



Defining Habitat Use by Declining Woodland Birds to Inform Restoration Programs

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Do you just look, or do you see?

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Abstract

Declines in woodland bird species across southern Australia is an ongoing concern, despite massive reductions in the rates of native habitat clearance. Species decline is particularly evident within the more isolated regions, such as the Mount Lofty Ranges (MLR) in South Australia. Within the MLR, a suite of woodland bird species are currently in decline, while a number of more common species are also now showing signs of being in trouble. It has been predicted that within the MLR, around 35 to 50 woodland bird species will eventually suffer local extinction under a status quo scenario. Proactive efforts aimed at addressing species decline and looming species loss need to be implemented. One such venture is the Para Woodland Reserve revegetation scheme. This 321 ha Reserve located within the north-central zone of the southern MLR, consists of cleared farming land that has been set aside for the re-establishment of woodland habitats through revegetation. The goal for these new woodlands, once established, is to attract and then support individuals of numerous woodland bird species that are currently in decline within the local region.

The first aim of this thesis was to determine if particular woodland types are better than other woodland types in supporting declining woodland bird species, or whether different species of woodland birds associated with different woodland types. If the latter, a mix of different woodland types would need to be reconstructed to maximise the range of bird species that the revegetated Para Woodland Reserve is likely to eventually support. Further, within a certain type of woodland, there is likely to be considerable spatial, structural and/or floristic heterogeneity that in turn may also influence which parts of that woodland will be used by individuals of different bird species. The second aim of this thesis was to document the extent of any heterogeneity within a woodland type and to determine if use by specific bird species was associated with specific habitat features within that woodland. Armed with such knowledge, a deliberate program that is aimed at incorporating specific habitat heterogeneity within the planting regime could be implemented to increase the likely use of new habitats by a suite of declining bird species.

Several novel approaches for surveying birds were employed during this project. In contrast to the popular twenty-minute / two-hectare survey method that is often used to infer broad habitat associations by birds, this project used survey areas that incorporated ten hectares. The strengths of this method are that the habitat associations of bird species

are reported at the scale of the surveyed area, and that the results do not need to be extrapolated beyond the habitat areas being surveyed (i.e. no assumptions are made regarding unsurveyed habitat for birds). Also, most bird surveys are concentrated on periods of peak bird activity (usually mornings) and are limited to times of fine weather conditions only. Instead, for the fine-scale habitat use surveys within individual home ranges, the birds were intensively surveyed for sustained periods between dawn and dusk over multiple days, regardless of the weather conditions. This approach avoided the typical survey biases mentioned above and helped to expose a more holistic view of habitat use by the birds. Much of the information that was gained from using this technique would have otherwise been missed had traditional bird survey methods been employed.

The key findings of this project include, 1) certain bird species responded to one (or two) particular woodland types, seemingly regardless of where within the landscape that woodland was located, 2) a relatively homogeneous woodland at ten hectares contained significant structural habitat heterogeneity, 3) numerous bird species were statistically associated with specific components of habitat structure within ten hectare sites, 4) the minimum home range requirements of birds belonging to several declining woodland species within the MLR were much larger than expected (i.e. 12 – 36 ha), 5) habitat use by birds within individual home ranges varied appreciably, in both time and space, 6) specific habitat attributes associated with either high use habitat areas (i.e. hot-spots) or low use habitat areas (i.e. cold-spots) helped to explain habitat associations by individuals of various bird species, and 7) these same habitat attributes helped to differentiate habitat use between birds of different species that co-existed within a common habitat area.

Recommendations for both the revegetation works on the Para Woodland Reserve and more broadly include, 1) revegetation should provide habitat areas that cover at least tens, if not hundreds, of hectares, 2) plant a variety of woodland types at large spatial scales (e.g. 10 ha each) to cater for a broader suite of locally declining bird species, 3) within each woodland type, incorporate structural (and floristic) heterogeneity at fine spatial scales (e.g. 30 m × 30 m), 4) identify and incorporate the habitat needs of individual bird species at fine spatial scales (e.g. multiple patches covering approximately 30 m × 30 m each), and 5) provide for multiple co-inhabiting bird species by planting a repeated mosaic of different habitat patches at these finer-scales, with each patch “type” catering to the individual habitat needs of one (or a few) pre-determined bird species.

Declaration

I certify that this work contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution in my name and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text. In addition, I certify that no part of this work will, in the future, be used in a submission in my name, for any other degree or diploma in any university or other tertiary institution without the prior approval of The University of Adelaide and where applicable, any partner institution responsible for the joint-award of this degree.

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Phillip John Northeast

01/10/2013

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Chapter 1

Project review: Revegetating the Para Woodland Reserve

History of land clearance across the Adelaide region

South Australia was colonised by European settlers and proclaimed as a new British colony on the 28th of December 1836. The new town of Adelaide (named after Queen Adelaide) was built near the banks of the River Torrens on the fertile floodplain between Gulf St. Vincent and the escarpment of the Mount Lofty Ranges (MLR). The MLR is an extensive north-south trending linear range that separates the Adelaide Plains to the west from the drier mallee plains to the east.

Prior to European settlement, the natural environment of the Adelaide Plains contained a mosaic of complex habitats and defined vegetation communities (Krahenbuehl 1996). These habitats included open forest, woodland, low woodland, both tall and low shrubland / heath, tussock grassland and reedbed sedgeland (Krahenbuehl 1996). An estimated 1,130 native vascular plant species existed over these Plains at the time of European settlement (Tait *et al.* 2005a). Plant species included those of the genus *Acacia*, *Allocasuarina*, *Banksia*, *Callitris*, *Eucalyptus*, *Melaleuca* and *Olearia*, and grasses dominated by *Danthonia* and *Stipa* (Krahenbuehl 1996). The species-rich understorey included numerous taxonomic Families, including Compositae, Gramineae, Leguminosae, Myrtaceae, Orchidaceae and Proteaceae (Facelli 2001 in Tait *et al.* 2005b). These floristically rich habitats once supported an estimated 290 bird species (including migratory and nomadic species), 56 reptile species, 40 mammal species, 11 freshwater fish species, seven amphibian species and tens of thousands of invertebrate species (Tait *et al.* 2005b).

Much of the natural habitats on the Adelaide Plains were quickly erased however by the early European settlers. The country was opened up for construction and farming and the felled timber provided for building materials and firewood (Krahenbuehl 2005). The reasons for some habitat clearances were less conventional though, such as the felling of a

Grey Box (*Eucalyptus microcarpa*) forest near the new township, for fear it may provide refuge for possible invading Russian mariners (Kraehenbuehl 2005). Land clearance escalated throughout this region during the 1870's, as settlements progressively spread outwards from Adelaide (Jones 2005). By the year 1900 only a few vestiges of remnant vegetation remained across these Plains (Kraehenbuehl 2005).

