

**THE INFLUENCE OF EXOTIC SHRUBS ON BIRDS OF URBAN  
YELLOW BOX-BLAKELY'S RED GUM (*E. melliodora*-*E. blakelyi*)  
WOODLAND IN CANBERRA.**

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## Abstract

This study considered the influence of exotic shrubs on birds in urban patches of Yellow Box-Blakely's Red Gum (*E. melliodora-E. blakelyi*) woodlands in the Australian Capital Territory, Canberra. The aim of this research was to identify native and exotic birds that have their abundance influenced by exotic shrubs. The purpose of this work was to provide more information to managers of this woodland about the potential impacts of weed control on birds living in woodland reserves adjacent to suburban areas.

Birds were sampled between 1996 and 1998 using the twenty minute area-search method to derive estimates of bird abundance. Around 680 twenty minute area-searches were completed, with 665 of these undertaken at 12 two hectare plots with varying levels of exotic shrub cover over a period of ten months. The abundance of birds for each of the ten months sampled were compared by pooling sites into four classes of exotic shrub cover. Classes of exotic shrub cover were nil, light, moderate and dense. Seasonal inferences were drawn from non-parametric analysis of variance. Non-parametric measures of association were used to test for correlation between the mean abundance of bird species at different woodland sites and percentage foliage estimates of exotic shrub cover. Percentage foliage estimates of native shrub cover were included in tests for correlation between bird abundance and exotic shrub cover by applying partial measures of association. To support statistical information, observations of birds in exotic shrub cover were also recorded. In one woodland site birds were sampled before and after the removal of exotic shrub cover. No statistical tests were applied to these samples because of a lack of replication; however, descriptive graphs of the abundance of selected birds following weed control are presented. To investigate the effect that exotic shrub invasion may have on the composition of bird assemblages in woodland cluster analysis and ordination of the 12 sites using the mean abundance of the 75 species recorded between July 1997 and June 1998 were also undertaken.

The presence of exotic shrub cover in *E. melliodora-E. blakelyi* woodland in Canberra was found to have differential effects on bird abundance. Wrens, finches, thornbills, whistlers and pigeons were more abundant in woodland sites where exotic shrubs were present when compared to sites with no or little exotic shrub cover. Fruit-eating birds, such as Silvereyes (*Zosterops lateralis*), Pied Currawong (*Strepera graculina*) and Crimson Rosella (*Platycercus elegans*), were more

abundant in woodland with exotic shrubs in winter and autumn when these shrubs provided food in the form of berries. The abundance of fruit-eating birds, and wrens and finches was reduced in a single woodland site following the removal of most of the exotic shrub cover. The Common Blackbird (*Turdus merula*) was the only exotic bird which showed a strong association with exotic shrubs in woodland, while the Laughing Kookaburra (*Dacelo novaeguineae*) was negatively correlated with exotic shrub cover, possibly because prey is harder to detect and capture in woodland with a shrubby understorey.

Cluster analysis and ordination of the 12 woodland sites did not group sites into the four experimental classes used to undertake analysis of variance. Multivariate analysis, however, did reveal that seasonal peaks in the abundance of fruit eating birds affected the composition of bird assemblages by increasing the mean abundance of these birds in densely invaded sites. Similarly, structural differences in the understorey resulted in some birds being more abundant in woodland sites invaded by exotic shrubs when compared to sites lacking a shrubby understorey. The distance between some sites confirmed this stark difference in bird life when plotted in three dimensions.

The results of this study suggest that exotic shrubs add food and structural complexity to woodland habitat. Benefits of structural complexity for small native birds in woodland include nest sites and protection from predators. These benefits may operate at certain thresholds of invasion, as the woodland site with the densest level of exotic shrub invasion showed a slight decrease in the number of wrens and finches. Adverse impacts from exotic shrub invasion may include reduced open ground in which to forage and loss of floristic diversity in the understorey. In effect, exotic shrubs add and remove resources in woodland habitat, benefiting some bird species and limiting others.

These findings suggest that the removal of exotic shrub cover in woodland located in urban landscapes simplifies the structural complexity of the understorey, reducing the quality of habitat for some birds. Thus, adverse impacts on biodiversity arising from the invasion of exotic shrubs in woodland need to be considered against the important role that a diverse bird population has in maintaining ecosystem function.

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# Chapter 1

## Introduction

### 1.1 Introduction

The conversion of temperate eucalypt woodlands into highly modified agricultural landscapes has resulted in local, regional and global extinction of flora and fauna in Australia (Yates and Hobbs 2000). It has been estimated that around 500 000km<sup>2</sup> of woodlands have been cleared in Australia with much of this clearing occurring in temperate woodland ecosystems (Yates and Hobbs 2000). The end result is that temperate woodlands are one of the most threatened wooded ecosystems in Australia (Robinson and Traill 1996).

Of the estimated 20 percent of temperate woodland that has not been cleared in Australia, weed invasion represents just one of many threats to its continued existence (Robinson and Traill 1996). Adair (1995) suggested that despite a lack of information, it was probable that weed invasions in natural ecosystems have had profound effects on the structure, function and composition of native plant and animal communities. More recently, Groves and Willis (1999) noted that weeds in Australia were responsible for the presumed extinction of four native plant species and usually have negative effects on ecosystem function. They did suggest, however, that weeds may also have positive impacts, depending on the group of organisms being considered.

The possibility that invasive exotic plants may have, in some contexts, positive and negative effects on wildlife has been considered before (Brown *et al.* 1991; Loyn and French 1991; French and Zubovich 1997). However, systematic investigations of the influence of environmental weeds on fauna in Australia remain relatively rare (Adair and Groves 1998). This study aims to contribute to addressing this gap in knowledge and to provide local managers of urban *E. melliodora*-*E. blakelyi* woodland with additional information on which to base their management of invasive exotic shrubs in urban woodland. Chapter 1 provides a brief overview of the study area and outlines the rationale, research focus and structure of this thesis.

## 1.2. The ACT and Canberra as an environment for birds

Canberra is the urban centre of the Australian Capital Territory (ACT) and lies at a latitude of between 35 and 36 degrees south of the equator (Australian Bureau of Statistics (ABS) 1999). The climate of the ACT is continental and characterised by significant variation in temperature between seasons, with warm to hot summers and cold winters (ABS 1999). Average annual rainfall for Canberra based on readings taken at the airport is 634 mm, and is usually evenly distributed throughout the year (ACT Commissioner for the Environment 1995). Information on monthly rainfall and mean temperature for the ACT in 1997 and 1998, is presented in Figures 1.1 and 1.2.

The total area of the ACT is just under 2400 km<sup>2</sup> and 60 percent of this land is hilly or mountainous (ABS 1999). Elevations vary from 450m above sea level to 1910m (ACT Commissioner for the Environment 1995). The presence of this high country, in combination with the nature of the climate in the ACT, influences the movements of a number of bird species and Canberra's bird population changes markedly with season (Taylor and Canberra Ornithologists Group 1992). Generally, the abundance and species richness of birds in Canberra is highest in the spring and summer months. However, there is also an influx of birds into Canberra from the high country in autumn and winter, with birds attracted to the milder conditions and food resources of the city (Taylor and Canberra Ornithologists Group 1992). Factors governing these bird movements and migrations are temperature, altitude and the availability of food (Frith 1976; Taylor and Canberra Ornithologists Group 1992).

A key difference between Canberra and many other Australian towns and cities is its carefully planned design. Integral to this design is the retention of natural bushland in reserves near urban settlement (ACT Government 1996a). This design is why bushland adjoins the suburbs of Canberra as part of Canberra Nature Park (ACT Government 1996a). Such close proximity means the human inhabitants of these suburbs enjoy easy access to these areas. Although there are many positive aspects to this design feature of Canberra, it does create a source of pressure on the natural values of these areas. Pressures which may result from close proximity to urban settlement include erosion from access tracks and roads, the presence of dogs, cats, horses and introduced birds, weed invasion and impacts from weed management, increased risk of fire,

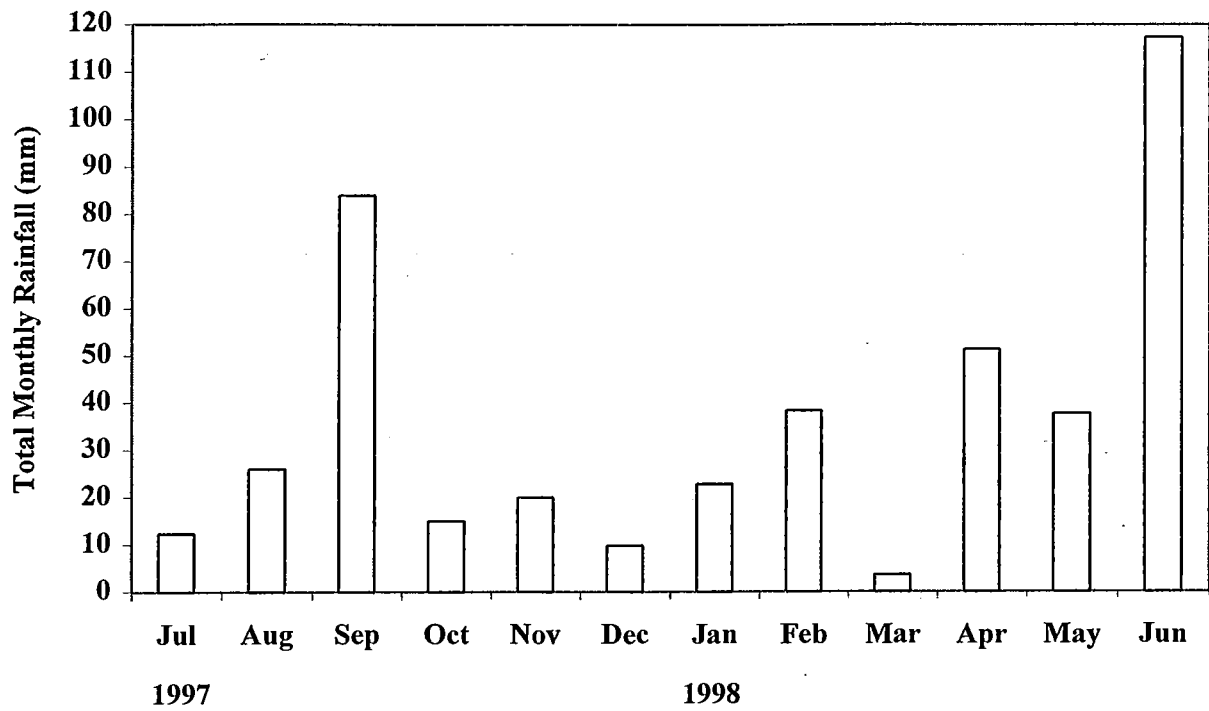


Figure 1.1 - Total monthly precipitation for Canberra Airport during the period July 1997 to June 1998

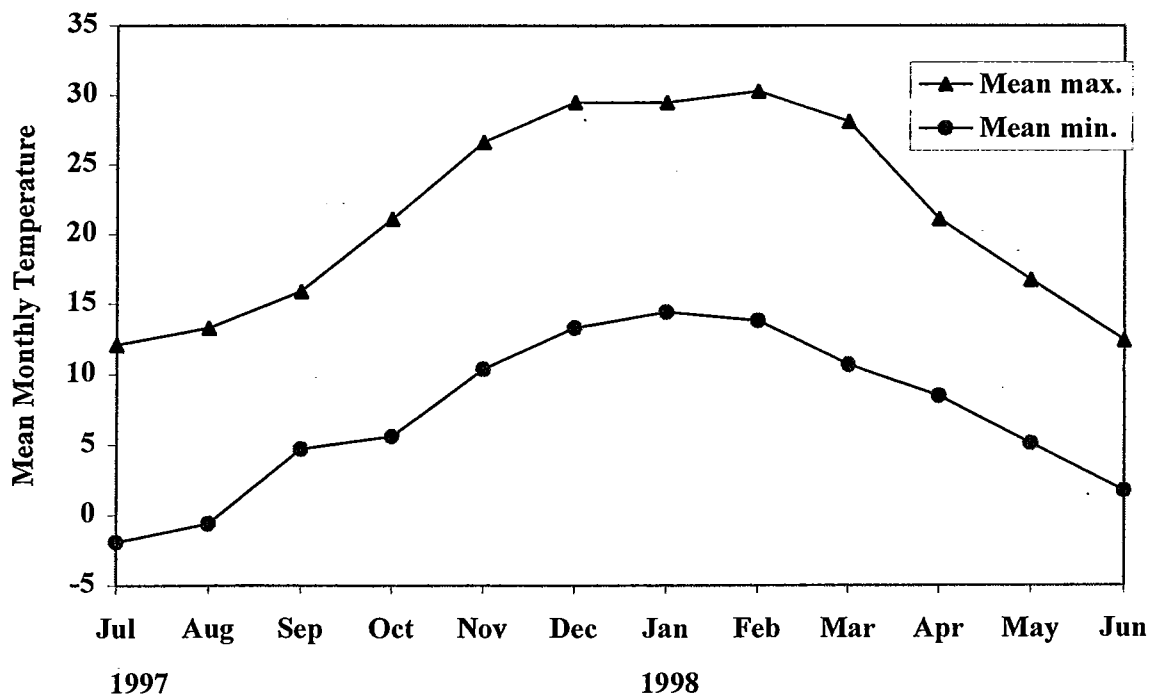


Figure 1.2 - Mean monthly maximum and minimum air temperature for Canberra Airport July 1997 to June 1998

direct disturbance of fauna and flora, collection of firewood, nutrient rich run-off from urban gardens and grazing (ACT Government 1996a).

Another important aspect of the urban design of Canberra is that a large number of exotic ornamental plants have been introduced to the city for landscaping (Mulvaney 1984). Over 40 percent of the vascular plants of the ACT are exotic and urban planners have deliberately planted at least 45 species belonging to the Genera *Pyracantha* and *Cotoneaster* (Mulvaney 1984). At least ten of these species of *Pyracantha* and *Cotoneaster* have since become naturalised in the bushland of Canberra (Mulvaney 1984). Moreover, studies have found that exotic vegetation in urban landscapes can have a profound influence on bird assemblages, generally favoring exotic birds (Green 1984).

### 1.3 Description and conservation status of temperate eucalypt woodland in Australia

The term woodland commonly refers to ecosystems that contain large, widely spaced trees with their crowns not touching (Yates and Hobbs 2000). Compared to forest trees, the crowns of woodland trees are usually more rounded and of greater depth, and vary in height from around 10 to 30 metres (ACT Government 1999). Temperate eucalypt woodland refers to these types of ecosystems located in a distinctive climatic and geographic zone, which are dominated by eucalypt species such as *Eucalyptus albens* (White Box), *Eucalyptus microcarpa* (Grey Box) and *Eucalyptus melliodora* (Yellow Box) (Robinson and Traill 1996).

Factors which determine the natural occurrence of eucalypt woodland in temperate Australia include annual rainfall of between 200-800mm, cooler temperatures, and nutrient poor and or drier soils in wetter areas (where woodland is essentially an extension of nearby forest areas) (Yates and Hobbs 1997). On the mainland of Australia these factors explain why most temperate eucalypt woodland is located between the semi-arid inland and the moister forests near the coast (Robinson and Traill 1996).

In structural terms, woodlands in Australia can be broadly classified as grassy woodlands or shrubby woodlands (Yates and Hobbs 2000). Grassy woodlands, in good condition, have a continuous species-rich ground layer of grasses and herbs, while shrubby woodlands have a patchy understorey of low trees and shrubs (Yates and Hobbs 1997). The extent and distribution

of these two broad structural types of temperate eucalypt woodland prior to the arrival of Europeans is not well known (Yates and Hobbs 1997). Factors influencing the floristics of temperate eucalypt woodland prior to the arrival of Europeans include climate, soil type, topography, hydrology, grazing, fire, and drought (Yates and Hobbs 1997).

In the last 200 or so years, over a million square kilometres of forests and woodlands have been cleared or thinned in Australia (Biodiversity Unit 1995). This clearing and disturbance was particularly severe in temperate woodlands (Robinson and Traill 1996). Clearing was also selective, with woodlands located on fertile soils suitable for agricultural production the main areas cleared (Yates and Hobbs 1997). This preference meant that often the most productive natural woodland areas were lost (Yates and Hobbs 1997).

As a result of this massive clearing, temperate eucalypt woodlands are the most threatened form of wooded ecosystem in Australia and around one quarter of birds living in this type of habitat appear to be in decline (Robinson and Traill 1996). Clearing for agriculture has been the main source of destruction of temperate woodland and it has been estimated that around 80 percent of woodland in eastern and south western Australia has been cleared for this form of land use (Robinson and Traill 1996).

Evidence of declines in biodiversity in temperate woodland has been documented by Robinson (1993), who noted a loss of woodland birds in Toolern Vale in Victoria. Saunders (1989) has also documented extinction and declines in woodland birds in the Wheatbelt of Western Australia. Sivertsen and Clarke (2000) documented regional declines in native mammals in temperate woodlands of New South Wales (NSW), while Paton *et al* (2000) documented regional extinction and decline of woodland birds in South Australia. Key reasons for these disturbing trends of declining biodiversity include direct loss of habitat and degradation of the ecological integrity of the remaining woodlands (Barret *et al.* 1994; Robinson and Traill 1996).

Some of the processes identified by Yates and Hobbs (1997) as contributing to ecological degradation and loss of biodiversity in temperate woodland include: continued clearing; habitat fragmentation; livestock grazing; nutrient enrichment; soil disturbance; altered hydrology and secondary salinity; invasion by exotic species; and damage by agricultural chemicals. Another important threatening process is the harvesting of timber for firewood. Robinson and Traill



(1996) noted that firewood production in Australia consumes around 6.1 million tonnes of wood a year. They also noted that in NSW and Victoria around 60 percent of the wood collected for firewood is obtained from temperate woodland and box-ironbark forests. Harvesting firewood has adverse impacts on temperate eucalypt woodlands by reducing the structural diversity of habitat, such as the number of hollows available for nesting birds (Robinson and Traill 1996).

#### 1.4 Description and conservation status of *E. melliodora*-*E. blakelyi* woodlands in the ACT

*E. melliodora*-*E. blakelyi* woodland is an open eucalypt woodland community in which *E. melliodora* is dominant or co-dominant with *E. blakelyi* (ACT Government 1999). A frequent associate is Apple Box (*Eucalyptus bridgesiana*) (ACT Government 1999).

In May 1997, the ACT Government scheduled *E. melliodora*-*E. blakelyi* woodland (described as Yellow Box-Red Gum Grassy Woodland in the action plan) as an endangered ecological community under section 21 of the *Nature Conservation Act 1980* (ACT Government 1999). This statutory listing reflects the fact that most of the woodland in the ACT has been cleared to make way for urban development or agriculture (ACT Government 1999). For instance, it has been estimated that prior to settlement Yellow Box-Red Gum Grassy Woodland covered about 32 000 hectares of the lowland of the ACT (ACT Government 1999). Currently, under 8 000 hectares remains, which represents about a 75 percent reduction in the size of this ecological community (ACT Government 1999).

Under section 23 of the *Nature Conservation Act 1980* an action plan has been developed to protect the natural values of Yellow Box-Red Gum Grassy Woodland. Key threatening processes identified in the action plan include: clearing for urban development; urban services and infrastructure development; firewood collection; clearing for agricultural and pastoral land use; failure of tree regeneration; rural tree die-back; deliberate modification of ground cover (rural pasture improvement); grazing by stock (and other grazing animals); invasion by weeds and feral animals; and fire (ACT Government 1999).

It is important to note that the scope of the legislative response described above is limited to Yellow Box-Red Gum Grassy Woodland remnants that meet specific criteria listed in the action plan (ACT Government 1999). Woodland that is considered degraded by destruction of the

original native understorey and replacement with exotic pasture species and woody weeds has largely been excluded from the scope of the plan (ACT Government 1999). The plan notes that excluded woodland remnants cover around 10 000 hectares (ACT Government 1999).

*E. melliodora-E. blakelyi* woodland provides habitat for around 50 species of birds and provides nest hollows and nest sites for a number of these birds (Taylor and Canberra Ornithologists Group 1992). Birds which use *E. melliodora-E. blakelyi* woodland in the ACT for at least part of their life cycle, and which have also been scheduled under section 21 of the *Nature Conservation Act 1980* by the ACT Government include: the Hooded Robin (*Melanodryas cucullata*); Swift Parrot (*Lathamus discolor*); Superb Parrot (*Polytelis swainsonii*); Brown Treecreeper (*Climacteris picumnus*); Regent Honeyeater (*Xanthomyza phrygia*); and Painted Honeyeater (*Grantiella picta*) (ACT Government 1999). Other birds found in lowland or open woodland which have not been scheduled under ACT legislation but which appear to be at risk or in decline include: the Diamond Firetail (*Stagonopleura guttata*); the Jacky Winter (*Microeca fuscina*); Speckled Warbler (*Chthonicola sagittata*); and the Rufous Songlark (*Cincloramphus mathewsi*) (ACT Commissioner for the Environment 1995).

### 1.5 Rationale and research focus of this thesis

Significant tracts of *E. melliodora-E. blakelyi* woodlands in the ACT are reserved within Canberra Nature Park and adjoin the suburbs of Canberra. This proximity to urban development, and the large number of exotic ornamental plants introduced to the ACT, has facilitated invasion of these woodlands by exotic woody weeds (Mulvaney 1984). As noted earlier, two important and invasive groups of exotic woody weeds found in *E. melliodora-E. blakelyi* woodland in Canberra belong to the genera *Pyracantha* and *Cotoneaster* (Berry and Mulvaney 1995). These exotic shrubs have been widely used as ornamental plants in the suburbs of Canberra and have been spread into adjoining woodlands by fruit eating birds, such as the Pied Currawong, which consume the berries and regurgitate viable seeds from woodland perches (Mulvaney 1984). Other important exotic woody weeds of woodland in Canberra include Hawthorn (*Crataegus spp*), Briar Rose (*Rosa rubiginosa*) and Blackberry (*Rubus fruticosus*) (Berry and Mulvaney 1995).

The implications of exotic woody weed invasion for the ecological health of urban remnants of *E. melliodora-E. blakelyi* woodland in Canberra are largely assumed to be adverse by managers

of this land (ACT Government 1996a). This assumption underpins key ACT Government policy and planning documents, such as the *Canberra Nature Park- Draft Management Plan 1996* and the *ACT Weeds Strategy* (ACT Government 1996a; ACT Government 1996b). The presence of weeds in Yellow Box-Red Gum Grassy Woodland, as defined in the action plan, also negatively influenced woodland quality ratings used to determine which patches of woodland in the ACT would be conserved under the *Nature Conservation Act 1980* (ACT Government 1999).

However, Australia wide, most of the research into the impact of environmental weeds has been concerned with their effects on vegetation and has largely ignored the impacts on animals (Adair and Groves 1998). This lack of research into faunal impacts represents a significant gap in our understanding of environmental weeds and leaves assumptions of adverse impact on the faunal component of woodland communities open to question.

A key ecological relationship between birds and shrubs in woodlands is the addition of structural diversity provided by the presence of shrub cover (Freudenberger 1999). Shrub cover, in the context of my research, provides some birds with shelter from adverse weather conditions, nest sites, protection from predators or aggressive competitors, roosts and perches, and surfaces on which to forage.

A second key area of ecological interaction concerns frugivorous birds. These birds are important because exotic shrubs produce berries that may increase the seasonal abundance of these birds in weed invaded woodlands. Of particular interest are birds that consume fruit without destroying the seeds (Forde 1986), as these birds disperse viable seeds of exotic shrubs into woodland habitat (Mulvaney 1984). The most likely and immediate impact of this process on woodland remnants is that exotic shrubs become established in dense clumps under woodland perches and compete for resources with native vegetation (Mulvaney 1984). Other impacts may arise from increases in the abundance of frugivorous birds due to the availability of exotic fruit.

Adair and Groves (1998) have reviewed the impact of weeds on biodiversity in Australia and the methods used to investigate these impacts. They noted that beneficial impacts of weeds were determined infrequently and mostly occurred on degraded land (Adair and Groves 1998). This finding raises the possibility that in an urban context, where woodland is often degraded, exotic shrubs may be of some benefit to native birds. The urban context of *E. melliodora*-*E. blakelyi* is

also of experimental interest because the presence of exotic shrubs may influence the invasion of exotic bird populations into these areas from nearby housing and suburban streets.

In this study I have aimed to determine whether there is a relationship between exotic shrubs and the abundance and species richness of birds of urban woodland. In particular I have examined whether exotic shrubs in urban woodland affected:

- 1) bird species richness;
- 2) the abundance of individual bird species; and
- 3) the composition of bird assemblages.

Thus, this thesis focuses on birds in urban *E. melliodora*-*E. blakelyi* woodland that have their abundance influenced by exotic shrubs. The findings will then provide a basis for discussing the likely conservation implications.

## 1.6 Structure of thesis

In Chapter 2 of this thesis I provide a brief review of habitat selection in birds and outline a theoretical and practical framework for considering the study of avifauna in response to habitat variables. In Chapter 3 the design of the study and the methods used are presented in detail, to ensure that the strengths and weaknesses of the approach used are transparent. In Chapter 4 results from statistical tests, models and field observations are presented (taxonomy for birds follows Pizzey and Knight (1997)). Chapter 5 provides a discussion of the results and is divided into five parts: 1) birds that use exotic shrubs in a structural context (e.g. shelter or protection); 2) exotic shrubs as a food resource for birds; 3) exotic shrubs as a threat to ecosystem function; 4) the implications of my findings for managers of urban woodland in Canberra; and 5) conclusions.

## Chapter 2

### Review of literature

#### 2.1 Introduction

This Chapter reviews some of the theory and practice relevant to the study of birds in eucalypt and weed invaded habitat. The purpose of this review is to show how this study relates to other research and to provide a theoretical framework in which to discuss the results in later chapters.

Eight broad environmental features that influence bird assemblages in eucalypt forest and woodland are reviewed in this chapter. These are: vegetation structure and floristic diversity; food availability; competition; predation; climate and seasonality; patterns of diurnal bird behaviour; nesting and reproduction; and human impacts, with particular emphasis on invasive plants. The chapter also includes a review of selected weed studies in Australia.

#### 2.2 The influence of vegetation structure and floristic diversity on bird assemblages

A number of studies of bird diversity have concluded that, generally, habitat that is more structurally complex will contain a higher number of bird species (Cody 1985; Wiens 1992a; Perrins and Birkhead 1983). Small birds, in particular, are thought to distinguish between habitat on the basis of structural characteristics (Cody 1985). Factors that contribute to structural complexity in temperate woodland in Australia include abundant groundcover, ground litter, rocks and logs, and a mixture of shrubs of different heights (Freudenberger 1999). In some types of habitat, however, the number and composition of different plant species (floristic diversity) may be more or as important to birds as the structure of the vegetation (Milledge and Recher 1985; Rotenberry 1985). These determinants of bird abundance and species richness are important to my study because invasive exotic shrubs can reduce floristic diversity (Loyn and French 1991; French and Zubovich 1997; Adair and Groves 1998) and alter structural complexity (French and Zubovich 1997). In *E. melliodora*-*E. blakelyi* woodland in the ACT, changes in the composition of bird assemblages are thought to be linked to the loss of a shrubby understorey following habitat simplification caused by grazing (Landsberg 2000).

Gilmore (1985) investigated the influence of vegetation structure on the density of insectivorous birds (including honeyeaters) in woodland plots in Victoria and Queensland. He suggested that most of the plant species present in woodland plots surveyed were relevant to insectivorous birds only through their proportional contribution to plant biomass. His findings suggested that bird assemblages in woodland are strongly influenced by the architecture or structure of the vegetation. This view is supported by Mac Nally (1990), who determined that structural features of forest and woodland habitat in Australia influences the density of birds.

Loyn (1985) studied the ecology, distribution and density of birds in Victorian forests. One of his findings was that the absence or presence of a shrub layer had a profound influence on the composition and abundance of bird assemblages. He found that shrubby areas had more birds that characteristically foraged in the understorey (e.g. the Golden Whistler, *Pachycephala pectoralis*), while forests with a sparse shrub layer resulted in these birds being absent or rare in the area. This work, and the findings of Gilmore (1985), supports the notion that a shrub layer has an important influence on the abundance and composition of bird assemblages in forest and woodland habitats.

Milledge and Recher (1985) compared forest bird communities from the south and mid-north coast of NSW. They found that the structural diversity of the vegetation in forest habitats was not the principal determinant of bird species diversity or of bird abundance in these forests. They considered that floristic diversity was also a significant factor in terms of increasing bird species diversity in the eucalypt forests sampled. This view was also supported by Shields *et al.* (1985) who concluded, from their work in forest of the Upper Hastings River, that both structural and floristic parameters influence birds and their relationship with habitat. They also determined that the amount of open habitat was the most important structural difference between forest types in terms of observed differences in bird species diversity.

Recher and Holmes (1985) investigated seasonal patterns of foraging ecology in forests on the Southern Tablelands. They found that plots of forest where ground, shrub and subcanopy vegetation was denser and more diverse influenced the abundance and species composition of bird assemblages. Species they found to be more abundant in forests with a denser and more diverse ground, shrub and sub-canopy layer included the Brown Thornbill (*Acanthiza pusilla*), the Golden Whistler (*Pachycephala pectoralis*) and the White-browed Scrubwren (*Sericornis*

*frontalis*). They also classified birds observed in their forest sites based on their foraging height distribution. Species that they determined as mainly foraging in the shrub layer (from 0.2m to 4m) were the: Silvereye (*Zosterops lateralis*); Red Wattlebird (*Anthochaera carunculata*); Golden Whistler (*Pachycephala pectoralis*); Rose Robin (*Petroica rosea*); Crescent Honeyeater (*Phylidonyris pyrrhoptera*); Brown Thornbill (*Acanthiza pusilla*); Eastern Spinebill (*Acanthorhynchus tenuirostris*); Rufous Fantail (*Rhipidura rufifrons*); and Eastern Whipbird (*Psophodes olivaceus*). Of these eight species the two most conspicuous and abundant were the Golden Whistler and the Brown Thornbill (Recher and Holmes 1985).

Ford (1985) suggested that the main features of forests that influence birds are the structure and composition of the vegetation, which provides refuges, foraging and breeding sites and food. He also noted that the structure of the vegetation determines whether safe refuges from predators and nesting sites are available. In forest and woodland of southeastern Australia, Recher *et al.* (1985) noted that changes to habitat from human activities will affect birds differently, because of differences in the way they use resources, including their response to habitat structure.

Smith (1985) examined the impact of logging on birds near Bega and determined that the Silvereye (*Zosterops lateralis*) was more common in areas of regeneration, which corresponded with a particularly dense and extensive layer of shrubs and saplings. The importance of a shrub layer in rehabilitating forest denuded by mining activities was determined by Collins *et al.* (1985) who studied re-colonisation of restored bauxite mine sites by birds in southwestern forests. They found that in the short to medium term (five to six years), rehabilitated mine sites which developed dense shrub cover attracted more native birds than sites without this shrub cover. The importance of shrub cover was also demonstrated by extensive revegetation of agricultural land in South Australia in the 1970s, which created around 200 ha of woodland near Monarto South (Paton *et al.* 2000). This 'new' woodland, supports a similar number of native birds to nearby natural woodland, but because it lacks a shrubby understorey, supports few ground and shrub dwelling birds, such as the Southern Scrub-Robin (*Drymoides brunneopygia*) or Fairy-wrens (*Malurus spp*) (Paton *et al.* 2000).

A number of studies have determined that small Australian birds, such as wrens, thornbills and finches, shelter and nest in dense shrubbery and that the presence of dense shrubs can influence abundance, distribution and social behaviour (Bell 1985; Nias 1984; Nias and Ford 1992; Recher

and Holmes 1985; Schodde and Tidemann 1988; Wilson 1994). These findings identify wrens, thornbills and finches as particularly dependent on the structural role played by shrub cover in eucalypt habitat.

### 2.3 The influence of food on the distribution and abundance of birds

Birds exploit a wide range of food types and must decide efficiently where to feed and which food items to select, in order to survive and reproduce successfully (Perrins and Birkhead 1983). The need for efficiency when foraging is based on the premise that food may be in short supply and that even when plentiful, efficient foraging leaves more time for other essential behaviour, such as avoiding predators (Perrins and Birkhead 1983).

Studies of the foraging patterns of birds in eucalypt forest and woodland in northeastern and southeastern Australia show that birds exploited food resources in different ways, that some were generalists and some were specialists, and that some were very similar in their habits and requirements (Ford *et al.*, 1985; Recher *et al.*, 1985). Ford *et al.* (1985) also suggested that resources, such as food need not limit bird populations. Birds consumed insects, seeds, fruit and nectar more frequently than other food items (Ford 1989). Ford (1989) also suggested that it is rarely possible to link the distribution of bird species in Australia to one or a few major types of food and that it is often difficult to separate the influence of food from the influence of habitat.

Ford (1989) also discussed the idea of optimal foraging as it relates to bird distribution in Australia. The theory of optimal foraging is based on the idea that some foods are more profitable than others and that feeding behaviours have costs and benefits to a bird (Perrins and Birkhead 1983; Ford 1989). The word profitable in this context refers to maximising energy intake per unit of time (Perrins and Birkhead 1983). Characteristics that make food sources more profitable, and therefore preferred, include being larger, richer in energy, or easier to obtain (Ford 1989). This aspect of optimal foraging (i.e. the quantity, quality and availability of food) is relevant to this study, in that exotic shrubs provide food for native and exotic birds in the form of fruit (Mulvaney 1984) and serve as foraging substrates (Loyn and French 1991). Exotic plants also displace food resources provided by native vegetation (Loyn and French 1991).



Mulvaney (1984) has summarised the appeal of the berries of Firethorn shrubs (*Pyracantha spp*) and Cotoneaster shrubs (*Cotoneaster spp*) to birds as their bulk per plant; their brightly coloured fleshy nature; their physical distribution and spacing on the branch; and their small size (about 6mm in diameter). He also determined that the nutritional value of the fruit is fairly low. These characteristics suggest that in terms of optimal foraging and food characteristics, exotic berries in Canberra woodland are a plentiful and easy to handle food (Mulvaney 1984), but not particularly rich in energy or nutrients.

Forde (1986) studied the relationship between birds and fruit in temperate Australia. He noted that 87 species of Australian birds have been recorded consuming fruit or arils in temperate southern Australia, including 26 honeyeaters. He concluded that only a small number of these species are specialist frugivores, with fruit forming only part of the diet for the majority of species observed. He also discussed seasonal influences, noting that fruit production in southern Australia peaks over spring-summer and summer-autumn and suggested that a lack of fruit in winter would force some birds to switch to other resources and or leave areas when fruit is no longer available. Other important relationships considered by Forde (1986) are the dispersal of seed by frugivorous birds and the influence that the characteristics of the fruit have to facilitate dispersal (e.g. colour, size, hardness of seed and nutritional value).

The characteristics of food sources, such as fruit, represent only one dimension of optimal foraging behavior. Another dimension is access to profitable food sources, which incorporates the presence of predators and or aggressive competitors and the energetic costs of vigilance and risk of injury or death (Ford 1989; Newton 1998). One way exotic shrubs may influence the efficiency and safety of foraging for food is by providing cover and places to watch for danger.

Insects are a very important part of the diet of a large proportion of Australian landbirds (Ford 1989; Barker and Vestjens 1989; Barker and Vestjens 1990). However, little is known about the role of exotic shrubs in terms of providing foraging habitat for insectivorous birds. Griffin *et al.* (1989) asserted that most introduced plants lack herbivorous insects in their foliage and suggested that displacement of native vegetation by the Tamarisk Tree (*Tamarix aphylla*) reduced insect prey for insectivorous birds. Mulvaney (1984) noted that few invertebrates feed, shelter or breed in *Pyracantha spp* and *Cotoneaster spp* shrubs, possibly because they produce toxic cyanogenic glycosides that prevent insect populations from thriving. Ekert and Bucher

(1999) found reduced insectivory in forest sites invaded by Large-leaved Privet (*Ligustrum lucidum*), although their study period did not coincide with the flowering of this species. These findings suggest that exotic shrubs may be a poor source of invertebrate prey for birds, with implications for the abundance of insectivorous birds. French and Zubovich (1997) found, however, that in some months insectivorous birds were more abundant in woodland invaded by Bitou Bush (*Chrysanthemoides monilifera*), suggesting that insect availability can be high in weed infested habitat.

The diet of birds and the availability of food in eucalypt habitat is influenced by seasonal variability (Recher 1985), with periods of food scarcity generally resulting in birds taking a wider variety of foods (Ford 1989). In general terms food in southern Australia is scarcer for birds in winter and peaks in spring and summer (Nix 1976; Recher and Holmes 1985; Ford 1989).

Dense shrubs provide nesting sites and shelter for small birds (Ford 1985). Small birds also represent an important food source for predatory birds, such as raptors (Olsen 1995). Given this relationship, it is possible that the distribution of raptors and other avian hunters of small birds are influenced by exotic shrubs in eucalypt habitat. French and Zubovich (1997) provided some evidence for this effect of exotic shrub invasion in their study of woodland invaded by Bitou Bush (*Chrysanthemoides monilifera*). They found that raptors and scavengers were lower in abundance and species richness in Bitou Bush infested woodland and offered as a possible explanation the availability of dense cover for prey. They implied that exotic shrubs might reduce hunting success, with dense shrubbery limiting the probability that small prey will be caught in the open. Alternatively, the presence of dense shrubbery may ensure a good supply of small birds to eat.

Environmental weeds such as African Boxthorn (*Lycium ferocissimum*) and Briar Rose (*Rosa rubiginosa*) are known to harbour pest animals, such as rabbits (Peter 2000). This may affect food supplies for birds because rabbits modify ground cover vegetation by grazing (Standing Committee on Conservation, Heritage and Environment 1994). An abundance of rabbits is also known to influence the abundance of predatory birds, such as the Australian Goshawk (Olsen 1995). Thus, woodland invaded by exotic shrubs may be more favourable to rabbits, which in turn may influence the availability of food for birds by changing understory vegetation and or providing prey for raptors.

Loyn and French (1991) suggested that native birds may use exotic plants as a source of food but that this food may be inferior to the food provided by displaced native vegetation. Griffin *et al.* (1989) determined that the trophic structure of bird assemblages were changed where the Tamarisk Tree (*Tamarix aphylla*) displaced native trees along a river system in the Northern Territory. Thus, exotic shrubs provide food for birds but also remove food resources by displacing indigenous vegetation.

In conclusion, the availability of food is fundamental to the distribution of birds. From the preceding discussion a number of key points emerge in relation to the way that exotic shrubs affect food, and hence bird distribution: 1) by providing fruit at different times of the year; 2) as foraging habitat for insectivorous birds; 3) by indirectly increasing access to food by providing sanctuary from aggressive competitors or predators; 4) by influencing the number and availability of small birds as a food source; 5) by increasing rabbit abundance; 6) by influencing the hunting efficiency of predators; and 7) by displacing native vegetation that provides food resources, such as insects.

