both plantings and paddocks. Species composition at a site was not related to its classification based on vegetation class. These results illustrate the great importance of the paddock matrix to foraging bats (Lumsden *et al.* 1995, Law *et al.* 1999, Law *et al.* 2000). These results are consistent with previous local studies that have found the same 11 bat species occurring in both rural and forest areas, with two species (*C. gouldii*, *M. planiceps* lp) regularly using the fringes of the Albury township to forage on insect concentrations near street lights (Kirsten and Klomp 1998, Adams 2000).

Despite plantings and paddocks supporting similar bat activity, we believe that there is a good case to value old, large plantings for bats. This is because recordings of bat activity in paddocks would be inflated due to a lack of sound attenuation by vegetation (Parsons 1996). Although we tried to minimise this effect by facing our detectors into vegetation gaps, these were often rather small in many plantings. Also, some plantings were dense throughout. A negative bias in activity would be of most relevance when assessing the value of old, large plantings, as they had twice the activity of paddocks, although the difference was marginally non-significant. Activity overlapped much more clearly between paddocks and other plantings of different sizes and ages, indicating that attenuation of sound has less influence on our conclusions for these comparisons. On this basis we can conclude that large plantings need to grow for 10-25 years to achieve greater bat activity (double) than that provided by paddocks with scattered trees. The other way of looking at this result is that old plantings (especially large ones) achieve about half the bat activity levels as that of remnant vegetation, which is not a bad outcome for only 10-25 years of growth. The noticeable difference in activity within older plantings could be due to the changing structure as the plantings grow. Vegetation patchiness was particularly evident in older plantings, which apparently resulted from mortality during past episodes of drought and drought-related grazing. It is interesting to note that stem density, another approximate measure of clutter, was lower in older than in younger plantings.

The similarity of bat activity between plantings and paddocks also needs to consider the nature of the paddocks we sampled. Half of our paddocks were treeless with the nearest tree being greater than 100 m distant and the other half were adjacent to an isolated paddock tree. Many bat species fly and forage around paddock trees (Law *et al.* 2000). On average, 12 % of the landscape within a 5 km buffer of our sites comprised forest woodland or plantings. This excludes scattered trees in paddocks and they are typical of the Albury region. It could be expected that bat activity would be lower in paddocks more distant from vegetation and thus tree-planting in such areas could prove more beneficial to bats.

High activity in remnants points to the importance of retaining as much of the remaining vegetation as possible (Law *et al.* 1999, Lumsden *et al.* 2002). Travelling Stock Routes (TSRs) constituted 60 % of our large remnant size-class emphasising their importance in preserving this vegetation class in rural Australia (Hibberd and Southberg 1991). In addition to supporting high activity levels, remnant vegetation also supports an abundance of large, hollow-bearing trees (Bennett *et al.* 1994), which are the preferred roosts of most bat species (Lumsden *et al.* 2002). The high use of small remnants by bats contrasts with studies on other fauna groups, such as reptiles and birds (Caughley and Gall 1995, Chapter 4 and 7). Indeed, the bird component of this study found that larger patches support more species and individuals (Chapter 4). Major *et al.* (2001) recommended retaining large remnants and restoring large patches because the aggressive noisy miner exerts less influence on declining small birds in such areas. Our study indicates that a focus on enhancing large remnants would not be to the detriment of bats, but that small patches must also be retained. Use of small patches is also relevant to revegetation programs, because small patches are the norm for plantings on farms (Bennett *et al.* 2000). Linear strips are also commonly planted on farms, for example as windbreaks, and these were also regularly used by bats.

Proximity to remnant vegetation had no effect on either total activity or species richness, probably due to the dispersal ability of flying bats and the capacity to forage around isolated paddock trees (Law *et al.* 2000). Planting age is another site characteristic that had no strong detectable effect on bat activity or species richness, although there appeared to be some interaction between age and patch size, so that large old plantings benefited bats. High tree species diversity is typically recommended for revegetation programs (Bennett *et al.* 2000), but it was not related to total activity or species richness of bats. Interestingly, tree diversity tended to be lower in remnant vegetation than in planted sites. A good example of bats using low diversity plantings was a 6 ha monoculture planting of Tasmanian blue gum *Eucalyptus globulus*, a non-indigenous species to the Albury area. A total of 212 calls was recorded at this site, with 171 of them identified as *Nyctophilus* spp., which was the highest activity recorded for this taxon in the study. It is possible that larger patch sizes, proximity to remnant vegetation, older planting age and a more diverse plantings support a more diverse invertebrate prey that could be available over a greater range of seasons or conditions than our snap-shot survey could reveal. We suggest that this is less likely for patch area and proximity to remnant vegetation as high activity in isolated, small patches has been observed in other studies of remnant vegetation in south-eastern Australia (Lumsden *et al.* 1995, Law *et al.* 1999). Although it is worth noting that when continuous forests of > 30,000 ha are included in such comparisons, some bat species appear sensitive to patch area (Law *et al.* 1999).

A negative relationship with shrub cover (including young eucalypts if < 5 m high) was the most consistent predictor of total activity and species richness. This pattern is in accordance with many studies that have demonstrated structural clutter is a primary driver of habitat use among bats as it has a negative influence on the foraging ability of many species (Brigham *et al.* 1997a, Humes *et al.* 1999, Law and Chidel 2002). This largely results from bat wing shape and echolocation call type, which limits where certain species can successfully forage (Aldridge and Rautenbach 1987, Findley 1993). *Nyctophilus* spp is the main clutter-tolerant taxon in our study area, because it is slow flying and highly maneuverable (Brigham *et al.* 1997b). Even though *Nyctophilus* spp. was widely distributed, it did have high activity in plantings with high stem density (e.g. 171 passes in a dense Tasmanian blue gum planting – 1000 stems ha⁻¹). Species that are slightly less maneuverable, but able to forage close to vegetation include *V. vulturnus*, *V. regulus* HF and *C. morio*. Based on ecomorphology we would predict these species to dominate the activity within dense vegetation, even though they also use more

open vegetation. The remaining clutter-sensitive species would be physically restricted to using more open vegetation. The fact that we only found an overall weak relationship with clutter/cover could be due to two factors. First, we specifically sampled gaps in the vegetation to maximise the reception distance of detector microphones. Second, our habitat plots covered a relatively small proportion of total available habitat in patches larger than the smallest size class, particularly in relation to foraging movements of bats.

The negative effect of clutter in the form of either high stem density or high shrub cover contrasts with findings on the importance of cover to small birds, especially in the woodland environments of south-eastern Australia (Watson *et al.* 2001, Seddon *et al.* 2003). The differential response of bats and birds emphasises the need for caution in extrapolating management recommendations from studies on one focal fauna group. Yet literature on the requirements of fauna in remnant vegetation is overwhelmingly biased towards diurnal birds. There is also a similar trend for a bias towards birds in recent and current studies on the importance of revegetation to fauna (Ryan 2000). Clearly there is a great need for further broad-based studies that consider the diverse requirements of a diverse fauna.

The distribution of roosts is also likely to be an important influence on activity levels. Tree hollows were more common in remnant vegetation than in plantations, which is likely to introduce some bias in activity levels to this vegetation category. Indeed, we excluded one site from analyses (a paddock tree) because very high activity levels (> 1000 passes in a night) suggested the presence of a roost. However, bats commonly commute from roosts to foraging areas (Lumsden *et al.* 2002), so that these two habitat components do not have to be contiguous. Roosts also appear to be reasonably widespread in the study area. Roosts have been found in both the rural and urban environment for *C. gouldii* and *V. vulturnus* in tree hollows, and for *M. planiceps* sp, *C. morio* and *V. regulus* HF in artificial roosts such as ceilings (M. Adams pers. comm., B. Law and M. Chidel. unpubl. data).

6. Nocturnal birds and arboreal marsupials

6.1 Introduction

The limited information that is available about the habitat value of revegetated sites in Australia has been derived primarily from studies on diurnal birds (Ryan 2000). No studies have been published on the use by nocturnal birds of eucalypt plantings in agricultural areas. Also, the mobility of birds may over-emphasise the habitat value of revegetated sites for other less mobile fauna groups, including arboreal marsupials. The little that is known about the use by arboreal marsupials of eucalypt plantings in agricultural areas includes evidence for several species of their acceptance of replanted areas once artificial nest hollows, the major limiting factor for most species, had been provided (Suckling and Macfarlane 1983, Smith and Agnew 1992). In both studies, the age of tree plantings was less than 15 years. The effects of fragmentation of remnant forest and woodland on nocturnal birds and arboreal marsupials are somewhat better known (Suckling 1982, 1984, Loyn 1985, Bennett 1987, Recher *et al.* 1987, Kavanagh and Stanton 1998, 2002, Lindenmayer *et al.* 2000, van der Ree 2002). These studies have shown that some gliders, especially the sugar glider *Petaurus breviceps* and squirrel glider *P. norfolcensis*, and some nocturnal forest birds, including the Australian Owlet-nightjar *Aegotheles cristatus*, barn owl *Tyto alba* and southern boobook *Ninox novaeseelandiae*, are capable of maintaining high population densities in linear roadside and riparian strips and in small (< 10 ha) patches of remnant vegetation. Two other common woodland species, the Common Brushtail Possum *Trichosurus vulpecula* and Common Ringtail Possum *Pseudocheirus peregrinus*, are usually recorded in larger fragments ($\geq$ 10-50 ha) (Kavanagh and Stanton 2002).

In this Chapter we report the results of standardised listening, call-playback and spotlighting surveys for all species of nocturnal birds and arboreal marsupials to determine the relative value of eucalypt plantings, remnant forest and woodlands, and cleared paddocks as habitat for these species.

6.2 Methods

The basis for sampling was a 200 m transect located at each of the 120 sites. After an initial 10 minute period of listening for any calling animals, one observer walked slowly along the transect line for approximately 20 minutes, listening and spotlighting with a 50 watt spotlight for any animals present. At the end of the transect, approximately 15 minutes was spent alternately broadcasting calls through a megaphone (Toa ER-66, 10W) and listening for any responses. Pre-recorded calls broadcast included the Barking Owl *Ninox connivens*, Masked Owl *Tyto novaehollandiae*, Squirrel Glider and, in larger remnants only, the Powerful Owl *Ninox strenua*. Often the call-playback session had to be attenuated, either in duration or volume, when houses or domestic stock were nearby (< 500 m), and in many cases was not attempted at all. Hence, call-playback data are provided for indicative purposes only. Perpendicular distances from the transect line were recorded for all observations. One visit was made to each site and all surveys were completed during mid-November to late December 2002.

6.3 Results

A total of 122 records of arboreal marsupials were made within 2 ha spotlight sampling plots at each site (i.e. within a strip 50 m wide on each side of the transect line) (Table 6.1). The Common Brushtail Possum and the Common Ringtail Possum were the species most frequently encountered, each occurring on about 25% of the sites. Both species were recorded more frequently in remnants, including narrow remnant strips, than in eucalypt plantings or in paddocks (Figs. 6.1 and 6.2). However, older (>10 years) eucalypt plantings, particularly in the 5-20 ha size-class, made a useful contribution to habitat for both species of possums. The Squirrel Glider was recorded at a total of 9 sites, including in a number of older plantings and in a similar number of remnants, including narrow roadside strips. This glider was recorded at only four sites within the 2 ha sampling plots (Table 6.1). The Sugar Glider was recorded only in one very large remnant (Woomargama National Park) and not on any of the 2 ha sampling plots. The Tuan *Phascogale tapoatafa* was recorded in one

older eucalypt planting (near Baranduda) that was adjacent to a narrow roadside remnant. During our pilot study (Kavanagh *et al.* 2001), one Koala *Phascolarctos cinereus* was heard calling during surveys in a very large remnant (Barambogie State Forest, now Chiltern-Mt Pilot National Park). On average, arboreal marsupials were recorded more frequently in remnants and in some of the larger and older eucalypt plantations (Fig. 6.3).

Table 6.1 Total counts of arboreal marsupials recorded within 2 ha sampling plots in each of the 12 vegetation categories sampled in this study. Abbreviations used for each vegetation category are also indicated. 0; species recorded, but outside the 2 ha plots.

Vegetation category		Common Brush-tail Possum	Common Ringtail Possum	Squirrel Glider
Plantings older than 10 years				
Large Plantations (20-100 ha)	POL	3	1	1
Medium-sized plantations (5-20 ha)	POM	7	14	0
Small plantations (<5 ha)	POS	1	2	
Narrow plantation strips (<50m)	POLi	1		1
Plantings younger than 10 years				
Young medium-sized plantations (5-20 ha)	PYM	2	0	
Young small plantations (<5 ha)	PYS	0		
Remnant forest and woodland				
Very large remnants (> 1000 ha)	RVL	4	5	0
Large remnants (>20-100 ha)	RL	20	21	0
Medium-sized remnants (5-20 ha)	RM	8	0	
Small remnants (<5 ha)	RS	5	1	
Narrow remnant strips (<50m)	RLi	6	15	3
Paddocks				
Paddocks with and without remnant trees	Pad		1	
Total individuals		57	60	5
Total no. 2 ha plots		26	22	4
Total no. sites		32	28	9

Nocturnal birds were recorded, usually by call, during standard spotlighting transects or during the following call-playback sessions. Calls were played only for a limited number of target species. The total number of sites where each species was recorded was as follows: Southern Boobook, 29 sites; Barking Owl, 8 sites; Barn Owl, 5 sites; Powerful Owl, 1 site; Tawny Frogmouth *Podargus strigoides*, 11 sites; White-throated Nightjar *Eurostopodus mystacalis*, 1 site; and Australian Owlet-nightjar, 5 sites.