Broad-acre habitat clearance for farming was not conducted randomly however. Common of many land clearance practices across southern Australia, land prized for its agricultural productivity was selectively cleared of its natural vegetation (Recher and Lim 1990; Hobbs *et al.* 1993a; Robinson and Traill 1996; Cofinas and Creighton 2001; Mac Nally *et al.* 2009; Thomson *et al.* 2009). On the Adelaide Plains, the more open woodlands were the first to be preferentially cleared for farming (Griffin and McCaskill 1986). These habitats were dominated by temperate grasslands and Box grassy woodlands (Davis 2000; Paton *et al.* 2000; Paton *et al.* 2004).

Broad-scale clearance of native vegetation has continued throughout most of the European occupation of the Adelaide region (Griffin and McCaskill 1986). For example, between the years 1945 and 1980, 62% of the remaining 240,000 ha of natural habitat across the southern MLR was cleared (Fig. 1.1). Natural habitats made way for the construction of suburbs and opened new land for cropping and pasture for the ever growing human population. Today, seventy-three percent of the State's 1.65 million people reside within the Adelaide metropolitan area, making it the most urbanised State within Australia (Australian Bureau of Statistics, <http://www.abs.gov.au>).

Today, less than 4% of the pre-European vegetation now remains across the Adelaide Plains (Paton *et al.* 2000). The current level of remaining natural habitat across this region as a whole has also been reduced to around 10% of pre-European extent (Westphal *et al.* 2003a; MLR NRM 2009). Much of this clearance has made way for horticulture and agriculture, which has expanded across the Mount Lofty region. Orchards, wine grapes and hobby farms are now scattered along the 100 or so kilometres between Willunga to the south and the Barossa Valley to the north. Intensive stock grazing also exists along the Fleurieu Peninsula to the south and on the drier plateau to the east. Arable farming and stock grazing are strongly evident on the plains further north.

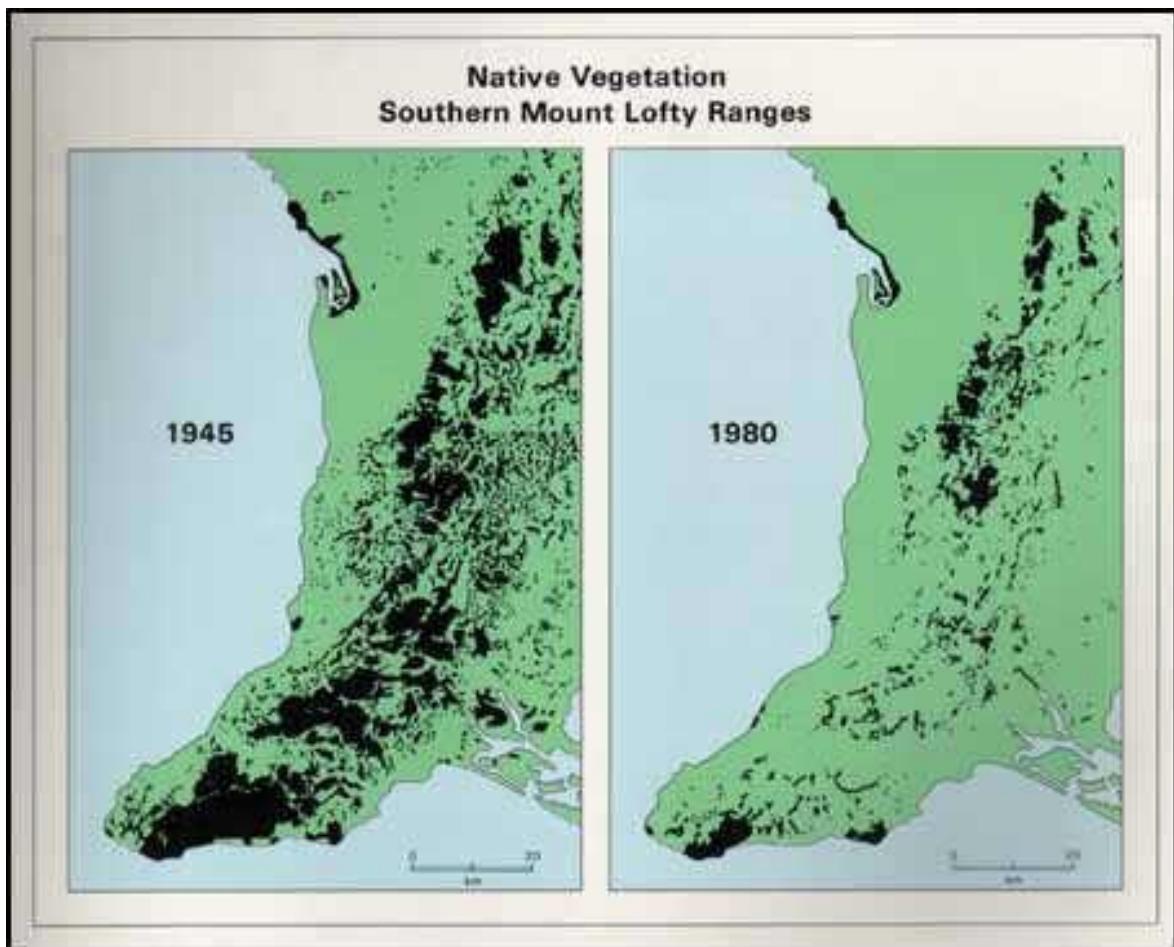


Figure 1.1: Extent of native vegetation clearance across the southern Mount Lofty Ranges, South Australia. Over 60% of the 240,000 ha of native vegetation was cleared between 1945 and 1980. The dark coloured areas indicate remaining native vegetation (Sourced from Griffin and McCaskill 1986).

As a result of disproportionate habitat clearance, much of the remnant vegetation has been left standing over areas of impoverished soils. Messmate Stringybark (*E. obliqua*) and Brown Stringybark (*E. baxteri*) grow over high rocky elevations, interspersed with Cup Gum (*E. cosmophylla*) and Long-leaved Box (*E. goniocalyx*; Croft and Brandle 2003). On the more fertile Adelaide Plains, remnants such as Pink Gum (*E. fasciculosa*) have been left standing over areas with shallow and/or sandy soils (Croft and Brandle 2003).