#### **2.4 Competition between birds and the effect of exotic shrubs on resources**

Bird species which are similar in aspects of their behaviour and diet may compete for resources, while individuals of the same species will almost certainly do so, if a shared resource becomes limited (Perrins and Birkhead 1983). Begon *et al.* 1996 defined competition as: “an interaction between individuals, brought about by a shared requirement for a resource in limited supply, and leading to a reduction in the survivorship, growth and or reproduction of at least some of the competing individuals concerned”. They also identified two types of competition: intraspecific competition, which relates to individuals of the same species and interspecific competition, that is, competition between species (Begon *et al.* 1996).

Ford (1989) defined competition in relation to birds as “use of a common resource that is in short supply, or restriction of access to a resource, even if not in short supply”. Competition between birds is relevant to this study because exotic shrubs change the resources present in woodland habitat.

Interspecific competition may occur through interference, whereby birds restrict access to a resource by territoriality and other forms of aggression (Ford 1989; Wiens 1992a). The second major pathway of interspecific competition occurs through exploitation, whereby access to an important resource is lost because it is depleted by another species without direct interaction between the competing birds (Ford 1989; Wiens 1992a). An example of interference competition is provided by larger honeyeaters, such as wattlebirds, in situations when they defend aggressively nectar sources from smaller honeyeaters (Ford 1989). An example of exploitation competition is smaller honeyeaters opportunistically feeding on rich, but highly variable, nectar sources scattered over greater distances (Ford 1989).

A key ecological condition required for interspecific competition to occur is that of limited resources (Perrins and Birkhead 1983; Ford 1989; Newton 1998). In Australia, Ford (1989) suggested that resource limitation is more likely to affect the abundance of coexisting species, rather than their ability to coexist at all. Further, he hypothesised that critical periods of resource limitation are likely to be cyclical, with ecological “crunch” times increasing the intensity of competition between bird species.

The theoretical importance of resource limitation in terms of interspecific competition is relevant in the context of woodland invaded by exotic shrubs because exotic shrubs provide resources such as fruit, nest sites and places to shelter. Exotic shrubs also decrease the availability of resources by out competing and displacing native vegetation. The influence this may have on bird assemblages is not immediately obvious. Specific impacts will depend on the foraging ecology of individual species and their use of resources in weed invaded habitat.

The increased availability of fruit provides a good example of resource change in weed invaded eucalypt forest or woodland. Under normal circumstances fruit is not a particularly important food source in these ecosystems (Ford 1985). However, there is considerable evidence to suggest that the Pied Currawong population in Canberra has increased substantially in size because berries provided by exotic shrubs have increased the food available to this species and reduced winter mortality (Taylor and Canberra Ornithologists Group 1992). Underlying this suggestion is the premise that prior to the establishment of an abundant supply of exotic fruit, food was an important limiting factor to the Pied Currawong population in Canberra. With food now less limiting, the related increase in the Pied Currawong population may have implications for other

bird species that compete for resources with Pied Currawongs. For instance, Gang-gang Cockatoos also feed on exotic berries in autumn and winter (Forshaw and Cooper 1991) and arrive in Canberra around the same time as Pied Currawongs (Frith 1976). Similarly, other birds that can make use of exotic shrubs may no longer be limited in ways that they once were, suggesting that interspecific competition may be altered by changes in resource availability provided by exotic shrubs. On the other hand, birds that have lost resources because exotic shrubs have displaced native vegetation important to their needs may face increased competition, as weed invasion limits the availability of resources. An example of a 'lost' resource in a woodland densely invaded by exotic shrubs is open ground in which to forage.

There is a strong body of evidence to indicate that exotic plants in native habitat have adverse effects on native vegetation, displacing indigenous species through competition for resources, such as light, space and nutrients (Adair and Groves 1998). It follows from this that fauna, such as birds, may also be affected adversely, particularly if the native component of the vegetation that is displaced by weeds was providing a resource necessary for that element of the fauna to persist or if ecosystem processes are under threat. Such adverse impacts on fauna have been demonstrated by a number of studies in Australia (Adair and Groves 1998). For example, the continued spread of *Mimosa pigra* may endanger the habitat of the Magpie Goose (*Anseranas semipalmata*) by replacing dense stands of native sedges needed for food and nesting (Groves 1991).

In this thesis I argue that the links between adverse impacts on native vegetation and adverse impacts on fauna are far from simple. Moreover, given the finding that exotic shrubs had differential impacts on bird species of coastal woodland (French and Zubovic 1997), it follows that removing exotic shrubs may have differential impacts on competition within and between bird species. For example, Dow (1977) demonstrated that a Noisy Miner (*Manorina melanocephala*) colony in open woodland successfully excluded nearly all other birds from their territory through an aggressive colonial defence system. However, in structurally more complex woodland, Noisy Miners were scarcer and less able to eliminate other birds (Dow 1977). Thus, in the case of the Noisy Miner, the presence or absence of a shrub layer appeared to influence the effectiveness of aggressive interference.

Competition within a species (intraspecific competition) also influences birds by placing additional limits on the capacity of habitat to sustain bird populations. Reduced intraspecific competition is one way that bird populations recover from mortality, with losses compensated for by the additional 'space' and resources made available to the survivors (Perrins and Birkhead 1983; Newton 1998). The addition of resources in woodland invaded by exotic shrubs, such as fruit, may also influence the effects of intraspecific competition on birds that can utilise this new resource and may influence mortality. The expansion of the Pied Currawong population in Canberra is quite plausibly an example of this effect, with the addition of fruit perhaps reducing competition between Pied Currawongs for food and reducing juvenile mortality in Pied Currawongs (Major *et al.* 1996).

## 2.5 The influence of predation on bird assemblages

Birds are eaten by a wide range of animals such as other birds, mammals and reptiles (Newton 1998). Mammalian predators important in Canberra woodland include cats and foxes (ACT Government 1996a). Typical avian predators of woodland in the ACT include raptors such as the Collared Sparrowhawk (*Accipiter cirrhocephalus*), Brown Goshawk (*Accipiter fasciatus*) and Australian Hobby (*Falco longipennis*), and non raptor species such as the Pied Currawong (*Strepera graculina*) (Taylor and Canberra Ornithologists Group 1992). These predators are typical of many bird predators, in that they can be classified as generalists, and are equipped to take a wide range of prey species in response to need and opportunity (Newton 1998).

Bird populations may be limited or regulated by direct predation, rather than through shortages in food and other ecological resources (Wiens 1992b; Newton 1998). For instance, in some passerines, 70-80 percent of all breeding attempts fail because of nest predation (Cody 1985). Predation in birds is seldom distributed evenly and may be concentrated on vulnerable components of bird populations (e.g. young and inexperienced birds) or on particular social classes (e.g. non-territorial birds in a territorial species) (Newton 1998). One effect of this selectivity in the taking of prey is that if non-breeding birds are targeted, the prey population may be better placed to recover any losses from predation through reproduction (Newton 1998). Increased nest predation, however, does not always reduce the population density of the prey species (Wiens 1992b), suggesting that predation and its influence on the population dynamics of prey is complex.

Predators may limit the numbers of their prey in a density-dependent manner (Begon *et al.* 1996; Newton 1998). A density-dependent association occurs where the proportion of prey that was killed increases with rising prey density (Begon *et al.* 1996; Newton 1998). This type of predator-prey relationship is driven by two main predator responses: a functional response, where individual predators kill more prey per unit of time as prey numbers increase; and a numerical response, where more predators arrive and increase in number in response to an increase in the abundance of prey (Begon *et al.* 1996; Olsen 1995; Newton 1998). These theories support the notion that an abundance of small birds in an area of habitat could influence the abundance of predators in that area seeking these small birds as prey.

Predator-prey interactions may also be independent of density (Begon *et al.* 1996; Newton 1998). For example, other habitat variables could limit the number of predators in an area (e.g. lack of nest sites), rather than the numbers of a particular species of prey (Newton 1998). Density independent predation can also occur when predators, whose numbers were determined elsewhere, suddenly move into an area (Newton 1998).

Density independent predator prey relationships are an important theoretical consideration of this thesis. This is because there is considerable evidence to suggest that the Pied Currawong population in Canberra is being sustained by the presence of exotic fruit in the suburbs and in weed invaded bushland (Taylor and Canberra Ornithologists Group 1992). As Pied Currawongs are mobile, highly effective predators of small birds (Wood 1998; Taylor and Canberra Ornithologists Group 1992; Major *et al.* 1996), an increase in their numbers because of fruit may represent the development of a density independent relationship being established in regard to birds of urban woodland in the ACT. In urban environments, the role of the Pied Currawong as a predator of small birds may be particularly significant, as this species is a dominant nest predator in urban areas (Major *et al.* 1996).

In theoretical terms, mortality from predation will have a significant impact on bird populations when the losses from predation are not compensated by reduced mortality from other causes (i.e. the effects of predation are additive) (Newton 1998). Compensatory effects on bird mortality following losses from predation, may include reduced intraspecific competition for nest sites and food, a reduced incidence of death from disease, and or the replacement of the birds killed by predation through increased breeding success (Perrins and Birkhead 1983; Newton 1998).

A key dimension of compensatory mortality and the impacts of predation on bird assemblages, is the importance of the breeding component of the prey population. Predators that target and kill breeding adults, on balance, are more likely to have a significant impact on prey populations than are predators that target the non-breeding component (Newton 1998; Olsen 1995). This is because the breeding component of a prey population provides the replacements for mortality, whatever the combined causes.

Rates of predation will vary with factors such as habitat, climate, the availability of refuges for prey and the diversity of prey (Wiens 1992b). Environmental factors that predispose birds to predation, such as the quality of protective cover, are thought to be as important to the study of predation as the agents of mortality themselves (e.g. predators, disease and starvation) (Newton 1998). For example, studies in the northern hemisphere have shown that density-dependent predator-prey relationships can be modified by dense nesting cover, with predation rates increasing more slowly in habitat with relatively more shrub cover (Newton 1998).

Ford (1989) considered that a major requirement of breeding success in Australian birds is a high quality territory, with safe breeding sites and refuge from predators. Thus, the level of nesting cover in temperate eucalypt woodland could be expected to have a profound influence on the rate of predation on breeding birds, and hence, the overall impact that predation has on bird populations in this habitat. This hypothesis is supported by the work of Nias and Ford (1992) who found that breeding success in the Superb Fairy-wren was correlated with larger areas of brambles (*Rubus* spp), which are used by the Superb Fairy-wren for nest sites and refuges (Nias 1984). Wrens living in territories with larger areas of brambles made more nest attempts and fledged more off-spring than did birds with smaller areas of brambles (Nias and Ford 1992). Nias and Ford (1992) concluded that increased breeding success by the Superb Fairy-wren was due to increased protection from nest predators.

Generalist predators of birds select a range of prey based on need and opportunity, with prey vulnerability an important component of this selection process (Newton 1998). Features of birds that may predispose their population to significant reductions in abundance through predation include: body size relative to a predator; the accessibility of nest sites; and the efficiency of behavioural defense strategies (Newton 1998). Ford (1989) suggested that in Australia, birds that forage on the ground might be more susceptible to predation. He also suggested that there is



safety in numbers and that solitary foragers in exposed habitat may be particularly vulnerable to predators.

Robinson (1993) noted that woodland birds that foraged on the ground for at least some of the time, represented 58 percent of woodland birds declining in southern Australia and suggested there may be common causes (although he did not specify predation as one of these). Robinson also noted that approximately 30 percent of woodland birds declining in abundance in southern Australia nested on the ground. This pattern of decline in ground foraging and nesting woodland birds is particularly noticeable in agricultural landscapes (Reid 1999; Barrett and Davidson 2000). A plausible hypothesis for these common patterns of decline is that a simplified understorey in woodland habitat deprives ground foraging birds of key resources, such as safe nest sites and refuge from predators.

Ford (1989) noted that dense shrubbery is an important component of territory quality for small birds and provides safe breeding sites and refuges from predators. He also discussed the prevalence of co-operative breeding strategies in Australia (around one third of Australian landbirds). He also suggested that co-operative breeding may in part be explained by the risk of predation to helpers, should they opt to disperse from the breeding territory into less optimal or poorer quality areas (e.g. areas that lack protective shrub cover). Thus, the abundance of small birds in woodland with dense shrubbery present may be indirectly related to the risk of predation outside these areas, with small birds concentrating in shrubby high quality territories because relatively safe habitat is a limited resource. This hypothesis is based on the assumption that the best chance for helpers to pass on their genes may be to wait for a breeding opportunity in a high quality territory, rather than risk death by predation which may result from dispersing to more open, and hence, more hazardous habitat (Ford 1989).

Alternative explanations for the prevalence of co-operative breeding in Australia include the highly variable climate of Australia and the lack of food (Rowley 1976; Thomas 1974). In contrast, Dow (1980) argued that predictable environments are more conducive to co-operative breeding strategies. He considered it likely that birds in Australia opt for communal breeding for a variety of reasons specific to the needs of their species, and perhaps varying with circumstance.

A final potential influence on predator-prey relationships is the impact the social structure of small birds may have on predator efficiency. Social behaviors such as the presence of helpers and mixed feeding flocks are thought to be related to the threat of predation, based on the general principle of safety in numbers (Ford 1989), although Hermes (1981) was unable to determine that mixed feeding flocks in the ACT influenced predation. Hence, if the availability of a suitable shrub layer increases the carrying capacity (Perrins and Birkhead 1983) of eucalypt habitat in terms of the abundance of small birds, it is possible that the social network established to thwart predation would be enhanced in habitat with plenty of shrub cover.

In conclusion, the influence of predation on birds is likely to be selective and dependent on prey vulnerability. Small size and foraging on the ground are two factors that may predispose some birds to predation (Ford 1989; Wiens 1992b; Newton 1998). The vulnerability of prey is also related to habitat, with the density and protective qualities of cover likely to influence predator-prey relationships (Nias and Ford 1992; Newton 1998). Predation that targets breeding adults is of particular importance, in the context of habitat quality, with the number of safe breeding sites likely to influence the impact of predation on bird populations (Newton 1998). In urban environments, increases in the abundance of Pied Currawongs may be particularly significant to the diversity of bird assemblages, as this species is a dominant nest predator in urban environments, and hence may lower recruitment in vulnerable prey species (Major *et al.* 1996).

## 2.6 The influence of climate and the seasons

The ACT has a continental climate, with warm to hot summers and cold winters, and lies at a latitude of 35 degrees and 36 degrees south of the equator (ABS 1999). The other principal determinants of the climate in the ACT are its elevation (ranging from 450m to 1910m above sea level) and its distance of 130km from the sea (ACT Commissioner for the Environment 1995).

The average annual rainfall in the ACT is around 634mm (ACT Commissioner for the Environment 1995). Maximum temperatures in summer are from 25 to 30 degrees celsius and from 12 to 15 degrees celsius in winter, while average winter minimums are close to 0 degrees celsius (Canberra Ornithologists Group 2000). The cloud free skies prevalent in winter result in some very cold nights, with around a 100 frosts recorded in Canberra per year (ACT Commissioner for the Environment 1995). *E. melliodora-E. blakelyi* woodland is

characteristically located in the warmer lowlands of the ACT, below about 700m, where frosts are relatively mild (Taylor and Canberra Ornithologists Group 1992).

Climatic extremes can have a significant influence on bird populations by increasing mortality during long periods of regional severity (such as drought), or through sudden episodic events, which kill many birds directly, such as severe storms (Newton 1998). More generally, climate and seasonal patterns in southern Australia influence bird populations by regulating the availability of key resources, such as food, while cold temperatures affect the physiological demands of birds by increasing the energy needed to stay warm (Nix 1976; Ford 1989; Newton 1998). Changing seasons may also provide environmental cues for the timing of breeding, with increasing day length, for example, thought to trigger the reproductive cycle of some birds (Ford 1989).

Seasonal patterns of food availability vary with the type of food and regional conditions (Nix 1976; Recher and Holmes 1985). Insect populations in eucalypt habitat in southern Australia are thought to peak in spring or summer and are at their most scarce in winter (Woinarski and Cullen 1984; Bell 1985; Recher and Holmes 1985; Ford 1989). Nectar is generally more abundant in winter and spring, while seeds and fruit are often more abundant in summer and autumn (Ford 1989). Nix (1976) hypothesised that the timing of breeding in the ACT is determined by increases in vegetative growth in the warmer seasons (summer and spring), as this also coincides with an increase in food resources.

In the ACT, the mix of resident and migrant bird species changes with climatic variation associated with season (Frith 1976). For example, in autumn and winter bird movements involve regional shifts from higher to lower altitudes, with many species overwintering in the milder conditions of Canberra (Frith 1976; Taylor and Canberra Ornithologists Group 1992). Based on their review of seasonal information concerning birds in the ACT, Er and Tidemann (1996) noted that low autumn temperatures in the ranges south-west of the lowland of the ACT stimulate altitudinal and north-south and east-west migration of several bird species. This results in a temporary influx of birds into the lowland of the ACT as birds pass through en route to their wintering grounds (Er and Tidemann 1996). Rising temperatures in spring increase the growth of vegetation and hence the abundance of food increases, attracting summer migrants (Er and Tidemann 1996).

Er and Tidemann (1996) undertook their own fieldwork and found that seasonal patterns of movement and migration have a profound influence on the composition and relative abundance of birds in *E. melliodora-E. blakelyi* woodland in the ACT. They inferred from seasonal data that *E. melliodora-E. blakelyi* woodland remnants help to maintain bird species diversity by providing: 1) bird refuges within the lowland landscape matrix of the ACT; 2) wintering sites essential for partial migrants and some resident species; 3) breeding sites for residents and summer migrants; and 4) refuges for nomadic species during periods of extreme drought. They also concluded that birds respond differently to their habitat at different times of the year and suggested that management decisions affecting birds should account for these seasonal variations. These findings suggest that the bird assemblages in the ACT are strongly influenced by seasonal changes.

Exotic shrubs in Canberra provide a significant fruit resource for birds in autumn and winter, a period when food is normally scarce (Mulvaney 1984). Milledge and Recher (1985) found that fruit increased species richness and bird abundance in eucalypt forest. Based on these findings, the presence of fruit laden shrubs in Canberra woodland may influence the seasonal composition of bird assemblages, by attracting fruit eating birds to these areas in winter and autumn. More generally, climatic variation is likely to influence fruit production.

Spring and early summer are the main breeding seasons for birds in southern Australia (Nix 1976; Ford 1989). This is also true of Canberra, although for some species the breeding season may extend into autumn, and some breeding activity has been recorded in late winter and autumn (Frith 1969; Taylor and Canberra Ornithologists Group 1992). Seasonal breeding activity will influence the abundance of bird populations by increasing the number of young birds recorded in an area at a particular time of the year. Breeding birds also tend to concentrate in areas with suitable nesting sites (Newton 1998). As exotic shrubs, such as African Boxthorn (*Lycium ferocissimum*), provide nest sites for native birds (Peter 2000), woodland invaded by exotic shrubs may show seasonal peaks in abundance that are related to breeding activity in these areas. French and Zubovich (1997) noted that birds were more abundant in Bitou Bush invaded woodland in spring but did not link this result with an increase in breeding activity.

The behaviour of birds also changes with season and may affect observed abundance because birds become easier or harder to detect (Er 1995). An example of this type of behavioural change is increased calling and conspicuous display in spring (Er 1995).

Seasonal changes in foraging behaviour have also been noted in insectivorous birds in sclerophyll forest in Tasmania (Cale 1994). Cale (1994) suggested that one of the implications of seasonal changes in the foraging behaviour of birds is that managers of forest should maintain a diversity of habitat and foraging substrates to accommodate this behaviour in birds.

The bird population in Canberra is strongly seasonal, with bird movement and migration into the city from the surrounding ranges a response to food shortages and colder temperatures in autumn and winter (Frith 1976; Taylor and Canberra Ornithologists Group 1992). Other birds move into Canberra from further afield, with many of these migrants arriving in summer and spring and staying to breed (Taylor and Canberra Ornithologists Group 1992). Resources, such as food, and increased physiological demands due to colder temperatures, are also governed by season and climate, and strongly influence patterns of bird behaviour in Canberra. Finally, Er and Tidemann (1996) demonstrated the seasonal nature of bird assemblages in *E. melliodora*-*E. blakelyi* woodland remnants in the ACT, and outlined the important and changing ecological functions provided to birds by this lowland ecosystem.

## 2.7 The influence of diurnal patterns of behaviour on birds

Ford (1989) noted that southern Australia experiences marked changes in weather from day to day, and suggested that this variability may influence bird populations. Leach and Watson (1994) studied diurnal patterns of bird abundance in softwood scrub and open eucalypt forest in south-east Queensland. Their study was unable to detect significant effects on bird abundance for time of day, based on the survey techniques they used. A general trend they reported was that the number of species and individuals recorded was usually low just after sunrise and then increased to a stable level for the remainder of the day.

Slater (1994) found that colder temperatures were negatively correlated with the number of birds seen and heard when conducting area-searches to determine bird abundance and species richness. Ford (1989) suggested that diurnal patterns of bird behaviour may be related to thermoregulation

of body temperature. He suggested that for some species in very cold temperatures, short periods of inactivity may be the most efficient option in terms of energy reserves and foraging. Adverse weather conditions such as high wind, extreme cold and driving rain also reduce bird activity (Er 1995). Clearly, in some circumstances, the energetic costs of being active in poor weather make the option of being inactive, and hence more difficult to detect by an observer, an efficient behavioural strategy. Thus diurnal patterns of behaviour are influenced by weather conditions, which may affect sampling of bird populations.

Ford (1989) also suggested that the abundance of food, such as insects may be influenced by variations in temperature. Thus, inactivity in cold periods of the day in insectivorous birds may be related to inactivity in their insect prey. This principle may also apply to raptors and other predators, with prey inactivity related to weather conditions, making it less efficient to hunt at certain times of the day or on certain types of days. Similarly, many of the larger raptors utilise thermals to soar in an energy efficient manner when searching for food (Olsen 1995). This requires sufficient warm air to be heated (often an afternoon phenomenon) and as a result, some raptor activity will be related to diurnal patterns associated with the occurrence of thermals (Olsen 1995).

Although diurnal patterns of behaviour in birds are not well studied, they are thought to relate to the availability of food and the energetic demands of being active under ambient conditions (Ford 1989). In adverse weather conditions, such as extreme cold or driving rain, birds will be less active (Er 1995), probably because it is more efficient to stay warm in a sheltered place using up energy reserves than it is to seek food under extreme conditions (Newton 1998).

## 2.8 The importance of reproduction to patterns of bird abundance

The breeding component of bird populations is fundamental to patterns of abundance and distribution (Perrins and Birkhead 1983; Newton 1998). Most extinction events involving birds have occurred because of impacts on the reproductive cycle, for example high levels of predation of nesting birds on islands (Wiens 1992b; Newton 1998). Similarly, drastic declines in the abundance of some raptor populations are now known to be caused by agricultural chemicals thinning egg shells, resulting in reproductive failure (Olsen 1995). Thus, changes in the

environment that modify reproductive success can profoundly change the abundance and distribution of birds.

Nesting sites are an important resource provided by the environment and can limit bird populations by limiting the number of breeding birds that successfully rear young (Perrins and Birkhead 1983). In forests that have been selectively logged research suggests that a significant long term impact on forest birds may be the reduction of hollows which provide nesting sites for hollow-dependent species (Smith 1985). For many smaller birds, the amount of protective shrub cover may set an upper limit on reproductive success (Ford 1989). This effect has been demonstrated in the Superb Fairy-wren with *Rubus spp* increasing breeding success (Nias 1984; Nias and Ford 1992). Thus, the quality and quantity of nesting sites may profoundly influence the 'carrying capacity' of habitat in relation to bird abundance (Perrins and Birkhead 1983).

Ford (1989) suggested that the relatively large number of co-operative breeders in Australia, (between 10 and 15 percent of birds (Recher 1985)), provides evidence that successful breeding in Australia is limited by the impacts of predation. He stated that:

*“For co-operative breeding to evolve there must be some advantage in staying with the family (or joining a new group) over solitary wandering in search of a territory. I have argued that the advantages of group-living, especially the detection and avoidance of predators, tip the scale in favour of belonging to a group even if this requires helping to rear young that are not your own.”*

He supported this statement with data on nesting failures and estimated that 80 percent of these failures are attributed to predators in Australia (Ford 1989). He also noted that the impact of predation on reproductive success will be heavily influenced by the availability and quality of breeding territories (Ford 1989). These conclusions support the notion that exotic shrub cover in woodland may influence bird assemblages by affecting the availability and quality of breeding territories of bird species that nest in or under exotic shrubs. Other explanations for the prevalence of cooperative breeding in Australia include the unpredictable climate (Rowley 1976) and general shortage of food (Thomas 1974), and alternatively, an array of unknown ecological factors acting in synergy with the needs of individual species (Dow 1980).

Birds renew their abundance through reproductive success. Where the reproductive cycle has been seriously impeded or limited over a long period of time, population decline and extinction

events have resulted (Newton 1998). An important component of the reproductive cycle in birds is the quality and quantity of nest sites. A key aspect of nest site quality is protection from predators. These theoretical considerations suggest a potentially influential role for exotic shrub cover in woodland in terms of bird populations – the provision of safe nesting sites and refuge from predators.

## 2.9 Human impacts on woodland habitat and birds in Canberra

Estimates suggest that around 90 percent of *E. melliodora*-*E. blakeyi* woodland has been cleared in the ACT (Landsberg 2000). Clearing for urban development remains a significant threat today, with areas of high conservation value earmarked by urban planners for housing developments (ACT Government 1999). Other threats to the patches of woodland that remain include: provision of urban services and infrastructure; firewood collection; failure of tree regeneration; rural tree dieback; pasture improvement; grazing; invasion by weeds and exotic/feral animals (including uncontrolled pets); and changed fire regimes (ACT Government 1999). This discussion will briefly consider the threats listed above, with particular attention to impacts on bird populations.

Clearing of woodland directly destroys bird habitat and contributes to the isolation and fragmentation of remaining patches of woodland. Freudenberger (1999) developed guidelines for enhancing the conservation values of grassy woodland in the ACT/NSW region. He found that woodland birds in the ACT appear to respond to patch size and isolation in different ways. For instance, the Hooded Robin (*Melanodryas cucullata*) is listed as vulnerable under the *Nature Conservation Act 1980* (ACT Government 1999), and was found to be sensitive to patch size, isolation and the structural complexity of woodland habitat (Freudenberger 1999). Freudenberger (1999) also identified the Speckled Warbler as being specialised in its habitat requirements. He suggested this bird requires structural complexity, proximity to other patches of woodland and moderate sized patches of woodland habitat to thrive. He also determined that a further 22 woodland birds were moderately sensitive to isolation, patch size and structural diversity in the ACT/NSW region, while eleven bird species were considered tolerant to these types of landscape disturbance.

Reid (1999) considered threatened and declining birds in the New South Wales Sheep-Wheat Belt. He determined that bird assemblages throughout southern Australian regions were in



decline, as evidenced by local extinctions, declining abundance of some species, and an overall reduction in the species richness of bird populations. His research identified 60 landbirds that were at risk in the NSW Sheep-Wheat Belt. Issues he listed as of most concern included: land degradation (e.g. salinity and dieback); grazing by feral herbivores; weed invasions; loss of and competition for hollows; lack of or limited regeneration in remnants; and predation by feral carnivores.

Barret, Ford and Recher (1994) considered conservation of woodland birds in landscapes fragmented by clearing for agriculture on the New England Tablelands near Armidale. They determined that the most bird species per unit of land occurred in patches of woodland between 20 and 400 hectares in size, while smaller patches or larger patches, had fewer species. They also determined that around 67 percent of birds recorded in their study were tolerant of moderate levels of woodland fragmentation, although they cautioned that their analysis was biased towards common, hardy species. These findings and the work of Freudenberger (1999) indicate that reduction of patch size, fragmentation and isolation caused by clearing of woodland, will have particularly severe consequences for a sub-group of woodland birds, while others may be relatively resilient to further fragmentation.

The removal of dead timber and branches during firewood collection is thought to impact on birds of temperate woodland by reducing the structural diversity of woodland habitat (Reid 1999; ACT Government 1999). Laven and MacNally (1998) determined that the species richness of birds in box-ironbark forest of central Victoria was associated with fallen timber. They argued that two of the most likely explanations for this were that fallen timber provides food and foraging opportunities, and possibly shelter and refuge from predators.

Landsberg (1990) determined that dieback of mature trees in Blakely's Red Gum (*E. blakelyi*) is associated with the density of herbivorous insects in the canopy, and suggested that chronic levels of insect attack may be more important than acute outbreaks. She also suggested that nutrient enrichment of the canopy may be implicated in rural tree dieback (Landsberg 1990). Exotic shrubs belonging to the Family Rosaceae (which includes *Pyracantha spp*, *Rubus spp* and *Rosa spp*) commonly develop an association with ectomycorrhizae fungi, which increases the availability of nutrients to plants (Mulvaney 1991) and influences nutrient cycles in woodland (Tommerup and Bougher 2000). This suggests that exotic shrub invasion of woodland may

influence the availability of soil nutrients, with implications for nutrient enrichment of the canopy. Other processes which enrich soil nutrients in ACT woodland include pasture improvement and fertiliser application on rural leases, while in woodland adjacent to urban areas, run-off from gardens and streets play a role (ACT Government 1996a).

A decline in the abundance and diversity of insectivorous birds may also be a factor in terms of dieback in woodland (Ford 1985; Loyn 1985; Landsberg 1990; Barrett and Davidson 2000). Ford and Bell (1980) have shown that the density of birds in eucalypt woodland is reduced as the severity of dieback increases. They also noted that small insectivorous birds disappeared more rapidly as tree health declined, than did larger birds, such as the Crimson Rosella. The relative health of mature trees in *E. melliodora-E. blakelyi* woodland is therefore likely to be important to the composition of bird assemblages.

Failure of tree regeneration in *E. melliodora-E. blakelyi* woodland in the ACT is thought to stem mostly from grazing pressure (ACT Government 1999) and grazing was identified as a key threat to birds of the NSW Sheep-Wheat Belt (Reid 1999). Grazing animals, such as sheep and cattle, diminish the understorey and remove seedlings of the dominant trees (ACT Government 1999). Reid (1999) suggested that grazing had simplified the understorey of woodland, resulting in less diverse bird assemblages. Landsberg (2000) noted that grazing continues to degrade temperate woodland in the ACT and that changes in bird assemblages are linked to the loss of shrubs and habitat simplification caused by grazing. This relationship is of interest because the removal of exotic shrubs in weeded woodland also simplifies habitat, through loss of shrub cover.

It is possible that tree regeneration may also be affected by exotic shrubs, which shade out native plant species and compete for other resources such as space, nutrients and water. Some weeds are also known to interfere chemically with competing vegetation by producing compounds that inhibit or damage other plants (a form of interference termed allelopathy) (Cousens and Mortimer 1995). Long term failure of tree regeneration would have major adverse impacts on biodiversity.

A less obvious, but important effect of the presence of environmental weeds in *E. melliodora-E. blakelyi* woodland is their impact on government policy concerning the assigning of conservation value. At sites where environmental weeds are present the status of a potentially threatened

community is likely to be downgraded (ACT Government 1999). For example, *Action Plan No. 10, Yellow Box/Red Gum Grassy Woodland - An endangered ecological community* stated:

*“Degradation of remnant woodlands of all kinds is frequently indicated by the presence of Briar Rosa rubiginosa, Hawthorn Crataegus monogyna, African Boxthorn Lycium ferocissimum and the many species of thistles which invade heavily grazed or disturbed ground. The presence of these woody weeds and introduced grasses has caused many otherwise significant woodland patches to be excluded from this Action Plan because their understorey no longer contains a sustainable level of native species.”* (ACT Government 1999).

In this thesis I argue that the decision to exclude otherwise significant patches of woodland from the scope of the above action plan, because of exotic shrubs and other exotic plants, ignores the potential importance of these shrubby woodlands to birds. Furthermore, a number of researchers have determined that even small, degraded patches of native habitat invaded by exotic vegetation can play a positive role in bird dispersal and conservation (Ekert and Bucher 1999; Date *et al.* 1991).

Hunting by domestic cats and strays is another significant pressure on birds of woodland in Canberra (Landsberg 2000). This potential problem is exacerbated by the proximity of people’s homes to some areas of woodland (ACT Government 1996a). The presence of exotic shrub cover in woodland may mitigate the impact of cats on birds, by providing safe nest sites and protective cover. Roaming dogs have been reported hunting in packs in Canberra Nature Park and disturb and or kill wildlife (ACT Government 1999). The impacts of dogs on woodland birds, if any, are not well known.

Foxes occur in woodland in Canberra Nature Park (ACT Government 1996a) and are also highly effective predators of small to medium sized birds (Saunders *et al.* 1995). Although known for their predatory impacts on native wildlife, foxes are also effective scavengers (Saunders *et al.* 1995). Being close to urban areas, it is possible that the density of foxes in woodland is higher than may be expected because of the scavenging opportunities provided by human settlements. Because Australian birds did not co-evolve with foxes, many species may not have developed effective defensive behaviours (Saunders *et al.* 1995). Saunders *et al.* (1995) have also suggested that habitat modification and fragmentation have probably exacerbated the predatory impact of foxes on native fauna. A study of the diet of urban foxes in Canberra, however, found that bird

was not a major part of the diet, suggesting that their impact on birds in Canberra may be minimal (Maston 1995).

Exotic shrubs in woodland may mitigate the impact of foxes on bird populations, in much the same way as cat predation may be mitigated (i.e. provision of refuges). However, studies of urban fox populations have shown that the availability of cover is also important to the density and distribution of foxes (Saunders *et al.* 1995). Hence, exotic shrub cover may provide protection and sanctuary to fox populations in urban woodland habitat and increase fox density. Foxes are also implicated in the spread of exotic shrubs, through the ingestion and excretion of viable seeds (Bass 1990; Peter 2000).

Rabbits are listed as a high priority pest species in the draft Management Plan for Canberra Nature Park (ACT Government 1996a). Adverse impacts on birds of woodland in the ACT may result from modification to vegetation and soil caused by grazing rabbits (Standing Committee on Conservation, Heritage and Environment 1994). Exotic shrubs, such as Boxthorn, protect rabbit warrens and provide rabbits with protective cover (Peter 2000). Hares are listed as a medium to low priority pest species of Canberra Nature Park and also have adverse impacts on native vegetation, such as nipping the tops off native seedlings (Standing Committee on Conservation, Heritage and Environment 1994).

Pest bird species are given a low priority in the Management Plan for Canberra Nature Park (ACT Government 1996a). Exotic birds can, however, have adverse impacts on native birds by competing for limited resources, such as hollows in trees. Birds that operate as "pests" may include native species, such as the Pied Currawong, which has increased in number in the ACT (Taylor and Canberra Ornithologists Group 1992) and may be implicated in the regional decline of small native birds, such as the Jacky Winter (Hoskin *et al.* 1991). Exotic shrubs may play a role in increasing the abundance of "pest" birds, by providing food and other resources, such as cover and nesting sites.

The presence of exotic shrubs in *E. melliodora*-*E. blakelyi* woodland provide food for exotic birds (Mulvaney 1986) and may also provide shelter. Exotic birds known to consume the berries of exotic shrubs include the Common Mynah (*Acridotheres tristis*), Common Starling (*Sturnus vulgaris*), Common Blackbird (*Turdus merula*), Song Thrush (*Turdus philomelos*), House

Sparrow (*Passer domesticus*) and European Greenfinch (*Carduelis chloris*) (Loyn and French 1991; Mulvaney 1986; Peter 2000).

The sources and sinks of nutrients in urban patches of *E. melliodora*-*E. blakelyi* woodland in Canberra may also have ecological impacts. Milewski (1986) correlated the presence of soil nutrients with the occurrence of invasive fleshy fruited plants. Mulvaney (1991) also suggested that the flow of nutrients from urban catchments into bushland may have influenced the bird dispersed spread of exotic woody weeds in south-eastern Australia. Moreover, he noted that some exotic shrubs are commonly associated with ectomycorrhizal fungi (Mulvaney 1991), which influences nutrient cycles in temperate woodland (Tommerup and Bougher 2000).

Cale and Hobbs (1991) suggested that nutrient enrichment of roadsides appeared to increase the percentage cover of native plants, but may also increase the competitiveness of exotic vegetation. They also found that nutrient enrichment of soil was present in a gradient, with elevated levels of nitrogen and phosphorous closest to the source (fertiliser run-off from agricultural land). They were unable to determine any associations between nutrient enrichment in the soil and insect defoliation in the canopy of eucalypt trees, suggesting that this work was confounded to some extent because of problems with the methods used.

In conclusion, large scale clearing of lowland woodland in the ACT has resulted in isolated patches of *E. melliodora*-*E. blakelyi* woodland, surrounded by a matrix of urban and rural land-use. This fundamental change in context threatens the conservation values of these areas but also provides ready access for management and thoughtful intervention.

## 2.10 Studies on the impact of environmental weeds on birds

The establishment of exotic vegetation in native habitat may have differential impacts on bird assemblages (French and Zubovic 1997). Similarly, the context of the native habitat and the nature and extent of weed invasion will affect the conservation implications of changes in floristics and structure. Some researchers have implied that eradication of exotic shrubs and trees may have adverse impacts on birds (Date *et al.* 1991; Ekert and Bucher 1999; Peter 2000). In all three cases an argument was made for staged eradication of exotic plants and planting of replacement native plants. Frith (1969) noted that the extensive plantings of both native and

exotic trees and shrubs, including exotic berry-bearing shrubs, greatly increased and diversified the food and cover for birds in the urban landscapes of Canberra. These observations suggest that the influence of exotic shrub invasion on bird assemblages of native habitat are far from obvious and vary with circumstance. A selection of studies that considered the impacts of environmental weeds on birds and native ecosystems are reviewed below.

Mimosa (*Mimosa pigra*) is a tall shrub from South America which has proven to be a highly invasive plant in the floodplains of the Northern Territory (Braithwaite *et al.* 1989). Braithwaite *et al.* (1989) compared native vegetation communities in tropical northern Australia that were infested by Mimosa with areas that were relatively free of this plant to determine impacts on native flora and fauna. Key findings in relation to fauna were that small mammals were more abundant, frogs were little affected, while many birds and lizards were lower in abundance in the weed invaded sites (Braithwaite *et al.* 1989).

Birds that appeared to have lowered abundance in Mimosa invaded sedgeland included Cisticolas (*Cisticola spp*), while in wet forests the Rufous-banded Honeyeater (*Conopophila albogularis*) and Peaceful Dove (*Geopelia striata*) were generally fewer in number in infested sites (Braithwaite *et al.* 1989). The Willie Wagtail (*Rhipidura leucophrys*), however, appeared to benefit greatly from the presence of Mimosa (Braithwaite *et al.* 1989).

Overall, Braithwaite *et al.* (1989) concluded that Mimosa will have adverse impacts on most terrestrial birds due to the loss of native vegetation in infested areas. They also suggested that the benefits to mammal species, in terms of increased abundance, are probably due to reduced predation by avian predators and that in the long term, this advantage may be outweighed by food shortages caused by loss of native vegetation.

