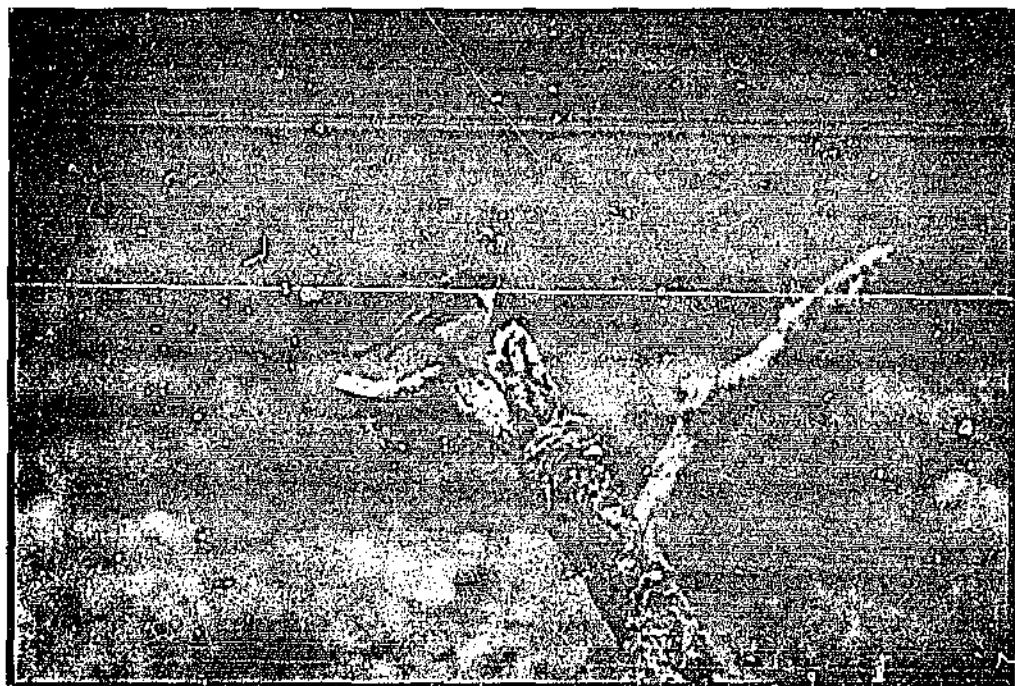


Foraging Ecology and Habitat Selection of the
Western Yellow Robin (*Eopsaltria griseogularis*) in
a Wandoo Woodland, Western Australia:
Conservation Ecology of a Declining Species.



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A thesis submitted in partial fulfilment of the requirements for the degree of Master of
Science in Environmental Management.

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ABSTRACT

This study examined the foraging ecology and habitat selection of the Western Yellow Robin in Wandoo Woodland at Dryandra Woodland, Western Australia. The foraging ecology component was comprised of an examination of foraging behaviour, perch-use selection and pounce-site characteristics. The habitat selection component was comprised of an examination of habitat characteristics of site occupancy and general nesting ecology. The implications of current management at Dryandra Woodland to the ecology of the Western Yellow Robin are discussed in reference to the findings of the present study.

Ground-pouncing was the dominant foraging behaviour throughout all seasons, with dead branches of live subcanopy Wandoo trees (*Eucalyptus wandoo*) and dead fallen timber as the dominant perch substrate throughout all seasons. However, interseasonal shifts were detected, exemplified by a decrease in proportion of ground pouncing, and an increase in foraging height and perching height during the warmer months. These results reflect seasonal changes in foraging behaviour influenced by the seasonality of invertebrate prey abundance, with lower abundance of leaf litter invertebrates and increased abundance of flying invertebrates in warmer months. Intraseasonal shifts in foraging behaviour reflect climatic differences between years.

Pounce site characteristics showed selection at multiple spatial scales. At the microhabitat scale, pounce sites had significantly more leaf litter and log material, and less bare ground than random points. At the macrohabitat scale, the distance of pounce sites to logs was significantly less than expected. Associations between foraging sites and logs represents selection for sites with a greater abundance of invertebrate prey associated with dead fallen timber. This pattern of selection is consistent throughout the year, indicating characteristics of ground pounce foraging locations remain the same year round regardless of the shift in foraging behaviours.

The habitat selection component of the study showed that sites occupied by Western Yellow Robins had higher canopy density, higher leaf litter and log density, higher proportions of Wandoo trees and *Gastrolobium* plants and higher fragment coefficients (indicating occupation of sites away from the woodland/agricultural ecotone). These results reflect selection of variables at multiple spatial scales; namely, selection for sites with abundant invertebrate prey habitat (microhabitat scale), selection

for highly productive habitat (macrohabitat scale), and selection for sites with a reduced edge effect (landscape scale).

Nest site characteristics showed selection at specific spatial scales. Although nest sites were located at a variety of heights, they were largely associated with the basal crown height of trees, affording them relatively unobstructed views of the ground to spot potential predators. Nesting trees were similar in height to surrounding trees, although nesting trees were almost always shorter than the highest of the surrounding trees. At a macrohabitat scale, habitat surrounding nest sites was no different to habitat at non-nest sites, indicating no selection for nest site habitat measured. Territory boundaries varied spatially and temporally, with a contraction of territory boundaries during the breeding season. Nests were normally located with the vicinity of the centre of the breeding territory.

The present study indicates the strong association of the Western Yellow Robin with the ground environment, with selection for dense leaf litter and logs at numerous spatial scales. The current Dryandra Management Plan strategies have the potential to detrimentally alter the ground environment on which the resource requirements of the Western Yellow Robins revolve. The impact of the proposed changes in land tenure of Dryandra Woodland to National Park would result in additional human pressure on the environment. The impact of an influx in weed invasion, introduction of the *Phytophthora* dieback fungus and vehicle disturbance by an increase in visitors, has the potential to detrimentally alter the ecology of the habitats occupied by Western Yellow Robins. Furthermore, the implementation of the current fire management plan and the push for tourism development has the potential to internally fragment Dryandra Woodland to the detriment of the Western Yellow Robin.

Future research needs to examine reasons for the continued decline in the distribution and abundance of the Western Yellow Robin in the wheatbelt. Such research should be undertaken at the landscape scale, by examining the effects of agricultural practices on the ecology of the species, such as the effect of remnant area, habitat loss, habitat fragmentation and isolation, grazing, weed invasion, altered ecosystem processes and inappropriate fire regimes. Research should also be undertaken on the impact of agricultural practices on the ground invertebrates of remnant native vegetation of the wheatbelt, as many of the above effects of agricultural practices has the potential to greatly disturb the ground environment. Such disturbance has the

potential to alter the abundance of ground invertebrates; an important foraging resource of the Western Yellow Robin.

I certify that this thesis does not, to the best of my knowledge and belief:

- (i) incorporate without acknowledgement any material previously submitted for a degree or diploma in any institution of higher education;
- (ii) contain any material previously published or written by another person except where due reference is made in the text; or
- (iii) contain any defamatory material.

Signed

A handwritten signature in black ink, appearing to read "Jarrad A. Cousin".

Jarrad A. Cousin

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PART I: BACKGROUND



Over the past half-billion years, the planet lost perhaps one species per million species each year, including everything from mammals to plants. Today, the annual rate of extinction is 1,000 to 10,000 times faster. If nothing more is done, one-fifth of all the plant and animal species now on earth could be gone or on the road to extinction by 2030. Being distracted and self-absorbed, as is our nature, we have not yet fully understood what we are doing. But future generations, with endless time to reflect, will understand it all and in painful detail. As awareness grows, so will their sense of loss.

-Edward O. Wilson,
"What Is Nature Worth?"
The Wilson Quarterly, Winter 2002

CHAPTER 1: INTRODUCTION

1.1 GENERAL INTRODUCTION

1.1.1 Overview of landscape modification

Anthropogenic landscape modification is the single-most threatening process to biodiversity on the face of the earth. On every continent there has been widespread modification of native ecosystems resulting in environmental degradation, major alterations to ecological communities and inevitable extinction of species (Burgman and Lindenmayer, 1998). With the onset of agricultural development, overexploitation of native ecosystems for agricultural and pastoral expansion has accelerated to such a degree that presently, the area of the world's deserts are increasing at a rate of 100,000km²/year (Sivertsen, 1995). An inevitable consequence of landscape-scale modification of native ecosystems is the fragmentation of remaining habitat.

Fragmentation has become somewhat of a panacea; "a proposed explanation intended to address a complex problem by trying to account for all possible contingencies but typically proving to be too broadly conceived and therefore oversimplified to be of any practical use" (Bunnell, 1999, page vii). It is beyond the scope of this thesis to cover all aspects of the term. Fragmentation can loosely be defined as the alteration in configuration of the landscape through natural or landuse modification (whether it be destroyed by severe weather events or cleared or burned etc.), whereby there is a separation of formerly continuous habitat into smaller, isolated fragments (modified definition of Villard *et al.*, 1999).

Habitat fragmentation has direct consequences to biota. It gives rise to a decrease in total habitat available, while isolating remaining fragments. Degree of isolation, time since isolation and connectivity are all factors that will influence the degree to which fragmentation affects the biota (Saunders *et al.*, 1991). Habitat fragmentation and isolation can also interrupt gene flow, affect population size and promote genetic inbreeding (Anciães and Marini, 2000). Changes in microclimate throughout the altered landscape also influences the vegetation of remaining fragments as well as affecting the fauna that rely on these fragments (Hobbs, 1993a). Alterations in radiation fluxes, wind patterns and water balance give rise to shifts in the equilibrium processes of remaining fragments (Saunders *et al.*, 1991). As a result of extensive clearing of native vegetation within agricultural regions, rising water tables, soil salinity and soil degradation have contributed to the degradation of many remaining fragments

of native vegetation throughout the wheatbelt of south-west and eastern Australia (Breckwoldt, 1986; Hobbs, 1993b).

1.1.2 A history of the wheatbelt of Western Australia

The wheatbelt of Western Australia covers approximately 140,000km² of the southwest corner of the state (Figure 1.1). It is an area that has been developed for agriculture over a very short time period. By 1890, only 500km² had been cleared, mostly around small towns just to the east of Perth (Saunders *et al.*, 1985; Saunders and Curry, 1990). Following the Second World War, the state government released land and provided incentives to clear vast areas of the present wheatbelt. By 1968, 130,000km² had been cleared (Saunders *et al.*, 1985), representing 93% of the wheatbelt.

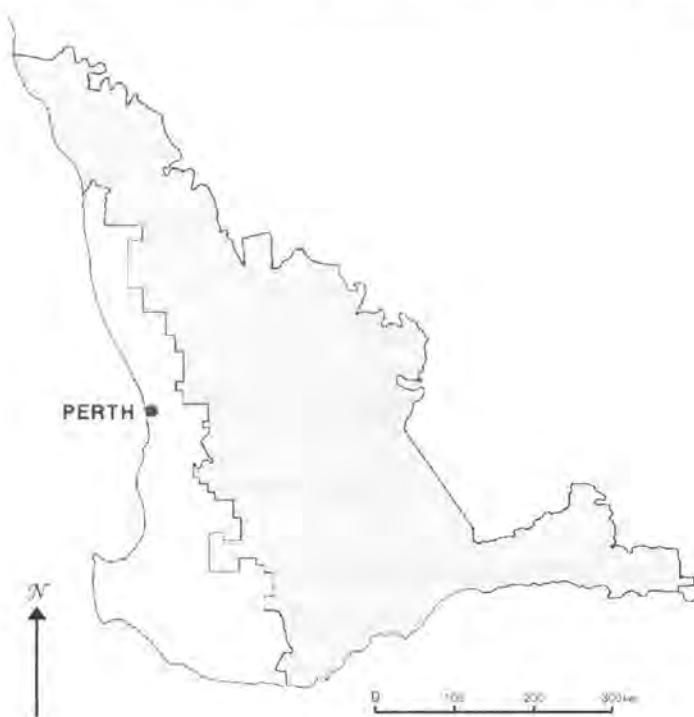


Figure 1.1 The wheatbelt of Western Australia (shaded grey).

During the period of rapid agricultural expansion, there were only a few areas set aside for nature conservation. At present there are thousands of remnants remaining in the wheatbelt. However, less than 15% of these are in conservation reserves (Saunders and Ingram, 1995). The majority of the remaining 85% of remnant vegetation is on private land, although it is not representative of the original vegetation. Components of these remnants are degraded by weed invasion, grazing by rabbits and domestic

stock, altered fire regimes and altered ecosystem processes (Table 1.1)(Saunders *et al.*, 1991; Hobbs, 1993a; Yates *et al.*, 1994; Yates *et al.*, 2000b).

Table 1.1 Ecosystem components affected by anthropogenic landuse modification (Table 2 in Hobbs and Hopkins, 1990, pg 97)

1. Vegetation	- above-ground biomass - below-ground organs
2. Propagule supply	- soil seed bank - on-plant store + vegetative propagules - dispersed seed - animal - wind, water
3. Other interacting organisms	- soil fungi, bacteria, mycorrhizae - soil and litter fauna - herbivores - pollinators - dispersal agents - other fauna
4. Soil properties	- structure/compaction - stability - pH
5. Hydrology	- water table - salinity
6. Nutrient Cycling	- nutrient pools - cycling processes
7. Population processes	- supply of propagules (2 above) - germination and establishment - altered mortality - competition
8. Community properties	- species richness - stability/resilience - vegetation dynamics

Remnants of native vegetation constituting the existing nature reserve system in the wheatbelt were primarily acquired during the 1960s and 1970s (Saunders and Curry, 1990). At present, there are 639 remnants gazetted as nature reserves, ranging in size between 0.4 ha and 309,000 ha with a median size of 114 ha (Saunders, 1989; Saunders and Curry, 1990).

1.1.3 Changes in the avifauna of the wheatbelt

As a result of clearing of 93% of the available habitat of the wheatbelt, there has been a decline in the abundance and distribution of a large number of birds. Ninety-five species of bird have declined in range and/or abundance during the twentieth century. This represents 49% of the bird species historically recorded throughout the wheatbelt (Saunders and Ingram, 1995). If we look more specifically at resident woodland avifauna, there is a recorded decline in 75% of all species (Saunders and Ingram, 1995).

Within the western wheatbelt (defined as the temperate slopes and plains of the Avon Wheatbelt, Mallee and Esperance IBRA regions), the “Action Plan for Australian Birds” (Garnett and Crowley, 2000) recognise 13 species of bird as endangered (Short-billed Black Cockatoo *Calyptorhynchus latirostris*), rare (Malleefowl *Leipoa ocellata*) or near threatened (including Crested Shrike-tit *Falcunculus frontatus*, White-browed Babbler *Pomatostomus superciliosus*, and Western Rosella *Platycercus icterotis*). Although classified as “of least concern” (Garnett and Crowley, 2000), the Western Yellow Robin (*Eopsaltria griseogularis*) is one such species that has declined in distribution and abundance throughout its range (Serventy, 1948; Masters and Milhinch, 1974; Saunders, 1989; Saunders and Curry, 1990; Cale, 1994b; Saunders and Ingram, 1995; Dilworth *et al.*, 2000).

1.2 WESTERN YELLOW ROBIN

1.2.1 Family Eopsaltridae (Australo-Papuan robins)

The Western Yellow Robin belongs to the Family Eopsaltridae. The Eopsaltridae (Australo-Papuan robins) are a group of perch-and-pounce insectivores found throughout Australia, New Guinea, Indonesia, New Zealand and other Pacific Island nations such as New Caledonia, Fiji, Samoa, Solomon Islands and Vanuatu (Oliver, 1974; Beehler *et al.*, 1986; Coates, 1990). The family contains 44 species from 13 genera, of which 21 species are found within Australia (Boles, 1988).

The species name (*Eopsaltria griseogularis*) means the “grey-throated dawn-harpist”. The Western Yellow Robin is similar in physical appearance to the Eastern Yellow Robin (*Eopsaltria australis*), differing in the plumage of the breast area. The Western Yellow Robin has a yellow ventral plumage from the belly to the base of the breast with a grey upper breast and throat, while the Eastern Yellow Robin has yellow ventral plumage from the belly to the throat. (Ford, 1979; Boles, 1988; Pizzey and Knight, 1997; Simpson *et al.*, 1999; Morcombe, 2000).

1.2.2 Systematics

The evolutionary relationship between the Eastern and Western Yellow Robins remains contentious, with many ornithologists divided on the presence of two species or of two subspecies, especially given their ecological similarity (Ford, 1963; Ford, 1971; Ford, 1979; Boles, 1988). Ford (1971) discusses the possible relationship between the two yellow robin species and the White-breasted Robin (*Eopsaltria georgiana*), found

in the south-west of Western Australia. Climatic oscillations during the latter phases of the Pleistocene are postulated to have provided the conditions for allopatric speciation through isolation processes that have given rise to the present pattern of species distribution of the three *Eopsaltria* species (Keast, 1984).

1.2.3 Western Yellow Robin habitat and distribution

The Western Yellow Robin is found in a diverse range of habitats from wet and dry sclerophyll forests, woodland, mallee and semi-arid acacia-scrub thickets (Ford, 1971). The species has two main populations, one in the southwest of Western Australia, with the second centred on the Eyre Peninsula in South Australia (Pizzey and Knight, 1997; Simpson *et al.*, 1999; Morcombe, 2000). In the wheatbelt, the Western Yellow Robin is most associated with open woodland vegetation types (Masters and Milhinch, 1974; Saunders and Curry, 1990; Saunders and Ingram, 1995).

There are two subspecies of Western Yellow Robin recognised (Figure 1.2); *griseogularis* found in the westernmost extent of its range; and *rosinae* found in the eastern half of the species' distribution. The two differ in plumage, with *griseogularis* having a yellow rump and *rosinae* having an olive-yellow rump (Schodde and Mason, 1999; Simpson *et al.*, 1999).



Figure 1.2 Map showing the distribution of the Western Yellow Robin (*Eopsaltria griseogularis*). The dark blue represents the distribution of the *rosinae* subspecies with the light blue representing the distribution of the *griseogularis* subspecies. The hatching region represents the zone of intergradation between the two subspecies (modified from Schodde & Mason, 1999).

1.2.4 The Western Yellow Robin as an appropriate study subject

The Western Yellow Robin is one of many species of bird that has declined in distribution and abundance throughout the wheatbelt. Saunders & Ingram (1995) used frequency of observation data to compare present and past distribution of wheatbelt birds. For the Western Yellow Robin, the frequency of observation decreased from 80% between 1900 and 1937, to 17% in 1990 (Saunders and Ingram, 1995).

The decline in the frequency of observation is most likely due to a decrease in the availability of appropriate habitat. Woodlands of the wheatbelt were preferentially cleared for agricultural development due to their highly productive soils (Saunders and Ingram, 1995) and as a result, up to 97% of the various woodland vegetation associations of the southwest/wheatbelt were cleared (Hobbs and Mooney, 1998). Evidence suggests that the Western Yellow Robin continues to decline in even the largest remnants of native vegetation in the wheatbelt. By understanding the ecology of the species, it may be possible to ascertain the processes leading to the observed decline in distribution and abundance.

1.3 AIMS AND THESIS OUTLINE

The primary research goal of the present study is to improve the ecological knowledge of the Western Yellow Robin. Specifically, the study aims to identify those factors contributing to the foraging ecology of the Western Yellow Robin in Dryandra Woodland, including seasonal changes in foraging behaviour, characteristics of perch-use selection and an examination of the characteristics of ground-pounce foraging locations. Factors contributing to habitat selection of the Western Yellow Robin in Dryandra Woodland will also be examined, including habitat associations of occupied sites and nesting ecology.

The thesis is divided into eight chapters (Figure 1.3). The first two chapters provide a background to the study; one being an introductory chapter while the second covers the description of study area and general methods. Chapters three through to five cover aspects of foraging ecology including a chapter each on seasonal changes in foraging behaviour, perch-use selection and pounce-site characteristics of the Western Yellow Robin. Chapters six and seven cover aspects of habitat selection, namely a chapter on site occupancy characteristics and a chapter on nesting ecology. The final chapter synthesises all the findings and presents overall conclusions.

Each of the foraging ecology and habitat selection chapters are structured as individual papers with an introduction, methods, results and discussion section. Citations used within each chapter appear in the reference section following the conclusion chapter.

Following the first mention of the Western Yellow Robin in chapters three to eight, the abbreviated code “WYR” (plural WYRs) will be utilised.

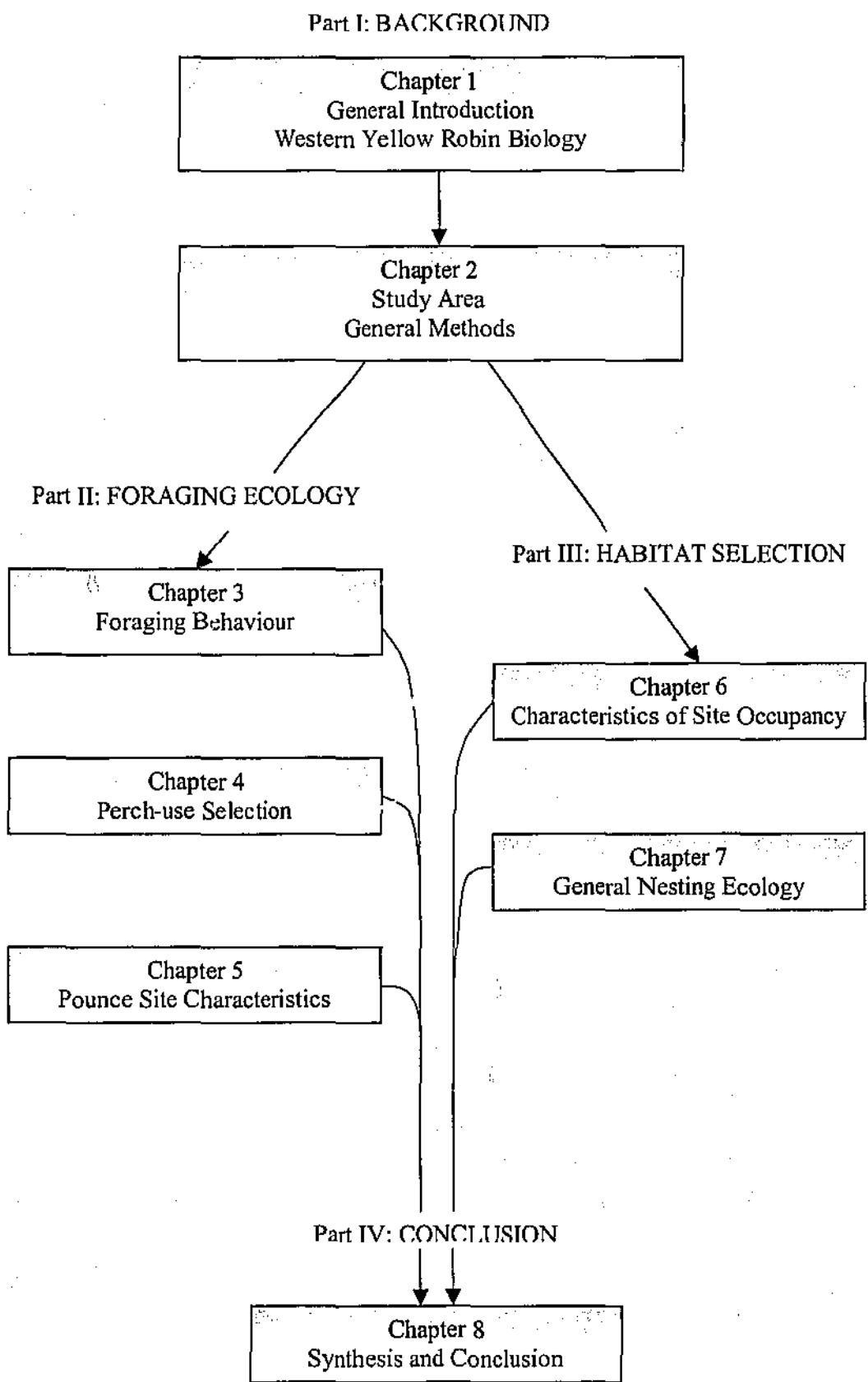


Figure 1.3 Thesis outline

CHAPTER 2: STUDY AREA AND GENERAL METHODS

2.1 STUDY AREA

2.1.1 Introduction

The present study was undertaken in Dryandra State Forest ($32^{\circ}49'S$, $116^{\circ}57'E$), within the central wheatbelt of Western Australia, 160 km south east of Perth (Figure 2.1). Dryandra State Forest was selected as an appropriate study site for two primary reasons. In order to understand reasons for the decline of the Western Yellow Robin, it is useful to understand its ecology in a relatively undisturbed and continuous habitat. Dryandra State Forest was chosen as an appropriate study area as it harbours these qualities, being one of the largest remaining fragments of native vegetation in the wheatbelt.

Due to its large size, Dryandra State Forest accommodates sufficient numbers of Western Yellow Robins for statistical analysis. The presence of various sized blocks also allows for analysis of edge effects and associated processes influencing site-occupancy for the habitat selection component of the study.

2.1.2 The wheatbelt

The wheatbelt of Western Australia covers an area of approximately $140,000\text{km}^2$, from the Murchison River in the north to Esperance in the southeast (Saunders *et al.*, 1985). It is characterised by a Mediterranean climate with hot dry summers and mild wet winters. Rainfall averages from 600mm in the west to 280mm in the east with mean maximum monthly temperatures ranging between 16°C in July and 34°C in January, with the north normally experiencing slightly elevated temperatures (Saunders *et al.*, 1993). Prior to European settlement, the vegetation of the wheatbelt consisted of woodlands of Salmon Gum (*Eucalyptus salmonophloia*), York Gum (*E. loxophleba*), Wandoo (*E. wandoo*), Gimlet (*E. salubris*) and other species on the red loam soils. The laterite, sandy and duplex soils were associated with scrub, heath and thickets (Saunders, 1989). As a result of patchy distributions of soils, there was an associated patchy distribution of vegetation associations, with woodlands scattered throughout (Saunders, 1989).

2.1.3 Dryandra State Forest

Dryandra State Forest is a Class A conservation reserve comprising of 17 blocks ranging in size from 87 ha to 12,283 ha, with a total area of 27,947ha (Figure 2.1). Dryandra State Forest is represented by three gazetted state forests; state forest 51 (Lol Gray), state forest 52 (Highbury) and state forest 53 (Montague) (Department of Conservation and Land Management, 1995). All further mention of Dryandra State Forest will refer to the 10 northernmost blocks, which I will denote as Dryandra Woodland (Figure 2.2).

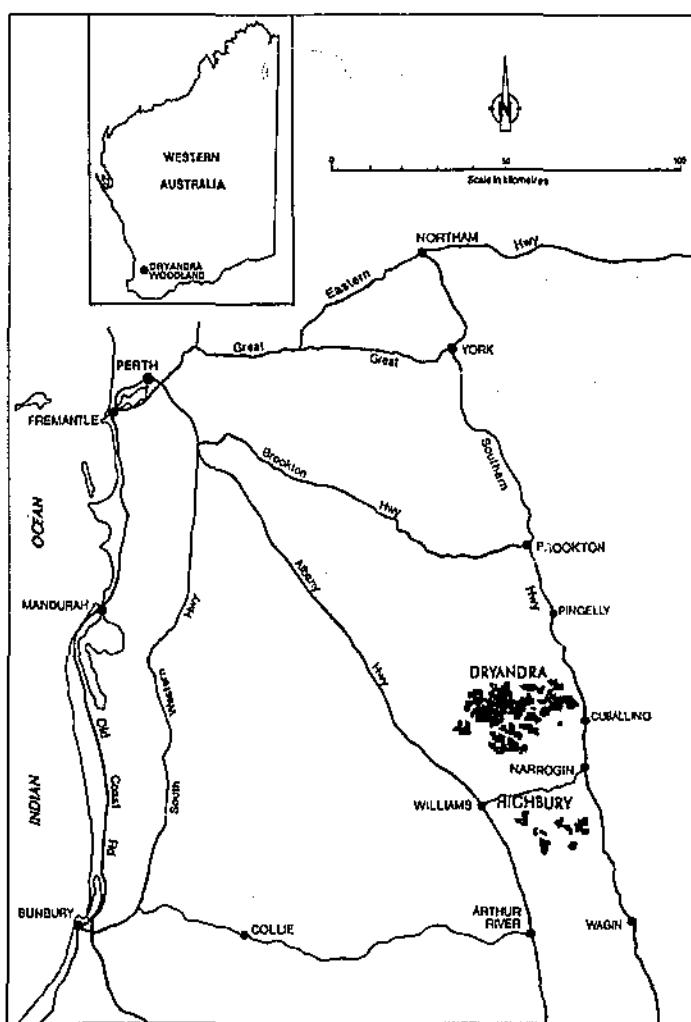


Figure 2.1: Location of Dryandra State Forest (Department of Conservation and Land Management, 1995).

Dryandra Woodland supports a rich assemblage of fauna as a result of its transitional position on the boundary between the Darling and Avon botanical provinces (Department of Conservation and Land Management, 1995). Dryandra Woodland is

important as a refuge for a large number of threatened marsupials including the Numbat (*Myrmecobius fasciatus*), Red-tailed Phascogale (*Phascogale calura*), Chuditch (*Dasyurus geoffroii*), Woylie (*Bettongia penicillata*) and Tammar Wallaby (*Macropus eugenii*) (Department of Conservation and Land Management, 1995). Of the 13 species of birds that are classified as endangered, vulnerable or near threatened within the western wheatbelt, seven are found within Dryandra Woodland including the Carnaby's Black-cockatoo (*Calyptorhynchus latirostris*), Malleefowl (*Leipoa ocellata*), Bush Stone-curlew (*Burhinus grallarius*), Western Rosella (*Platycercus icterotis*), White-browed Babbler (*Pomatostomus superciliosus*), Crested Shrike-tit (*Falcunculus frontatus*) and Crested Bellbird (*Oreoica gutturalis*) (Garnett and Crowley, 2000).

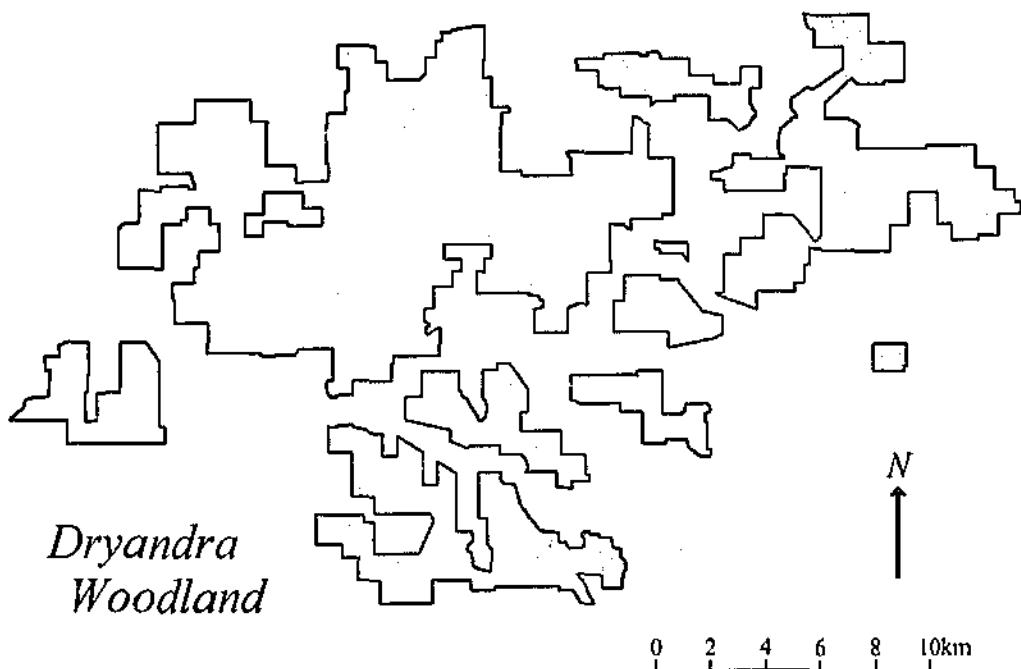


Figure 2.2 Map showing the ten blocks of Dryandra Woodland

2.1.4 History of Dryandra Woodland

Aboriginal heritage

Archaeological evidence suggests the occupation of the south-west of Western Australia by Noongar aborigines over the last 40,000 to 50,000 years (Department of Conservation and Land Management, 1995). Within Dryandra Woodland, the presence of archaeological sites including an ochre quarry, scarred tree, stone arrangements, and

artefact scatters provides evidence of occupation by the people of the Wiilman Clan of the Narrogin district (Department of Conservation and Land Management, 1995).

European occupation

European occupation of the Dryandra surrounds was first recorded during the 1860s with the issuing of pastoral leases to early settlers. To supplement the income of the developing agricultural industry, local farmers turned to the mallet stands of the region. During the early part of the 1900s, the bark of the Brown Mallet (*E. astringens*) was a major export commodity of Western Australia, supporting the large tannin industry of the time. Brown Mallet is primarily restricted to break-away slopes within Dryandra Woodland, although the majority was harvested to support the industry of the time. Apart from mallet bark stripping, the area was also utilised by sandalwood cutters and kangaroo and possum hunters (Department of Conservation and Land Management, 1995). By 1908, the Woods and Forests Department noted the danger of a shortage in mallet due to over-exploitation, and in 1924, the first portions of Dryandra Woodland were reserved by the Forest Department as Crown reserves for the purpose of protecting natural mallet stands and to extend the range of the species by establishing plantations (Department of Conservation and Land Management, 1995). Plantations were established in Dryandra Woodland during the 1920s, and by 1962, almost 30% (8,316 ha) of the area of Dryandra Woodland comprised of planted Mallet. Although a large portion of the area was set aside, areas surrounding Dryandra Woodland underwent major changes as a result of clearing of remnant vegetation surrounding the woodland (Figure 2.3).

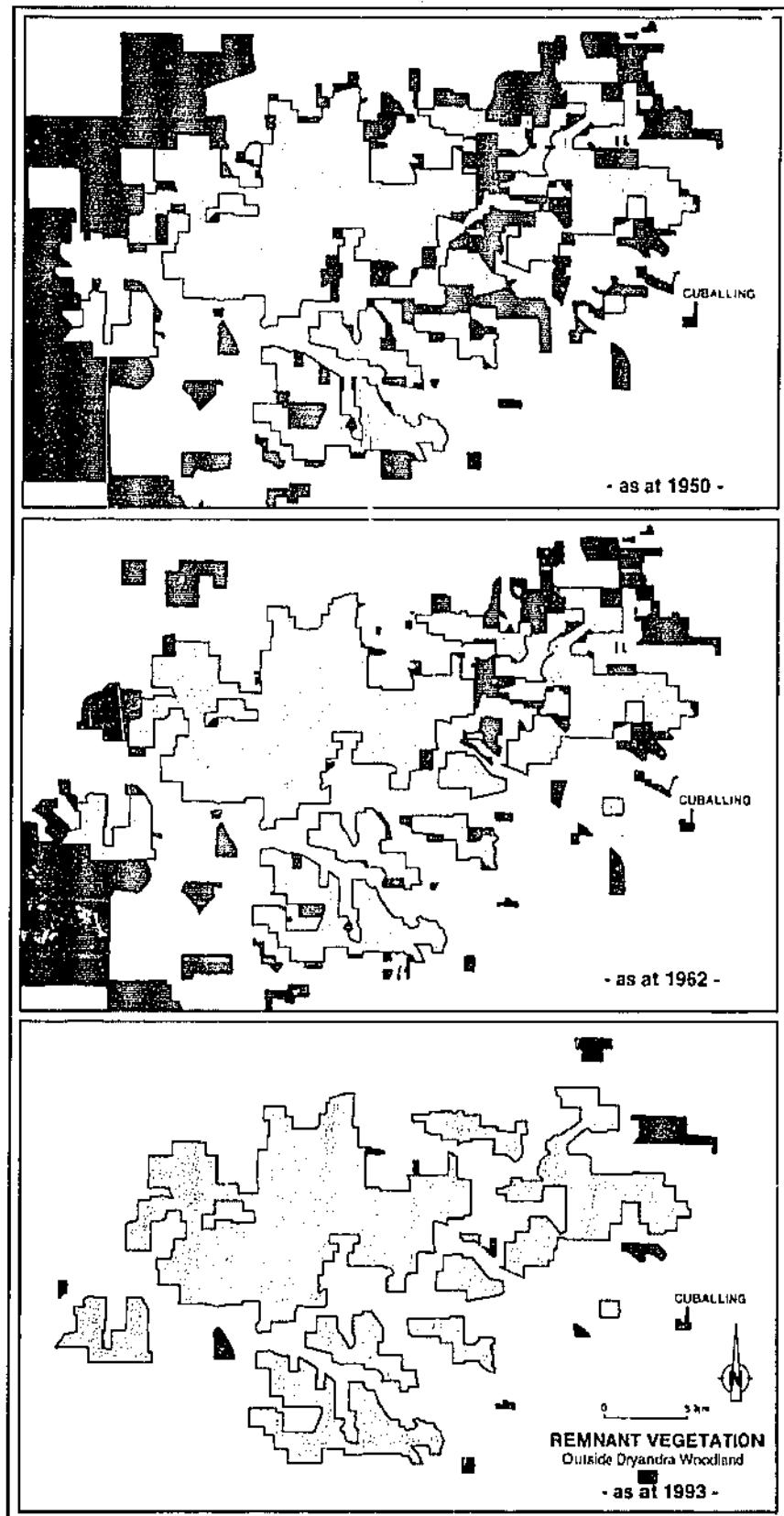


Figure 2.3 History of the changes in vegetation surrounding Dryandra Woodland between 1950 and 1993 (Department of Conservation and Land Management, 1995).