Remnant vegetation across the MLR is also far from being pristine. For example, more than one third of the plant species recorded across the Adelaide Plain's park and reserve systems are introduced (Tait *et al.* 2005b). Remnant stands also typically have high edge to core ratios, as the native vegetation has been left both fragmented and relictual

(McIntyre and Hobbs 1999; Paton *et al.* 2004). The physical structures of many mature trees within remnant Stringybark forests have also been compromised. Early woodcutters preferentially selected the largest trees to fell. The resultant coppice regrowth from the residual stumps produce thin, multi-stemmed trees which dominate today's forests (Griffin and McCaskill 1986; Tait *et al.* 2005b). Some remnant vegetation is also not considered to be self-sustaining. Mature scattered "paddock trees" often experience ongoing recruitment failure due to high stock grazing pressure, threatening the long-term existence of these plants within the landscape (Manning *et al.* 2006).

The traditional savannah woodlands which once adorned the Adelaide Plains are now restricted due to their broad-scale removal. These include woodlands of South Australian Blue Gum (*E. leucoxylon*), Manna Gum (*E. viminalis*), River Red-gum (*E. camaldulensis*), Peppermint Box (*E. odorata*) and Grey Box (Croft and Brandle 2003). As a direct result of past disproportionate clearance, grassy woodlands dominated by Peppermint Box and Grey Box are classified as Critically Endangered and Endangered respectively, within the MLR (EPBC Act 1999; <http://www.environment.gov.au/epbc/index.html>).

Local and global concerns regarding the effects of native land clearance on the environment prompted the South Australian Government to establish the Department of Environment and Conservation in 1972. This new agency oversaw an enquiry into the extent of native vegetation clearances within the State. The findings of this report were released to the public in 1977, which gained front page exposure in the local printed media (Fig. 1.2). The report prompted calls for urgent action to slow the rate of land clearance within the State. This heralded the introduction of Heritage Agreements in 1980, whereby private landowners voluntarily assumed legal responsibility for maintaining and/or improving native vegetation of high conservation value on their property. However, a study by the Nature Conservation Society of South Australia revealed that the voluntary nature of Heritage Agreements did not adequately halt the rate of native vegetation clearance. In response, the Government introduced statutory controls on land clearance through the Planning Act of 1982. The Native Vegetation Management Act 1985 was then introduced to financially compensate landowners whose land clearance applications were not approved. The Native Vegetation Act in 1991 then superseded the 1985 Act by discontinuing the financial compensation. More importantly, this latter Act also formally brought an end to any future broad-acre clearances within the State.

The Advertiser June 1977

Curb land clearance, SA urged

By Conservation Writer KVM TILBROOK

Slowing SA's rate of land clearance is urgently needed, according to a State Government report issued yesterday.

Leaving the report, the Minister for the Environment (Mr. Simmonds) said that about 73 per cent of the State inside the 250-million-acre agricultural boundary had been cleared for agriculture.

Of the remaining 26 per cent, 5 per cent was in national parks, he said.

It could have been a good idea if the report had been set up 10 or 20 years ago, he said.

The report says uncleared vegetation is a finite and "potentially declining resource" which will continue to be needed for community needs.

It says principles of sound land use and management should be applied.

The report was commissioned by the Cabinet in July, 1976, because of concern at the rate of land clearance.

The study, "Vegetation Clearance in SA," makes 14 recommendations which it hopes will slow clearance.

It discusses financial incentives for keeping natural vegetation and compensation for potential productive loss.

The report suggests a Heritage Agreements Act under which a landowner

could contract with an independent body, typically the Government, to use his land in an agreed way.

The agreement would not be valid without the agreement of both parties and would remain as an encumbrance, covering any subsequent owners.

The report says the Crown Solicitor should be asked to draft a model agreement which would allow natural vegetation to be kept on private land in perpetuity and also ensure that a chosen incentive was not abused.

The 57-page report recommends:

- Immediate moves to decide reasonable and effective ways to encourage landholders to keep native vegetation.

- Attention should first be directed towards varying State and local government rates and taxes.

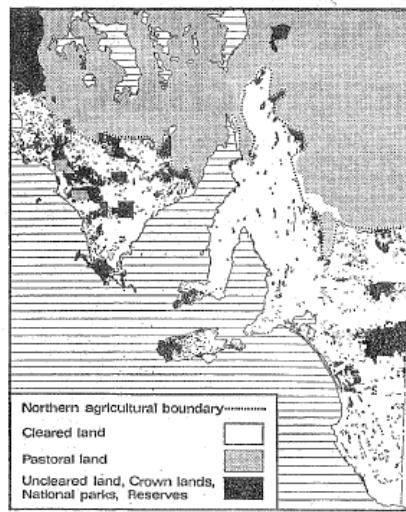
- All possible incentive schemes should be considered and evaluated.

It says many existing State and Commonwealth rates and taxes are a disincentive to retain natural vegetation.

"Some measure of relief from their provisions would probably encourage landowners to retain natural vegetation," the report says.

It suggests the Income Tax Assessment Act could be amended to allow the deduction of land clearing costs only for approved clearance procedures.

● Contd. on Page 8.



Cut SA land clearing – report

● Contd. from Page 1.

An amendment could also allow deductions for the cost of maintaining uncleared land.

The report says pressure from land tax to reduce land tax on cultivable and some relief appears needed.

"At present, land used for primary production receives a statutory exemption of \$46,000, but land held for conservation purposes receives no relief," it says.

The report says local government rates vary from district to district but are highest near urban areas.

This does not benefit the landowner who wishes to remain on the land and who wants to undertake subdivision and/or clearance, it says.

"Local government rates are generally higher than land tax rates, although exemptions from them may be a more decisive incentive for the retention of vegetation."

Compensation for potential productive losses may be extremely expensive and difficult to administer, the report says.

The committee proposes that in the interim regulations for land clearance should be scrutinised by the Department for the Environment.

However, it is well aware that if liaison is not established with rural interests, first, this measure could provoke a hostile reaction.

It says it is important when considering introduced species that farmers should not be inconvenienced or delayed by administrative procedures.

It recommends that the issue be handled by a diverse panel of the Department for the Environment, assisted by an advisory committee on vegetation clearance.

The measure would be monitored and, if necessary, reviewed after three years.

The report says the logical long-term answer must be the establishment of a land use authority supported by statute.

But short-term control is necessary.

The committee also found that:

- SA's older settled regions have been almost totally cleared of native vegetation.

- The native vegetation clearance since World War II was caused by the physical environment, agricultural, economic factors, Government policies, and technological developments in land utilisation and management.

- Clearance has dropped in recent years, but a future demand for increased agricultural production may cause clearance of more land in existing agricultural regions.

- The future should require more justification.

Mr. Simmonds said yesterday: "I am looking at the recommendations to see to what extent they should be followed."

Why so much clearance from the public was wanted because action was "urgently necessary".