They hypothesised that forest areas of the floodplains may eventually be completely converted into Mimosa shrubland, due to Mimosa suppressing tree regeneration and seedling growth. They attributed the predicted severity of Mimosa's impact on native ecosystems to three main characteristics: the high dominance of the invading plant; the gross change in vegetation structure caused by Mimosa infestation; and the conversion of a wide range of structural vegetation types into a single homogenous shrub layer. They also predicted that if Mimosa invasion continues in

the manner they have suggested, the adverse impacts on biodiversity will be massive in scale and scope.

Tamarisk trees (*Tamarix Aphylla*) grow up to six metres high, are salt tolerant and are native to arid areas of Asia minor, north-western India and north-eastern Africa (Griffin *et al.* 1989). Griffin *et al.* (1989) studied the impact of exotic Tamarisk Trees on native ecosystems of the Finke river system in arid central Australia. They found that displacement of eucalypts by Tamarisks was facilitated by flooding which tore out native trees on the edge of the river. They thought that the susceptibility of river banks to flood damage of this type was probably facilitated by poor land management practices, such as over-grazing. They also found that in areas where native trees had been displaced by Tamarisks, the species richness of flora and fauna was reduced, and bird and reptile abundance lowered. Saline conditions resulting from land disturbance, were also thought to have contributed to the success of invading Tamarisk trees along the Finke River (Griffin *et al.* 1989).

Griffin *et al.* (1989) also reported that only four percent of the total number of birds in Tamarisk invaded habitat were seen in the Tamarisk trees. They observed that many of these birds roosted in the thick foliage at night, that none appeared to feed in the exotic trees, and only one species (the Willie Wagtail, *Rhipidura leucophrys*) appeared likely to be nesting in Tamarisk trees. They also noticed that the number of birds foraging on the ground away from trees was small and was similar at invaded and non-invaded river sites. They attributed the reduced bird numbers in Tamarisk invaded river banks to fewer native gums in these areas.

Another key finding of their study was that the trophic structure of bird assemblages in Tamarisk invaded habitat was different to that of non-invaded areas. Predators, scavengers, insectivores and nectarivorous/frugivorous birds were significantly less abundant in areas of Tamarisk invasion, aerial insectivores were more abundant, while granivores occurred in similar proportions (Griffin *et al.* 1989). Thus, the composition and structure of bird assemblages of Tamarisk invaded habitat appears to have changed in response to this invasion.

Griffin *et al.* (1989) proposed the following reasons for reduced bird abundance and species richness in Tamarisk invaded habitat: firstly, Tamarisk trees produce only tiny flowers, and hence may attract few insects and no nectarivorous birds; secondly, foliage of the Tamarisk tree appeared

to be devoid of herbivorous insects; thirdly, the litter beneath Tamarisk trees appeared to suppress ground-dwelling insects; and finally, the number of hollows and amount of fallen timber in invaded sites was reduced. These characteristics point to Tamarisk invasion resulting in a net reduction in resources for birds, where these exotic trees have displaced eucalypts (Griffin *et al.* 1989). Not surprisingly, Griffin *et al.* (1989) concluded by suggesting that if Tamarisk trees continue to spread along arid river systems, native biodiversity will be adversely affected.

Griffin *et al.* (1989) sampled birds by observing them for five minutes at twenty points, located 100m apart, at a study site invaded by Tamarisk Trees and a study site dominated by River Gums (*E. camaldulensis* and *E. microtheca*). Sampling of birds was limited to four days and sites were sampled on alternate days, which means two days of sampling per treatment (i.e. infested areas and uninfested areas). Although their conclusions from this work, as described earlier, seem reasonable, the limited nature of the data set makes it difficult to generalise confidently from their results. The short time frame for observing birds may also explain why they found no significant differences in the species richness of birds at invaded and uninvaded sites. Alternatively, differential effects on foraging groups resulted in no net differences in the diversity of birds, although the composition of species was different (e.g. there were more aerial insectivores in invaded areas) (Griffin *et al.* 1989).

French and Zubovic (1997) found little evidence to suggest that Bitou Bush (*Chrysanthemoides monilifera*) infestation negatively affected bird assemblages in coastal woodland on the south coast of NSW. They found that overall bird abundance did not differ between infested and non-infested sites, while the species richness of birds differed only in one area. They also determined that weed infestations have a differential impact on foraging groups. Canopy-feeding generalists and understorey insectivores occurred in greater numbers in invaded woodland, with some areas also showing higher species richness for these sub-groups of birds (French and Zubovic 1997). They suggested that this finding indicated that weeds were providing some resources for canopy generalists and understorey insectivores.

French and Zubovic (1997) identified two foraging guilds which showed lower abundance and reduced species richness in Bitou Bush infested woodland. These were birds that relied fairly exclusively on plant material (mainly fed on nectar, fruits, flowers and or seeds); and scavengers and raptors which fed on vertebrates and large insects. They suggested that weeding in coastal



woodland is likely to have a positive effect on these two guilds. They also concluded that changes in the floristic composition of the woodland due to infestation of Bitou Bush is likely to affect the first group of birds adversely because of their more specialised dietary needs. Assigning birds into groups, however, as acknowledged by French and Zubovich (1997), conceals negative and positive effects on individual species. Thus, the generalisation concerning adverse impacts on birds that mainly fed on nectar, fruits, flowers and or seeds is open to challenge, particularly as the Crimson Rosella was included in this group. This species consumes the fruit of Bitou Bush (Loyn and French 1991), and in southern NSW feeds extensively on *Crataegus spp*, *Cotoneaster spp* and *Pyracantha spp* berries (Foreshaw and Cooper 1991), suggesting that exotic shrubs have expanded the food available to this Rosella.

One interesting hypothesis which was raised by French and Zubovic (1997) was that bird species may use exotic areas for particular activities (such as protection from predators at particular times of the day) but return to uninfested areas to meet other daily needs. This hypothesis was based around their finding that there was temporal variation in bird numbers and use of weed infested woodland. They also noted the importance of possible long terms effects of Bitou Bush invasion and suggested that it is likely that competition with Bitou Bush will reduce the establishment of a tall understorey and canopy species in infested sites.

An important relationship between Australian birds and exotic shrubs is the consumption of fruit by birds (Forde 1986). Bass (1991) compared bird dispersal of Hawthorn (*Crataegus spp*) seeds and St Lucies Cherry (*Prunus mahaleb*) seeds in the northern tablelands of New South Wales, which differ in their invasiveness in this region. He noted that both these species are members of the Rosaceae, they co-occur and they are dispersed by birds and mammals. Bass (1991) also noted, however, that Hawthorn was deliberately introduced over a much wider geographical area and suggested that this is one reason why Hawthorn has spread over more of the region.

Bass (1991) identified seven birds as dispersal agents of St Lucie's Cherry (Pied Currawong, Olive-backed Oriole, Noisy Friarbird, Red Wattlebird, Mistletoebird, Silvereye and Australian Raven), and two for Hawthorn (Pied Currawong and Silvereye), and noted that the dispersal ecologies of these two plants differ greatly. One important difference is that ripe fruit is only available on the cherry trees for one month, compared to up to nine months for Hawthorn (Bass 1991). A second difference he observed was that the more important dispersers of cherry seeds

(Noisy Friarbirds and Red Wattlebirds) operated over distances greater than 100m, compared to distances of 1000m or greater by the gregarious and abundant Pied Currawong. Bass (1991) summarised factors that limited the dispersal range of the cherry as the short fruiting season and limited geographical introduction of the plant. He also noted that autumn and winter are a time of low invertebrate abundance, with Pied Currawongs switching to fruit in these times of invertebrate shortage. This, in combination with the long fruiting season and widespread Hawthorn population, means that the fruit is available to dispersers over a long time and a wide area (Bass 1991). Mammals listed by Bass (1991) as dispersing seeds of Hawthorn included foxes, possums, and wallaroos.

Bass (1990) also considered the role of Pied Currawongs in seed dispersal in an earlier study located in Armidale. He found that Pied Currawongs fed extensively on Firethorn (*Pyracantha spp*) and Hawthorn (*Crataegus spp*), and regurgitated viable seeds. The main fruit eaten was from *Pyracantha angustifolia*, *Pyracantha rogersiana*, *Crataegus monogyna*, *Ligustrum lucidum* and *Ligustrum sinense*, and frugivory was observed from late March through to September (Bass 1990). Feeding behaviour of the Pied Currawong was described as follows by Bass (1990), who stated:

*“Fruits were ingested whole and after 5-15 minutes undigested pulp and seeds were regurgitated as pellets, approximately 5cm long and 2cm in diameter. Pyracantha spp., with yellow, orange or red fruits were the first to be eaten (March-April) and formed the major part of the winter diet of Pied Currawongs. Whole shrubs were stripped of fruit, but some shrubs retained fruit until early summer. Red C.monogyna fruits were eaten in small quantities from April to early July, with many shrubs retaining fruit until summer.”*

Bass also noted that feeding flocks of Pied Currawongs varied from as few as four to five birds, and up to about 100 birds, and that some birds flew upwards of 300m from where they had fed before regurgitating pellets. An important finding of his work was the regularity of Pied Currawong movements into and out of Armidale along set routes, providing good opportunities to monitor their population. A second important finding was that Pied Currawongs appeared capable of spreading seeds of exotic shrubs over many kilometres (Bass 1990).

African Boxthorn (*Lycium ferocissimum*) is a bird dispersed weed introduced into Australia from South Africa in the 19th Century. It forms a dense shrub protected by fierce spines (Peter 2000). Its fruit consists of red berries, 10mm in diameter and each berry has from 30 to 70 seeds inside

the flesh of the fruit (Peter 2000). Peter (2000) studied birds and their interactions with Boxthorn in Victoria in the autumn and winter of 1999.

Peter (2000) listed species that he observed consuming the seeds or fruits of Boxthorn berries and also birds that used the Boxthorn for shelter and sanctuary. He supplemented his own observations with historical records to compile a list of birds recorded in the study area consuming Boxthorn berries. Species observed consuming berries included the Crimson Rosella, House Sparrow, Mistletoebird, Silvereeye and Common Blackbird. Birds observed using clumps of Boxthorn as a refuge from danger included the Superb-Fairy Wren, Yellow-rumped Thornbill and the Red-browed Finch. Disused nests of the Superb Fairy-Wren and the Yellow-rumped Thornbill were located in Boxthorn shrubs, while the Red-browed Finch had been recorded nesting in Boxthorn in earlier studies (Peter 2000). Other observations from his study were an association between rabbit burrows and clumps of Boxthorn and the presence of undigested Boxthorn seeds in the scats of Red foxes.

Peter (2000) concluded that Boxthorn is a destructive weed. However, he also suggested that because birds depend on it for food and refuge, an alternative understorey of native plants should be established before it is weeded to minimise adverse impacts on dependent fauna.

Forde (1986) studied the relationships between birds and fruits in temperate Australia. He determined that there are few fruit specialists, with most birds taking fruit in an opportunistic manner. He also suggested that most birds in temperate Australia have been recorded taking fruit from only one or a few plants, although the Silvereeye, Emu and the Singing and Spiny-cheeked honeyeaters take fruit more frequently and obtain fruit from more than ten genera of plants. Forde (1986) also discussed the importance of bird dispersal of seed and the characteristics of fruit that made them attractive to birds.

Forde (1986) noted that birds have been recorded consuming the fruits of over 100 plant species in temperate Australia from 36 plant families. He also noted that most of the fruit producing plants visited by birds as a food source produce fleshy fruits for a two to four month period over spring-summer or summer-autumn. Thus, based on Forde's research, there is a clear seasonal peak of fruit availability in summer throughout much of southern Australia, with fruit availability much more limited in winter.

Williams and Karl (1996) examined the importance of fruit from native and exotic plants in the diets of birds of forest remnants in New Zealand. They determined that endemic birds ate the least adventive fruit, while Silvereyes (non-endemic) ate the widest range of indigenous and adventive fruits. Exotic birds that ate significant amounts of adventive fruit were the Common Blackbird and Song Thrushes. They also determined that the presence of exotic fruiting vegetation extended the availability of fruit into winter, although they found that seasonal variation in the numbers of birds caught in their mist nets was small. They concluded that where fruiting weeds replaced native vegetation, the habitat for endemic birds was inferior. They also considered that the dispersal of seed by non-endemic and adventive birds would have adverse impacts on biodiversity.

Forde (1986), Loyn and French (1991) and Mulvaney (1986) made a key distinction between birds that consume fruit and noted that some grind up and destroy the seed (i.e. are seed predators), while others consume the fruit and excrete viable seeds (i.e. are seed dispersers). This distinction is important because it affects the process of seed dispersal and hence, weed invasion of woodland habitat.

Loyn and French (1991) reviewed the relationship between birds and environmental weeds in south-eastern Australia. They concluded that exotic plants are less valuable to native birds than the native vegetation they replace. They noted that exotic plants provide a source of food and cover that attracts native birds and may sustain some populations. They argued that the apparent benefits of exotic plants to birds were more easily observed than adverse effects but were potentially less important than negative impacts arising from changes to vegetation structure and composition through weed invasion.

This view neatly captures the dilemma facing managers who are, to some extent, forced to intervene with an incomplete picture of how their intervention may impact on whole ecosystem function and resilience. Loyn and French (1991) also predicted in their review that the invasion of environmental weeds into native ecosystems would change the ecology and composition of bird assemblages and birds would continue to influence the dispersal of weeds.

Loyn and French (1991) concluded that birds of open woodland and farmland make more use of weeds than birds of forests, where weeds are usually less abundant. They suggested that birds

may use weeds as a resource out of dire need, but that the food or resource provided by the weed may not be as beneficial as native equivalents or replacements. The evidence they provided to support this argument was inconclusive. Moreover, the reverse may be true, as many of the exotic shrubs I considered in my study provided dense protective thickets with spines to deter predators. These characteristics may make them functionally superior to endemic native shrubs in terms of providing safe refuge for small native birds (the resource in this instance being cover rather than food).

Ekert and Bucher (1999) studied the winter use of Large-leaved Privet (*Ligustrum lucidum*) by birds in forest stands located in suburban Lismore, NSW. They determined that during the fruiting season of *L. lucidum*, seventeen species of birds used the Privet, with most being frugivores. They also noted that at the site with the lowest Privet abundance, significantly less insectivory took place. A key finding of their work was that Privet trees in mixed stands supported a greater range of bird activity and were used by more birds than Privet trees in near-pure stands. They concluded that eradication programs aimed at Privet will reduce resources available to birds in and around the Lismore area, unless these programs are gradual and include replacement with suitable native species. Other findings they discussed were the role of exotic vegetation as 'stepping stones' for birds moving between patches of native vegetation fragmented and isolated by clearing and the seasonal importance of fruit as a food resource.

The idea that exotic vegetation may assist bird movement by acting as a stepping stone between fragmented native remnants has been explored by Date *et al.* (1991). They determined that rainforest frugivores, such as pigeons, used exotic vegetation, such as Camphor Laurel (*Cinnamomum camphora*), to move across open, fragmented landscapes to larger areas of continuous native forest. They concluded that in the short term, exotic species that provide winter food, such as Camphor Laurel, should be retained in northeastern NSW to assist these movements. In the long term, Date *et al.* (1991) suggested exotic plants fulfilling the stepping stone role should be replaced by a network of native species.

Mulvaney (1986) discussed the role of birds in the dispersal of *Pyracantha spp* and *Cotoneaster spp* into bushland in Canberra. He observed that birds that fed regularly on the fruit and seeds of these exotic shrubs included the Gang-gang Cockatoo (*Callocephalon fimbriatum*), Australian King Parrot (*Alisterus scapularis*), Crimson Rosella (*Platycercus elegans*), Eastern Rosella

(*Platycercus eximius*), Common Blackbird (*Turdus merula*), Silvereye (*Zosterops lateralis*) and Pied Currawong (*Strepera graculina*). Based on his studies he concluded that *Pyracantha spp* was mainly a food source for the Blackbird, Silvereye and Pied Currawong, and was of less importance to the Rosellas and Starling. *Cotoneaster spp*, however, was fed on fairly exclusively by the Gang-gang Cockatoo, Australian King Parrot and the Rosellas. He suggested that *Pyracantha spp* was more widely distributed than *Cotoneaster spp* because the birds that mostly fed on *Cotoneaster spp* were seed predators (i.e. they destroy the seed when feeding). In contrast, the Blackbird, Starling, Pied Currawong and Silvereye consumed the fruit of *Pyracantha spp* and passed viable seed (i.e. are seed dispersers) (Mulvaney 1986). Thus, Mulvaney (1986) concluded that the preference of seed dispersing birds for *Pyracantha spp* was related to its distribution as an environmental weed.

The studies reviewed here suggest that the impacts of exotic shrubs on birds of native habitat are complex and vary with circumstance. The influence of exotic shrubs on birds may be adverse, neutral and or beneficial. Factors determining these effects include changes to habitat structure and floristic diversity, both of which influence the resources birds need to sustain their populations.

## Chapter 3

### Methods

#### 3.1 Selection of study sites

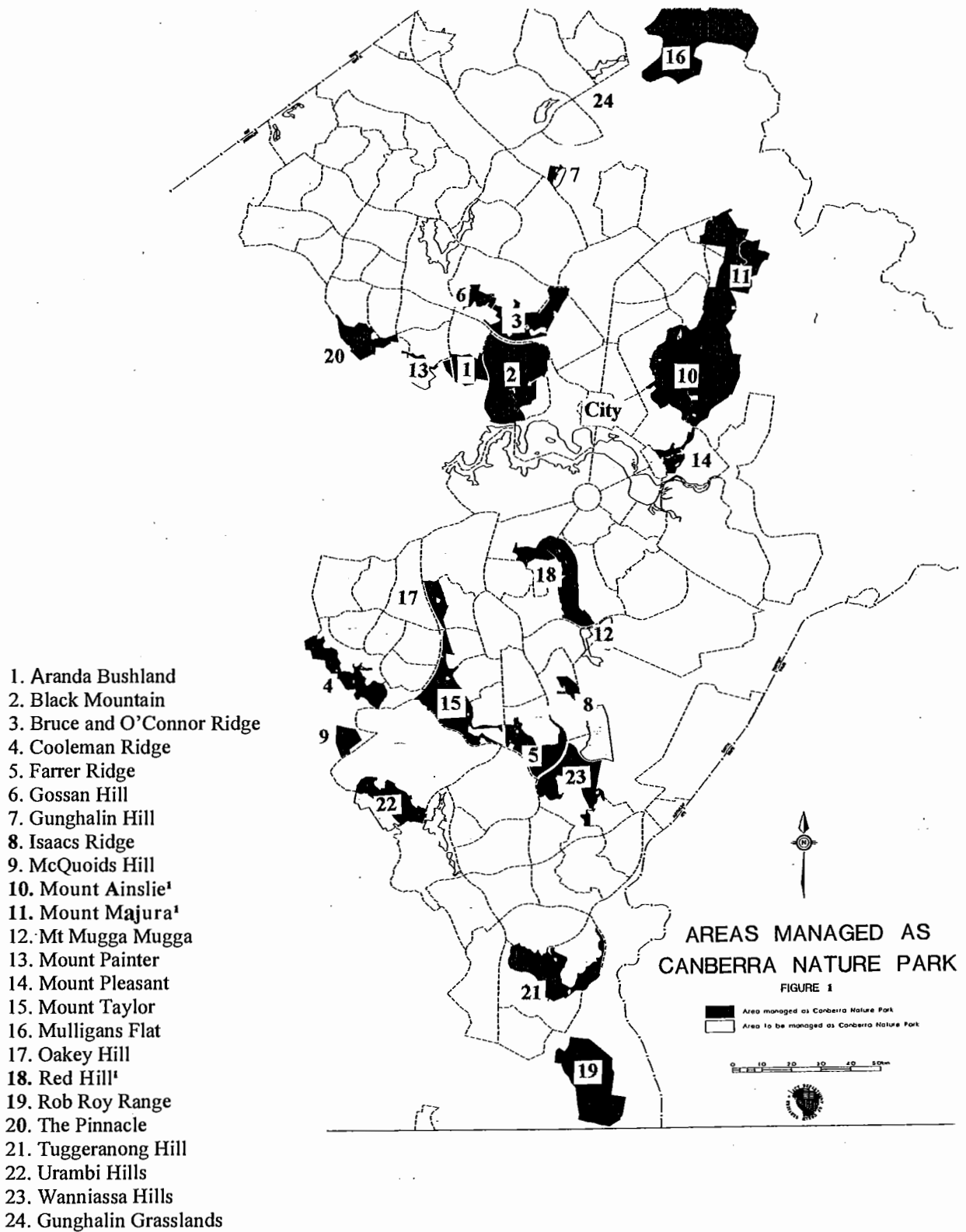
This study was designed as a multi-site comparison of woodland sites with varying levels of exotic woody weed invasion present at the different sites. Twelve 2-hectare *E. melliodora*-*E. blakelyi* woodland sites were marked out in reserve units comprising parts of Canberra Nature Park (with the exception of site 8, which was approximately 200m outside of Canberra Nature Park). Figure 3.1 shows the location of the reserve units of Canberra Nature Park where woodland sites were located. Figure 3.2 shows the approximate location of woodland sites in urban Canberra.

Woodland sites were selected using the following criteria:

- 1) each contained a minimum of two hectares of contiguous woodland suitable for establishment of a rectangular survey plot (100m X 200m);
- 2) each was a minimum of 500m distance from the next nearest site;
- 3) each edge patch was located near urban development;
- 4) each was ungrazed or rarely grazed by domestic stock;
- 5) each had varying levels of exotic woody weed invasion; and
- 6) each was preferably located in Canberra Nature Park.

These criteria were developed around the use of the area search method to sample bird populations, as described by Loyn (1986), Slater (1994) and Er (1995) (the use of the 20 minute area-search method to estimate bird abundance is discussed later in this chapter).

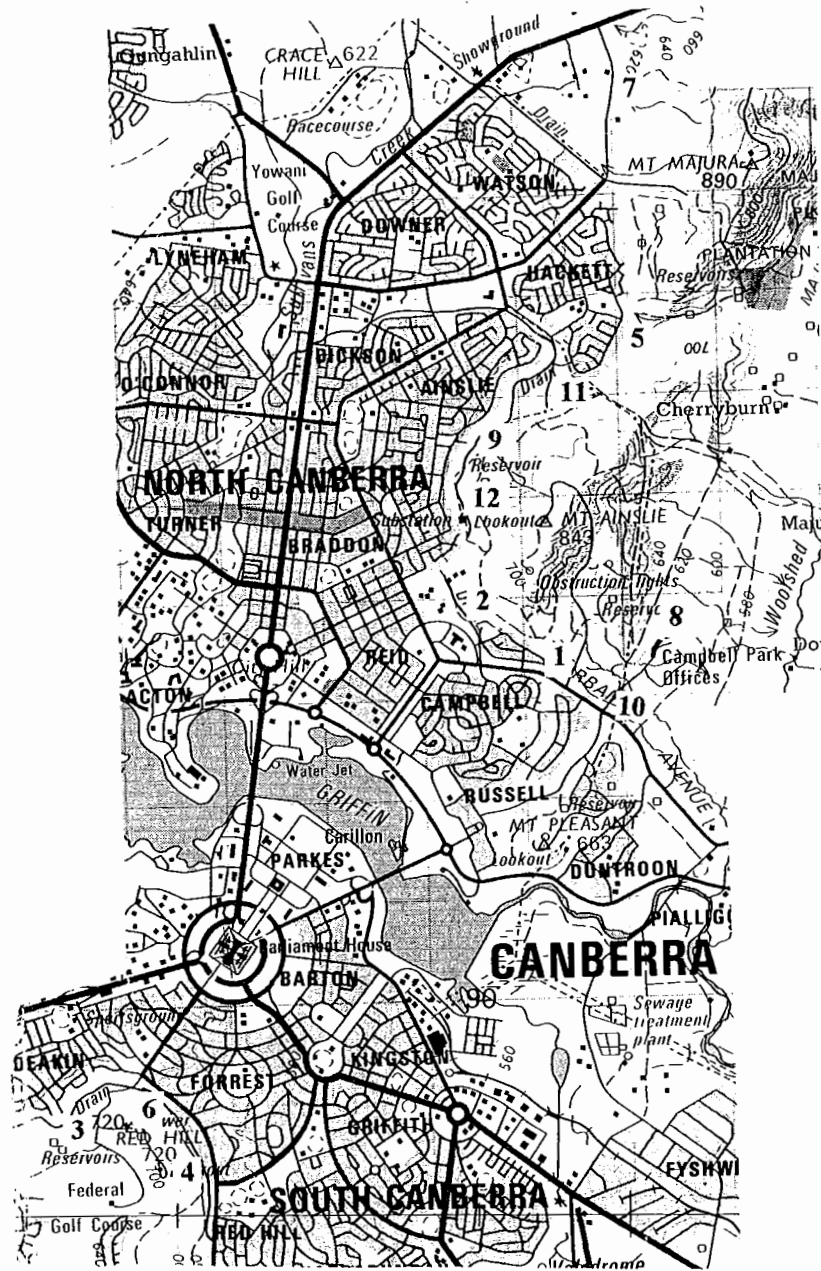
The sites were chosen after initial examination on foot of the following units of Canberra Nature Park: Red Hill; Mounts Ainslie/Majura; Mount Pleasant; Mulligans Flat; McQuoids Hill; Tuggeranong Hill; and Mount Mugga Mugga (Figure 3.1). Areas outside Canberra Nature Park also considered for site selection included woodland remnants near Hall, Castle Hill and Tharwa (within the town and nearby sections of the Murrumbidgee River Corridor Reserve).



**Figure 3.1 – Areas managed as Canberra Nature Park**

(1) Woodland sites were located in these areas (see Figure 3.2)  
 Source: ACT Government 1996.





**Figure 3.2 - Location of the woodland sites used to sample birds**

Source: Australian Surveying and Land Information Group (Commonwealth of Australia 1989)

Scale - 1:75 000

From this broad selection of possible areas, the number of study sites was reduced to the lower slopes of Red Hill, Mount Majura and Mount Ainslie (one site just outside Campbell Park). These areas of land met the listed criteria, while providing enough suitable tracts of woodland to ensure replication of plots which were a minimum of 500m apart.

Woodland located on Mount Mugga Mugga was rejected because of the presence of grazing cattle and horses and because it was situated at a greater distance from housing than considered desirable. Mulligans Flat, an extensive woodland area in the north of Gunghalin, was also excluded from the study population because the surrounding land use was rural, rather than urban. Grazing and the rural nature of surrounding land use for woodlands at Castle Hill and near Tharwa also made these areas unsuitable. The remaining areas listed did not have sufficient contiguous patches of urban edge woodland to enable the marking out of replicate plots with varying levels of exotic shrub cover, which were at least 500m apart.

### **3.2 Rationale for site selection criteria**

The rationale behind the criteria used to select sites is as follows. Two hectares was chosen for the size of the plots because this is a standard size recommended by Birds Australia for the Atlas of Australian Birds project (Birds Australia 1997) and because standardising plot size is important when monitoring areas over time (Loyn 1986). This approach also ensures that the data compiled is compatible with the Birds Australia Atlas of Australian Birds project, increasing the usefulness of the data set.

The selection of patches of woodland adjacent to or near to urban development was considered to be an important component of site selection because these areas comprise an ecological interface, or transition zone, between woodland and the urban environment. From a practical perspective this zone was also of most interest because exotic woody weed invasion in Canberra Nature Park commonly occurs where urban gardens and woodlands are in close proximity (Mulvaney 1984). Urban reserves are also of experimental interest because exotic birds that have established themselves as urban specialists have an opportunity to establish an ecological role in adjacent reserves (Wood 1996). Locating sites on the edges of urban reserves therefore facilitates investigation of the idea that the presence of exotic shrubs may influence the abundance of exotic bird populations in these areas.

Some areas of *E. melliodora-E. blakelyi* woodland considered for site selection carried grazing cattle and horses. Because of the possible effects of grazing on bird populations, these sites were excluded from the study. An exception to this was site 8, which on two occasions was found to have small numbers of cattle grazing in the area during bird sampling.

The level of exotic shrub invasion in tracts of *E. melliodora-E. blakelyi* woodland also influenced the availability and location of sites. Evaluation of the level of exotic shrub invasion was based on a visual estimate of the number, genus and size of exotic shrubs present in tracts of woodland examined on foot. Exotic shrubs included in this aspect of site selection were *Pyracantha spp*, *Cotoneaster spp*, *Rosa spp* and *Crataegus spp*. I also considered dead exotic shrubs in this process, as observations from earlier bird sampling in weed invaded woodland had showed that dead exotic shrubs were visited by birds (D. Evans unpublished data).

Preference was given to areas of *E. melliodora-E. blakelyi* woodland located within units of Canberra Nature Park. The main reason for this preference was that intensive removal of exotic shrubs from selected patches of woodland in Canberra Nature Park resulting from weed management practices provided the opportunity to compare sites with varying levels of exotic shrubs present. Two other reasons for giving preference to woodland in Canberra Nature Park were that, firstly, these areas were unlikely to be affected by development or other disturbances before sampling was complete and that, secondly, management of these areas would remain reasonably consistent during the study period.

Exotic woody weeds were subject to management by ACT Government departments and agencies. Examples of management included spraying of exotic woody shrubs and hand weeding and removal by community volunteers belonging to Parkcare groups. These activities affected much of the study area. Many areas showed signs of earlier weeding, such as the presence of dead free standing exotic shrubs or the stumps of exotic shrubs that had been cut, poisoned and removed. Potential sites that were likely to change in their shrub cover or status during bird sampling were avoided to minimise this source of habitat variability. Despite this, sites 3, 9 and 11 were weeded before bird survey work was completed, although only site 9 was extensively weeded (see Appendix 3).

The minimum distance between 2-hectare plots recommended for the Atlas of Australian Birds project is 400m (Birds Australia 1997). In this study the minimum distance between sites was 500m. I selected this distance on the basis that it provided for a sufficient buffer between sites to ensure that for most birds, the counts were independent of each other, while at the same time providing scope for relatively homogenous experimental material in the form of comparable tracts of woodland habitat.

### 3.3 Use of the 20 minute area-search method for bird counts

The method used to count birds was the 20 minute area-search (Loyn 1986; Slater 1994; Er 1995). This method entails the observer moving freely within the two hectare area and counting and recording all birds seen. Counts commenced at least thirty minutes after sunrise. In her experimental work on the area-search method, Slater (1994) commenced counts at least 37 minutes after sunrise to avoid the initial dawn activity at the first site where counting took place. My decision to commence counts thirty minutes after sunrise was based on the need to mitigate any bias caused by dawn activity.

Loyn (1986) suggested that counts could be undertaken throughout the day, except when the wind is strong or the afternoon is very hot. In this project, counting birds ceased a maximum of three and a half hours after sunrise. The rationale for limiting bird counting to this time frame was based on the work of Slater (1994), who found a negative correlation between species seen and time after sunrise, causing a slight reduction in species observed the later it became in the morning. Arnold (1989) found a relatively constant number of species recorded in Wandoo woodland for the first four hours after sunrise, with some reduction in species observed after that time. This finding also influenced the three and a half hour bird sampling time frame implemented during the study.

Variables recorded were abundance (the number of individual birds counted in 20 minutes) and the number of species (the total number of species recorded in 20 minutes). Birds observed flying over the site were not included in counts or listed as present. I avoided counting birds in rain, heavy fog or very windy conditions to reduce bias caused by extreme weather.

I selected the 20 minute area-search method for counting birds because it is simple, flexible and provided sufficient total time is spent on site, a comprehensive species list can be compiled (Loyn 1986; Slater 1994). Slater (1994) summarised factors affecting the area search method as duration and repetition of counting periods, density of vegetation, abundance of individual species, weather and time of day. She found that the total amount of time spent on site, rather than the number of repetitions of counts, was the most influential variable in terms of compiling a comprehensive list of bird species. She also determined that if density estimates of birds are less important to a researcher's aims than compiling a complete species list, fewer area searches of longer duration (up to 60 minutes long) represented a more efficient approach to bird sampling than a 20 minute area-search. She also concluded that in sites with a sparse understorey a total time of 100 minutes would allow most species to be recorded, whilst a dense understorey would require about 150 minutes of searching. In this study each site was visited a minimum of five times per month, resulting in a minimum of 100 minutes of searching effort per site per month.

The 20 minute duration of counts has been established as a standard approach because experience has shown that longer periods may result in over estimates due to birds being counted more than once (Loyn 1986). Time periods of less than 20 minutes are considered insufficient to provide for an adequate search and many replicates of shorter counts would be biased towards species that behave in a conspicuous manner (Loyn 1986). Loyn (1986) also considered that the mobility of the observer using the area-search method improved the chance of detecting inconspicuous birds due to flushing.

An important strength of the area-search method is that the mobility of the individual allows close observation of birds and can reduce observer bias as a function of distance (Er 1995). A difficulty with the area-search method is the concern that individual birds may be counted more than once, thus resulting in inaccurate estimates of abundance (Er 1995). Experience has shown that standardising of time and area can reduce the influence of this bias (Loyn 1986; Slater 1994).

Common alternatives to the area-search counting method include a fixed-width strip transect or a point count (Er 1995). Fixed-width strip transects are widely used in Australia to estimate species richness and abundance of birds in woodland and this approach is extremely popular with a number of researchers (Er 1995). This method involves measuring and marking a strip and

moving along a transect line at a constant speed counting birds (Er 1995). This method is simple, repeatable and reduces the risk of counting birds more than once (Er 1995).

Arnold *et al.* (1987) found that the transect method gave lower estimates of abundance when compared to the area search method in remnant Wandoo woodland and farmland in Western Australia. They attributed this difference to the area search method flushing birds in trees and on the ground. In the present study I decided not to use the fixed-width strip transect method because of the dense and clumped distribution of the exotic woody shrub cover in invaded sites and the concern that birds in this microhabitat would be overlooked. Moreover, it has been suggested that an area-search approach is buffered against bias caused by extremes of weather to a greater degree than the transect method (Slater 1994). As counts were undertaken during autumn and winter the transect method may also have resulted in greater bias due to variation in weather, as Canberra weather can become very cold in these months.

Another counting option is the use of point counts (Bibby *et al.* 1992). In brief, this method involves standing in one place for a set length of time and counting all the birds seen and heard from that point (Bibby *et al.* 1992). Bibby *et al.* (1992) considered this method to be very efficient, noting that a single observer might visit 10 points in a single morning with relative ease. They also noted that it is a method requiring high levels of observer skill.

Er (1995) noted that point counts were originally developed to estimate bird abundance in habitat that was structurally complex and where bird calls were the most likely method of identification. He considered an important advantage of the point count method to be the simplicity and repeatability of setting up survey points. He also concluded that point counts were best applied in closed forest and in the breeding season, as it is a very useful technique for assessing abundance of songbirds. A key criticism of stationary bird sampling methods in general is that they are more biased toward conspicuous, loudly calling species, than the fixed width transect and area-search method (Loyn 1986).

I determined the point count method to be inappropriate for the open woodland surveyed in this study. This decision was based on the stationary nature of the point count method, which may have lead to cryptic species in dense clumps of exotic shrubs being overlooked. I also considered that visual identification and counting of birds within 2-hectare plots, as described for the 20

minute area-search (Loyn 1986; Slater 1994; Er 1995), was better suited to my skills as an observer. The 20 minute area-search method also required the discipline of locating and visually identifying birds within the boundaries of a 2-hectare plot, allowing habitat variables within this plot to be matched with these observations. I also decided that the flexibility of moving around a 2-hectare plot provided more scope than the point count method in terms of recording observations of birds interacting with exotic shrubs. Other variables recorded when counting birds, were the presence of dogs, cats, foxes, rabbits and hares, and people on site during a 20 minute area-search . Birds observed perching on exotic woody weeds or consuming fruit from these plants were recorded so that any statistical relationships with exotic shrubs could be assessed in light of field observations. Table 3.1 summarises site variables measured or recorded for birds.

**Table 3.1 - Description of site variables recorded or measured for birds**

Name of variable	Description of data item
Total bird abundance	The total number of all birds counted in a 20 minute area-search
Abundance of individual bird species	The total number of a particular species of bird counted in a 20 minute area-search
Species richness	The total number of bird species recorded in a 20 minute area-search
Percentage of counts present	The number of area-searches where a species is recorded, expressed as a percentage of the total number of area-searches completed for that site
Percentage of counts observed in exotic shrubs	The number of area-searches where a species was observed in or on exotic shrubs, expressed as a percentage of the number of area-searches where that species was present.
Birds observed consuming fruit or seeds from exotic shrubs	List of species observed consuming fruit or seeds.

### 3.4 Bird sampling strategy

Bird life in Canberra is heavily influenced by seasonal patterns of migration and movement (Taylor and Canberra Ornithologists Group 1992). In order to account for the influence of season on bird abundance and assemblages in the twelve woodland sites sampled, each calendar month was treated as a discrete subset of data. Bird counts began in March 1996 on Red Hill. Sampling at all twelve sites commenced in July 1997 and ended in June 1998 (personal commitments resulted in November and December of 1997 being omitted from the sampling program). Table 3.2 presents the number of 20 minute area-searches completed for each of the twelve sites over the ten month period covered. Table 3.3 shows the number of 20 minute area-searches completed per month.

**Table 3.2 - Total number of 20 minute area-searches per site (July 1997 to June 1998)**

Site No.	No. of area-searches
1	55
2	56
3	56
4	56
5	56
6	55
7	57
8	55
9	54
10	55
11	55
12	55
<b>Total</b>	<b>665</b>

### 3.5 Vegetation surveys and other supporting data sets

Vegetation was sampled at each of the twelve sites. The general approach taken to recording vegetation followed McDonald *et al.* (1990). For exotic shrubs and native shrubs and saplings, this comprised marking out either one or two 200m transects, along the length of the rectangular



shaped sites. The number of transects used was based on sampling at least twelve measurements in each site (McDonald *et al.* 1990).