2.1.5 Location of seasonal study sites

Within Dryandra Woodland, four primary study sites were selected within the largest block. At each of these sites, seasonal changes in foraging behaviour, perch use and pounce-site characteristics were examined. These particular study sites (subsequently named Koomal, Tommingley, Norn and Ochre) were chosen, as they were the only sites in which Western Yellow Robins were consistently seen over three weeks of pilot surveys during Autumn 2000 (Figure 2.4).

At each of the study sites, a minimum convex polygon method was used to determine resident Western Yellow Robin territory boundaries during the non-breeding season. This was established by measuring the furthest distance a Western Yellow Robin was seen in at least nine measured directions from a fixed point in the vicinity of the theoretical territory centre (see section 7.2.4). By adjoining these points, an estimate of the territory boundary and size was calculated.

Koomal (32° 45' 45"S, 116° 55' 22"E)

Koomal was a Wandoo/Powderbark Wandoo open woodland (Plate 2.1). The site was located on lower slopes, with Powderbark Wandoo trees associated with gentle upper slopes on the north and eastern sides of the site. Koomal had the lowest proportion of Wandoo trees (80.4%), and highest proportion of Powderbark Wandoo trees (14.6%) of the study sites. Koomal also had the highest density of sub-canopy trees (97 trees/ha). Of the study sites, Koomal had the highest proportion of *Astroloma* shrubs.

Tommingley (32° 47' 53"S, 116° 53' 55"E)

Tommingley was a Wandoo open woodland, with 97.5% of trees represented by Wandoos (Plate 2.2). Tommingley is similar to Koomal in that the site is found on lower slopes, bounded to the north and east by lateritic breakaways. Bounding the south of the site is Tommingley Road, beyond which is an intermittent creek line. Tommingley has the highest density of sapling and canopy trees (66 and 101 trees/ha respectively). Tommingley is also characterised as harbouring the highest density of shrubs (4834 shrubs/ha), as well as the highest proportion of *Gastrolobium* shrubs.

Norn (32° 48' 23"S, 116° 53' 57"E)

Norn was a Wandoo open woodland, with 97.1% of trees represented by Wandoos (Plate 2.3). Norn was different from other study sites in being on a relatively flat relief, not bounded by lateritic breakaways or gentle upslopes. The north of the site was bounded by Powderbark Wandoo woodland and a large expanse of

mallee/shrubland. Norn had the highest proportion of *Acacia* shrubs and the highest abundance of herbs.

Ochre (32° 47' 26"S, 116° 56' 27"E)

Ochre was a Wandoo open woodland with 84.2% of trees represented by Wandoo, with Marri (*E. calophylla*) and Powderbark Wandoo contributing 7.5% and 4.2% respectively (Plate 2.4). The Ochre site was bounded to the south and east by an extensive lateritic breakaway, with the east breakaway covered in dense *Hakea* shrubland. The density of sapling, subcanopy and canopy trees, as well as the aggregation of saplings was lowest at Ochre.

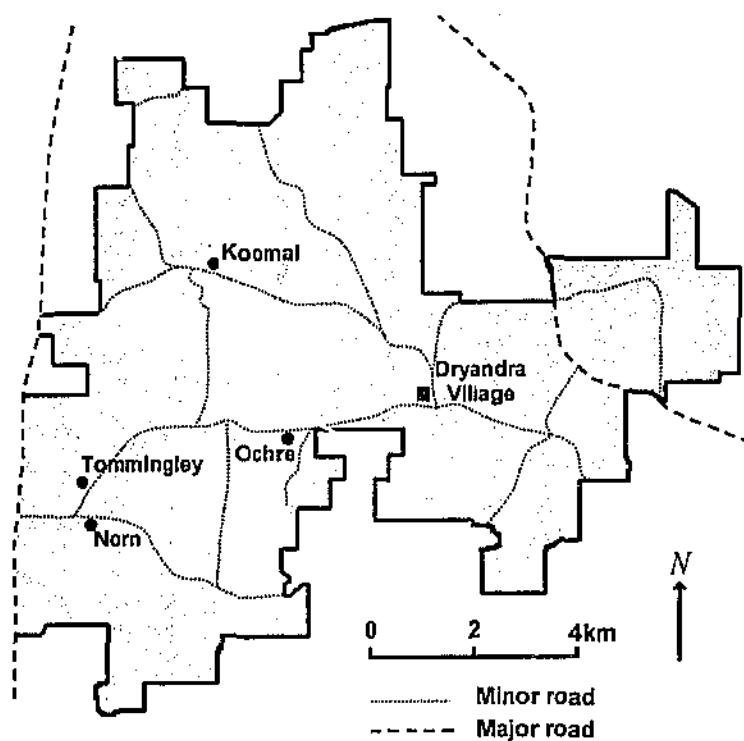


Figure 2.4 Location of the seasonal study sites within the main block of Dryandra Woodland



Plate 2.1 Site photograph of Koomal (yellow arrow represents north, Photo: J. Cousin)



Plate 2.2 Site photograph of Tommingley (yellow arrow represents north, Photo: J. Cousin)



Plate 2.3 Site photograph of Norn (yellow arrow represents north, Photo: J. Cousin)



Plate 2.4 Site photograph of Ochre (yellow arrow represents north, Photo: J. Cousin)

2.1.6 Climate and Vegetation of Dryandra Woodland

Climate

The climate of Dryandra Woodland is typical of the western wheatbelt with a long-term yearly average rainfall of 503mm. The long-term mean monthly maximum temperature range varies between 14.7°C in July to 30.9°C in January (annual mean of 22.3°C) with the long-term mean monthly minimum temperature range varying between 5.6°C in August and 14.7°C in February (annual mean of 9.8°C). Due to the absence of a weather recording station at Dryandra Woodland, all rainfall and temperature averages were obtained from the nearby town of Narrogin (32°56'03"S, 117°10'47"E), 25km to the southeast.

Rainfall

The total annual rainfall during the study period dropped from 453.0mm in 2000 to 383.2mm in 2001, representing 90% and 76% respectively of the 110-year average of 502.6mm. Most notable during the study period was the unseasonal pattern of rainfall (Figure 2.5), with very high rainfall recorded during January 2000 (133.4mm); the highest monthly rainfall for the study period, approaching the 110-year high January rainfall total of 155.0mm. Following this was a particularly dry autumn 2000, including a May 2000 rainfall total of 3.8mm (representing the lowest ever May total recorded in 110 years) as well as relatively late winter rain.

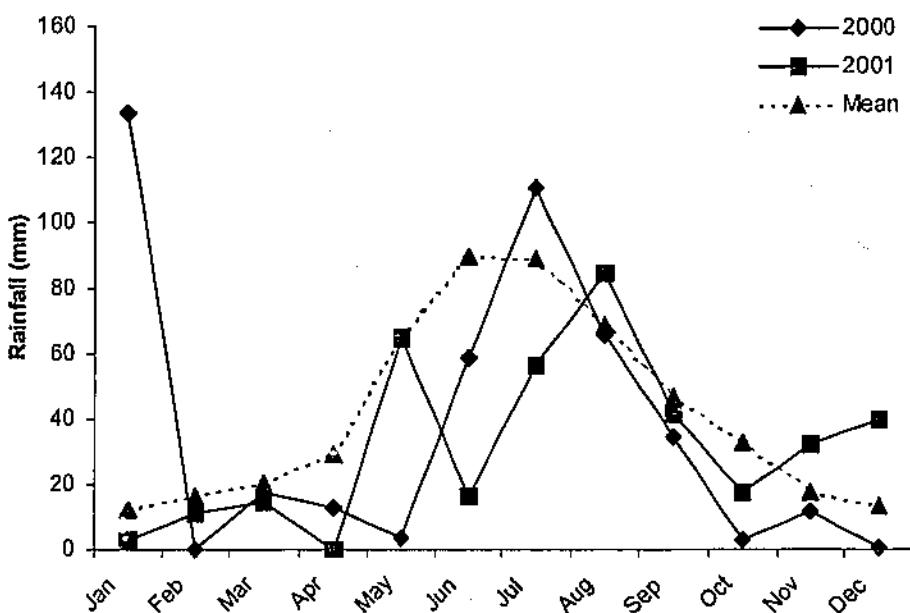


Figure 2.5 Total monthly rainfall for 2000 and 2001, and long-term mean monthly rainfall, all recorded from Narrogin (Bureau of Meteorology, acquired February 2002).

During 2001, there was a dry summer/early autumn, with 44.6mm falling in the seven months between October 2000 and April 2001, compared to the 143.3mm long-term average for the same time period. This was followed by a dry winter (including the driest ever June monthly rainfall total of 16.7mm compared to the long-term June monthly average of 89.8mm). Spring 2001 rainfall figures showed similar monthly rainfall to the long-term mean.

Temperature

The mean daily maximum temperatures during the study period ranged between 2.6°C higher and 3.4°C lower than the long-term mean daily maximum temperatures for April 2001 and December 2001 respectively (Figure 2.6). The mean daily minimum temperatures ranged between 1.6°C higher and 2.6°C lower than long-term mean daily minimum temperatures for January 2001 and July 2001 respectively. Averaged throughout the two-year study period, the mean daily maximum temperature averaged 0.5°C higher than the long-term average, with the mean daily minimum temperature averaging 0.5°C lower than the long-term average.

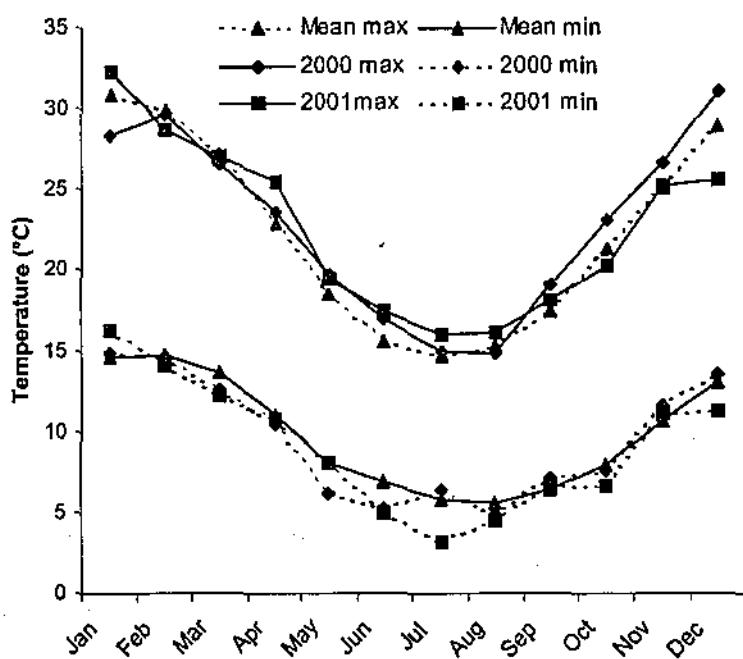


Figure 2.6 Mean monthly maximum and minimum temperatures for 2000 and 2001 compared to the long-term average, recorded from Narragin (Bureau of Meteorology, acquired February 2002).

Vegetation associations

Past weathering of the Yilgarn Block plateau on which Dryandra Woodland lies, has given rise to the development of three broad landform units: Norraine (lateritic uplands), Noombaling (valley slopes) and Biberkine (valley floors), from which vegetation associations are closely linked (Department of Conservation and Land Management, 1995). Table 1 shows the twelve recognised vegetation associations along with their landscape position, soil type and frequency of occurrence within the Dryandra Woodland system.

Dryandra Woodland lies on the boundary between the Darling and Avon botanical districts of the South-west Botanical Province. This transitional position between the moist Jarrah (*E. marginata*) and Marri forests of the Darling Range, and the Wandoo, Salmon Gum and York Gum woodlands of the interior has given rise to the high diversity of plant life within Dryandra Woodland. A total of 853 species of plant have been identified within Dryandra Woodland, including 70 introduced species (Coates, 1993). The vegetation is however dominated by Wandoo and Powderbark Wandoo woodland, accounting for 50% of the total area (Department of Conservation and Land Management, 1995), with Mallet representing 30% of the area.

2.2 GENERAL METHODS

2.2.1 Fieldwork

The foraging ecology of the Western Yellow Robin was studied between Winter 2000 and Spring 2001. During each of the six seasons, a minimum of four weeks was spent in the field. Aspects of nesting ecology were investigated throughout the 2000 breeding season (July – December) and the earlier part of the 2001 breeding season (July to November). Habitat characteristics of site occupancy were measured during the 2001 breeding season.

It was assumed that the same individuals were sampled throughout the study, on the basis that the territory boundaries remained fairly constant and Western Yellow Robins are resident and sedentary (Saunders, 1989; Pizzey and Knight, 1997; Anon., 1999b) much like the ecologically similar Eastern Yellow Robin (Anon., 1998; Anon., 1999a). However, birds were not banded, so turnover of individuals during the study may have occurred.

Detailed methods on data collection and statistical analysis for each component of the study are provided in the relevant chapters.

Table 2.1 Vegetation associations of Dryandra Woodland (modified from Coates, 1993)

Vegetation association	Landscape Position	Soil Type	Extent of presence
<i>Eucalyptus accedens</i> (powderbark wandoo) woodland	Steep to gentle upper slopes below the lateritic plateau and small gravelly rises in mid slope position	Sand or sandy loam and gravelly duplex soils	Extensive
<i>Eucalyptus wandoo</i> (whitegum, wandoo) woodland	mid to lower slopes, occasionally sandier upper slopes, low lying areas and drainage lines	Sand to sandy loam ± gravel over clay	Extensive
<i>Eucalyptus calophylla</i> (marri) woodland	Lower and mid slopes	Grey sandy soils	Small areas
<i>Acacia acuminata</i> (jam wattle) low forest	Lower slopes, low lying areas often associated with granite or drainage lines	Loam soils sometimes in association with granite	Small areas, mostly in mixed associations
<i>Eucalyptus astringens</i> (brown mallet) Woodland	Naturally occurring on steep slopes adjoining breakaways or escarpments	Clayey soils with laterite	Common as small areas
<i>Eucalyptus loxophleba</i> (York gum) Woodland	Lower slopes often in association with granite or drainage lines	Loam soils over clay	Very restricted
<i>Allocasuarina huegeliana</i> (rock sheoak) Low Forest	Slopes below the lateritic plateau	Sandy soils in association with granite outcrops	Relatively common
Lateritic plateau Woodlands <i>Eucalyptus accedens</i> , <i>Eucalyptus calophylla</i> , <i>Eucalyptus marginata</i> <i>Eucalyptus wandoo</i> species dominant or co-dominant over short distances	Lateritic plateau remnants usually bounded by escarpments, spurs to lower slopes	Duricrust, sand and sandy loam ± gravel in depressions, shallow gravelly soils over ironstone	Common in interior parts of plateau
<i>Eucalyptus wandoo / Allocasuarina huegeliana</i> (rock sheoak) Forest	Mid and lower slopes	Sandy soils	Common but in small areas
Short kwongan (diverse mixed shrubland < 2 metres)	Occasionally on the lateritic plateau usually on slopes below. Sometimes associated with granite rock	Shallow gravelly soils, deeper sands and gravels, gravelly duplex soils	Common
Dryandra and Petrophile Shrubland (Tall kwongan) > 2 metres when mature	Duricrust, usually forming a fringe around the tops of lateritic residuals	Shallow gravelly soils over ironstone	Small areas and merges with lateritic plateau woodlands
Lithic complex – granite	Slopes below the lateritic plateau	Rock surface and associated soils	Small outcrops relatively common

PART II: FORAGING ECOLOGY



The tree which moves some to tears of joy is in the eyes of others only a green thing that stands in the way. Some see nature all ridicule and deformity . . . and some scarce see nature at all. But to the eyes of the man of imagination, nature is imagination itself.

-William Blake (1757–1827)

*In the end, we will conserve only what we love.
We will love only what we understand.
We will understand only what we are taught.*

- Baba Dioum
Senegalese ecologist

CHAPTER 3: FORAGING BEHAVIOUR

3.1 INTRODUCTION

To understand the ecology of a species, one needs to determine as many aspects of its life history as possible. Within any forest or woodland ecosystem, birds exhibit spatial and temporal differences in the foraging resources that they exploit. For example, some utilise all strata of the habitat from the canopy to the ground (e.g., Golden Whistler *Pachycephala pectoralis*, see Recher and Davis, 1998), while others forage only within a single stratum (e.g., Diamond Dove *Geopelia cuneata*, see Recher and Davis, 1997). Understanding the spatial and temporal breadth of a species' foraging niche allows one to potentially understand why there is an effect of landuse modification on foraging resource requirements.

3.1.1 Spatial and temporal considerations

What is often overlooked in studies of foraging ecology is the seasonality of resource utilisation, with requirements varying not only among species, but also seasonally within species (Bunnell and Chan-McLeod, 1999). A few studies have examined temporal variation in avian foraging behaviour (e.g., Recher *et al.*, 1987a; Hejl and Verner, 1990; Lundquist and Manuwal, 1990; Miles, 1990; Szaro *et al.*, 1990; Recher and Majer, 1994; González, 1997; Luck *et al.*, 2001; Wilson and Recher, 2001), with many pooling data across seasons (see Holmes and Robinson, 1981; Schooley, 1994 for comments).

Seasonal variation in foraging resource use is driven by seasonal patterns of rainfall, temperature, humidity, wind velocity and plant phenology (Grubb, 1975; Grubb, 1977; Grubb, 1978; Szaro *et al.*, 1990; Kleintjes and Dahlsten, 1995). The underlying patterns to seasonal changes in foraging behaviour, in many cases, are related to seasonality in the availability of prey, with many studies demonstrating seasonal changes in invertebrate biomass (Cameron, 1985; Noske, 1985; Karr and Brawn, 1990; Recher *et al.*, 1996).

Seasonal variation in avian foraging ecology may have a significant effect on the management decisions we draw on estimates of habitat use, foraging resource utilisation and niche breadth (Miles, 1990).

3.1.2 Ground-foraging

While many species utilise the ground as a foraging substrate at least some time during the year (e.g., Beck and George, 2000), a large proportion of the avifauna of forest and woodland ecosystems utilise the ground as a major substrate for the acquisition of prey throughout all seasons (Recher *et al.*, 1985; Recher and Davis, 1997; Recher and Davis, 1998).

3.1.3 Aims

Of the foraging guilds within Australia, the ground foragers are among the most detrimentally threatened. At present, 58% of threatened bird species are present within the ground-foraging guild (Table 5.2 in Garnett and Crowley, 2000). The Western Yellow Robin (*Eopsaltria griseogularis*) is one species whose foraging ecology is dependent on the ground as a foraging substrate, due to its predominant ground-pouncing mode of foraging (Recher and Davis, 1998; Recher *et al.*, 2002). Although Recher and Davis (1998) provide information on the foraging ecology of the WYR, the data are limited, having been collected over nine days in Spring 1995.

In the present chapter, an investigation into interseasonal and intraseasonal changes in foraging behaviour (manoeuvres, substrates and heights), will contribute to the knowledge of the ecological resource requirements of the WYR. Furthermore, this information will provide valuable information towards an understanding of possible reasons why this guild of birds is threatened.

3.2 METHODS

3.2.1 Study sites

Foraging behaviour was recorded within the four study sites described in section 2.1.5. All data from each season at each site were collected within the boundaries of the observed territories (as described in section 2.1.5).

3.2.2 Foraging behaviour

Foraging behaviour was recorded between Winter 2000 and Spring 2001. Collection of data within a season occurred over a two to three week period to reduce the effect of unseasonal weather on foraging behaviour. To account for any diurnal changes in foraging behaviours, data collection was spread evenly throughout the day.

Data were not collected during rainy or windy weather, or when temperatures exceeded 35°C.

Following the detection of an individual, a minimum of 60 seconds was allowed to elapse to avoid biasing first observations towards conspicuous behaviours (Kleintjes and Dahlsten, 1995), and to reduce recording behaviours influenced by my presence. A minimum distance of 20 metres separated myself from foraging WYRs to reduce observer disturbance.

A minimum of 200 foraging behaviours was recorded per site per season from between two and five resident individuals, which included foraging manoeuvre, substrate and height[†]. For each individual encountered, between 20 and 40 sequential observations of foraging manoeuvre, foraging substrate and foraging height were recorded whether the attempts were successful at attaining prey or not (similar methodology to Adams and Morrison, 1993).

The recording and analysis of sequential foraging observations is recognised as resulting in an inaccurate representation of the proportional contribution of foraging manoeuvres and substrates (see Hejl *et al.*, 1990 for review). Furthermore, collection of sequential data, especially from the same individual, is often labelled as pseudoreplication due to the lack of independence of replicate data (Bell *et al.*, 1990). However, sequential sampling is often preferred as it increases the likelihood of sampling uncommon and inconspicuous foraging behaviours (Morrison, 1984; Recher *et al.*, 1985). Recher and Gebski (1990) comment how sequential observations are often preferred in cases where little is known of a species' behaviour, or when individuals are hard to locate; conditions applicable in the present study. Recher and Gebski (1990) also suggest that large sample sizes could potentially overcome problems associated with the lack of statistical independence of sequential data.

Nonetheless, during the present study, the non-independence of data (behavioural observations recorded sequentially, with sites resampled between seasons) was addressed by reducing the statistical significance level to 1% ($\alpha = 0.01$).

Definitions

Foraging manoeuvres were recorded as pounce, glean, probe, snatch, hawk or hover, with definitions following Recher *et al.* (1985)(Table 3.1).

[†] Foraging substrate data were collected from Spring 2000 to Spring 2001

Table 3.1 Foraging manoeuvre definitions (modified definitions of Recher *et al.*, 1985).

<i>Foraging Manoeuvre</i>	<i>Definition</i>
<i>Pounce</i>	The bird flies or drops down from a perch to take a prey organism from the ground or low vegetation. The bird lands, takes the prey, and then returns to the same or different perch.
<i>Glean</i>	A standing or hopping bird takes prey from nearby substrates.
<i>Probe</i>	In probing, a bird extracts prey from within or under a substrate such as soil, litter, crevices or soft wood. A probe was recorded after a pounce only if the bird travelled along the ground more than two metres.
<i>Snatch</i>	The bird flies or jumps up to take a prey from a nearby substrate. In most cases, the bird spots the prey when perched. The bird often returns back to the same perch. Most often, prey is snatched from surfaces a short distance above the bird.
<i>Hawk</i>	The bird flies from a perch to capture a flying insect after which it usually flies to a different perch.
<i>Hover</i>	A bird flies from a perch and hovers in the air for a brief period while picking a food item from a substrate, after which it returns to a perch.

Foraging substrates were recorded as ground, trunk bark, branch bark, foliage, air or flower according to specific criteria (Table 3.2).

Table 3.2 Foraging substrate definitions

<i>Foraging Substrate</i>	<i>Definition</i>
<i>Ground</i>	Foraging on leaf litter and bare ground
<i>Trunk (Bark)</i>	Foraging on the bark of a trunk of a standing live or dead tree, as well as log
<i>Branch (Bark)</i>	Foraging on the bark of any branch on any standing live or dead tree, as well as branches of logs
<i>Foliage</i>	Foraging on any live green leaf material, including the canopy of trees, leaves of shrubs, herbs and grasses
<i>Air</i>	Foraging on any flying insect in the air column
<i>Flower</i>	Foraging off any plant inflorescence, including in the canopy of trees and the flowers of shrubs, herbs and grasses

Analysis

Foraging manoeuvres and substrates

Foraging manoeuvre and foraging substrate data were analysed to determine the proportional contribution of the most common foraging behaviours, with seasonal and site differences tested by chi-square analyses. Due to the high proportion of pounce manoeuvres, other ‘active’ manoeuvres were pooled into an “other” category prior to analysis. ‘Trunk bark’ and ‘branch bark’ was combined as “bark” prior to analysis of foraging substrates. There were only three records of flowers as a foraging substrate, so these were not included in analyses.

Multiple comparison tests were performed between each season pair for foraging manoeuvres using chi-square with a Bonferroni corrected *p*-value of 0.0007 (0.01/15).

Foraging heights

Non-parametric two-factor analysis of variance by tie ranks was used to test for season and site differences in foraging heights. This test was undertaken as the distributions of foraging heights could not be transformed to normality, and sample variances were heterogenous. This form of non-parametric analysis is an extension of the Kruskal-Wallis test, using tied ranks, which calculates a Kruskal-Wallis (*H*) statistic, closely approximated by X^2 (Zar, 1984).

Non-parametric Tukey-type multiple comparison tests were performed to determine between which seasons any significant differences occurred (Zar, 1984 pg 199).

3.3 RESULTS

3.3.1 Foraging manoeuvres

A total of 5,677 records of foraging manoeuvres by WYRs was collected from four study sites over six seasons (Winter 2000 to Spring 2001). WYRs foraged predominantly by pouncing (81 – 87% of foraging manoeuvres per site; Table 3.3). Prey were also obtained by gleaning (4 – 6%), snatching (3 – 5%) and hovering (3 – 4%), with hawk and probe manoeuvres infrequently recorded (each representing ≤ 3% of total foraging manoeuvres per study site).

Table 3.3 Percentage contribution of foraging manoeuvres of the Western Yellow Robin at each of the four study sites from six seasons. Sample size given in parentheses.

	Season						
	Winter 2000	Spring 2000	Summer 2000/01	Autumn 2001	Winter 2001	Spring 2001	Combined
Koomal	(232)	(203)	(218)	(226)	(309)	(219)	(1,407)
Pounce	87	77	75	85	87	74	81
Glean	7	7	8	2	6	5	6
Probe	4	2	0	1	0	1	1
Snatch	0	4	7	5	2	10	5
Hawk	0	3	3	3	1	8	3
Hover	1	6	7	4	3	2	4
Tommingley	(275)	(217)	(218)	(215)	(261)	(216)	(1,402)
Pounce	89	87	80	86	89	70	84
Glean	7	4	6	3	4	5	5
Probe	1	0	0	0	0	1	1
Snatch	1	2	2	7	5	14	5
Hawk	0	1	9	1	1	5	3
Hover	1	5	3	3	1	5	3
Norn	(237)	(236)	(213)	(251)	(246)	(206)	(1,389)
Pounce	89	82	79	82	94	79	84
Glean	7	2	6	6	2	8	5
Probe	3	0	0	1	0	0	1
Snatch	0	4	6	7	1	5	4
Hawk	0	3	4	2	1	4	2
Hover	0	9	6	3	2	4	4
Ochre	(213)	(228)	(259)	(303)	(249)	(227)	(1,479)
Pounce	85	86	81	94	91	81	87
Glean	9	3	6	1	2	4	4
Probe	1	0	1	0	0	0	0
Snatch	1	3	4	2	3	4	3
Hawk	0	4	1	0	4	2	2
Hover	3	4	7	2	0	9	4

With sites combined, there were significant differences in foraging manoeuvres between seasons ($\chi^2 = 107.8$, df = 5, $p < 0.0001$). This seasonality in foraging manoeuvres was present within sites at Koomal ($\chi^2 = 30.7$, df = 5, $p < 0.0001$), Tommingley ($\chi^2 = 46.2$, df = 5, $p < 0.0001$), Norn ($\chi^2 = 33.8$, df = 5, $p < 0.0001$) and Ochre ($\chi^2 = 32.5$, df = 5, $p < 0.0001$). With seasons combined, there were significant differences in foraging manoeuvres observed between sites ($\chi^2 = 16.1$, df = 3, $p < 0.005$), owing to a high proportion of “other” foraging manoeuvres at Koomal and low proportion of “other” foraging manoeuvres at Ochre (Table 3.3).

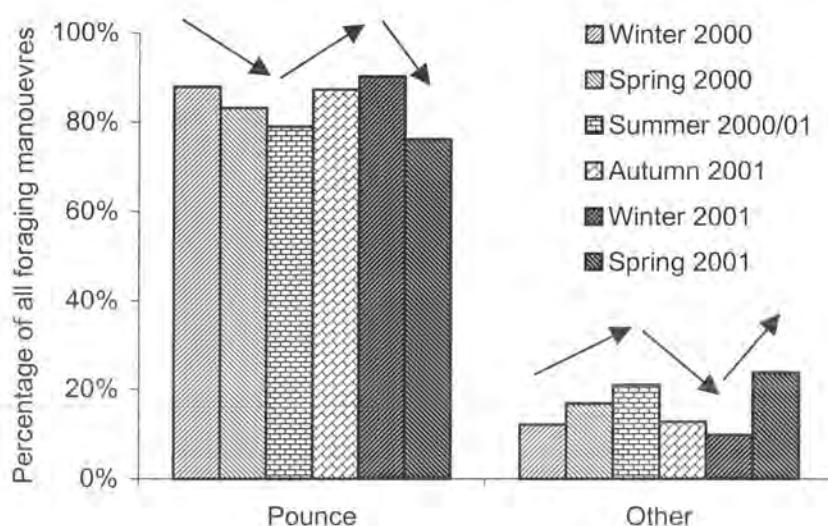


Figure 3.1 Seasonal changes in percentage of pounce and other foraging behaviours pooled from four study sites.

An examination of the seasonal comparisons in foraging manoeuvre percentages indicates a seasonal cycle of proportion of pounce and other foraging manoeuvres (Figure 3.1). The seasonal pattern is exemplified by a significant decline in the proportion of pounces from Winter 2000 to Summer 2000/01 ($\chi^2 = 26.9$, $df = 1$, $p < 0.0001$, Table 3.4), followed by a significant increase in proportion of pounces from Summer 2000/01 to Winter 2001 ($\chi^2 = 49.0$, $df = 1$, $p < 0.0001$). This is followed by a decline in proportion of pounces in Spring 2001 ($\chi^2 = 70.3$, $df = 1$, $p < 0.0001$).

Table 3.4 Pairwise multiple comparisons of foraging manoeuvre proportions (sites combined) between seasons (* = significant difference at corrected p -value of 0.0007).

Comparison	Chi-square	df	p-value
Winter 2000 v Spring 2000	8.4	1	0.0038
Winter 2000 v Summer 2000/01	26.9	1	< 0.0001*
Winter 2000 v Autumn 2001	0.2	1	0.6547
Winter 2000 v Winter 2001	2.9	1	0.0886
Winter 2000 v Spring 2001	43.0	1	< 0.0001*
Spring 2000 v Summer 2000/01	5.1	1	0.0239
Spring 2000 v Autumn 2001	6.3	1	0.0121
Spring 2000 v Winter 2001	21.5	1	< 0.0001*
Spring 2000 v Spring 2001	13.2	1	0.0003*
Summer 2000/01 v Autumn 2001	23.3	1	< 0.0001*
Summer 2000/01 v Winter 2001	49.0	1	< 0.0001*
Summer 2000/01 v Spring 2001	2.0	1	0.1573
Autumn 2001 v Winter 2001	4.6	1	0.0320
Autumn 2001 v Spring 2001	38.7	1	< 0.0001*
Winter 2001 v Spring 2001	70.3	1	< 0.0001*

There was no difference in foraging manoeuvres between Winter 2000 and Winter 2001 ($\chi^2 = 2.9$, $df = 1$, $p > 0.05$). However, there was a significant difference

between Spring 2000 and Spring 2001 ($X^2 = 13.2$, df = 1, $p < 0.0005$) owing to a reduced proportion of pounces in Spring 2001.

3.3.2 Foraging substrate

A total of 4,699 records of foraging substrates utilised by WYRs was collected from four study sites over five seasons (Spring 2000 to Spring 2001). WYRs foraged predominantly on the ground (77 – 86% of foraging substrates per site; Table 3.5). Prey was also taken off the bark of tree trunks (7 – 14%), with foraging from foliage, air and bark of branches infrequently recorded (each representing $\leq 3\%$ of total foraging substrates per study site).

Table 3.5 Percentage contribution of foraging substrate of the Western Yellow Robin at each of the four study sites from five seasons. Sample size given in parentheses.

	Spring 2000	Summer 2000/01	Season Autumn 2001	Winter 2001	Spring 2001	Combined
Koomal	(193)	(218)	(226)	(309)	(219)	(1,165)
Ground	80	68	81	84	71	77
Trunk	12	23	12	8	14	14
Branch	3	3	1	4	5	3
Foliage	2	3	4	3	1	3
Air	4	3	3	1	8	3
Tommingley	(213)	(218)	(215)	(261)	(216)	(1,123)
Ground	85	75	84	89	70	81
Trunk	11	9	13	6	18	11
Branch	0	4	0	2	2	2
Foliage	3	3	2	2	4	3
Air	1	9	1	1	5	3
Norn	(233)	(213)	(251)	(246)	(206)	(1,149)
Ground	83	75	74	88	74	79
Trunk	12	15	21	8	12	13
Branch	2	3	2	2	5	3
Foliage	1	3	2	1	5	2
Air	2	4	2	1	4	2
Ochre	(226)	(258)	(303)	(248)	(227)	(1,262)
Ground	85	79	93	92	83	86
Trunk	7	14	4	5	8	7
Branch	1	3	0	0	2	1
Foliage	4	4	3	0	5	3
Air	3	1	0	4	2	2

With sites combined, there were significant differences in foraging substrates between seasons ($X^2 = 97.9$, df = 12, $p < 0.0001$). This seasonality in foraging substrates was present within sites at Koomal ($X^2 = 47.0$, df = 12, $p < 0.0001$), Tommingley ($X^2 = 54.7$, df = 12, $p < 0.0001$), Norn ($X^2 = 36.8$, df = 12, $p < 0.0005$) and Ochre ($X^2 = 58.4$, df = 12, $p < 0.0001$). With seasons combined, there were significant differences in

foraging substrates utilised between sites ($\chi^2 = 55.4$, df = 9, $p < 0.0001$), owing to a low proportion of bark foraging and a high proportion of ground foraging at Ochre (Table 3.5).

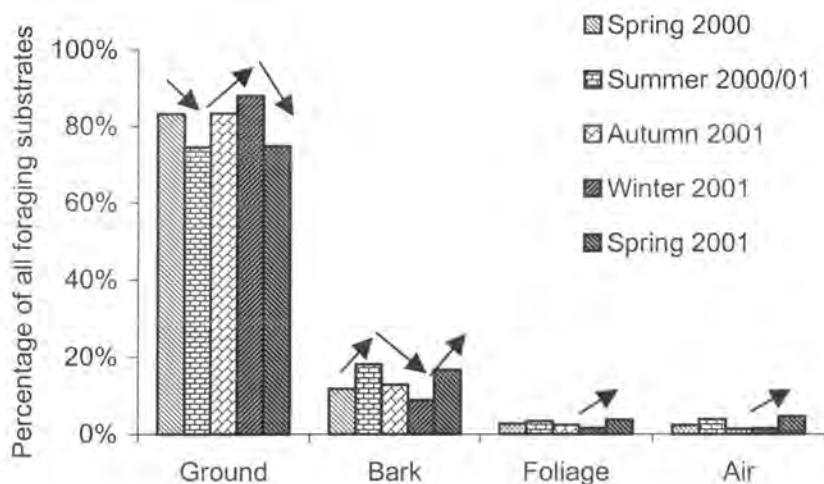


Figure 3.2 Seasonal changes in percentage of foraging substrates pooled from four study sites.