The report was prepared by officials of the Department for the Environment, the Department of Agriculture and Fisheries, the Lands Department and the Woods and Forests Department.

The committee chairman is Mr. C. Barnes of the Department for the Environment.

Figure 1.2: This article appearing on the front page of The Advertiser (June 1977) called for land clearance across South Australia to be curbed. The article followed the public release of an extensive enquiry by the South Australian Government into the extent of native vegetation clearance within the State.

Local extinctions and species decline

An estimated 128 native plant and animal species native to the Adelaide region at the time of European settlement have become locally extinct (Tait *et al.* 2005a). In the study of Tait *et al.* (2005a), between the years 1836 and 2002 the Adelaide region lost 85 (7.5%) native plant species, 20 (50%) mammal species, 21 (7.0%) bird species and two (3.6%) reptile species. The physical removal of large habitat areas and anthropogenic changes to the

landscape are obvious drivers of species loss. Another significant factor is that the woodlands and open forests of the MLR are geographically isolated from similar woodland systems in eastern Australia, being separated by arid mallee and chenopod habitats (Paton *et al.* 2004). This renders the remnant habitats of the MLR a virtual island for many of its inhabiting species (Szabo *et al.* 2011). With many of these species physically isolated from larger populations in eastern Australia, biotic losses within the MLR are not readily replenished from source populations located elsewhere (e.g. Schodde and Mason 1999). Evidence for this suspected genetic isolation is apparent with several MLR bird species having developed morphological characteristic that are distinct from their eastern State counterparts (Schodde and Mason 1999).

The great majority of the bird species within the MLR also exist on the western extremity of their national range (see Schodde and Mason 1999). As the decline of a species is often first evident in the periphery of its range, further declines and species loss within the MLR are to be expected (Paton *et al.* 2004). Based on the amount of natural habitat now standing within the MLR, Ford and Howe (1980) forecasted that between 35 and 50 woodland bird species would eventually suffer local extinction within this region. In support, Paton *et al.* (2004) indicated that 10 of these bird species are now considered to be regionally extinct within the MLR. In addition, a further 50 or so bird species are considered to be in decline, either in their distribution and abundance, or just their abundance. The paper of Paton *et al.* (2004) also highlighted that these declining species are spread over a range of foraging guilds and across a number of different habitat types. This suggests any attempts to address species decline within the MLR must take on a multi-species approach with a regional perspective.

Latter studies have also confirmed the declining status of numerous MLR bird species. Tait and Daniels (2005b) reported a 13% reduction in Adelaide's woodland bird species between the years 1836 and 2002, which has escalated sharply since the 1950's. Possingham *et al.* (2006) considered 19 rare bird species occupying Stringybark habitats within the MLR are of local conservation concern, but are not necessarily of national conservation significance. A more recent study by Szabo *et al.* (2011) supported the earlier prediction by Paton *et al.* (2004), that species decline will be proportionally greater in the disproportionately cleared Gum woodland habitats. Disturbingly, Szabo *et al.* (2011) also indicated that several bird species that are currently regarded as being common

within these habitats are now also starting to show signs of decline. These species belong to a range of foraging guilds and include the Black-faced Cuckoo-shrike (*Coracina novaehollandiae*), Grey Fantail (*Rhipidura fuliginosa*), Mistletoebird (*Dicaeum hirundinaceum*), Musk Lorikeet (*Glossopsitta concinna*), New Holland Honeyeater (*Phylidonyris novaehollandiae*), Red Wattlebird (*Anthochaera carunculata*), Striated Thornbill (*Acanthiza lineata*) and Superb Fairy-wren (*Malurus cyaneus*).

Clearance of native vegetation of conservation significance within the MLR has effectively ceased since 1982. Despite this, declines in woodland bird species within this region is ongoing and appears to be escalating. Indeed, the MLR has been declared a National Biodiversity Hotspot, in part due to the imminent threat of local extinction many of the species inhabiting this region now face (Long 1999). The current rate of species decline within the MLR suggests that the extinction debt as proposed by Tilman (1994) from past land clearance is far from being paid in full (Paton *et al.* 2004; Szabo *et al.* 2011). The plight of these bird species also echoes similar concerns for birds within other Australian woodland systems (e.g. Robinson 1991; Ford *et al.* 1995; Saunders and Ingram 1995; and many others). The parlous state of the woodland bird community within the MLR prompted Szabo *et al.* (2011) to describe this region as a “canary landscape” for woodland systems elsewhere. Predictions of multiple species loss across the MLR are starting to become a reality, which should fire a warning shot for those concerned over the plight of birds that are reliant on temperate woodland systems across southern Australian.

Revegetation and the Para Woodland Reserve

Time lags between initial habitat clearance and eventual faunal species losses provide a window of opportunity to redress current species decline (Paton *et al.* 2004). The delay in eventual species loss may be due to the relatively long lived nature of many Australian birds (Ford *et al.* 2001). This suggests however that small bird populations within fragmented remnants may not be viable and will be lost without adequate human intervention (Tait and Daniels 2005).

A logical response in addressing ongoing species decline is to return native habitat through revegetation (Saunders *et al.* 1993; Bennett *et al.* 2000; Hobbs and Harris 2001; Vesk and

Mac Nally 2006). To date though, revegetation across the MLR typically does not replace the habitats that once graced the areas with the more fertile soils and higher rainfall (Paton *et al.* 2004). Revegetating these areas that were disproportionately cleared remains a priority to help stabilise and restore populations of declining bird species (Thomson *et al.* 2009).

Opportunities to revegetate large parcels of good quality land fortunately do exist. In 2003, Elizabeth Law-Smith on behalf of her late husband David bequeathed a 118 ha block of land within a larger holding that had previously been managed for stock grazing and cropping (Bickerton 2005). This property (the Yaringa Block) is situated 3 km south-east of the rural township of Gawler and some 40 km north of Adelaide, South Australia (Fig. 1.3). This land contains areas of native vegetation of some conservation value, however it is typically an agricultural landscape (Bickerton 2005). The land was donated to the Nature Foundation of South Australia (NFS) and the peoples of South Australia. This donation came with long-term funding for the re-establishment of a native woodland system on this property. The specific intention for this revegetation is to help address the on-going declines of woodland birds within this region. The State Government has since purchased additional land adjacent to this area, together with around 70 ha approximately 1.5 km to the south-east (the Barossa Block; Fig. 1.3). The collective 321 ha is now known as the Para Woodland Reserve and is jointly managed by the NFS and the Department of Environment, Water and Natural Resources, South Australia (Geelen 2003).