The large home-range sizes for the owls, and the method of detection, makes it impossible to assign them to any particular vegetation category because most of the landscape is so variegated that any one category would be insufficient to support these species. The Powerful Owl may be an exception because it was recorded only in one very large remnant (Barambogie State Forest, now Chiltern-Mt Pilot National Park). The smaller-ranging frogmouths and nightjars were recorded mainly in the larger remnants of forest and woodland in the region, but also in some of the older eucalypt plantings.

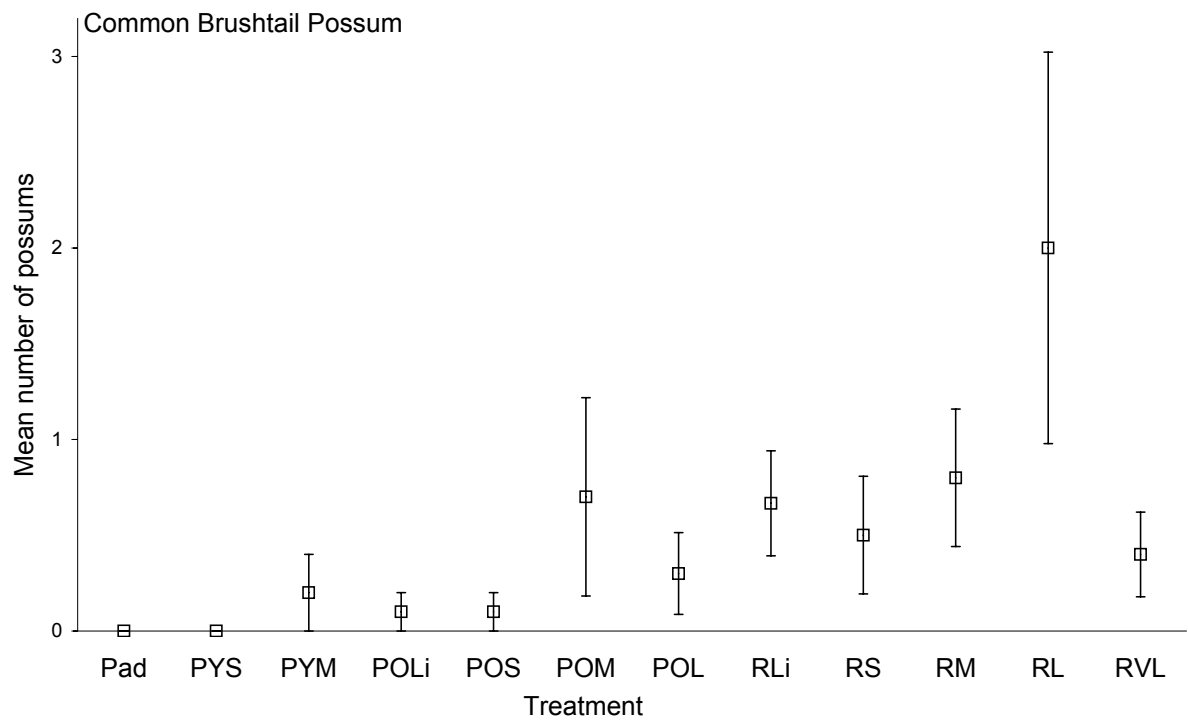


Fig. 6.1 Mean numbers of the Common Brushtail Possum (with standard errors) counted on 200 m transects (n=10) in each treatment category. See Table 6.1 for treatment codes.

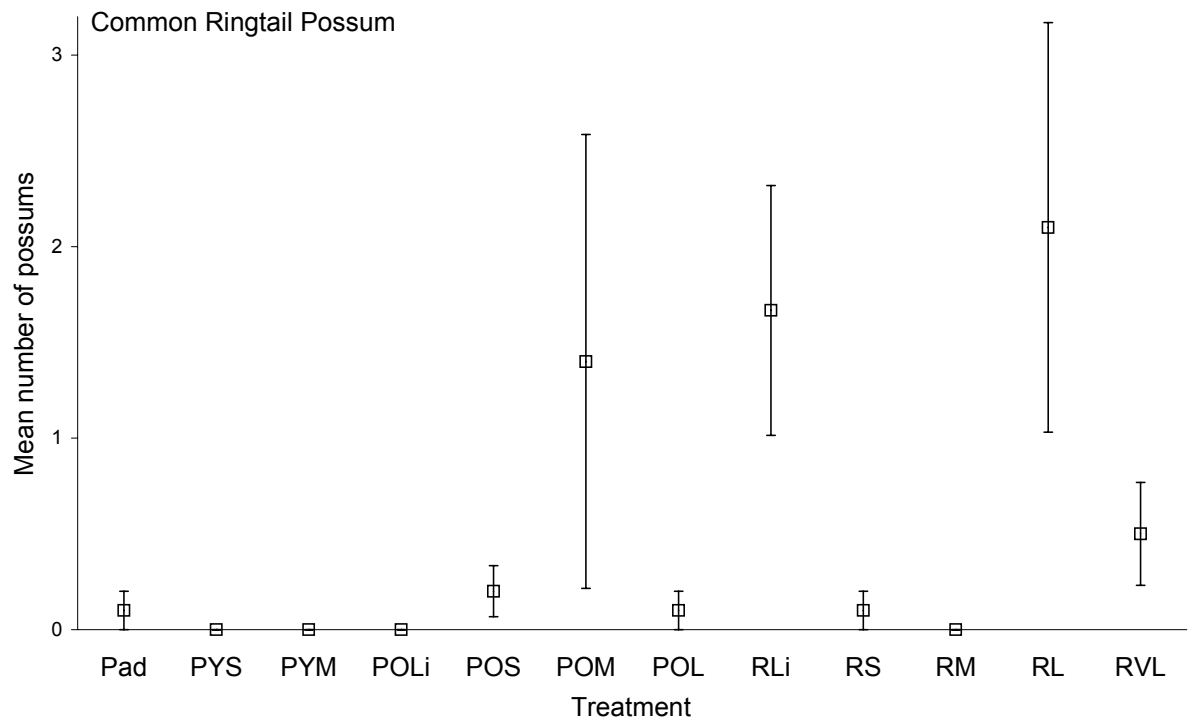


Fig. 6.2 Mean numbers of the Common Ringtail Possum (with standard errors) counted on 200 m transects (n=10) in each treatment category. See Table 6.1 for treatment codes.

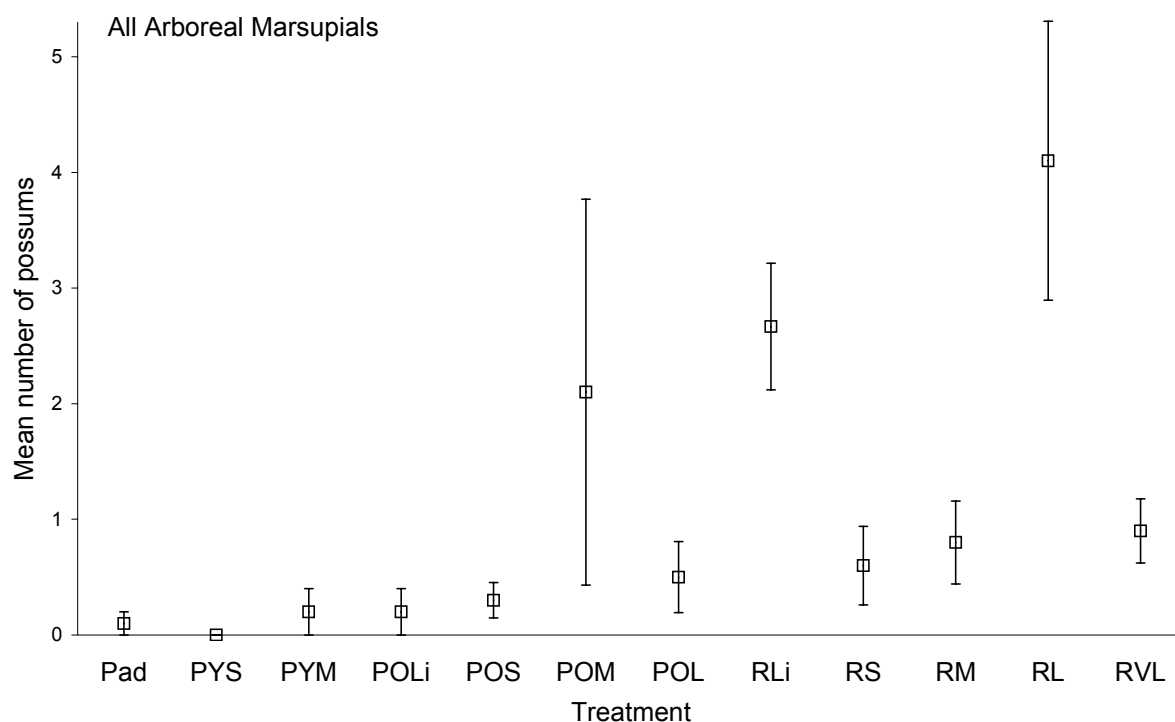


Fig. 6.3 Mean numbers of three species of arboreal marsupials (with standard errors) counted on 200 m transects (n=10) in each treatment category. See Table 6.1 for individual species data and treatment codes.

6.4 Discussion

Possums, gliders, owls and the Australian Owllet-nightjar require hollows in old trees for breeding and, with the exception of the Common Ringtail Possum which builds dreys, also for diurnal shelter. Two other species encountered, the Tawny Frogmouth and White-throated Nightjar, do not require tree hollows because they make stick nests or nest on the ground. Clearly young tree plantings cannot provide this essential resource for many decades and, thus, the spatial context of eucalypt plantings in relation to old trees and remnant forest and woodland is likely to be an important factor accounting for the use of planted vegetation by these species. Unfortunately, this study has shown that arboreal marsupials and nocturnal birds are missing from many of the remnants in the region, presumably because they are so small and degraded (see also Murphy 1999). Even several of the very large remnants sampled were impoverished, probably because they were comprised of lower quality habitat for these species. The greater relative abundance of several species in linear strips (< 50 m) of remnant forest and woodland embedded within the agricultural matrix suggests that the areas originally cleared more than a century ago were of greater productivity and habitat quality than several of the very large remnants (> 1000 ha) that remain uncleared to this day (see also van der Ree and Bennett 2001). This view is supported by the relatively high population densities recorded for some species within our category large remnants (20-100 ha) which was comprised mainly of travelling stock routes (TSR's). These areas represent the largest patches of remnant forest and woodland still existing within the agricultural matrix and they would probably also have been cleared had they not provided an important community service. The occurrences of the Common Brushtail Possum, Squirrel Glider and, in particular, the Common Ringtail Possum in older (> 10 years) plantings established near remnant forest and woodland demonstrates the potential of eucalypt plantings to provide a useful contribution to habitat for these species in fragmented landscapes. Eucalypt plantings have an important role in increasing the effective size and habitat capability of existing remnant vegetation.

There was no clear effect of patch size in relation to species detectability, although some medium-sized patches (> 5 ha) of eucalypt plantings and remnant forest and woodland had the greatest numbers of the Common Brushtail Possum and Common Ringtail Possum. These species are also known to occur regularly in small patches of remnant forest (Bennett 1987, Lindenmayer *et al.* 2000, Lindenmayer 2004, Kavanagh and Stanton 2002). Surveys of paddock trees in northern NSW revealed the Common Brushtail Possum as the most common arboreal marsupial, whereas the Common Ringtail Possum was not recorded in this situation (Law *et al.* 2000).

This study is complementary to the regional study of arboreal marsupials and nocturnal birds inhabiting the larger remnants and continuous forest occurring just to the east of the present study area (Kavanagh and Stanton 1998). Another similar study of arboreal marsupials was done in northern Victoria (Bennett *et al.* 1991). These studies confirm that the Common Brushtail Possum and the Common Ringtail Possum are the two most common arboreal marsupials inhabiting remnant forest and woodlands in this region, and that most of the other species likely to occur were also recorded, albeit in very low numbers. Species not recorded but expected include the Feathertail Glider and Eastern Pygmy Possum (Bennett *et al.* 1991, Kavanagh and Stanton 1998, Kavanagh 2004). Both species are small and difficult to detect, but the Feathertail Glider is known to forage in eucalypt plantation in southern Queensland (Kirk *et al.* 2000). The Sugar Glider was not recorded at elevations below 510 m by Kavanagh and Stanton (1998) and only one record of this species was made during our pilot study in the region (Kavanagh *et al.* 2001). The Squirrel Glider occurred exclusively in the drier forests below 300 m elevation (Bennett *et al.* 1991), and was reported by van der Ree (2002) as maintaining high population densities throughout a system of linear roadside remnants similar to those sampled in this study where the species was also recorded.

The large home-range sizes for the owls, and the method for their detection, makes it impossible to assign them to any particular vegetation category. Certainly, the Barking Owl was recorded at locations that had a high proportion of eucalypt plantings, but there was also a high proportion of remnant forest and woodland in the vicinity. The occurrence of this species in fragmented landscapes at low elevation (this study) contrasts markedly with their absence from higher elevation and continuously forested areas nearby (Kavanagh and Stanton 1998), and may indicate that the preferred habitat of the Barking Owl (and also likely for the Squirrel Glider and the Tuan) has been greatly reduced by clearing for agriculture in the region. The records of these two mammals in older eucalypt plantings suggests that plantings may eventually serve a useful role in habitat restoration for these species, and possibly also for the Barking Owl.