Due to the close correlation of foraging manoeuvre and substrate, no analysis of pairwise seasonal comparisons was undertaken due to the similar seasonal patterns (Figure 3.2). The only comparison made was the within-seasonal comparison of foraging substrate utilisation between Spring 2000 and Spring 2001, which showed a significant difference ($\chi^2 = 19.5$, df = 3, $p < 0.0005$), owing to a reduced proportion of ground as a foraging substrate in Spring 2001. This was undertaken for the purpose of discussion.

3.3.3 Foraging height

A total of 4,260 foraging height records was collected between Spring 2000 and Spring 2001, with the majority (76.9%) at 0m (Figure 3.3). Foraging heights ranged between 0cm and 1,050cm (mean \pm S.E = 41.9 ± 1.9 cm).

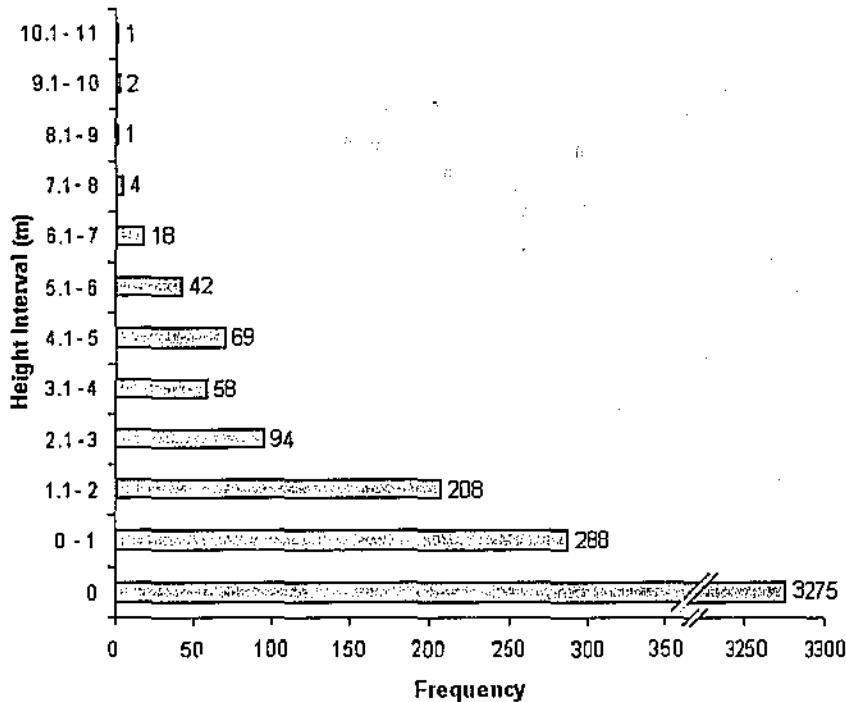


Figure 3.3 Frequency distribution of records of WYR foraging heights. Note the split X-axis and the presence of data values adjacent the relevant bars.

Analysis indicated a significant difference in the foraging height between seasons (Table 3.6).

Table 3.6 Non-parametric two-factor analysis of variance of foraging height differences between four sites and five seasons.

Source of Variation	SS	df	H	p-value
Cells	94190841.3	19		
Season	56807634.0	4	41.35	< 0.0001
Site	20179853.5	3	14.69	0.0021
Season x Site	17203353.8	12	12.52	0.4049

A difference existed between the average foraging heights between sites, attributed to site habitat differences. There was no significant season x site interaction effect, indicating the seasonal differences in foraging height were similar at each of the four study sites (Table 3.6).

An examination of the seasonal differences in foraging height (Figure 3.4), indicate a seasonal cycle, exemplified by a decrease from $55.1 \pm 4.4\text{cm}$ in Summer 2000 to $16.6 \pm 2.3\text{cm}$ in Winter 2001 ($q = 60.2$, $df = 5$, $p < 0.01$). Following this, there is an increase to $67.7 \pm 5.8\text{cm}$ in Spring 2001 ($q = 58.0$, $df = 5$, $p < 0.01$).

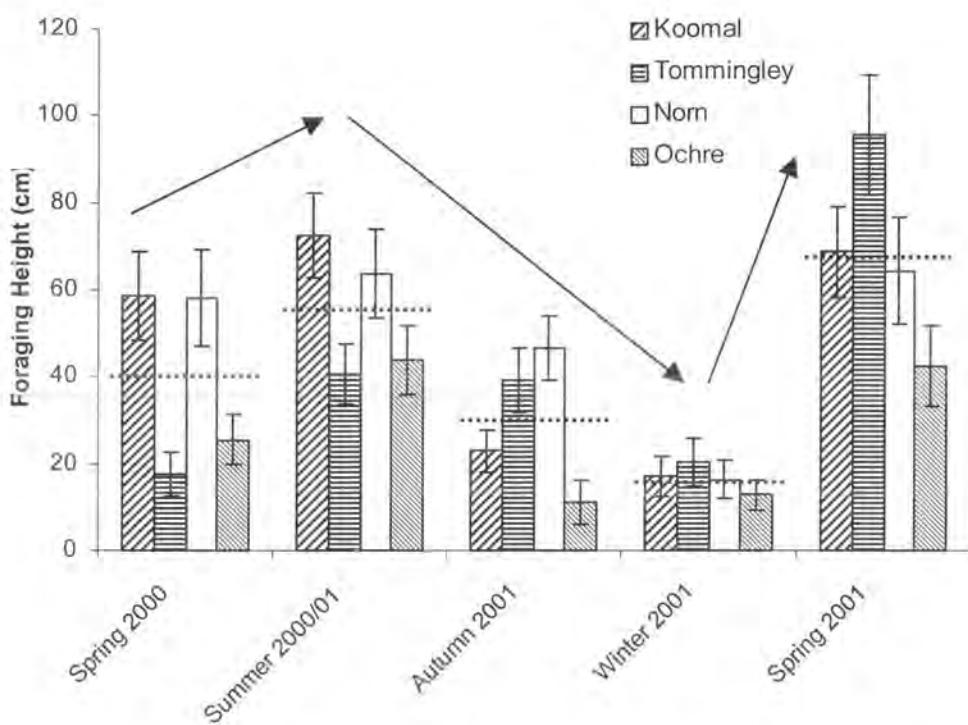


Figure 3.4 Average foraging height (\pm standard error) for each site between Spring 2000 and Spring 2001. The average foraging height within each season (sites pooled) is represented by the dashed line.

Results of pairwise multiple comparison tests indicate a significant difference in foraging height between season pairs except Spring 2000/Autumn 2001 and Summer 2000(01)/Spring 2001 (Table 3.7). The average foraging height during Spring 2001 was significantly higher than during Spring 2000 ($q = 32.6$, $df = 5$, $p < 0.01$) owing to a decrease in proportion of pounce foraging manoeuvres and ground foraging substrates from Spring 2000 to Spring 2001 (see section 3.3.1).

Table 3.7 Non-parametric Tukey-type multiple comparisons test of seasonal differences in foraging heights using pairwise differences of rank sums of the Kruskal-Wallis test. (* = significant difference p -value of 0.01)

Comparison	Difference	SE	q	df	p-value
Spring 2000 v Summer 2000/01	145399.0	4176.7	34.81	5	< 0.0001*
Spring 2000 v Autumn 2001	16660.5	4176.7	3.99	5	0.0398
Spring 2000 v Winter 2001	106022.5	4176.7	25.38	5	< 0.0001*
Spring 2000 v Spring 2001	136051.5	4176.7	32.57	5	< 0.0001*
Summer 2000/01 v Autumn 2001	162059.5	4176.7	38.80	5	< 0.0001*
Summer 2000/01 v Winter 2001	251421.5	4176.7	60.20	5	< 0.0001*
Summer 2000/01 v Spring 2001	9347.5	4176.7	2.24	5	0.5073
Autumn 2001 v Winter 2001	89362.0	4176.7	21.40	5	< 0.0001*
Autumn 2001 v Spring 2001	152712.0	4176.7	36.56	5	< 0.0001*
Winter 2001 v Spring 2001	242074.0	4176.7	57.96	5	< 0.0001*

3.4 DISCUSSION

3.4.1 Observed patterns of foraging behaviour

Pouncing was the most common foraging manoeuvre recorded in all sites and seasons, representing 84.3% of foraging manoeuvres. This result is less than other studies of the foraging behaviour of WYRs. At Dryandra Woodland, across a range of habitats (including sites surveyed during the present study), Recher and Davis (1998) and Recher *et al.* (2002) recorded pouncing as representing 97% and 94% of foraging manoeuvres respectively. However, their studies were highly seasonal, with data collected during winter and spring. Recher and Holmes (1985) and Ford *et al.* (1986) recorded pouncing as representing 75% and 67% of foraging manoeuvres respectively, in the ecologically similar Eastern Yellow Robin (*E. australis*) in eucalypt forest of New South Wales. However, these studies were also seasonal, having been undertaken during late spring and summer.

In the present study, the ground represented 81.1% of foraging substrates; less than Recher and Davis (1998) and Recher *et al.* (2002) whom recorded ground as representing 96% and 94% of the foraging substrate of WYRs respectively. Recher and Holmes (1985) and Ford *et al.* (1986) recorded ground as representing 77% and 73% of the foraging substrate of Eastern Yellow Robins respectively. Although foraging manoeuvres, foraging substrates and heights of prey acquisition attempts were recorded separately, all are intimately tied in that a “pounce” manoeuvre was usually onto the “ground” at a height of “0.0cm”, and a “hawk” manoeuvre was always performed in the “air” at a given height higher than the ground

3.4.2 Seasonality of foraging behaviour

Western Yellow Robins showed seasonal changes in foraging manoeuvres, foraging substrates and foraging heights. During Autumn and Winter, foraging behaviour was characterised by a high proportion of pounces to the ground. During Spring and Summer, pouncing was still the main foraging manoeuvre, although there was a significant increase in snatch, hawk and hover foraging manoeuvres. This was associated with increases in the proportion of other foraging substrates (trunk, branch, air, foliage), and average foraging heights.

Scasonality of foraging behaviour is a common pattern in birds from forest (Recher *et al.*, 1987a; Recher, 1989; Martin and Karr, 1990; Recher and Majer, 1994;

Kleintjes and Dahlsten, 1995; Franklin, 1999; Beck and George, 2000) and woodland (Hejl and Verner, 1990; Miles, 1990; Keane and Morrison, 1999; Luck *et al.*, 2001; Wilson and Recher, 2001) ecosystems. Often, a shifted pattern of resource selection is attributed to opportunistic behaviours (Franklin, 1999) although in the majority of cases, the shift in foraging behaviour is a seasonal and yearly pattern of occurrence.

The seasonal shift in foraging behaviour in the present study is most likely a result of seasonal shifts in the abundance of leaf litter, arboreal and aerial prey. The abundance of leaf litter invertebrates is often highest during the wetter periods of the year (Moeed and Meads, 1986; Majer and Abbott, 1989; Cale, 1994a; Fox *et al.*, 1997; Burgess *et al.*, 1999), although this pattern varies spatially and temporally (e.g., the effect of regional and yearly differences in temperature, humidity and rainfall, see Majer and Koch, 1982; Majer *et al.*, 1990; Recher *et al.*, 1996). As the leaf litter layer heats and desiccates during the warmer months, any remaining invertebrates may vertically migrate down into moister regions of the organic layer (Detsis, 2000), thus reducing available leaf litter prey to WYRs. Concurrently, a reduction in the abundance of leaf litter invertebrates during warmer months is accompanied by an increase in the abundance of arboreal and flying prey, as a result of their higher activity in warmer months (Recher *et al.*, 1983).

Intraseasonal foraging behaviour was similar between Winter 2000 and Winter 2001. However, differences in temperature and rainfall may have driven the intraseasonal shifts in foraging behaviour of WYRs between Spring 2000 and Spring 2001. A combination of less yearly rainfall during 2001, late winter 2001 rainfall and warmer temperatures during the Spring 2001 sampling period (see Figure 2.5 and 2.6) may have given rise to an influx in numbers of flying invertebrates, resulting in a higher average foraging height, brought on by a higher proportion of snatch, hawk and hover manoeuvres from bark, foliage and air substrates. Similarly, the reduced rainfall during much of 2001 may have brought about a ‘drought effect’, giving rise to a decline in available leaf litter invertebrates compared to 2000. Patterns of yearly variation in foraging behaviour have previously been noted, usually attributed to seasonal differences in rainfall and food availability (for example, see Szaro *et al.*, 1990).

No one factor can adequately explain the seasonal shift in foraging behaviour demonstrated in the present study of the WYR (Figure 3.5). Seasonal effects on prey abundance probably represent the principal influence (albeit indirectly), giving rise to the observed pattern of seasonal changes in foraging behaviour. The effect of season on

its own could also directly influence the foraging behaviour of WYRs. Grubb (1975) found birds reduced their foraging heights with decreasing temperature and increasing air/wind movement. Energetic demands could also influence seasonal changes in foraging behaviour of the WYR. As a result of breeding commitments and reduced hours per day available for foraging during winter, the Eastern Yellow Robin increases its foraging efficiency by reducing its time investment in energetically expensive ‘active’ foraging behaviours (Haylock and Lill, 1988). Dietary demands associated with preferential foraging of high calcium content invertebrates during the breeding season, have also been suggested to influence seasonal foraging behaviour (Davies, 1977; Beck and George, 2000).

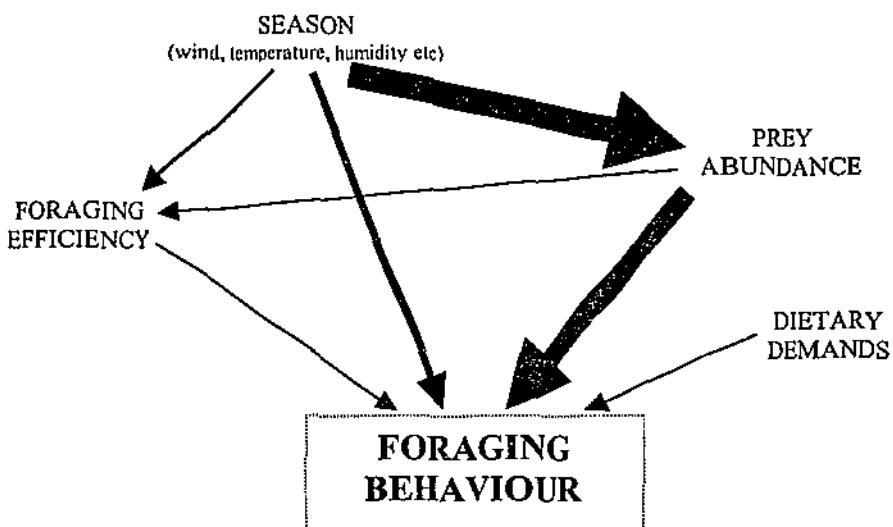


Figure 3.5 Schematic representation of the inter-relationship of factors possibly giving rise to the observed pattern of seasonal shifts in foraging behaviour. The perceived relative importance of factors in the present study is graded from most important (boldest black lines) to least important (least bold line).

CHAPTER 4: PERCH-USE SELECTION

4.1 INTRODUCTION

Broadening the knowledge of the foraging ecology of the Western Yellow Robin (*Eopsaltria griseogularis*) relies on understanding perch-use in addition to foraging behaviour. The Western Yellow Robin is a ground-pouncing bird, scanning the ground for potential prey from a perch, before pouncing onto small invertebrate and vertebrate prey (Recher *et al.*, 2002). As a result, the availability of perches plays an integral role in the efficient foraging of ground-pouncing birds.

Perching substrates are also an important component of the foraging ecology of birds in other guilds throughout all habitats. Throughout the world, there are many examples of the close association between bird and perch. Studies have noted the importance of perches in the conservation of Bald Eagles (*Haliaeetus leucocephalus*) (Caton *et al.*, 1992) and Madagascar Fish-eagles (*Haliaeetus vociferoides*) (Berkelman *et al.*, 1999), with both species selecting lakes based on the density and location of hunting perches. In Texas USA, the density of perches reflects territory quality in the Loggerhead Shrike (*Lanius ludovicianus*) (Yosef and Grubb, 1992; Chavez-Ramirez *et al.*, 1994), while the density of perches surrounding Song Sparrow (*Melospiza melodia*) nests in New York USA, increases their risk of detection and eventual parasitism by Brown-headed Cowbirds (*Molothrus ater*) (Hauber and Russo, 2000).

Apart from exhibiting important roles in individual species ecology, perches exhibit ecosystem conservation roles, having been implicated as a potential management tool for accelerating ecological succession, especially for reforesting disturbed land, through bird-mediated seed deposition (McClanahan and Wolfe, 1993).

4.1.1 Spatial and temporal shifts in perch-use selection

Perch use is intimately tied to foraging behaviour, with specific foraging manoeuvres often associated with characteristic perch types. Utilisation of various perches for foraging does not remain static throughout the year, varying spatially and temporally.

Spatial variation in the utilisation of perches is influenced primarily by vegetation structure. Birds foraging in a heterogeneous environment are hypothesized to select foraging locations in such a way to maximize energy intake (Davies, 1977). Recher *et al.* (2002) comment how the area around a perch in which prey is located is a

function of the height of the perch, the openness of the immediate habitat and the behaviour of the bird. As such, spatial selection of perches from which foraging manoeuvres are undertaken is influenced by the detectability of available prey. Following mowing of understorey weedy vegetation in Florida USA, Loggerhead Shrikes altered their foraging behaviour and associated perch use, with a higher abundance of low perches used in mowed territories, where ground invertebrates were more visible (Yosef and Grubb, 1993).

Utilisation of perches also varies temporally, with seasonal changes primarily influenced by the seasonal availability and distribution of invertebrate prey (Greig-Smith, 1983). As such, birds may alter the habitat in which they forage seasonally, selecting for territories or areas within home ranges with specific perch characteristics.

4.1.2 Aims

In the present chapter, an investigation into interseasonal and intraseasonal changes in perch use will contribute to the knowledge of the foraging ecology of the WYR. Furthermore, an investigation into tree species and size selection, as well as an analysis of perching heights, will provide valuable information towards perch use understanding.

As ground-foraging birds are amongst the most threatened in Australia (Garnett and Crowley, 2000), an understanding of perch requirements, an integral component of the ground-foraging guild, is essential to understanding factors giving rise to the decline of these birds.

4.2 METHODS

4.2.1 Study sites

Data on perch-use selection of WYRs were collected from the four study sites described in section 2.1.5. All data from each season at each site were collected within the boundaries of the observed territories (as described in section 2.1.5).

4.2.2 Perch-use selection

Perch-use selection data were collected between Winter 2000 and Spring 2001 at the same time as the foraging behaviour data. As such, all the conditions governing the collection of perch-use data are the same as those for the foraging behaviour component of the study (see section 3.2.2).

A minimum of 400 perch-uses was recorded per site per season from between two and five resident individuals at the same time as foraging behaviour data collection (section 3.2.2). For each individual encountered, between 50 and 80 sequential observations of perch-use were recorded whether a foraging attempt was made or not. However, sequential perches were only recorded if they were more than one metre apart.

The complications associated with sequential observations are the same as those discussed in section 3.2.2, namely, the non-independence of sequential data (Bell *et al.*, 1990). To account for this, the statistical significance level was reduced to 1% ($\alpha = 0.01$). The higher number of sequential and total observations of perch-use compared to the foraging behaviour data, (see section 3.2.2) reflects the use of a multiple number of perches between foraging manoeuvres.

Definitions

Following analysis of preliminary data collected in Autumn 2000, eight perch-use categories were employed (Table 4.1). Wandoos (*Eucalyptus wandoo*) trees were the most abundant trees in each of the study sites (see section 2.1.5). Due to the high utilisation of the abundant Wandoos, the use of perches on live Wandoos by WYRs was divided into trunk, live branch and dead branch (LWT, LWLB and LWDB respectively). Preliminary data also indicated the importance of the trunk and dead branches of dead (standing) trees (DTT and DTDB respectively). The remaining utilised perches were divided into dead fallen timber (DFT), live shrubs (LS) and dead shrubs (DS), with all other perches from other species of trees included in an “Other” category. The definitions of each of the categories are presented in table 4.1.

Analysis

Perch-use selection data were analysed to determine the proportional contribution of the most utilised perches, with seasonal and site differences calculated by chi-square analyses.

Multiple comparison tests were performed between each season pair for perch-use selection using chi-square with a Bonferroni corrected p-value of 0.0007 (0.01/15).

Table 4.1 Perch-use category definitions and abbreviations used in the text.

<i>Perch-use Category (Abbreviation)</i>	<i>Definition</i>
<i>Live Wandoos</i>	<i>Trunk (LWT)</i> The bird perches on the main trunk of a live sapling, subcanopy or canopy Wandoo (<i>Eucalyptus wandoo</i>).
	<i>Live Branch (LWLB)</i> The bird perches on any section of a live branch of a live sapling, subcanopy or canopy Wandoo.
	<i>Dead Branch (LWDB)</i> The bird perches on any section of a dead branch of a live sapling, subcanopy or canopy Wandoo, including dead branch material suspended from a live tree.
<i>Dead Standing Tree</i>	<i>Trunk (DTT)</i> The bird perches on the main trunk of a dead standing sapling, subcanopy or canopy tree of any species.*
	<i>Dead Branch (DTDB)</i> The bird perches on any section of a dead branch of a dead standing sapling, subcanopy or canopy tree of any species, including any suspended dead branch material.
<i>Dead Fallen Timber (DFT)</i>	The bird perches on any part of any fallen timber* having a diameter of at least 50mm and a length at least 800mm.
<i>Live Shrub (LS)</i>	The bird perches on the trunk, live branch, dead branch or foliage of any live shrub at least 300mm in height.
<i>Dead Shrub (DS)</i>	The bird perches on the trunk, dead branch or dead foliage of any dead shrub at least 300mm in height.
<i>Other</i>	All other perches from Powderbark (<i>E. accedens</i>), Marri (<i>E. calophylla</i>), Jam Wattle (<i>Acacia acuminata</i>), Casuarina sp., <i>Xanthorrhoea</i> sp., <i>Hakea</i> sp. and <i>Dryandra</i> sp.

* The species of dead standing and dead fallen timber were not identified

4.2.3 Species and size of utilised perching trees

Between Winter 2000 and Spring 2001, the characteristics of utilised perching trees were recorded. Throughout the six seasons, the species of utilised trees was noted. Additionally, in Winter and Spring of 2001, the size of the tree was also recorded. These data were collected to compare to the expected use of the species and size of trees as measured within each site. All counts of perches on live Wandoos (LWT, LWLB and LWDB perch categories from section 4.2.2) were grouped as “Wandoo”. All counts of perches on live Powderbarks (Powderbark subcategory from “Other” category; see Appendix 4.1) were grouped as “Powderbark”. All counts of perches on other live tree

species (Marri, Jam Wattle and Casuarina subcategories from “Other” category; see Appendix 4.1) were grouped as “Other”. Counts of the size of utilised trees were denoted as sapling (1m – 5m), subcanopy (5m – 10m) or canopy (trees > 10m).

The expected proportion of perching tree species and size was calculated from data collected during the habitat selection component of the study (see section 6.2.3). Briefly, the density of tree species and sizes was calculated by point-quarter analysis (Cottam and Curtis, 1956). From these densities, the proportional contribution of the various tree species and sizes within each site were calculated. These proportions were converted to expected numbers (for comparison to utilised numbers) by multiplying the expected proportions by the total number of observed records for each site during each season.

Analysis

For each season at each site, the observed number of records of perches from each tree species and size was compared to the expected number, and was tested by chi-square analysis. Due to the different expected proportions of tree species and sizes between sites, each site was analysed separately.

In the analysis of perching tree species, the low calculated expected number of “Powderbark” records at Tommingley and Norn, resulted in the grouping of “Powderbark” and “Other” groups the “Other” category. For Koomal and Ochre, “Wandoo”, “Powderbark” and “Other” categories were used.

For tree species analysis, the level of significance of the chi-square tests was reduced to 0.00167 (0.01/6), as a result of performing six chi-square analyses for each site (Winter 2000 to Spring 2001) while for tree size analysis, the level of significance was reduced to 0.005 (0.01/2), as a result of performing two chi-square analyses for each site (Winter 2001 and Spring 2001).

Multiple comparison tests were undertaken to determine which tree species and which tree size contributed to statistical significance of the chi-square statistic of each season for each site. These were calculated through the use of Bonferroni confidence intervals (Neu *et al.*, 1974). These confidence intervals are in the form of:

$$\bar{p}_i - z_{(1-\alpha/2k)} \sqrt{\bar{p}_i (1-\bar{p}_i) / n} \leq p_i \leq \bar{p}_i + z_{(1-\alpha/2k)} \sqrt{\bar{p}_i (1-\bar{p}_i) / n}$$

where \bar{p}_i = the proportion of perches in the i th perch-tree species or size category

z = the z-statistic with

α = significance level (0.99)

k = number of perch-tree species or size categories

n = the sample size

4.2.4 Perch characteristics of sapling, subcanopy and canopy trees

To assist in the interpretation of any perching tree size preference, perch structural profiles of sapling, subcanopy and canopy trees were produced. The numbers of perches in 25cm intervals on sapling, subcanopy and canopy trees was recorded for the first three metres for each tree size category. Three metres was chosen as the upper bound of recording as 75% of perch heights were below this height (see section 4.3.5). Within each 25cm interval, the abundance of perches was estimated on a scale from zero (no perches) to five (abundant perches). A perch was defined as any dead or live branch < 30mm in diameter and > 100mm in length, protruding from the main trunk or secondary branches within 150cm of the main trunk.

Analysis was not undertaken on this component of the study, as it was only intended to provide an insight into any patterns of tree size selection. Perch structural profiles of sapling, subcanopy and canopy trees were produced from the average abundance of perches within each of the height intervals.

4.2.5 Perching height

The height of a perch is a function of the area around the perch in which prey is sought, the openness of the surrounding habitat and inevitably, the behaviour of the bird (Recher *et al.*, 2002). For this reason, the perching height is intimately tied to the foraging behaviour of the WYR.

Between Spring 2000 and Autumn 2001, perching heights were recorded from all sites. From these recorded heights, a frequency histogram of all perches was produced. No analysis was undertaken on these data.

4.2.6 Height comparison of random and pounce perches

Perching height records from each site were delineated into “random” and “pounce” perches. “Random” refers to those perches in which a foraging attempt was not subsequently performed. “Pounce” refers to those perches in which a pounce foraging attempt followed.

Analysis

A two-way analysis of variance was performed to test for differences between height of random and pounce perches, differences between sites and presence or otherwise of any interaction effect. Bartlett's test statistic was calculated in order to determine if the all samples had equal variances; an assumption of the two-way ANOVA test (Zar, 1984). Following square-root transformation, the variances of all samples were not significantly different (Corrected $B = 6.13$; $p > 0.05$).

4.3 RESULTS

4.3.1 Perch-use selection

A total of 10,578 records of perch-use selection by WYRs was collected from four study sites over six seasons (Winter 2000 to Spring 2001). Of these, LWDB contributed the highest proportion (27 – 48% of perch substrates per site; Table 4.2), followed by DFT (12 – 27%). LS contributed the smallest proportion (1 – 3%). The “other” category contributed between 2% and 15% of all perches recorded per site (see Appendix 4.1 for content of “other” perches).

Table 4.2 Percentage contribution of all perch substrates at each of the four study sites from six seasons. Sample size given in parentheses. (See Table 4.1 for definitions of perch-use abbreviations)

	Season						
	Winter 2000	Spring 2000	Summer 2000/01	Autumn 2001	Winter 2001	Spring 2001	Combined
Koomal	(413)	(514)	(405)	(379)	(462)	(455)	(2,628)
LWT	12	3	3	10	8	3	6
LWLB	6	6	6	6	3	4	5
LWDB	36	42	57	45	34	50	44
DTT	8	6	8	3	4	6	6
DTDB	7	4	1	3	4	3	4
DFT	13	15	4	12	16	12	12
DS	13	7	6	9	19	7	10
LS	1	4	0	1	1	1	2
Other	5	12	15	11	11	16	12
Tommingley	(424)	(444)	(419)	(404)	(427)	(515)	(2,633)
LWT	5	3	3	5	4	4	4
LWLB	9	5	5	3	1	3	4
LWDB	33	49	52	32	26	33	38
DTT	3	10	3	5	7	8	6
DTDB	9	8	2	5	4	3	5
DFT	22	13	17	30	35	27	24
DS	14	5	7	9	18	17	12
LS	3	0	4	6	4	3	3
Other	1	8	7	4	1	2	4
Norn	(500)	(422)	(457)	(408)	(423)	(446)	(2,676)
LWT	9	5	7	8	5	2	6
LWLB	4	10	8	3	4	1	5
LWDB	40	59	54	51	43	44	48
DTT	4	8	6	9	5	10	7
DTDB	7	6	3	3	7	7	6
DFT	28	9	19	19	28	27	22
DS	4	1	1	3	6	7	4
LS	0	1	1	0	1	1	1
Other	3	2	1	2	2	1	2
Ochre	(427)	(409)	(424)	(440)	(478)	(463)	(2,641)
LWT	3	7	8	7	4	9	6
LWLB	3	9	7	4	2	3	5
LWDB	30	40	30	20	16	26	27
DTT	2	2	8	8	7	6	5
DTDB	9	8	6	4	7	4	6
DFT	31	16	22	20	36	40	28
DS	4	6	4	5	9	3	5
LS	0	3	3	3	5	2	3
Other	17	10	13	29	15	5	15

With all sites combined, there were significant differences in perch-use selection between seasons ($\chi^2 = 638.5$, df = 40, $p < 0.0001$). This seasonal pattern of perch-use selection was evident within sites at Koomal ($\chi^2 = 289.8$, df = 40, $p < 0.0001$), Tommingley ($\chi^2 = 359.4$, df = 40, $p < 0.0001$), Norn ($\chi^2 = 247.9$, df = 40, $p < 0.0001$) and Ochre ($\chi^2 = 344.2$, df = 40, $p < 0.0001$). However, with all seasons combined, there were significant differences in the proportions of perch-use substrates between sites (χ^2

= 976.7, df = 24, $p < 0.0001$) owing to low proportional utilisation of DFT at Koomal, low proportional utilisation of LWDB at Ochre, and significantly varied proportional utilisation of “Other” perches. With all sites combined, perch-use selection varied seasonally (Figure 4.1).

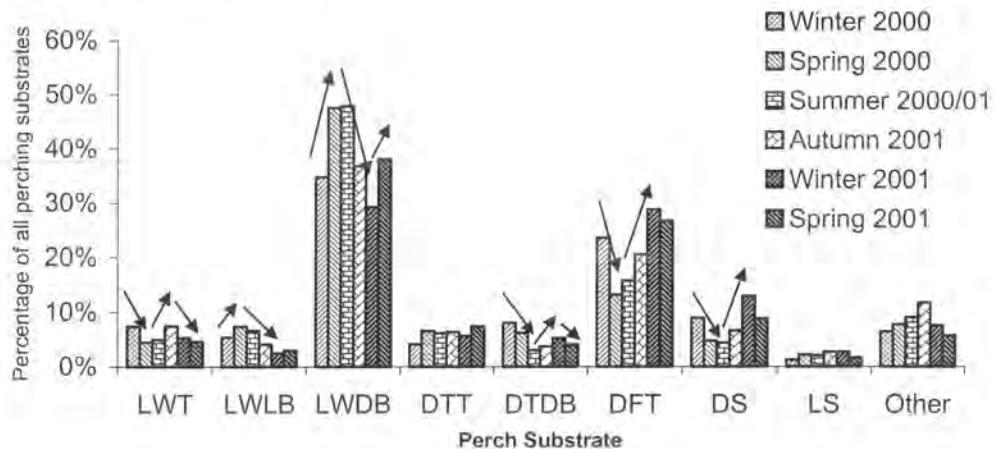


Figure 4.1 Seasonal changes in percentage of perching substrates pooled from four study sites (see Table 4.1 for explanation of abbreviations).

The seasonal pattern is exemplified by a significant increase in proportion of LWDB from Winter 2000 to Summer 2000/01 (contributing to $\chi^2 = 156.2$, df = 8, $p < 0.0001$), and a significant increase in use of DFT from Spring 2000 to Winter 2001 (contributing to $\chi^2 = 303.2$, df = 8, $p < 0.0001$, Table 4.3).

Table 4.3 Pairwise multiple comparisons of perching substrate proportions (all sites combined) between seasons (* = significant difference at corrected p-value of 0.0007).

Comparison	Chi-square	df	p-value
Winter 2000 v Spring 2000	151.1	8	< 0.0001*
Winter 2000 v Summer 2000/01	156.2	8	< 0.0001*
Winter 2000 v Autumn 2001	82.8	8	< 0.0001*
Winter 2000 v Winter 2001	83.0	8	< 0.0001*
Winter 2000 v Spring 2001	73.5	8	< 0.0001*
Spring 2000 v Summer 2000/01	26.2	8	0.001
Spring 2000 v Autumn 2001	111.3	8	< 0.0001*
Spring 2000 v Winter 2001	303.2	8	< 0.0001*
Spring 2000 v Spring 2001	178.1	8	< 0.0001*
Summer 2000/01 v Autumn 2001	68.5	8	< 0.0001*
Summer 2000/01 v Winter 2001	266.6	8	< 0.0001*
Summer 2000/01 v Spring 2001	140.6	8	< 0.0001*
Autumn 2001 v Winter 2001	106.8	8	< 0.0001*
Autumn 2001 v Spring 2001	77.6	8	< 0.0001*
Winter 2001 v Spring 2001	55.1	8	< 0.0001*

As for within-seasonal comparisons, perch-use differed between Winter 2000 and Winter 2001 ($X^2 = 83.0$, df = 8, $p < 0.0001$), and between Spring 2000 and Spring 2001 ($X^2 = 178.1$, df = 8, $p < 0.0001$). The only seasonal comparison showing no significant difference was that between Spring 2000 and Summer 2000/01 although the low p -value suggests that it is almost significant ($X^2 = 26.2$, df = 8, $p = 0.001$).

4.3.2 Species and size of utilised perching trees

Perching tree species

Koomal

Perch tree species utilisation was significantly different than expected during Winter 2000, Spring 2000 and Spring 2001 (Table 4.4). Bonferroni confidence intervals indicate that during Winter 2000, utilisation of Wandoos was significantly higher and utilisation of Powderbarks was significantly lower than expected. Similarly, in Spring 2000, utilisation of Powderbarks was significantly lower, and in Spring 2001 utilisation of “other” perches was significantly lower than expected (Figure 4.2a).

Table 4.4 Chi-square analysis of difference in perch-tree species utilisation to expected at Koomal for all seasons between Winter 2000 and Spring 2001 (* refers to significant adjusted p-value). Included is significance of species comparison Bonferroni confidence intervals, with ‘higher’ and ‘lower’ corresponding to significantly higher or significantly lower proportion of utilised (compared to expected) species, respectively. N.S indicates no significant difference in utilised compared to expected proportion of species.

	X^2	df	p -value	Bonferroni confidence interval		
				Wandoo	Powderbark	Other
Winter 2000	23.91	2	<0.0001*	higher	lower	N.S
Spring 2000	16.26	2	0.0003*	N.S	lower	N.S
Summer 2000	3.43	2	0.1799	N.S	N.S	N.S
Autumn 2001	5.75	2	0.0565	N.S	N.S	N.S
Winter 2001	4.19	2	0.1232	N.S	N.S	N.S
Spring 2001	15.89	2	0.0004*	N.S	N.S	lower

The proportion of Wandoo, Powderbark and other species exhibit a seasonal pattern of fluctuation (Figure 4.2a). Most notable is the initial decline in the use of Wandoo species between Winter 2000 and Summer 2000, before increasing again during Autumn 2001. A seasonal pattern is also present in the Powderbark species whereby there is an initial increase between Winter 2000 and Summer 2000 followed by a decrease during Autumn 2001.