Native vegetation is at best limited within the surrounding landscape of the Para Woodland Reserve, with many properties still under primary production. Using information sourced from the South Australian Lands Titles Office dated January 1881, Bickerton (2005) interpreted the pre-European vegetation in this area to have consisted of the following. Overstorey dominated by Peppermint Box and/or Black Mallee-box (*E. porosa*), +/- South Australian Blue Gum and/or River Red-gum, and Drooping Sheoak (*Allocasuarina verticillata*). The vegetation at the lower storey levels included *Xanthorrhoea* sp., *Acacia* spp (including Kangaroo Thorn, *Acacia paradoxa*), various low shrubs (including heath species) and native grasses belonging to the genus *Aristida* and *Austrostipa*.

However, several large stands of remnant vegetation do exist within a 15 km radius of the Para Woodland Reserve. The higher elevations to the south-east of the Reserve are

dominated by Long-leaved Box and Stringybark woodland and open forest. These form part of a 2,573 ha block of native vegetation that covers this upland region (Department of Environment 2005). As this elevated zone trends to the north, it progressively loses topography and opens onto valley floors and low undulating hills. Remnant vegetation within these lowland areas is severely restricted due to their extensive clearance for agriculture (Paton *et al.* 2004). Several remnant habitats do exist though over sandy soil to the east of the Reserve. These woodlands are dominated by Pink Gum and Slender Cypress-pine (*Callitris gracilis*).

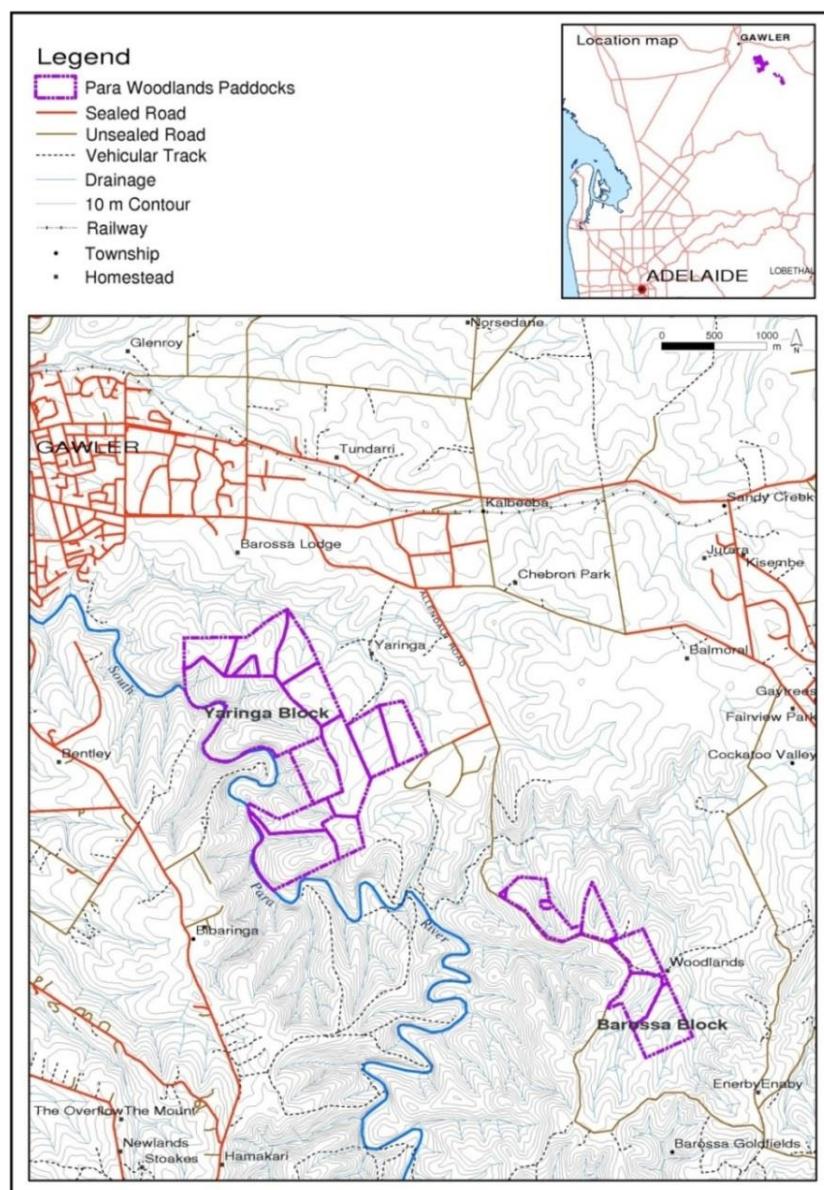


Figure 1.3: The Para Woodland Reserve consisting of two parcels of land 1.5 km apart, the Yaringa Block and the Barossa Block (shown in purple outlines). This property lies 1.5 km south-east from the rural township of Gawler and some 40 km north of Adelaide, South Australia (Sourced from Bickerton 2005).

The Ph. D project

Management of the Para Woodland Reserve is controlled by the Para Woodland Steering Committee. This Committee funded my Ph. D project for the first three years. In return, the project is to provide information and advice specifically to aid in the revegetation works on the Para Woodland Reserve. The Committee required a multi-species approach that was based on the bird species known to be declining within the local region. It was assumed that the bird species that are most likely to eventually colonize any new revegetation are likely to be those species that occupy and come from nearby areas. The species of woodland birds that were deemed to be declining were referenced from Paton *et al.* (2004). At the time of planning this project, this paper contained the most complete list of bird species that are regarded as being in decline across the MLR. The information in this paper was compiled using a variety of sources, namely Baxter (1980), Ford and Howe (1980), Paton *et al.* (1994), Ashton (1996), Eckert (2000) and D.C. Paton (unpubl. data).

This Ph. D project was set around the remnant woodlands that were located within the north-central zone of the southern MLR, within a 15 km radius of the Para Woodland Reserve. The project was designed to reveal habitat associations by woodland birds at a range of spatial scales. At the coarse-scale, information was sought on the habitat associations by bird species with broad habitat type/s that the birds used (and/or did not use). To achieve this, the habitat associations by woodland birds were studied at a scale of ten hectares. Three habitat types were considered, being grassy woodland, open heath woodland and closed heath woodland. Across this region, 15 ten hectare sites were selected, with five each for each of the three woodland types. With native vegetation generally restricted across the lower lying regions due to previous land clearance, some areas could only accommodate a plot size of ten hectares for the assignment of either a grassy woodland site or an open heath woodland site. Within these areas, a plot size greater than ten hectares would usually then consist of more than one broad woodland type.