**Table 3.3 Total number of 20 minute area-searches completed per month (July 1997 to June 1998)**

Month and Year	No. of area-searches
July 1997	60
August 1997	72
September 1997	64
October 1997	72
November 1997	0
December 1997	0
January 1998	70
February 1998	61
March 1998	73
April 1998	61
May 1998	72
June 1998	60
<b>Total</b>	<b>665</b>

The starting point of each transect was the middle of the 100m base of the rectangular site. The direction of the transect from this point was parallel with the 200m boundaries of the site. Sampling along each transect involved walking a zig-zag pattern along its length, sampling shrubs and saplings encountered going towards or across the transect in the direction P to Q (see Figure 3.3). All exotic shrub species encountered while sampling as described above were sampled, while for native shrub cover only shrubs and trees estimated as less than four metres tall were sampled. Variables measured for each shrub or sapling encountered were crown width and distance to the next nearest sapling or shrub crown encountered, using the transect and the towards and across rule, to guide the zig-zag pattern walked (McDonald *et al.* 1990) (Figure 3.3). The height of each species was estimated, not measured. The raw shrub data for each site was converted into percentage crown cover, which estimates the proportion of a site that would be covered by the crowns of the sampled plants if their foliage were completely opaque. This value was converted into percentage foliage cover by multiplying percentage crown cover by crown type percentage (see Figure 3.4) (McDonald *et al.* 1990), which is based on visual estimates of

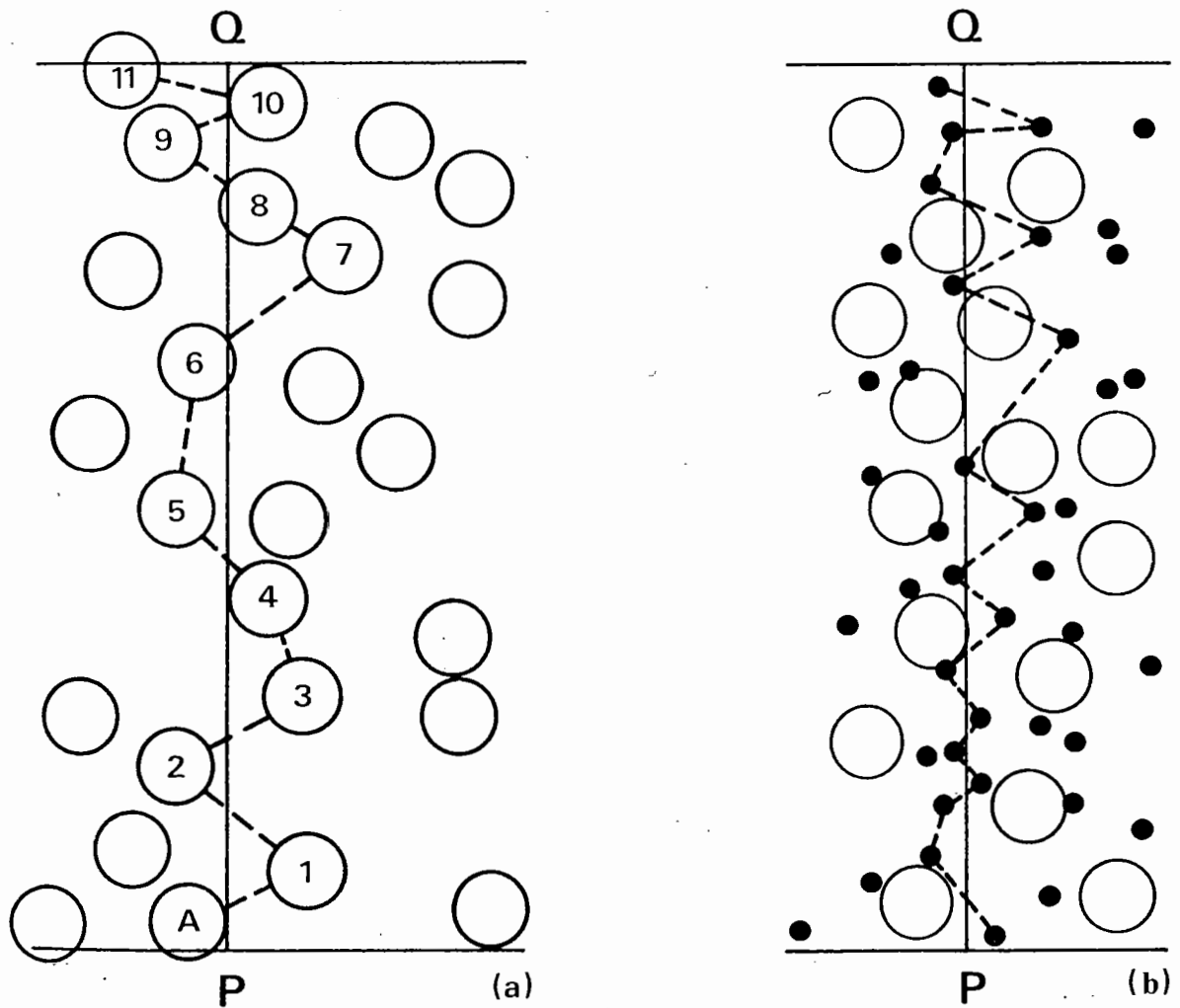


Figure 3.3 - Zig-zag transect used to sample distance between tree and shrub crowns and crown widths (from McDonald *et al.* 1990)

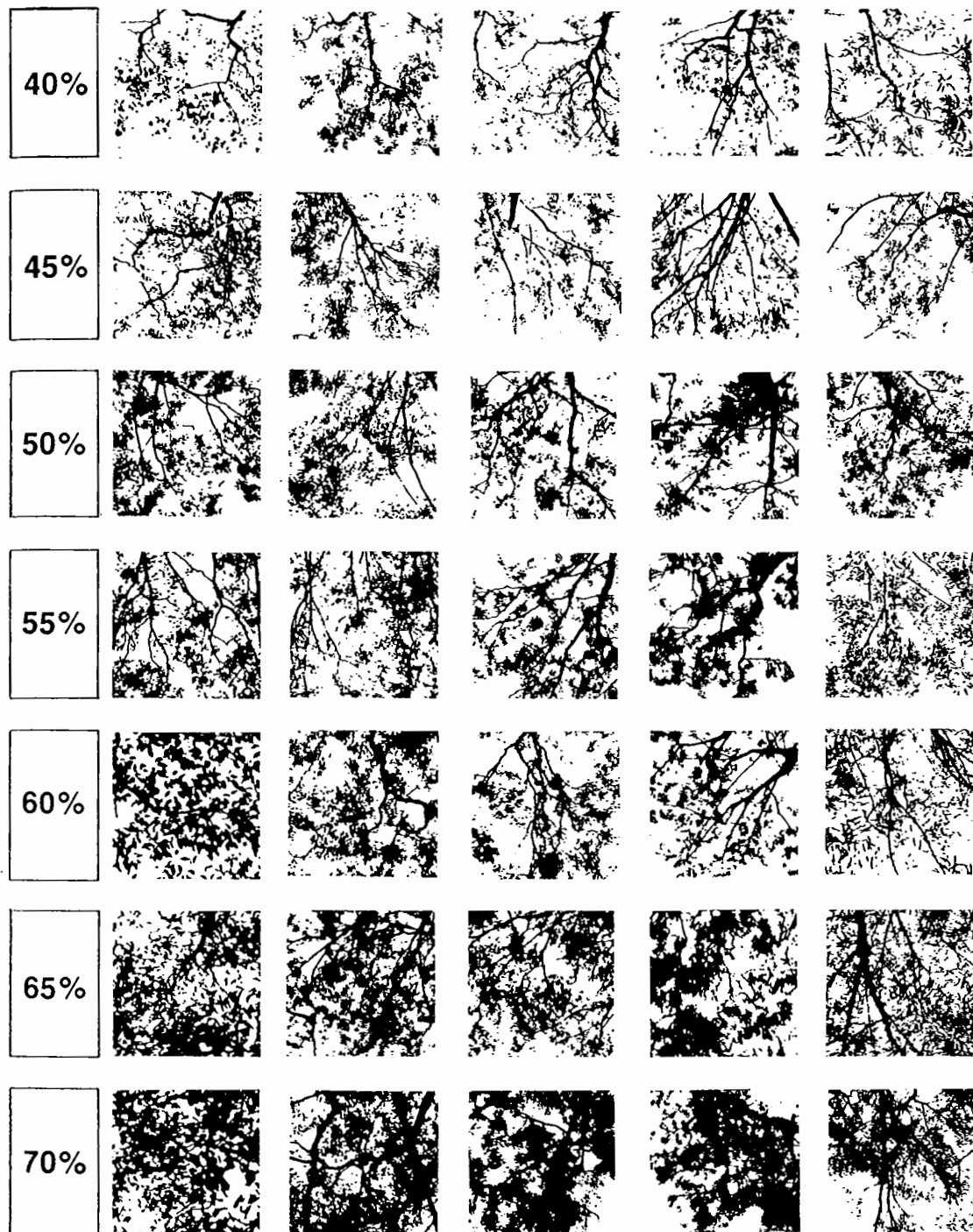


Figure 3.4 - Crown types used to estimate the openness of tree and shrub crowns (from McDonald *et al.* 1990)

the density of different patterns of foliage. Table 3.4 lists the crown type percentage I used for each of the exotic shrub species sampled in this study to convert percentage crown cover into percentage foliage cover.

Table 3.4 – Crown type used to estimate foliage cover for exotic shrubs

Exotic shrub	Crown type
Briar Rose	55%
Blackberry	70%
Cotoneaster	70%
Firethorn	70%
Hawthorn	45%
Privet	45%
All dead shrubs	30%

The method used for the shrub layer of each site was also applied to native trees, with estimates of percentage foliage cover for native trees present in woodland sites derived using crown width, distance to next nearest crown and visual estimates of crown type.

Descriptions of groundcover vegetation for each site were derived from identifying plants in ten to fifteen 1m x 1m quadrats per site and occurred in November 2001. Table 3.5 lists and describes vegetation variables measured or described for each site.

Mean monthly temperature and rainfall data during the survey period were obtained from the Australian Bureau of Statistics publication *Australian Capital Territory in Focus* and are presented in Chapter 1. Other variables describing each of the twelve sites were obtained from field observations and are listed in Table 3.6.

**Table 3.5 - Vegetation variables sampled or described for each site**

Site variable	Description of data item
Exotic shrub cover	Percentage foliage cover
Native shrub cover (native shrubs and trees less than 4 metres high).	Percentage foliage cover
Native tree cover	Percentage foliage cover,
Groundcover	Described.

**Table 3.6 - Other site variables or descriptive features**

Other site variables or descriptive feature
Urban infrastructure in or near (<50m distant) sites.
Description of weed control activity in site.
No. of cats, foxes, and rabbits and hares counted in a 20 minute area-search.
No. of people observed within a site in a 20 minute area search.
No. of dogs observed within a site in a 20 minute area-search and % of dogs not on leads.

These variables were noted to provide supporting information for the bird survey work and to enable consideration of variables other than exotic shrub cover when examining the results of the bird data analysed in Chapter 4.

### 3.6 Statistical analysis

Univariate tests applied included measures of association and analysis of variance. All univariate tests used were non-parametric tests, as the data did not follow a normal distribution and in many cases sample sizes were quite small ( $n < 30$ ) (Siegel and Castellan 1988; Cho 1991).

The approach to testing for statistical association between exotic shrub cover and mean species richness, mean total bird abundance and the mean abundance of individual bird species for the twelve sites sampled entailed two tests of correlation: 1) Spearman rank-order correlation coefficient ( $R_s$ ); and 2) Kendall rank-order correlation coefficient  $T$  (generalised to a partial coefficient) (Siegel and Castellan 1988; Cho 1991).

The Spearman rank-order correlation coefficient ( $R_s$ ) was applied first and was used to determine if mean species richness and mean total bird abundance for each site was associated with exotic shrub cover at each site. This test was also applied to test for correlation between exotic shrub cover at each site and the abundance of individual bird species at each site, entailing multiple tests. Following this step,  $R_s$  coefficients determined as statistically significant ( $\alpha \leq 0.05$ ) were then re-tested with the variable native shrub cover included in the analysis and eliminated as a source of variance by applying the Kendall rank-order correlation coefficient  $T$ , generalised to a partial coefficient (Siegel and Castellan 1988; Cho 1991). This second step was developed to ascertain if native shrub cover was a confounding influence on the associations identified using the Spearman rank-order correlation coefficient ( $R_s$ ).

A Kruskal-Wallis One-Way ANOVA was also used to test if sites with varying levels of exotic shrub cover were statistically different in terms of total bird abundance, abundance of individual bird species and species richness. Sites were blocked into four classes of exotic shrub invasion, based on the percentage foliage cover of exotic shrubs determined for each site. These classes were nil-weed; light weed; moderate weed; and dense weed. Table 3.7 shows the number of sites in each class of exotic shrub invasion and the range of percentage foliage cover of exotic shrub cover corresponding to that class.

Table 3.7 - Key characteristics of exotic shrub invasion classes

Exotic shrub invasion class	No. of sites	% exotic foliage cover (range)
Nil-weed	4	0.09 to 0.39
Light weed	4	0.99 to 2.87
Moderate weed	2	4.5 to 5.37
Dense weed	2	8.30 to 14.38

The Kruskal-Wallis One Way ANOVA was tested separately for each of the ten months of data collected to account for the influence of monthly variations in bird abundance and species richness (Wood 1995).

Multi-variate analysis was undertaken on the mean abundance of individual bird species by site (i.e. the data matrix consisted of the twelve sites and the mean abundance of the 75 bird species recorded in the study). The analysis used was a hierarchical clustering of sites and determination of site groups (Manly 1995). This was done to determine which sites resembled each other in terms of their bird life and to see if the arbitrary exotic shrub classes imposed matched the similarity of sites as determined by comparing their bird life.

The data was untransformed prior to analysis. This decision was based on the need to ensure that sites with higher mean abundance values for some species were given a high weighting in the analysis. The distance measure chosen to determine association of sites was the Kulczynski coefficient (Belbin 1994). Belbin (1994) considered this coefficient to be a good measure of association for ecological problems. Hierarchical clustering was used to group sites (Belbin 1994).

The distance matrix generated using the Kulczynski coefficient was used to construct a three dimensional plot of the sites, showing their relationship based on the mean abundance of their bird life. The method used to develop this plot was non-metric multidimensional scaling (Manly 1995). Interpretation of the three dimensional plot of sites was based on a visual examination of a three dimensional plot of individual bird species and the vectors used to plot individual birds in three dimensions. The raw data showing mean abundance of bird species at each site was also used to help interpret this plot (Appendix 9).

### **3.7 Specific Observations made at site 4 – before and after weed control**

Site 4 was subject to intensive weed control by volunteers from ACT Parkcare in June 1996 and June 1997. Twenty minute area-searches were conducted before and after control of exotic shrubs and exotic shrub cover was also estimated before and after weed control. Table 3.8 summarises the number of 20 minute-area searches completed before and after weed control. The method used to estimate exotic shrub cover in this site before and after weed control was quite different

from the method used for the multi-site comparison, described earlier. Random numbers were used to generate 100 sample points. If any part of an exotic shrub projected over or beneath the sample point, the genus was recorded. The number of times a shrub species was recorded was then expressed as a percentage. No statistical tests are presented here to compare data before and after weed control because of a lack of replication. Selection of bird species for before and after comparisons of abundance were based on observations of these species using the exotic shrubs for food or cover prior to weed control.

**Table 3.8 – Number of 20 minute area-searches undertaken before and after weed control in site 4**

<b>Site 4 level of exotic shrub invasion</b>	<b>No. 20 minute area-searches</b>	<b>Time period</b>
24%	10	19/3/96 to 18/6/96
6%	6	22/6/96 to 18/6/97
< 1%	40 <sup>1</sup> to 65	19/6/97 to 30/6/98

**(1) There were fewer counts for fruit-eating birds because only counts from winter and autumn months were included.**



## Chapter 4

### Results

#### 4.1 Composition of the vegetation at each site

The sites varied in the amount of living and dead shrub cover provided by exotic shrubs (Table 4.1). The composition of exotic shrubs at sites also varied considerably (Table 4.1). Estimates of foliage cover for native tree and shrub species are presented in Table 4.2 and Table 4.3.

Descriptions of groundcover vegetation in the different sites are presented in Appendix 1.

Table 4.1 - Percentage foliage cover estimates of exotic shrubs for each site and arbitrary exotic shrub cover class based on these estimates.

Site No.	BR	BR dead	B	C	C dead	F	F dead	H	PR	Total	Exotic shrub cover site class
1	0.005	0	0	0	0	0	0	0	0.05	0.06	nil
2	0.08	0	0	0	0	0	0.12	0	0.04	0.24	nil
3	0.02	0	0.09	0.07	0	0.1	0	0.001	0.03	0.31	nil
4	0.01	0.02	0	0.11	0	0.1	0.12	0	0.03	0.39	nil
5	0.34	0.14	0	0	0	0.47	0.04	0	0	0.99	light
6	0	0.06	0	0.21	0.5	0.2	0.41	0	0	1.38	light
7	0.53	0	0	0	0	0	0	1.13	0	1.66	light
8	2.47	0	0	0	0	0.23	0	0.13	0.04	2.87	light
9	0	0	3.02	0.11	0	0.25	0	0.06	1.06	4.50	moderate
10	2.45	0	0	0	0	0.64	0	2.28	0	5.37	moderate
11	0.35	0	0	2.9	0	4.41	0	0	0.64	8.30	dense
12	0.06	0	0	7.55	0	6.25	0	0	0.52	14.38	dense

**Key to abbreviations**

BR = Briar Rose (*Rosa rubiginosa*)

B = Blackberry (*Rubus spp*)

C = Cotoneaster (*Cotoneaster spp*)

F = Firethorn (*Pyracantha spp*)

H = Hawthorn (*Crataegus spp*)

PR = Privet (*Ligustrum spp*)

Dead indicates dead shrubs of that genus.

There were differences between sites for attributes other than exotic shrub cover, such as the composition, relative health, and openness of tree cover (Table 4.2). Other differences were in native shrub cover (Table 4.3), ground cover vegetation (Appendix 1), the presence of urban infrastructure (Appendix 2), the presence or absence of weed control before and or during the multi-site comparison (Appendix 3), and the level of use by people and their dogs (Appendix 4).

Table 4.2 - Percentage foliage cover estimates of trees for sites and percentage of eucalypts in poor condition

Site No.	YB	BRG	AB	C	OW	N	Sh	P	Total	% trees in poor condition (1)
1	13.8	2.0	4.6	0	0	0	0	0	20.4	12
2	14.1	0	5.4	0.5	0.2	0	0	0	20.2	6
3	9.0	0	0	0	2.1	0	0	0	11.1	33
4	13.8	0	0	0.7	0.9	0	0.9	0	16.3	20
5	5.9	2.6	0	0	0.4	0	0	0	8.9	21
6	13.8	0.4	0	1.4	0	0	0	0	15.6	25
7	3.9	7.6	2.0	0	0.3	0.9	0	0	14.7	40
8	14.6	1.7	2.6	1.7	0	0.8	0	0	21.4	23
9	21.8	2.9	0	1.1	0.6	0	0	0	26.4	8
10	14.5	2.1	2.3	0	0	0	0	0	18.9	29
11	10.8	3.9	1.3	2.1	0.5	0.7	0	0	19.3	43
12	17.1	0	0	3.4	0.4	0	0	1.8	22.7	9

**Key to abbreviations**

YB = Yellow Box (*E. melliodora*)

BRG = Blakely's Red Gum (*E. blakelyi*)

AB = Apple Box (*E. bridgesiana*)

C = Cootamundra Wattle (*Acacia baileyana*)

OW = Other wattles (Hickory, Silver and Black) (*Acacia spp*)

N = Native Cherry (*Exocarpus cupressiformis*)

Sh = She-oak (*Casuarina stricta*)

P = Pine (*Pinus radiata*)

(1) Eucalypts only, crown type estimate < or = to 50%. Note that 80 % of sampled *E. blakelyi* trees were showing signs of poor condition, compared to 13% of *E. melliodora* trees and 0% of *E. bridgesiana*.

The most obvious differences between sites were that: 1) sites 3, 4, 5, 6 and 7 were sparser and more open than the other sites (9-16 percent foliage tree cover, compared to 19-26 percent foliage tree cover) (Table 4.2); 2) sites 3, 4, 6, 9 and 11 were affected by weed control during or just before the multi-site comparisons began (Appendix 3); 3) sites 7 and 8 were more distant from housing than other sites (see Figure 3.2) and; 4) site 10 was lightly grazed by cattle. All the sites had dead or dying eucalypt trees present, with *E. blakelyi* typically the least healthy of the trees (Table 4.2). Sites with the highest proportion of eucalypt trees (height greater than 4 metres), whose foliage looked sparse and unhealthy (foliage cover class < or = to 50 percent), were sites 3, 7, 10 and 11 (Table 4.2).

Table 4.3 - Percentage foliage cover estimates of native shrubs<sup>1</sup> for sites

Site No.	E	B	C	H	S	W	N	SH	CA	Total
1	0.76	0.06	0.12	0.07	0	0	0	0	0.20	1.21
2	1.23	0	0	0.28	0	0.21	0	0		1.72
3	1.48	0	0.18	0.3	1.65	0	0	0	1.69	5.30
4	3.21	0	0.48	0.11	0	0	0	0.3	0.23	4.33
5	2.23	0	0	0.07	0	0	0.16	0	0	2.46
6	1.30	0	0.74	0.04	0	0	0	0	0.19	2.27
7	0.28	0.03	0	0.11	0	0	0.10	0	0	0.52
8	1.12	0	0.21	0	0	0	0	0	0.31	1.64
9	0.29	0.31	0.24	0.14	0	0	0	0	0.25	1.23
10	1.07	0	0	0	0	0	0	0	0.02	1.09
11	0.45	0.39	0.98	0.05	0	2.65	0	0	0.61	5.13
12	0.79	0	1.45	0	0	0.08	0.29	0	0.29	2.90

**Key to abbreviations**E = *Eucalyptus* spp (*E. melliodora*, *E. blakelyi*, *E. bridgesiana*)B = Black Wattle (*A. decurrens*)C = Cootamundra Wattle (*A. baileyana*)H = Hickory Wattle (*A. falciformis*)S = Silver Wattle (*A. dealbata*)W = Wedge-leaf Wattle (*A. pravissima*)N = Native Cherry (*Exocarpus cupressiformis*)SH = She-oak (*Casuarina stricta*)CA = Cassinias (*Cassinia* spp)

(1) Trees less than 4m tall were classed as shrubs.

**4.3 Birds observed in exotic shrubs, July 1997 to June 1998**

Thirty-two species of birds were observed perched in exotic shrubs in woodland sites (Table 4.4).

This represented 43 percent of all bird species recorded in the sites (Appendix 9).

**4.4 Birds that nest or construct roosts in exotic shrubs**

Five bird species used exotic shrubs for nesting and two species of finch constructed permanent roosts in exotic shrubs (Table 4.5). Most nests were found in *Pyracantha* spp and *Crataegus* spp. The finches were only observed roosting in *Pyracantha* spp.

**Table 4.4 - Birds observed in exotic shrubs in *E. melliodora*-*E. blakelyi* woodland, July 1997 to June 1998**

Common Name	% of counts present in sites <sup>1</sup>	% of counts in exotic shrub cover <sup>2</sup>	BR	B	C	F	H
Crested Pigeon	16.69	9.94	✓			✓	✓
Australian King-Parrot	16.39	2.75			✓	✓	✓
Crimson Rosella	89.47	9.58	✓	✓	✓	✓	✓
Eastern Rosella	49.62	2.73				✓	✓
Superb Fairy-wren	45.11	68.33	✓	✓	✓	✓	✓
White-browed Scrubwren	6.02	62.50		✓	✓	✓	
Speckled Warbler	15.04	45.00	✓	✓	✓	✓	✓
Western Gerygone	1.50	30.00				*✓	✓
Brown Thornbill	12.18	25.93	✓		✓	✓	✓
Buff-rumped Thornbill	19.10	11.02	✓			✓	✓
Yellow-rumped Thornbill	13.08	25.29	✓		✓	✓	✓
Striated Thornbill	13.68	8.79	✓		✓	✓	✓
Yellow Thornbill	00.60	50.00	✓			*✓	
Weebill	53.38	7.04	✓		✓	✓	✓
Noisy Friarbird	23.46	0.64					✓
Noisy Miner	17.90	0.84					✓
White-eared Honeyeater	5.71	10.53	✓		✓	✓	
Eastern Spinebill	1.65	9.09			✓	*✓	
Scarlet Robin	3.91	3.85			*✓		
Golden Whistler	9.32	6.45			✓	✓	✓
Rufous Whistler	3.76	4.00					✓
Grey Shrike-thrush	5.87	2.56				✓	
Grey Fantail	40.90	16.91	✓	✓	✓	✓	✓
Black-faced Cuckoo-shrike	15.79	0.95				✓	
Pied Currawong	46.17	3.91			✓	✓	✓
Red-browed Finch	8.42	83.93	✓		✓	✓	✓
House Sparrow	0.90	33.33				✓	
Mistletoebird	1.05	14.29	✓				
Silvereye	20.30	31.85	✓	✓	✓	✓	✓
Common Blackbird	15.04	45.00		✓	✓	✓	✓
Common Starling	22.71	1.32				✓	
Common Mynah	15.79	0.95				✓	

**Abbreviations**

✓ = observed in

BR = Briar Rose (*Rosa rubiginosa*)

B = Blackberry (*Rubus fruticosus*)

C = Cotoneaster (*Cotoneaster spp*)

F = Firethorn (*Pyracantha spp*)

H = Hawthorn (*Crataegus spp*)

\*Dead - indicates dead shrubs of that species.

(1) Expressed as the number of twenty minute area searches where a bird species was present over the total number of twenty minute area searches conducted (n=665).

(2) Expressed as the number of twenty-minute area searches where a bird species was observed in exotic shrubs over the total number of twenty minute area searches that species was present.

Table 4.5 - Birds that nested or constructed permanent roosts in exotic shrubs in *E.melliodora-E.blakelyi* woodland

Common Name	Exotic shrub	Evidence
Crested Pigeon	<i>Pyracantha spp</i>	active nest
Superb Fairy-wren	<i>Pyracantha spp &amp; Crataegus spp</i>	disused nests
Grey Fantail	<i>Rosa spp</i>	active nest
Red-browed Finch	<i>Pyracantha spp</i>	*active roosts
Double-barred Finch	<i>Pyracantha spp</i>	*active roosts
Silvereye	<i>Crataegus spp</i>	disused nest
Common Blackbird	<i>Crataegus spp</i>	disused nest

\*Both finches constructed roosts in *Pyracantha spp* and immature birds of both species were observed in them, however I was unable to inspect the roosts closely enough to confirm breeding.

#### 4.5 Birds observed eating fruit or seeds from exotic shrubs

Seven species of birds were observed eating the seeds or fruit of exotic shrubs (Table 4.6). This behavior was common for the Crimson Rosella (*Platycercus elegans*), Silvereye (*Zosterops lateralis*) and Pied Currawong (*Strepera graculina*), and for the Gang-gang Cockatoo (*Callocephalon fimbriatum*) in 1996 only. The King Parrot (*Alisterus scapularis*) and Eastern Rosella (*Platycercus eximius*) were seen consuming seeds on only a few occasions. The Common Blackbird (*Turdus merula*) was seen feeding on *Pyracantha spp* berries twice, while the Black-faced Cuckoo Shrike (*Coracina novaehollandiae*) was observed feeding on fruit from *Pyracantha spp* on only one occasion (Table 4.6).

#### 4.6 Correlation between bird abundance and exotic shrub cover

The mean abundance of nine bird species in woodland sites was found to correlate with estimates of exotic shrub cover (Spearman's Rank-Order Correlation, Table 4.7). Eight of these correlations were positive and one was negative (Table 4.7). These associations were also significant ( $p \leq 0.05$ ) using the Kendall Partial Rank-Order correlation to test for association, with native shrub cover included in the analysis (Table 4.7).

**Table 4.6 - Birds observed consuming fruit or seeds of exotic shrubs in *E. melliodora-E. blakelyi* woodland, March 1996 to June 1998<sup>1</sup>.**

Common Name	Fruit/seeds of exotic woody shrubs consumed	Season(s) observed feeding on fruit/seeds of exotic shrubs
Gang-Gang Cockatoo <sup>2</sup>	F	winter, autumn
Australian King-Parrot	C, F, H.	winter
Crimson Rosella	BR, C, F, H.	winter, spring, summer, autumn
Eastern Rosella	C, F, H.	winter, spring, summer, autumn
Black-faced Cuckoo Shrike	F.	winter <sup>3</sup>
Pied Currawong	C, F.	winter, autumn
Silvereye	C, F, H.	winter, summer
Blackbird	F.	winter <sup>3</sup>

**Key to abbreviations**

**BR** = Briar Rose (*Rosa spp*)

**C** = Cotoneaster (*Cotoneaster spp*)

**F** = Firethorn (*Pyracantha spp*)

**H** = Hawthorn (*Crataegus spp*)

(1) All listed birds, with the exception of the Black-faced Cuckoo Shrike, were observed eating seeds or fruit in site 4 before weed control in June 1996.

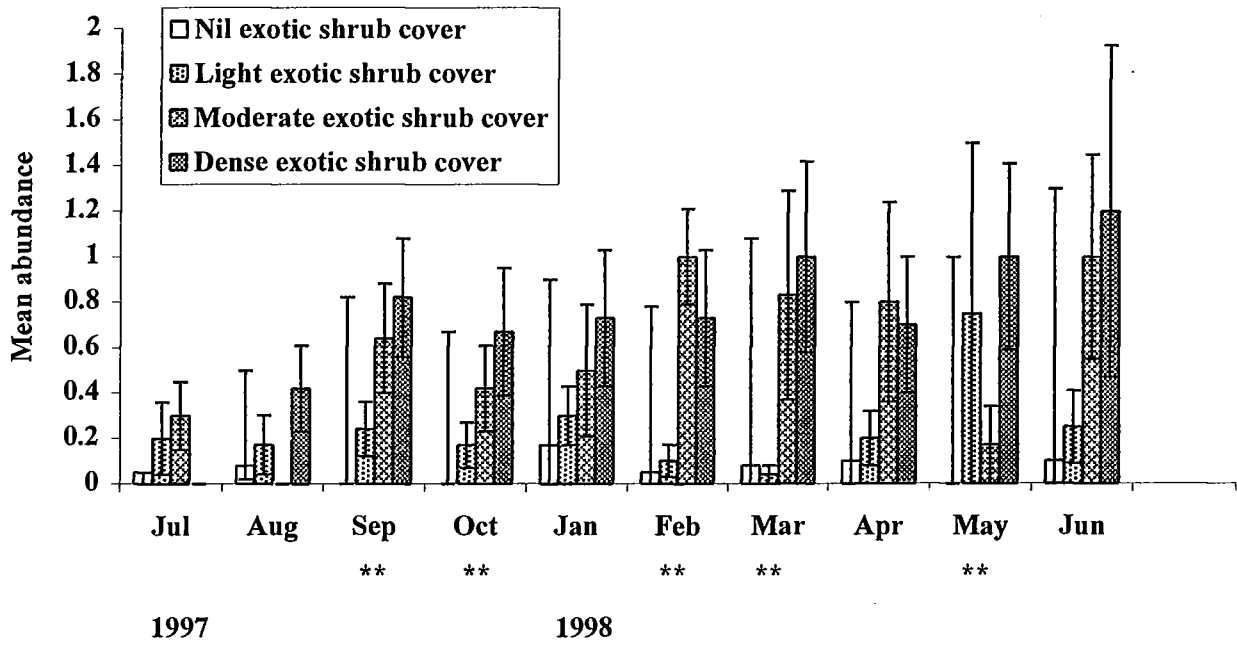
(2) Observed consuming seeds only in 1996.

(3) One occasion only for the Black-faced Cuckoo Shrike, two observations of the Blackbird.

## 4.7 Observations on the major groups of birds

### 4.7.1 Family Columbidae (Pigeons and Doves)

Two species of pigeon recorded in woodland sites were the Common Bronzewing (*Phaps chalcoptera*) and the Crested Pigeon (*Ocyphaps lophotes*) (Appendix 5). The Crested Pigeon was most abundant in *E.melliodora-E.blakelyi* sites classed as having moderate and dense levels of exotic shrub cover (Figure 4.1, Appendix 5). The abundance of the Crested Pigeon was determined as significantly different between exotic shrub cover classes in September,



**Figure 4.1 - Mean monthly abundance and standard error of the Crested Pigeon in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.

October, February, March and May (Figure 4.1). Site 11 had the highest mean abundance (Figure 4.2). The Crested Pigeon was observed nesting in *Pyracantha spp.* (Table 4.5) in site 11 and was observed in exotic shrub cover in approximately 10 percent of the 20 minute-area searches where this bird was present (Table 4.4). Mean abundance for the Crested Pigeon was positively correlated with increasing exotic shrub cover ( $R_s = 0.61$ ,  $p \leq 0.04$ , Table 4.7).

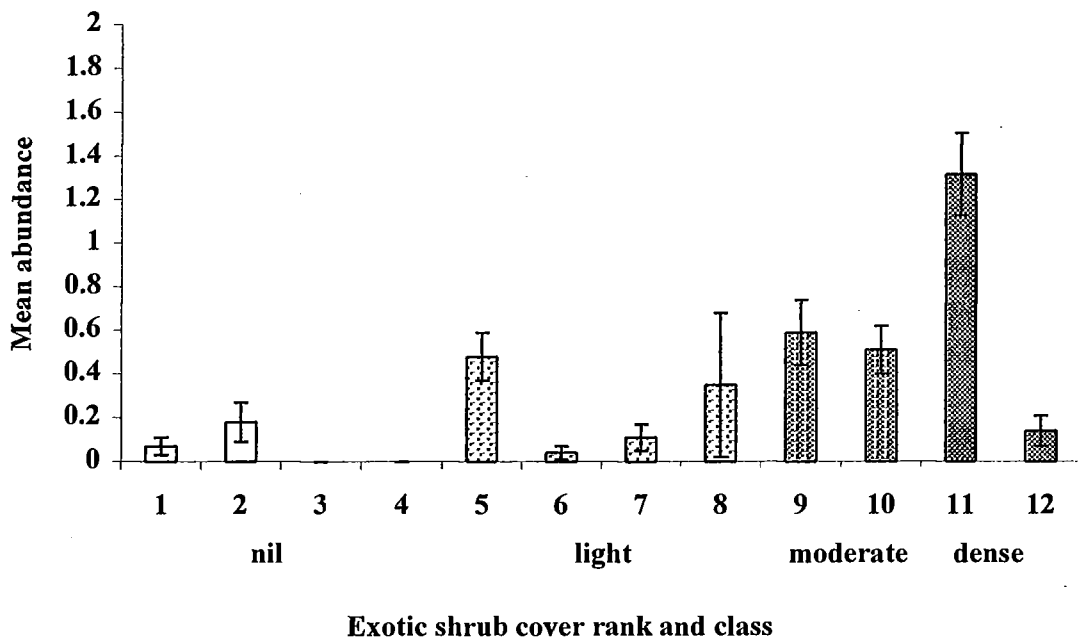
**Table 4.7 – Birds with a significant correlation with estimates of exotic shrub percentage foliage cover in *E.melliodora-E.blakelyi* woodland sites<sup>1</sup>**

COMMON NAME	Correlation coefficient (R <sub>s</sub> )	Probability for R <sub>s</sub>	Partial Correlation coefficient (Kendalls tau-b) <sup>2</sup>
Crested Pigeon	0.61	≤ 0.0368	0.47
Laughing Kookaburra	- 0.63	≤ 0.0315	- 0.49
Common Blackbird	0.72	≤ 0.0084	0.59
Golden Whistler	0.79	≤ 0.0021	0.69
Grey Fantail	0.81	≤ 0.0014	0.63
Superb Fairy-wren	0.67	≤ 0.0168	0.45
Speckled Warbler	0.78	≤ 0.0030	0.62
Brown Thornbill	0.74	≤ 0.0060	0.55
Red-browed Finch	0.61	≤ 0.0369	0.47

(1) Correlations were calculated on the twelve mean values for each species at each site (i.e. n=12).

(2)  $P \leq 0.05$  (Siegel and Castellan 1988, Table S, p 364).





**Figure 4.2 - Mean abundance and standard error for the Crested Pigeon in *E. melliodora-E. blakelyi* woodland ranked by exotic shrub cover (lowest to highest, all months combined)**

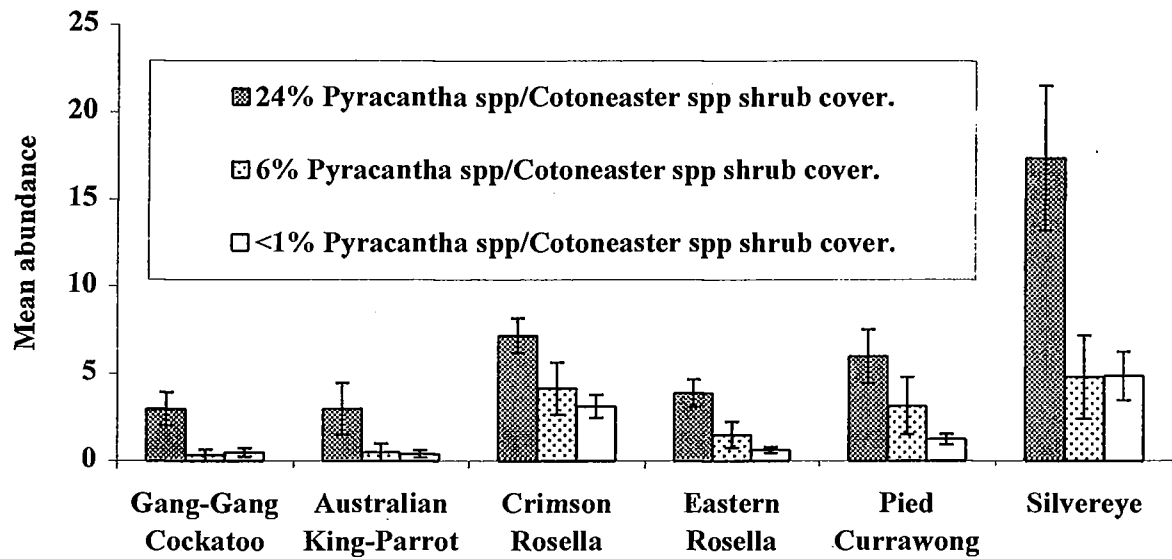
#### 4.7.2 Family *Cacatuidae* (Cockatoos, Corellas and Cockatiel)

The Gang-gang Cockatoo (*Callocephalon fimbriatum*), the Galah (*Cacatua roseicapilla*) and Sulphur-crested Cockatoo (*Cacatua galerita*) were recorded at woodland sites between July 1997 and June 1998 (Appendix 5). The Gang-gang Cockatoo was recorded consuming berry seeds from *Pyracantha spp* shrubs in the autumn and winter months of 1996 (Table 4.6). It was less abundant in winter and autumn in site 4 following weed control of exotic shrubs in this site (Figure 4.3). Between July 1997 and June 1998 the Gang-gang Cockatoo was not observed consuming berries from exotic shrubs in woodland sites (Table 4.6). Mean abundance for the Gang-Gang Cockatoo was similar for sites with no exotic shrubs and with dense exotic shrub cover between July 1997 and June 1998 (Appendix 5). The other two cockatoo species were not recorded in exotic shrubs (Table 4.4) and there was little evidence that exotic shrub cover influenced their abundance (Appendix 5).