Tommingley

Perch tree species utilisation was significantly different from expected during Spring 2000, Summer 2000, and Autumn 2001 (Table 4.5). Bonferroni confidence intervals indicate that during Spring 2000, utilisation of Wandoos was significantly lower and utilisation of other species was significantly higher than expected. Neither the Wandoo nor the other species comparisons showed any significant differences in utilisation and availability in Summer 2000 and Autumn 2001, even though the overall difference was significant (Table 4.5).

Table 4.5 Chi-square analysis of difference in perch-tree species utilisation to expected at Tommingley for all seasons between Winter 2000 and Spring 2001 (* refers to significant p-value). Included is significance of species comparison Bonferroni confidence intervals, with 'higher' and 'lower' corresponding to significantly higher and lower proportion of utilised (compared to expected) species, respectively. N.S indicates no significant difference in utilised compared to expected proportion of species.

	χ^2	df	p-value	Bonferroni confidence interval	
				Wandoo	Other
Winter 2000	0.17	1	0.6786	N.S	N.S
Spring 2000	66.10	1	< 0.0001*	lower	higher
Summer 2000	15.59	1	< 0.0001*	N.S	N.S
Autumn 2001	10.10	1	0.0015*	N.S	N.S
Winter 2001	0	1	1	N.S	N.S
Spring 2001	0.15	1	0.7008	N.S	N.S

As with Koomal, the proportion of Wandoo and other species exhibits a slight seasonal pattern of fluctuation (Figure 4.2b). An initial decline in the use of Wandoo species between Winter 2000 and Summer 2000, before increasing again during Autumn 2001 is a pattern evident in Tommingley as it was in Koomal. A seasonal pattern is also present in the other species whereby there is an initial increase between Winter 2000 and Summer 2000 followed by a decrease during Autumn 2001.

Norn

Perch tree species utilisation was not different from expected during any of the six seasons (Table 4.6). Bonferroni confidence intervals indicate that throughout all seasons, proportional utilisation of Wandoo and other species was not significantly different than expected. (Table 4.6).

Table 4.6 Chi-square analysis of difference in perch-tree species utilisation to expected at Norn for all seasons between Winter 2000 and Spring 2001. Included is significance of species comparison Bonferroni confidence intervals. N.S indicates no significant difference in utilised compared to expected proportion of species.

	χ^2	df	p-value	Bonferroni confidence interval	
				Wandoo	Other
Winter 2000	0.52	1	0.4729	N.S	N.S
Spring 2000	0.93	1	0.3355	N.S	N.S
Summer 2000	1.65	1	0.1987	N.S	N.S
Autumn 2001	1.16	1	0.2813	N.S	N.S
Winter 2001	1.33	1	0.2492	N.S	N.S
Spring 2001	2.74	1	0.0976	N.S	N.S

The seasonal pattern of fluctuation evident in Koomal and Tommingley is not evident in Norn, with utilised proportions approximating expected proportions (Figure 4.2c).

Ochre

Perch tree species utilisation was significantly different from expected during all seasons except Spring of 2000 (Table 4.7). Bonferroni confidence intervals indicate that during Winter 2000 and Autumn 2001, utilisation of Wandoos was significantly lower than expected. During Winter and Summer 2000, utilisation of Powderbarks was significantly higher than expected. Proportional use of other species was significantly lower in Summer 2000 and Spring 2001 and significantly higher during Autumn 2001 (Table 4.7).

Table 4.7 Chi-square analysis of difference in perch-tree species utilisation to expected at Ochre for all seasons between Winter 2000 and Spring 2001 (* refers to significant p-value). Included is significance of species comparison Bonferroni confidence intervals, with 'higher' and 'lower' corresponding to significantly higher or significantly lower proportion of utilised (compared to expected) species, respectively. N.S indicates no significant difference in utilised compared to expected proportion of species.

	χ^2	df	p-value	Bonferroni confidence interval		
				Wandoo	Powderbark	Other
Winter 2000	64.15	2	< 0.0001*	lower	higher	N.S
Spring 2000	3.47	2	0.1755	N.S	N.S	N.S
Summer 2000	77.25	2	< 0.0001*	N.S	higher	lower
Autumn 2001	57.63	2	< 0.0001*	lower	N.S	higher
Winter 2001	13.98	2	0.0009*	N.S	N.S	N.S
Spring 2001	15.87	2	0.0004*	N.S	N.S	lower

The proportion of Wandoo, Powderbark and other species exhibit a seasonal pattern of fluctuation (Figure 4.2d), although in an opposite direction as seen in Koomal and Tommingley. Most notable is the initial incline in the use of Wandoo species between Winter and Spring 2000, before decreasing to Autumn 2001. The pattern of increasing utilisation repeats during Winter and Spring of 2001. The seasonal pattern is evident in the other species also, which shows initial decline and then increased utilisation.

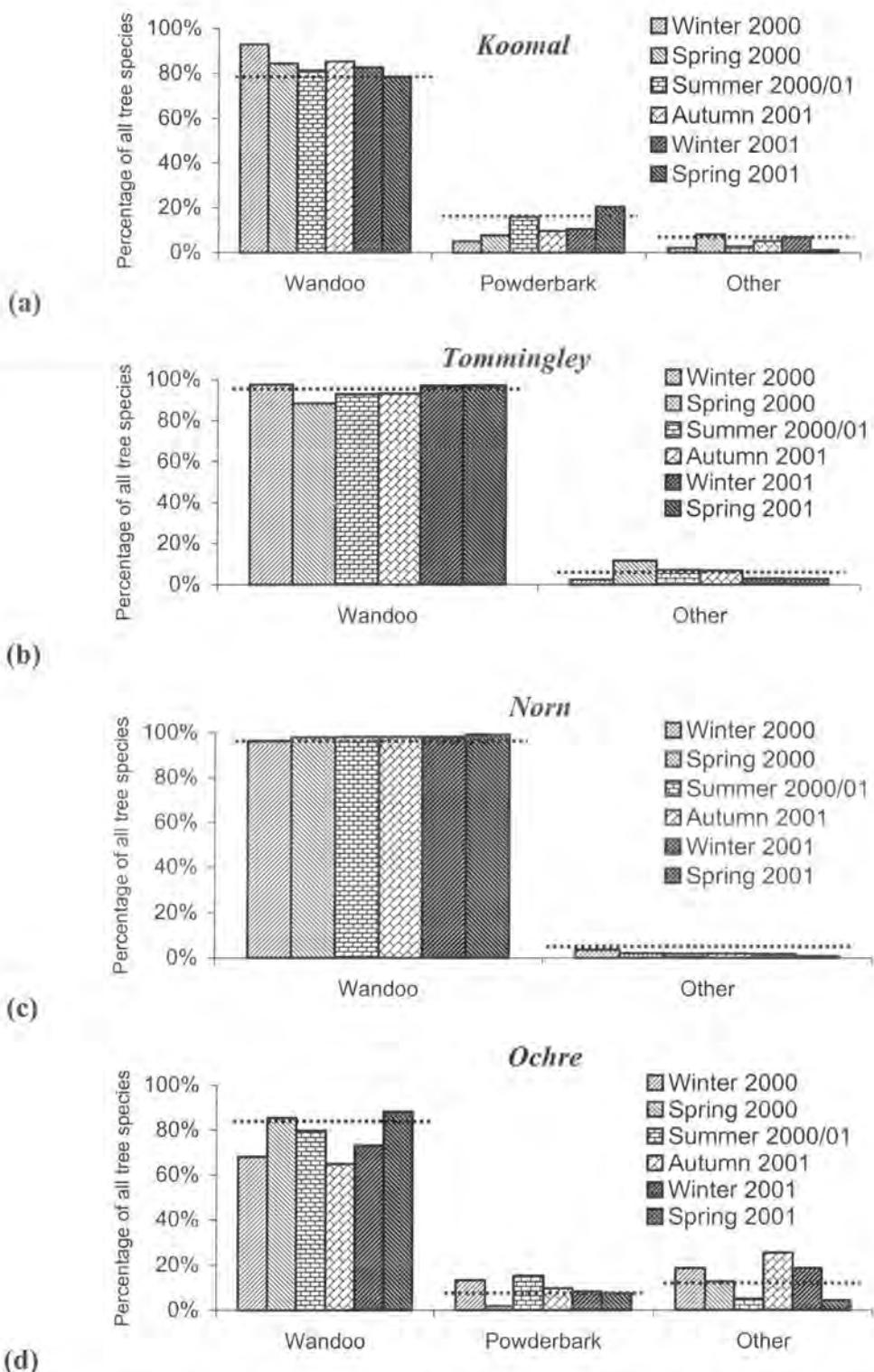


Figure 4.2 Percentage use of Wandoo, Powderbark and other species as perching trees at a) Koomal, b) Tommingley, c) Norn and d) Ochre between Winter 2000 and Spring 2001. Only the proportional use of Wandoo and other are included at Tommingley and Norn (see text). Expected proportion of each species represented by dotted line.

Perching tree size

Koomal

Perch tree size utilisation was significantly different than expected in Winter and Spring 2001 (Table 4.8). Bonferroni confidence intervals indicate that during Winter and Spring 2001, utilisation of subcanopy trees was significantly higher than expected. During Winter 2001 utilisation of canopy trees was significantly lower than expected (Table 4.8).

Table 4.8 Chi-square analysis of difference in perch-tree size utilisation to expected at Koomal for Winter and Spring 2001 (* refers to significant p-value). Included is significance of species comparison Bonferroni confidence intervals, with ‘higher’ and ‘lower’ corresponding to significantly higher or significantly lower proportion of utilised (compared to available) size, respectively. N.S indicates no significant difference in utilised compared to expected proportion of tree size.

	χ^2	df	p-value	Bonferroni confidence interval		
				Sapling	Sub-canopy	Canopy
Winter 2001	17.99	2	0.0001*	N.S	higher	lower
Spring 2001	14.49	2	0.0007*	N.S	higher	N.S

The proportion of saplings utilised is higher during Winter than during Spring, while the proportion of utilised canopy trees are lower during Winter (Figure 4.3a). The proportion of subcanopy trees utilised are similar in seasons.

Tommingley

Analysis for Tommingley indicates that perch-tree size utilisation was significantly different than expected in Winter and Spring 2001 (Table 4.9). Bonferroni confidence intervals indicate that during Winter and Spring 2001, utilisation of subcanopy trees was significantly higher than expected. During Winter 2001 utilisation of canopy trees was significantly lower than expected and during Spring 2001, utilisation of sapling trees was significantly lower than expected (Table 4.9).

Table 4.9 Chi-square analysis of difference in perch-tree size utilisation to expected at Tommingley for Winter and Spring 2001 (* refers to significant p-value). Included is significance of species comparison Bonferroni confidence intervals, with ‘higher’ and ‘lower’ corresponding to significantly higher or significantly lower proportion of utilised (compared to available) size, respectively. N.S indicates no significant difference in utilised compared to expected proportion of tree size.

	χ^2	df	p-value	Bonferroni confidence interval		
				Sapling	Sub-canopy	Canopy
Winter 2001	53.04	2	< 0.0001*	N.S	higher	lower
Spring 2001	55.43	2	< 0.0001*	lower	higher	N.S

The pattern of proportional use of sapling, subcanopy and canopy trees follows the same pattern as at Koomal. A higher proportion of saplings was utilised during Winter, similar proportions of subcanopy trees were utilised in Winter and Spring, and lower proportional utilisation of canopy trees during Winter (Figure 4.3b).

Norn

Perch-tree size utilisation was significantly different than expected in Winter and Spring 2001 (Table 4.10). Bonferroni confidence intervals indicate that during Winter and Spring 2001, utilisation of subcanopy trees was significantly higher than expected. During Winter 2001 utilisation of canopy trees was significantly lower than expected and during Spring 2001, utilisation of sapling trees was significantly lower than expected (Table 4.10).

Table 4.10 Chi-square analysis of difference in perch-tree size utilisation to expected at Norn for Winter and Spring 2001 (* refers to significant p-value). Included is significance of species comparison Bonferroni confidence intervals, with ‘higher’ and ‘lower’ corresponding to significantly higher or significantly lower proportion of utilised (compared to available) size, respectively. N.S indicates no significant difference in utilised compared to expected proportion of tree size.

	χ^2	df	p-value	Bonferroni confidence interval		
				Sapling	Sub-canopy	Canopy
Winter 2001	83.58	2	< 0.0001*	N.S	higher	lower
Spring 2001	39.95	2	< 0.0001*	lower	higher	N.S

The pattern of proportional use of sapling, subcanopy and canopy trees follows exactly the same pattern as at Koomal and Tommingley, the only exception being the slightly higher utilisation of subcanopy trees during Winter (Figure 4.3c).

Ochre

Perch-tree size utilisation was significantly different than expected only in Winter 2001 (Table 4.11). Bonferroni confidence intervals indicate that during Winter 2001, utilisation of subcanopy trees was significantly higher than expected, and utilisation of canopy trees was significantly lower than expected (Table 4.11).

Table 4.11 Chi-square analysis of difference in perch-tree size utilisation to expected at Ochre for Winter and Spring 2001 (* refers to significant p-value). Included is significance of species comparison Bonferroni confidence intervals, with 'higher' and 'lower' corresponding to significantly higher or significantly lower proportion of utilised (compared to available) size, respectively. N.S indicates no significant difference in utilised compared to expected proportion of tree size.

	X^2	df	p-value	Bonferroni confidence interval		
				Sapling	Sub-canopy	Canopy
Winter 2001	30.88	2	< 0.0001*	N.S	higher	lower
Spring 2001	0.93	2	0.6278	N.S	N.S	N.S

The pattern of proportional use of sapling, subcanopy and canopy trees follows the same pattern as at Norn, the only exception being similar utilisation of sapling trees during both seasons (Figure 4.3d).

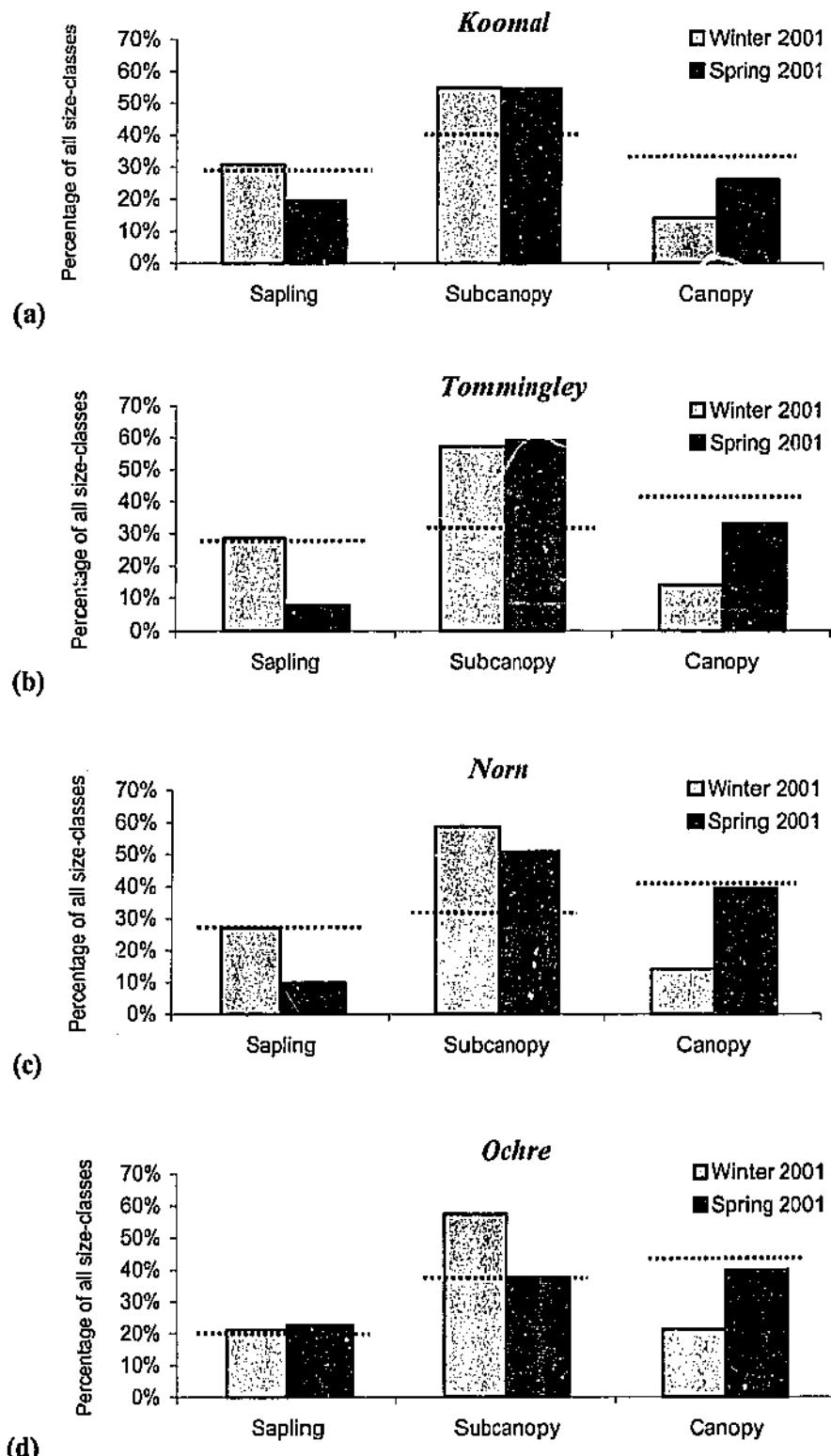


Figure 4.3 Percentage use of sapling, subcanopy and canopy trees as perching trees at a) Koomal, b) Tommingley, c) Norn and d) Ochre between Winter and Spring 2001. Expected proportion of each perch-tree size represented by dotted line.

4.3.3 Perch characteristics of sapling, subcanopy and canopy trees

Average perch structural profiles of saplings ($n = 9$), subcanopies ($n = 14$) and canopy ($n = 12$) trees show higher density of perches between 0 – 3m on sapling trees (Figure 4.4). The density of perches between 75cm and 175cm is notable in sapling trees, although these usually represented live branches with abundant foliage. The number of available perches on subcanopies was smaller although they consisted mainly of dead branches. The trunks of canopy trees rarely provided available perches below 3m in height.

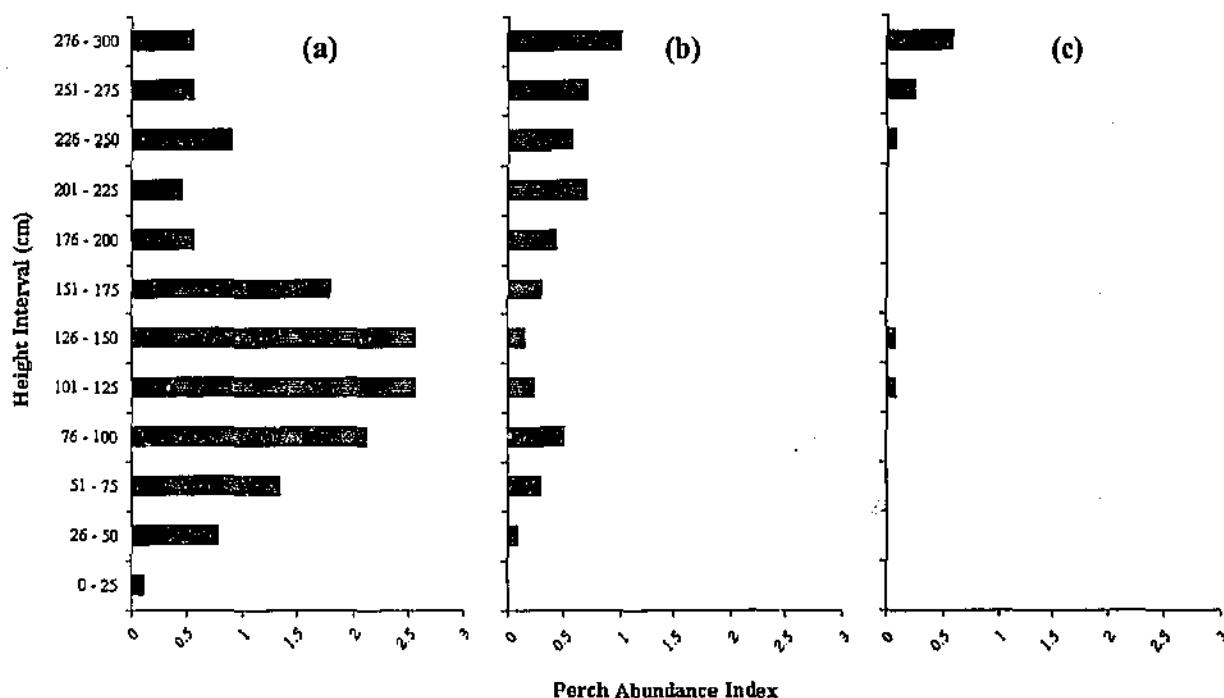


Figure 4.4 Perch structural profiles of a) sapling, b) subcanopy and c) canopy trees showing perch abundance indexes for each 25cm height interval up to 3m (see text).

4.3.4 Perching height

The average height of perches recorded throughout the study equated to $2.2 \pm 0.1\text{m}$ ($n = 469$) with a minimum at five centimetres (dead fallen timber) up to 8.0m (Wandoo canopy). The distribution of perching heights exhibits a positively skewed distribution (Figure 4.5), with 315 records (67% of all records) below 2.5m, and only 42 records (9%) above 5.0m. The high frequency of 4.0m and 6.0m could reflect observer-error categorisation of heights into nearest metre measures.

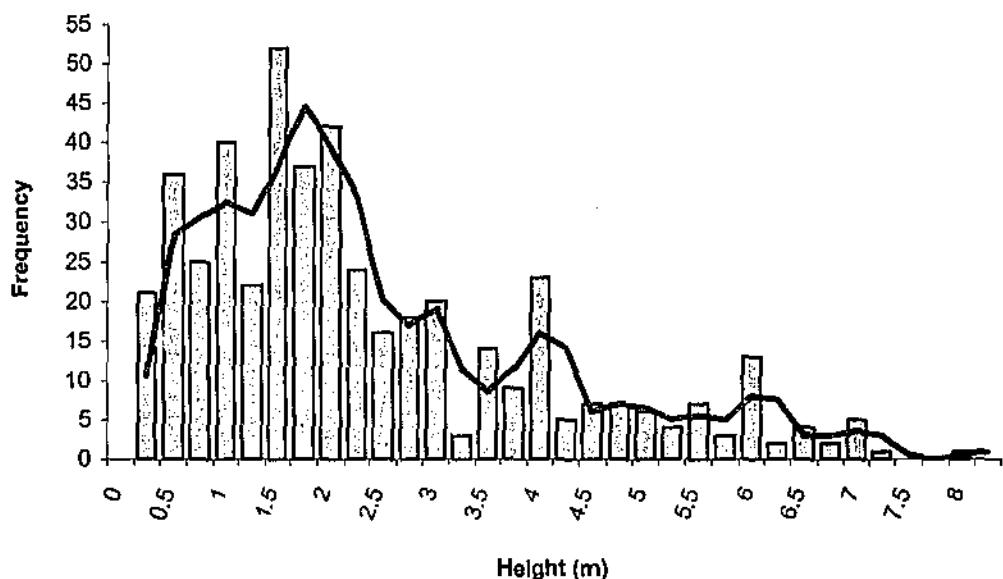


Figure 4.5 Frequency histogram of perching height ($n = 469$) showing a 2-point moving average trendline.

4.3.5 Height comparison of random and pounce perches

The height of all random perches averaged $2.7 \pm 0.2\text{m}$, significantly higher than the height of perches subsequently resulting in pounces, which averaged $1.7 \pm 0.1\text{m}$ ($p < 0.0001$; Table 4.12).

Table 4.12 Two-way ANOVA of the height of random compared to pounce-resulting perches between each of the four-study sites.

Source of Variation	SS	df	MS	F	p-value
Total	75.27	247			
Cells	58.89	240			
Perch Type	5.40	1	5.40	22.00	<0.0001
Site	4.08	3	1.36	5.54	0.0011
Perch type x Site	6.90	3	2.30	9.38	<0.0001

A significant difference was also noted in the average perching height between the four study sites, most likely as a result of inherent differences in the structural composition of the vegetation at each site. The significant difference in the site x perch type (random or pounce) interaction indicates a significant interaction of the two factors between sites (Table 4.12). This result is attributed to Ochre, where in contrast to the other sites, “pounce” perch height was greater than “random” perch height (Figure 4.6).

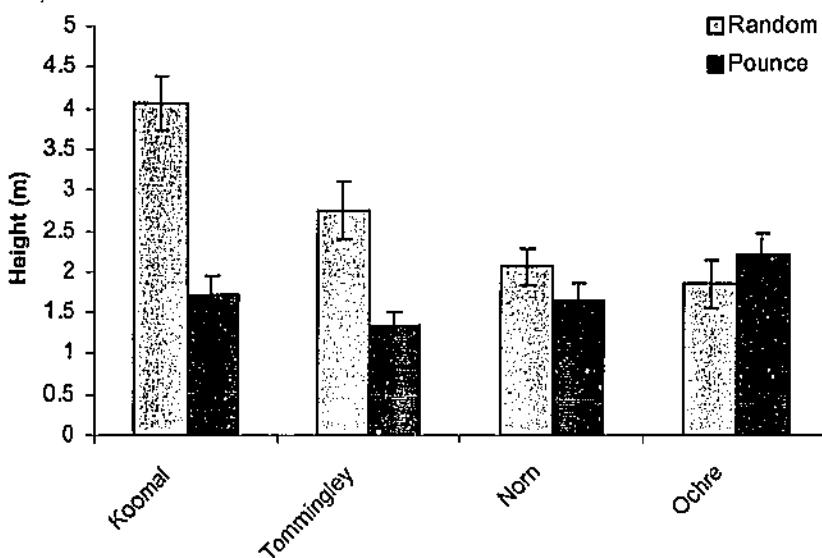


Figure 4.6 Average perching heights (\pm S.E) of random perches and perches subsequently resulting in pounces of all four study sites.

4.4 DISCUSSION

4.4.1 Observed pattern of perch-use

Utilisation of perches is an important component of the foraging ecology of the WYR. As a perch and pounce insectivore, the utilisation of perches is intimately tied to foraging behaviour. Dead branches of live wandoos (LWDB) were the most common perching substrates used throughout all sites and seasons, representing 39.1% of all perches, while dead fallen timber (DFT) represented 21.5% of all perches. Patterns of perch-use varied seasonally, corresponding with shifts in foraging behaviour.

4.4.2 Seasonality of perch-use

The higher proportion of DFT and dead shrub (DS) as perching substrates during Winter 2000 and 2001 compared to other seasons corresponds with a higher proportion of ground pouncing manoeuvres (see section 3.3.1). Similarly, the high proportion of LWDB and live branches of live wandoos (LWLW) as perches during Spring and Summer compared to other seasons corresponds to a higher proportion of 'active' foraging manoeuvres (see section 3.3.1). These results suggest seasonal perch-use is closely associated with seasonal changes in foraging manoeuvres. Stemming from this, seasonal changes in perch-use reflect selection of perches in closer proximity to potential prey, a pattern commonly encountered in other studies (see Caton *et al.*, 1992).

The significant intraseasonal difference in perch-use selection between Winters and Springs was driven by higher proportions of DFT and DS perches during 2001 compared to 2000. This pattern would be expected to favour ground-pouncing manoeuvres, although an increase in proportion of ‘active’ foraging manoeuvres was detected in 2001 (shown in the foraging behaviour of Spring 2001 compared to Spring 2000, section 3.3.1 – 3.3.3).

4.4.3 Perch-tree species selection

There was no consistent pattern of selection of perching tree species across seasons and sites, with Wandoo preferentially utilised at Koomal during some seasons, and Powderbark preferentially utilised at Norn during some seasons. Within Koomal and Ochre, there were seasonal fluctuations in perching tree species utilisation (see Figure 4.2) possibly due to habitat differences between breeding and non-breeding seasons (see also section 7.3.4). For example, during the breeding season at Ochre, Powderbark trees represented a greater proportion of the breeding territory than Wandoo, while during the non-breeding season, the territory extended into primarily Wandoo dominated woodland, resulting in an increased proportion of utilisation of Wandoo as perches.

4.4.4 Perch-tree size selection

Two clear patterns of perching tree size selection were evident. Within seasons at each site (except Spring 2001 at Norn), subcanopy trees were utilised in higher proportion than expected. Subcanopy trees, although providing fewer perches than sapling trees (see Figure 4.4), provide the most uninterrupted view of the ground from the average perching height of $2.2 \pm 0.1\text{m}$ (section 4.3.4). Sapling tree perches below this average perching height were primarily comprised of leafy material, thus restricting observation of the ground for prey. Canopy trees were not selected as perching trees, as they provide lower numbers of perches at much greater heights (Figure 4.4c).

WYRs showed temporal changes in perch-use selection. The higher proportional use of low perches (from sapling and subcanopy trees) in Winter 2001 compared to Spring 2001, is associated with a higher proportion of ground-pouncing foraging manoeuvres during Winter 2001 (see section 3.3.1). Lower proportional utilisation of sapling trees during Spring 2001 at two sites reflects the shift in utilisation to higher perches (subcanopy and canopy trees, Figure 4.3), associated with an increase in

‘active’ foraging manoeuvres and increase in foraging height. As with the overall seasonality of perch-use selection component (section 4.4.2), seasonal changes in perching tree size reflect selection of perches on trees in closer proximity to potential prey, as previously found by Caton (1992) with Bald Eagles in Montana USA.

4.4.5 Height comparison of random and pounce perches

The average height of perches resulting in pounces was significantly lower than random perches (those perches not resulting in a foraging manoeuvre). Recher and Davis (Recher *et al.*, 2002) found similar results with robins in Western Australian woodlands, where mean perch heights of ground pouncing robins were significantly lower than other perch heights of hawking and snatching robins. The results of the present study suggest that among a pool of available perches, WYRs select lower perches from which to undertake pounces. This pattern of lower heights of pounce resulting compared to random perch heights was observed consistently in three of the sites, but not at Ochre. However, the average height of random perches at Ochre approached the height of pounce perches of the other sites (see Figure 4.6). This result supports the idea that regardless of the range of available perches, WYRs utilise perches to pounce from, within a narrow range of low heights that is consistent between sites and independent of available perch heights. Utilisation of lower perches for pounces indicates the greater visibility afforded by these perches for detecting ground invertebrates.

Appendix 4.1 Percentage contribution of various perch substrates to 'other' category of Table 4.2 at each of the four study sites from six seasons. Sample size given in parentheses.

	Season						
	Winter 2000	Spring 2000	Summer 2000/01	Autumn 2001	Winter 2001	Spring 2001	Combined
Koomal	(19)	(60)	(62)	(40)	(53)	(71)	(305)
Powderbark	63	40	84	65	49	93	68
Marri	21	32	11	35	23	3	19
Jam Wattle	5	7	2	0	9	1	4
Casuarina	0	3	2	0	0	1	1
Xanthorrhoea	0	0	0	0	2	0	0
Hakea	11	5	2	0	17	0	5
Dryandra	0	13	0	0	0	1	3
Tommingley	(5)	(34)	(30)	(15)	(4)	(9)	(97)
Powderbark	80	79	40	0	0	0	44
Marri	0	0	17	20	50	0	10
Jam Wattle	0	9	0	27	0	11	8
Casuarina	20	9	7	33	50	56	19
Xanthorrhoea	0	0	0	0	0	0	0
Hakea	0	3	6	0	0	0	3
Dryandra	0	0	30	20	0	33	15
Norn	(17)	(7)	(6)	(10)	(7)	(4)	(51)
Powderbark	53	0	0	0	17	25	22
Marri	0	0	0	0	18	25	2
Jam Wattle	6	43	0	50	3	0	20
Casuarina	0	57	100	0	17	50	24
Xanthorrhoea	41	0	0	50	0	0	33
Hakea	0	0	0	0	0	0	0
Dryandra	0	0	0	0	45	0	0
Ochre	(73)	(39)	(56)	(126)	(71)	(24)	(389)
Powderbark	41	13	64	17	17	63	31
Marri	33	67	18	34	18	33	32
Jam Wattle	12	15	0	1	3	0	5
Casuarina	12	5	4	9	17	4	10
Xanthorrhoea	0	0	0	0	0	0	0
Hakea	1	0	0	1	0	0	1
Dryandra	0	0	14	39	45	0	23

CHAPTER 5: POUNCE SITE CHARACTERISTICS

5.1 INTRODUCTION

The scale at which a bird selects a foraging site is determined by numerous factors that operate at multiple temporal and spatial scales. George and Zack (2001) noted the existence of a hierarchy of selection scales, from the geographical range of the species, down to the selection of a feeding site, and then on to the actual food items that an animal chooses from those available. For example, at a geographical scale, Yellow-plumed Honeyeaters (*Lichenostomus ornatus*) exhibit selection for woodland and mallee in the higher rainfall districts of the wheatbelt (Saunders and Ingram, 1995). Within the woodland of the western wheatbelt, they exhibit macrohabitat scale selection of more productive woodlands dominated by Wandoo (*Eucalyptus wandoo*) trees (Wilson and Recher, 2001). At the microhabitat scale, Yellow-plumed Honeyeaters forage on larger trees, selecting to glean from leaves and bark, although this selection pattern varies seasonally, according to the availability of prey (Wilson and Recher, 2001).

The scale at which a foraging site is chosen is not only dependent on the bird, but also on the type of resource being utilised (Beck and George, 2000). Primarily, selection of a foraging site is dependent on the visibility of potential prey, which itself varies spatially and temporally. For example, during the drier warmer season, when the leaf litter layer heats up and desiccates, invertebrates vertically migrate into moister regions of the organic layer (Detsis, 2000). Seasonality in the detection of leaf litter prey, results in the selection of foraging substrates favouring easier detection of leaf litter prey. As such, microhabitat foraging substrate selection varies both spatially and temporally, and must be considered in studies determining foraging resource requirements.

5.1.1 Importance of ground substrate to the Western Yellow Robin

The ground represents the dominant substrate from which prey is attained by WYRs, contributing over 81% of all foraging substrates (section 3.3.2). WYRs typically utilise the lower stratum of vegetation, usually perching and foraging below 2.5m (section 4.3.5). Due to its importance in the foraging ecology of the WYR, characteristics of pounce site substrate provides information on the specific microhabitat foraging requirements of the species. This knowledge would allow one to potentially

ascertain how anthropogenic landuse practices alter the microhabitat foraging requirements of the WYR.

5.1.2 Aims

The aim of the present chapter is to examine characteristics of ground foraging sites of WYRs at two spatial scales, including ground substrate composition at exact ground pounce sites, and, at a larger scale, habitat characteristics directly surrounding ground pounce sites. Seasonal changes in selection of ground pounce sites will also be examined.

5.2 METHODS

5.2.1 Study sites

Data on pounce site characteristics of WYRs were collected from the four study sites described in section 2.1.5. All data from each site were collected within the boundaries of the observed territories (as described in section 2.1.5).

5.2.2 Ground substrate composition

The ground substrate composition of ground pounce sites was assessed using a 300mm x 300mm grid, consisting of a wooden frame divided into a 10 x 10 grid with twine (equating to one hundred 30mm x 30mm squares). The middle of the grid was placed over the exact pounce site, with the orientation of the grid randomly allocated by spinning a reference twig, from which the grid axis was matched. The proportional contribution of leaf litter, plant material, bare ground and log material within the 300mm x 300mm area was recorded. Log material was designated as any dead fallen timber (in all classes of decay) with a diameter larger than 50mm and at least 800mm long. For the example provided in figure 5.1, leaf litter contributed approximately 21%, plant material 29%, bare ground 20% and log material 30% of the total area.