More precise information on the actual habitat use by bird species was also required. Accordingly, intense studies were conducted on fine-scale habitat use by individuals of nine declining woodland bird species within their respective home ranges. Within a home range, discrete areas of habitat that were used disproportionately more by the birds being

surveyed compared to the immediately surrounding habitat, are referred to throughout this thesis as being “hot-spots”. Within the same home range, any areas that were clearly underutilised by the birds during the survey are referred to throughout this thesis as being “cold-spots”. Any general principles and/or specific habitat features that could be identified from this project was to be used to help drive the re-establishment of woodland habitats on the Para Woodland Reserve.

With the requirement for the new revegetation on the Para Woodland Reserve to ultimately cater for an array of declining bird species, judicious choice in the species that were studied was required. Bird species used for the fine-scale habitat studies were chosen based on several criteria. First, the birds were biased towards insectivorous species that forage on or near the ground. Of the bird species that are listed as declining across south-eastern Australia, greater than 80% are woodland species, with more than half of these using the ground at some point to forage (Robinson 1991; 1994; Reid 1999; Garnett and Crowley 2000; Antos and Bennett 2006). Second, there needed to be a sufficient number of birds of each species in which to survey. Of the bird species regarded as declining within the MLR, some were more prevalent than others and thus were easier to locate within the time constraints of this study. For this, the ten hectare bird surveys (Chapter 3) were used to select the species for further study. Third, each species should differ in how they forage within woodland environments. With the requirement for the Para Woodland revegetation to cater for multiple bird species, the survey work needed to incorporate a range of species that differed in their ecologies, notably their foraging ecologies. As such, the bird species that were selected were each associated with a particular foraging guild, based upon the substrates that they used and typical foraging manoeuvres that they employed (see Recher and Holmes 1985). The foraging guilds used in this project were “glean” (taking prey from substrate surfaces), “probe” (prey taken from within or under substrates, including crevices, loose bark and soil), “snatch” (a moving bird grabs prey from substrate surfaces), “pursue” (actively chasing prey) and “pounce” (bird scans the ground from an elevated vantage point, then drops / flies down onto detected prey). Descriptions of these foraging manoeuvres follow the terminology of Recher and Holmes (1985) and Recher and Davis (1997a). A cross-section of these typical foraging guilds covers a broad range of ecologies and body sizes of birds that are typically found within southern temperate woodlands in Australia (Holmes and Recher 1986).

Nine declining woodland bird species were selected for the fine-scale component of this project. They were the Brown Treecreeper (*Climacteris picumnus*), Buff-rumped Thornbill (*Acanthiza reguloides*), Hooded Robin (*Melanodryas cucullata*), Rufous Whistler (*Pachycephala rufiventris*), White-browed Babbler (*Pomatostomus superciliosus*), White-winged Chough (*Corcorax melanorhamphos*), Willie Wagtail (*Rhipidura leucophrys*), Yellow-rumped Thornbill (*A. chrysorrhoa*) and Yellow Thornbill (*A. nana*; Fig. 1.4).

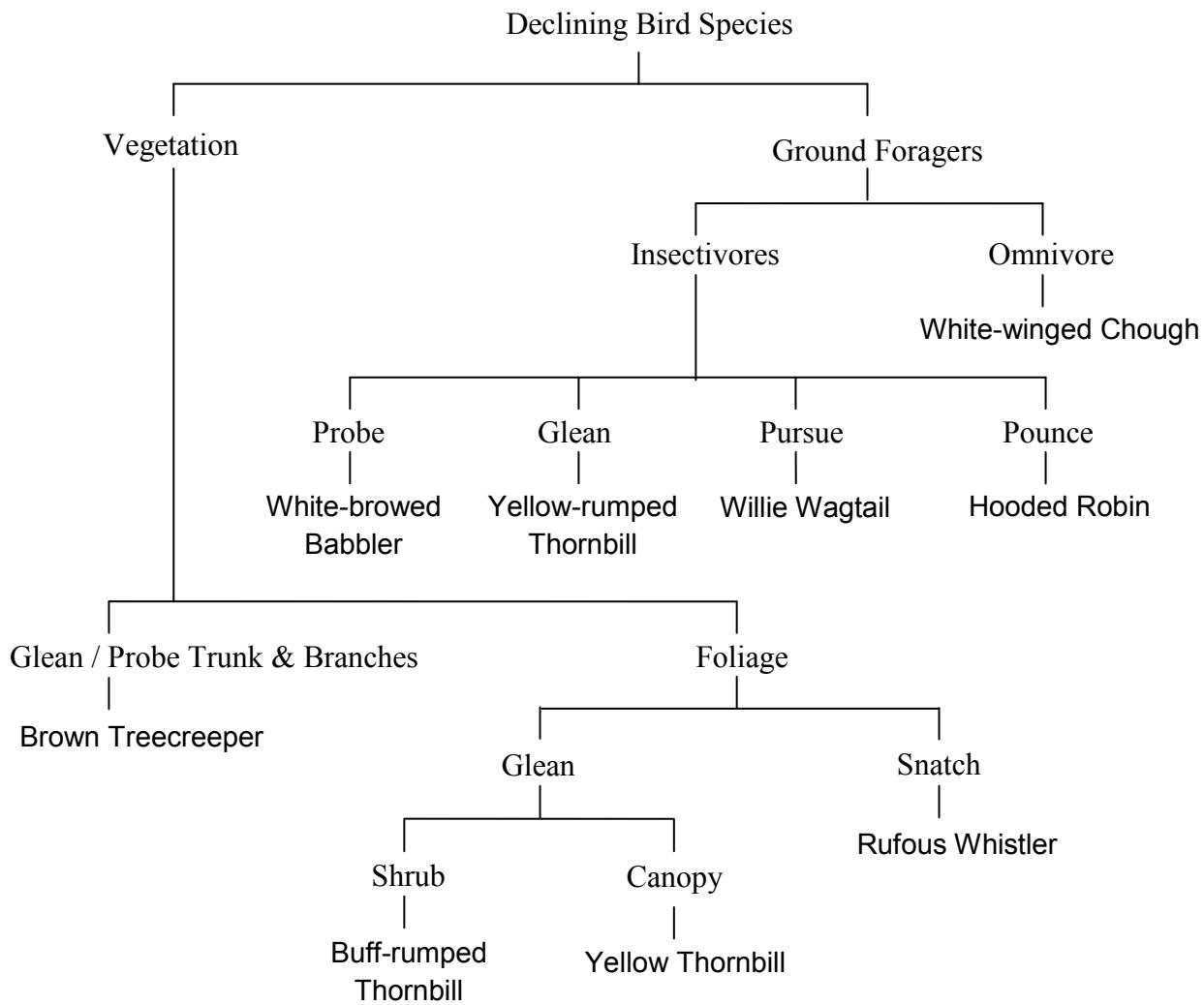


Figure 1.4: A schematic diagram for a selection of foraging guilds that can be found within remnant woodland of the north-central zone of the southern Mount Lofty Ranges, South Australia. The nine bird species that are shown are representatives for each of the specific foraging guilds (after Recher and Holmes 1985, and Recher and Davis 1997a).