#### 4.7.3 Family *Psittacidae* (Parrots, Rosellas, Ringnecks and Budgerigars)

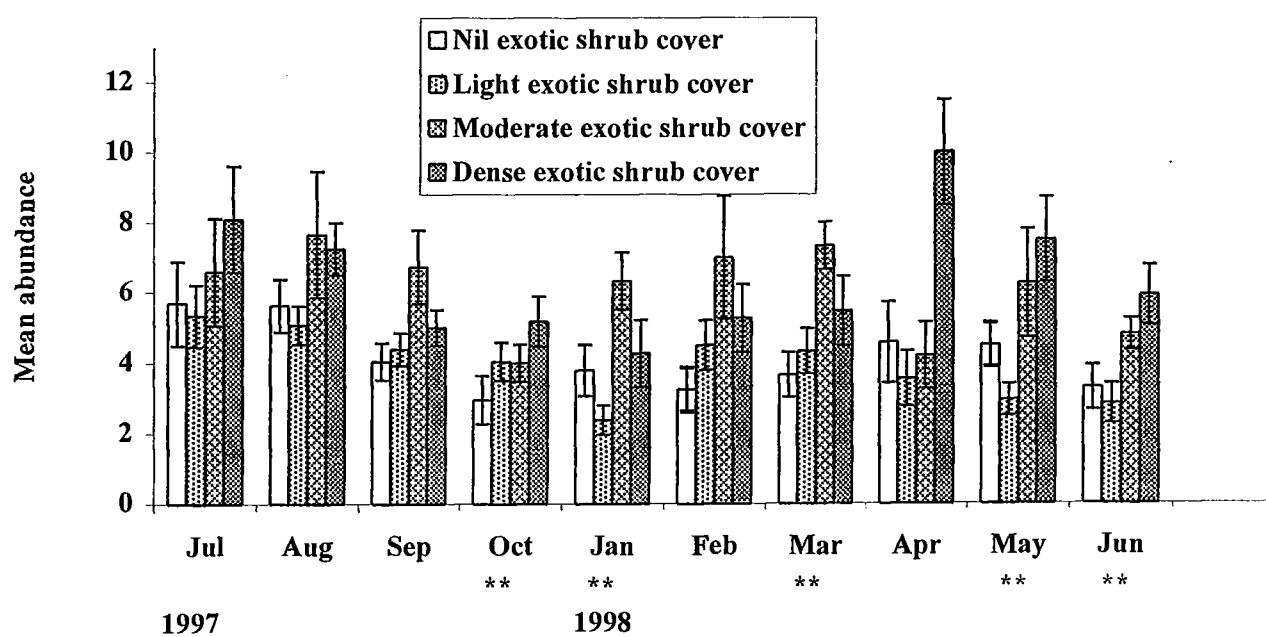
The Australian King-Parrot (*Alisterus scapularis*), the Crimson Rosella (*Platycercus elegans*), Eastern Rosella (*Platycercus eximius*) and Red-rumped Parrot (*Psephotus haematonotus*) were recorded at woodland study sites (Appendix 5). Species observed consuming the seeds of exotic shrubs during the study period were the Australian King-Parrot and both Rosellas (Table 4.6). The Eastern Rosella was most abundant in sites classified as having nil exotic shrub cover, whereas the Crimson Rosella was most abundant in sites with dense exotic shrub cover (Appendix 5).

Increases in abundance in autumn and winter in sites with dense exotic shrub cover were recorded for the Crimson Rosella (Figure 4.4). These seasonal rises in abundance corresponded with feeding observations in exotic shrubs (Table 4.6). The abundance of the Crimson Rosella was determined as being significantly different between the four exotic shrub cover classes in October, January, March, May and June (Figure 4.4). The Eastern Rosella was most abundant in woodland sites with dense exotic shrub cover in July 1997 (Figure 4.5). The abundance of the Eastern Rosella was significantly different between the four exotic shrub cover classes in September, March and June (Figure 4.5). The Eastern Rosella was less frequently recorded in exotic shrub cover than the Crimson Rosella (Table 4.4). The mean abundance of the Eastern and



**Figure 4.3 - Mean abundance of selected frugiverous birds, following weed control(i) of *Pyracantha spp* and *Cotoneaster spp*, site 4, Canberra Nature Park, Red Hill (autumn and winter months only)**

(1) Site 4 was intensively weeded twice: prior to weed control exotic shrub cover was approximately 24%; intensive weeding on a single day reduced this to about 6% in June 1996 and ; more weed control reduced this to less than 1% in June 1997 (see methods for more detail concerning the timing of bird sampling).



**Figure 4.4 - Mean monthly abundance and standard error of the Crimson Rosella in *E. melliodora*-*E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) between exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.

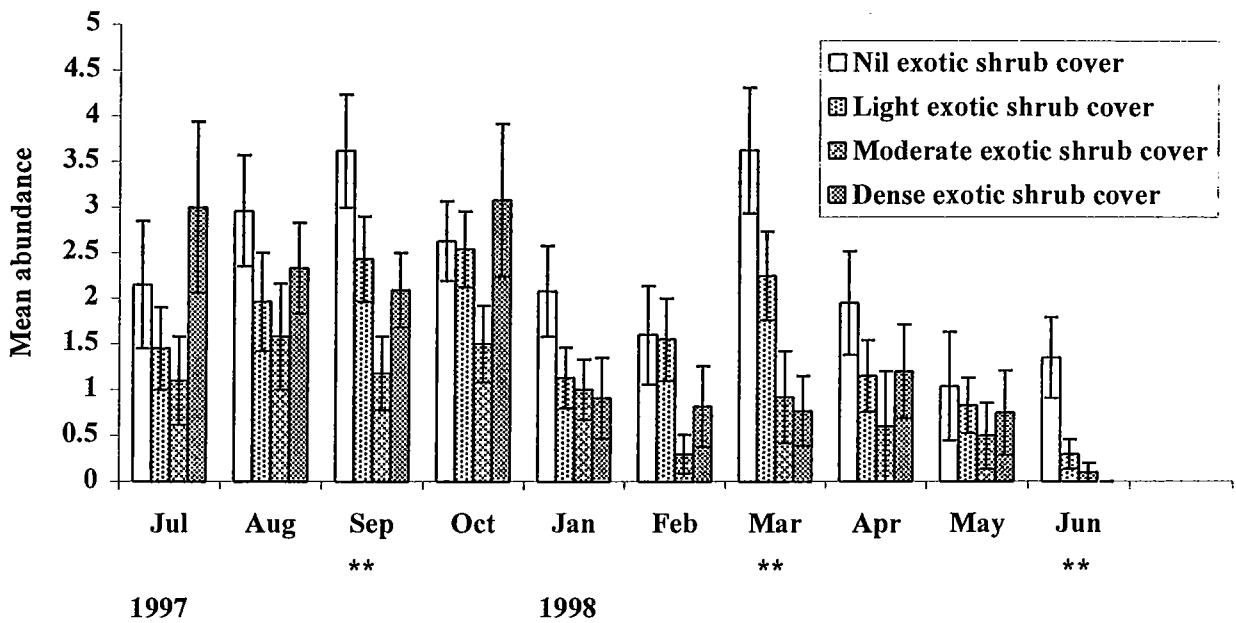


Figure 4.5 - Mean monthly abundance and standard error of the Eastern Rosella in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover

\*\* Mean monthly abundance significantly different ( $P < 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.

Crimson Rosella was lower in autumn and winter following weed control of exotic shrubs in site 4 in 1996 and 1997 (Figure 4.3).

The mean abundance of the Australian King-Parrot was highest at woodland sites with nil exotic shrub cover (Appendix 5). This species was less abundant in site 4 following weed control in 1996 and 1997 (Figure 4.3).

#### 4.7.4 Family Halcyonidae (Tree Kingfishers)

The Laughing Kookaburra (*Dacelo novaeguineae*) and the Sacred Kingfisher (*Todiramphus sanctus*) were recorded during the study (Appendix 5), although there were few records for the Sacred Kingfisher (Appendix 5). The Laughing Kookaburra was not observed in exotic shrub cover (Table 4.4). Mean abundance by site suggests this bird is less abundant in woodland with dense levels of exotic shrub cover, although mean abundance was low in all sites (Appendix 5). Mean abundance for the Laughing Kookaburra was negatively correlated with increasing levels of exotic shrub cover ( $R_s = -0.63$ ,  $p \leq 0.04$ , Table 4.7).

#### 4.7.5 Family Maluridae (Fairy-wrens, Emu-wrens and Grasswrens)

The Superb Fairy-wren (*Malurus cyaneus*) was present at woodland sites (Appendix 6) and was frequently observed in exotic shrubs (Table 4.4). The disused nests of this species were located in *Crataegus spp* and *Pyracantha spp* (Table 4.5). The Superb Fairy-wren was least abundant in sites classified as having nil exotic shrub cover and this pattern was fairly consistent between July 1997 and June 1998 (Figure 4.6). The highest mean abundance of Superb Fairy-wrens occurred at sites with moderate and light exotic shrub cover (Appendix 6), with the highest mean abundance being at sites 7 and 10 (Figure 4.7). The abundance of the Superb Fairy-wren differed significantly between the four exotic shrub cover classes in all months sampled (Figure 4.6). The mean abundance for the Superb Fairy-wren was lower following weed control of *Pyracantha spp* and *Cotoneaster spp* in site 4 in 1996 and 1997 (Figure 4.8). The mean abundance of the Superb Fairy-wren was positively correlated with exotic shrub cover ( $R_s = 0.67$ ,  $p \leq 0.02$ , Table 4.7).

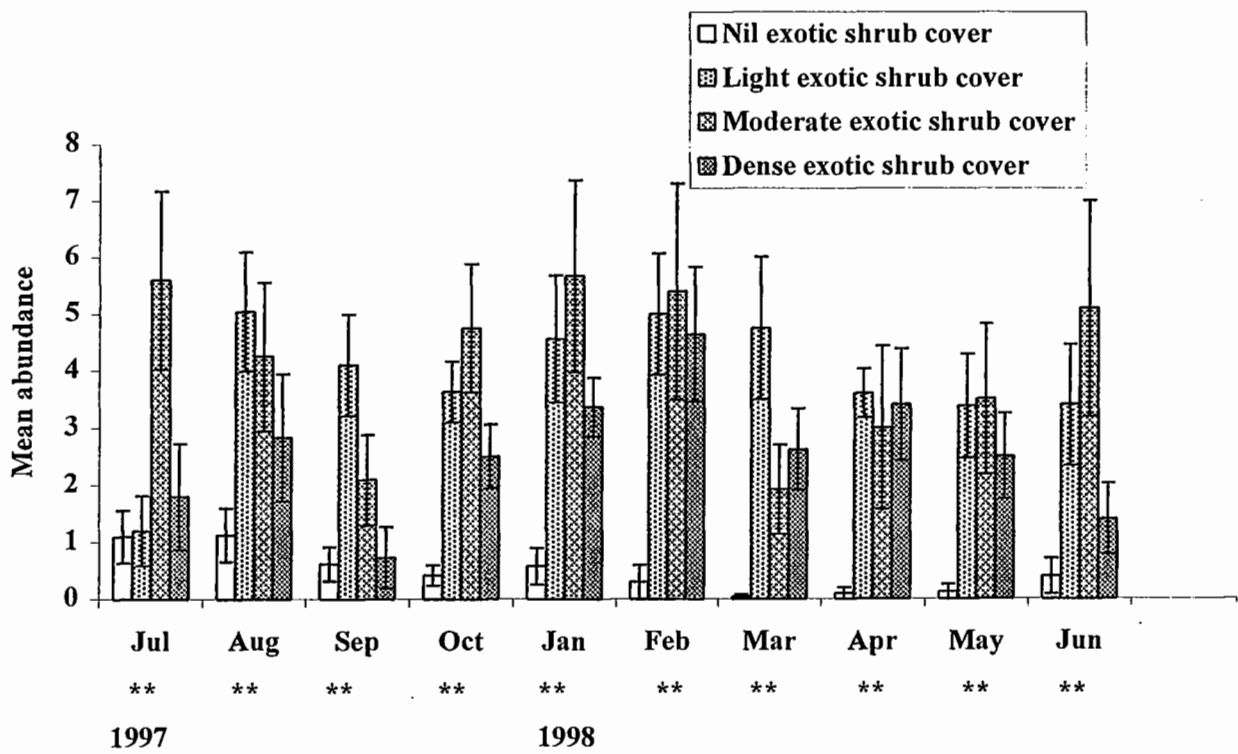
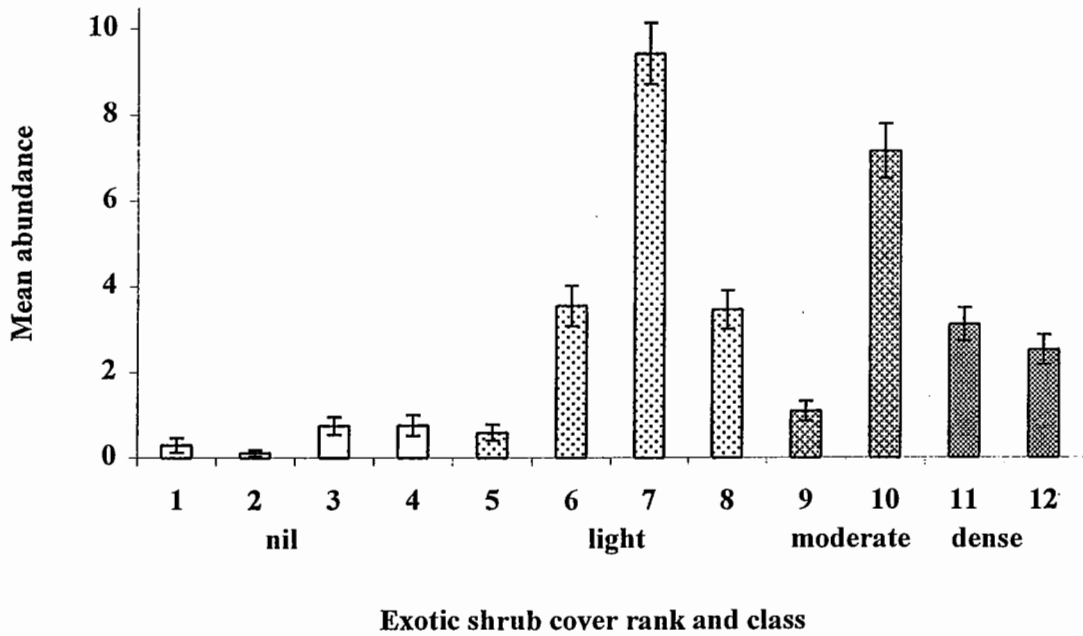


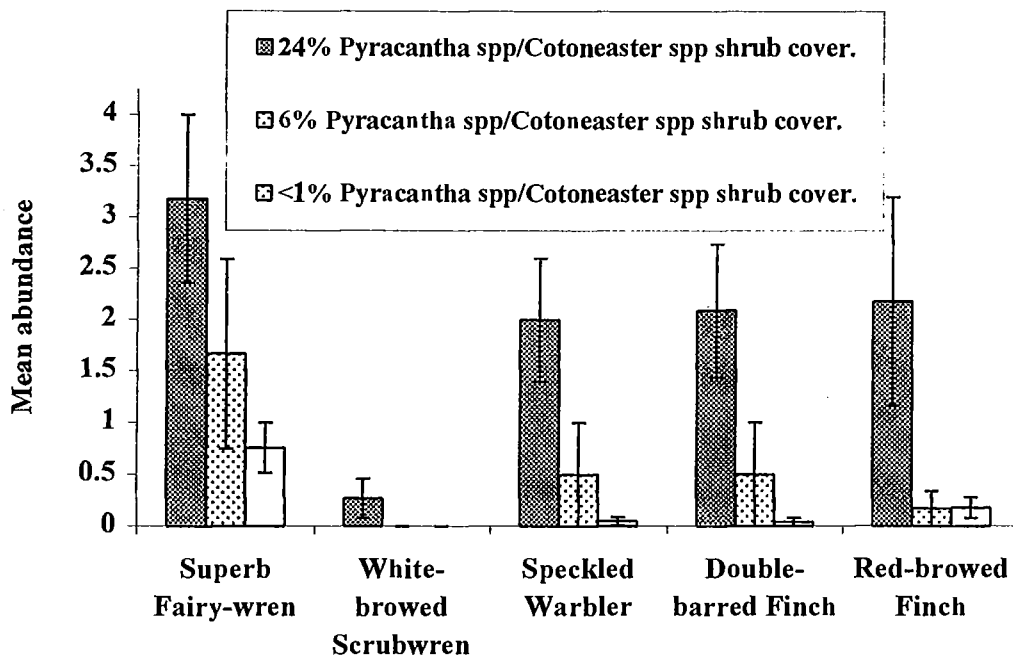
Figure 4.6 - Mean monthly abundance and standard error of the Superb Fairy-wren in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover

\*\* Mean monthly abundance significantly different ( $P < 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



**Figure 4.7 - Mean abundance and standard error for the Superb Fairy-wren in *E. melliodora-E. blakelyi* woodland ranked by level of exotic shrub cover (lowest to highest, all months combined)**





**Figure 4.8 - Mean abundance of selected small birds following weed control<sup>(1)</sup> of *Pyracantha* spp and *Cotoneaster* spp, site 4, Canberra Nature Park, Red Hill**

(1) Site 4 was intensively weeded twice: prior to weeding exotic shrub cover was approximately 24%; intensive weeding on a single day reduced this to 6% in June 1996 and ; more weed control reduced this to less than 1% in June 1997 (see methods for more detail concerning the timing of bird sampling).

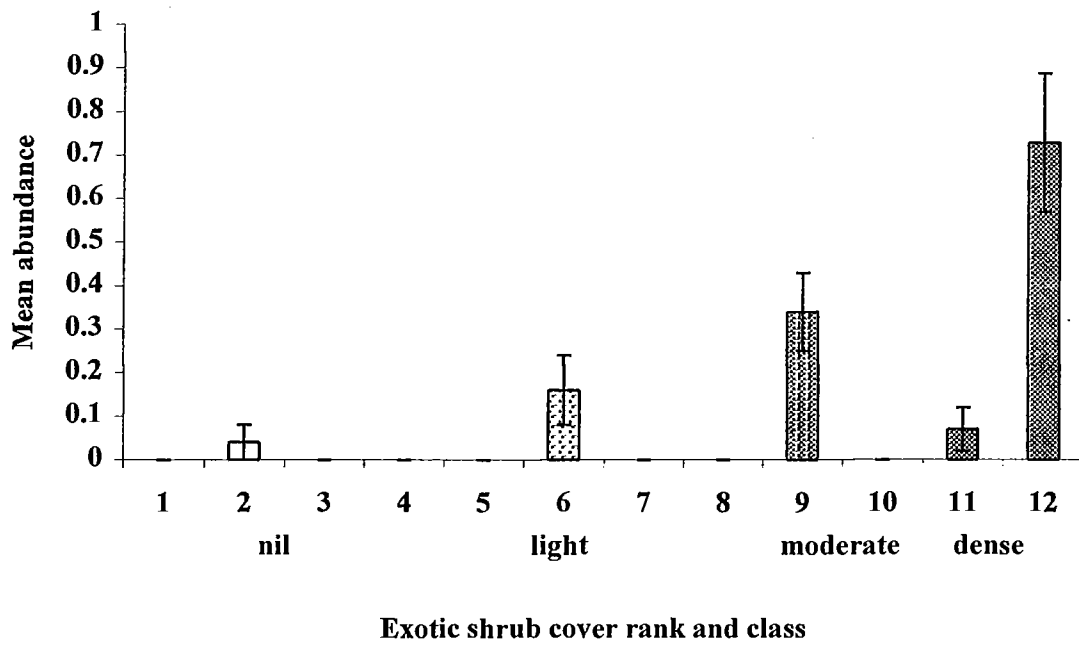
4.7.6 Family Pardalotidae (*Pardalotes*, Bristlebirds, Sandstone Warblers, Scrubwrens, Heathwrens, Fieldwrens, Gerygones, Thornbills and Whitefaces)

Twelve species belonging to the family Pardalotidae were recorded at woodland study sites (Appendix 6). Nine of these birds were recorded in exotic shrub cover: the White-browed Scrubwren (*Sericornis frontalis*); the Speckled Warbler (*Chthonicola sagittata*); Western Gerygone (*Gerygone fused*); all five thornbill species; and the Weebill (*Smicrorhis brevirostris*) (Table 4.4).

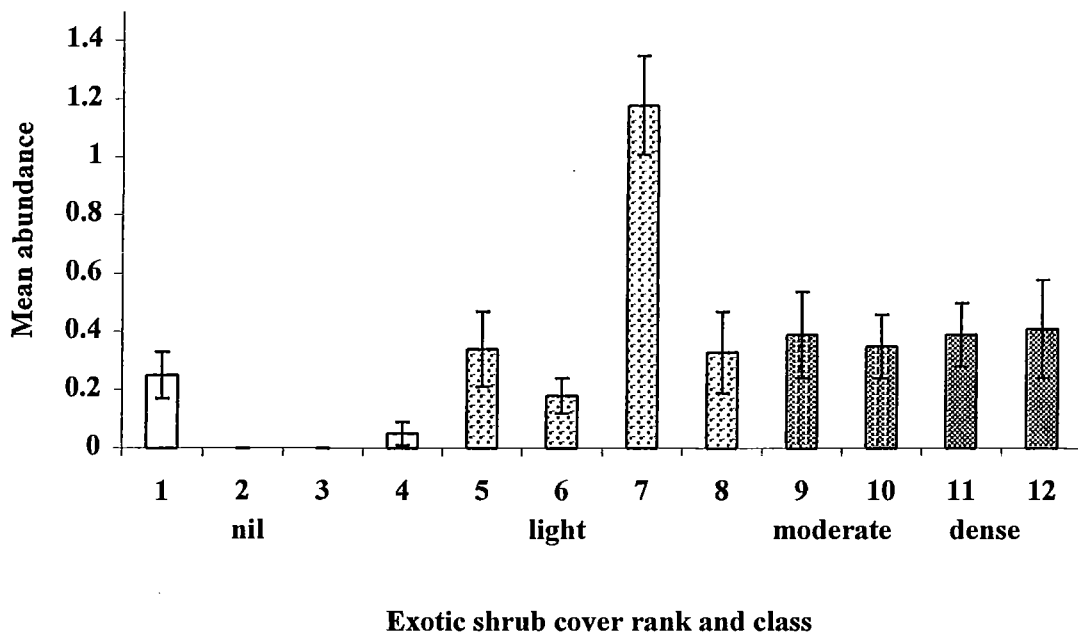
The mean abundance of the White-browed Scrubwren was low in woodland sites (Appendix 6). When observed, the scrubwren was commonly seen in *Rubus spp*, *Cotoneaster spp* and *Pyracantha spp* (Table 4.4). The White-browed Scrubwren was most abundant in sites with dense exotic shrub cover (Appendix 6). This species was observed in only five of the twelve sites and was most abundant in site 12 (Figure 4.9). Following weed control in site 4 in 1996 and 1997, the White-browed Scrubwren was not seen again in this site (Figure 4.8).

The Speckled Warbler was observed in *Rosa spp*, *Rubus spp*, *Cotoneaster spp*, *Pyracantha spp* and *Crataegus spp* (Table 4.4). This species showed a preference for sites with some exotic shrub cover (light to dense), with the lowest mean abundance generally recorded in woodland with an absence of exotic shrubs (Figure 4.10). Mean abundance was highest in the winter months of 1997 and generally lower in the spring months of 1997 (Figure 4.11). There was a significant difference in the abundance of the Speckled Warbler between the four exotic shrub cover classes in July, August and September (Figure 4.11). The highest mean abundance was recorded at site 7 (Figure 4.10). The mean abundance for the Speckled Warbler in site 4 was lower following weed control of exotic shrubs in 1996 and 1997 (Figure 4.8). The mean abundance of the Speckled Warbler was positively correlated with increasing exotic shrub cover ( $R_s = 0.78$ ,  $p \leq 0.003$ , Table 4.7).

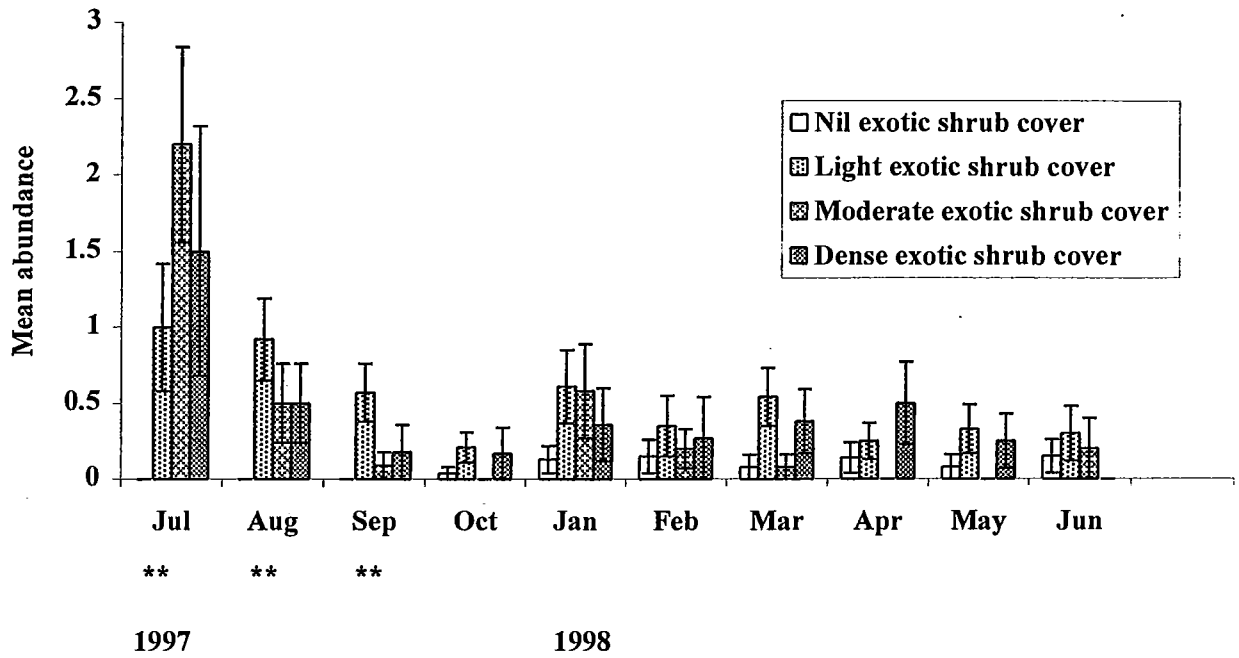
The Buff-rumped Thornbill (*Acanthiza reguloides*) and Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*) were observed in exotic shrubs (Table 4.4). The Buff-rumped Thornbill was widespread and abundant in all woodland sites, showing no clear preference for woodland with different levels of exotic shrub cover (Appendix 6). The Yellow-rumped Thornbill was most abundant in sites with light exotic shrub cover (Appendix 6). The Brown Thornbill (*Acanthiza*



**Figure 4.9 - Mean abundance and standard error for White-browed Scrub-wren in *E. melliodora*-*E. blakelyi* woodland ranked by level of exotic shrub cover (lowest to highest, all months combined)**



**Figure 4.10 - Mean abundance and standard error of the Speckled Warbler in *E. melliodora*-*E. blakelyi* woodland ranked by exotic shrub cover (lowest to highest, all months combined)**



**Figure 4.11 - Mean monthly abundance and standard error of the Speckled Warbler in *E. melliodora*-*E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.

*pusilla*) was observed in exotic shrub cover (Table 4.4) and was most abundant in woodland with dense levels of exotic shrub cover (Appendix 6). The highest mean abundance for the Brown Thornbill was recorded at site 11 (Figure 4.12). Mean abundance for the Brown Thornbill peaked in woodland with dense levels of exotic shrub cover in July 1997 and October 1997, with abundance generally lower in all sites in 1998 (Figure 4.13). The abundance of the Brown Thornbill differed significantly between the four exotic shrub cover classes in August, October, February and March (Figure 4.13). Mean abundance for the Brown Thornbill was positively correlated with increasing exotic shrub cover ( $R_s = 0.74$ ,  $p \leq 0.006$ , Table 4.7).

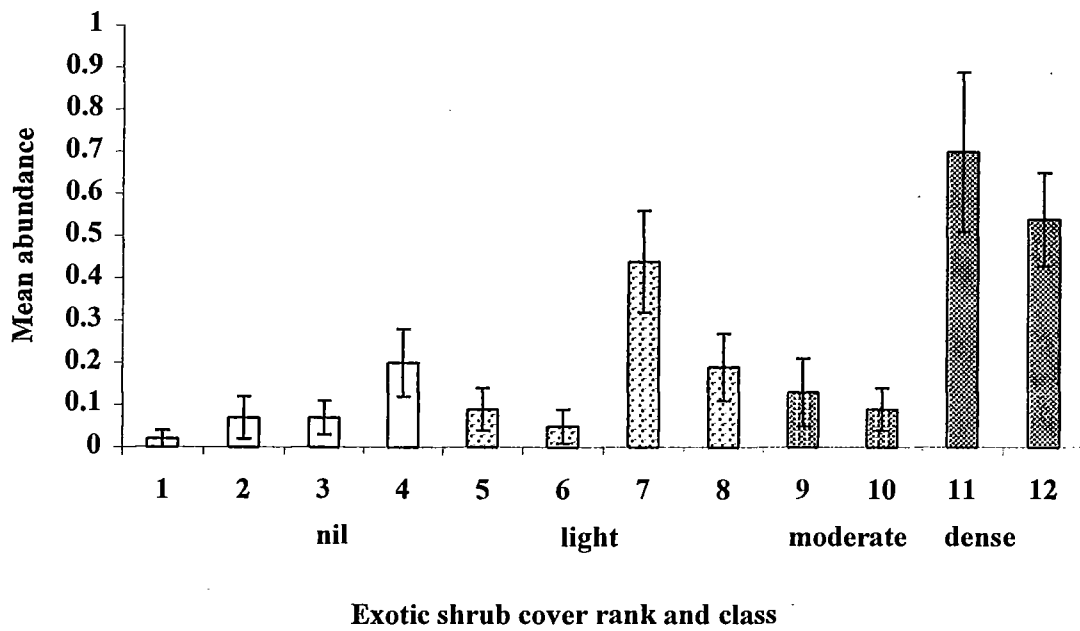
The Striated Thornbill (*Acanthiza lineata*), Yellow Thornbill (*Acanthiza nana*), Weebill (*Smicrornis brevirostris*) and Western Gerygone (*Gerygone fusca*) were observed in exotic shrubs (Table 4.4). The Weebill and Striated Thornbill were most abundant in woodland sites with light levels of exotic shrub cover (Appendix 6), while the Yellow Thornbill and Western Gerygone were rarely seen in the study sites (Appendix 6).

#### 4.7.7 Family Meliphagidae (Honeyeaters)

Nine species of honeyeater were recorded at the study sites (Appendix 6), with four of these observed in exotic shrubs (Table 4.4). The Eastern Spinebill (*Acanthorhynchus tenuirostris*) was infrequently recorded (Appendix 6) but was more abundant in sites with dense exotic shrub cover (Figure 4.14). The Noisy Miner (*Manorina melanocephala*) generally had a higher mean abundance in woodland which had an absence of exotic shrub cover (Figure 4.15). This species was not recorded in woodland with dense exotic shrub cover (Figure 4.15). The abundance of the Noisy Miner differed significantly between the four exotic shrub cover classes in July, October, January and March (Figure 4.15).

#### 4.7.8 Family Pachycephalidae (Shrike-tits, Crested Bellbird, Whistlers and Shrike-thrushes)

Birds from the Pachycephala family observed in woodland sites were the Golden Whistler (*Pachycephala pectoralis*), Rufous Whistler (*Pachycephala rufiventris*) and Grey Shrike Thrush (*Colluricincla harmonica*) (Appendix 7). All three species were reported in exotic shrubs (Table 4.4). The Golden Whistler and Rufous Whistler were both more abundant at sites 7 and 10, although the Golden Whistler was most abundant at site 12. (Figure 4.16).



**Figure 4.12- Mean abundance and standard error for the Brown Thornbill in *E. melliodora-E. blakelyi* woodland ranked by level of exotic shrub cover (lowest to highest, all months combined)**

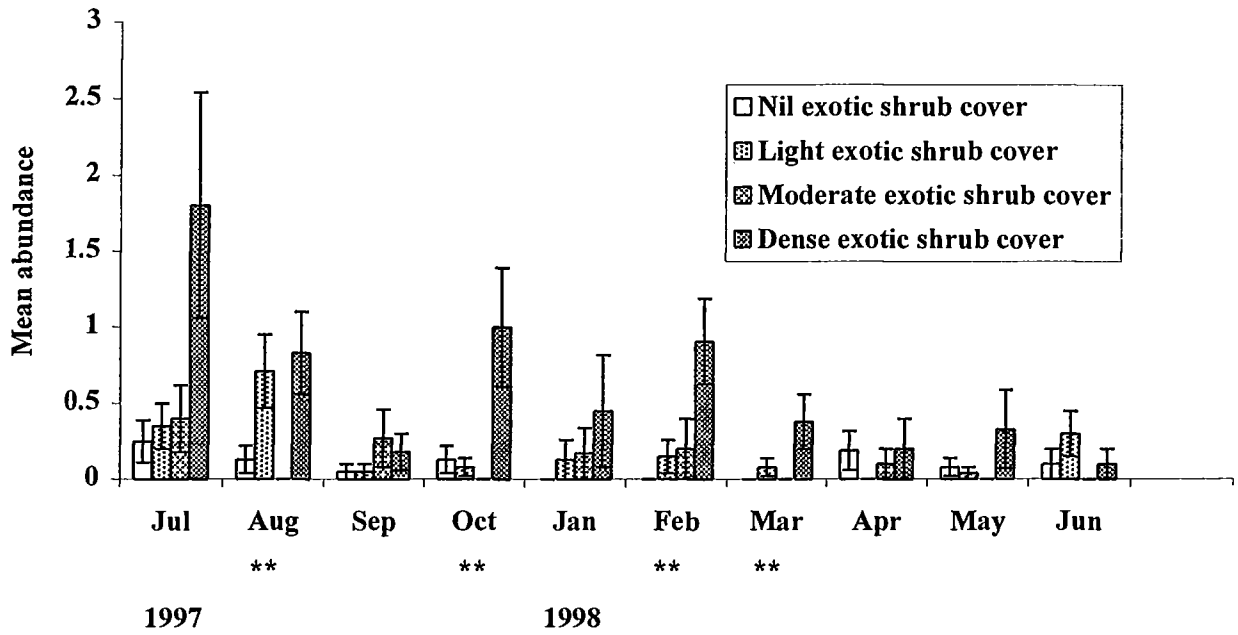


Figure 4.13 - Mean monthly abundance and standard error of the Brown Thornbill in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



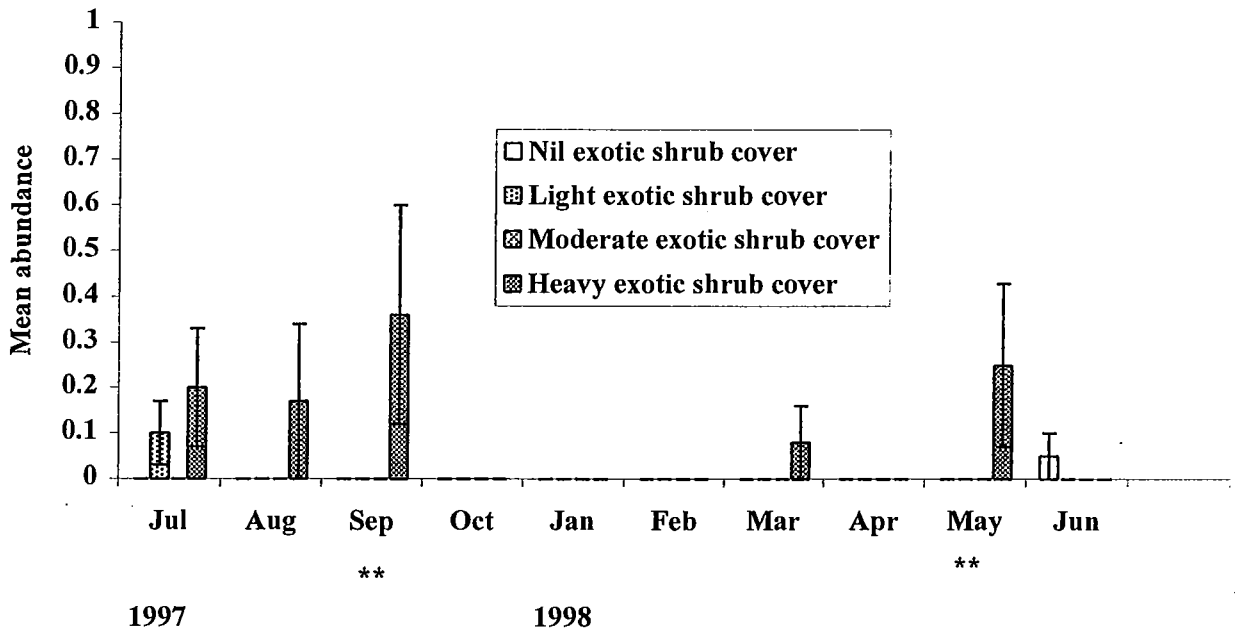
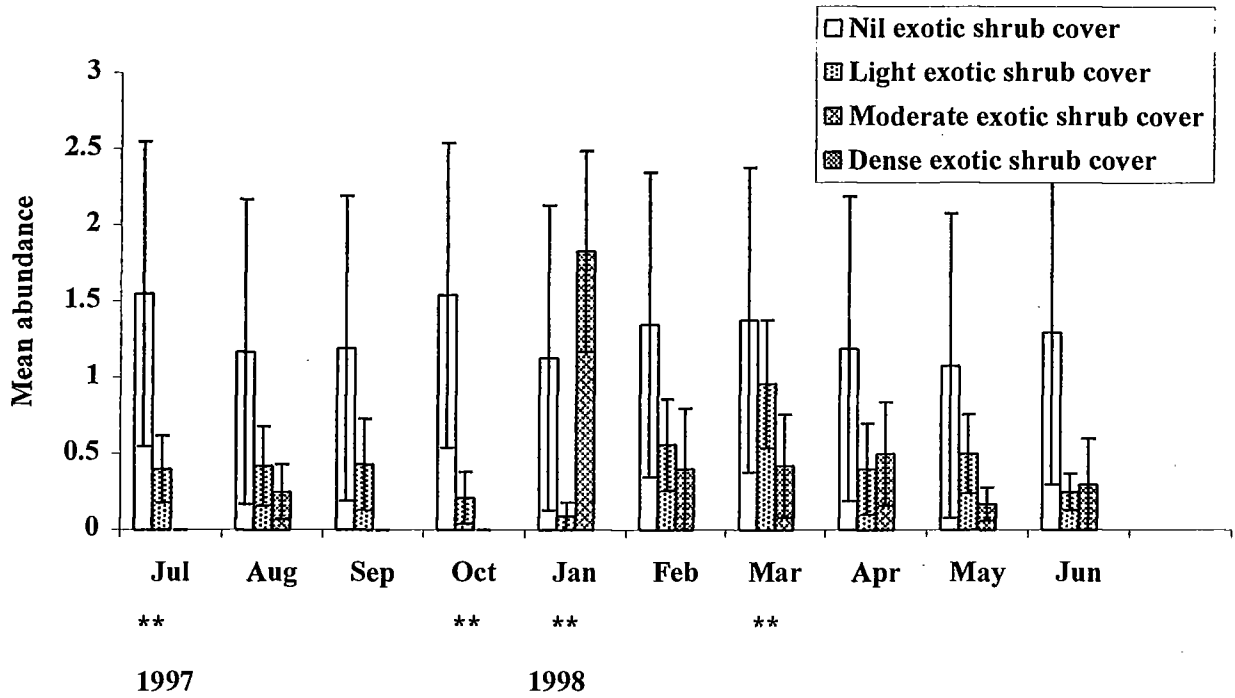


Figure 4.14 - Mean monthly abundance and standard error of the Eastern Spinebill in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover

\*\* Mean monthly abundance significantly different ( $P \leq 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



**Figure 4.15 - Mean monthly abundance and standard error of the Noisy Miner in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.