Justification for the design

The decision to use a 300mm x 300mm area to assess pounce site selection was based on the premise that terrestrial prey would not traverse further than approximately 150mm from the time that the WYR detected the prey, to the point of acquisition of prey. Furthermore, this was also a scale at which an appropriate microhabitat scale selection could be accurately assessed. Beck and George (2000) suggest that the usual adopted 0.04ha circular mesoplots incorporate much of the heterogeneity of the

surrounding habitat, potentially masking microhabitat scale selection of ground substrates important for foraging.

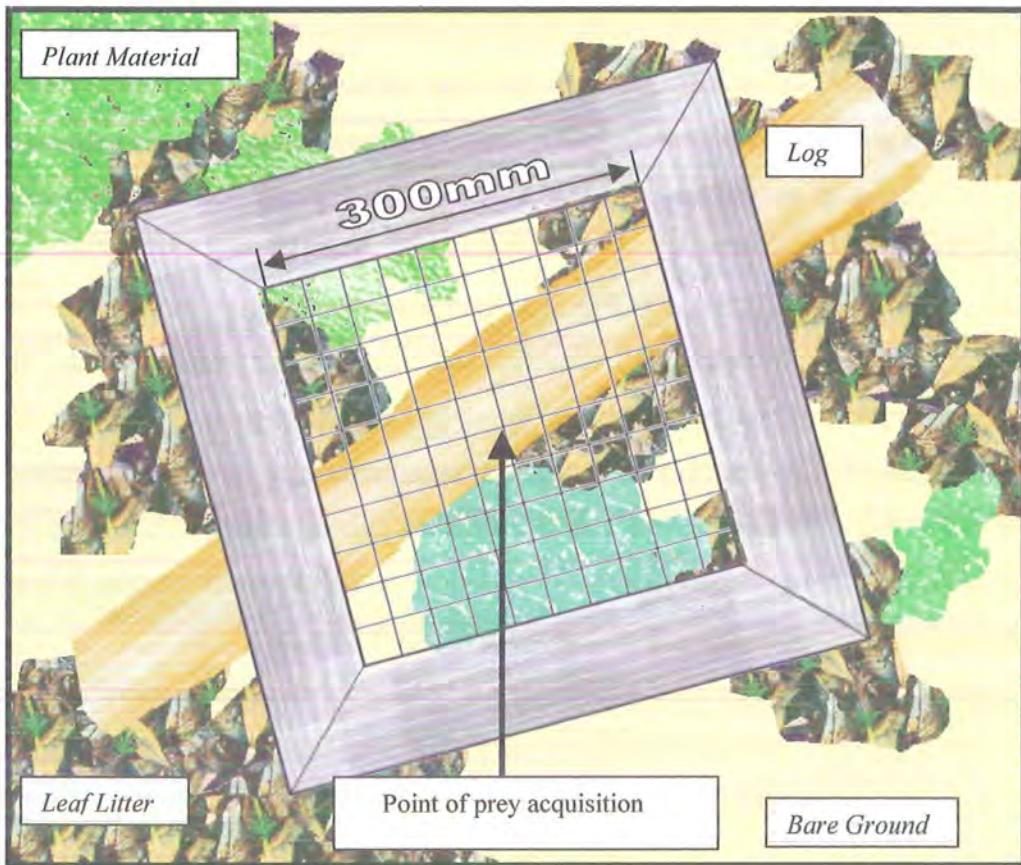


Figure 5.1 Diagrammatic representation of the 300mm x 300mm grid used to determine the proportional contribution of leaf litter, plant material, bare ground and log at pounce and random points.

5.2.3 Seasonal changes in pounce site characteristics

Seasonal changes in ground substrate composition of ground pounce sites were assessed between two seasons, namely Spring 2000 (following a cool wet winter) and Autumn 2001 (following a hot dry summer). Following the detection of an individual, a minimum of 60 seconds was allowed to elapse to avoid recording conspicuous behaviours (Kleintjes and Dahlsten, 1995), and to reduce recording behaviours influenced by my presence. The exact point of the ground pounce was noted and the ‘grid’ was placed on the centre point (Figure 5.1).

A maximum of three sequential records of pounce site characteristics were collected during any one recording period. As such, collection of data within a season occurred over a one to two week period to reduce the effect of unseasonal weather on selection of pounce sites. To account for diurnal changes in selection of ground

substrates as pounce sites, data were collected throughout the day. A total of 15 pounce sites were recorded in each of the four sites in both seasons.

Analysis

Separate tests were performed on each of the ground substrate categories; leaf litter, plant material, bare ground and log material. Before analysis, all data were transformed and tested for normality according to the Shapiro-Wilk test. Leaf litter was transformed by arcsin transformation and subsequent conversion to radians (Fowler *et al.*, 1998). Bartlett's test statistic was calculated in order to determine if all samples had equal variances; an assumption of the two-way ANOVA test (Zar, 1984). The variances of all samples were not significantly different (Corrected $B = 12.49$; $p > 0.05$). Following this, a two-way analysis of variance was undertaken to compare season and site differences as well as to determine the presence or otherwise of interaction effects. The proportions of plant material, bare ground and log material were positively skewed and could not be transformed to normality, so analysis was undertaken by a non-parametric two-factor analysis of variance by tied ranks (Zar, 1984). This form of non-parametric analysis is an extension of the Kruskal-Wallis test, using tied ranks, which calculates a Kruskal-Wallis (H) statistic, closely approximated by X^2 (Zar, 1984).

5.2.4 Comparison of pounce site to random points

A comparison was undertaken on the ground substrate composition of the abovementioned pounce sites to "random" points within each of the four study sites, to assess selection for specific ground attributes for foraging. Random points were sampled along a 530m transect within the territory boundaries of each of the four study sites, as part of the habitat selection component of the study (undertaken during Spring 2001; see section 6.2.3).

Analysis

Analysis was undertaken as in section 5.2.3 with the only difference being the replacement of the season variable, with a 'treatment' variable (pounce site and random point as the two levels). The leaf litter component was transformed by arcsin transformation and subsequent conversion to radians (Fowler *et al.*, 1998). The variances of all samples were not significantly different (Corrected $B = 7.98$; $p > 0.05$).

5.2.5 Distance of pounce sites and random points to habitat attributes

To determine the effect of surrounding habitat attributes on the selection of pounce sites, the distance to nearest shrub (> 300mm in height and width), tree (> 1m in height) and log were recorded for each pounce site and random point.

Analysis

Separate tests were performed on each of the distance categories; distance to shrub, tree and log. Before analysis, all data were transformed and tested for normality according to the Shapiro-Wilk test.

Distance to shrub was transformed by a modified inverse/reciprocal transformation:

$$x' = \frac{1}{(x + 1)}$$

where x = pre-transformed value
 x' = transformed value

Distance to tree was transformed to normality by a modified log transformation procedure:

$$x' = \log_{10}(x + 1)$$

where x = untransformed variable
 x' = transformed variable

Distance to log was transformed to normality using a standard square root transformation.

Bartlett's test statistic was calculated and the variances of each transformed sample within each test were not significantly different (Corrected B ranged between 1.37 and 10.50; $p > 0.05$). A two-way analysis of variance was undertaken comparing treatment (pounce and random point distances to shrub, tree and log) and site differences, as well as the presence or otherwise of any interaction effects.

5.3 RESULTS

5.3.1 Seasonal changes in pounce site characteristics

Leaf Litter

Leaf litter represented the largest proportion of the grid area of pounces, ranging from $50.8 \pm 8.0\%$ at Tommingley during Autumn 2001 up to $83.0 \pm 3.2\%$ at Ochre during Spring 2000 (see Figure 5.2a).

Results of the two-way ANOVA (Table 5.1) shows the effect for season was not significant ($F = 0.68, p > 0.05$), indicating no seasonal change in the proportion of leaf litter at pounce sites between Spring 2000 and Autumn 2001. There also was no significant season by site interaction effect ($F = 2.18, p > 0.05$). Significant site differences ($F = 4.97, p < 0.01$) are most likely attributed to inherent differences in the structure of the vegetation at each site, thus giving rise to different ground substrate composition.

Table 5.1 Two-way ANOVA of the proportion of leaf litter comparing season, site and season x site interaction effect.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>MS</i>	<i>F</i>	<i>p-value</i>
Total	17.34	119			
Cells	14.48	112			
Season	0.09	1	0.09	0.68	0.4097
Site	1.93	3	0.64	4.97	0.0028
Season x Site	0.84	3	0.28	2.18	0.0946

Plant Material

The proportion of plant material in the grid area of pounces ranged from $4.7 \pm 1.8\%$ at Koomal during Autumn 2001 to $16.2 \pm 4.5\%$ at Norn during Spring 2000 (see Figure 5.2b).

Results of the non-parametric two-way ANOVA (Table 5.2) shows the effect for season was significant ($H = 3.85, p < 0.05$), indicating seasonal change in the proportion of plant material at pounce sites between Spring 2000 and Autumn 2001. There was no significant site difference ($H = 3.35, p > 0.05$) or season by site interaction effect ($F = 3.12, p > 0.05$).

Table 5.2 Non-parametric two-factor analysis of the proportion of plant material, comparing season, site and season x site interaction effect.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>H</i>	<i>p-value</i>
Cells	12502.3	7		
Season	4662.5	1	3.85	0.0496
Site	4058.8	3	3.35	0.3401
Season x Site	3781.0	3	3.12	0.3728

Bare Ground

The proportion of bare ground in the grid area of pounces at all sites and in both seasons varied more than plant material or leaf litter, ranging from $1.8 \pm 0.8\%$ at Ochre in Spring 2000, to $21.1 \pm 8.8\%$ at Norn in Autumn 2001 (see Figure 5.2c).

Results of the non-parametric two-way ANOVA (Table 5.3) shows the effect for season was not significant ($H = 3.75, p > 0.05$), indicating no seasonal change in the proportion of bare ground at pounce sites between Spring 2000 and Autumn 2001. There was no significant site difference ($H = 7.53, p > 0.05$) or season by site interaction effect ($F = 0.67, p > 0.05$).

Table 5.3 Non-parametric two-factor analysis of the proportion of bare ground comparing season, site and season x site interaction effect.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>H</i>	<i>p-value</i>
Cells	14444.9	7		
Season	4538.7	1	3.75	0.0528
Site	9107.8	3	7.53	0.0569
Season x Site	798.5	3	0.67	0.8826

Log Material

Logs contributed the lowest proportion of ground substrate in the pounce site grids, with no record of log in pounce sites at Ochre in Autumn 2001. The proportion of logs at pounce sites showed a maximum of $6.2 \pm 2.9\%$ at Norn in Spring 2000.

Results of the non-parametric two-way ANOVA (Table 5.4) shows that season ($H = 2.15$), site ($H = 1.15$) and season x site interaction effects ($H = 2.82$) were not significant ($p > 0.05$), indicating no seasonal or site difference in the proportion of log material at pounce sites between Spring 2000 and Autumn 2001.

Table 5.4 Non-parametric two-factor analysis of the proportion of log, comparing season, site and season x site interaction effect.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>H</i>	<i>p-value</i>
Cells	7402.2	7		
Season	2604.0	1	2.15	0.1424
Site	1387.4	3	1.15	0.7658
Season x Site	3412.8	3	2.82	0.4201

Summary of seasonal changes in pounce site characteristics

The proportion of leaf litter, bare ground and logs in pounce sites showed no observable seasonal changes (Table 5.5). However, minor seasonal change in the proportion of plant material was shown. There were no season x site interaction effects, indicating that the effect of season on proportion of ground substrate composition at pounce sites was similar between each study site.

Table 5.5 Summary of significance of season and season x site interaction effects for proportion of leaf litter, plant material, bare ground and log of pounce sites. (N.S = Not significant, * = $p < 0.05$, ** = $p < 0.01$).

	<i>Season</i>	<i>Site</i>	<i>Interaction</i>
Leaf Litter	N.S	**	N.S
Plant Material	*	N.S	N.S
Bare Ground	N.S	N.S	N.S
Log	N.S	N.S	N.S

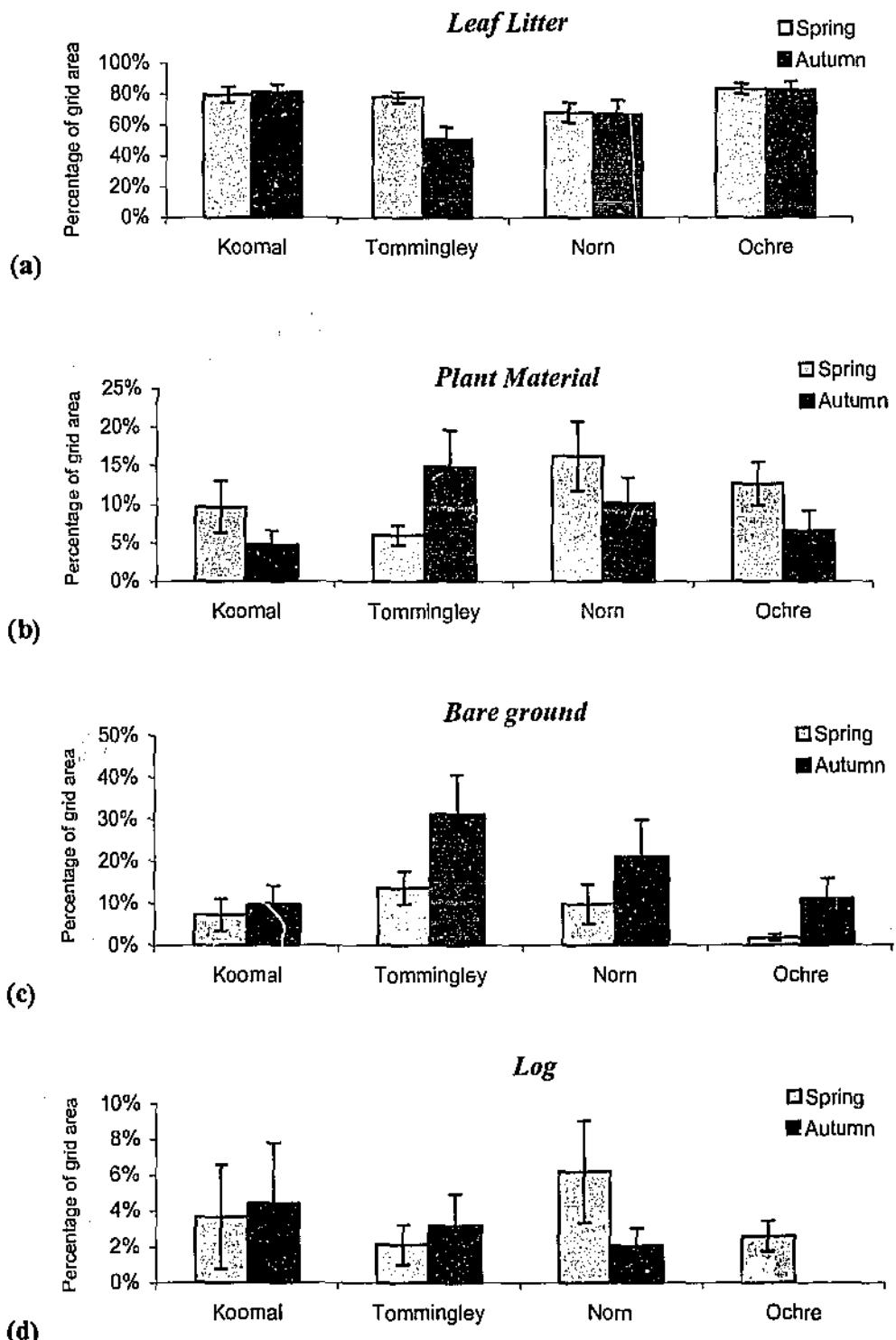


Figure 5.2 Percentage (\pm S.E) of a) leaf litter, b) plant material, c) bare ground and d) log between the four study sites, comparing Spring 2000 to Autumn 2001.

5.3.2 Comparison of pounce site characteristics at pounce and random points

In the previous section, it was established that there were no seasonal differences in the proportion of ground substrates of pounce sites within each study site; the only difference being for plant material ($p = 0.0496$). As a result of lack of seasonal differences, data on pounce site proportions of ground substrates for Spring 2000 and Autumn 2001 were pooled. These results were then compared to ground substrates recorded at random points.

It was not assessed whether there were seasonal changes in the composition of ground substrates within the study sites, although observations indicated that there were seasonal influxes in the abundance of weedy annuals associated with clumps of herb vegetation, which themselves were present throughout the year. The result is similar ground substrate composition throughout the year. Spring 2000 and Autumn 2001 pounce site data were pooled to compare to the Spring 2001 random point data. Any result of significant change in ground substrate composition between pounce site and random points, would therefore be interpreted as an actual difference, and not a relict of seasonal differences.

Leaf Litter

Leaf litter proportion at pounce sites ranged between $64.5 \pm 5.0\%$ (Tommingley) and $82.7 \pm 3.0\%$ (Ochre) while at random points, leaf litter proportion was lower at all sites, varying between $50.3 \pm 5.5\%$ (Nom) and $67.3 \pm 5.5\%$ (Tommingley) (Figure 5.3a).

Results of the two-way ANOVA (Table 5.6) shows the effect for treatment was significant ($F = 11.37, p < 0.001$), as a result of a difference in the proportion of leaf litter at pounce sites compared to random sites. There was also a significant treatment by site interaction effect ($F = 3.26, p < 0.05$), although there were no significant site differences ($F = 2.50, p > 0.05$). The significant treatment x site interaction is due to the different response at Tommingley, whereby there was a slightly higher proportion of leaf litter in random points than pounce points, contrary to patterns at Koomal, Nom and Ochre (Figure 5.3a).

Table 5.6 Two-way ANOVA of the proportion of leaf litter, comparing treatment, site and treatment x site interaction effect

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>MS</i>	<i>F</i>	<i>p-value</i>
Total	43.13	239			
Cells	38.39	232			
Treatment	1.88	1	1.88	11.37	0.0009
Site	1.24	3	0.41	2.50	0.0605
Treatment x Site	1.62	3	0.54	3.26	0.0224

Plant Material

Plant material proportion at pounce sites ranged between $7.2 \pm 1.9\%$ (Koomal) and $13.1 \pm 2.8\%$ (Norn) while at random points, plant material proportion varied between $9.4 \pm 2.6\%$ (Ochre) and $24.2 \pm 5.1\%$ (Norn) (Figure 5.3b).

Results of the non-parametric two-way ANOVA (Table 5.7) shows the effect for treatment ($H = 0.03$), site ($H = 5.30$) and treatment x site interaction effect ($H = 3.63$) were not significant ($p > 0.05$), indicating no treatment or site difference in the proportion of plant material at random and pounce sites.

Table 5.7 Non-parametric two-factor analysis of the proportion of plant material, comparing treatment, site and treatment x site interaction effect.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>H</i>	<i>p-value</i>
Cells	43225.8	7		
Treatment	165.0	1	0.03	0.8532
Site	25550.6	3	5.30	0.1486
Treatment x Site	17510.2	3	3.63	0.3039

Bare Ground

The proportion of bare ground at pounce sites ranged between $6.4 \pm 2.6\%$ (Ochre) and $22.4 \pm 5.3\%$ (Tommingley), while at random points, the proportion of bare ground varied between $15.2 \pm 4.1\%$ (Tommingley) and $25.6 \pm 5.2\%$ (Ochre).

Results of the non-parametric two-way ANOVA (Table 5.8) shows the effect for treatment was significant ($H = 9.36, p < 0.01$), indicating a difference in the proportion of bare ground at pounce and random sites. There was no significant site difference ($H = 1.51, p > 0.05$), although there was a significant treatment by site interaction effect ($F = 11.15, p < 0.05$). This result is attributed to a higher proportion of bare ground at pounce sites compared to random sites within Tommingley, contrary to patterns at Koomal, Norn and Ochre (Figure 5.3c).

Table 5.8 Non-parametric two-factor analysis of the proportion of bare ground, comparing treatment, site and treatment x site interaction effect.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>H</i>	<i>p-value</i>
Cells	106092.1	7		
Treatment	45100.4	1	9.36	0.0022
Site	7267.0	3	1.51	0.6805
Treatment x Site	53724.6	3	11.15	0.0110

Log Material

The proportion of logs at pounce sites varied between $1.3 \pm 0.5\%$ (Ochre) and $4.1 \pm 1.5\%$ (Norn), while at random points, the proportion of logs varied between $1.3 \pm 1.0\%$ (Norn) and $2.7 \pm 1.4\%$ (Ochre) (Figure 5.3d).

Results of the non-parametric two-way ANOVA (Table 5.9) show the effect for treatment was significant ($H = 5.83, p < 0.05$), indicating a change in the proportion of logs between pounce and random points. There was no significant site difference ($H = 0.67, p > 0.05$) or treatment by site interaction effect ($F = 1.19, p > 0.05$).

Table 5.9 Non-parametric two-factor analysis of the proportion of log, comparing treatment, site and treatment x site interaction effect.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>H</i>	<i>p-value</i>
Cells	37065.8	7		
Treatment	28080.1	1	5.83	0.0158
Site	3238.2	3	0.67	0.8798
Treatment x Site	5747.5	3	1.19	0.7548

Summary of pounce site characteristics at pounce and random points

The proportion of leaf litter, bare ground and logs showed significant differences between pounce sites and random points (Table 5.10). The proportion of leaf litter and log material was significantly higher at pounce sites than random points. While the proportion of plant material was not shown to be significantly different between pounce and random points, figure 5.3b shows a smaller proportion of plant material in pounce sites compared to random points. The significant difference in the proportion of bare ground showed an unclear pattern between pounce sites and random points, resulting in a significant interaction effect.

Table 5.10 Summary of significance of treatment, site and treatment x site interaction effects for proportion of leaf litter, plant material, bare ground and log. (N.S = Not significant, * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$).

	<i>Treatment</i>	<i>Site</i>	<i>Interaction</i>
Leaf Litter	***	N.S	*
Plant Material	N.S	N.S	N.S
Bare Ground	**	N.S	*
Log Material	*	N.S	N.S

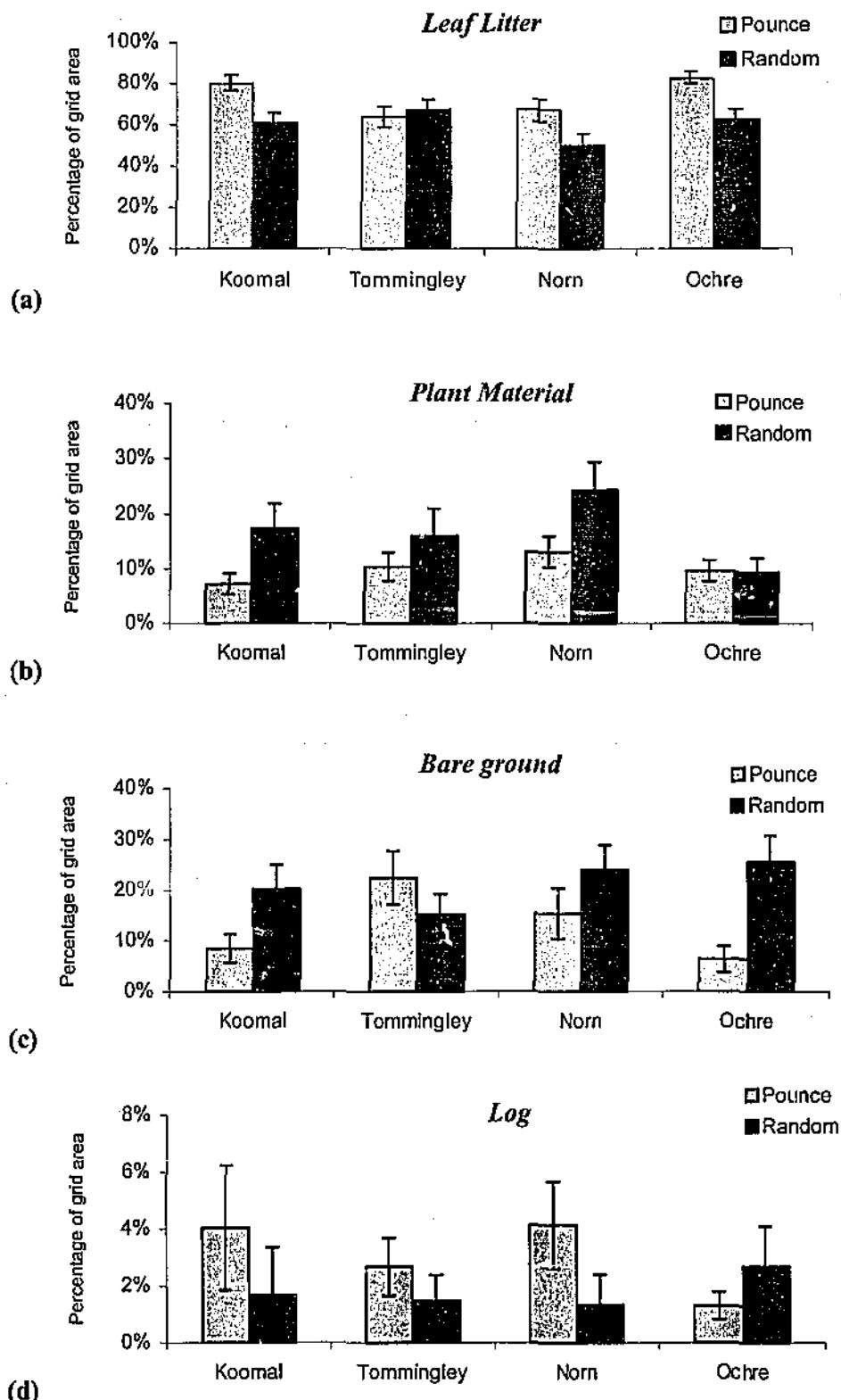


Figure 5.3 Percentage (\pm S.E) of a) leaf litter, b) plant material, c) bare ground and d) log between the four study sites, comparing random points and pounce sites.

5.3.3 Distance of pounce site and random points to habitat attributes

Shrub

The mean shortest distance to a shrub from actual pounce sites varied between sites from 0.77 ± 0.08 m (Tommingley) to 1.57 ± 0.23 m (Ochre), with distances from random points to shrubs between 0.87 ± 0.18 m (Tommingley) up to 1.95 ± 0.55 m (Koomal) (Figure 5.4a).

Results of the two-way ANOVA (Table 5.11) shows the effect for treatment was not significant ($F = 2.70, p > 0.05$), indicating no difference in the mean shortest distance to shrub between pounce sites and random points. There was a significant site effect ($F = 3.35, p < 0.05$), indicating a difference in the mean shortest distance to shrub between sites. There was no significant treatment x site interaction effect ($F = 0.53, p > 0.05$).

Table 5.11 Two-way ANOVA of mean shortest distance to shrub, comparing treatment, site and treatment x site interaction effects.

Source of Variation	SS	df	MS	F	p-value
Total	12.21	239			
Cells	11.49	232			
Treatment	0.13	1	0.13	2.70	0.1015
Site	0.50	3	0.17	3.35	0.0198
Treatment x Site	0.08	3	0.03	0.53	0.6644

Trees

The mean shortest distance to a tree actual pounce sites varied between 2.35 ± 0.31 m (Koomal) to 3.54 ± 0.41 m (Norn), with distances from random points to trees between 3.04 ± 0.38 m (Tommingley) and 4.29 ± 0.39 m (Ochre) (Figure 5.4b).

Results of the two-way ANOVA (Table 5.12) shows the effect for treatment was not significant ($F = 3.10, p > 0.05$), indicating no significant difference in the mean shortest distance to tree between pounce sites and random points. There was a significant site effect ($F = 3.37, p < 0.05$), indicating a difference in the mean shortest distance to tree between sites. There was no significant treatment x site interaction effect ($F = 1.82, p > 0.05$).

Table 5.12 Two-way ANOVA of mean shortest distance to tree, comparing treatment, site and treatment x site interaction effects.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>MS</i>	<i>F</i>	<i>p-value</i>
Total	11.18	239			
Cells	10.35	232			
Treatment	0.14	1	0.14	3.10	0.0797
Site	0.45	3	0.15	3.37	0.0193
Treatment x Site	0.24	3	0.08	1.82	0.1449

Log

The mean shortest distance to a log from pounce sites varied between 1.40 ± 0.24 m (Ochre) and 4.26 ± 0.53 m (Koomal), with distances from random points to trees between 3.05 ± 0.47 m (Ochre) and 5.15 ± 0.63 m (Norn).

Results of the two-way ANOVA (Table 5.13) shows the effect for treatment was significant ($F = 35.68, p < 0.0001$), indicating a difference in the mean shortest distance to log between pounce sites and random points. There was also a significant site effect ($F = 9.24, p < 0.0001$), indicating a difference in the mean shortest distance to logs between sites. There was no significant treatment x site interaction effect ($F = 1.88, p > 0.05$).

Table 5.13 Two-way ANOVA of mean shortest distance to log, comparing treatment, site and treatment x site interaction effects.

<i>Source of Variation</i>	<i>SS</i>	<i>df</i>	<i>MS</i>	<i>F</i>	<i>p-value</i>
Total	164.22	239			
Cells	126.55	232			
Treatment	19.47	1	19.47	35.68	< 0.0001
Site	15.12	3	5.04	9.24	< 0.0001
Treatment x Site	3.08	3	1.03	1.88	0.1330

Summary of distance of pounce site and random points to habitat attributes

The distance to shrub and tree at pounce sites was not significantly different to random points (Table 5.14), although five of the eight comparisons (representing 62.5%) presented in figures 5.4a and 5.4b indicate that pounce sites are closer to the shrubs and trees than are the random points. Including the distance to log measures, nine of the 12 comparisons (75%) show shorter distances from pounce sites to shrubs, trees and logs than random points.

Only the distances to nearest log from pounce sites and random points was significantly different however, with shorter distances from pounce sites than from random points. These results indicate that WYRs pounce significantly closer to logs than would be expected. Significant site differences for distance to shrub, tree and log reflect different densities of shrubs, trees and logs between sites.

Table 5.14 Summary of significance of treatment, site and treatment x site interaction effects for distance to shrub, tree and log between pounce sites and random points. (N.S = Not significant, * = $p < 0.05$, *** = $p < 0.001$)

	Treatment	Site	Interaction
Shrub	N.S	*	N.S
Tree	N.S	*	N.S
Log	***	***	N.S

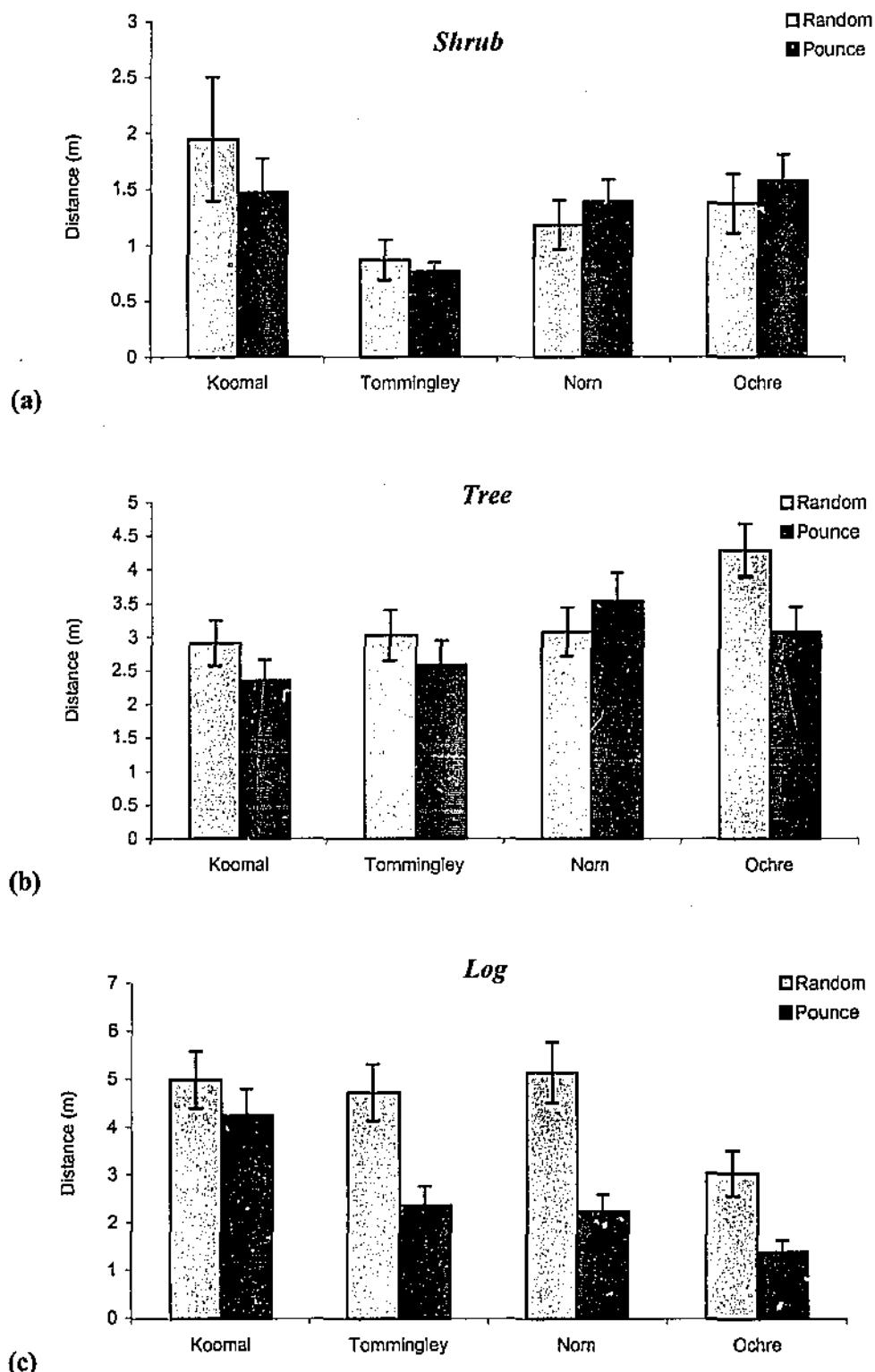


Figure 5.4 Mean shortest distance (\pm S.E) to a) shrub, b) tree and c) log of random points and actual pounce sites from all study sites.

5.4 DISCUSSION

In a study on the foraging ecology of ground-pouncing robins, Recher *et al.* (2002) showed that leaf litter contributed $57 \pm 28\%$ (mean \pm SD) of ground substrates in a three metre radius around the perch from which WYRs attacked prey in Dryandra Woodland. In the present study, the proportion of leaf litter at the 300mm x 300mm pounce site microhabitat scale was higher, representing between 65% and 83% of the pounce site grid. Conversely, the proportion of plant material in the Recher *et al.* (2002) study ($21 \pm 22\%$; mean \pm SD) was higher than in the present study (between 7% and 13%). However, the proportion of bare ground ($17 \pm 19\%$; mean \pm SD) and logs ($4 \pm 5\%$; mean \pm SD) in the Recher *et al.* (2002) study was similar to the present study (between 6% and 22% for bare ground and 1% and 4% for logs).

Differences in ground substrate composition at pounce sites between the present study and Recher *et al.* (2002) indicate the effect of scale on foraging site selection, with a higher proportion of leaf litter and lower proportion of plant material selected at the 300mm x 300mm scale compared to the larger scale utilised in Recher *et al.* (2002).

5.4.1 Seasonal changes in pounce site characteristics

Analysis of the Spring 2000 and Autumn 2001 data indicate that there were no seasonal differences in the proportional composition of leaf litter, plant material and log material at pounce sites. However, the proportion of bare ground at pounce sites was significantly higher in Autumn 2001 than Spring 2000 ($p = 0.036$). This result reflects a decrease in plant material, and an associated increase in bare ground following the dry summer season.

Results of seasonal changes in foraging behaviour (chapter three), showed seasonal changes in foraging substrates (ground, bark, air, foliage etc), reflecting changes in invertebrate abundance throughout the seasons (Moeed and Meads, 1986; Majer and Abbott, 1989; Cale, 1994a; Fox *et al.*, 1997; Burgess *et al.*, 1999). Present results tend to indicate that even though there were significant changes in the proportion of ground pouncing throughout the seasons (Chapter three), the ground substrate composition at pounce sites on which the WYRs foraged upon remained unchanged between the two seasons.