The Barn Owl is associated primarily with agriculturally-dominated landscapes (Taylor 1994, Kavanagh and Stanton 2002), so it is surprising that this species was recorded only at five sites, although it was not targeted specifically in sampling. The lack of any records for the Masked Owl, and only one record for the Powerful Owl, both forest species that were targeted, is also surprising. The widespread occurrence of the Southern Boobook, both in remnants and in continuous forest, is well known although numbers appeared less than expected (Kavanagh and Stanton 1998, 2002). The Australian Owlet-nightjar was also recorded less commonly than expected (Kavanagh and Stanton 1998, 2002). The Tawny Frogmouth appears to be a species well suited to older-aged plantings (or regrowth forest; e.g. Kavanagh *et al.* 1995) and several records were made in this vegetation category.

7. Ground-dwelling fauna (reptiles, amphibians and mammals)

7.1 Introduction

Ground-dwelling fauna could be expected to respond quite differently from flying and arboreal animals to the processes of fragmentation and revegetation, due to their reliance on ground habitat features. Reptiles and frogs form a significant component of the Australian vertebrate fauna in any landscape. As ectotherms, they are pre-adapted to the variable conditions of Australia that can limit food availability and both groups have diversified and spread across the landscape. Relatively few studies have looked at either group in relation to fragmented rural landscapes and those available have concentrated on reptiles. Work by Sarre *et al.* (1995) in Western Australia showed that one of two species of gecko coped well within a region of the wheat belt split into small fragments whereas the other did not. The latter was highly dependent on the quality of each retained patch and was not able to move between patches, greatly increasing the chances for long-term extinctions. MacNally and Brown (2001) looked at four different patch sizes and large reference areas in woodlands of central Victoria (similar to many areas in south-eastern New South Wales). They found that the most common species in reference areas were very rare in fragments, but other species showed the reverse trend. Recent work in south-eastern Australia by Fisher *et al.* (2003) found various factors to influence reptile distribution for each species. One skink preferred areas with few rocks, many spiders and high tree cover. Another was more likely to be found at sites with many ants and beetles and a high tree cover. A third skink and a legless lizard were significantly more likely to be detected in areas with a simple microhabitat structure. Overall species richness was highest in box woodlands and sites with highly variable habitat structure. Grazing was considered a potentially significant problem because it simplified the vegetation in patches.

In general, many species of reptiles have been recorded from rural lands (see Cogger 2000), however, the need for retained remnant vegetation for nearly all of these species is unknown. For most smaller species (most skinks, geckoes and small snakes) the presence of logs, rocks, grass tussocks or deep leaf litter as ground cover within an area is likely to be critical as they require such material for shelter from predators and more extreme environmental conditions. Larger species are more mobile and may be able to use sites lacking suitable cover if appropriate resources are available nearby.

Both major radiations of frogs within Australia, the tree frogs and ground frogs, are well represented in the region (Cogger 2000). The extant species represent a wide range of habits and some frogs can be expected to use all of the available habitats. Many are well known for inhabiting agricultural environments and for breeding in artificial dams (e.g. spotted grass frog, common froglet and perons tree frog; Healy *et al.* 1997, Lemckert 1998, Hazell *et al.* 2004). Hence, riparian sites within plantations are likely to represent suitable breeding habitats for many species.

The non-breeding habitat requirements of most frog species are poorly known, but should include terrestrial habitats that can be some distance from the breeding site (Lemckert 2004). As for reptiles, leaf litter and other ground cover are likely to be of great significance as general shelter sites, particularly given the permeable nature of a frog's skin, which requires a frog to find protection from desiccating conditions. When conditions become much drier they will seek protected "retreat" environments, which can be a riparian area, or could be underground, within vegetation, under a rock or in a tree cavity. Many Australian frogs can persist within modified agricultural landscapes (Hazell 2003), although their long-term ability to do so is uncertain.

In contrast, the abundance and species richness of small ground mammals is correlated positively with remnant size (Suckling 1982, Bennett 1987, Dunstan and Fox 1996). Some species of ground mammals are known to persist in forested corridors (Downes *et al.* 1997) and in resource-rich small patches if disturbance levels are low (Dunstan and Fox 1996). When remnants are retained in a matrix of pine plantations and native forest, small mammals, such as the Brown Antechinus *Antechinus stuartii* and the Bush Rat *Rattus fuscipes*, have a high probability of occurrence (Lindenmayer *et al.*

1999). Low viability in rural areas may be because small remnants do not provide the variety of resources required (including loss of cover as these areas are frequently grazed by domestic stock), and because they are vulnerable to predation when dispersing through farmland (Suckling 1982, Law and Dickman 1998).

While we have some knowledge about how mammals use remnant native vegetation, virtually nothing is known about their use of native tree plantations established on farmland. Plantations of exotic species (e.g. Radiata Pine) tend to favour introduced small mammals over native species (Barnett *et al.* 1977), although retained patches of native vegetation improves the probability of occurrence for native species (Lindenmayer *et al.* 1999). Law and Chidel (2002) found two native species of ground mammal (Bush Rat and Swamp Rat) in one year old eucalypt plantations on the north coast of NSW, compared to seven native species in surrounding native vegetation remnants, and none in paddocks.

7.2 Methods

Herpetofauna

This group was surveyed using three different approaches:

a) A combined timed visual/habitat search. In this case, the standard transects were searched on foot for 20 person-minutes per transect, with searches being restricted to within 10m either side of the midline. The search started at one end of the transect, with the searcher/s first visually surveying the ground, ground cover and tree trunks to locate and identify any active reptiles. After the visual search, the surveyor/s proceeded to walk slowly along the transect, turning over or looking in/under any available cover to locate reptiles or amphibians on the transect (active search). Visual and active searches were interspersed over the 20 person-minutes as required. All sites were surveyed once in November 2003.

b) Cover boards. These were placed at the start and end of each transect between December 2002 and March 2003. These generally consisted of 10 wooden boards (116cm x 15cm x 2cm) positioned as two layers on the ground to provide a “shelter” site. This arrangement meant that there were spaces both under and between boards for herpetofauna to use and approximately 0.97m² of cover was made available by the boards. We also added one metal pipe (50cm long and 1.5cm diameter) and/or 2 pieces of black plastic tubing (30cm and 2cm) to 16 sets of the cover boards to provide additional refuge sites. These tubes were placed adjacent to the boards with one end at or under the edge of the boards. The boards were checked during surveys undertaken in November 2003.

c) Audio-visual surveys of ponds/riparian areas. Ponds known to be present within the designated vegetation area were searched at night, listening and looking for frogs. We initially listened for any frogs present and calling at the sites for a period between three and six minutes. The species and numbers of calling frogs were recorded at the end of the period. We then searched the banks of the water body looking for any frogs that were present and not calling and recorded these. The time period for this depended on how long it took to cover the area thoroughly. This search was never less than five minutes and could be as long as 30 minutes. Surveys were undertaken at various times between March 2001 and November 2003.

We attempted to identify any herpetofauna first by sight to avoid disturbing other animals. If this could not be done, we attempted to catch the animal by hand to allow identification using Cogger (2000). If the reptile could not be captured, it was recorded as an unidentified member of that family (only skinks in this case).

Small Terrestrial Mammals

Two techniques were used to survey for small mammals. The first used aluminum box traps (Elliott-type A) and this was only used in a pilot study of 21 sites that spanned the range of our full set of treatments. A single transect comprising 10 Elliott traps was set at each of the 21 study sites. Traps were spaced at 10 metre intervals and set for two consecutive nights. Traps were baited with a mixture

of peanut butter, honey and rolled oats. In addition two large cage traps baited with raw meat were placed at either end of the transect. This pilot survey was undertaken in March 2001.

Due to low capture rates in Elliott traps (see Results), hair tubes were used as an alternative technique for the full survey of our 120 sites in November-December 2002. A transect of 5 small hair tubes lined with double-sided tape to catch hairs from small mammals was set at each site and left *in situ* for 10-14 days. Hair tubes were alternately baited with a mixture of peanut butter and rolled oats or sardines, and they were placed either on the ground or nailed 2 m high on a tree trunk. Hair samples were identified by Barbara Triggs, Mallacoota, Victoria.

Analysis

Due to the limited data obtained, statistical analyses were not appropriate. Therefore, all data are presented as graphs or figures to provide a visual assessment of the patterns related to the sampled areas.

7.3 Results

Reptiles

Searches of all transects on all sites produced very few records of reptiles or frogs. In total, only 165 reptiles were counted, either visually or under boards, representing 10 species. The vast majority of these were skinks, the dominant ones being *Lampropholis guichenoti* and *Hemiergis decresiensis* (see Table 7.1). Both were widespread, the former across all habitats, the latter in remnants of any size. Very few snakes were seen and only one turtle carapace found. The additional species of reptiles recorded off transects (Table 7.1) were found opportunistically, generally when surveyors were crossing roads when travelling between transects. Hence the species are present in the region, but were not located during the formal searches.

Table 7.1 Reptile species recorded during the surveys (*species not recorded on transects)

Species	Common name	No. counted	No. sites
<i>Carlia tetradactyla</i>	Rainbow Skink	17	14
<i>Christinus marmoratus</i>	Marbled Gecko	5	5
<i>Cryptoblepharus virgatus</i>		18	8
<i>Ctenotus robustus</i>	Robust Skink	5	4
<i>Egernia striolata</i>	Tree Skink	5	2
<i>Hemiergis decresiensis</i>	Three-toed Skink	29	13
<i>Lampropholis guichenoti</i>	Garden Skink	52	28
<i>Morethia boulengeri</i>	Boulengers Skink	8	6
<i>Tiliqua scincoids</i>	Eastern Blue-tongue	1	1
	Skink Unidentified	22	17
<i>Pseudonaja textiles</i>	Brown Snake	2	2
	Tortoise spp.	1	1
<i>Eulamprus tympanum</i> *	Southern Water Skink	0	NA
<i>Diplodactylus vittatus</i> *	Stone Gecko	0	NA
<i>Rhamphotyphlops nigrescens</i> *	Blind Snake	0	NA

The results of the reptile searches are presented graphically in Figs 7.1 and 7.2. These provide good visual comparisons of the mean numbers of reptiles and mean numbers of species recorded in the different site types, demonstrating the very low mean counts and species richness in the region. The majority of records came from areas of remnant vegetation, with larger remnants having noticeably greater reptile numbers and diversity. Smaller plantations held very few reptiles. Where reptiles were located, they were associated with rock outcrops, downed timber or large grass tussocks. No reptiles were recorded in paddocks during the searches and none were seen in this vegetation category at any stage.

Transect counts were far more successful in obtaining records of reptiles than were the cover boards where only 5 individuals from 3 species were located (all of which were small skinks).

Frogs

Many frogs were heard in various parts of the study area, but few were actually located within any of the sampling sites. In all, we conducted searches of 14 riparian sites, one swamp and 15 ponds, and located only five species of frogs (Table 7.2). Two ponds did not have a single frog recorded around them. There was insufficient coverage of ponds in different site types to provide any useful comparisons of their values as habitat to frogs in the region, however, we noted that essentially all ponds with water supported some frogs, regardless their location in the landscape. The other species of frogs recorded in Table 7.2 were recorded opportunistically during periods of rainfall. One notable species was the *Uperoleia* recorded at a pond near Albury. This was the only site at which this frog was recorded and its identification remains to be verified. There are records of *Uperoleia laevis* for this area, but the site is at the western edge of its range and the habitat was very different from that normally encountered for this species. It is possible that this is a new species.

Table 7.2 Species of frogs recorded during the surveys (*species not recorded on transects)

Species	Common name	No. counted	No. sites
<i>Litoria peroni</i>	Perons Tree Frog	18	5
<i>Crinia signifera</i>	Common Froglet	3	1
<i>Crinia parinsignifera</i>	Plains Froglet	14	3
<i>Neobatrachus sudelli</i>	Sudells Frog	1	1
<i>Limnodynastes tasmaniensis</i>	Spotted Grass Frog	12	4
<i>Limnodynastes dumerilii</i> *	Eastern Banjo Frog	0	NA
<i>Limnodynastes interioris</i> *	Giant Banjo Frog	0	NA
<i>Uperoleia laevis</i> ?*	Smooth Toadlet	0	NA

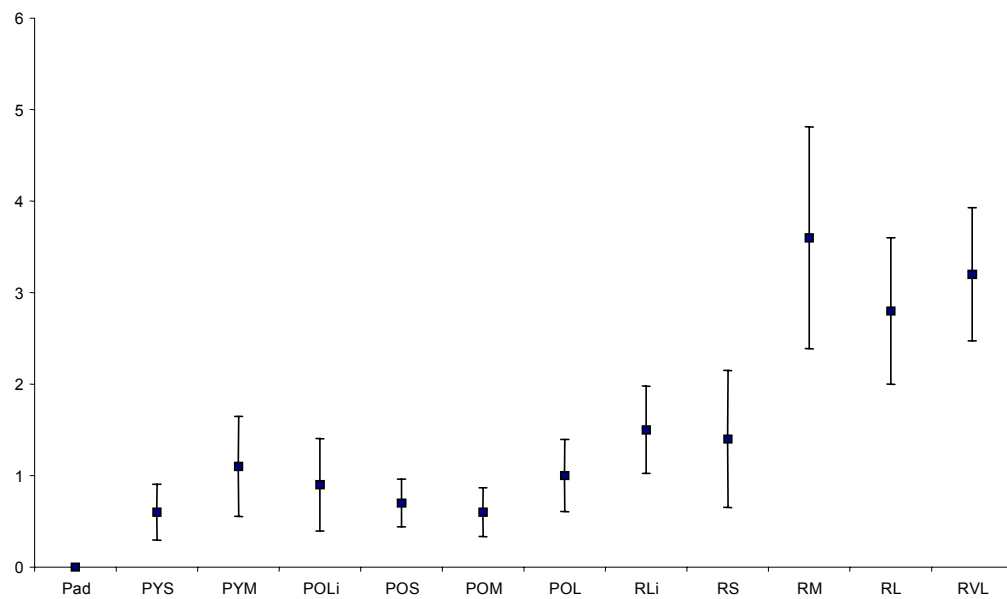


Figure 7.1 Mean numbers of reptiles (with standard errors) counted per transect in the differing site types.