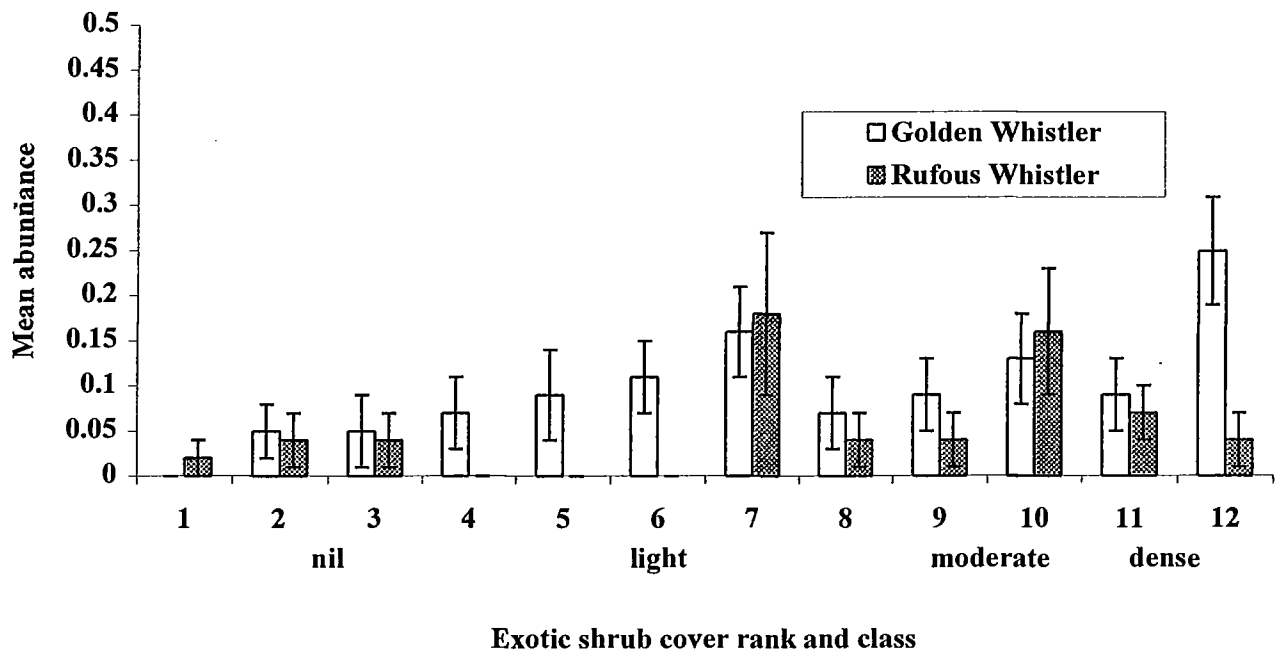


Figure 4.16 - Mean abundance and standard error of the Golden Whistler and Rufous Whistler in *E. melliodora-E. blakelyi* woodland ranked by exotic shrub cover (lowest to highest, all months combined)

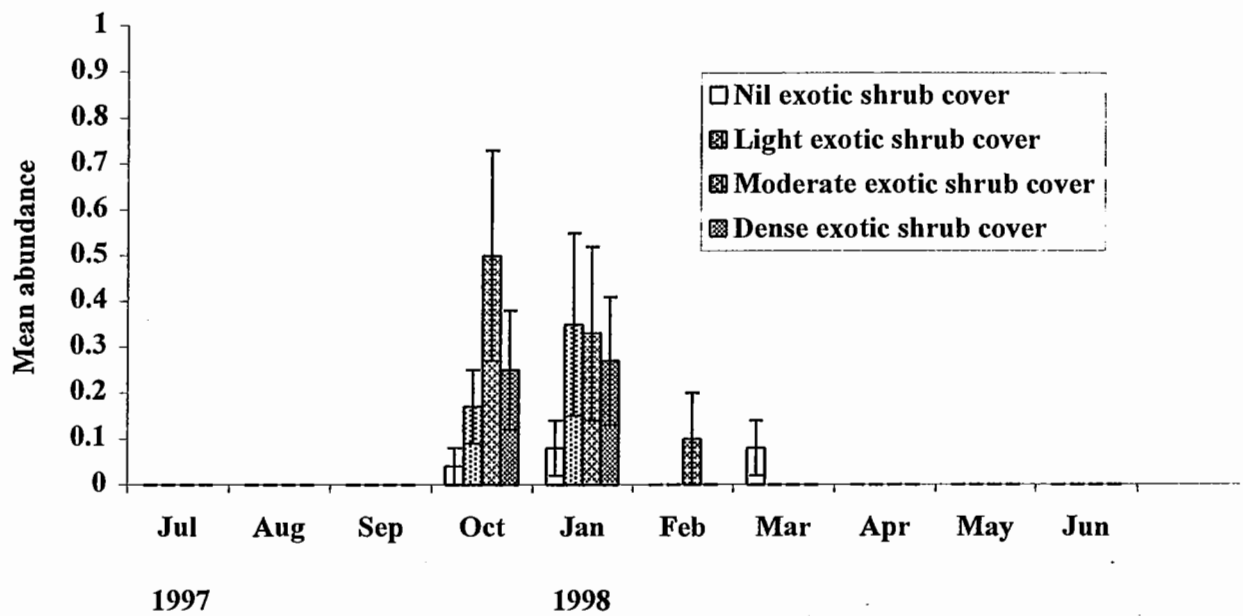
The Rufous Whistler reached its highest mean abundance in the warmer months (summer and spring) and was largely absent in autumn and winter (Figure 4.17). There were no significant differences in the abundance of the Rufous Whistler between the four exotic shrub cover classes for any month (Figure 4.17). The Golden Whistler was most abundant in the winter, autumn and spring months and absent in summer (Figure 4.18). The abundance of the Golden Whistler did not differ significantly between the four exotic shrub cover classes for any month (Figure 4.18), but was positively correlated with increasing exotic shrub cover ( $R_s = 0.79$ ,  $p \leq 0.003$ , Table 4.7). The Grey Shrike-thrush was more abundant at sites with moderate exotic shrub cover in July and in sites with dense exotic shrub cover in October (Figure 4.19). The abundance of the Grey Shrike-thrush differed significantly between the four exotic shrub cover classes in July and October (Figure 4.19).

#### 4.7.9 Family Dicruridae (Boatbills, Monarchs, Flycatchers, Magie-larks, Fantails and Drongoes)

Four species from Family Dicruridae were recorded at the study sites (Appendix 7). The Magpie Lark (*Grallina cyanoleuca*) was most abundant in sites with nil exotic shrub cover (Appendix 7) and was not recorded in exotic shrubs (Table 4.4). The Grey Fantail (*Rhipidura fuliginosa*) was observed in exotic shrubs (Table 4.4) and was seen nesting in Briar Rose (*Rosa rubiginosa*) (Table 4.5). This species was most abundant from September 1997 to April 1998 and at study sites with some exotic shrub cover (light to dense) (Figure 4.20). The abundance of the Grey Fantail was significantly different between the four exotic shrub cover classes in September, October, January, February and May (Figure 4.20). The highest mean abundance for this species was recorded at sites 7, 10 and 11 (Figure 4.21). Mean abundance for the Grey Fantail was positively correlated with exotic shrub cover ( $R_s = 0.81$ ,  $p \leq 0.002$ , Table 4.7).

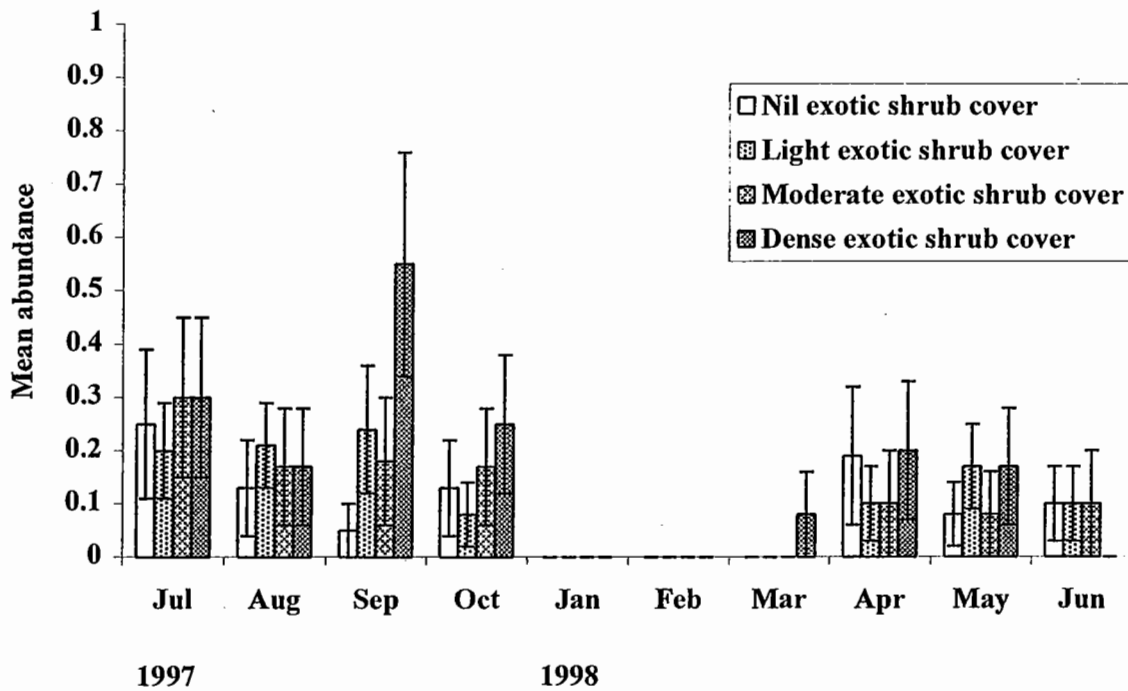
#### 4.7.10 Family Campephagidae (Cuckoo-shrikes, Cicadabird and Trillers)

The Black-faced Cuckoo Shrike (*Coracina novaehollandiae*) and White-winged Triller (*Lalage sueurii*) were observed at the study sites between July 1997 and June 1998 (Appendix 7). The Black-faced Cuckoo Shrike was observed in exotic shrubs (Table 4.4) and on one occasion was seen to consume the berries of *Pyracantha spp* (Table 4.6).



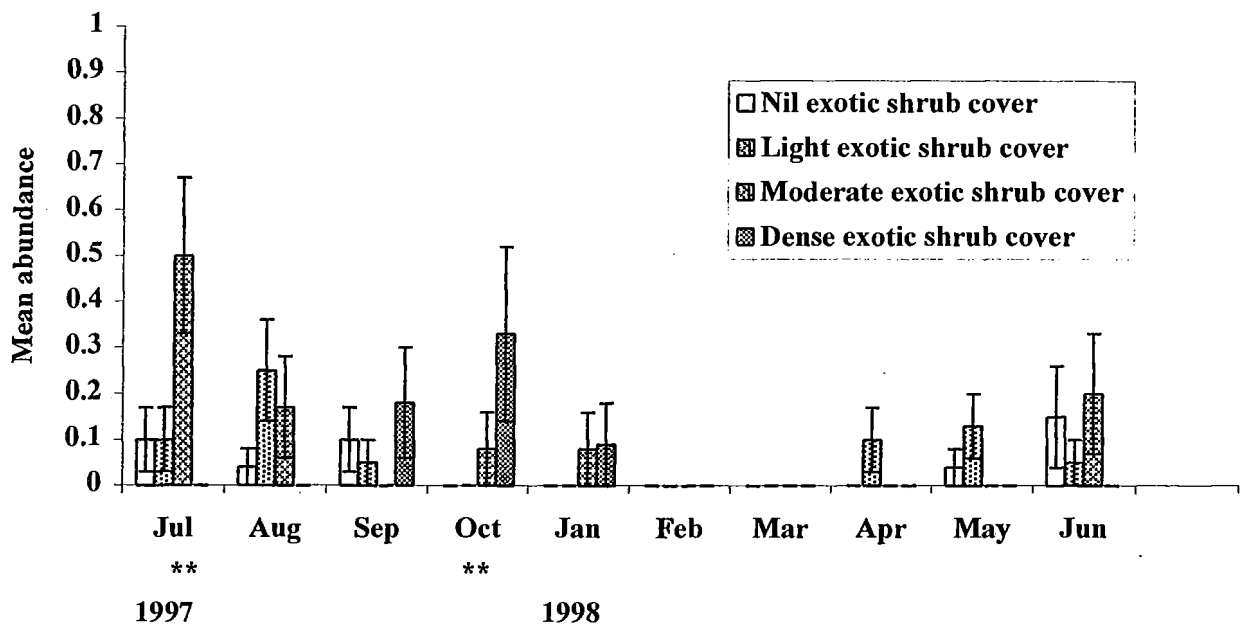
**Figure 4.17 - Mean monthly abundance(1) and standard error of the Rufous Whistler in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover**

(1) Mean monthly abundance was not significantly different ( $P > 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



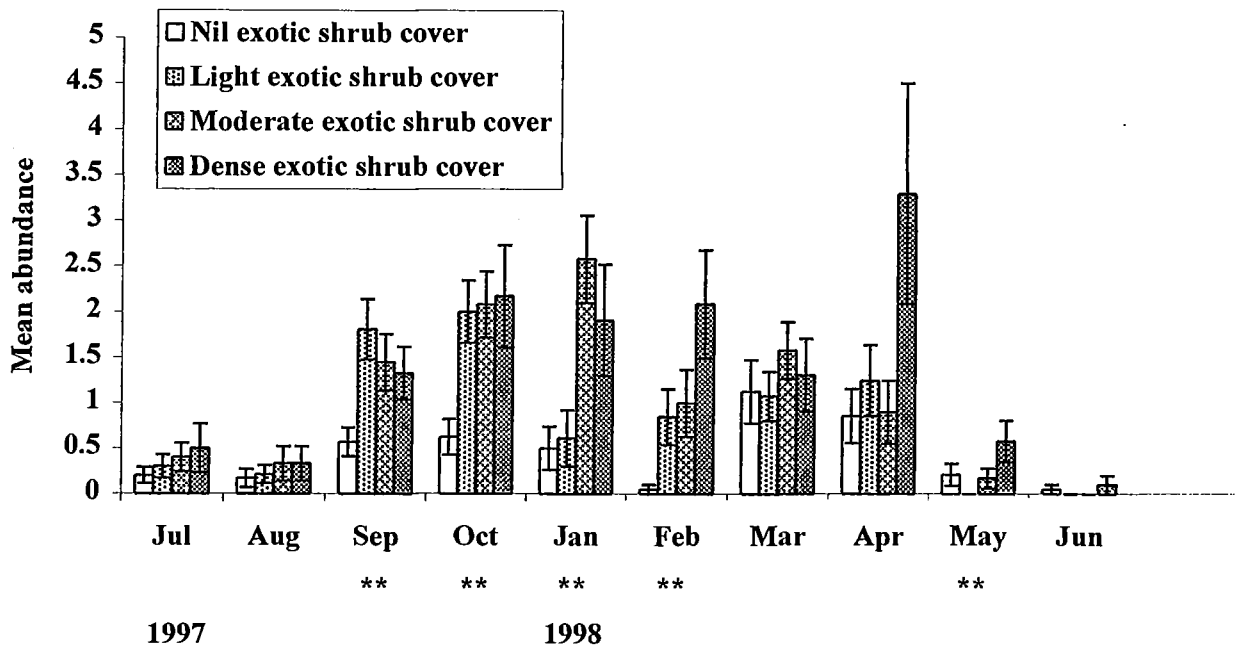
**Figure 4.18 - Mean monthly abundance(i) and standard error for the Golden Whistler in *E. melliodora*-*E. blakelyi* woodland with four levels of exotic shrub cover**

(1) Mean monthly abundance was not significantly different ( $P > 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



**Figure 4.19 - Mean abundance and standard error for the Grey Shrike-Thrush in *E. melliodora*-*E. blakelyi* woodland with four levels of exotic shrub cover**

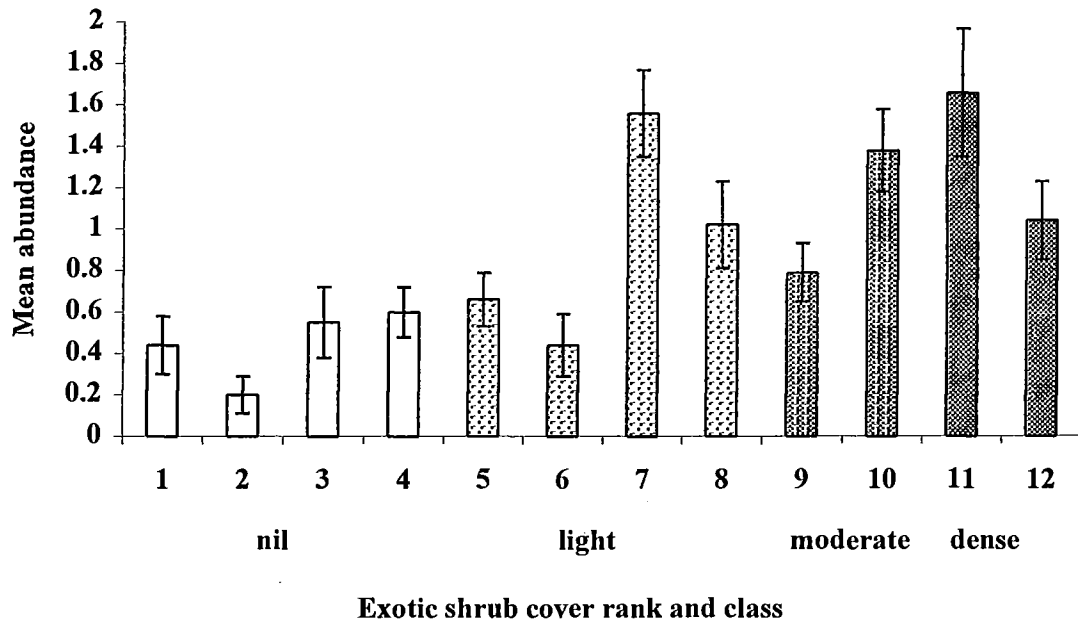
**\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.**



**Figure 4.20 - Mean monthly abundance and standard error of the Grey Fantail in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.





**Figure 4.21 - Mean abundance and standard error for the Grey Fantail in *E. melliodora-E. blakelyi* woodland ranked by level of exotic shrub cover (lowest to highest, all months combined)**

#### 4.7.11 Family Artimidae (*Woodswallows, Butcherbirds, Australian Magpies and Currawongs*)

Five species belonging to the family Artimidae were recorded at the study sites (Appendix 7). Only the Pied Currawong (*Strepera graculina*) was observed in exotic shrubs (Table 4.4), with many records of this species consuming the berries of *Pyracantha spp* and *Cotoneaster spp* shrubs (Table 4.6). The Pied Currawong showed seasonal peaks in abundance in the winter and autumn months at study sites with dense exotic shrub cover (Figure 4.22). The abundance of the Pied Currawong differed significantly between the four exotic shrub cover classes in July, September, January and May (Figure 4.22). Following weed control of *Pyracantha spp* and *Cotoneaster spp* in site 4, the abundance of the Pied Currawong in autumn and winter months was lower (Figure 4.3).

#### 4.7.12 Family Corvidae (*Ravens and Crows*)

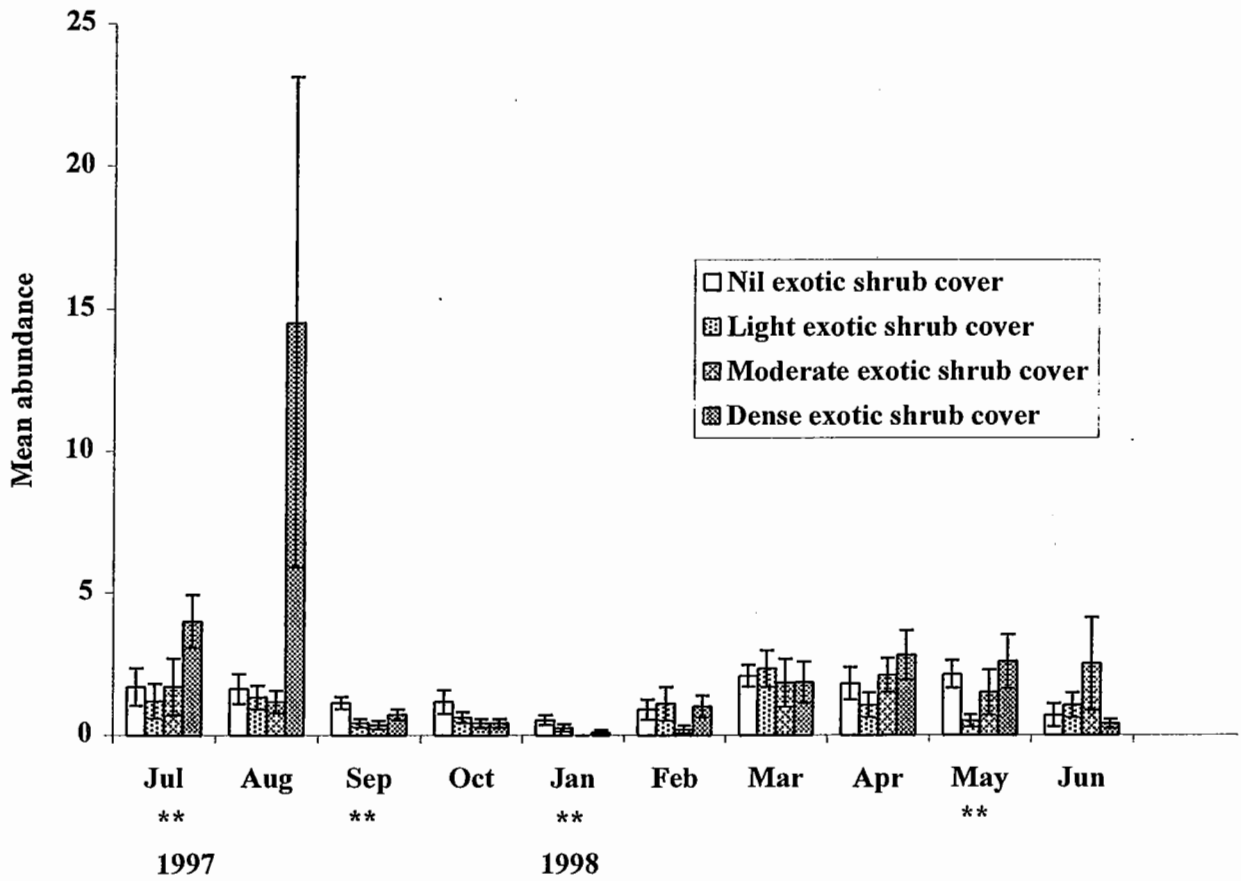
The only Raven species recorded in woodland sites was the Australian Raven (*Corvus coronoides*) (Appendix 7). Site 2 recorded the highest mean abundance of this species (Figure 4.23). The other sites had similar mean abundance for this Raven (Figure 4.23).

#### 4.7.13 Family Corcoracidae (*Australian Mud-Nest Builders*)

The White-winged Chough was recorded at study sites (Appendix 7). This species was not recorded in sites with dense exotic shrub (Figure 4.24). Sites 1, 2 and 8 recorded the highest mean abundance for the White-winged Chough (Figure 4.24).

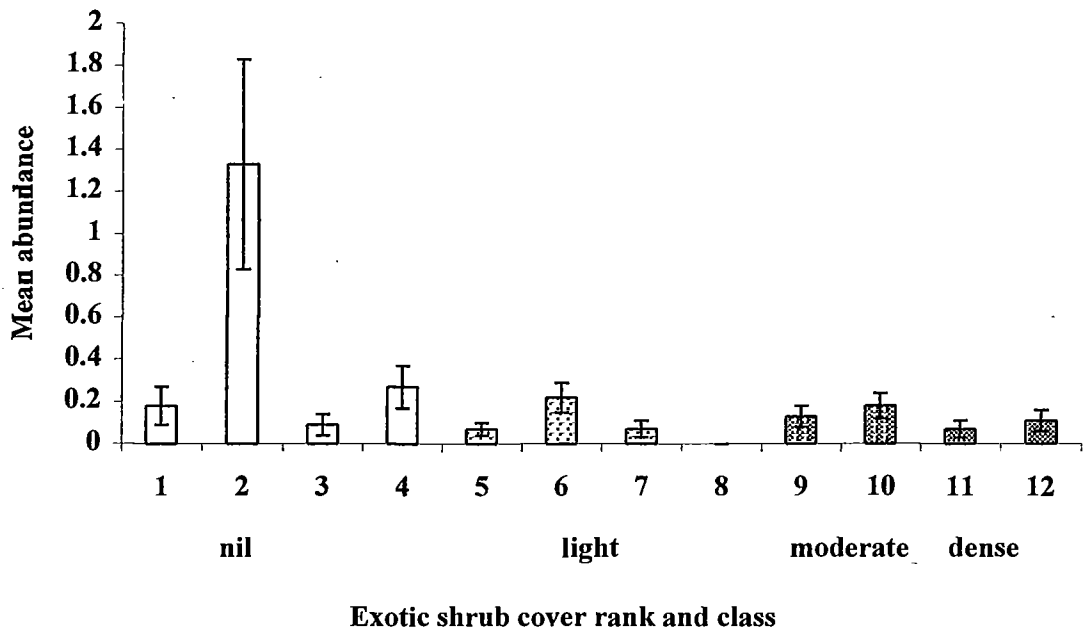
#### 4.7.14 Family Passeridae (*Old World Sparrows, Weavers, Waxbills, Grassfinches, Mannikins*)

The Red-browed Finch (*Neochmia temporalis*) was often recorded in exotic shrub cover when observed in woodland (Table 4.4) and constructed roosts in *Pyracantha spp* (Table 4.5). The Red-browed Finch was more abundant at sites with dense to moderate exotic shrub cover (Appendix 7). Mean abundance in sites with dense exotic shrub cover was highest in the spring and summer months (Figure 4.25). The abundance of the Red-browed Finch was determined as significantly different between the four exotic shrub cover classes in September, October, January, February, March and April (Figure 4.25). Site 11 and site 10 recorded the highest mean

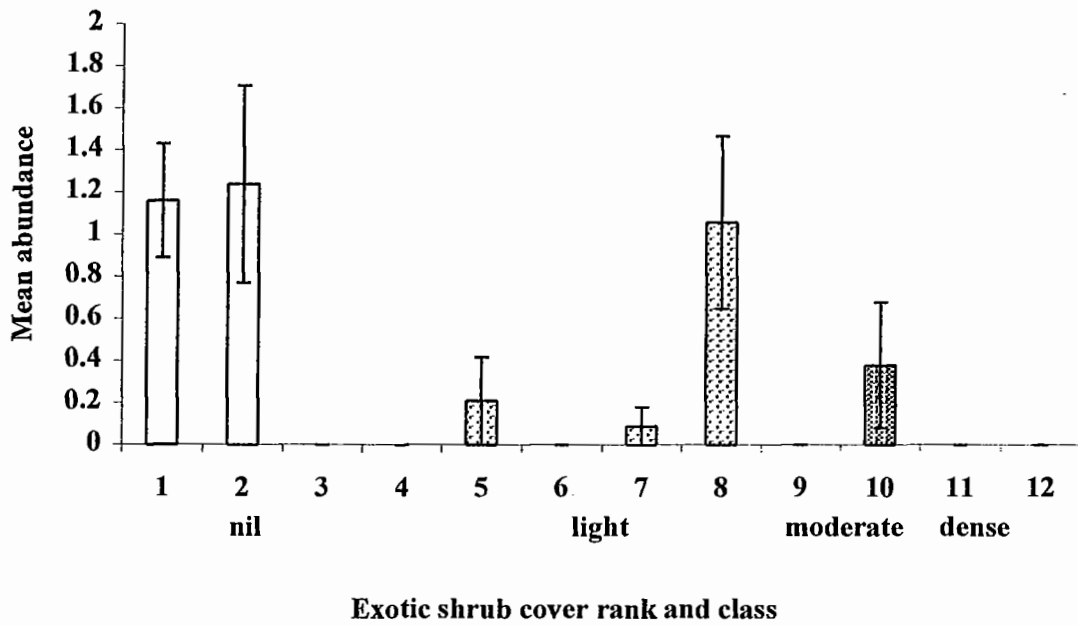


**Figure 4.22 - Mean monthly abundance and standard error for the Pied Currawong in *E. melliodora*-*E. blakelyi* woodland with four levels of exotic shrub cover**

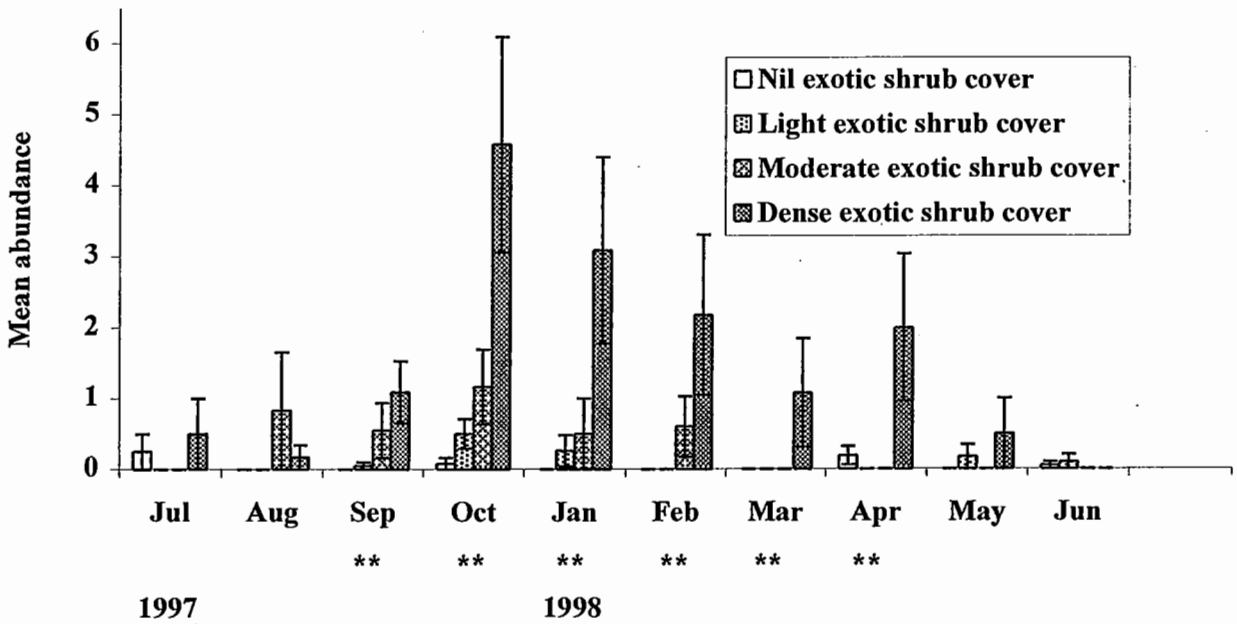
\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



**Figure 4.23 - Mean abundance and standard error for the Australian Raven in *E. melliodora-E. blakelyi* woodland ranked by exotic shrub cover (lowest to highest, all months combined)**



**Figure 4.24 - Mean abundance and standard error for the White-winged Chough in *E. melliodora*-*E. blakelyi* woodland ranked by exotic shrub cover (lowest to highest, all months combined)**



**Figure 4.25 - Mean monthly abundance and standard error for the Red-browed Finch in *E. melliodora*-*E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.

abundance for the Red-browed Finch (Figure 4.26). Mean abundance for the Red-browed Finch by site was positively correlated with exotic shrub cover ( $R_s = 0.61$ ,  $p \leq 0.04$ , Table 4.7).

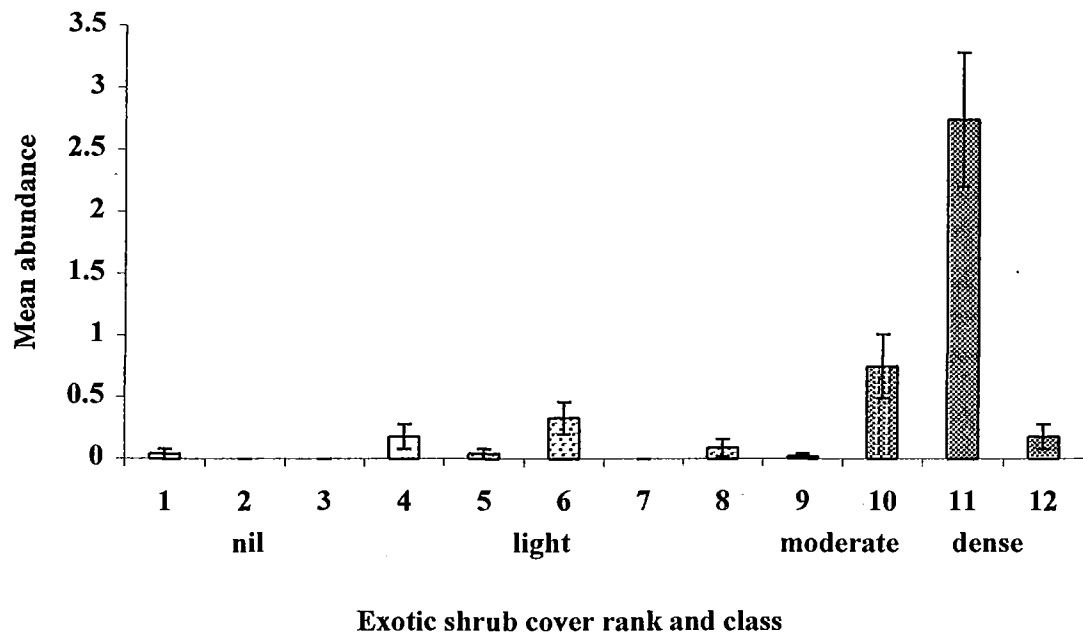
The Double-barred Finch (*Taeniopygia bichenovii*) was only rarely seen at study sites between July 1997 and June 1998 (Appendix 7). The Double-barred Finch built roosts in *Pyracantha spp* (Table 4.5). Both the Red-browed Finch and the Double-barred Finch were lower in abundance at site 4, following weed control in this site (Figure 4.8).

#### 4.7.15 Family Zosteropidae (White-eyes)

The Silvereye (*Zosterops lateralis*) was observed consuming fruit in exotic shrubs (Table 4.6). Disused nests of the Silvereye were also found in *Crataegus spp* (Table 4.5). This species was most abundant at sites with dense exotic shrub cover in the winter months of 1997 (Figure 4.27). Seasonal peaks in abundance in July and August 1997 corresponded with winter feeding on the berries of exotic shrubs (Table 4.6). The abundance of the Silvereye differed significantly between the four exotic shrub cover classes in July, February and May (Figure 4.27). The Silvereye was less abundant in autumn and winter months at site 4 following weed control at this site (Figure 4.3).

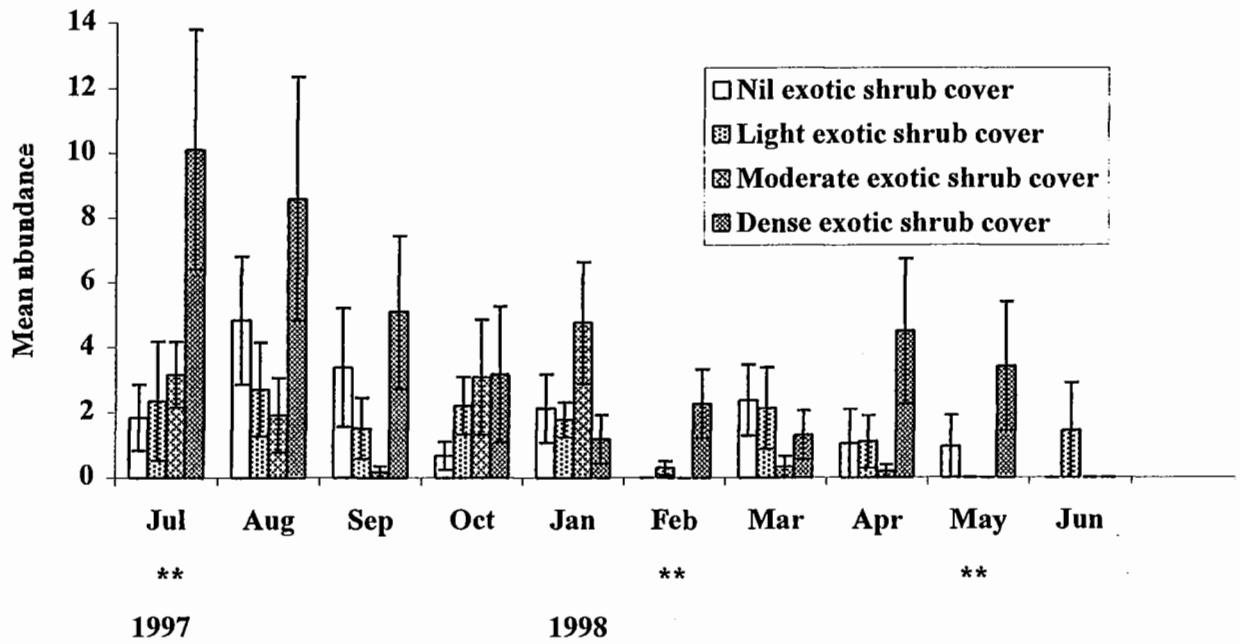
#### 4.7.16 Family Muscicapidae (Thrushes)

The Common Blackbird (*Turdus merula*) was observed in exotic shrubs at study sites (Table 4.4) and disused nests were located in *Crataegus spp* (Table 4.5). This species was also observed feeding on the berries of *Pyracantha spp* on two occasions (Table 4.6). Mean abundance for this species was highest in sites with moderate and dense exotic shrub cover (Figure 4.28). The abundance of the Common Blackbird differed significantly between the four exotic shrub cover classes in July, August, September, October, February, April, May and June (Figure 4.28). Site 12 recorded the highest mean abundance (Figure 4.29). Mean abundance for the Common Blackbird was positively correlated with exotic shrub cover in woodland sites ( $R_s = 0.72$ ,  $p \leq 0.009$ , Table 4.7).