5.4.2 Comparison of pounce site characteristics at pounce and random points

WYRs selected pounce sites that had significantly more leaf litter and log material compared to random points. Selection of pounce sites with abundant leaf litter reflects selection for sites with a high abundance of habitat for leaf litter invertebrates preyed upon by WYRs. Laven and Mac Nally (1998) sighted the provision of greater amounts of food and/or foraging opportunities as a reason for the utilisation of accumulations of logs, with two species of robin; the Eastern Yellow Robin (*Eopsaltria australis*) and the Flame Robin (*Petroica phoenicea*), associated with accumulations of logs in the Box-Ironbark forest of Victoria.

5.4.3 Distance of pounce site and random points to habitat attributes

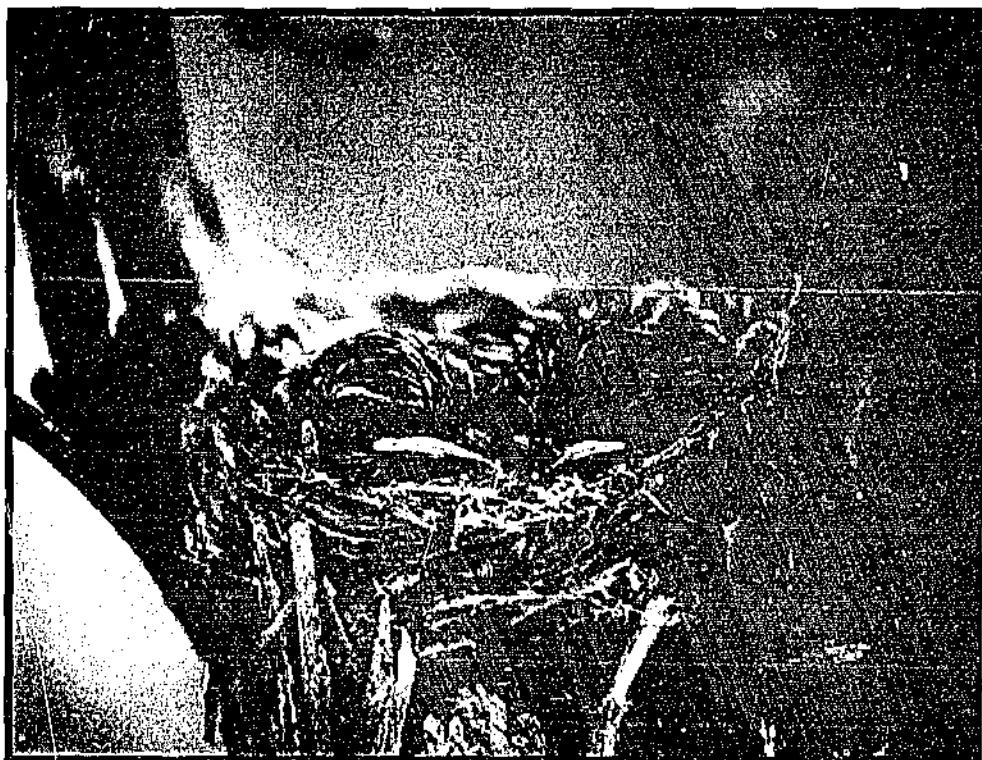
As well as selection at the foraging micro-site, there was also evidence to suggest selection of pounce sites based on the surrounding habitat. Pounce sites were significantly closer to logs than were random points across all study sites, with no pattern of selection based on distance to tree or shrub. Pouncing in close proximity to logs may represent a balance between:

- active selection of pounce sites in close proximity to logs, and
- an artifact associated with a pounce executed in close proximity to a utilised log as a perching substrate

These results reinforce the importance of logs in the foraging site selection of the WYR, as a result of the greater amounts of food associated with dead fallen timber (Laven and Mac Nally, 1998).

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PART III: HABITAT SELECTION



This curious world which we inhabit is more wonderful than it is convenient, more beautiful than it is useful; it is more to be admired and enjoyed than used.

- Henry David Thoreau (1817-1862)

Thousands of tired, nerve-shaken, overcivilized people are beginning to find out that going to the mountains is going home; that wilderness is a necessity.

- John Muir (1838-1914)

CHAPTER 6: CHARACTERISTICS OF SITE OCCUPANCY

6.1 INTRODUCTION

Habitat selection is influenced by a large number of decisions that ultimately leads to the settling of an organism in a particular location (George and Zack, 2001). In birds, an association with specific habitat attributes is believed to represent an evolved behaviour to select habitat providing appropriate resources for reproduction and survival (Steele, 1993). This evolved behaviour is shaped by a bird's morphology, which allows them to forage efficiently by maximising their ability to find, catch and handle prey in certain vegetation structures (Cody, 1981).

6.1.1 Factors contributing to habitat selection

There are a large number of factors that govern the evolved behaviour leading to habitat selection, including food availability, foraging sites, perching sites, nesting sites, favourable microclimates, and an avoidance of competitors, predators and parasites. These factors are by no means mutually exclusive in that reproductive output and survival is affected by food availability, provision of adequate nesting sites and nest predators.

6.1.2 Spatial and temporal scales

Habitat selection is not restricted to any single spatial or temporal scale, with an association existing between both (Figure 6.1). The choice of a foraging site by an insectivorous bird changes continuously in response to food availability, presence of predators, temperature and wind (George and Zack, 2001), with foraging behaviour shifting within seconds (small scale temporal changes) as well as seasonally (large scale temporal changes, see Chapter three).

Habitat selection also operates at multiple spatial scales, from the micro-site selection of specific nesting sites (Calder, 1973) to the selection of habitat at a landscape scale influenced by size, shape, isolation, and configuration of remnant habitat patches (George and Zack, 2001). Furthermore, the ability of a species to persist within a highly fragmented environment is dependent on the species' dispersal ability. Therefore, habitat selection is also shaped on a landscape scale.

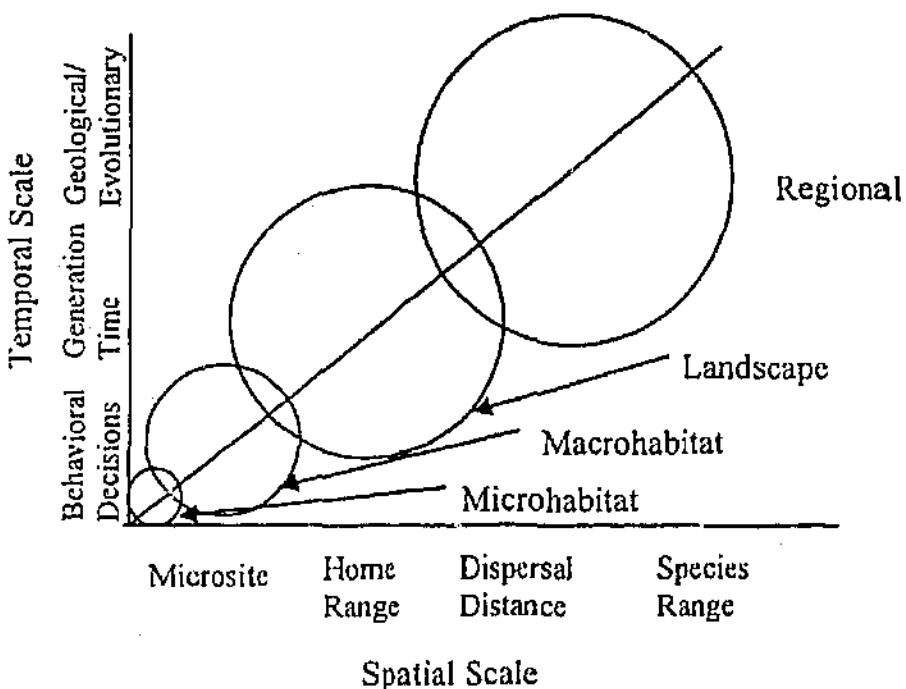


Figure 6.1 Relationship between temporal and spatial scales of habitat selection, with processes operating at small spatial scales occurring over short time periods and processes at large spatial scales operating over longer time periods (George and Zack, 2001).

6.1.3 Importance of habitat selection of the Western Yellow Robin

In the present chapter, an attempt was made to characterise habitat selection of WYRs, during the breeding season, at the three-hectare patch/territory scale. This spatial scale was chosen as it represented the approximate size of breeding territories of WYRs (see section 7.3.4). The breeding season was deemed the most appropriate time to assess habitat selection, as the ability of a population of animals to persist in an environment is dependent of the suitability of the habitat during the breeding season. This does not rule out the importance of non-breeding territories to the ecology of the WYRs, merely that the breeding territory characteristics are of more potential importance in understanding the overall habitat requirements of the species.

6.1.4 Aims

The aim of the present chapter was to determine habitat attributes associated with site occupancy of the WYR during the breeding season. Even though habitat selection of the WYR was assessed at the three-hectare patch/territory scale, variables

were incorporated from a range of scales from the microhabitat to the landscape scale (see section 6.2.3 for variables included).

6.2 METHODS

6.2.1 Study area

Analysis of characteristics of site occupancy of the WYR was undertaken in 52 sites in seven of the ten blocks constituting Dryandra Woodland (Figure 6.2). The 52 sites included the four foraging ecology study sites (Koomal, Tommingley, Norn and Ochre).

Each of the sites were chosen independently, by driving along tracks within Dryandra Woodland and spacing sites at least 1000m apart. It was assumed that the presence of WYRs within two adjacent sites at least 1000m apart would represent different territories. The only exception to the minimum distance decree was the location of site 'HS 8' (see Appendix 6.1), which was located in close proximity to the 'Tommingley' site, although WYRs were not present at 'HS 8' in any case. An effort was made to select sites:

- a) incorporating diverse woodland vegetation types
- b) with uniform vegetation structure and diversity
- c) within the outlying blocks
- d) at varying distances to the woodland/farmland ecotone.

Selection of sites in kwongan or heath vegetation was avoided. At two sites, heath represented over 60% of the site area surveyed and so were later omitted (therefore the absence of 'HS 11' and 'HS 18' sites; Appendix 6.1). The reason for such a decision was as follows: at 'HS 11', WYRs were recorded in woodland (representing about 20% of the surveyed area) directly adjacent to a large patch of heath (representing 80% of the surveyed area). In attempting to quantify vegetation attributes associated with site occupancy of the WYR, the high mean density of shrubs and low mean density of trees represented by the dominant heath vegetation brings about an inaccurate representation of the habitat selected by WYRs.

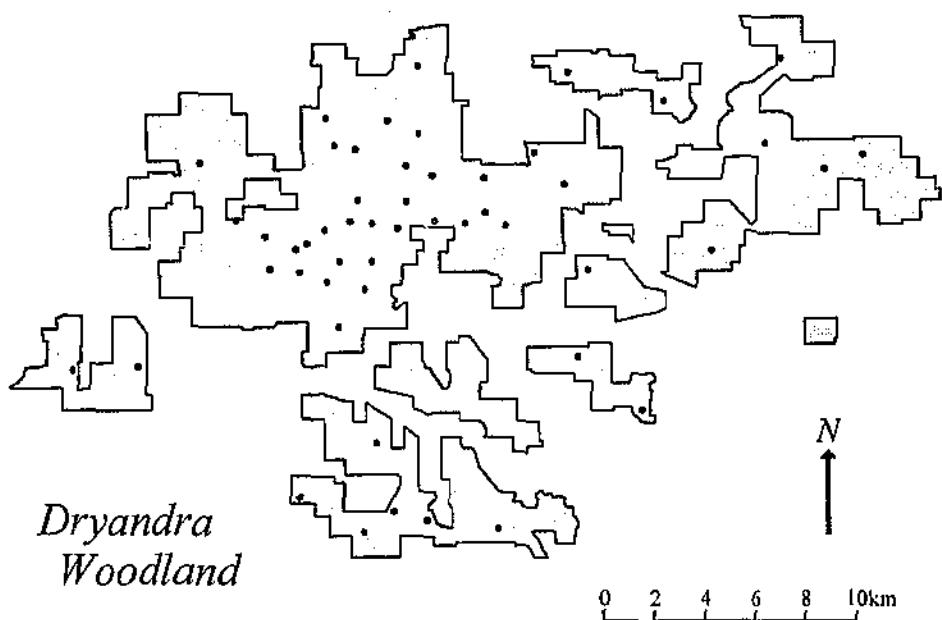


Figure 6.2 Map showing the 52 study sites throughout Dryandra Woodland.

6.2.2 Site sampling methodology

During 2000, the average breeding territory size of the WYRs was estimated as being approximately 2.6ha ($n = 2$; see section 7.3.4). As the habitat selection component of the study was being undertaken during the 2001 breeding season, it was deemed appropriate to determine habitat associations attributed to site occupancy at or near to this scale (see section 6.1.3).

A 530 metre transect was walked within each site in a rectangular shape (Figure 6.3). Along this transect, a large number of vegetation and habitat variables were recorded (see section 6.2.3). An area out to 40 metres either side of the transect was assessed for the presence of any WYRs. The site was deemed an ‘occupied’ site if at least one WYR was recorded in any part of the surveyed area. If no WYRs were present within the boundaries of the surveyed area, then the site was deemed an ‘unoccupied’ site.

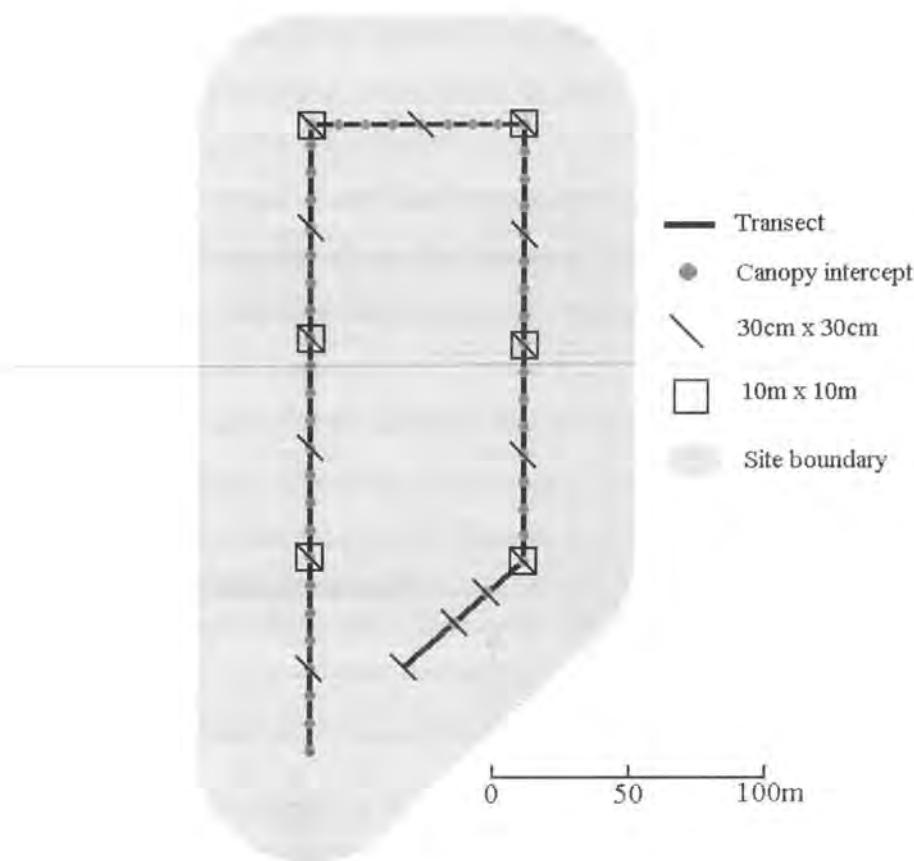


Figure 6.3 Diagrammatical representation of the transect shape, and the sampling methodology employed during vegetation and habitat sampling of sites.

6.2.3 Vegetation and habitat variables

A total of 41 vegetation and habitat variables were recorded for each of the 52 surveyed sites. Variables ranged from a microhabitat scale (e.g. leaf litter in 300mm x 300mm area) to a landscape scale (e.g. fragment co-efficient). Table 6.1 categorises each variable into groups according to the type of variable represented.

Overstorey cover

Every ten metres along the 530 metre transect, overstorey cover was assessed as cover or no cover (Figure 6.3), with overstorey cover designated as sapling and/or subcanopy and/or canopy tree. This resulted in the calculation of the four overstorey cover variables one through to four (see Table 6.1).

Ground substrate composition

Every 40 metres along the transect, the ground substrate composition in a 300mm x 300mm area was recorded, as in the pounce site characteristics component of the study (section 5.2.2). The proportional composition of leaf litter, plant material, bare

ground and log was recorded, giving rise to variables 38, 35, 40 and 33 respectively (see Table 6.1). The distance to nearest tree, shrub and log was also measured as in the pounce-site characteristics component of the study (section 5.2.2), giving rise to variables 14, 24 and 34 respectively.

Tree and shrub density and aggregation

Every 80 metres along the transect, the density of sapling, subcanopy and canopy trees was recorded (see Figure 6.3), through the use of a modified point-quarter method (Cottam and Curtis, 1956). The sampling point was firstly divided into quarters, with the axis of each quarter aligning the axis of a spun reference twig. The distance to the nearest sapling, subcanopy and canopy tree was recorded within each quarter (Figure 6.4). If any nearest tree fell along an axis line, a coin was flipped and designated to one of the quarters accordingly.

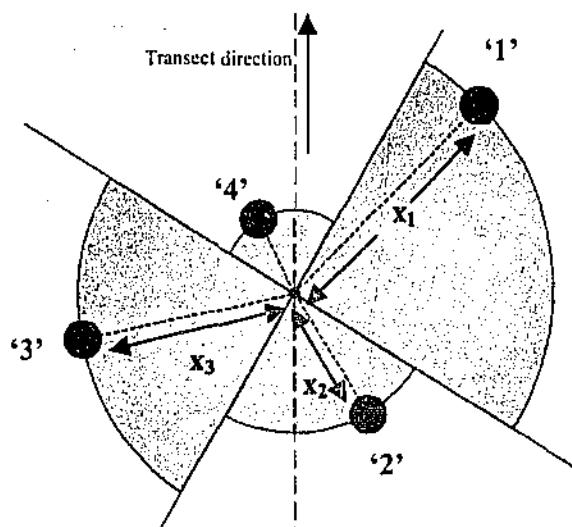


Figure 6.4 Diagrammatic representation of the point-quarter method showing the four closest trees within each quarter and the respective areas of the quarter circles. Note also the tilted axis of the quarters aligned with a spun reference stick.

The area of each quarter circle was calculated as:

$$\text{Area} = \Delta x_1 = \frac{1}{4}(\pi x_1^2)$$

where: x_1 = distance to tree 1 (in metres)

Δx_1 = area of the quarter circle for tree 1

From this measure, the average density in trees per hectare were calculated for each sample point as:

$$\text{Density} = 10,000 / \left(\frac{\sum_{i=1}^4 \Delta x_i}{4} \right)$$

Table 6.1 Categories and names of the 43 individual vegetation and habitat variables measured as part of the habitat characteristics of site occupancy component of the study.

Category	Variable name	Variable number
Tree		
Overstorey Cover	% Sapling Overstorey % Subcanopy Overstorey % Canopy Overstorey % Total Overstorey	1 2 3 4
Species Composition	% Wandoo % Powderbark % Marri % Jam Wattle % Casuarina % sapling Wandoo % subcanopy Wandoo % canopy Wandoo Shannon Diversity trees	5 6 7 8 9 10 11 12 13
Tree Density	Distance to tree Sapling trees/hectare Subcanopy trees/hectare Canopy trees/hectare Number of saplings in 10m x 10m Number of subcanopies in 10m x 10m Number of canopies in 10m x 10m	14 15 16 17 18 19 20
Aggregation	Aggregation of saplings Aggregation of subcanopies Aggregation of canopies	21 22 23
Shrub		
Shrub Density	Distance to Shrub Shrubs per hectare % Live shrub in 10m x 10m % Dead shrub in 10m x 10m	24 25 26 27
Species Composition	% shrubs <i>Gastrolobium</i> Shannon Diversity shrubs	28 29
Shrub Size/Aggregation	Aggregation of shrubs Average shrub height Average shrub width	30 31 32
Log		
Log Abundance	% Log in 30cm x 30cm Distance to log	33 34
Ground Substrate		
Plant Material	% Plant in 30cm x 30cm % Herbs in 10m x 10m % Grasses in 10m x 10m	35 36 37
Leaf Litter	% Leaf litter in 30cm x 30cm Leaf litter in 10m x 10m	38 39
Bare Ground	% Bare ground in 30cm x 30cm	40
Landscape Position		
Edge/Shape Index	Fragment co-efficient	41

The average density of the subject throughout the site was calculated as the average density of all sampling points within the site. In the present study, a total of six point-quarter measures were taken per site (Figure 6.3). Using this method, the average density of sapling, subcanopy and canopy trees (variables 15, 16 and 17 respectively), as well as shrubs (variable 25) were calculated. Using this data, an index of aggregation (clumping) was calculated for each of these variables (see Roth, 1976), giving rise to variables 21, 22, 23 and 30 respectively. Aggregation is calculated as the average density per site, divided by the standard deviation of the densities at each of the six sampling points.

Tree species diversity

As well as recording the density of sapling, subcanopy and canopy trees, the species of each of the trees was also noted. This gave rise to the calculation of proportional species composition of trees within each of the sites (variables five through to nine), as well as the proportional contribution of Wandoos as sapling, subcanopy and canopy trees (variables ten, eleven and twelve). From this data, the diversity of trees was recorded as Shannon diversity (variable 13).

Shrub morphology and species diversity

For each of the shrubs recorded during density measures, the average shrub height (variable 31) and width (variable 33) was recorded, as were proportional composition of *Gastrolobium* (variable 28) and Shannon diversity (variable 29; Table 6.1).

Proportional cover of understorey vegetation and leaf litter measures

Every 80 metres along the transect, corresponding with the point-quarter measures, the quarter axes were formed into four '5m x 5m' quadrats. Within these, the percentage cover of live shrubs (variable 26), dead shrubs (variable 27), herbs (variable 36) and grasses (variable 37) were calculated and subsequently averaged over the four quadrates to give a value for a 10m x 10m quadrate. As the categories were not mutually exclusive, the total cover could exceed 100%, as live shrubs could cover 80% of the area, with herbs also covering 80%. Leaf litter density was also calculated within each of the 5m x 5m quadrates (variable 39). A total of nine indexes of leaf litter abundance were utilised between sparse to dense. These leaf litter indexes were averaged over the four quadrates to give a value for each 10m x 10m quadrate area.

Area counts of trees

Within each of the 5m x 5m quadrates, the total number of sapling, subcanopy and canopy trees was recorded. The values of each 5m x 5m quadrat were summed to give total number of sapling (variable 18), subcanopy (variable 19) and canopy trees (variable 20) per 10m x 10m quadrat.

Distance to edge/fragment shape index

The final variable (41) recorded was the ‘fragment co-efficient’ variable. This variable is an extension of the popular distance to edge measure, although takes into account the distance to edge of a site in relation to the shape of the fragment (see Figure 6.5).

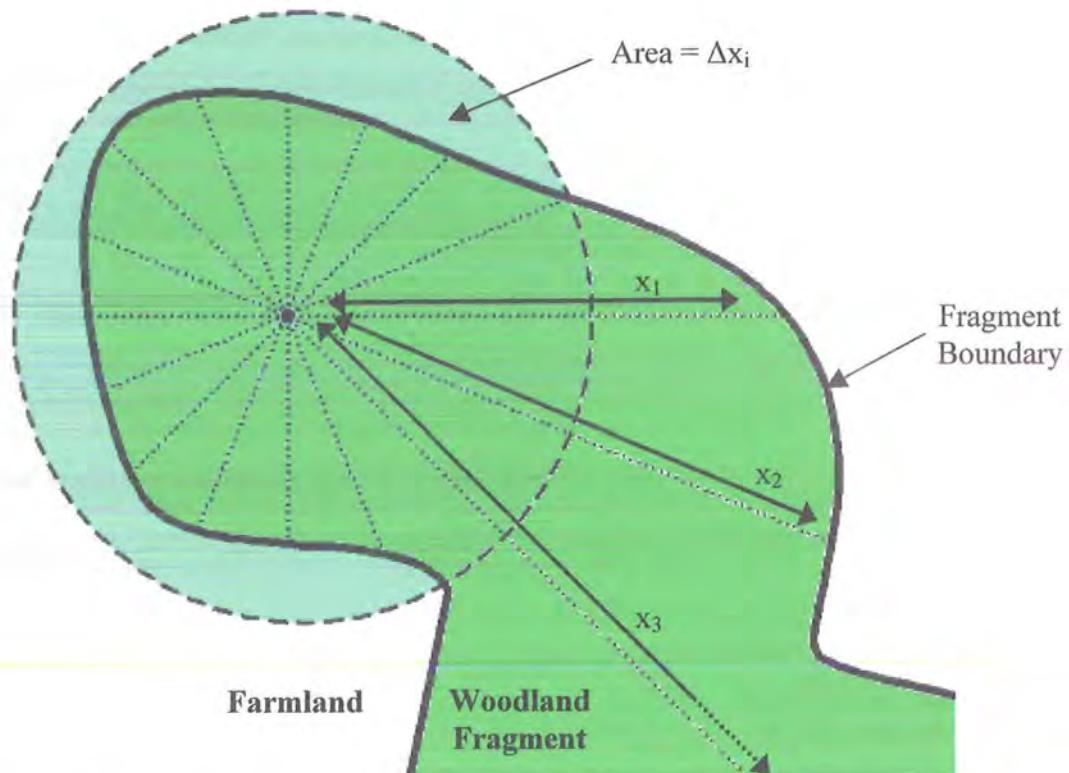


Figure 6.5 Diagrammatical representation of the calculation of the ‘fragmentation co-efficient’ variable in a hypothetical fragment, showing the distance to fragment edge (x_i ; dotted lines) in 16 directions, and the area of the circle with a radius of the average distance (Δx_i ; dashed line).

The co-efficient was calculated by firstly measuring the average distance to the edge of a fragment in sixteen cardinal directions. From this, the area of a circle with a radius of the average distance to edge was calculated:

$$\text{Area} = \Delta x_i = \pi \bar{x}_i^2$$

and was divided by the standard deviation of all 16 distances, to give the final fragment co-efficient value:

$$\text{Fragment Co-efficient} = \Delta x_i / \sigma_{x_i}$$

The resulting ‘fragment co-efficient’ value is a non-unit value, comparable between sites at the same landscape scale.

Analysis

All of the vegetation/habitat variables were assessed for co-linearity through the calculation of a correlation matrix of all variables. Factor analysis was also undertaken, which reduces a large number of variables into a smaller number of factors (Tabachnick and Fidell, 1996). ‘Factors’ are generally interpreted by the variables that correlate with them. Rather than replacing co-linear variables with a factor variable, the most reliable and accurate co-linear variable was retained. Following this process, the remaining variables were transformed and tested for normality according to the Shapiro-Wilk test. Following co-linearity and factor analysis, 15 variables were eliminated from further analysis (see Table 6.2). Of the remaining 26 variables, all percentage unit variables were transformed to normality by ARCSIN transformation with remaining variables transformed to normality by square root transformation. Four of the remaining 26 variables could not be transformed to normality (see Table 6.2) and were removed from further analysis, resulting in the omission of 19 variables, thus leaving 22 variables for further analysis.

Table 6.2 Variables omitted from further analysis, showing variable retained in its place (if any) as well as the justification of omitting the variable.

	<i>Variable Omitted</i>		<i>Variable retained</i>	<i>Justification</i>
1	% Sapling Overstorey			Minor, non-normality
2	% Subcanopy Overstorey	4	% Total Overstorey	Correlation
3	% Canopy Overstorey			Correlation
6	% Powderbark			Correlation, factor analysis
10	% sapling Wandoo	5	% Wandoo trees	Correlation, factor analysis
11	% subcanopy Wandoo			Correlation, factor analysis
12	% canopy Wandoo			Correlation, factor analysis
7	% Marri			Minor, non-normality
8	% Jam Wattle			Minor, non-normality
9	% Casuarina			Minor, non-normality
14	Distance to tree	15	Sapling trees/hectare	Correlation, accuracy
18	Number of saplings in 10m x 10m			Correlation, accuracy
19	Number of subcanopies in 10m x 10m	16	Subcanopy trees/hectare	Correlation, accuracy
20	Number of canopies in 10m x 10m	17	Canopy trees/hectare	Correlation, accuracy
24	Distance to Shrub	24	Shrubs per hectare	Correlation, factor analysis
26	% Live shrub in 10m x 10m			Correlation, factor analysis
32	Average shrub width (cm)	31	Average shrub height (cm)	Factor analysis
36	% Herbs in 10m x 10m	35	% Plant in 30cm x 30cm	Correlation, accuracy
39	Leaf litter in 10m x 10m	38	% Leaf litter in 30cm x 30cm	Correlation, factor analysis

Univariate analysis

Univariate analyses were undertaken on the remaining 22 transformed variables prior to multivariate analysis. Anderson and Shugart (1974) discuss how univariate analysis is a standard procedure used to eliminate unimportant variables prior to multivariate analysis. Univariate analyses were undertaken using two-sample t-tests, testing for differences among sites found to be occupied or unoccupied by WYRs. Due to the multiple number of tests performed, Bonferroni correction was taken into account when interpreting the calculated *p*-value. These analyses provide a preliminary insight

into the underlying factors giving rise to site occupancy of the WYR within the 52 study sites.

Multivariate analysis

In order to further reduce the number of variables, Principal Components Analysis (PCA) was undertaken. Eleven variables with eigenvector scores larger than |0.3| within the first three extracted principal component axes were retained (Tabachnick and Fidell, 1996)(see Table 6.3).

Table 6.3 Final variables used in PCA

<i>Variable</i>	
4	% Total overstorey
5	% Wandoo
13	Shannon Diversity trees
17	Canopy trees/hectare
21	Aggregation of saplings
28	% shrubs <i>Gastrolobium</i>
30	Aggregation of shrubs
31	Average shrub height (cm)
34	Distance to log
38	% Leaf litter in 30cm x 30cm
41	Fragment co-efficient

Following this, a final PCA was undertaken. ‘PCA reduces the original number of variables to a smaller set of new, uncorrelated variables (components) that still account for a large part of the total variation in the original data set’ (Wiens and Rotenberry, 1981: pg 25). Three components were extracted, with each new component extracting less variance than previous components (Tabachnick and Fidell, 1996).

Following extraction of principal components, correlations of variables with extracted components are displayed in a loading matrix. Variables with high similar factor loadings (larger than |0.30|) within each component were collected together in order to develop a concept unifying them all (Tabachnick and Fidell, 1996).

Principal component scores of occupied and unoccupied sites were analysed using a two-sample t-test to determine the degree to which each component (and its correlated variables) delineated occupied from unoccupied sites on a PCA plot. Due to the inequality of the standard deviations of the principal component scores of occupied and unoccupied sites on PCI ($F = 2.598$; $p < 0.05$), a Mann-Whitney *U*-test was undertaken instead.

Habitat Complexity Analysis

The structural complexity of vegetation has been correlated with species abundance and diversity (MacArthur and MacArthur, 1961; MacArthur *et al.*, 1966). In order to determine if WYRs are selecting for sites with high habitat complexity, a comparison was made of habitat complexity of occupied and unoccupied sites. A measure of vegetation structural complexity was calculated for each of the 52 sites, through the calculation of habitat complexity scores based on five habitat components (Table 6.4). The scores for each habitat complexity component were determined from one of the measured variables believed to best represent the component. Each component was rated on a scale of one to five, with the sum of all component scores providing a final habitat complexity score (HCS) for each site.

Table 6.4 Components comprising the habitat complexity score, the variables they were calculated from, as well as the variable ranges for each component complexity score.

	<i>Variable</i>	<i>Component</i>	<i>Habitat complexity score</i>				
			1	2	3	4	5
4	% Total overstorey	Canopy	< 20%	20 – 30%	30 – 40%	40 – 50%	50 – 60%
34	Distance to log	Logs	> 7m	6 – 7m	5 – 6m	4 – 5m	< 4m
25	Shrubs per hectare	Shrub	< 500	500 – 1000	1000 – 2000	2000 – 5000	> 5000
35	% plant in 30cm x 30cm	Herbage	< 5%	5 – 10%	10 – 20%	20 – 30%	> 30%
38	% leaf litter in 30cm x 30cm	Leaf litter	< 50%	50 – 60%	60 – 70%	70 – 80%	> 80%

Sites with a low HCS (< 10) are indicative of those sites with minimal vegetation structural complexity, with low canopy and understorey cover. Sites with a high HCS (> 18) are indicative of those sites with high vegetation structural complexity, with high canopy cover, a high density of understorey shrubs, ground herbage, logs and leaf litter.

Analysis was undertaken through a two-sample t-test comparing the average HCS of occupied and unoccupied sites. In order to determine which of the habitat complexity components were contributing to the final t-test result, multiple t-tests were performed on each component, comparing occupied and unoccupied sites. Significance levels were adjusted for the multiple habitat component tests according to the Bonferroni correction, with $\alpha = 0.01$ ($0.05/k$; where $k = 5$ habitat complexity components).

6.3 RESULTS

6.3.1 Habitat characteristics of occupied sites

Univariate Analysis

Univariate analyses indicated that sites occupied by WYRs had a higher abundance of logs, as indicated by a significantly higher proportion of log material in the 30cm x 30cm plot ($t = 3.61$; $p < 0.001$, Table 6.5) and a significantly shorter distance to log ($t = 2.77$; $p < 0.01$).

Table 6.5 Univariate analysis of habitat variables showing average value of each variable in sites occupied and sites unoccupied by WYRs, and the associated t statistic. The variables are sorted from lowest to highest p -value.

	Variable	Unoccupied	Occupied	t	p -value
33	% Log in 30cm x 30cm	0.6%	1.7%	3.61	0.0007
4	% Total Overstorey	38.1%	45.9%	3.16	0.0027
13	Shannon Diversity trees	0.78	0.50	3.04	0.0038
41	Fragment co-efficient	13491	22827	3.00	0.0042
28	% shrubs <i>Gastrolobium</i>	30.7%	52.9%	2.27	0.0078
34	Distance to log (m)	6.08	4.80	2.77	0.0078
40	% Bare ground in 30cm x 30cm	21.6%	16.6%	2.16	0.0352
5	% Wandoo	55.1%	71.1%	2.02	0.0485
30	Aggregation of shrubs	0.36	0.43	1.69	0.0971
38	% Leaf litter in 30cm x 30cm	59.8%	65.7%	1.50	0.1393
31	Average shrub height (cm)	78.41	70.59	1.30	0.1984
29	Shannon Diversity shrubs	1.23	1.10	1.16	0.2535
23	Aggregation of canopies	0.77	0.66	1.03	0.3064
16	Subcanopy trees/hectare	103	116	0.99	0.3258
21	Aggregation of saplings	0.78	0.70	0.99	0.3273
17	Canopy trees/hectare	110	120	0.81	0.4213
22	Aggregation of subcanopies	0.77	0.70	0.74	0.4625
35	% Plant in 30cm x 30cm	18.0%	15.9%	0.71	0.4781
27	% Dead shrub in 10m x 10m	3.2%	3.0%	0.36	0.7231
15	Sapling trees/hectare	67	66	0.18	0.8545
37	% Grasses in 10m x 10m	8.1%	8.7%	0.10	0.9208
25	Shrubs per hectare	2096	2025	0.09	0.9305

Total overstorey cover was significantly higher in occupied sites ($t = 3.16$; $p < 0.005$). The proportion of bare ground at occupied sites was significantly lower than unoccupied sites ($t = 2.16$; $p < 0.05$), indirectly reflecting a higher leaf litter proportion beneath a denser canopy cover provided at occupied sites. The proportion of Wandoo trees and *Gastrolobium* shrubs was significantly higher at occupied sites ($t = 2.02$; $p <$

0.05 and $t = 2.77$; $p < 0.01$ respectively), with a significantly lower diversity of trees at occupied sites ($t = 3.04$; $p < 0.005$). The fragment co-efficient variable was significantly higher in the occupied sites than the unoccupied sites ($t = 3.00$; $p < 0.005$).