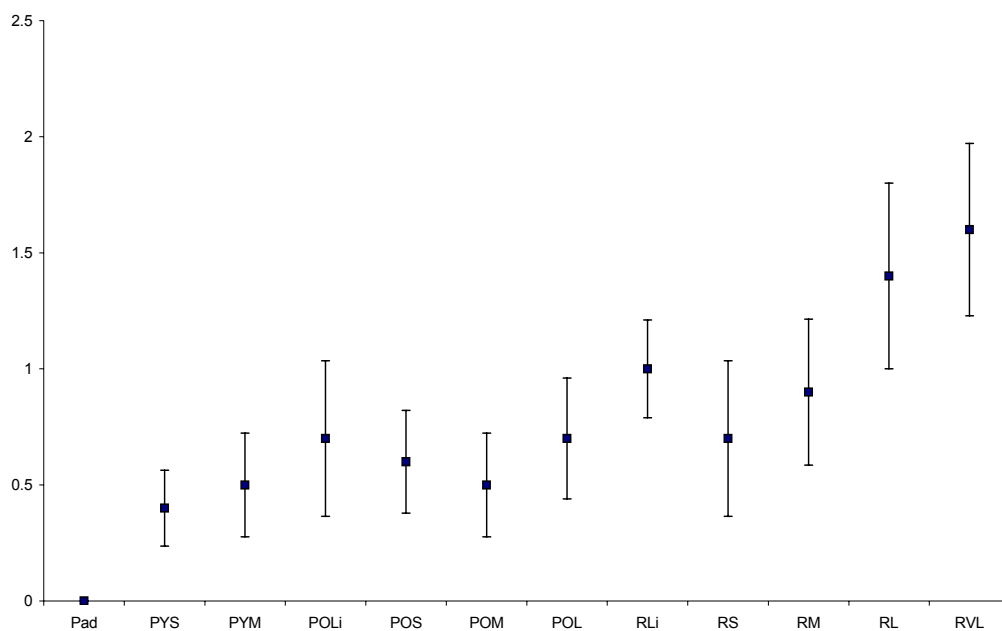


Figure 7.2 Mean numbers of reptile species (with standard errors) recorded on transects in the differing site types.

Small Mammals

In the pilot study, we accumulated 420 trap-nights using Elliott traps, but recorded no captures of small native mammals. Indeed, we had very low capture rates overall, comprising just two Black Rats *Rattus rattus* and 15 House Mouse *Mus domesticus*. The House Mouse *Mus domesticus* was widespread, but sparse, in all vegetation classes. Black Rats *Rattus rattus* were recorded only from remnant woodlands and old large plantings. No mammals were captured in the cage traps.

A number of other ground mammals, large and small, were observed opportunistically during field sampling for other species. They included Fox *Vulpes vulpes*, Cat *Felis catus*, Echidna *Tachyglossus aculeatus*, Rabbit *Oryctolagus cuniculus*, Hare *Lepus capensis* and Eastern Grey Kangaroo *Macropus giganteus*.

For the main study, 5 % of the 600 hair tubes contained hair and 10 % of the 120 sites recorded a hair. However, hair identification revealed that there were no native small mammals. Three records were from *Rattus* (not identified to species), five from Common Brushtail Possum *Trichosurus vulpecula*, one from Common Ringtail Possum *Pseudocheirus peregrinus* and nine records were of cats *Felis catus*. The remaining records were either cattle, sheep or dog hair.

Systematic spotlighting for arboreal marsupials (Chapter 6) provided two incidental records of small mammals. The semi-arboreal Brush-tailed Phascogale *Phascogale tapoatafa* was recorded in an old large planting and the Yellow-footed Antechinus *Antechinus flavipes* was recorded in a very large remnant (Tabletop Reserve).

7.4 Discussion

Reptiles

The plantations sampled in this study provided little useful habitat for the reptiles. The majority of those reptiles recorded came from large to small patches of remnant vegetation and remnants always outperformed their plantation counterparts. Even remnant roadside strips provided some useful reptile habitat, despite their very small size. No reptiles were recorded in paddocks. The major apparent difference between the two environments was the presence of ground cover. During the pilot surveys of the area, sites that had, at least, grass tussocks generally had rainbow skinks also present in them. By the time of the main survey, due to drought and grazing, this grass was gone and no skinks were seen. During these dry conditions, reptiles were almost exclusively associated with either rock or log cover. Even in the driest conditions, reptiles were still recorded in the large remnants, all associated with large, fallen, woody debris. As an example of the importance of cover, a survey in March 2003 at one plantation site found just two reptiles. These were located under a pallet that had been left after works at the site. There was no other cover available along the entire transect (no grass, logs or rocks). This pallet was absent in later surveys and no reptiles were recorded after this time.

Grazing appeared to be an important issue in our study sites, but the impacts of grazing on reptiles have not been well studied in Australia and the few available studies provide mixed results that may only apply to the arid zone. Heavy grazing has been identified as detrimental to reptiles by Smith *et al.* (1996), Woinarski and Ash (2002) and James (2003), either by decreasing abundance or diversity of reptile species. However, James (2003) noted that at least one species increased in abundance in heavily grazed areas and Read (2002) could find no effect of grazing. Morton (1990) suggested that many reptiles have been resilient to grazing due to their metabolic efficiency and non-herbivorous nature. This information though, does tend to support the belief that heavy grazing is likely to be detrimental to many reptiles and that grazing combined with drought almost certainly had a major impact on reptiles in this study. Ground cover of all types was almost completely absent in many sites and would have impacted any species reliant on ground cover. Species able to use standing trees as cover may better tolerate such effects.

The influence of patch size on reptiles is not well studied, but information suggests that it will depend on the species involved. Species dependent on forest or woodland resources are unlikely to cross cleared lands to reach new areas or exchange populations. More mobile and habitat-generalist

species can do so over potentially many kilometres of cleared lands. Sarre *et al.*(1995) reported one gecko species as capable of doing so, whereas another could not. Anderson and Burgin (2002) noted that there were different impacts of fragmentation edges on the three skinks they studied. One was most prevalent in the interior of a patch, one at the edge, and one scattered throughout but with the adults and juveniles more prevalent in the middle and subadults on the outside. The high edge to core ratio of very small patches or thin strips would then be unfavourable habitat for two species, but may favour the third. MacNally and Brown (2001) also noted that distributions of different species of reptiles in a range of patch sizes varied by species and there was no specific general response. Hence, impacts on reptiles will depend on the species involved.

Cover boards would seem to have presented a good opportunity to sample reptiles as they represented new ground cover in areas otherwise devoid of this shelter. However, boards were rarely used by reptiles. We believe that this was because the boards were placed out when drought conditions had already begun and the grass cover already depleted. Under these circumstances, the boards represented some shelter, but there was no food source associated with them and they probably did not have time to bed properly to thus provide insulated protection under them. We tried a few cover stations that had double the number of boards present but these were no more successful, probably because the drought had already impacted on reptiles. Using boards in better years might provide better results and confirm our belief that ground cover is of paramount importance in retaining reptile diversity and abundance in new plantations.

Frogs

Since the introduction of farm dams in Australia's rural landscape, some species of frogs should be greatly advantaged through the creation of huge numbers of potential breeding sites. Any species that can breed at such sites and can tolerate the fragmentation of habitat and loss of some native vegetation cover is likely to have increased significantly in number. The main question is how much habitat loss and fragmentation can they tolerate before the habitat change is no longer suitable for them. This is unknown. Hazell (2003) reviewed the impacts of habitat modification and fragmentation on frogs noting that we have little knowledge or understanding of what the impacts may be. Hazell *et al.*(2004), in looking at species using ponds in southern NSW, found some species to be more prevalent in ponds among natural vegetation and others more prevalent in cleared areas, while some species were only apparently found in natural breeding sites. Whilst the response is variable, we suggest the presence of any native vegetation within a region of cleared lands will provide some benefit to nearly all frogs within that landscape and may be essential for the retention of some species.

We were unable to make any real conclusions regarding the use and specific benefits of the different habitats within the region. Frogs are very difficult to sample away from breeding sites and even when a breeding site is present, it is difficult to determine where the frogs have come from (this may be hundreds of metres or even kilometres). Frogs were detected at nearly all of the ponds sampled, but generally in very low numbers (less than 5 individuals). Also, no ponds were available for sampling in the larger remnants, which were more suitable for reptiles and potentially also for frogs. The low frog numbers can be attributed to the drought, but their presence at almost all potential breeding sites indicates that landscape composition (patch size and landscape position) may not be very important to the frogs currently found in the region. The extensive use of pitfall traps on an opportunistic basis at times when wet weather precipitates frog movements would probably be the only means of providing clear information on the distribution and abundance of frogs across the landscape. Such work is expensive and time consuming and was beyond the scope of this study, but would seem essential to understand use of this environment by frogs.

Grazing has been demonstrated to be detrimental to frogs. Healy *et al.*(1997), Jansen and Healy (2003) and Bull and Hayes (2002) have all shown that increased grazing around breeding sites leads to decreases in frog numbers at those breeding sites. The impact depends on the species being studied, but is related mainly to changes in the riparian vegetation at the sites. This vegetation forms important shelter for resident frogs and components may be essential for successful breeding. Stirring of the water and pollution from stock droppings are also potentially negative influences on site quality for breeding frogs. It was notable in our study that no pond surveyed was in a relatively natural condition.

All had been subjected to some grazing and all had poor coverage of vegetation along the pond banks. Some ponds had essentially no vegetation on the banks and these ponds also had no or very few frogs present in them. Whilst we cannot demonstrate a statistically significant relationship between grazing and the abundance of frogs at the ponds, we believe that there has been a strong negative impact on frogs through grazing of the breeding sites.

Small Terrestrial Mammals

Although small mammals have undoubtedly declined west of the Divide (Dickman *et al.* 1993), the lack of native, small mammal captures in our study, in both remnant vegetation and in plantings, is a concern. A similar result was obtained by another study in a woodland remnant (225 ha) near Wagga Wagga which failed to record any native small mammals (Murphy 1999). Our main survey was undertaken during drought conditions and in late spring when male dasyurid populations would be depleted due to post-breeding die-off. However, our pilot survey was conducted during autumn, a more suitable time for trapping small dasyurids, and similar results were found. Nonetheless, it could be expected that small mammal numbers were low during our study. Notably, in one of our previous studies, *Antechinus* sp. was detected using hair-tubes in extensive forest and woodland of nearby Woomargama National Park (Stanton and Anderson 1998), but they were not recorded in the current study.

It is likely that *Antechinus flavipes*, *A. agilis*, *Sminthopsis crassicaudata* and *S. murina* are the only extant small, terrestrial, native mammals in the area (Strahan 1995). *Antechinus flavipes* appears to be more common in open woodland and it was the only native small mammal detected in River Red Gum woodland west of Albury, where it was associated with higher levels of woody debris (Mac Nally *et al.* 2001). In a local study of fauna in a mosaic of pine plantation, native plantings and remnant vegetation, it was only recorded in remnant vegetation (Klomp *et al.* 2001). *Antechinus flavipes* is known to persist and reproduce in linear remnants where large trees dominate (van der Ree 2003), but it is usually absent from linear and small patches (> 90 % of sites – van der Ree *et al.* 2003). The scarcity of native small mammals in our study region contrasts with remnants and eucalypt plantations of northern NSW which provide habitat for low numbers of native small mammal species, including *A. flavipes* (Law and Chidel 2002). The spotlight observation of the semi-arboreal Brush-tailed Phascogale *Phascogale tapoatafa* is a highly significant record of this locally uncommon (Stanton and Anderson 1998) and vulnerable species (NSW TSC Act, 1995). The record indicates that plantings do provide resources used by specialised species, in this case, at least when some old trees are present nearby.

8. Influence of landscape context on birds and bats in young plantings

8.1 Introduction

Plantation patch characteristics, including tree age, planting size, plant species composition and grazing history, are all important factors influencing bird species assemblages and, to some degree, bat species assemblages occurring in predominantly agricultural landscapes (Chapters 4 and 5). Birds and bats, probably because they are more mobile than other fauna groups, have shown the greatest response to plantation establishment in this study. Birds and bats are less likely to be restricted by poor habitat connectivity compared to other fauna groups, especially ground-dwelling vertebrates (small mammals, reptiles, frogs) and arboreal marsupials (possums, gliders). Instead, landscape context is likely to be an important determinant of species occurrence and hence, the potential for young tree plantings to be utilised by these more mobile species. Tree cover has previously been identified as a highly significant predictor of the richness of woodland birds at the landscape scale (Bennett and Ford 1997, Radford and Bennett 2004). Both studies, one situated just south-west of the present study area, reported the numbers of woodland birds to be more strongly correlated with total tree cover than with the degree of fragmentation, suggesting that the total amount of habitat in the landscape may be an important cue for bird species. McIntyre and Barrett (1992) also recognised the distinction between a “variegated” landscape and a “fragmented” landscape in terms of its suitability as habitat for many bird species.

The aim of this Chapter was to determine whether eucalypt plantings differ in their capacity to provide habitat for birds and bats depending on the amount of remnant forest and woodland in the local landscape. We felt that it was important to evaluate the landscape context of farmland in terms of the effectiveness of native vegetation plantings in restoring habitat for wildlife. We expected that revegetation would be more effective if it was planted on farms that were located in areas that had greater amounts of existing remnant forest and woodland.