The terminology used to describe the foraging guilds and the subsequent choice of bird species should in some instances be interpreted as a guide only, as some species are more flexible in how they forage. For instance, although the Brown Treecreeper is the only species selected in this study that regularly uses tree trunks and major branches as a foraging substrate, this species also forages for an equal amount of time on the ground (e.g. Noske 1991; Rogers 1998; James 2003; Martin and Possingham 2005; Barker 2007; Ford *et al.* 2009). Also, while the Buff-rumped Thornbill can be regularly found gleaning foliage within the shrub layer (Bell 1985; Recher 1989), this species may forage anywhere between the ground and the overstorey canopy layer (e.g. Lyon 1980; Leach and Hines 1987; Wood 1995).

Of the other bird species, the Hooded Robin is a noted pounce forager. This species uses low exposed perches (usually up to 3 m from the ground) to scan for and then fly down onto invertebrate prey (Sullivan 1993; Antos and Bennett 2006). The Rufous Whistler is a snatcher of invertebrates, mainly from the foliage of trees (Keast 1985; Loyn 1985). The White-browed Babbler and White-winged Chough are probbers (and gleaners) of various ground substrates, although their diets generally differ (Heinsohn 1995; Recher and Davis 1998). The Willie Wagtail pursues aerial prey close to the ground, which are either flushed by active searching or detected from a perched position (Harrison 1976; Recher and Davis 1997a). The Yellow-rumped Thornbill gleans invertebrates off the ground more than most other Thornbills (Green 1995; Saunders and Ingram 1995), while the Yellow Thornbill is an arboreal forager that gleans foliage within the canopy and sub-canopy stratum (Leach and Hines 1987; Recher 1989).

The selection of bird species used for the fine-scale habitat studies was also biased towards insectivorous species for reasons other than their overall conservation status. Although invertebrate prey may express seasonal fluxes, they are regularly available throughout the year. A typical pattern within temperate climates is high arthropod abundances during the warmer summer months and low abundances during the cooler winter months (Ford 1985). Due to the predictable year-round availability of invertebrates, insectivorous bird species tend to reside year-round within dry temperate woodlands (Recher *et al.* 1983; Blakers *et al.* 1984). In doing so, insectivorous species significantly contribute to woodland bird species richness (typically 40-70%; Antos and Bennett 2005). Of the species selected for this project, the Brown Treecreeper and White-browed Babbler also hold permanent

territories (Schodde and Mason 1999). In contrast, the availability and timing of resources such as nectar and seeds sees nectarivorous bird species responding to resource pulses at landscape scales (Paton 1986; Paton *et al.* 2004), while granivorous bird species may relocate locally in response to diet shifts between seed-bearing grasses, herbs and invertebrates (Read 1994).

Thesis overview

Chapter 1 introduces the history of the study region, the plight of many of the inhabiting woodland bird species and the purpose of this project.

Chapter 2 then examines the literature for general findings and recommendations that may be used to help inform in the effective revegetation of the Para Woodland Reserve. Emphasis is placed on woodland birds and in particular species of conservation concern.

Chapter 3 investigates the use of three broad woodland types by woodland bird species (grassy woodland, open heath woodland and closed heath woodland). This study seeks to identify if particular bird species are associated with certain woodland types within the north-central zone of the southern MLR. The habitat types studied ranged from open Stringybark forest to open grassy Gum woodland, with the latter habitat said to have previously existed over the Para Woodland site and its surrounds.

Chapter 4 uses the habitat data that were collected from the 15 ten hectare sites in Chapter 3 to measure the degree of structural habitat heterogeneity that existed within each of the 15 study sites. Habitat heterogeneity was measured by dividing each ten hectare site into around 165, 25 m × 25 m quadrats and scoring the habitat features within each to document spatially the degree of heterogeneity and habitat patchiness of various habitat features across each site. Should the study sites contain significant amounts of habitat heterogeneity, this information can be used to help inform on the extent of heterogeneity that may be necessary when planning new revegetation.

Chapter 5 then uses the habitat heterogeneity data from Chapter 4, to investigate if certain woodland bird species associated with particular habitat structures within a ten hectare site.

If individual bird species use particular patches of habitat within the ten hectare site, this further builds a case for incorporating such heterogeneity into revegetation works and for the need to define habitat use by birds at fine-scales.

Chapter 6 reports on the spatial use of habitats by pairs and cohesive groups of nine declining woodland bird species within the MLR. The area requirements of birds belonging to a range of species relates directly to the size of an area that revegetation should cover (or revegetation that is associated with other native habitats should cover). For this study, only relatively short-term area use by pairs and cohesive groups of birds were measures, thus the area estimates can only represent the minimum area requirements of the birds that were surveyed.

Chapter 7 then reports on variable habitat use by birds within their individual home ranges of the nine declining species that were intensively studied. This Chapter (and the next) reports on the use of habitats within the confines of the habitat area in which the birds were contained during the survey (i.e. minimum home range area).

Chapter 8 examines the fine-scale habitat use by birds within their individual home ranges of the nine species that were studied. The habitat attributes that were used (and not used) by the birds are sought by comparing the habitat attributes associated with high use areas (i.e. hot-spots) with habitat attributes associated with low use areas (i.e. cold-spots). Habitat attributes that appeared to either promote or hinder habitat use by individual birds of particular species need to be considered (with any beneficial habitat attributes incorporated) when planning revegetation for target species.

Chapter 9 then investigates habitat use by birds of numerous species that co-inhabited within a common habitat area, by exploring the extent of spatial overlap in their use of discrete patches of habitat by the different birds. This study seeks to establish if individuals from a range of bird species that shared a common habitat area used distinct but separate patches of habitat. If so, this has direct ramifications on the planting design of revegetation, in that the different habitat features need to be provided for as discrete patches within the revegetation, to cater for a diversity of woodland bird species. If supported, this would argue that reconstructed woodlands should not consist of a ubiquitous, highly homogeneous planting, but rather as a spatially diverse woodland

habitat that is made up of repeated subsections that each contains specific habitat elements, where importantly each subsection “type” should differ from any adjacent subsections.

Chapter 10 summarises the objectives and specific outcomes for each of the preceding chapters. I also discuss issues relating to the effective return of temperate woodland habitats for bird species. Suggestions are then made for future bird survey methods that aim to calculate more accurate species density estimates, to identify species-habitat associations, and to help predict the distribution of species across wide habitat areas.