With Bonferroni correction, only the proportion of log material in the 30cm x 30cm plot is significant at the $\alpha = 0.0023$ ($0.05 / k$; where k represents the 22 variables) level of significance.

Multivariate analysis

The first three extracted components of the final PCA (eigenvalues > 1.00) accounted for 61% of the variation in the eleven habitat variable matrix (Table 6.6).

Table 6.6 Principal Component Analysis of WYR site occupancy showing the factor loadings of eleven habitat variables for each of the three principal components (only those factor loadings over $|0.3|$ were included in the final table).

	<i>Principal Component</i>	<i>I</i>	<i>II</i>	<i>III</i>
	Eigenvalue	2.86	2.06	1.80
	% Variation	26.0	18.7	16.4
	Cum. % Variation	26.0	44.7	61.0
4	% Total overstorey	0.454		
38	% Leaf litter in 30cm x 30cm	0.453		
34	Distance to log (m)		-0.370	
17	Canopy trees/hectare	0.403		
21	Aggregation of saplings			-0.327
5	% Wandoo		0.505	0.412
30	Aggregation of shrubs		0.343	-0.403
31	Average shrub height (cm)		0.320	-0.494
28	% shrubs <i>Gastrolobium</i>	0.425		
13	Shannon Diversity trees		-0.421	-0.430
41	Fragment co-efficient	0.370		

The first principal component explained 26.0% of the variation and correlated best with % total overstorey, % leaf litter in 30cm x 30cm, canopy trees/hectare, % shrubs *Gastrolobium*, and to a lesser extent, the fragment co-efficient variable. The first three of these variables reflect occupancy of sites with higher tree density, which reflects the high canopy incidence and the indirectly associated higher proportion of leaf litter in 30cm x 30cm plot. The first principal component however, is complicated in that both the proportion of *Gastrolobium* and fragment co-efficient is included in this component, indicating the importance of all these variables rather than a collection of unified variables.

The second principal component explained 18.7% of the variation and was negatively correlated with distance to log indicating occupation of sites with high log abundance. Other correlates indicated that occupied sites were characterized by high proportion of Wandoo trees and an associated low diversity of trees. Both aggregation and average height of shrubs were also important correlates of site occupancy on the second principal component. The second principal component is also represented by a collection of variables.

The third principal component explained only 16.4% of the variation and correlated with four of the five same variables as the second principal component, although also correlated with the aggregation of saplings.

Figure 6.6 presents a diagrammatical representation of the principal component scores of PCI and PCII. Analysis of the principal component scores indicates that PCI successfully delineates occupied from unoccupied sites ($U = 127.00$; $p < 0.0005$), whereas PCII ($t = 1.85$; $p > 0.05$) and PCIII ($t = 1.74$; $p > 0.05$) do not. This result indicates that only those factors correlating with PCI (contributing 26.0% of the variation) are important in delineating occupied from unoccupied sites (Figure 6.6).

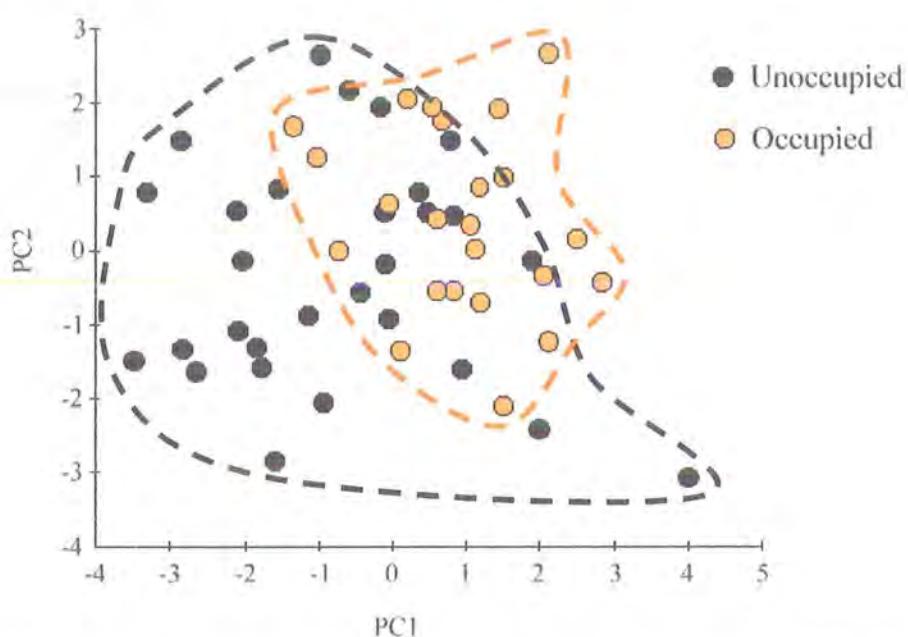


Figure 6.6 PCA ordination showing the distribution of occupied and unoccupied sites according to PCI and PCII. Note grouping of occupied sites toward the top-right of the ordination.

Habitat Complexity Analysis

The average habitat complexity score of occupied sites ($HCS = 16.43$) was significantly higher than unoccupied sites ($HCS = 14.10$, $t = 3.47$; $p < 0.005$) indicating occupancy of sites with significantly higher complexity of vegetation structure (Figure 6.7).

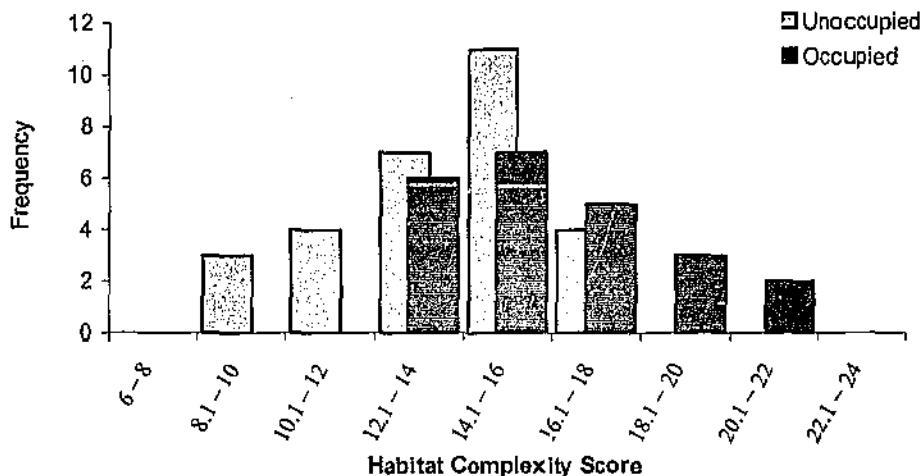


Figure 6.7 Frequency histogram of the habitat complexity scores of unoccupied and occupied sites. The distribution of occupied sites show higher scores than the distribution of unoccupied sites.

Of the five habitat complexity components making up the final site habitat complexity score, the log component showed a higher score at occupied sites (3.65) compared to unoccupied sites (2.72) at the $\alpha = 0.05$ level of significance ($t = 2.49$; $p < 0.05$). The canopy component however, was the only component showing significantly different scores between occupied sites (3.96) and unoccupied sites (3.14) at the Bonferroni corrected significance level of 0.01 ($t = 3.26$; $p < 0.005$), indicating occupation of sites with a higher canopy cover.

6.4 DISCUSSION

6.4.1 Characteristics of occupied sites

Selection for sites with abundant invertebrate/prey habitat

High canopy cover incidence, high density of canopy trees, high abundance of leaf litter and high abundance of logs were shown to be important components of sites occupied by WYRs. These variables indicate occupation of sites with abundant invertebrate prey habitat. Selection for habitat with high canopy cover and density of trees is indicative of sites with lower ground surface temperature, potentially favouring

a higher abundance of leaf litter invertebrates compared to those areas with low canopy cover. Concurrently, these components also favour the production of a denser leaf litter layer on which the WYRs forage. A number of studies have shown the importance of woody detritus as invertebrate and small vertebrate habitat (Taylor, 1990; Laven and Mac Nally, 1998; Andrew *et al.*, 2000; Buddle, 2001; Willett, 2001), so selection for sites with a high abundance of logs provides additional habitat for WYR prey.

Selection for highly productive sites

Occupation of sites with high proportions of Wandoo trees and *Gastrolobium* shrubs, indicates selection for sites with a high productivity compared to unoccupied sites. Wandoo trees are typically positioned on mid to lower slopes, usually being associated with low-lying areas and drainage lines possessing more productive downslope soils (Coates, 1993). Selection of sites with higher productivity is indicative of sites with higher invertebrate biomass.

Selection for sites with reduced edge effect

Occupation of sites with a significantly higher fragment coefficient than unoccupied sites, reflects selection of habitat away from edges, perhaps as a result of the edge effects associated with the agricultural/woodland ecotone. This avoidance of edges simultaneously reflects an inherent area-sensitiveness of the WYR, a result previously proposed by Robert Lambeck in his paper on the focal species approach to nature conservation (Lambeck, 1997; Dilworth *et al.*, 2000).

Appendix 6.1 Latitude (south) and longitude (east) positions of all habitat selection sites. Note the exclusion of sites 'HS 11' and 'HS 18' (see text). 'Pr/Ab' refers to the presence (+) or absence (-) of WYRs from each respective site.

Site	Lat	Long	Pr/Ab	Site	Lat	Long	Pr/Ab
Koomal	32° 45' 45"	116° 55' 22"	+	HS 25	32° 50' 03"	117° 00' 59"	+
Tommingley	32° 47' 53"	116° 53' 55"	+	HS 26	32° 47' 52"	117° 04' 13"	-
Nom	32° 48' 23"	116° 53' 57"	+	HS 27	32° 46' 25"	117° 00' 38"	-
Ochre	32° 47' 26"	116° 56' 27"	+	HS 28	32° 44' 09"	117° 00' 45"	-
HS 1	32° 47' 16"	116° 58' 11"	-	HS 29	32° 44' 46"	117° 03' 03"	+
HS 2	32° 47' 14"	116° 57' 21"	+	HS 30	32° 43' 50"	117° 05' 56"	-
HS 3	32° 47' 14"	116° 55' 47"	+	HS 31	32° 45' 41"	117° 05' 37"	-
HS 4	32° 47' 15"	116° 55' 15"	+	HS 32	32° 45' 53"	117° 08' 02"	-
HS 5	32° 47' 25"	116° 54' 35"	+	HS 33	32° 46' 10"	117° 07' 08"	-
HS 6	32° 47' 43"	116° 54' 05"	-	HS 34	32° 45' 06"	116° 56' 13"	-
HS 7	32° 47' 03"	116° 58' 40"	+	HS 35	32° 45' 24"	116° 57' 01"	-
HS 8	32° 47' 21"	116° 59' 10"	-	HS 36	32° 50' 19"	116° 49' 57"	-
HS 9	32° 48' 18"	116° 53' 08"	-	HS 37	32° 46' 18"	116° 58' 38"	-
HS 10	32° 46' 48"	116° 55' 26"	+	HS 38	32° 53' 38"	116° 59' 00"	-
HS 12	32° 45' 36"	116° 54' 50"	+	HS 39	32° 53' 26"	116° 57' 12"	+
HS 13	32° 45' 04"	116° 54' 32"	-	HS 40	32° 53' 17"	116° 56' 20"	-
HS 14	32° 46' 07"	116° 56' 37"	-	HS 41	32° 53' 41"	116° 55' 35"	-
HS 15	32° 44' 09"	116° 57' 00"	-	HS 42	32° 53' 02"	116° 53' 57"	-
HS 16	32° 43' 29"	116° 56' 42"	-	HS 43	32° 51' 52"	116° 55' 57"	-
HS 17	32° 46' 51"	116° 56' 34"	+	HS 44	32° 46' 05"	116° 51' 27"	-
HS 19	32° 48' 42"	116° 55' 35"	+	HS 45	32° 50' 27"	116° 48' 22"	+
HS 20	32° 48' 32"	116° 54' 38"	-	HS 46	32° 45' 47"	116° 59' 51"	-
HS 21	32° 47' 37"	116° 53' 07"	+	HS 47	32° 46' 16"	116° 57' 19"	+
HS 22	32° 47' 13"	116° 52' 21"	+	HS 48	32° 48' 07"	116° 55' 50"	+
HS 23	32° 48' 16"	117° 01' 10"	-	HS 49	32° 49' 07"	116° 55' 33"	+
HS 24	32° 51' 08"	117° 02' 39"	-	HS 50	32° 47' 57"	116° 55' 00"	+

CHAPTER 7: GENERAL NESTING ECOLOGY

7.1 INTRODUCTION

Knowledge of the nesting ecology of the Western Yellow Robin (*Eopsaltria griseogularis*) is lacking, and comprises of a few records from casual observations (e.g., Whittell, 1933) and from the nest-recording scheme (Higgins and Peter, 2002). The WYR forms an open-cup nest from 0.5m to 22m high, in a variety of species of trees and shrubs 1m to 25m in height (Higgins and Peter, 2002). However, no data exists on aspects of nest site selection, or breeding territory sizes; crucial components affecting the reproduction and survival of birds (Steele, 1993).

Factors governing habitat selection operate at numerous spatial and temporal scales, with the availability of suitable nest sites during the breeding season being potentially the most important (Steele, 1993; Huhta *et al.*, 1999). In most birds, selection of a nesting site is undertaken following selection of a suitable territory. This is especially evident in sedentary birds that occupy year-round territories providing suitable foraging and nesting habitat (Luck, 2000). However, selection of a nesting site is often dependent on habitat attributes at numerous spatial scales within the territory (e.g., selection by the Rufous Treecreeper *Climacteris rufa* for hollow bearing trees for nesting within a pool of large trees utilised for foraging, Luck, 2000). Additionally, within a territory, nest sites are often located in close proximity to the highest quality foraging habitat (Rosenberg and McKelvey, 1999).

In addition to spatial selection of nest sites, there is also spatial and temporal variation in territory boundaries, especially between breeding and non-breeding seasons (Yarrow and Root, 1970). Commonly, there is a reduction of the territory size during the breeding season (Yarrow and Root, 1970). Such variations are believed to represent a response to optimise foraging efficiency and minimise predation risk during the breeding season, but could also relate to a higher density of food during this time (Recher *et al.*, 1987b).

7.1.1 Aims

The aim of the present study is to determine the nest site selection of the WYR at two spatial scales, including nest site characteristics at a microhabitat scale (e.g., nest height, nesting tree height), and a macrohabitat scale (characteristics of nest site habitat patch). Additionally, the breeding and non-breeding territory sizes will be presented.

7.2 METHODS

Searches for nests of the WYR were undertaken in the four foraging ecology study sites (see section 2.1.5 for site descriptions) during the breeding seasons of 2000 and 2001 ($n = 19$), with opportunistic recording from habitat selection sites during the breeding season of 2001 ($n = 3$). Nests were located by following birds exhibiting characteristic breeding behaviours, including aggressive interactions and diversion behaviours such as ‘broken-wing’ and ‘leaf-tossing’ displays. Any individual carrying nesting material or food was followed, as it was assumed it could potentially be returning to a nest. An assessment of breeding success was not undertaken, as it was beyond the scope of the project to do so. All measures of heights of nests and habitat surrounding nests were undertaken following the vacancy of the nestling. This was done to reduce the impact of my presence on nesting adults and nestlings, and to reduce the potential of attracting nest predators such as the Grey Currawong (*Strepera versicolor*) and Grey Shrike-thrush (*Colluricinclla harmonica*) (see Kilgo *et al.*, 1996 for comments on the effect of human visitation on nest success).

7.2.1 Nest tree characteristics

Following the vacancy of the nest by the breeding birds, the height of the nest (measured from the base of the tree to the base of the nest cup) and the height and species of the nesting tree was recorded. A comparison of the height of nesting trees to surrounding trees was also undertaken. The average height of the nearest tree in each of four quarters surrounding the nesting tree was recorded (see Appendix 7.1).

Analysis

No analysis was undertaken on nest height and the height and species of the nesting tree. A paired-sample *t*-test was undertaken to compare the average height of the four surrounding trees with nest trees for each of the 22 nests.

7.2.2 Density of trees at nest site locations

The distances to nearest tree in four quarters surrounding the nesting tree were recorded in order to determine the average density of trees of nesting sites. During analysis, for each nest, one of the distances to nearest tree was randomly replaced with a ‘zero’ distance, to take into account the ‘zero’ distance of the nesting tree at the central point. Density of trees were then calculated (with the modified zero distance) according

to the point-quarter method (Cottam and Curtis, 1956). The density surrounding each nest was then compared to the density of trees at 20 random points within the respective site (also calculated with the point-quarter method). A total of 20 random points were chosen in order sample the variability in tree density across the site.

Analysis

Analysis of difference in density of trees at nest sites and random points was undertaken separately for each of the four study sites using a non-parametric two-sample significance test; the Mann-Whitney *U* Test, due to the non-normal distribution of the scores and heterogeneity of variances.

7.2.3 Characteristics of nest site locations

The characteristics of the nest site location were examined in a 10m x 10m quadrate surrounding the nest tree. The percent cover of live shrubs, dead shrubs, herbs and grasses was calculated for the 10m x 10m quadrate. As categories were not mutually exclusive, the total cover could exceed 100%, as live shrubs and herbs could each 80% of the area. Leaf litter density was also calculated for the 10m x 10m quadrate area, according to one of nine indexes of leaf litter abundance between sparse and dense.

To determine preference for nest sites within study sites with specific habitat characteristics, a comparison was made between nest sites and non-used ‘random point’ sites (see Martin and Roper, 1988; and Sieg and Becker, 1990 for comparable methodology). The percent cover of live shrubs, dead shrubs, herbs and grasses as well as the index of leaf litter density in the 10m x 10m area surrounding nest sites was compared to 20 random point 10m x 10m plots throughout the respective site.

Analysis

Analysis of differences between nest site plots and random plots was undertaken separately for each vegetation variable, for each of the study sites. All measures were square-root transformed and tested for normality using the Shapiro-Wilk test. For those *t*-tests in which the variances were not homogenous, a non-parametric two-sample significance test, the Mann-Whitney *U*-Test, was undertaken. A non-metric multi-dimensional scaling (MDS) ordination was constructed (from a normalised Euclidean distance similarity matrix) of the nest sites and random points within each site. The function of the MDS is to construct a “sample map whose inter-point distances have the same rank order as the corresponding dissimilarities between samples” (Clarke and

Warwick, 1994, pg 5.9). It provides a diagrammatical representation of the similarity of nest sites to random points.

7.2.4 Breeding and non-breeding territory size

During 2000, the non-breeding territory size of the resident WYR was mapped and area calculated at Ochre. During the 2000 breeding season, the breeding territories of resident WYRs were mapped and area calculated at Koomal and Ochre. Territory boundaries were established according to a modified Minimum Convex Polygon (MCP) territory estimate method. The majority of MCPs are produced from radiotelemetry, with territory boundaries shaped from stationary locations without known trajectories (Aebischer *et al.*, 1993). However, in the present study, a polygon (not necessarily convex) was produced by adjoining points representing the furthest distance a WYR was seen, in at least nine measured directions, from a fixed point in the vicinity of the theoretical territory centre. Furthest points were usually distinguished by a unique behaviour exhibited by the WYR, whereby they would sit on a high perch for an extended period of time, before flying a considerable distance back toward the centre of the territory. These points were marked with flagging tape, and, following two to three weeks of similar observations, the furthest of such points were joined to form the territory boundary. Interestingly, the high perches identified as furthest points in one week, were often re-identified as furthest points in subsequent weeks.

The locations of the 2000 breeding season nests were also mapped according to the breeding territories of each site. This was done to determine where within the territories the nests were located.

This section of the chapter is almost entirely descriptive so no statistical analyses were undertaken.

7.3 RESULTS

7.3.1 Nest tree characteristics

The average nest height ($n = 22$) from all sites was $4.18 \pm 0.78\text{m}$ ($\pm \text{S.E.}$). Of the 22 nests, 18 (81.8%) were located in Wandoo (*Eucalyptus wandoo*) trees, with two nests (9.1%) found each in Powderbark (*E. accedens*) and Marri (*E. calophylla*) trees.

A significant positive correlation existed between the height of the nest and height of the nesting tree (Pearson $r = 0.929$, $p < 0.0001$; Figure 7.1). The average

height of all nests was approximately 0.44 ± 0.04 times the height of the nesting trees, corresponding with the basal crown height (the height of the lowest branches).

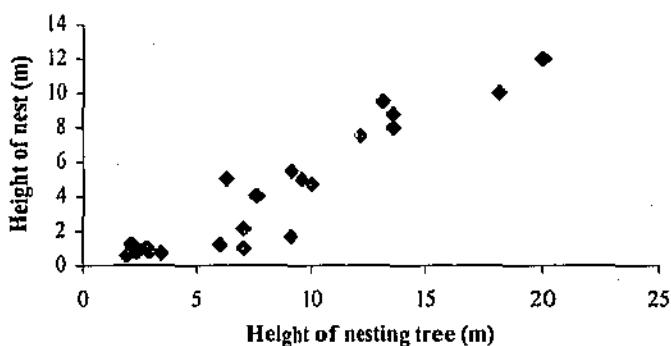


Figure 7.1 Scattergram showing relationship between height of nest and height of nesting tree.

The average height of nesting trees (8.34 ± 1.08 m) was smaller than surrounding trees (9.19 ± 0.56 m), but not significantly so (paired $t = -0.95$, $p > 0.05$; Figure 7.2). However, at only two (9.1%) of the 22 nests was the nesting tree higher than the four surrounding trees, whereas, at eight (36.4%) of the 22 nests, the nesting tree was the smaller than the four surrounding trees (Appendix 7.1).

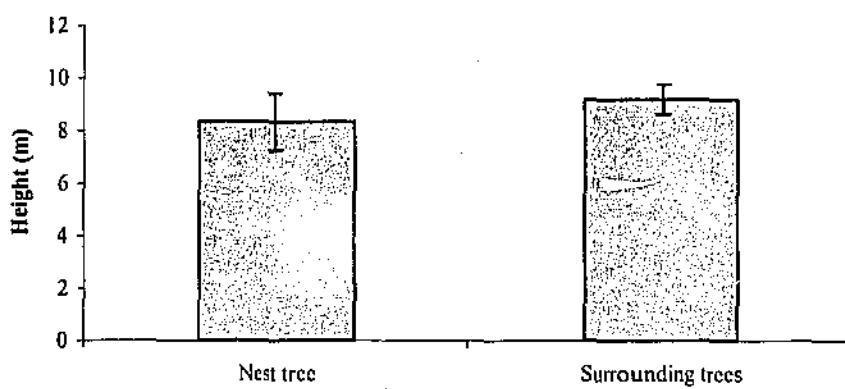


Figure 7.2 Comparison of the height of nesting trees with surrounding trees (mean \pm S.E.).

7.3.2 Density of trees at nest site locations

The density of trees at nest sites was higher than at random points within Tommingley, Norm and Ochre sites, but was only significantly higher at Tommingley ($U = 101$, $p < 0.05$). At Koomal, the density of trees was higher at random points than nest sites, but was not significantly different ($U = 63$, $p > 0.05$).

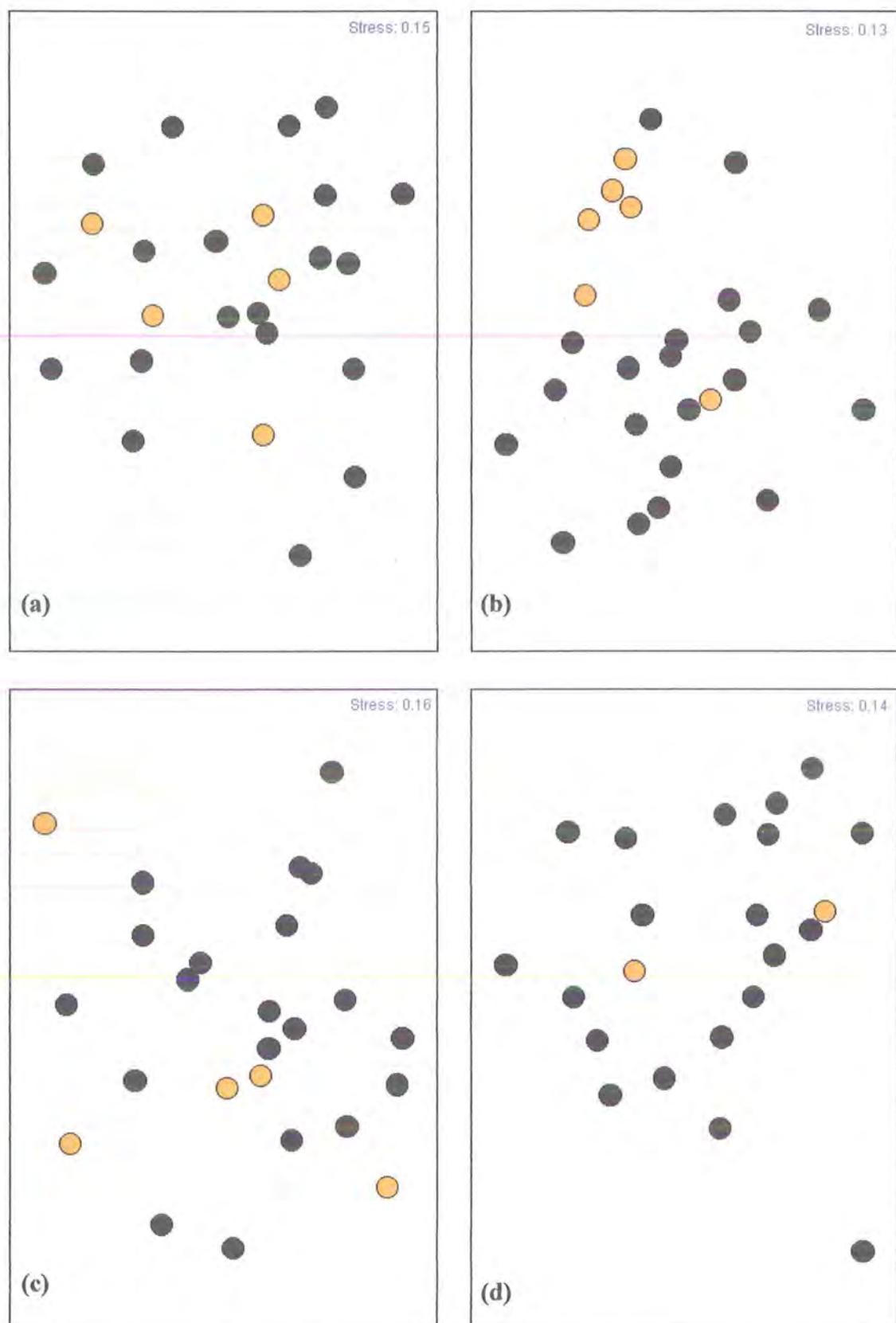


Figure 7.3 Two-dimensional MDS ordinations of the nest sites (orange spots) and random points (black spots) within breeding territories at a) Koomal, b) Tommingley, c) Norn and d) Ochre based on a normalised Euclidean distance similarity matrix of the five vegetation variables.

7.3.4 Breeding and non-breeding territory size

Ochre

The non-breeding territory size of WYRs at Ochre during 2000 was 12.0ha (Figure 7.4) and was longitudinally aligned with Tommingley road to the north. The concave nature of the non-breeding territory towards the north-west of the territory reflects both the influence of Tommingley Road to the north as well as a dense thicket of *Gastrolobium* shrubs towards that region of the site. Although this does not conform to the MCP methodology (thus the modified MCP approach outlined in section 7.2.4), it is believed that this is a more accurate representation of the utilised territory.

The breeding territory size at Ochre during 2000 was 3.13ha, representing 26.1% of the size of the non-breeding territory, indicating contraction of the boundaries during the breeding season. Following the 2000 breeding season, the territory size increased once more, although shifted slightly east, with an approximate overlap of 70 – 80% of the 2000 non-breeding territory (pers. obs.; not shown in Figure 7.4). The two nests of the 2000 breeding season at Ochre are located with the central region of the breeding territory.

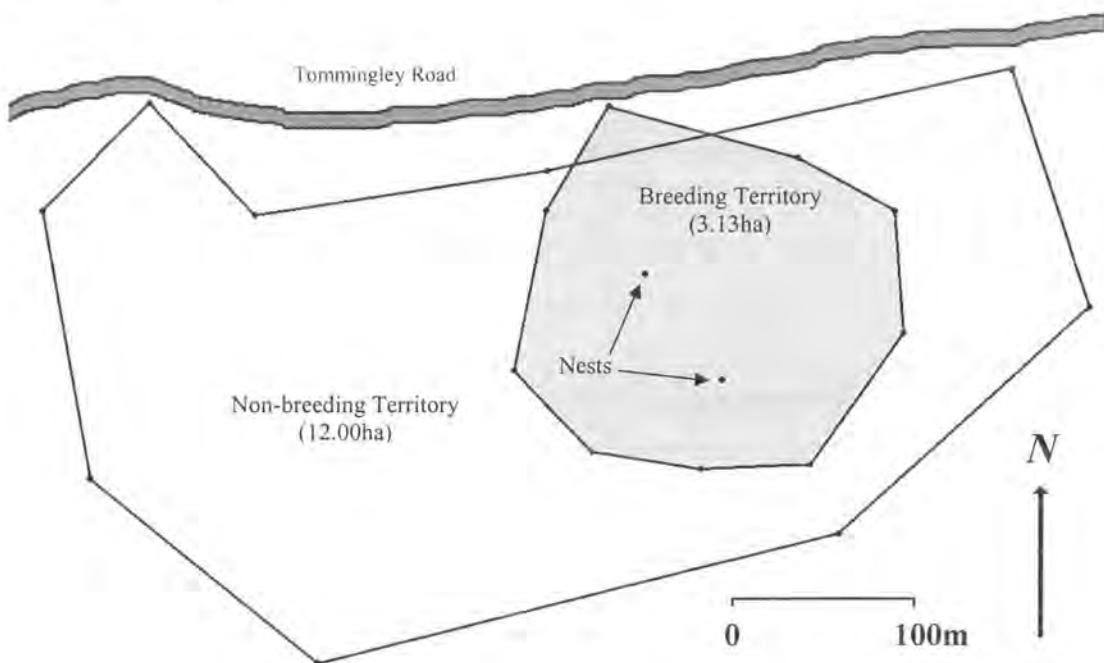


Figure 7.4 Map showing the shape of the breeding and non-breeding territories, and location of nests at Ochre during 2000.

Koomal

The breeding territory size at Koomal during the 2000 breeding season was 2.02ha (Figure 7.5). As with Ochre, the non-breeding territory size at Koomal was also considerably larger than the breeding territory size (pers. obs.). The location of the two nests of the 2000 breeding season correspond with the central region of the 2000 breeding territory.

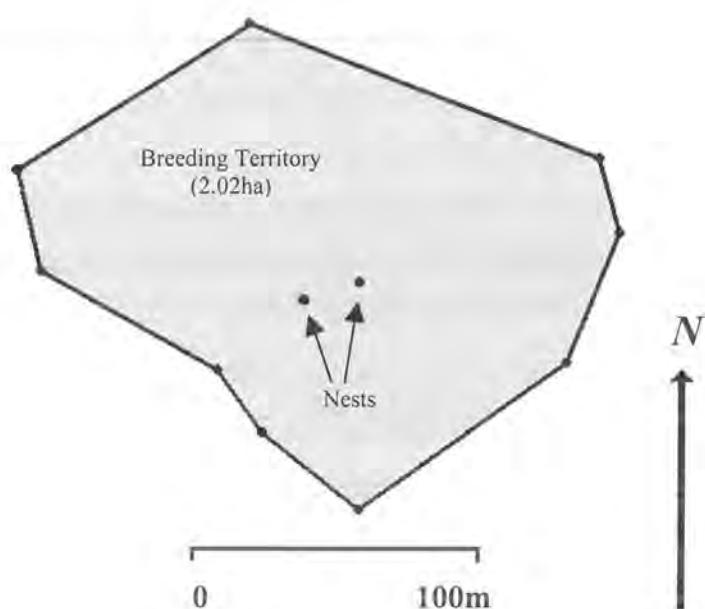


Figure 7.5 Map showing the breeding territory shape and location of nests at Koomal during the 2000 breeding season.

7.4 DISCUSSION

7.4.1 Microhabitat scale selection

The average nesting height ($4.18 \pm 0.80\text{m}$) corresponded with the placement of nests primarily within subcanopy trees, with fewer nests placed in sapling and canopy trees. The placement of nests in the vicinity of the basal crown height provides a relatively unobstructed view of the ground to spot potential predators, as well as affording the protection of an overstorey canopy. However, Marchant (1984) commented that the nests of the Eastern Yellow Robin (*E. australis*), in similar appearance, were often in “absurdly conspicuous positions, thus lending them vulnerable to predation”.

The absence of a difference in the height of nesting trees compared to surrounding trees indicates an absence of selection for nesting tree height. However, the variability of the surrounding tree heights (see Appendix 7.1) may contribute to the absence of significant differences. If the heights of all surrounding trees were examined,

instead of the average height, selection patterns are revealed. For example, the low occurrence of nesting trees of greater height than all surrounding trees (only nests 18 and 21 in Appendix 7.1) indicates selection of nesting trees affording cover from surrounding canopy. This could be in response to a lower conspicuousness of the nest from aerial predators provided by the additional canopy cover of surrounding trees.

7.4.2 Macrohabitat scale selection

None of the vegetation variables were strongly associated with nest site selection across all sites, apart from minor associations at Tommingley and Norn (section 7.3.3). The apparent absence of selectivity of nest sites in the present study is in contrast to most studies. Selection of sites with a higher abundance of leaf litter (Rodewald and Yahner, 2000), patchiness of shrubs (Hoover and Brittingham, 1998) and overall density of vegetation (Kilgo *et al.*, 1996) have been recorded as being associated with nest site selection. These studies suggest selection of nesting sites in areas with high food availability and more pronounced nest concealment. The absence of selectivity of nest sites in the present study may reflect the low number of nests utilised in the analysis. Alternatively, the absence of any apparent selectivity in nest sites could simply indicate the absence of measured variables actually important to WYRs in selection of nest sites. Nest sites may be selected in areas with higher productivity; a factor not looked at in the present study.

Nests were located toward the centre of breeding territories. This finding supports the central-place foraging theory, which suggests “(the) use of resources decreases with increasing distance from a central place such as a nest site” (Rosenberg and McKelvey, 1999, pg 1029).