8.2 Methods

Sixteen new sites, representing 8 sites in young (< 10 years old), medium-sized (5-20 ha) plantings and 8 sites in young, small-sized (<5 ha) plantings were located approximately 2-20 km further west of the existing 120 sites described and sampled above (Figs. 2.1 and 2.2). Efforts were made to locate these new sites in areas which did not differ significantly from existing sites in terms of geology, rainfall and forest/woodland type, but differing only in the proportions of remnant forest and woodland in the landscape. These differences are evident after inspection of the landsat image in Fig. 2.2. A change in native vegetation, where Cypress Pine (*Callitris* spp.) became a dominant feature, prohibited the location of new sites further to the west. This new “landscape” did not have significant areas of new plantings among the older age class (> 10 years), so comparisons of the birds and bats occupying these young plantings were made only with the 20 existing sites representing similar-sized (medium and small) young (< 10 years) plantings.

Quantifying landscape context

Differences in the extent of vegetation in each landscape were quantified by comparing the areas of dense tree cover and scattered tree cover as represented in the digitised spatial layers of Tree Canopy Density (DLWC, NSW) and Tree Density (DSE, Victoria). We did not redefine areas of remnant vegetation as we did for our original 120 sites because we could not access high quality colour aerial photographic imagery for the new, “cleared” landscape. Because Victoria and NSW used different criteria for classifying vegetation as dense, we derived a new classification of cover so that the two layers were comparable. The new classification was based on visual assessment against air photos in both NSW and Victoria. We constructed two cover classes: “dense” comprised dense cover from Victoria and > 20 % cover from NSW, while “scattered” comprised medium and scattered cover from Victoria and 5-20 % cover from NSW. The total area of vegetation remaining in each landscape

was then calculated as the total area of dense tree cover and scattered tree cover occurring within a 5 km radius around each site. This was then expressed as a percentage of the total area around each site (within a 5 km radius) and averaged across all sites in each landscape.

Bird sampling

Birds were sampled from 31 March-5 April 2004. The same sampling procedure as used in Chapter 4 was applied to the 2004 survey. In brief, each of the 16 new and 20 old sites were sampled twice, once each by two observers, using two fixed time, fixed area (10 minute, 50 m radius) sampling plots at each site. Bird counts were undertaken from shortly after dawn until late morning. The data used in analyses were the mean counts of bird species and individuals recorded per 10 minute count (4) at each site.

Bat sampling

Bats were sampled from 29 March-2 April 2004. The same sampling procedure as used in Chapter 5 was applied to the 2004 survey. In brief, one Anabat detector sampled each site for a full night by recording files to a lap-top computer via a zero-crossing interface (ZCAIM). Detectors were positioned so that the microphones pointed into gaps within the planted vegetation. Bat activity at a site was expressed for each species as the number of passes per night.

Identification of Bat Calls

Bat calls were identified using the procedure outlined in Chapter 5. That is, automated identification was enabled using Anascheme software and an identical identification key. A new feature was available for use in the version of Anascheme used for the analysis of the 2004 survey data. This was a set of filters that could be written to identify particular species prior to applying the identification key. Filters are particularly useful in identifying species that have calls which alternate in frequency. That is, consecutive calls which step up and down in frequency by 2-3 kHz. *Chalinolobus gouldii* is a good example where filters aid identification, because the calls of this species alternate in frequency, such that they are easy to identify manually, but the key used in Chapter 5 was unable to identify this feature and instead relied upon other call characteristics to identify species.

A filter was therefore designed to identify the alternating calls of *C. gouldii* and this was tested on reference (known) calls. The *C. gouldii* filter could identify 57 % of *C. gouldii* calls compared to 22 % using the key alone (n=23 calls). Two other species with overlapping, but non-alternating calls were also tested. The filter attributed *C. gouldii* identifications to 0 % of *Scotorepens balstoni* calls (n=28 calls) and 1 % of *Mormopterus planiceps* sp calls (n=70 calls), respectively. This low level of false identifications was considered acceptable for the present analysis. Thus, our procedure for identifying bat calls using Anascheme was to first apply the *C. gouldii* filter, then identified calls were removed allowing the automated identification key to be applied to the remaining set of calls.

Statistical Analysis

Differences between “treatments” (landscape context and planting size) were assessed using pre-planned contrasts and poisson regression with a log link function (implemented in SAS using Proc Genmod; SAS Version 8.2). Dependent variables included the total numbers of species and of individuals recorded during the four 10 minute counts, and within 50m radius of the two sampling plots, at each site. Analyses were also made on the components of these two variables that were designated as either “woodland-dependent” or “non woodland-dependent” birds (see Table 4.1). Tests for the poisson distribution of these variables were undertaken by visual inspection of simple plots of the count data. Plots were also made of the mean counts (with standard errors) for species and individuals by treatment type.

A two factor analysis of variance was used to test the effects of landscape context and planting size on bat activity and bat species richness. All bat data were log₁₀ transformed prior to analysis to ensure variances were homogenous and data were approximately normally distributed. As for birds, there were two levels within landscape context (variegated and cleared; e.g. McIntyre and Barrett 1992) and two levels within planting size (medium and small). Because landscape context was not replicated,

any significant differences detected for this factor were assigned only tentatively to the “treatment”. Differences in landscape context were thus attributed to differences between the two *areas*, because it is possible that other more subtle differences or past histories could potentially explain any statistically significant results. Careful matching of the characteristics of the two landscapes was done to minimise any potentially confounding effects.

8.3 Results

Comparison of Landscape Context

The two landscapes differed markedly in the extent of remaining indigenous vegetation (Table 8.1). The variegated landscape (i.e. the areas surrounding the initial 20 young planted sites) contained about twice as much vegetation with dense cover as the cleared landscape (i.e. the areas surrounding the new 16 young planted sites). The maximum percentage of dense vegetation surrounding a site in the cleared landscape was 7.78 % (minimum 2.73 %). The variegated landscape also contained more than twice as much scattered trees as the cleared landscape. Over all 120 sites in the variegated landscape, the average amount of dense and sparse vegetation cover retained was 12.13 % (± 0.10) and 13.93 % (± 0.09), respectively. These results clearly illustrate the high level of fragmentation and clearing that has occurred in both landscapes, but also the greater and quite extreme level of fragmentation and clearing that has occurred within the cleared landscape.

Table 8.1 The mean percentage (and standard error) of dense and scattered vegetation remaining within 5 km radius buffers surrounding study sites in the two landscapes.

Vegetation Cover	Variegated	Cleared
Dense	9.25 (± 0.34)	4.82 (± 0.09)
Scattered	17.17 (± 0.55)	7.50 (± 0.39)

The reliability of the combined spatial layer from Victoria and NSW was compared to our independently-derived mapping (based on colour aerial photographs) of remnant vegetation for the variegated landscape. Across the variegated landscape, the two methods were comparable. “Dense” vegetation comprised on average 12 % of the variegated landscape using the calibrated layer compared to 11 % for remnant vegetation (or 13 % including plantations) calculated by our independently-derived spatial layer. We did not attempt to calculate the area of scattered trees from the colour aerial photographs.

As expected, sites in the cleared landscape were more isolated from remnants. The median distances to forest and woodland remnants larger than 10 ha in size in each of the two treatment categories were 1.26 km (± 0.38) and 3.28 km (± 0.35) for the variegated and cleared landscapes, respectively.

Birds

A total of 83 bird species was recorded from the 36 sites sampled in early autumn 2004 (Table 8.2). The data available for analysis comprised 1649 birds from 69 species recorded within 50 m radius of the sampling plots during standard census counts. Of these, 764 birds from 57 species were recorded at 20 sites within the variegated landscape, and 785 birds from 39 species were recorded at 16 sites within the cleared landscape (Table 8.2).

No differences were found in the total numbers of bird species recorded in each landscape (Fig. 8.1, Table 8.3). This effect was the same in comparisons of the same-sized planting patches, and when they were lumped (Table 8.3). There was clearly a strong effect of planting patch size, with

medium patches (5-20 ha) having more species than small patches (< 5 ha) in both landscapes (Fig. 8.1), but this test was not specified in the pre-planned contrasts as the comparison had already been made in Chapter 4. However, significant differences were found in the total numbers of individual birds recorded, with more birds being recorded in the cleared landscape (Fig. 8.2, Table 8.3). Again, there were clear, but untested, differences between the numbers of individual birds recorded and planting patch size (Fig. 8.2).

So, while there were no differences between landscapes in the total numbers of bird species recorded, there were major differences in the composition of the bird species assemblages. The variegated landscape had significantly more woodland-dependent bird species than the cleared landscape, and this was due mainly to the increased capability of the medium-sized plantings and not to any extra contribution from the small plantings (Fig. 8.3a, Table 8.3). By corollary, medium-sized plantings in the cleared landscape had significantly more non woodland-dependent bird species (Fig. 8.3b, Table 8.3). The small planted patches in both landscapes were similar in having only few woodland-dependent bird species.

Including both categories of planting area, there was no significant difference in the total numbers of individual woodland-dependent birds recorded between landscapes (Table 8.3). However, as above, more woodland-dependent birds were recorded in the medium-sized plantings of the variegated landscape than in the cleared landscape (Fig. 8.3c, Table 8.3). Again, the cleared landscape had more non woodland-dependent birds, and this was due mainly to the greater numbers recorded in the medium-sized plantings within the cleared landscape compared to similarly-sized plantings in the variegated landscape (Fig. 8.3d, Table 8.3).

Major differences between the two landscape types included the greater abundance of the Superb Fairy-wren and Grey Fantail in the variegated landscape and the Crested Pigeon and Noisy Miner in the cleared landscape (Table 8.2).

Table 8.2 Mean numbers of birds recorded within 50m radius of sampling plots within small and medium-sized planted areas in each landscape during formal census counts. n=10 sites for each size category in the variegated landscape and n=8 sites for each size category in the cleared landscape.

Species	Variegated landscape			Cleared landscape			Total % all sites
	Small (< 5 ha)	Medium (5-20 ha)	% of Sites	Small (< 5 ha)	Medium (5-20 ha)	% of Sites	
Willie Wagtail	0.48	0.85	70	0.31	0.94	63	67
White-plumed Honeyeater	1.18	1.38	70	1.41	3.25	56	64
Red-rumped Parrot	0.98	0.40	45	1.06	1.88	75	58
Striated Pardalote	0.33	0.38	60	0.69	0.84	56	58
Eastern Rosella	0.35	0.20	35	0.56	0.91	63	47
Australian Magpie	0.33	0.13	50	0.19	0.41	44	47
Crested Pigeon	0.05	0.03	10	0.28	0.91	69	36
Superb Fairy-wren	0.83	1.48	55		0.03	6	33
Yellow-rumped Thornbill	0.05	0.20	25	0.16	0.22	38	31
Noisy Miner	0.15		10	1.19	0.47	50	28
Grey Shrike-thrush	0.05	0.25	35	0.03	0.13	19	28
Grey Fantail	0.13	1.05	45				25
Crested Shrike-tit		0.20	25	0.13	0.06	19	22
Welcome Swallow	0.05	0.63	20	0.16	0.09	25	22
Yellow Thornbill		0.73	15	0.47	0.44	25	19
Weebill	0.05	0.40	15	0.31	0.25	19	17
Crimson/Yellow Rosella	0.03	0.25	25				14
Silvereye		0.58	20		0.13	6	14
Galah		0.03	5		0.41	19	11
Noisy Friarbird	0.43	0.10	20				11
Golden Whistler		0.28	20				11
Rufous Whistler		0.20	20				11
Black-faced Cuckoo-shrike	0.08	0.03	15		0.13	6	11
White-winged Chough	0.10	0.23	10	0.19	0.47	13	11
Common Starling	0.10	0.08	10		0.56	13	11
Common Bronzewing	0.10	0.08	15				8
White-throated Gerygone	0.03	0.03	10	0.06		6	8
Striated Thornbill		0.60	15				8
Blue-faced Honeyeater	0.05	0.03	10	0.03		6	8
Yellow-faced Honeyeater	0.05	0.33	15				8
Grey Butcherbird		0.03	5	0.03	0.06	13	8
Australian Raven				0.03	0.22	19	8
Diamond Firetail	0.10	0.08	15				8
Tree Martin					0.91	19	8
Common Blackbird		0.08	10		0.03	6	8
Spotted Pardalote		0.13	5		0.03	6	6
Brown Thornbill		0.35	10				6
White-fronted Chat				0.19	0.03	13	6
Restless Flycatcher		0.05	10				6
Pied Butcherbird	0.03		5		0.03	6	6
Fairy Martin		0.10	5		1.81	6	6
Australian Wood Duck				0.06		6	3
Hoary-headed Grebe				0.03		6	3
Brown Goshawk					0.06	6	3
Australian Hobby					0.06	6	3
Nankeen Kestrel					0.03	6	3
Peaceful Dove		0.05	5				3
Sulphur-crested Cockatoo		0.15	5				3
Australian King Parrot		0.03	5				3
Brush Cuckoo					0.03	6	3
Southern Boobook	0.03		5				3
Laughing Kookaburra		0.03	5				3

Species	Variegated landscape			Cleared landscape			Total % all sites
	Small (< 5 ha)	Medium (5-20 ha)	% of Sites	Small (< 5 ha)	Medium (5-20 ha)	% of Sites	
Rainbow Bee-eater		0.03	5				3
White-throated Treecreeper		0.05	5				3
White-browed Scrubwren		0.05	5				3
Red Wattlebird		0.03	5				3
Little Friarbird	0.05		5				3
Brown-headed Honeyeater		0.05	5				3
Eastern Spinebill		0.03	5				3
Jacky Winter		0.03	5				3
Red-capped Robin		0.05	5				3
Flame Robin		0.03	5				3
Eastern Yellow Robin		0.05	5				3
Grey-crowned Babbler				0.09		6	3
Magpie-lark	0.05		5				3
House Sparrow				0.56		6	3
Eurasian Tree Sparrow				0.50		6	3
Red-browed Finch		0.40	5				3
Mistletoebird	0.03		5				3
Stubble Quail							
Little Pied Cormorant							
Collared Sparrowhawk							
Brown Falcon							
Dusky Moorhen							
Gang-gang Cockatoo							
Little Lorikeet							
Australian Ringneck							
Budgerigar							
White-bellied Cuckoo-shrike							
Olive-backed Oriole							
Pied Currawong							
Singing Bushlark							
(Eurasian) Goldfinch							
Total	6.20	12.90	57 spp.	8.72	15.81	39 spp.	