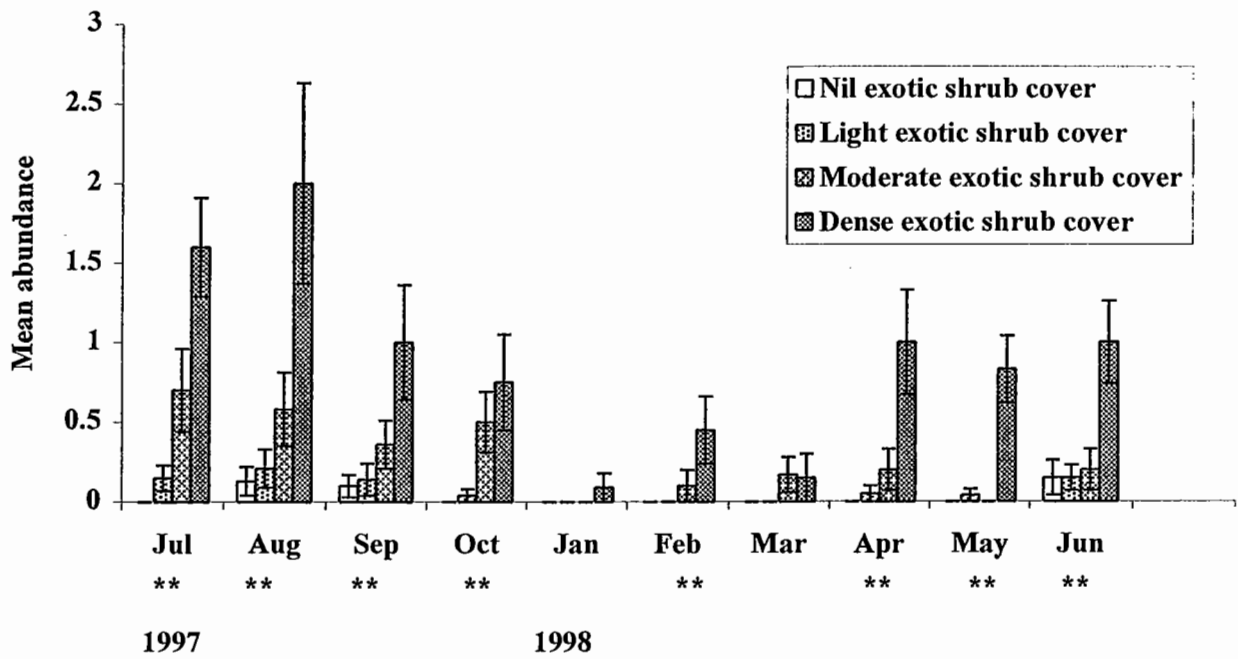
**Figure 4.26 - Mean abundance and standard error for the Red-browed Finch in *E. melliodora-E. blakelyi* woodland ranked by exotic shrub cover (lowest to highest, all months combined)**





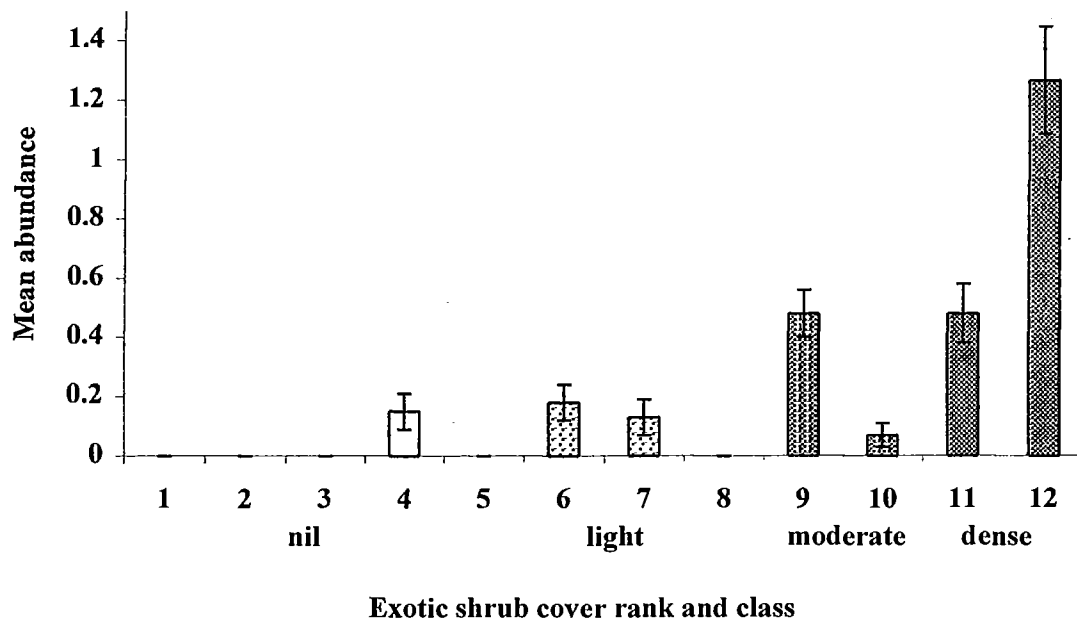
**Figure 4.27 - Mean monthly abundance and standard error for the Silvereye in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly abundance significantly different ( $P \leq 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



**Figure 4.28 - Mean monthly abundance and standard error for the Common Blackbird in *E. melliodora*-*E. blakelyi* woodland with four levels of exotic shrub cover**

**\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.**



**Figure 4.29 - Mean abundance and standard error for the Common Blackbird in *E. melliodora*-*E. blakelyi* woodland ranked by level of exotic shrub cover (lowest to highest, all months combined)**

#### 4.7.17 Family Sturnidae (Starlings and Mynahs)

The Common Mynah (*Acridotheres tristis*) and Starling (*Sturnus vulgaris*) were both observed in exotic shrub cover (Table 4.4). The Common Mynah was not reported at sites with dense exotic shrub cover for six of the ten months that sites were surveyed (Figure 4.30). The abundance of the Common Mynah differed significantly between the four exotic shrub cover classes in March (Figure 4.30). The mean abundance of the Common Starling was lowest in sites with dense exotic shrub cover (Appendix 8).

#### 4.8 Bird species richness and total abundance

Mean species richness in study sites with four levels of exotic shrub cover varied considerably during the ten months sampled (Figure 4.31). In July 1997 woodland with moderate and dense exotic shrub cover recorded the most bird species (Figure 4.31). In October 1997, woodland with light, moderate and dense exotic shrub cover recorded more bird species than woodland with nil exotic shrub cover (Figure 4.31). The summer months also showed more species of birds being recorded in woodland with light, moderate and dense levels of exotic shrub cover compared to sites with nil exotic shrub cover (Figure 4.31). Mean species richness in February was highest overall in woodland sites with dense exotic shrub cover (Figure 4.31). Generally, from March 1998 through to June 1998, there was a decline in the number of species recorded in all sites when compared to the period July 1997 to February 1998 (Figure 4.31). Mean species richness differed significantly different between the four exotic shrub cover classes in October, January and February (Figure 4.31). Sites 3 and 4 recorded the lowest mean bird species richness, sites 10 and 11 the highest (Figure 4.32).

Mean total abundance of birds was highest at sites with dense exotic shrub cover in July and August 1997, with abundance in woodland with moderate exotic shrub cover highest in July 1997 (Figure 4.33). Total abundance of birds in January 1998 was highest at sites with moderate exotic shrub cover. In April 1998 sites with dense exotic shrub cover had the highest mean abundance of birds (Figure 4.33). Mean total bird abundance differed significantly between the four exotic shrub cover classes in October, January and February (Figure 4.33). Sites 7 and 10 recorded the highest mean total abundance, sites 3 and 4 the lowest (Figure 4.34).

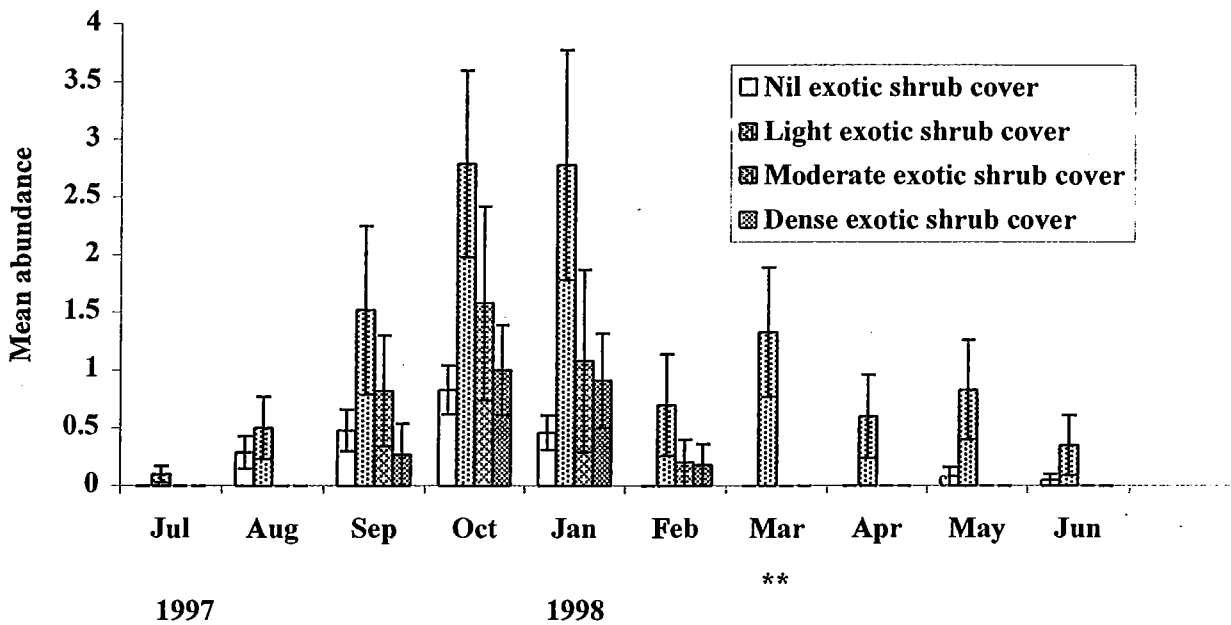
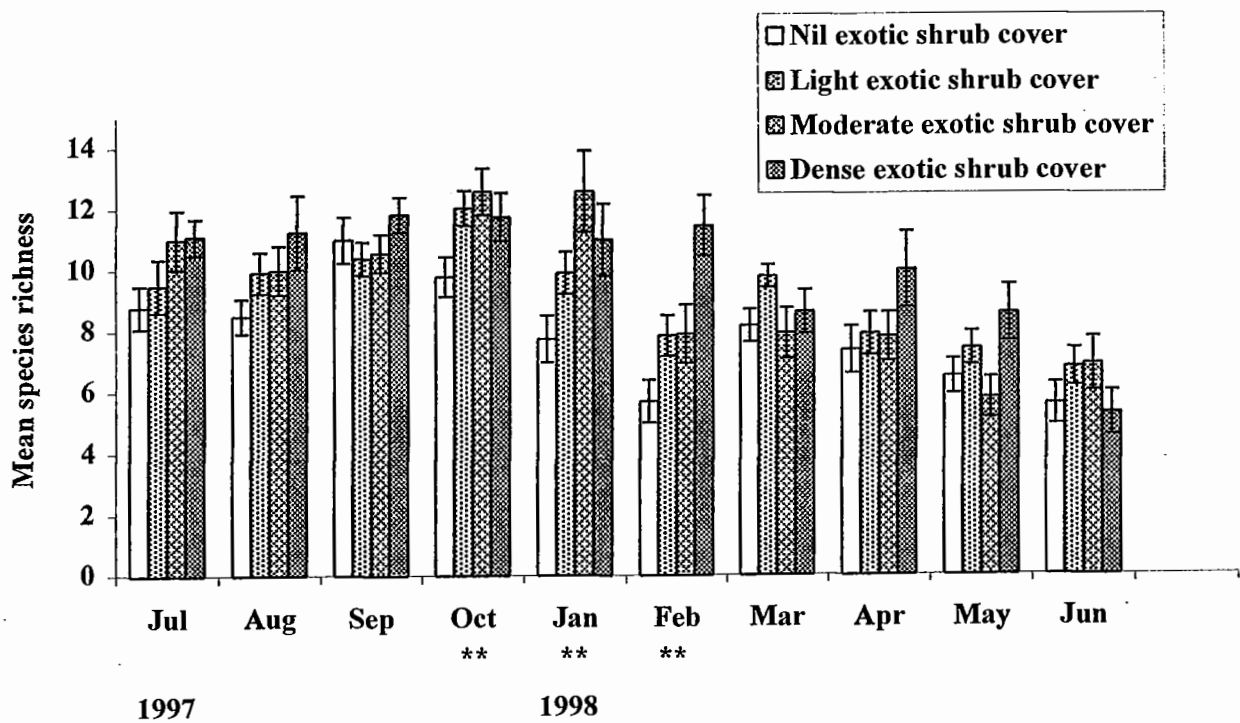


Figure 4.30 - Mean monthly abundance and standard error for the Common Mynah in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



**Figure 4.31 - Mean monthly bird species richness and standard error in *E. melliodora*-*E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly species richness significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.

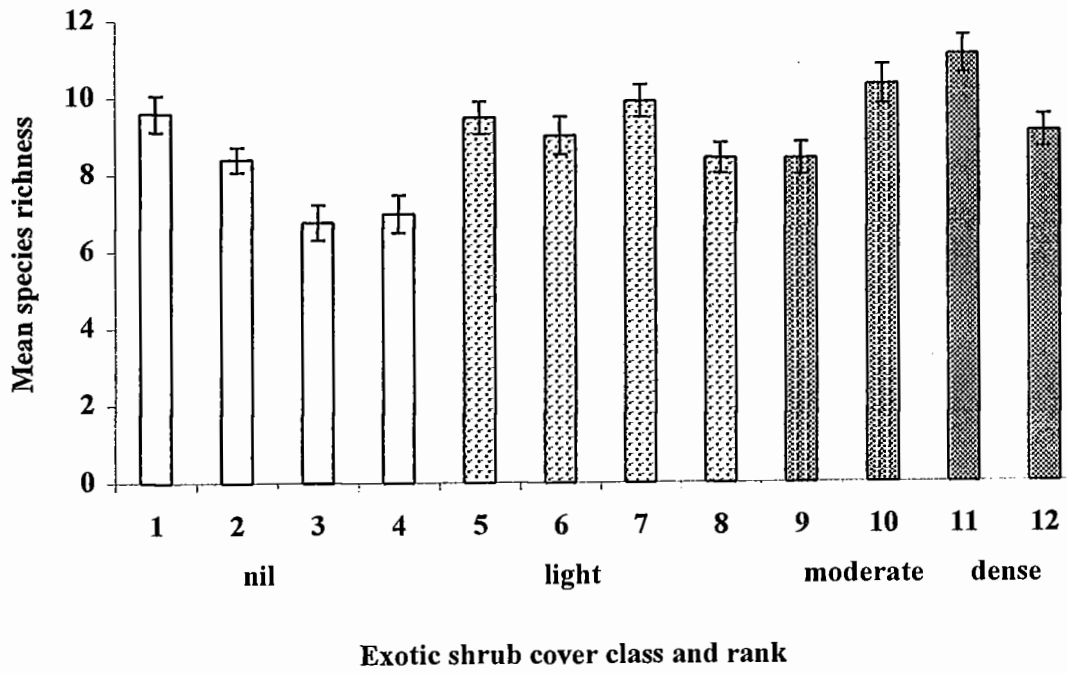
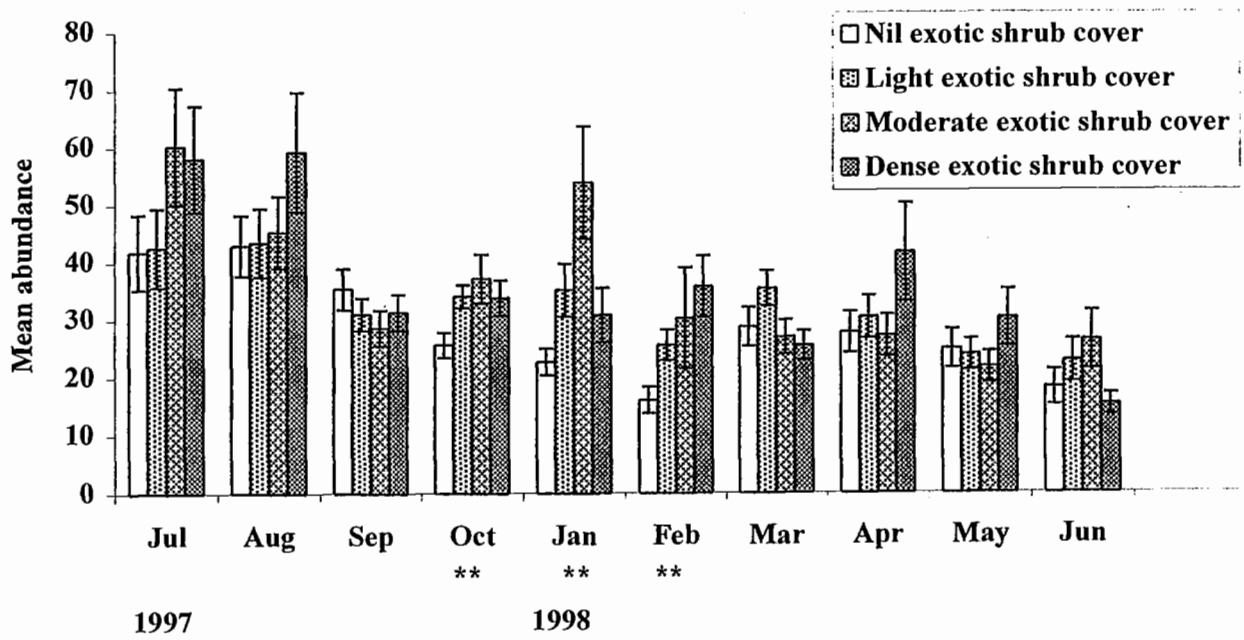


Figure 4.32 - Mean bird species richness and standard error in *E. melliodora*-*E. blakelyi* woodland ranked by exotic shrub cover (lowest to highest, all months combined)



**Figure 4.33 - Mean monthly total bird abundance (all species combined) and standard error in *E. melliodora-E. blakelyi* woodland with four levels of exotic shrub cover**

\*\* Mean monthly abundance significantly different ( $P < \text{or} = 0.05$ ) for exotic shrub classes, as determined by a Kruskal Wallis One Way ANOVA.



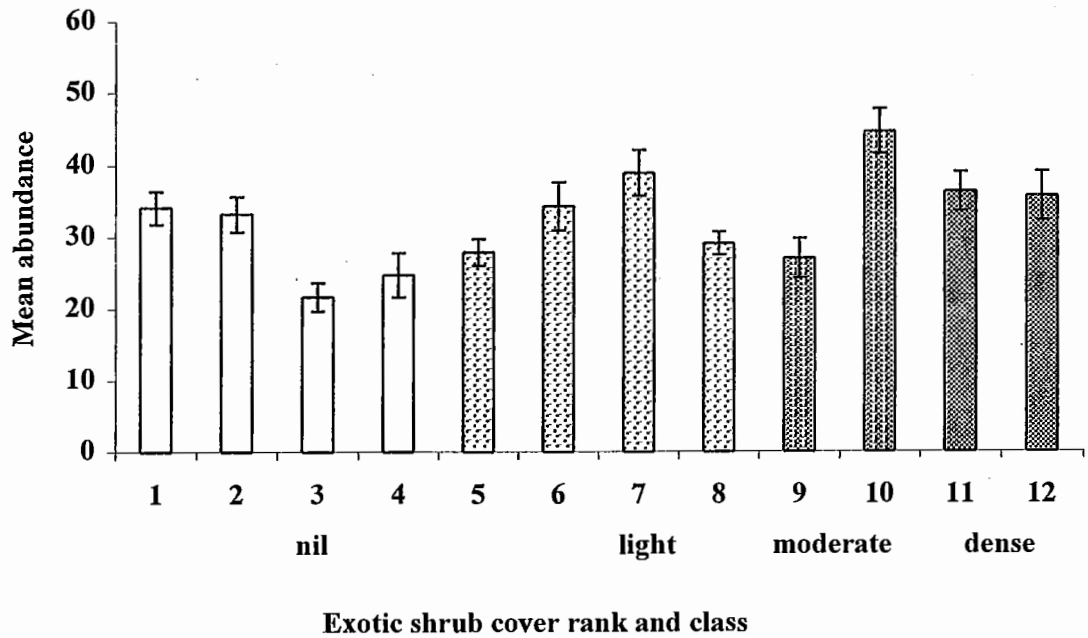
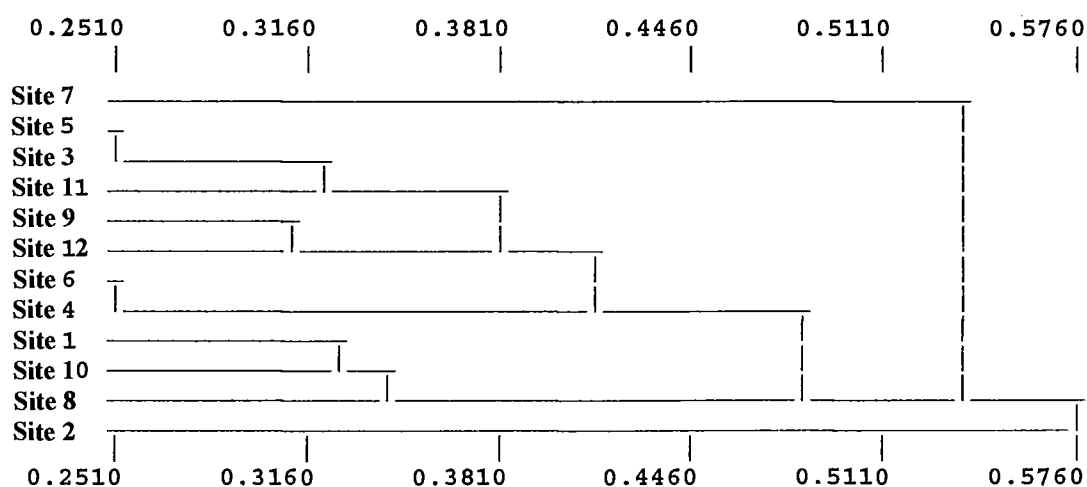


Figure 4.34 - Mean total bird abundance (all species combined) and standard error for *E. melliodora*-*E. blakelyi* woodland ranked by exotic shrub cover (lowest to highest, all months combined)

#### 4.9 Ordination and cluster analysis on the mean abundance of all birds observed

Cluster analysis on the abundance of the 75 species of birds (Appendix 9) observed in study sites revealed four groups (Table 4.8). Site 2 and site 7 were the most different of all the woodland sites in terms of their bird life (Figure 4.35). There is reasonable agreement between the identified groups and the distances between sites using cluster analysis, when compared to multidimensional scaling of the sites plotted in three dimensions (Figure 4.36).



**Figure 4.35 - Dendrogram of woodland sites showing the distance between them based on mean abundance of seventy-five bird species using hierarchical clustering. Note site numbering relates to exotic shrub rank (see Table 4.1 and Table 4.8)**

**Table 4.8 – Site groups formed by cluster analysis**

Group No.	Site No. ( weed rank <sup>1</sup> )
1	7
2	3, 4, 5, 6, 9, 11, 12
3	1, 8, 10
4	2

(1) Site numbers correspond to their exotic shrub rank. Sites 1 to 4 were classed as having nil exotic shrub cover, sites 5 to 8 were classed as having light exotic shrub cover, sites 9 and 10 moderate cover, and sites 11 and 12 dense cover (see Table 4.1).

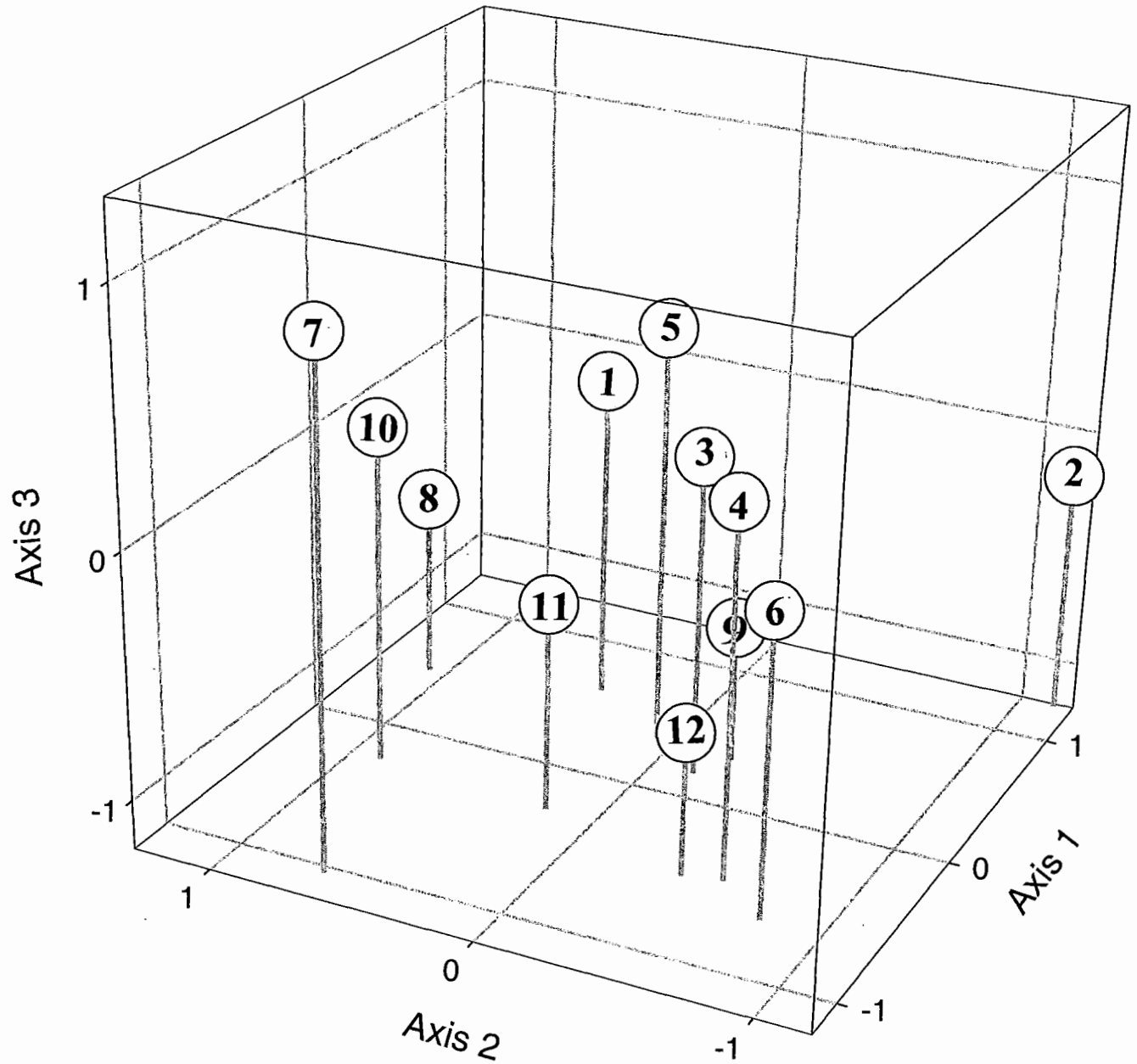


Figure 4.36 - Three dimensional plot of woodland sites based on the mean abundance of seventy-five bird species (non-metric multi-dimensional scaling). Note site numbering relates to exotic shrub rank (see Table 4.1 and Table 4.8).

Group 1 was comprised of site 7. This site was different from all the others in terms of bird life because the Sacred Kingfisher, Leaden Flycatcher, Speckled Warbler and Superb Fairy-wren were all most abundant at this site, and the Pied Currawong was least abundant (Appendix 9).

Group 2 comprised seven sites (Table 4.6). Differences within this group were also apparent (Figure 4.36). For example, site 11 was different from other sites in Group 2 because it had the highest mean abundance for the Red-browed Finch and Crested Pigeon, while sites 5 and 11 are close together and had the two highest means for Silvereye abundance (Appendix 9).

Group 3 included sites 1, 8 and 10. The relatively high abundance of the Common Starling distinguished these sites from the others (Appendix 9). The Common Bronzewing was only observed in Sites 10 and 8 and was absent from all other sites, which explains the proximity of sites 10 and 8 within Group 3 when plotted in three dimensions (Figure 4.36).

Group 1 (site 7), Group 2, and Group 3 were distant from Group 4 (site 2) (Figure 4.36). Group 4 (site 2) was different from all other sites because it had the highest mean abundance for the Noisy Miner, Australian Raven, Australian King-Parrot and Eastern Rosella (Appendix 9).

Site 2 and site 7 were the most distant sites when plotted in three dimensions (Figure 4.36). An important difference in their bird life was that the Speckled Warbler and Superb Fairy-wren were most abundant in site 7 and least abundant and absent from site 2, respectively (Appendix 9).

## Chapter 5

### Discussion

#### 5.1 Introduction

In this study I found that the presence of exotic shrubs had differential impacts on the abundance of birds and the composition of bird assemblages in *E. melliodora*-*E. blakelyi* woodland within urban reserves in the ACT. The patterns of abundance and species composition that I observed were supported by observations of birds using exotic shrubs as a specific resource in woodland (e.g. the Crested Pigeon and the Crimson Rosella) (Table 4.5 and Table 4.6, respectively). Exotic shrubs were important for some birds and provided food, nest sites and shelter. Other birds had little or no direct interaction with exotic shrubs and their abundance was not obviously affected. There were also some birds which were absent or less abundant in woodland with dense levels of exotic shrub invasion. The notion of shrub weeds having differential impacts on birds supported here, and by earlier work in coastal woodland (French and Zubovic 1997), suggests that intensive control of exotic shrubs, including their complete removal, may benefit some birds and harm others. My results are discussed below in the context of specific relationships between birds and exotic shrubs and the possible conservation implications of these relationships.

#### 5.2 The structural effect of exotic shrubs on birds of urban woodland

In functional terms, exotic shrubs in urban woodland added structural complexity to bird habitat. Finches and wrens were two sub-groups of birds that responded positively to this structural complexity and used exotic shrubs as protection from predators, shelter and as nesting sites (Table 4.4 and Table 4.5). The removal and control of exotic shrubs in urban woodland is likely to adversely affect these types of birds in the short to medium term and reduce their abundance (Figure 4.8). Birds of most concern in terms of this effect are the Speckled Warbler and the Double-barred Finch. Both these birds are in decline in the ACT (ACT Commissioner for the Environment 1995), and hence the risk of habitat simplification following weed control of exotic shrubs is more pressing for these birds than others whose conservation status is secure.

The Speckled Warbler is a ground nesting bird (Schodde and Tidemann 1988). Exotic shrubs, such as *Rosa rubiginosa*, may help to conceal and protect the nests of Speckled Warblers from

predators, such as the Pied Currawong (J. Gardner pers.comm.). However, this view remains to be tested. The Speckled Warbler can persist in relatively small patches of woodland (less than ten hectares), provided that habitat complexity is high, and they are not too isolated from other woodland patches (Freudenberger 1999). I argue here that exotic shrubs contribute to habitat complexity.

In this study I have provided qualitative information suggesting an association between the Double-barred Finch and the presence of exotic shrubs in woodland. This species was less abundant in a single woodland site following removal of exotic shrubs (Figure 4.8) and built roosts in *Pyracantha spp.* (Table 4.5). An alternative explanation for the observed reduction in the abundance of this finch following removal of exotic shrubs was the general influence on birds of the drought during 1997 and 1998, an event that occurred after weed control.

There was some indication that for certain wrens (e.g. the Speckled Warbler and Superb Fairy-wren), abundance was lower at high levels of exotic shrub cover (Figure 4.10 and Figure 4.7). The Red-browed Finch was also less abundant at the site most heavily modified by exotic shrub invasion (Figure 4.26). At high levels of exotic shrub invasion, the structural benefits of shrub cover may be mitigated by the loss of open ground in which to forage. The loss of native plants from the understorey may also reduce resources, such as food. The evidence from my study for the presence of any threshold effects, however, is weak and requires further investigation.

The Crested Pigeon was correlated with exotic shrub cover in woodland habitat (Table 4.7) and was seen nesting in *Pyracantha spp.* (Table 4.5). Canberra is at the eastern limit of the range for this pigeon (Taylor and Canberra Ornithologists Group 1992). Changes in the local population have fluctuated, rising in drought years and falling when local conditions worsened, such as a series of harsh winters (Taylor and Canberra Ornithologists Group 1992). More recently, Crested Pigeons have become established in Canberra gardens and the urban environment, with a significant increase in breeding records noted in 1988-89 and 1993-94 (Canberra Ornithologists Group 2000). It is probable that exotic vegetation, such as exotic shrubs, have helped this species to become established in Canberra, as it feeds extensively on the seeds of exotic plants (Higgins and Davies 1996) and nests in *Pyracantha spp.* (Table 4.5).

The Laughing Kookaburra, Magpie Lark and White-winged Chough are known to favour open habitat (Schodde and Tidemann 1986; Taylor and Canberra Ornithologists Group 1992; Strahan 1996). These birds were most abundant in woodland with little or no exotic shrub cover (Appendix 5). In these instances exotic shrubs and other invasive plants, may remove key resources including: open ground; a relatively undisturbed native understorey; and food inhabiting the litter layer. More research is needed, however, to establish if exotic shrubs pose a threat in this context, as I have little evidence to support or discount this hypothesis.

The Laughing Kookaburra was the only bird species negatively correlated with exotic shrub cover. However, the differences in its abundance in woodland sites with varying levels of exotic shrub cover were small in magnitude. In coastal woodland invaded by Bitou Bush (*Chrysanthemoides monilifera*), French and Zubovic (1997) found that raptors and scavengers, such as ravens, were less abundant in densely invaded woodland. One theory that they proposed to explain this was that there was more shrub cover for prey, reducing the efficiency of prey detection and capture. The Laughing Kookaburra has a preference for open habitat in which to perch and pounce on vertebrate and invertebrate prey on the ground (Strahan 1994). Exotic shrub cover may reduce the open terrain the Laughing Kookaburra needs in order to hunt effectively. This likelihood would complement the earlier argument that exotic shrubs protect small birds, such as finches and wrens, from predators.

Pied Currawongs are dominant bird predators in urban landscapes (Major *et al.* 1996; Wood 1998). For instance, Major *et al.* (1996) reported that of 141 predatory attacks on artificial nests observed in urban areas, 59 percent could be attributed to the Pied Currawong. In my study I saw little direct predation on birds, and did not sample birds in November, when breeding activity is high (Taylor and Canberra Ornithologists Group 1992). Further study may be needed to resolve whether Pied Currawongs are efficient nest predators in exotic shrub cover.

Silvereyes are widespread throughout Australia and have adapted well to urban landscapes (Schodde and Tidemann 1988). Their ability to use exotic shrubs in urban woodland for food (Table 4.6) and nesting substrate (Table 4.5) provides more evidence of their capacity to adapt to modified habitats. Silvereyes belong to a widely-distributed family (Zosteropidae) and are fairly uniform in size, shape and plumage, suggesting comparatively recent evolution and colonisation of Australia (Pizzey and Knight 1997). This evolutionary background and the distribution of

Zosteropidae through Africa and Asia (Pizzey and Knight 1997) may explain why Silvereyes have adapted so well to exotic shrubs introduced to Canberra from China and Africa (e.g. *Pyracantha spp* and *Cotoneaster spp*). However, Silvereyes also used *Crataegus spp*, which is introduced from Europe. Their dominance of urban habitat in Canberra (Lenz 1989) and wide distribution suggests they are consummate generalists, which respond to habitat structure more strongly than the floristic composition of vegetation.

The Golden Whistler and Grey Fantail were both observed in exotic shrubs and their abundance was positively correlated with exotic shrub cover. The Rufous Whistler was also seen in exotic shrubs and has been observed nesting in *Rosa rubiginosa* in *E. melliodora*-*E. blakeyi* woodland in Canberra (A. Overs pers. comm.). The conservation status of the Golden Whistler and Grey Fantail is currently secure and both are widely distributed (Schodde and Tidemann 1988). Reid (1999) noted that the Rufous Whistler was in decline in the NSW Sheep Wheat Belt, probably as a result of simplification and clearing of habitat. Freudenberger (1999) noted that the Rufous Whistler was infrequently observed in woodland sites smaller than ten hectares or in sites with little structural diversity. He suggested that the Rufous Whistler may be a good candidate for use as a focal species to assess the impact of planting native vegetation to revitalise the habitat value of grassy woodland remnants. Moreover, he predicted the presence of this bird in revegetated sites will provide an early indication that the habitat value of grassy woodland has been enhanced. In this context, the effect of removing exotic shrubs from woodland in Canberra could perhaps be tested using focal species, such as Whistlers, to determine the impact of removing exotic shrubs in grassy woodland on habitat value.

The Brown Thornbill was correlated with exotic shrub cover in woodland (Table 4.7) and was often observed in exotic shrubs (Table 4.4.). Brown Thornbills live as territorial pairs (Bell 1985). Bell (1985) noted that the size of a Brown Thornbill territory was inversely correlated with the percentage of territory covered in understorey. This suggests that exotic shrubs in the understorey may reduce territory size, increasing the density of Brown Thornbills in woodland with a shrubby understorey. The offspring of Brown Thornbills also suffer higher rates of predation after fledging than other species of thornbills, such as the Buff-rumped Thornbill, whose young join social groups or clans (Bell 1985). This difference in social organisation (i.e. territorial pairs as opposed to clan living) may influence rates of predation, with Brown



Thornbills relatively more dependent on the cover provided by exotic shrubs than other thornbill species.

The Common Blackbird was the only exotic species in urban woodland found to utilise exotic shrubs on a regular basis and nested and sheltered in exotic shrubs. The Common Blackbird may compete with native birds in woodland, although I have no evidence to support any strong competitive effects. The native Bassian Thrush (*Zoothera lunulata*) is a close relative of the Common Blackbird and is distributed in the ranges of the ACT, although some birds are found at lower altitudes in winter (Taylor and Canberra Ornithologists Group 1992). Exotic shrubs along river corridors have assisted the Common Blackbird in becoming established in non-urban areas of the ACT (Taylor and Canberra Ornithologists Group 1992). There is some concern that if the Blackbird becomes established in the ranges it will directly compete with, and perhaps displace, the native thrush (Taylor and Canberra Ornithologists Group 1992). Dense thickets of blackberry in gullies have been identified as particularly favorable to this species, in terms of it invading native forests (Loyn 1985).

### 5.3 Exotic shrubs as a food resource

Nine species of birds were observed eating exotic fruit and seeds from shrubs in urban woodland between 1996 and 1998 (Table 4.6). Only some fruit eating birds showed seasonal peaks in abundance corresponding with the availability of fruit. The conservation status of all nine birds identified consuming fruit is currently secure nationally and in the ACT. Conservation issues mostly relate to whether fruit eating birds spread the seed of exotic shrubs and if their population is increased by fruit, to the detriment of other birds.

The Silvereye, Common Starling and Pied Currawong were observed eating fruit from *Pyracantha spp* and *Cotoneaster spp* shrubs between 1996 and 1998 (Table 4.6). These birds are implicated in seed dispersal of these shrubs, favoring *Pyracantha spp* over *Cotoneaster spp* (Mulvaney 1986). The Pied Currawong is also known to disperse *Crataegus spp* seeds (Bass 1990). The Crimson Rosella, Eastern Rosella, Australian King Parrot and Gang-gang Cockatoo are seed predators and destroy the seeds of *Pyracantha spp* and *Cotoneaster spp* (Mulvaney 1986).