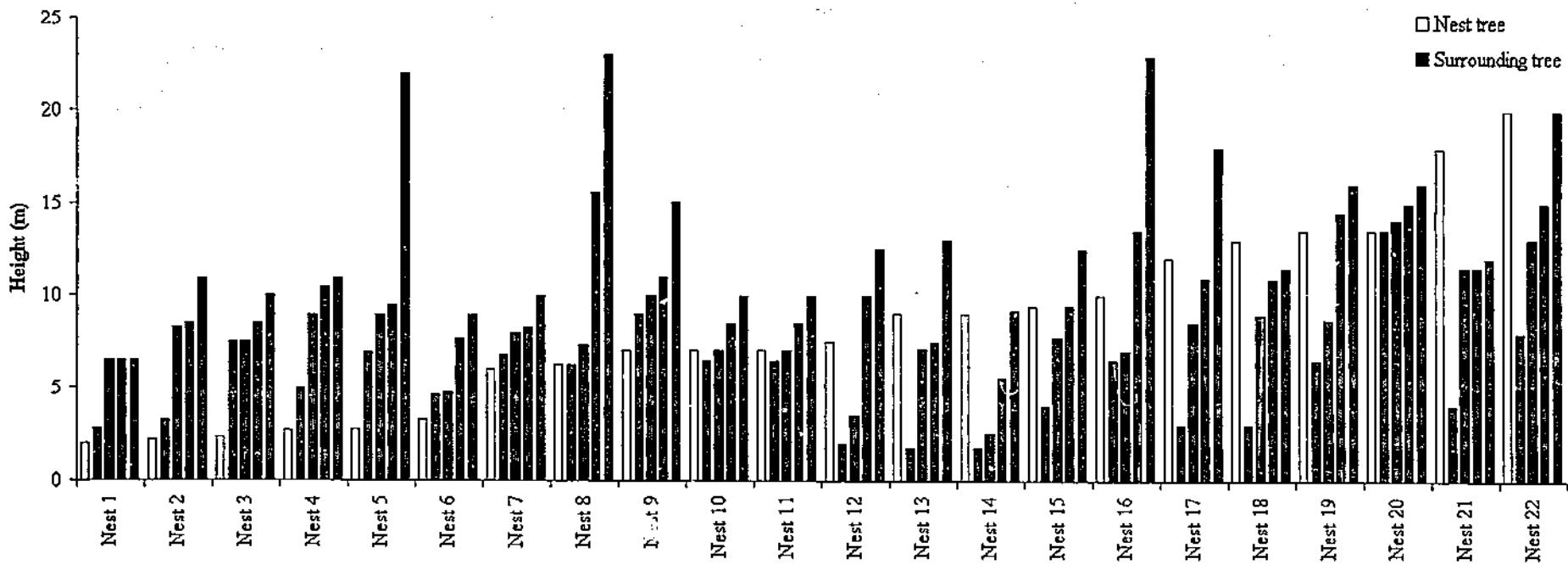
7.4.3 Temporal shift in territories

Following on from the central-place foraging theory (section 7.4.2), the selection of nesting sites is believed to correspond with high-quality foraging habitat (Rosenberg and McKelvey, 1999). During the nesting season, higher quality foraging habitat is required to meet the nutritional demands of the brooding female and then the nestlings. The actual boundaries of breeding territories are determined from the placement of nests that are themselves determined by high-quality habitat surrounding the nesting area, resulting in an apparent hierachal scale of nest, nest patch and breeding territory selection that could all be influenced by food availability.

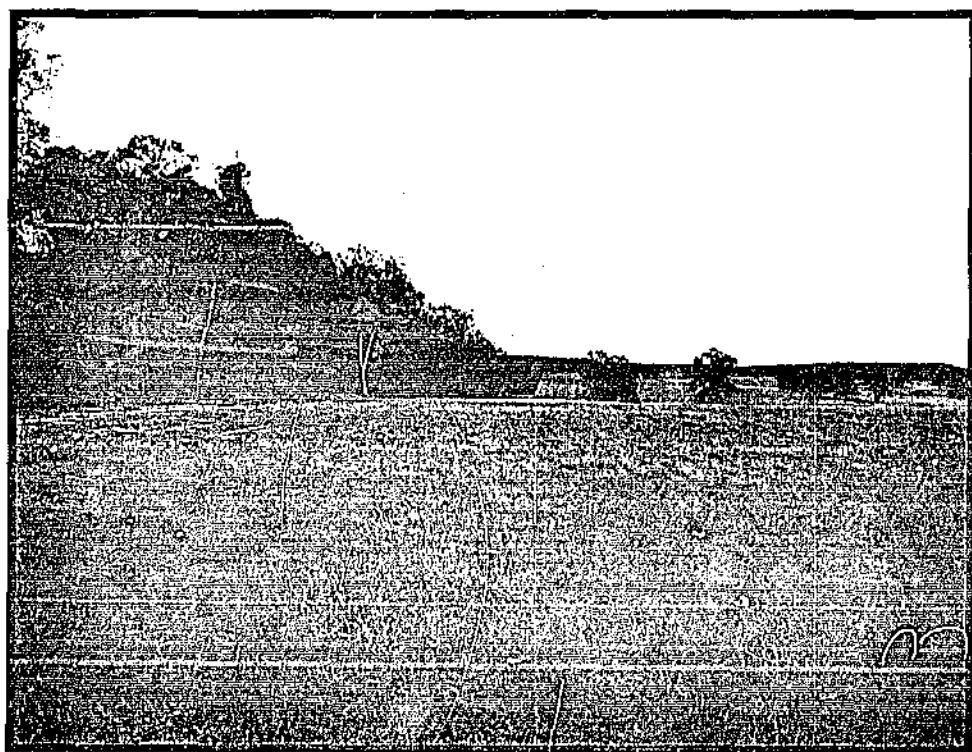
The decrease in area of breeding territories compared to non-breeding territories shown in the present study, has previously been shown in a number of other bird species, and is believed to relate to a need to shorten flight distances to and from the nest to conserve energy during the energetically demanding breeding season (Willson, 1966). Yarrow and Root (1970) found that the foraging area of American Redstarts (*Setophaga ruticilla*) changed according to the stage of the breeding cycle. During the nestling feeding stage, the foraging area was 19% of the size of the foraging area of the “self-maintenance” and fledgling feeding stage. In the present study, the reduced size of the breeding territory of the WYR was consistent throughout the breeding season and not just during any particular stage of the breeding cycle such as the nestling feeding stage.

Contraction of territory boundaries from the non-breeding to the breeding season, could also simply reflect a higher abundance of food available during the breeding season, with WYRs not needing to fly too far from the nest for food. This result emphasises the importance of looking at the ecology of birds during breeding and non-breeding seasons, as management decisions based on requirements of breeding birds, do not encapsulate resource requirements and habitat extent of birds in the non-breeding season.

Appendix 7.1 Height of nest tree (yellow column) and four surrounding trees (green columns) for each of the 22 nests. Note order of nest tree from lowest (nest 1) to highest (nest 22).



PART IV: CONCLUSION



*What have they done to the earth?
What have they done to our fair sister?
Ravaged and plundered,
And ripped her and bit her,
Stuck her with knives in the side of the dawn,
And tied her with fences and dragged her down.*

- Jim Morrison (1943-1971) of The Doors,
"When the Music's Over,"
1967

*How to be green? Many people have asked us this important question.
It's really very simple and requires no expert knowledge or complex
skills. Here's the answer. Consume less. Share more. Enjoy life.*

-Penny Kemp and Derek Wall,
"A Green Manifesto for the 1990s"
1990

CHAPTER 8: SYNTHESIS AND CONCLUSION

The primary research goal of the present study was to examine the foraging ecology and habitat selection of the Western Yellow Robin (*Eopsaltria griseogularis*) at Dryandra Woodland. The foraging ecology component revealed interseasonal and intraseasonal changes in foraging behaviour (Chapter 3). It also identified seasonal changes in utilisation of specific perches (Chapter 4), closely tied to foraging behaviour. An examination of ground-pounce foraging locations revealed selection for microhabitat and macrohabitat attributes (Chapter 5). Habitat selection analysis revealed occupied sites with specific vegetation attributes at numerous spatial scales (Chapter 6). Examination of the nesting ecology of the WYR revealed selection of nest site location at the microhabitat scale, but an absence at the macrohabitat scale (Chapter 7).

8.1 FORAGING ECOLOGY

8.1.1 Temporal changes and links between foraging behaviour and perch use

The foraging ecology component of the study revealed temporal shifts in foraging behaviour as well as spatial selection for specific habitat attributes at numerous micro and macrohabitat scales. All components of foraging ecology from foraging manoeuvre to selection of perch tree size were linked, with the WYR altering all aspects of its foraging ecology to maximize foraging efficiency in its seasonally variable environment.

Throughout the study, ground pouncing accounted for the majority of foraging behaviours. Similarly, dead branches of live Wandoos (*Eucalyptus wandoo*), and dead fallen timber represented the primary perching substrates year-round. However, there were interseasonal and intraseasonal changes in the proportion of all foraging behaviours and perching substrates. The warmer months (Spring and Summer) were characterised by a lower proportion of pounce manoeuvres, lower proportion of ground as a foraging substrate and lower foraging heights in comparison to the cooler months. This was accompanied by utilisation of lower perches (dead fallen timber and dead shrubs), with a lower average perching height in warmer months compared to cooler months.

WYRs favoured the use of subcanopy trees year-round for perching. This result reflects selection for perches at the optimal height, from which there were abundant perches with an uninterrupted view of ground invertebrate prey. However, there were

seasonal changes, with canopy trees utilised more and sapling trees less in warmer months than in cooler months. The heights of perches resulting in ground pounce foraging manoeuvres were lower than random perches.

The seasonal shifts in foraging behaviours and perch use are attributed to seasonal shifts in the abundance of invertebrate prey. During the warmer months, the abundance of leaf litter invertebrates decreases as a result of the desiccation of the leaf litter layer (Moeed and Meads, 1986; Majer and Abbott, 1989; Cale, 1994a; Fox *et al.*, 1997; Burgess *et al.*, 1999), and the abundance of arboreal and flying prey increases (Recher *et al.*, 1983). As a result, the WYRs alter their behaviour accordingly, maximising their foraging efficiency by foraging on a resource in higher supply.

However, this pattern varies spatially and temporally as a result of yearly differences in temperature, humidity and rainfall (see Majer and Koch, 1982; Majer *et al.*, 1990; Recher *et al.*, 1996), explaining the significant intraseasonal difference in foraging behaviour between Spring 2000 and Spring 2001. Higher temperatures and lower rainfall in 2001 compared to 2000, gave rise to a reduced proportion of ground pounces and higher foraging heights. This intraseasonal pattern is in contrast to perch use, which showed an increase in the proportion of low perches (dead fallen timber and dead shrubs) in Winter and Spring 2001 compared to 2000.

8.1.2 Pounce site selection at multiple spatial scales

Characteristics of actual pounce sites showed selectivity, although this was restricted to spatial selection. At the microhabitat scale, pounce sites had significantly more leaf litter and log material, and less bare ground than random points. At the macrohabitat scale, the distance of pounce sites to logs was significantly less than expected. Associations between foraging sites and logs represents selection for sites with a greater abundance of invertebrate prey associated with dead fallen timber (Taylor, 1990; Laven and Mac Nally, 1998; Willett, 2001; Luck, 2002). These results suggest selection by WYRs of pounce sites at a microhabitat and macrohabitat scale, in close proximity to abundant invertebrate prey habitat. This pattern of selection is consistent throughout the year, indicating that although there are temporal shifts in foraging behaviour, characteristics of ground pounce foraging locations remain the same year round.

8.2 HABITAT SELECTION

8.2.1 Site occupancy selection at multiple spatial scales

Sites occupied by WYRs were characterised by variables at multiple spatial scales; namely, selection for sites with abundant invertebrate prey habitat (microhabitat scale), selection for highly productive habitat (macrohabitat scale), and selection for sites with a reduced edge effect (landscape scale).

Sites that were occupied had a high canopy density, and a higher density of leaf litter and logs compared to unoccupied sites. These factors indicated occupation of sites with lower ground surface temperatures, and more leaf litter habitat favouring a higher abundance of leaf litter prey and prey associated with logs compared to more open sites with a low canopy density and low log density. At a macrohabitat scale, occupied sites were characterised as those sites with high proportions of Wandoo trees and *Gastrolobium* plants. These plants are typically associated with soils on lower slopes within Dryandra Woodland, indicating selection of more productive downslope sites. At the landscape scale, WYRs also occupied sites with a high fragment coefficient, indicating occupation of sites away from the woodland/agricultural ecotone.

8.2.2 Nest site selection at specific spatial scales

Nest site selection exhibits selection ranging from the microhabitat scale to the territory size scale. Nest sites were located at a variety of heights, but were normally associated with the basal crown height of trees. This placement affords them relatively unobstructed views of the ground to spot potential predators, as well as providing an overstorey to protect nests from aerial predators.

In conjunction with the occupancy of sites dominated by Wandoo trees, WYRs predominantly nested in subcanopy Wandoo trees. These nesting trees were no different in height, although rarely higher, than all surrounding trees. At a macrohabitat scale, habitat surrounding nest sites was no different to habitat at non-nest sites, indicating no selection for nest site habitat measured. Territory boundaries varied spatially and temporally, with a contraction of territory boundaries during the breeding season to approximately 25% of the non-breeding season size. Within the contracted breeding season territory boundary, nests were located near the centre of the territory.

8.3 IMPLICATIONS OF CURRENT MANAGEMENT AT DRYANDRA WOODLAND TO THE WESTERN YELLOW ROBIN

Management of Dryandra Woodland is administered by a ten-year management plan drafted by the Department of Conservation and Land Management in 1995, on behalf of the National Parks and Nature Conservation Authority and the Lands and Forest Commission (Department of Conservation and Land Management, 1995). The principal objectives of the Management Plan are divided into three sections, according to the existing and proposed land tenures of parts of Dryandra Woodland. At present, Dryandra Woodland is vested in the Lands and Forest Commission for the purpose of multiple uses, with the conservation of flora and fauna the primary use, and mallet timber production and recreation as secondary uses.

The proposed changes to land tenure at Dryandra Woodland, outlined in the Management Plan, would see over 60% (representing 16,337 hectares) of the area converted to National Park (see Figure 8.1). This is proposed to “fulfill as much of the demand for recreation by members of the public as is consistent with the proper maintenance and restoration of the natural environment, the protection of indigenous flora and fauna, and the preservation of any feature of archaeological, historic or scientific interest” (Department of Conservation and Land Management, 1995, p. i).

Almost the entire remaining area (represented by all Mallet plantations) will be managed as State Forest (see Figure 8.1), to achieve the purposes of conservation, recreation and timber production. Only a very small area, represented by the Barron Hills block, will be vested as nature reserve (see Figure 8.1), to “maintain and restore the natural environment, and to protect, care for, and promote the appreciation and study of, indigenous flora and fauna, and to preserve any feature of archaeological, historic or scientific interest” (Department of Conservation and Land Management, 1995, p. i).

8.3.1 Potential impact of increased human pressure

In Dryandra Woodland, WYRs are primarily associated with sites characterised by areas of natural Wandoo woodland, away from the woodland/farmland ecotone. Within occupied sites, WYRs selected sites with a dense canopy cover and dense leaf litter and logs. The proposed conversion of large contiguous areas of natural woodland to national park in Dryandra Woodland has the potential to detrimentally affect the ecology of WYR, by impacts associated with increased human pressure.

Weed invasion

With an increase in the number of visitors to Dryandra Woodland, there is an increased risk of the introduction of weeds. Increasing the density of weeds in natural woodland alters the structural components of the habitat selected for by WYRs. The structure of ground substrates utilised as foraging sites by ground foraging species such as the WYR can be described as a mosaic of bare ground, leaf litter and ground vegetation, with any process altering this mosaic (e.g. an increase in weeds), affecting the foraging ecology of the species (Recher *et al.*, 2002).

***Phytophthora dieback* fungus**

With an increase in the number of cars and people traversing through Dryandra Woodland, there is an increased risk of infection by the *Phytophthora* dieback fungus (Breckwoldt, 1986). Many of the vegetation associations of Dryandra Woodland area are highly susceptible (e.g. moist Jam Wattle sites), or have a high proportion of susceptible species (Department of Conservation and Land Management, 1995). Infection by the *Phytophthora* dieback fungus results in an alteration of the structural complexity of the vegetation, due in part to the loss of crown cover and the death of many understorey plants (Breckwoldt, 1986). As WYRs select for sites with dense canopy cover, the contraction of the *Phytophthora* fungus in Dryandra Woodland has the potential to detrimentally affect the habitat selected for by WYRs.

Vehicle disturbance

With an increase in the number of cars and commercial tour operator buses, there will be an increase in disturbance along existing tracks in the form of noise, rubbish, air pollution and dust. In the habitat selection component of the study, results suggested WYRs avoided the woodland/farmland ecotone. It is possible that alterations to the habitat surrounding highly disturbed tracks may lead to the creation of a woodland/track ecotone; habitat potentially avoided by WYRs. As a result, Dryandra Woodland may become, effectively, internally fragmented into isolated blocks, reducing the amount of undisturbed core habitat available to WYRs.

8.3.2 Impact of fire management

Present fire management in Dryandra Woodland addresses multiple objectives, including protection of life, property, and commercial Brown mallet plantations, and maintenance of habitat and species diversity. The primary strategy revolves around the requirement for fire prevention, involving maintenance of 50 – 100m wide burn buffers

throughout Dryandra Woodland (see Figure 8.2), to assist in containment of wildfires to blocks of about 400 – 600 hectares. Such fire buffers are usually associated with existing roads, with alternative sides of the road burnt each time to reduce the frequency of burning of any particular strip (Department of Conservation and Land Management, 1995).

Fire buffer strips

The use of buffer strips for fire management, primarily along existing tracks, further contributes to the internal fragmentation of Dryandra Woodland suggested in the *disturbance from vehicles* section above, by dividing the woodland into blocks separated by fire buffers up to 100m wide. Furthermore, the fire management plan states it does not practice fuel reduction burning near the perimeter of any of Dryandra Woodland to reduce the potential for weed invasion (Department of Conservation and Land Management, 1995), although the fire management map (see Figure 8.2) shows a number of locations where fire buffer strips align along the perimeter.

Timing and intensity of fire

The degree of impact of fire management is largely influenced by the timing and intensity of the control burn. For example, fire is a primary factor governing the availability of logs. Logs are usually destroyed during intense fires; being replaced by hollow trees that fall as a result of the intense fire. The control burning of buffer strips, undertaken during spring (pers. obs.), are usually of low to mild intensity, leading to the effective destruction or accelerated decomposition of logs. This is accompanied by an absence of resupply from falling trees. In the present study, the ecology of WYRs was intimately tied to the provision of logs, both in terms of foraging ecology and habitat selection aspects. Logs were an integral component of perching substrates, with pounce sites also found in close proximity to logs as a result of the invertebrate prey they provided. At a larger spatial scale, the provision of logs was an important determinant governing site habitat selection. The destruction of logs through buffer strip control fires will see a reduction in the availability of invertebrate prey in these areas, leading to the absence of utilisation of these areas by foraging WYRs. Furthermore, fuel reduction burning during spring may lead to a reduction in the availability of invertebrates during the breeding season, leading to a potential reduction in breeding success of the WYR.

8.3.3 Impact of tourism development (e.g. construction of buildings, roads)

The Access section of the management report (Department of Conservation and Land Management, 1995) states that general access strategies revolve around the maintenance and possible upgrade of the existing road network in Dryandra Woodland. Similarly, the fire management plan strategies exist to maintain a network of roads enabling safe and rapid access for fire control to 400 - 600 hectare blocks. Maintenance in this example refers to the improvement of existing tracks, with minimal construction of new firelines during wildfire suppression (Department of Conservation and Land Management, 1995). The management plan also aims to “avoid the construction of any other tracks for management purposes except when values of a high priority are threatened” (Department of Conservation and Land Management, 1995, p. 32). However, new roads are still being constructed throughout the large central block of Dryandra Woodland (pers. obs.). For example, around the new walk through enclosure and woodland interpretive centre, one of the existing roads has been redirected, and a new road constructed from the settlement straight to the enclosure. This has occurred despite the presence of numerous main tracks already approaching the area from two directions. Furthermore, due to the high economic value of the interpretative centre and walk through enclosure, a very large area surrounding the infrastructure was control burnt prior to Spring 2002 (pers. obs.). Such burning practices, if undertaken frequently, will result in the creation of a barrier to movement of WYRs between regions of good quality habitat.

The construction of new tracks throughout Dryandra Woodland aids in internal fragmentation of the woodland. Secondly, it also facilitates the introduction of pollution and weeds along new track edges, further reducing the amount of undisturbed core habitat available to WYRs.

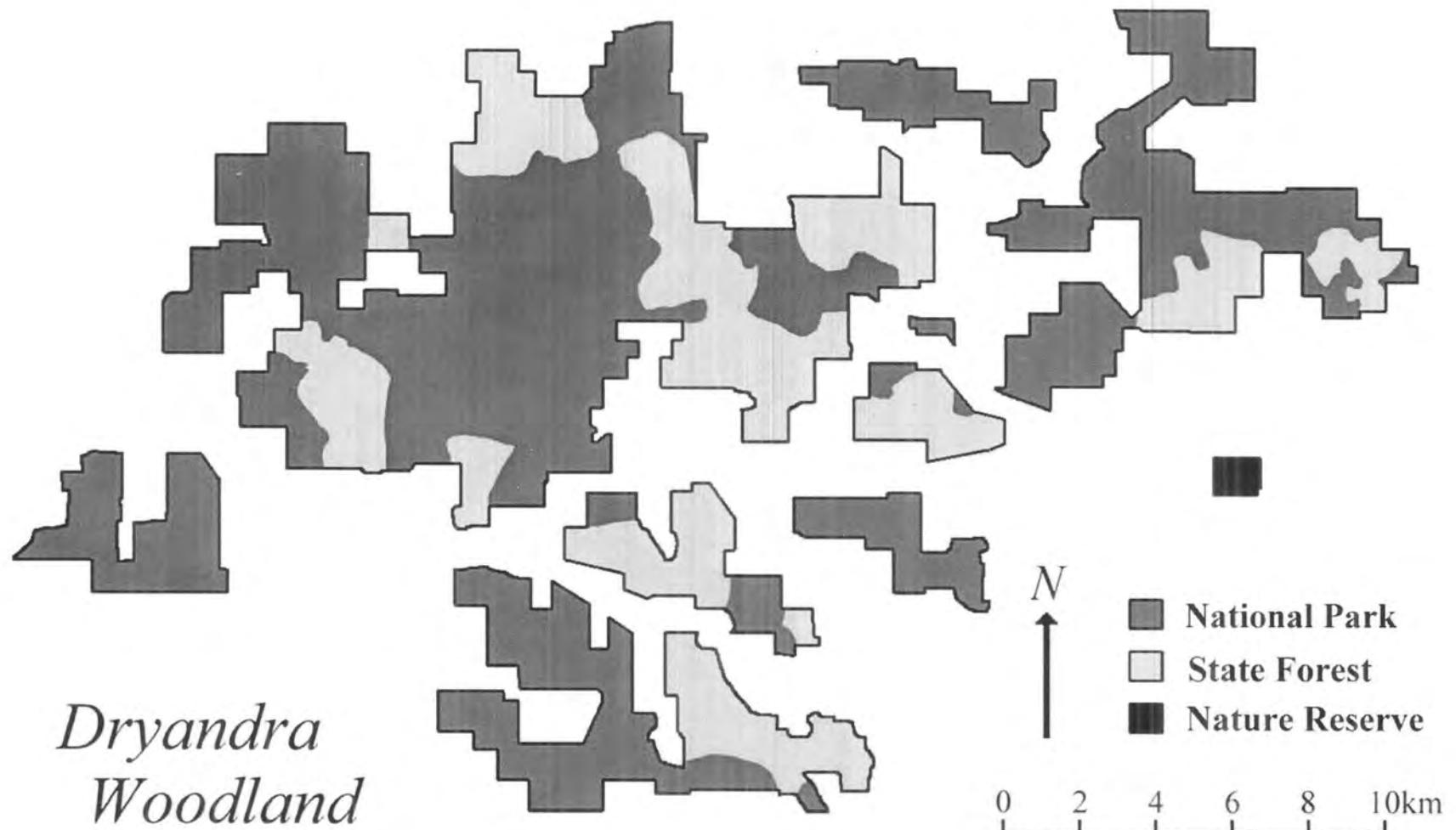


Figure 8.1 Map of Dryandra Woodland showing the proposed land tenure (modified from Department of Conservation and Land Management, 1995)

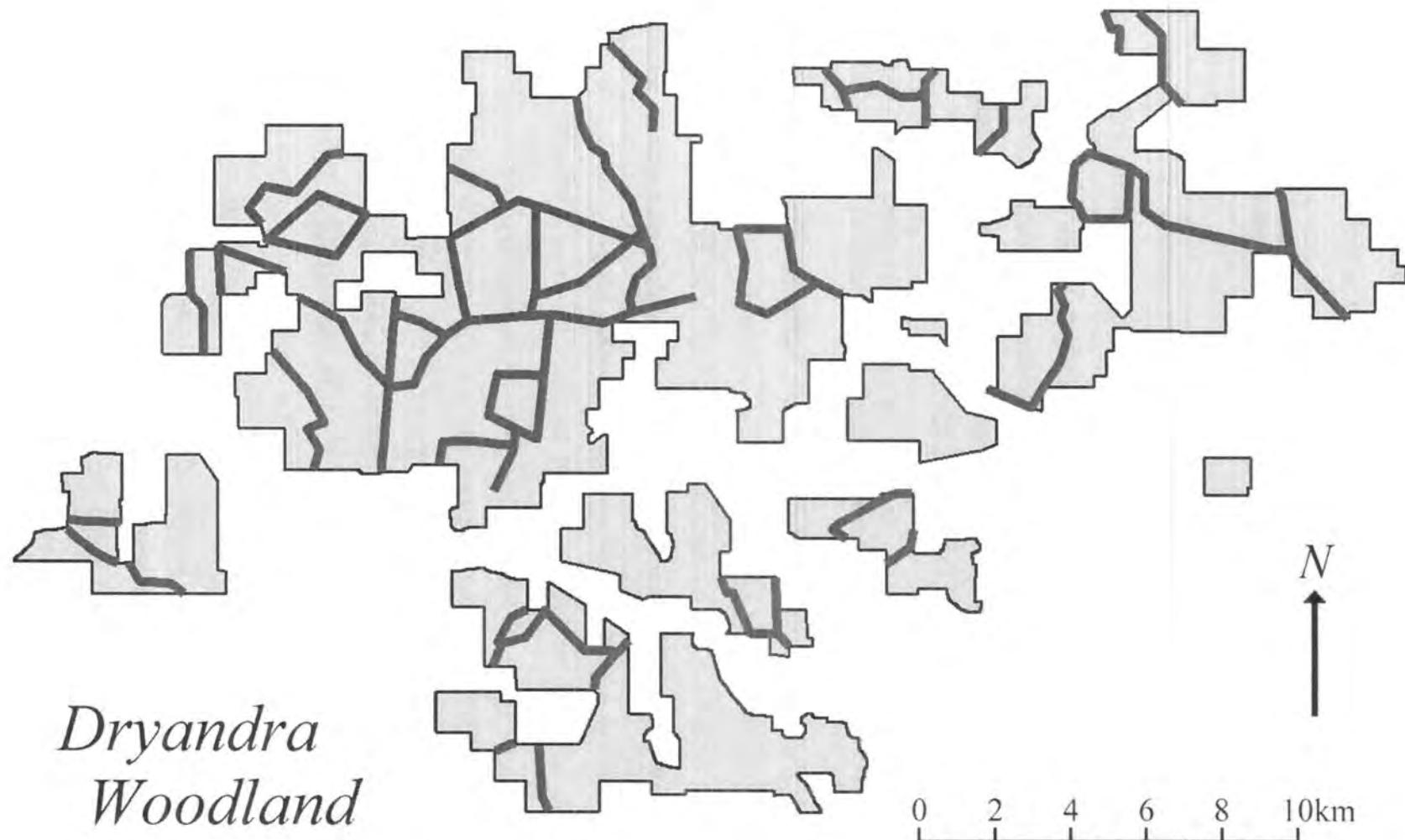


Figure 8.1 Map of Dryandra Woodland showing the proposed fuel reduce buffers (modified from Department of Conservation and Land Management, 1995)

8.4 FUTURE RESEARCH

In the wheatbelt of Western Australia, the WYR continues to decline in distribution and abundance, especially in open woodland, which it is most associated with (Masters and Milhinch, 1974; Saunders and Curry, 1990; Saunders and Ingram, 1995). The primary research goal of the thesis was to improve the ecological knowledge of the WYR, in order to understand reasons for its observed decline in distribution and abundance.

To halt the decline of the WYR requires management decisions at a number of spatial scales, including local scales (e.g. management within Dryandra Woodland) and at landscape and regional scales (e.g. management throughout the wheatbelt). The present study examined the ecology of the WYR within Dryandra Woodland, so implications of current management for that location could be outlined (section 8.3). However, knowledge generated in the present study can also be applied to guiding management and future research in the wheatbelt.

8.4.1 Landscape scale research

Results from the present study can be used to test the effect of the processes operating in the fragmented agricultural landscape on the ecological requirements of foraging resources and habitat attributes identified within Dryandra Woodland. If it is assumed that the foraging resource requirements and habitat attributes selected in Dryandra Woodland represent idealized factors selected for, then any deviation from this condition could represent an unfavourable environment for the WYR. For example, the ground substrate composition is a very important component of the foraging ecology and habitat selection of the WYR, so any process that altering this in the agricultural landscape this has the potential to alter the ecology of the WYR in that landscape.

However, the persistence of the WYR in a fragmented agricultural landscape is by no means restricted to processes influencing the ecology within any one remnant, but is also dependent on factors operating at a landscape scale. In particular, the degree of fragmentation and isolation of remaining remnants of native vegetation need to be taken into consideration when testing the factors important in the persistence of the WYR in a fragmented landscape such as the wheatbelt. Other factors potentially influencing the ecology of the WYR, pertinent in an agricultural landscape such as the wheatbelt include:

- extent of habitat loss

- the area of remaining remnants,
- grazing by domestic and introduced animals
- weed invasion
- altered ecosystem processes
 - hydrological cycles
 - nutrient cycling processes
 - remnant microclimate
- inappropriate fire regimes

The importance of the above factors is dependent on their relative influence on all aspects of the ecology of the WYR. Research on the relative influence of each of these factors will allow management of remnants for the protection of the ecological requirements of the species. Figure 8.3 represents the various factors associated with the fragmented agricultural landscape, and the degree to which I perceive them as potentially influencing each other and eventually the ecology of the WYR.

8.4.2 Area-sensitivity research

A good example of the need for a landscape scale approach is exemplified by the finding, in 1997, of the inherent area-sensitive and isolation sensitive nature of the WYR. Dr Robert Lambeck identified the WYR as a ‘focal’ species for remnant patch size and remnant isolation, indicating the area-sensitive and dispersal limited nature of the WYR (Lambeck, 1997). The approach involved identifying the species most sensitive to certain threatening processes (such as habitat loss and isolation) and determining the management required to meet the needs of the most sensitive species (Dilworth *et al.*, 2000). For each threat, the species requiring the most comprehensive approach is identified (Lambeck, 1997). The identification of the WYR as a focal species supported prior results presented in Peter Cale’s PhD study in the Kellerberrin region of the wheatbelt, which indicated the WYR as being restricted to patches in excess of 43ha (Cale, 1994b).

Future research needs to examine what processes lead to the area-sensitive response of the WYR. Small remnant patches have altered habitat characteristics including reduced habitat complexity (Watson, 1999), higher weed incidence (often associated with stock grazing) (Gelfedder and Kirkpatrick, 1998; Yates *et al.*, 2000a; Yates *et al.*, 2000b) as well as altered ecosystem processes such as disrupted

hydrological and nutrient cycles, and altered microclimates, which could detrimentally alter the ecology, and eventually, the persistence of birds such as the WYR in remnant vegetation.

8.4.3 ‘Food-shortage hypothesis’ research

In a study on the area sensitive Eastern Yellow Robin (*Eopsaltria australis*), Zanette *et al.* (2000) found that the area sensitive effect exhibited by the species was primarily due to the shortage of invertebrate prey in small fragments. This shortage in available prey resulted in impaired reproduction in smaller fragments. Similarly, Sekercioglu *et al.* (2002) found a reduced number of insectivorous bird species in small habitat remnants as a result of impoverished ground-invertebrate prey. In the present study, an assessment of the abundance of leaf litter invertebrates was not undertaken. To understand reasons for the decline of the WYR in remnants of the wheatbelt may require research into the effect of remnant size on prey abundance.

8.4.4 In the meantime...

Apart from long-term research into the potential effect of landuse practices in the fragmented agricultural landscape on the ecology of the WYR, there are several recommendations that should be adopted immediately. There should be a stop to all clearing of woodland vegetation throughout the wheatbelt, in order to ease the pressure of the continued fragmentation and isolation of native vegetation. Furthermore, retention of all large remnants of native vegetation will provide security for all species, such as the WYR, which are potentially area sensitive. Ultimately, the above recommendations should be complemented with the reclamation of farmland, especially around existing remnants, to increase the area and decrease the severity of edge effects associated with fragmented native vegetation.

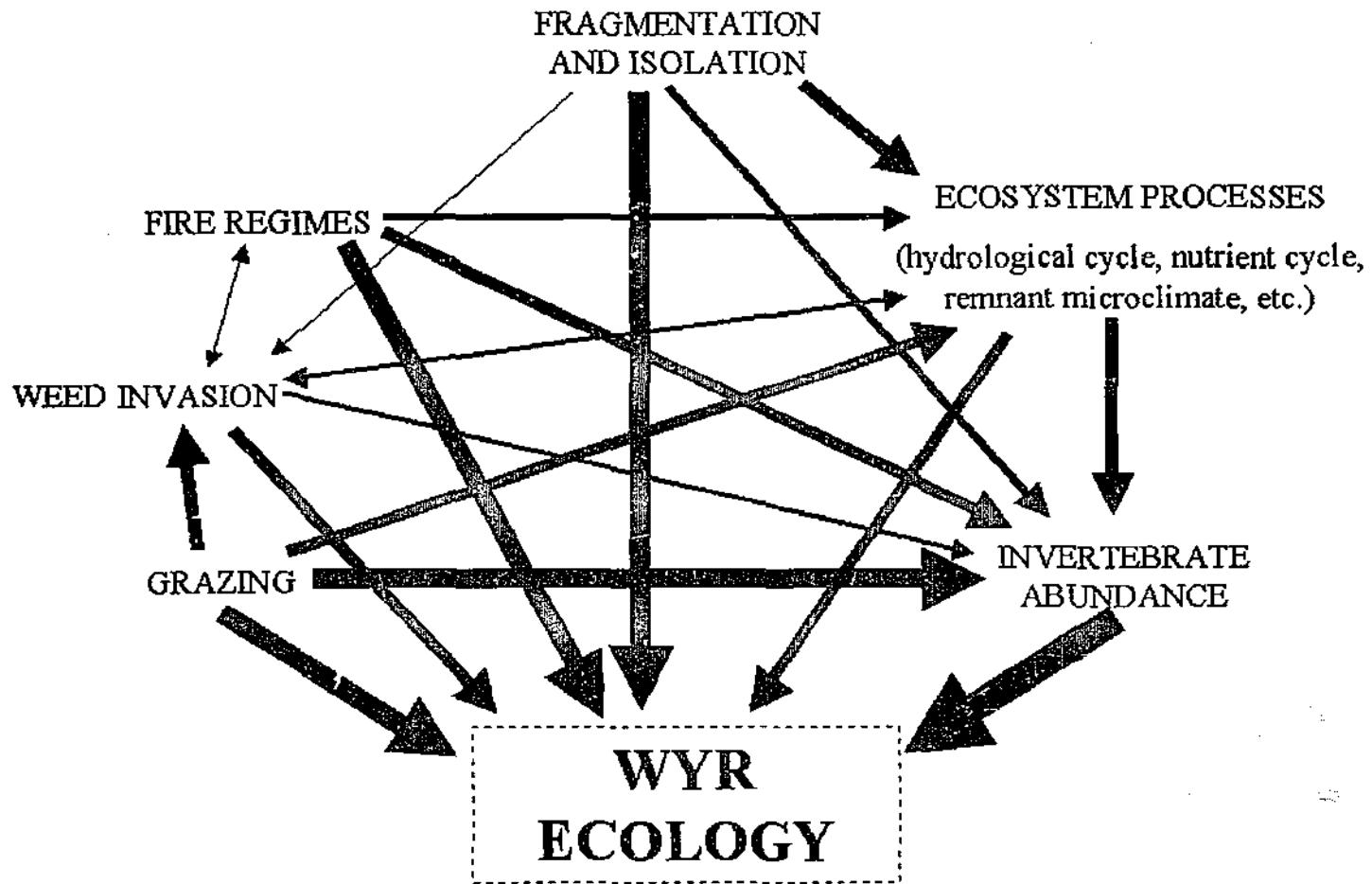


Figure 8.3 Schematic representation of the inter-relationships of factors associated with fragmented agricultural landscapes, and their possible influence on the ecology of WYRs in remnant vegetation of the wheatbelt of Western Australia. The perceived relative importance of factors is graded from most important (boldest black lines) to least important (least bold line).

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