Table 8.3 Results of pre-planned statistical contrasts between treatments for total numbers of species and individuals. Data are Chi-square values and their significance levels (at df. 1). ***, P<0.01; **, P<0.05. WD, woodland-dependent. Codes for treatments: P, variegated landscape; N, cleared landscape; Y, young plantings; S, small; M, medium.

Contrast	Variable					
	Non WD species	WD species	Total species	Non WD birds	WD birds	Total birds
PYM+PYS vs NYM+NYS	5.3**	9.8***	0.4	31.4***	3.6	25.7***
PYM vs NYM	15.3***	23.0***	0.8	50.8***	4.2**	10.6***
PYS vs NYS	0.0	0.2	0.1	2.7	13.7***	15.3***



Fig. 8.1 Mean numbers of bird species (with standard errors) counted in 10 minute, 50 m radius sampling plots in each treatment category (n=40 for variegated landscape categories, n=32 for cleared landscape categories).

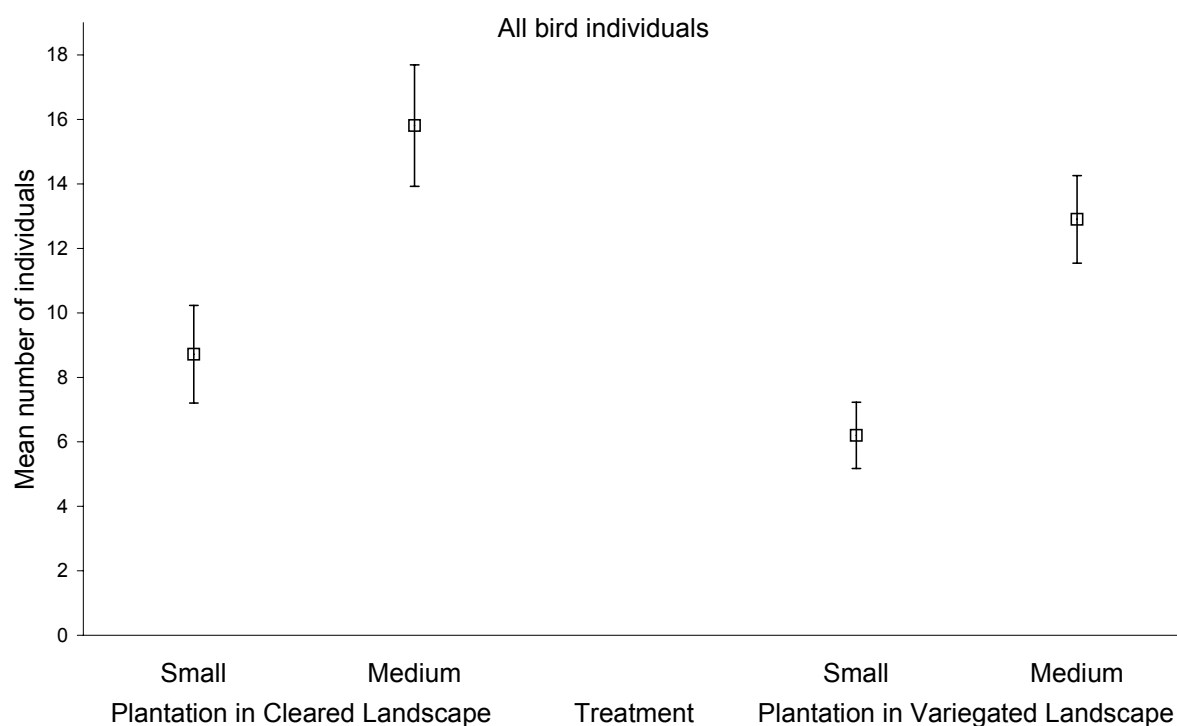
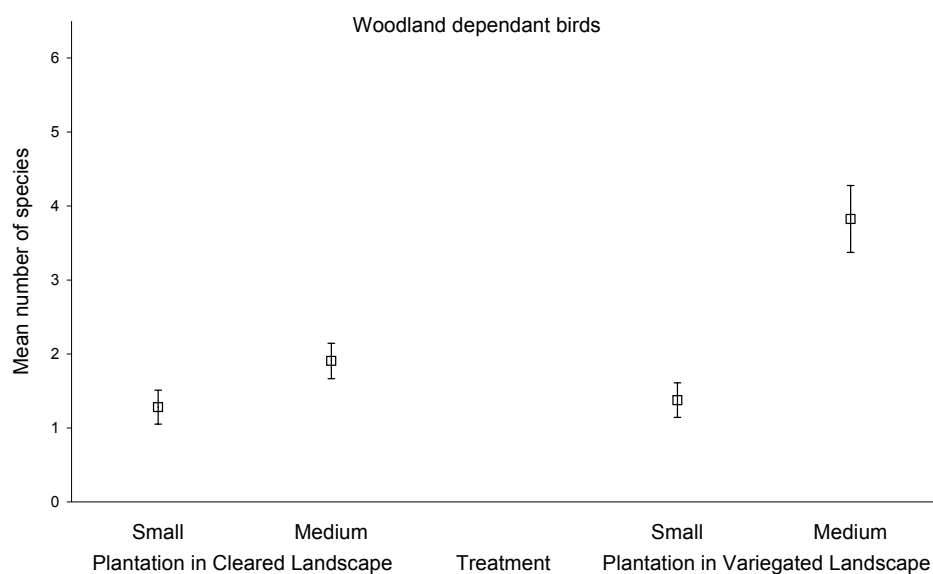


Fig. 8.2 Mean numbers of individual birds (with standard errors) counted in 10 minute, 50 m radius sampling plots in each treatment category (n=40 for variegated landscape categories, n=32 for cleared landscape categories).

a.



b.

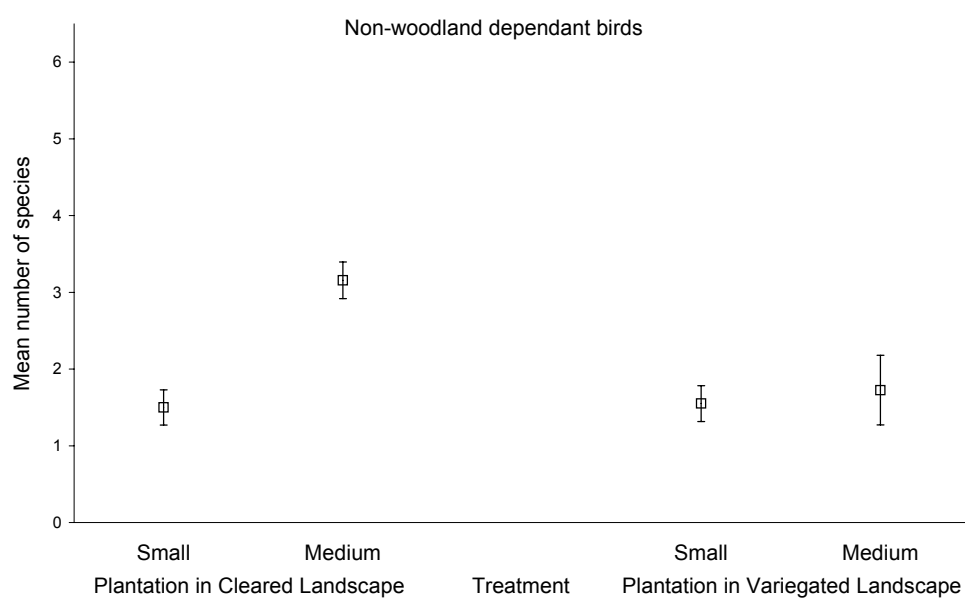


Fig. 8.3 a,b Mean numbers of woodland-dependent and non woodland-dependent bird species (with standard errors) counted in 10 minute, 50 m radius sampling plots in each treatment category (n=40 for variegated landscape categories, n=32 for cleared landscape categories).

c.



d.



Fig. 8.3 c,d Mean numbers of woodland-dependent and non woodland-dependent individual birds (with standard errors) counted in 10 minute, 50 m radius sampling plots in each treatment category (n=40 for variegated landscape categories, n=32 for cleared landscape categories).

Bats

Anabat detectors recorded for 396 hours across the 36 sites, resulting in 8397 Anabat files. Anascheme recognized 3,505 of these as calls, with 72 % identified to species level. Eleven species were recorded in total, with *V. vulturnus* by far the most frequently recorded species (Table 8.4).

Total bat activity was equally distributed across the two landscape contexts. There was a trend for higher activity in large plantings within both landscapes (Fig 8.4), but there was considerable variation in the data resulting in a lack of statistical significance (Table 8.5). The pattern for species richness was similar in that landscape context had no influence on the number of species recorded, but there was a strong trend for more species in larger plantings (Fig 8.5), although this was only marginally non-significant ($P=0.08$ – Table 8.5).

Table 8.4 Bat taxa recorded from 36 study sites and the number of passes per taxa (common names are given in Table 5.1, p 33).

Species	Number of passes
<i>Vespadelus vulturnus</i>	1266
<i>Tadarida australis</i>	383
<i>Nyctophilus</i> spp.	291
<i>Mormopterus planiceps</i> lp	213
<i>Chalinolobus gouldii</i>	210
<i>Vespadelus regulus</i> HF	40
<i>Vespadelus darlingtoni</i>	38
<i>Chalinolobus morio</i>	32
<i>Mormopterus planiceps</i> sp2	26
<i>Saccolaimus flaviventris</i>	8
<i>Scotorepens balstoni</i>	5

For activity levels of individual species, 2-factor ANOVAs revealed just one significant effect (Table 8.5). This was for *C. gouldii*, which was more active in medium-sized plantings than small plantings (Fig 8.6), but there was no significant landscape effect or interaction for this species (Table 8.5). None of the other species displayed any significant differences between the 2 factors. *Mormopterus planiceps* sp and *V. darlingtoni* were both very rare in the cleared landscape, being recorded at just a single site for each of the small and medium-sized plantings (Fig 8.6). *Scotorepens balstoni* and *S. flaviventris* were rare in both landscapes.

Table 8.5 Results for bat activity and species richness of a two-factor (planting size x landscape context) ANOVA. Species rarely recorded (*S. balstoni*, *S. flaviventris*, *M. planiceps* sp and *V. darlingtoni*) were not analysed. *F*- and *P*-values are shown, with those significant at the 0.05 indicated by *.

	Size	Landscape	SxL
Total activity	2.3, 0.14	0.0, 0.93	0.2, 0.64
Species richness	3.3, 0.08	2.3, 0.14	1.2, 0.27
<i>Chalinolobus gouldii</i>	7.1, 0.01 *	0.6, 0.46	2.4, 0.13
<i>Chalinolobus morio</i>	1.0, 0.33	0.0, 0.92	0.5, 0.51
<i>Mormopterus planiceps</i> lp	1.5, 0.23	0.6, 0.43	0.4, 0.55
<i>Nyctophilus</i> spp.	1.1, 0.31	0.0, 0.83	0.4, 0.54
<i>Vespadelus vulturnus</i>	0.5, 0.47	0.7, 0.42	0.0, 0.84
<i>Vespadelus regulus</i> HF	0.1, 0.72	2.7, 0.13	0.1, 0.72
<i>Tadarida australis</i>	0.0, 0.88	2.2, 0.15	0.3, 0.61

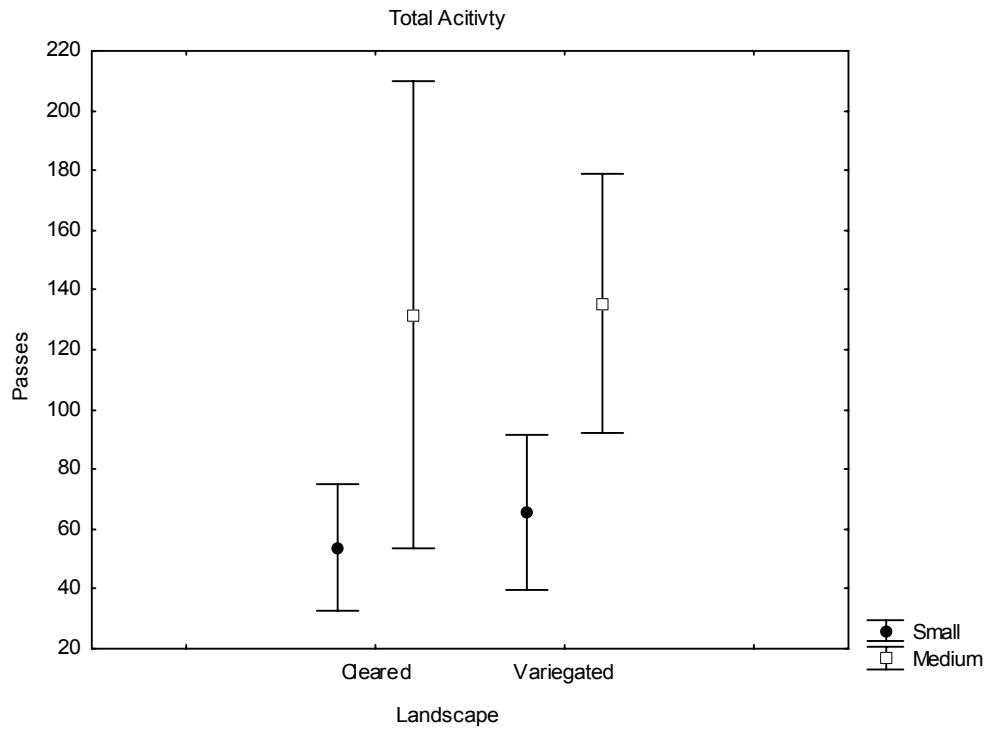


Fig 8.4 Total bat activity (passes per night) in plantings (small and medium-sized) within two landscapes (variegated and cleared). Means $\pm$ standard errors are shown.