Fruit-bearing shrubs, particularly *Pyracantha spp* and *Cotoneaster spp*, are implicated in increases in the abundance of the Pied Currawong population in Canberra because the highly palatable berries provide an abundant food source for currawongs in autumn and winter (Taylor and Canberra Ornithologists Group 1992). This increase in abundance is of concern because Pied Currawongs are known to be highly effective nest predators of small native birds (Wood 1998; Major *et al.* 1996) and may have contributed to the decline of the Jacky Winter in the Sydney Region (Hoskins *et al.* 1991). Pied Currawongs regurgitate viable seed of *Pyracantha spp* and *Cotoneaster spp* (Mulvaney 1984). The seasonal use of these shrubs as a food source suggests a mutually reinforcing relationship may have become established (i.e. an increase in exotic shrubs leads to an increase in Pied Currawong populations which leads to the increased spread of exotic shrubs). Further studies are required to test the extent to which this is the case.

Removal of exotic shrubs through weed control removes much of the fruit available to Pied Currawongs and is thought to be one way of reducing Pied Currawong numbers (Strahan 1996). However, control of exotic shrubs in Canberra, when it occurs selectively in urban woodland, removes protective cover and nest sites for small native birds. Ample supplies of exotic shrubs and their berries are not generally subject to weed control because they are located in urban areas and pine forests. Because fruit available in these areas is largely unaffected by weed control, the net impacts on the Pied Currawong population from weeding of exotic shrubs in woodland may be negligible as only a proportion of the fruit resource is being removed. As a result, the potential impact on small native birds is likely to remain, since Pied Currawongs readily move between reserves and urban landscapes to feed (Wood 1996). Moreover, since some small birds, such as the Speckled Warbler, are largely restricted to woodland habitat, the net effect of weed control in such areas may be to remove protective cover from small woodland specialists. The seed rain from Pied Currawongs will also continue as long as there are exotic shrubs beyond the reach or will of managers to control.

Weed control in woodland, however, may influence some aspects of dispersal ecology, such as the distance that seed travels into native habitat before being regurgitated. Bass (1991) noted that Pied Currawongs dispersed *Crataegus spp* seed more than 1000m in the northern tablelands of NSW.

Speckled Warblers were positively correlated with increasing exotic shrub cover (Table 4.7) and are also known to eat fruit (Ford 1989). However, this species was not observed feeding on the berries of exotic shrubs (Table 4.6). Whether or not exotic berries are important in the diet of Speckled Warblers in weed invaded *E. melliodora*-*E. blakelyi* woodland may be worthy of more research.

Although I did not examine the productivity of exotic shrubs in terms of invertebrate prey items, insect eating birds, such as whistlers, thornbills and honeyeaters, were observed in exotic shrub cover (Table 4.4). Mulvaney (1984) noted that few invertebrates feed, breed or shelter in *Pyracantha spp* or *Cotoneaster spp*, suggesting that these genera may be a poor source of food for insectivorous birds. However, whether or not exotic shrub cover is a good or poor source of invertebrate prey requires further investigation.

#### 5.4 Exotic shrubs and their management– a threat to ecosystem function?

Although it is clear that exotic shrubs in urban woodland threaten native vegetation in some contexts, the precise effects of exotic plants invading urban bushland in the ACT are not known (Standing Committee on Conservation, Heritage and Environment 1994). Moreover, in an urban context some pressures on birds, such as structural simplification through the collection of firewood, isolation from other woodland patches, the presence of cats and foxes, and an abundant Pied Currawong population, may, as I have argued earlier, be mitigated by the presence of exotic shrub cover. The effects of exotic shrubs on less obvious ecosystem functions, such as nutrient cycles, is largely unknown, and may be profound. For example, Tommerup and Bougher (2000) have outlined the essential role played by ectomycorrhizal fungi in nutrient cycling in temperate eucalypt woodland. They suggested that weeds may interfere with nutrient cycling in woodland by harming or displacing endemic fungi. Tommerup and Bougher (2000) also noted that some ectomycorrhizal fungi are sensitive to soil disturbance, losses of organic matter and rises in soil temperature associated with bare open ground. Weed control may influence all these factors. These considerations were well beyond the scope of my study. I mention them here, however, to make two points: 1) a lack of evidence of threats to ecosystem function from exotic shrubs does not mean a threat does not exist; and 2) ecosystem effects may be complex and not easily deduced or assumed.

Animals play an important part in ecosystem function in woodland. For instance, birds regulate the abundance of herbivorous insects in the foliage of woodland trees, which in turn influences the occurrence of dieback and tree health (Loyn 1985; Landsberg *et al.* 1990; Barrett and Davidson 2000) and small mammals help to spread ectomycorrhizal fungi (Tommerup and Bougher 2000). Birds also play a role pollinating plants in woodland habitat (Robinson 1993). Efforts to sustain degraded woodland, such as woodland in agricultural landscapes, may depend on revegetation strategies that attract diverse bird populations to reduce the severity of insect attack (Barrett and Davidson 2000). Thus, the loss of birds from degraded woodland areas following weed control, may influence ecosystem function and tree health in these areas.

Noisy Miners were most abundant at site 2, which lacked a shrubby understorey (Table 4.1 and Table 4.3). This site proved to be an outlier when plotted in three dimensions following an ordination on the abundance of birds observed in woodland sites (Figure 4.36). Noisy Miners are insectivorous, colonial and aggressively exclude other insectivorous birds in defense of their insect prey (Dow 1977). This behaviour may have implications for the health of woodland trees. Some studies have suggested that a diverse mixture of insectivorous birds reduces the abundance of herbivorous insects in the canopy of eucalypts more effectively than a colony of Noisy Miners, and insect outbreaks are implicated in some forms of dieback (Dow 1977; Ford and Bell 1980; Landsberg *et al.* 1990). The lack of shrubs in site 2 and the cleared area beneath transmission wires (Appendix 2) probably contributed to the dominance of Noisy Miners at this site, suggesting that exotic shrubs may mitigate the influence of aggressive competition. However, this hypothesis remains to be tested and the evidence in my study for an effect on the colonial Noisy Miner is weak.

The herbicides used in weed removal could potentially interfere with fungi important to nutrient cycles in temperate woodland (Tommerup and Bougher 2000). Ideally, any assessment of adverse impacts on woodland birds from herbicide use would be informed by regional evidence of ecosystem threats from weed invasion, gathered before intervening.

Adair and Groves (1998) noted the importance of thresholds of weed invasion when considering the risks posed to biodiversity from alien plants. Adverse impacts on ecosystem function in temperate woodland, arising from exotic shrub invasion, may depend on invasion reaching

certain thresholds. These thresholds may be beyond the levels of invasion considered in this study (Table 4.1) and this limitation should be kept in mind when considering the results.

### 5.5 Study findings - implications for managers of urban woodland in Canberra

The possibility that exotic shrubs may be of some benefit to native birds was not discussed in the final version of the *ACT Weeds Strategy* (ACT Government 1996b), although the *ACT Nature Conservation Strategy* noted that environmental weeds may provide food, nest sites and protection from predators (ACT Government 1998). The *ACT Weeds Strategy* did, however, outline the importance of weed control measures being integrated with site rehabilitation and suggested that integrated control programs, although more complex, are more effective in the long term (ACT Government 1996b).

Several of the woodland sites where I sampled birds were subject to removal of exotic shrubs (Appendix 3). Sites 11 and 12 of this study were weeded intensively after I completed 20 minute area-searches in June 1998 (Appendix 3). The fact that this weeding of sites 11 and 12 took place after my bird sampling was complete presents managers of this woodland with the opportunity to monitor the impact on birds in these areas. This monitoring would complement the actions outlined in *Canberra Nature Park - Draft Management Plan 1996* (ACT Government 1996a), which stated that areas where exotic plants have been removed may need rehabilitation to minimise erosion and prevent re-establishment of pest plants (ACT Government 1996a). The purpose of monitoring birds in these sites would be to determine if the reduced abundance of some native birds noted in site 4 following intense weed control (Figure 4.3 and Figure 4.8), is repeated. The weeded sites would also provide managers with the opportunity to experiment with integrated control programs, as recommended in the *ACT Weeds Strategy* (ACT Government 1996b). As large amounts of shrub cover have been removed, the planting of local, native shrub species could be considered in this process.

A final point which emerges from my study is that excluding *E. melliodora-E. blakeyi* woodland from the scope of *Action Plan No. 10, Yellow Box/Red Gum Grassy Woodland - An endangered ecological community* (ACT Government 1999), because of the presence of exotic shrubs in the understorey, may have been short sighted. This argument is based on the possibility that these areas are important refuges for a number of small native birds, such as the Speckled Warbler. The

decision to exclude patches of “otherwise significant woodland” because the understorey is made up of exotic plants does not adequately consider the high value of structurally complex, albeit exotic understoreys, to native birds.

## 5.6 Conclusion

Exotic shrubs in *E. melliodora*-*E. blakelyi* woodland provided some birds with food and increased the value of this habitat for a sub-set of small native birds, probably because they mitigated the impact of predation. These benefits may operate at certain thresholds of invasion, although more work is needed to confirm this hypothesis. I believe that the rationale for removing exotic shrubs in temperate urban woodland in Canberra needs to be stronger than is currently the case. This argument is based on the risks of weed control, which reduces habitat complexity in the short to medium term, and the unknown implications for habitat value and ecosystem function in the long term. While there are examples throughout Australia of exotic shrubs posing dire threats to ecosystems through structural changes, evidence of this effect in Canberra is lacking. In fact, rather than being plants in the wrong place at the wrong time, for some small native birds, exotic shrubs in Canberra woodland are in the right place at the right time.

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## Appendix 1 – Description of groundcover vegetation

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### Description of groundcover at woodland sites

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Site 1 – Most common native grass *Danthonia spp.* *Poa spp* and *Bothriochloa spp* also present. *Helichrysum spp* most common native daisies. Modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Nasella spp*; *Acaena spp*; *Briza spp*; *Bromus spp*; and *Trifolium spp*.

Site 2 – Most common native grass *Danthonia spp.* *Poa spp* and *Stipa spp* present, and largest patch of *Themeda spp* of all sites. *Helichrysum spp* most common native daisies. Modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Phalaris spp*; *Nasella spp*; *Acaena spp*; *Avena spp*; *Briza spp*; *Bromus spp*; and *Trifolium spp*.

Site 3 – *Poa spp* most common native grass. *Danthonia spp* , *Stipa spp* and *Themeda spp* present. Modified by exotic understorey plants. Common exotics included: *Hypericum spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Nasella spp*; *Acaena spp*; *Bromus spp*; and *Trifolium spp*.

Site 4 – Most common native grass *Danthonia spp*, with some *Poa spp*, *Stipa spp* and *Bothriochloa spp* present. Modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Nasella spp*; *Acaena spp*; *Briza spp*; *Bromus spp*; *Avena spp* and *Trifolium spp*. Endangered daisy *Rutidosia leptorrhynchoides* present.

Site 5– Most common native grass *Danthonia spp*, with some *Poa spp* present, small number of *Themeda spp*. Modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Phalaris spp*; *Nasella spp*; *Acaena spp*; *Avena spp*; *Briza spp*; *Bromus spp*; and *Trifolium spp*.

Site 6 – Most common native grass *Danthonia spp*, with some *Poa spp*, *Stipa spp* and *Themeda spp* present .Modified by exotic understorey plants. Common exotics included: *Hypericum spp*; *Hypochoeris spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Nasella spp*; *Acaena spp*; *Avena spp*; *Bromus spp*; and *Trifolium spp*.

Site 7 – Most common native grass *Danthonia spp*, with some *Poa spp* and *Stipa spp* present. *Helichrysum spp* most common native daisies. Heavily modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Phalaris spp*; *Nasella spp*; *Acaena spp*; *Briza spp*; *Bromus spp*; *Avena spp*; *Hypericum spp*; and *Trifolium spp*.

Site 8 – Most common native grass *Danthonia spp.* *Helichrysum spp* most common native daisies. Heavily modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Hypericum spp*; *Nasella spp*; *Acaena spp*; *Avena spp*; *Briza spp*; *Bromus spp*; and *Trifolium spp*.

Site 9 – Most common native grass *Danthonia spp.* *Poa spp*, *Bothriochloa spp* and *Stipa spp* present. *Helichrysum spp* most common native daisies. Heavily modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Hypericum spp*; *Nasella spp*; *Acaena spp*; *Avena spp*; *Briza spp*; *Bromus spp*; and *Trifolium spp*.

Site 10 – Most common native grass *Danthonia spp.* *Stipa spp* present. *Helichrysum spp* most common native daisies. Heavily modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Hypericum spp*; *Nasella spp*; *Acaena spp*; *Avena spp*; *Briza spp*; *Bromus spp*; and *Trifolium spp*.

Site 11 – Most common native grass *Danthonia spp*, with some *Poa spp* and *Stipa spp* present. Small number of *Themeda spp.* *Helichrysum spp* most common native daisies. Heavily modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Hypericum spp*; *Rumex spp*; *Nasella spp*; *Acaena spp*; *Avena spp*; *Briza spp*; *Bromus spp*; and *Trifolium spp*.

Site 12 – Most common native grass *Danthonia spp*, with some *Poa spp* and *Stipa spp* present. *Helichrysum spp* most common native daisies. Heavily modified by exotic understorey plants. Common exotics included: *Enneapogon spp*; *Chondrilla spp*; *Plantago spp*; *Rumex spp*; *Phalaris spp*; *Nasella spp*; *Acaena spp*; *Briza spp*; *Bromus spp*; and *Trifolium spp*.

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## Appendix 2 – Urban infrastructure in or near (<50m distance) sites

Site No.	Transmission wires	Dirt roads or permanent tracks	Open drains	Fences
1	Yes (near)	Yes (in)	No	Yes (in)
2	Yes (in)	Yes (in)	Yes (near)	Yes (near)
3	Yes (near)	Yes (in)	Yes (near)	Yes (in)
4	Yes (near)		Yes (near)	Yes (in)
5	Yes (near)	Yes (in)	Yes (near)	Yes (near)
6	Yes (near)	Yes (in)	Yes (near)	Yes (in)
7	Yes (near)	Yes (in)	No	Yes (in)
8	No	Yes (in)	No	Yes (near)
9	Yes (near)	Yes (in)	Yes (near)	Yes (near)
10	Yes (in)	Yes (near)	Yes (in)	Yes (in)
11	Yes (near)	Yes (in)	Yes (near)	Yes (near)
12	Yes (near)	Yes (in)	Yes (near)	Yes (near)

### Appendix 3 – Recent weed control in sites (1996 to 1998<sup>3</sup>)

Site No.	Description
1	No recent weed control. Stumps of removed exotic shrubs evident and one or two dead exotic shrubs present.
2	No recent weed control. Stumps of removed exotic shrubs evident.
3	Stumps of removed exotic shrubs evident. Some cut and paint removal of regenerating exotic weeds between July 1997 and June 1998 (i.e. young exotic shrubs removed).
4	Sprayed and intensively weeded in June 1996 and June 1997, with significant removal of exotic shrub cover from site (from 24% exotic shrub cover to 1%).
5	No recent weed control. Stumps of removed exotic shrubs evident and some dead exotic shrubs evident.
6	Exotic shrubs sprayed in 1996 and 1997 (before July 1997).
7	No sign of weed control.
8	No sign of weed control.
9	Some weed control between July 1997 and June 1998, with <i>Pyracantha spp</i> shrubs removed. Blackberry patch unweeded in this time. Weed control is estimated <sup>1</sup> to have removed a significant amount of exotic shrub cover from site.
10	No sign of weed control.
11	Minor weed control in 1998, with some exotic shrubs removed. Weed control is estimated <sup>2</sup> to have had little impact on overall level of exotic shrub cover in site.
12	No sign of weed control in site.

(1) Two large clumps of *Pyracantha spp* were completely removed from opposite ends of the site, altering the understorey.

(2) Only a few individual shrubs were removed from the edge of the site.

(3) Sites 11 and 12 were intensively weeded in 1999, with significant amounts of exotic shrub cover removed.

**Appendix 4 – Mean number of people, dogs, cats, rabbits and hares, and foxes, observed within woodland plots during 20 minute area-searches, July 1997 to June 1998**

Site No.	People	SE(1)	Dogs(2)	SE	Cats	SE	Rabbits & Hares	SE	Foxes	SE
1	1.91	± 0.27	0.28	± 0.07	0	± 0	0	± 0	0	± 0
2	0.89	± 0.20	0.25	± 0.10	0	± 0	0.05	± 0.03	0	± 0
3	0.40	± 0.10	0.24	± 0.08	0.04	± 0.03	0	± 0	0	± 0
4	0.55	± 0.15	0.25	± 0.08	0	± 0	0	± 0	0	± 0
5	0.95	± 0.18	0.27	± 0.07	0	± 0	0.05	± 0.03	0	± 0
6	0.35	± 0.10	0.13	± 0.06	0	± 0	0	± 0	0	± 0
7	0.09	± 0.04	0.04	± 0.03	0	± 0	0.25	± 0.08	0	± 0
8	0.20	± 0.09	0.04	± 0.04	0	± 0	0.39	± 0.08	0.06	± 0.03
9	2.04	± 0.28	0.79	± 0.15	0.02	± 0.02	0.02	± 0.02	0	± 0
10	0.02	± 0.02	0.04	± 0.04	0	± 0	0.45	± 0.12	0.11	± 0.05
11	0.29	± 0.09	0.16	± 0.07	0.02	± 0.02	0.11	± 0.04	0	± 0
12	1.25	± 0.21	0.43	± 0.10	0	± 0	0	± 0	0	± 0

(1) SE = Standard error.

(2) 78% of dogs observed in woodland plots were not on leads and 2% were seen chasing Kangaroos.

**Appendix 5 - Mean abundance for birds recorded in woodland sites with four levels of exotic shrub cover, July 1997 to June 1998 (Family Anatidae to Family Halcyonidae)**

<b>Taxon</b>	<b>Nil</b>	<b>SE</b>	<b>Light</b>	<b>SE</b>	<b>Mod.</b>	<b>SE</b>	<b>Dense</b>	<b>SE</b>
<b>Family Anatidae</b>								
<i>Chenonetta jubata</i>	0.22	± 0.07	0.21	± 0.07	0.14	± 0.05	0.05	± 0.03
<i>Anas superciliosa</i>	0.04	± 0.02	0.02	± 0.01	0.02	± 0.01	0.01	± 0.01
<b>Family Ardeidae</b>								
<i>Egretta novaehollandiae</i>	0	± 0	0	± 0	0.01	± 0.01	0	± 0
<b>Family Accipitridae</b>								
<i>Accipiter cirrhocephalus</i>	0	± 0	0.01	± 0.01	0.05	± 0.02	0.01	± 0.01
<i>Accipiter fasciatus</i>	0.01	± 0.01	0.01	± 0.01	0	± 0	0	± 0
<b>Family Falconidae</b>								
<i>Falco longipennis</i>	0.03	± 0.01	0.01	± 0.01	0	± 0	0	± 0
<b>Family Columbidae</b>								
<i>Phaps chalcoptera</i>	0	± 0	0.03	± 0.01	0.02	± 0.01	0	± 0
<i>Ocyphaps lophotes</i>	0.06	± 0.03	0.25	± 0.09	0.55	± 0.09	0.73	± 0.11
<b>Family Cacatuidae</b>								
<i>Callocephalon fimbriatum</i>	0.28	± 0.07	0.11	± 0.04	0.07	± 0.04	0.28	± 0.1
<i>Cacatua roseicapilla</i>	0.56	± 0.09	1.09	± 0.12	1.15	± 0.17	1.02	± 0.16
<i>Cacatua galerita</i>	0.12	± 0.03	0.89	± 0.22	0.06	± 0.03	0.28	± 0.09
<b>Family Psittacidae</b>								
<i>Alisterus scapularis</i>	1.25	± 0.21	0.37	± 0.1	0.71	± 0.17	0.38	± 0.11
<i>Platycercus elegans</i>	4.14	± 0.25	3.94	± 0.2	6.12	± 0.38	6.05	± 0.33
<i>Platycercus eximius</i>	2.32	± 0.19	1.58	± 0.14	0.9	± 0.14	1.5	± 0.19
<i>Psephotus haematonotus</i>	0.01	± 0.01	0	± 0	0	± 0	0	± 0
<b>Family Cuculidae</b>								
<i>Cuculus pallidus</i>	0	± 0	0.01	± 0.01	0.01	± 0.01	0	± 0
<i>Cacomantis flabelliformis</i>	0.03	± 0.01	0.01	± 0.01	0.01	± 0.01	0.02	± 0.01
<b>Family Podargidae</b>								
<i>Podargus strigoides</i>	0	± 0	0.01	± 0.01	0	± 0	0	± 0
<b>Family Halcyonidae</b>								
<i>Dacelo novaeguineae</i>	0.24	± 0.04	0.23	± 0.04	0.1	± 0.04	0.08	± 0.03
<i>Todiramphus sanctus</i>	0.02	± 0.01	0.04	± 0.01	0.03	± 0.02	0.04	± 0.02



**Appendix 6 -Mean abundance for birds recorded in woodland sites with four levels of exotic shrub cover, July 1997 to June 1998 (Family Coraciidae to Family Meliphagidae)**

	Nil	SE	Light	SE	Mod.	SE	Dense	SE
<b>Family Coraciidae</b>								
<i>Eorystomus orientalis</i>	0.02	± 0.01	0.01	± 0.01	0.05	± 0.02	0.03	± 0.02
<b>Family Climacteridae</b>								
<i>Cornobates leucophaeus</i>	0.05	± 0.02	0.03	± 0.01	0.02	± 0.01	0.12	± 0.03
<i>Climacteris picumnus</i>	0	± 0	0	± 0	0.04	± 0.03	0	± 0
<b>Family Maluridae</b>								
<i>Malurus cyaneus</i>	0.48	± 0.09	4.24	± 0.33	4.1	± 0.45	2.81	± 0.26
<b>Family Pardalotidae</b>								
<i>Pardalotus punctatus</i>	0.59	± 0.14	0.49	± 0.13	1.05	± 0.42	1.04	± 0.2
<i>Pardalotus striatus</i>	3.04	± 0.7	3.08	± 0.76	3.03	± 1.14	2.58	± 0.75
<i>Sericomis frontalis</i>	0.01	± 0.01	0.04	± 0.02	0.17	± 0.05	0.4	± 0.09
<i>Chthonicola sagittata</i>	0.08	± 0.02	0.51	± 0.07	0.37	± 0.09	0.4	± 0.1
<i>Gerygone fusca</i>	0.01	± 0.01	0.01	± 0.01	0.05	± 0.03	0	± 0
<i>Gerygone olivacea</i>	0.03	± 0.01	0.16	± 0.03	0.07	± 0.03	0.11	± 0.03
<i>Acanthiza pusilla</i>	0.09	± 0.03	0.19	± 0.04	0.11	± 0.04	0.62	± 0.11
<i>Acanthiza reguloides</i>	0.92	± 0.13	0.73	± 0.13	0.67	± 0.15	0.6	± 0.15
<i>Acanthiza chrysorrhoa</i>	0.28	± 0.08	1.23	± 0.17	0.38	± 0.11	0.05	± 0.04
<i>Acanthiza lineata</i>	0.26	± 0.07	0.75	± 0.11	0.25	± 0.09	0.46	± 0.12
<i>Acanthiza nana</i>	0	± 0	0.02	± 0.01	0.03	± 0.03	0.03	± 0.03
<i>Smicromis brevirostris</i>	1.6	± 0.13	2.26	± 0.14	1.68	± 0.24	1.5	± 0.19
<b>Family Meliphagidae</b>								
<i>Anthochaera carunculata</i>	0.78	± 0.14	0.5	± 0.06	0.89	± 0.14	0.8	± 0.12
<i>Philemon comiculatus</i>	0.39	± 0.06	0.46	± 0.06	0.5	± 0.1	0.48	± 0.1
<i>Manorina melanocephala</i>	1.28	± 0.15	0.42	± 0.08	0.4	± 0.11	0	± 0
<i>Lichenostomus chrysops</i>	0.36	± 0.14	0.22	± 0.06	0.27	± 0.12	0.46	± 0.17
<i>Lichenostomus leucotis</i>	0.04	± 0.01	0.12	± 0.03	0.05	± 0.03	0.08	± 0.04
<i>Lichenostomus fuscus</i>	0.01	± 0.01	0.05	± 0.02	0	± 0	0.01	± 0.01
<i>Lichenostomus penicillatus</i>	0.01	± 0.01	0.03	± 0.02	0.1	± 0.06	0	± 0
<i>Melithreptus lunatus</i>	0.01	± 0.01	0.01	± 0.01	0	± 0	0.08	± 0.05
<i>Acanthorhynchus tenuirostris</i>	0.01	± 0.01	0.01	± 0.01	0	± 0	0.11	± 0.04

**Appendix 7 - Mean abundance for birds recorded in woodland sites with four levels of exotic shrub cover, July 1997 to June 1998 (Family Petroicidae to Family Passeridae)**

<b>Taxon</b>	<b>Nil</b>	<b>SE</b>	<b>Light</b>	<b>SE</b>	<b>Mod.</b>	<b>SE</b>	<b>Dense</b>	<b>SE</b>
<b>Family Petroicidae</b>								
<i>Petroica multicolor</i>	0.08	± 0.02	0.09	± 0.03	0	± 0	0.04	± 0.02
<i>Eopsaltria australis</i>	0	± 0	0	± 0	0.03	± 0.02	0	± 0
<b>Family Neosittidae</b>								
<i>Daphoenositta chrysoptera</i>	0.08	± 0.05	0.08	± 0.04	0.33	± 0.12	0.08	± 0.06
<b>Family Pachycephalidae</b>								
<i>Pachycephala pectoralis</i>	0.05	± 0.02	0.11	± 0.02	0.11	± 0.03	0.17	± 0.04
<i>Pachycephala rufiventris</i>	0.02	± 0.01	0.05	± 0.02	0.1	± 0.04	0.05	± 0.02
<i>Colluricincla harmonica</i>	0.04	± 0.01	0.07	± 0.02	0.1	± 0.03	0.06	± 0.03
<b>Family Dicruridae</b>								
<i>Myiagra rubecula</i>	0.03	± 0.02	0.1	± 0.04	0.04	± 0.03	0.12	± 0.04
<i>Rhipidura leucophrys</i>	0.03	± 0.01	0.02	± 0.01	0.01	± 0.01	0.02	± 0.01
<i>Rhipidura fuliginosa</i>	0.45	± 0.07	0.92	± 0.09	1.08	± 0.12	1.35	± 0.18
<i>Grallina cyanoleuca</i>	0.52	± 0.08	0.1	± 0.04	0.24	± 0.07	0.04	± 0.04
<b>Family Campephagidae</b>								
<i>Coracina novaehollandiae</i>	0.25	± 0.04	0.17	± 0.04	0.36	± 0.07	0.23	± 0.06
<i>Lalage sueurii</i>	0	± 0	0	± 0	0.14	± 0.07	0	± 0
<b>Family Oriolidae</b>								
<i>Oriolus sagittatus</i>	0.03	± 0.01	0.01	± 0.01	0.03	± 0.02	0.01	± 0.01
<b>Family Artamidae</b>								
<i>Artamus cyanopterus</i>	0	± 0	0.09	± 0.06	0.2	± 0.09	0	± 0
<i>Cracticus torquatus</i>	0	± 0	0	± 0	0.02	± 0.01	0.01	± 0.01
<i>Gymnorhina tibicen</i>	1.31	± 0.11	1.17	± 0.11	1.04	± 0.13	0.92	± 0.11
<i>Strepera graculina</i>	1.39	± 0.14	0.99	± 0.13	1.15	± 0.23	2.91	± 0.99
<i>Strepera versicolor</i>	0.04	± 0.02	0.02	± 0.02	0.01	± 0.01	0.03	± 0.02
<b>Family Corvidae</b>								
<i>Corvus coronoides</i>	0.46	± 0.13	0.09	± 0.02	0.15	± 0.04	0.09	± 0.03
<b>Family Corcoracidae</b>								
<i>Corcorax melanorhamphos</i>	0.6	± 0.14	0.34	± 0.12	0.19	± 0.15	0	± 0
<b>Family Ptilonorhynchidae</b>								
<i>Ptilonorhynchus violaceus</i>	0.01	± 0.01	0.01	± 0.01	0	± 0	0	± 0
<b>Family Passeridae</b>								
<i>Taeniopygia bichenovii</i>	0.01	± 0.01	0.01	± 0.01	0	± 0	0.02	± 0.02
<i>Neochmia temporalis</i>	0.05	± 0.03	0.11	± 0.04	0.38	± 0.13	1.54	± 0.3
<i>Passer domesticus</i>	0.01	± 0.01	0.04	0.02	0	± 0	0.03	± 0.02

**Appendix 8 - Mean abundance for birds recorded in woodland sites with four levels of exotic shrub cover, July 1997 to June 1998 (Family Dicaeidae to Family Sturnidae)**

<b>Taxon</b>	<b>Nil</b>	<b>SE</b>	<b>Light</b>	<b>SE</b>	<b>Mod.</b>	<b>SE</b>	<b>Dense</b>	<b>SE</b>
<b>Family Dicaeidae</b>								
<i>Dicaeum hirundinaceum</i>	0	± 0	0.02	± 0.01	0.05	± 0.03	0.01	± 0.01
<b>Family Hirundinidae</b>								
<i>Hirundo neoxena</i>	0.03	± 0.02	0.01	± 0.01	0	± 0	0	± 0
<b>Family Zosteropidae</b>								
<i>Zosterops lateralis</i>	1.77	± 0.37	1.57	± 0.34	1.22	± 0.34	3.92	± 0.73
<b>Family Muscicapidae</b>								
<i>Turdus merula</i>	0.04	± 0.02	0.08	± 0.02	0.28	± 0.05	0.88	± 0.11
<b>Family Sturnidae</b>								
<i>Stumus vulgaris</i>	1.27	± 0.21	0.9	± 0.16	3.76	± 0.54	0.04	± 0.02
<i>Acridotheres tristis</i>	0.23	± 0.04	1.19	± 0.19	0.39	± 0.14	0.24	± 0.07

Appendix 9 - Means of all birds observed in woodland sites (1)

SPP no.	SITE01	SITE02	SITE03	SITE04	SITE05	SITE06	SITE07	SITE08	SITE09	SITE10	SITE11	SITE12
188	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.018	0.000	0.000	0.000
208	0.109	0.036	0.000	0.000	0.000	0.000	0.000	0.093	0.018	0.018	0.018	0.000
202	0.418	0.473	0.000	0.000	0.000	0.000	0.491	0.370	0.000	0.291	0.000	0.091
221	0.018	0.000	0.018	0.000	0.018	0.000	0.018	0.000	0.000	0.000	0.000	0.000
222	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.056	0.091	0.000	0.000	0.018
235	0.000	0.000	0.091	0.018	0.018	0.000	0.018	0.000	0.000	0.000	0.000	0.000
34	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.111	0.000	0.036	0.000	0.000
43	0.073	0.182	0.000	0.000	0.491	0.036	0.109	0.352	0.582	0.509	1.345	0.145
268	0.055	0.345	0.182	0.564	0.000	0.436	0.000	0.000	0.109	0.036	0.200	0.236
273	1.364	0.182	0.382	0.182	1.345	0.182	0.691	2.148	1.291	0.964	1.691	0.345
269	0.036	0.182	0.164	0.091	2.255	0.455	0.436	0.426	0.091	0.036	0.364	0.204
281	0.545	3.836	0.309	0.255	0.091	0.418	0.018	0.963	1.164	0.218	0.418	0.345
282	5.291	3.691	4.636	2.473	4.782	3.055	2.655	5.259	6.982	5.091	5.618	6.491
288	1.945	4.582	1.873	0.855	1.745	2.836	0.564	1.167	0.945	0.836	1.945	0.855
295	0.036	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
337	0.000	0.000	0.000	0.000	0.055	0.000	0.000	0.000	0.000	0.018	0.000	0.000
338	0.000	0.000	0.000	0.127	0.018	0.018	0.000	0.000	0.018	0.000	0.018	0.018
313	0.000	0.000	0.000	0.000	0.036	0.000	0.000	0.000	0.000	0.000	0.000	0.000
322	0.327	0.218	0.127	0.255	0.236	0.200	0.364	0.111	0.200	0.000	0.145	0.018
326	0.000	0.036	0.000	0.036	0.019	0.000	0.091	0.037	0.056	0.000	0.036	0.036
318	0.000	0.018	0.018	0.036	0.000	0.000	0.000	0.056	0.000	0.091	0.018	0.036
357	0.000	0.109	0.000	0.000	0.000	0.000	0.000	0.037	0.000	0.000	0.000	0.000
424	0.455	0.218	0.164	0.182	0.182	0.327	0.164	0.037	0.327	0.382	0.382	0.091
430	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.291	0.000	0.000
991	0.000	0.000	0.000	0.145	0.000	0.182	0.127	0.000	0.491	0.073	0.455	1.273
380	0.018	0.036	0.055	0.200	0.036	0.127	0.182	0.000	0.000	0.000	0.073	0.000
392	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.055	0.000	0.000	0.000
398	0.000	0.055	0.055	0.073	0.091	0.109	0.164	0.074	0.073	0.127	0.091	0.236
401	0.018	0.036	0.036	0.000	0.000	0.000	0.182	0.037	0.036	0.164	0.073	0.036
408	0.091	0.000	0.000	0.073	0.073	0.109	0.000	0.093	0.055	0.145	0.018	0.109
365	0.073	0.018	0.036	0.000	0.018	0.018	0.382	0.000	0.073	0.000	0.218	0.018
361	0.436	0.200	0.545	0.600	0.673	0.436	1.564	1.019	0.800	1.382	1.691	1.056
364	0.036	0.000	0.073	0.018	0.018	0.000	0.055	0.000	0.000	0.018	0.036	0.000
529	0.309	0.109	0.745	0.764	0.600	3.545	9.436	3.463	1.036	7.164	3.036	2.564
488	0.000	0.036	0.000	0.000	0.000	0.164	0.000	0.000	0.345	0.000	0.073	0.745
504	0.255	0.000	0.000	0.055	0.345	0.182	1.182	0.333	0.309	0.345	0.400	0.418
465	2.455	0.418	1.236	2.127	2.236	1.909	2.400	2.500	0.345	2.982	1.891	0.982
463	0.055	0.000	0.000	0.000	0.018	0.000	0.036	0.000	0.000	0.109	0.000	0.000
453	0.091	0.000	0.018	0.000	0.145	0.000	0.255	0.241	0.000	0.145	0.200	0.018
475	0.018	0.073	0.073	0.200	0.091	0.055	0.436	0.185	0.109	0.091	0.618	0.545
484	1.127	0.164	1.309	1.127	1.073	0.400	1.327	0.111	0.145	1.200	0.636	0.582
486	0.218	0.000	0.436	0.491	1.182	0.691	3.036	0.000	0.000	0.764	0.109	0.000
471	0.000	0.000	0.000	0.000	0.000	0.036	0.000	0.038	0.000	0.055	0.000	0.000
470	0.455	0.091	0.218	0.291	0.545	0.564	1.636	0.241	0.145	0.364	0.600	0.345
549	0.273	0.000	0.036	0.000	0.000	0.000	0.273	0.037	0.000	0.673	0.164	0.000

**Appendix 9 - Means of all birds observed in woodland sites (1) (continued)**

558	0.091	0.018	0.018	0.055	0.073	0.000	0.036	0.019	0.018	0.019	0.109	0.127
555	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.073	0.000	0.000
638	1.345	0.309	0.455	0.909	0.800	0.273	0.709	0.204	0.727	1.073	0.855	0.759
645	0.527	0.436	0.218	0.400	0.745	0.218	0.709	0.185	0.764	0.236	0.745	0.236
634	0.236	4.582	0.345	0.000	0.418	0.000	0.000	1.278	0.000	0.800	0.000	0.000
614	0.491	0.273	0.364	0.327	0.218	0.255	0.291	0.111	0.273	0.273	0.182	0.764
617	0.000	0.036	0.018	0.093	0.218	0.073	0.200	0.000	0.018	0.091	0.018	0.145
613	0.000	0.036	0.018	0.000	0.091	0.000	0.127	0.000	0.000	0.000	0.000	0.018
625	0.000	0.000	0.018	0.000	0.000	0.000	0.073	0.037	0.000	0.200	0.000	0.000
578	0.000	0.000	0.055	0.000	0.000	0.000	0.036	0.000	0.000	0.000	0.091	0.055
591	0.000	0.000	0.000	0.018	0.000	0.000	0.036	0.000	0.000	0.000	0.018	0.200
564	0.000	0.000	0.000	0.000	0.055	0.000	0.018	0.000	0.091	0.018	0.000	0.018
565	0.273	0.745	0.745	0.636	0.436	0.855	0.236	0.444	0.109	1.982	1.291	0.764
976	3.218	3.473	2.000	3.564	1.873	3.818	6.345	0.296	3.273	2.836	1.782	3.164
574	0.509	0.400	1.436	4.800	0.164	5.273	0.691	0.167	1.091	1.364	2.745	5.018
995	0.000	0.000	0.018	0.000	0.109	0.056	0.000	0.000	0.000	0.000	0.000	0.055
662	0.036	0.000	0.000	0.182	0.036	0.327	0.000	0.093	0.018	0.745	2.800	0.327
655	0.000	0.000	0.000	0.036	0.000	0.036	0.000	0.000	0.000	0.000	0.036	0.000
999	4.582	0.036	0.218	0.055	0.145	0.200	0.200	3.130	0.036	7.545	0.000	0.073
998	0.200	0.327	0.218	0.182	0.091	4.055	0.000	0.630	0.309	0.473	0.345	0.145
671	0.091	0.000	0.018	0.018	0.000	0.000	0.018	0.000	0.055	0.000	0.018	0.000
679	0.000	0.000	0.000	0.036	0.000	0.055	0.000	0.000	0.000	0.000	0.000	0.000
693	1.200	1.236	0.000	0.000	0.000	0.000	0.091	1.056	0.000	0.382	0.000	0.000
415	1.618	0.273	0.036	0.182	0.200	0.164	0.000	0.019	0.345	0.145	0.073	0.000
547	0.000	0.000	0.000	0.000	0.345	0.000	0.000	0.000	0.000	0.400	0.000	0.000
702	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.036	0.000	0.018	0.000
705	0.945	1.945	1.364	0.709	2.309	0.382	0.582	1.389	1.727	0.291	1.073	0.800
694	0.927	2.455	1.164	1.036	1.127	2.073	0.255	0.500	1.800	0.527	1.018	4.764
697	0.127	0.000	0.000	0.000	0.018	0.000	0.073	0.000	0.000	0.018	0.036	0.018
930	0.182	1.327	0.091	0.273	0.073	0.218	0.073	0.000	0.127	0.182	0.073	0.109

(1) Species code is from the Canberra Ornithologist Group's data sheet (attached).