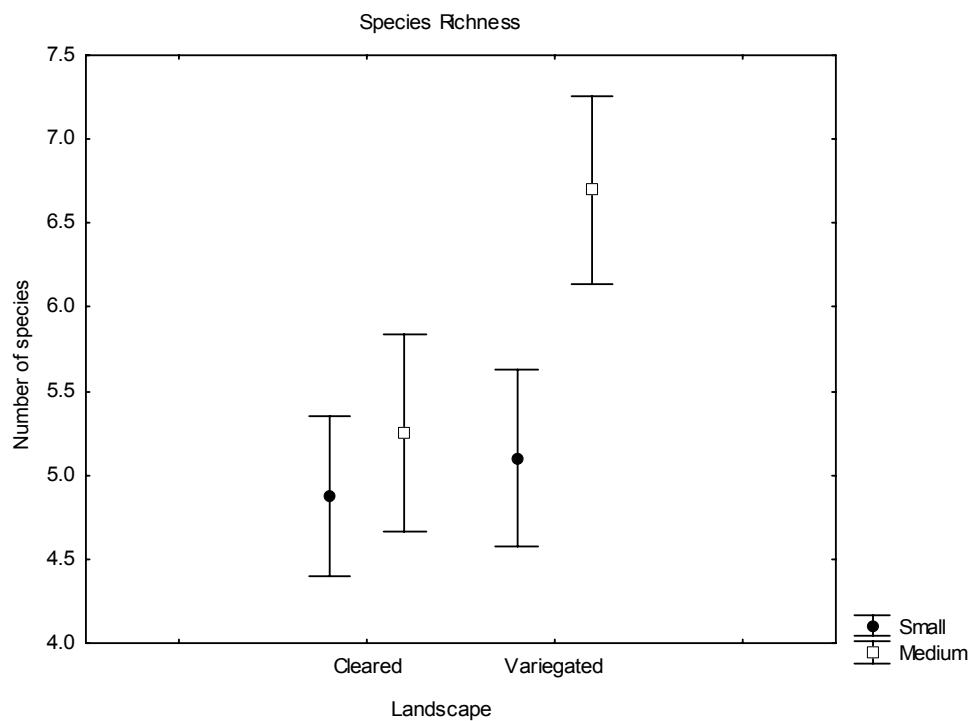


Fig 8.5 Species richness in plantings (small and medium-sized) within two landscapes (variegated and cleared). Means $\pm$ standard errors are shown.

Fig 8.6 Box plots of bat species' activity in plantings (small and medium-sized) within two landscapes (variegated and cleared). Medians, 25-75 % quartiles and non-outlier extremes are shown.

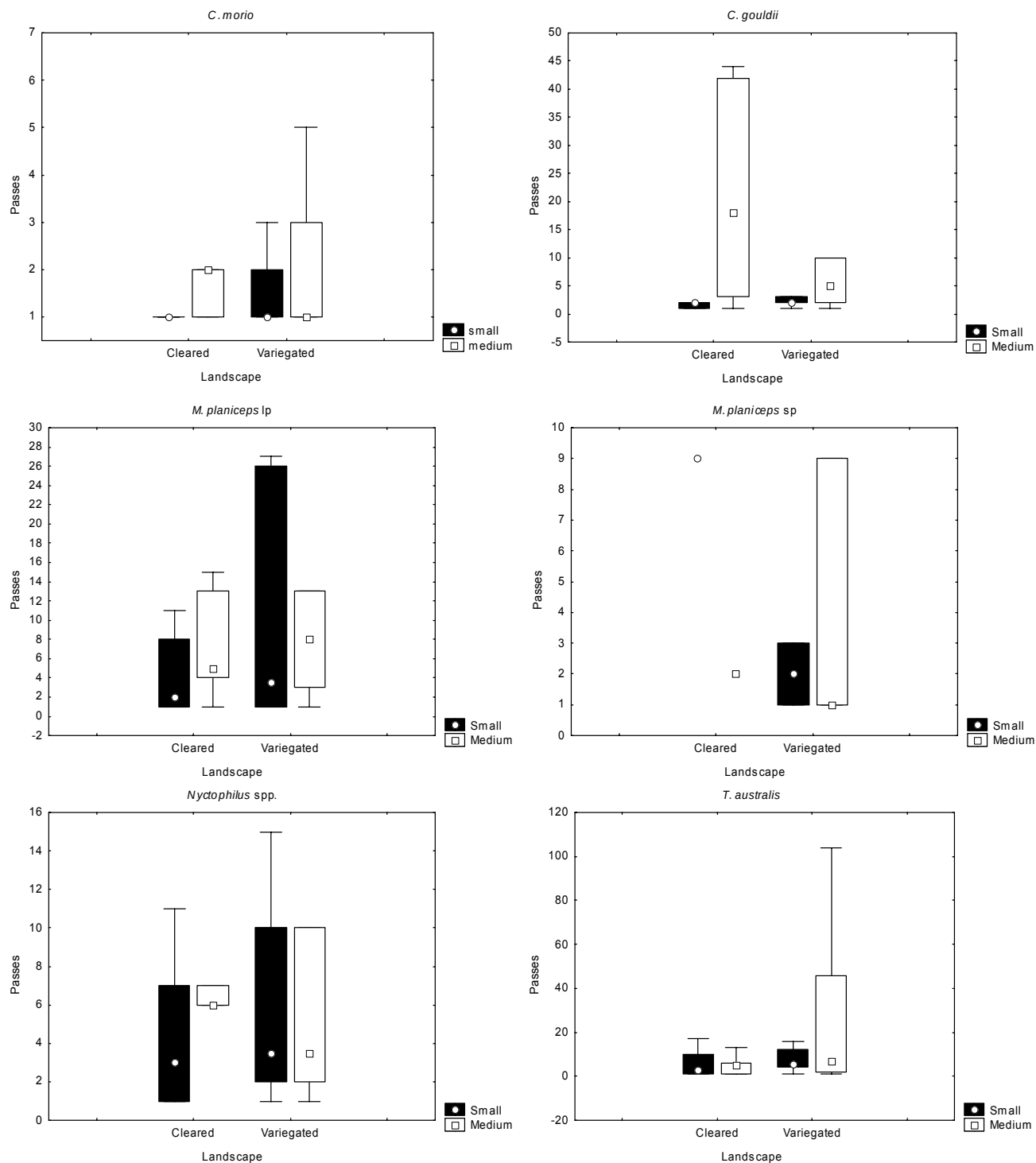
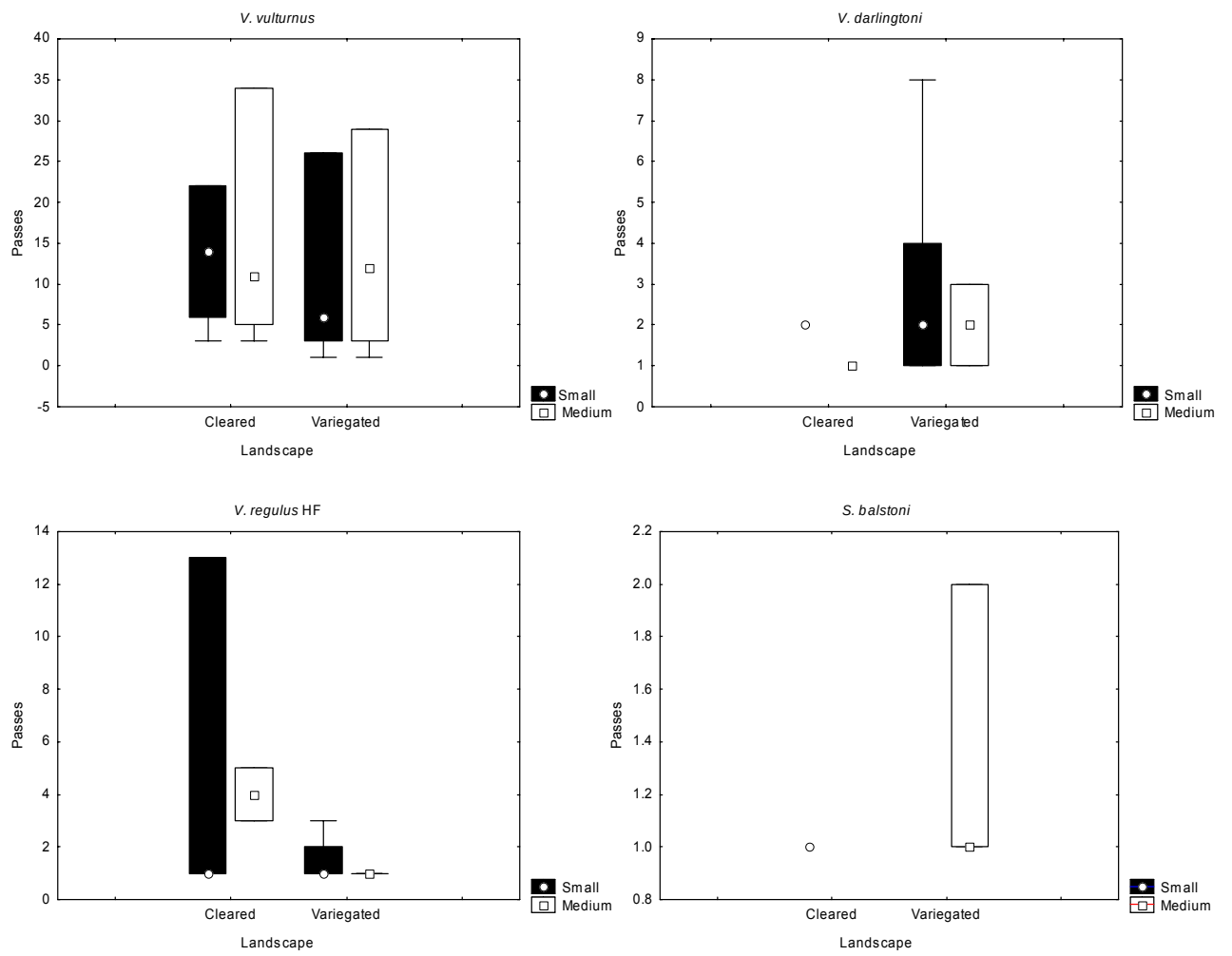


Fig. 8.6 continued



8.4 Discussion

Birds

About 50% more birds were recorded from the same 20 sites within the variegated landscape in late spring-early summer 2002 (1142 birds from 69 species; Table 4.2), compared to counts in autumn 2004, indicating the annual or seasonal variation that may be experienced in bird count data. Also, about 20% fewer bird species were recorded on these sites between the two sample periods (69 and 57). Aside from the likely climatic differences existing between the two sample periods (i.e. longer exposure to drought conditions prior to the 2004 sampling), a significant cause of the variation in counts of individual birds may be due to differences in bird detectability associated with the spring-early summer breeding period. Nonetheless, comparisons between the two landscapes in autumn 2004 are valid because all 36 sites were sampled at the same time.

The principal finding is that landscape context appears to have a significant effect on the uptake and occupancy of young (10 years old) native tree and shrub plantings by woodland-dependent birds. Sites proposed for planting that are situated in landscapes having greater amounts of remnant forest and woodland could be expected to be colonised or used by more species of woodland-dependent birds than if sites were planted in more sparsely treed landscapes. The potential role of eucalypt plantings, especially in variegated landscapes, appears to be a substantially new finding. However, relationships between bird species composition *in remnants* and the amount of remnant vegetation in the landscape has been reported recently (Bennett and Ford 1997, Major *et al.* 2001, Seddon *et al.* 2003). All plantings investigated in this study were used by birds, so even plantings in essentially cleared landscapes are likely to be useful as they provide habitat for many non woodland-dependent bird species.

Another key finding, while not the subject of statistical testing in this Chapter, is the influence of planting patch size (area) on bird occupancy. Plantings larger than 5 ha appear to be far more important to birds than smaller patches, and this supports the results presented in Chapter 4 which were based on many more samples. The role of patch size of remnant vegetation in influencing bird species assemblages is now becoming well known (e.g. Bennett and Ford 1997, Major *et al.* 2001, Seddon *et al.* 2003), but this finding has not yet been reported for eucalypt plantings.

While the floristic and structural composition of the remnant and planted vegetation is likely to have a major influence on the composition of bird communities, factors that were recognised and partially controlled in this study, it is likely that the influence of biotic factors, such as the presence of the Noisy Miner *Manorina melanocephala* could also be important (e.g. Major *et al.* 2001). Indeed, it is interesting to note that only four Noisy Miners were recorded at three of the 20 sites sampled in the variegated landscape but 29 Noisy Miners were recorded at 10 of the 16 sites sampled in the cleared landscape.

Bats

Bat species richness and activity was remarkably consistent in the variegated landscape between autumn 2004 and spring 2002. On average, five species were recorded in small plantings and seven species were recorded in medium-sized plantings in both 2002 and 2004. Similarly for bat activity, small plantings supported 50 passes in 2002 compared with 67 in 2004, while medium-sized plantings supported 96 passes in 2002 compared with 135 passes in 2004. Note that these small differences were well within the range of standard errors. This consistent pattern provides strong support for the reliability of our results and the patterns reported, despite the fact that sites were only sampled for bats on a single night.

Analysis of these results also produced similar findings, in that planting size was not found to have a significant effect on bat activity levels, except for *C. gouldii* and species richness (where the effect was marginally non-significant). The significant difference for *C. gouldii* was not apparent in the 2002 results, as both sizes of young plantings had low activity (< 5 passes) for this species. It is possible that improved identification of *C. gouldii* through the application of a filter to recognize its alternating calls improved the reliability of results for this species. The trend of recording more species in medium-sized

plantings in 2004 was detected in 2002 results. The generally weak influence of patch size on bats was discussed in Chapter 5.

For this part of our study the key finding concerning bats was that landscape context had no effect on bat activity or species richness. Although large remnants of woodland were absent from the cleared landscape, bats maintained their activity levels. Again, this result is consistent with analyses of the 2002 results that investigated the influence of landscape connectivity. For the 120 study sites in the variegated landscape, neither the amount of woody vegetation in buffers with radii of 0.5 km or 5 km influenced bat activity. The lack of an isolation effect was also found in another nearby study on insectivorous bats in fragmented vegetation on the south-western slopes of NSW (Law *et al.* 1999). Three species (*M. planiceps* sp, *V. darlingtoni* and *S. balstoni*) were each restricted to a single site in the cleared landscape, suggesting that their sensitivity to landscape context warrants further study. It is possible that their absence is not due to the level of clearing, but to an absence of some other key habitat feature. For instance, both *M. planiceps* sp and *V. darlingtoni* were shown to be commonly associated with River Red Gum *E. camaldulensis* forest (Chapter 5), which was absent from the areas sampled in the cleared landscape because the nearest large waterway was 10 km distant.

One important consideration in the interpretation of these results is the presence of scattered trees in paddocks. While our cleared landscape had little in the way of intact remnant vegetation, scattered trees were moderately common (7.5 % of the landscape, not including isolated trees in paddocks). It is well known that many species of bats forage around paddock trees (Law *et al.* 2000) and these, typically old, trees also provide some roosting opportunities (Lumsden *et al.* 2002). However, this is the first study to directly demonstrate that bats maintain their activity levels in a landscape where the indigenous vegetation is comprised mostly of scattered trees. Future studies on the importance of landscape context to bats should attempt to include landscapes where scattered trees are less prominent.

Thus, we conclude that small and medium-sized plantings in cleared landscapes are used by bat species to a similar extent to similar plantings in variegated landscapes, where more remnant vegetation is present. It is not known whether paddocks are used less frequently in cleared landscapes. If so, plantings would have greater relative value to bats than in variegated landscapes where paddocks are used as much as young plantings (Chapter 5).

9. General discussion

It is now generally recognised that the current network of native remnant vegetation in agriculturally-dominated landscapes is not sufficient to ensure the long-term persistence of biological diversity, and requires enhancement (Hobbs 1993, Saunders and Hobbs 1995). Revegetation is the primary tool for reconstructing habitat for a wide range of species, either *in situ* within planted areas or by augmenting the carrying capacity of the habitat provided by nearby remnants. The principal research issues are to provide guidance about the biological values of plantings of native trees and shrubs and the optimal spatial scale and landscape context for these plantings to achieve the best conservation outcomes. While there are a number of current studies underway to address these issues, few have been published. The recent review by Ryan (2000) could find only three relevant studies, all of which were about one major fauna group, diurnal birds. Since that time, Kinross (2004) has reported the avian use of planted narrow windbreaks on farms.

This study has shown that significant improvements in biodiversity occur when trees are planted in agricultural landscapes, compared to that existing in cleared paddocks. This is a substantially new and welcome finding, providing encouragement to those concerned about the potential and methods for restoring habitat in degraded or fragmented landscapes. However, there are still significant differences between the fauna of eucalypt plantings and those of native forests and woodlands. Birds and bats (both highly mobile) appear to be the fauna groups most capable of exploiting the new habitat provided by eucalypt plantings. Importantly, eucalypt plantings provided habitat for many woodland-dependent bird species, including many that have been classified as declining species. Indeed, similar findings have been reported for birds using narrow planted windbreaks (Kinross 2004). Other mammal species, reptiles and frogs were much slower to recolonise eucalypt plantings, and this is due partly to the low abundance of these species in the remnant forests and woodlands of the region. The value of eucalypt plantings for these species may depend largely on the habitat quality of each plantation patch (e.g. number of remnant trees nearby, shrub species cover and diversity, the amount of coarse woody debris, availability of water) and the proximity to remnants. Coarse woody debris, shrub cover and the availability of old hollow-bearing trees or their recruits have been identified as critically limiting factors for wildlife in agricultural landscapes (Bennett *et al.* 1994, Law *et al.* 2000, MacNally *et al.* 2001, Seddon *et al.* 2003).

The greatest contribution of eucalypt plantings to conserving biodiversity may be to greatly improve the quality of the landscape “matrix” around remnants. This would result in the creation of new habitat for many species, and provide increased opportunities for foraging and dispersal by other species that are dependent on habitat within remnants, thus augmenting the biodiversity value of existing remnants in the landscape. For example, arboreal marsupials appeared to be the group most dependent on remnants, but eucalypt plantings were used by these species when they were established adjacent to remnant vegetation.

Patch area appeared to strongly influence the numbers of birds occupying eucalypt plantings, and indeed remnants, with more species and individuals recorded in the larger (> 5 ha) planted and remnant areas than in smaller ones. Even in patches generally smaller than 5 ha, Kinross (2004) reported that wider planted windrows had significantly more birds than narrow planted windrows. She also recorded 24 of 65 species as breeding in linear plantings, including several regionally-declining woodland species. However, there was no evidence that planting patch size or remnant size was important in maintaining bat activity. This was also reported for a number of bat species in an earlier study of forest remnants in the Tumbumba region (Law *et al.* 1999).

Similarly, the age of eucalypt plantings appeared to be more important for birds than bats, with plantings older than 10 years generally having more birds than younger plantings. The greater diversity of plant species recorded in some younger eucalypt plantings may explain the greater numbers of birds recorded in young medium-sized plantings. An increase in the patchiness of vegetation (e.g. gaps) in large, old plantings appeared to be beneficial for bats.

Landscape context was shown to be an important factor affecting occupancy of eucalypt plantings by birds, but again, this was not the case for bats. Most bats in this study appear to be capable of

utilising revegetation in cleared landscapes provided there are at least some scattered old hollow trees nearby for roosting or foraging. It is well known that many species of bats forage around paddock trees (Law *et al.* 2000) and these, typically old, trees also provide some roosting opportunities (Lumsden *et al.* 2002).

9.1 Recommendations for restoring habitat on farms

Some, although relatively few, species appeared to be virtually restricted to remnant forests and woodlands, illustrating the importance of native remnants in regional conservation planning. Many other species were recorded most frequently in remnant vegetation. Efforts to restore habitat for animals on farms should begin with an understanding of the habitat value of existing remnant vegetation and its position in the landscape. Remnant vegetation should become the focal point for restoration efforts. The importance of mature, hollow-bearing trees for many species indicates that the loss of these trees in remnants subject to clearing is unlikely to be offset by new areas of revegetation.

New plantings should target creek lines and dry gullies, and where possible should adjoin patches of remnant forest and woodland. Many species, including *Antechinus flavipes*, are significantly more abundant in mesic gullies within generally dry forest landscapes (Soderquist and MacNally 2000), and these areas should be targetted for new planting sites.

Retention of relictual (paddock) trees within new plantings should provide a focus for foraging and offer a local source of roosts and nest hollows for animals (Law *et al.* 2000). Nest boxes have been used successfully in some eucalypt plantations to provide a local source of roosts for bats and other fauna (Smith and Agnew 2002), and a range of nest box designs is now available for different species.

Planting patch size is an important factor influencing occupancy by many species. While planted patches of all sizes and shapes, including narrow linear plantings, are used by wildlife, the results of this study suggest that planted areas larger than 5 ha in size are significantly more useful for animals.

New plantings should contain a mix of native tree species and, importantly, shrub species to increase their use by wildlife. Selecting tree species that are appropriate to the site, which are locally endemic, and which have complementary patterns of flowering, should give the best outcome.

For bats, gaps or more widely-spaced plantings might be beneficial as dense cover had a negative influence on bat activity. Alternatively, the presence of tracks is likely to benefit bats. In forests where vegetation is dense, small tracks large enough to permit 4WD access are used extensively as flyways and foraging areas by many bats (Law *et al.* 1998, Law and Chidel 2002). Other management practices that reduce clutter, such as thinning of high-density stands, might also be beneficial.

The inclusion of a dam is likely to be beneficial to bats and other fauna, especially frogs. Bat activity is known to be high around water, with dams being used heavily by bats (Lumsden and Bennett 1995, Law *et al.* 1998). Bats visit water to drink and also to feed on the typically high numbers of flying invertebrates found near water.

The provision or retention of cover in the form of logs, litter and/or rocks is crucial for the habitat of most species of ground-dwelling mammals, reptiles and frogs. Habitat restoration for some of these species has been demonstrated by MacNally and Horrocks (2002) who reported a significant increase in numbers of *Antechinus flavipes* following the addition of large woody debris (logs and large limbs) in flood-plain forests of south-eastern Australia.

Site preparation for plantation establishment often removes log cover; this practice is highly detrimental to reptiles using the site. This is exacerbated by grazing which often occurs as soon as the planted trees become taller than domestic stock. Grazing removes the grass tussocks and shrubs that provide cover, and nesting habitat, for small ground mammals, lizards and some birds. Log removal and grazing are probably also detrimental for the non-breeding habitat requirements of frogs. Efforts should be made to retain any available log cover and to exclude grazing, especially during drought, from planted sites.

Stock grazing is also likely to have a deleterious effect on frogs through modification of their breeding habitats. All ponds searched by us that were in areas subject to stock grazing lacked any significant bank coverage of vegetation or emergent aquatic vegetation. Both are significant as shelter for calling and resting frogs and for tadpoles using ponds (Healy *et al.* 1997, Jansen and Healy 2003) and their loss must impact on frogs attempting to breed in these water bodies. We recognise that the prevention of stock from trampling and grazing at water bodies is not likely to be an option as such watering points are necessary. However, an alternative is to fence off portions of water bodies to exclude stock, while still allowing them access to water; this allows some of the pond to develop natural vegetation and a diversity of habitats for different frogs.

9.2 Future research needs

Eucalypt plantings clearly provide shelter and foraging habitat for many species of animals. Unknown, however, is the extent to which eucalypt plantings can provide breeding habitat for these species. A large proportion of Australia's forest and woodland fauna utilise, and many require, hollows in old trees for breeding (Gibbons and Lindenmayer 2002). Eucalypt plantings will be unable to provide this resource for many decades, or perhaps never if they are managed on a short rotation for wood production.

Research is needed to document the role that eucalypt plantings play in contributing to the breeding and foraging habitat requirements of animals, and the complementary role of nearby remnant vegetation in providing these resources. Comparative studies of species nesting success are required.

Opportunities for improving the value of eucalypt plantings for wildlife need to be explored. For example, the role of nest boxes, the provision of coarse woody debris or artificial cover, the exclusion of grazing, the provision of water, and the response to increased diversity of planted tree and shrub species.

The extent to which eucalypt plantings can augment the carrying capacity of remnants for wildlife is an important issue, especially given the isolation, small size, and ongoing threats to many remnants.

The potential of eucalypt plantings for assisting the recovery of threatened species needs to be documented.

There are now a number of "biodiversity toolkits" in preparation to underpin the development of both market-based and government-subsidy schemes for effecting land-use change in rural areas (e.g. Parkes *et al.* 2003, Oliver and Parkes 2003). These toolkits depend on assumptions made about the response of "biodiversity" to varying levels or classes of vegetation type and condition, and proximity to remnant vegetation. Habitat rankings for different surrogates, including vegetation type, patch size, vegetation condition, connectivity and landscape context need to be evaluated for a wide range of species. This is needed to provide confidence that the components of these toolkits are relevant and that the rankings for different restorative actions have been properly calibrated.

9.3 Environmental services

Managing for biodiversity, identified as one of a number of "environmental services" in rural landscapes (Grieve 2003), is not like managing other values such as carbon or salinity. Biodiversity represents, potentially, thousands of species at any one site, whereas carbon or salinity represent only one unit (or "species"). Each species responds in its own way to the physical and biological features of its environment, and in its response to disturbances and restorative measures. Certainly, species can be grouped together as having similar broad habitat requirements, but there are many such groupings. Each species group may require environmental attributes (i.e. elements of habitat) that are frequently independent of the requirements of other groups, and which may often be inversely-related to the requirements of other groups. Thus, the process of "optimisation" of biodiversity benefits is not simple because species frequently do not respond in a similar manner, yet some will be of greater conservation concern than others. Clear statements are required about the goals of management, particularly in relation to the identification of priority species for management, because it is not possible to maximise the "status" of all species at each location.

10. References

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