Appendix A provides photographs of each of the 15 ten hectare sites used in this project.

Appendix B details the bird species that were observed to be breeding during this project and the plant species that those birds used.

Appendix C lists the resources the birds were observed to be foraging upon. These include seeds, vegetation and invertebrates, but does not include floral resources.

Chapter 2

Literature review: Effective revegetation for woodland birds

Introduction

An estimated 85 – 90% of the original temperate woodlands in Australia have been cleared in the past 200 years or so (SECA 1996; Lindenmayer *et al.* 2005). As a consequence, a number of woodland bird species have become locally extinct, while many others are of grave conservation concern (Ford and Howe 1980; Saunders 1989; Recher and Lim 1990; Robinson and Traill 1996; Recher 1999; Paton 2000; MacHunter *et al.* 2006). This is particularly so within the more geographically isolated regions, such as the Mount Lofty Ranges in South Australia (Paton *et al.* 2004; Szabo *et al.* 2011). In many cases, human intervention is required in order to redress species decline and to stabilise vulnerable bird populations.

This review of the literature identifies what information is currently available to help inform the return of useful woodland habitat for bird species of conservation concern. Accordingly, this review will focus on the general principles and specific findings that can be applied in order to successfully return woodland habitat for vulnerable species. This review also asks the underlying question, is the information within the literature sufficient to adequately inform the restoration process? Importantly, any knowledge limitations that are exposed can provide the basis for further research.

The findings from this literature review will help guide the design for this Ph. D project. The primary aim of the project is to inform the conservation works on the Para Woodland Reserve in South Australia. This Reserve consists of over 300 ha of previously farmed land that has now been set aside for restoration and revegetation. Specifically, the aim is to re-establish woodland habitat that supports a suite of declining woodland bird species within this region (Bickerton 2005).

Fate of woodland birds in Australia

Nearly half of all Australian threatened ecosystems involve eucalypt woodlands and forests (Sattler and Creighon 2002). Among the faunal groups whose habitats have been compromised by broad-scale clearing, woodland birds are widely understood to have been impacted. Research has shown that eucalypt woodlands and forests host the greatest concentration of threatened and declining bird species within Australia (Robinson 1994; Robinson and Traill 1996; Reid 1999).

Although woodland birds are not the only fauna to be adversely affected by land clearance, a major reason for their dominance within the peer-reviewed literature is that birds are a relatively easy and charismatic group in which to study (Kimber *et al.* 1999; Munro *et al.* 2007). Birds are practical research tools, being numerous, conspicuous, typically diurnal and their mobility promotes landscape scale patch use (Lambeck 1999; Kavanagh *et al.* 2005). Due to their mobility, birds are also considered to be likely candidates to rapidly colonise any new revegetation (Munro *et al.* 2011).

Different species of woodland birds also generally use different aspects of woodland habitats, so they are able to co-exist by using different parts of the same woodland. Collectively, this allows for a rich assemblage of bird species to have the ability to use the same woodland system (Recher *et al.* 2002). In doing so, the bird community interacts with many aspects of the woodland environment, ranging from the ground and shrub layer, throughout the various tree layers, together with the spaces above and between the tree canopies (Ford 1985). As a consequence, birds are often used as bio-indicators to signal for environmental changes. Being near the top of the food chain they are sensitive to change, so their status may be used as an indicator of general environmental wellbeing (Williams and Gaston 1994). Also, birds interact with their environment in a number of significant ways. Many birds (e.g. honeyeaters) provide a pollination service for plants (Paton 1986). Birds also disperse seeds (Forde 1986) and consume a variety of problem invertebrates (Ford 1986). Significant changes in the status of the bird community would be evident by changes in the ecological services that they provide. For example, a low abundance of honeyeaters has resulted in inadequate pollination of native plants within the Mount Lofty region (Paton 2000).

Prior to the year 1970 however, little had been published in relation to the biology or ecology of woodland birds in Australia (Recher 1985b). Since this time, several National and State bird atlases have been produced to map bird species distributions (e.g. Blakers *et al.* 1984; Paton *et al.* 1994; Saunders and Ingram 1995; Barrett *et al.* 2003; Possingham *et al.* 2004; Possingham *et al.* 2006). Atlas style data and a myriad of independent studies have provided strong evidence that a number of temperate woodland bird species have become locally extinct in some areas, while a suite of others are declining in their distribution and/or abundance in many regions across southern Australia (e.g. Ford and Howe 1980; Saunders 1989; Recher and Lim 1990; Robinson 1994; Robinson and Traill 1996; Recher 1999; Garnett and Crowley 2000; Paton *et al.* 2000; Ford *et al.* 2001; Paton *et al.* 2004; MacHunter *et al.* 2006; Woinarski *et al.* 2006).

Current declines in woodland-dependent birds have been reported across a wide variety of systems, including woodlands in south-western Australia (Saunders and Ingram 1995), South Australia (Paton *et al.* 1994), Victoria (Robinson 1991) and New South Wales (Ford *et al.* 1995). Ominously, some woodland bird species that are regarded as common and of least conservation concern are also now starting to show signs of decline in some areas (Mac Nally *et al.* 2009; Szabo *et al.* 2011).

Extinction debt

The decline of many temperate woodland bird species is clearly continuing, despite massive reductions in ongoing native habitat clearance across much of southern Australia (e.g. Robinson 1991; Paton *et al.* 1994; Ford *et al.* 1995). The trend in multiple species decline has been linked to broad-scale habitat loss from past vegetation clearance, the effects of fragmentation and degradation of remnant habitats, and ongoing perturbations such as increased predation rates (e.g. Bennett 1987; McFarlane *et al.* 1993; Nulsen 1993; Glanznig 1995; Kimber *et al.* 1999; Bennett *et al.* 2000; Mac Nally *et al.* 2000a; Ford *et al.* 2001; Cale 2003; Watson *et al.* 2005; Adkins 2006). For example, Bennett (1987) and Soulé *et al.* (1988) both showed that the numbers of individuals of different species of birds within remnant habitats continued to decline with an increase in time since clearing of the surrounding habitat.

Evidence for ongoing species decline lends strong support to dire predictions that the extinction debt is yet to be paid in full (Tilman *et al.* 1994; Recher 1999; Szabo *et al.* 2011). Species are not necessarily in equilibrium with their current environment (Ernoult *et al.* 2006) and it may take many bird generations for the full impact of habitat loss to result in the eventual demise of a species (Tilman *et al.* 1994). By the time these on-going declines are recognised it may be too late to re-establish enough suitable habitat to save some species (Fahrig 2001). Even if habitat availability was instantaneously increased, modelling by Schrott *et al.* (2005) suggested that some species that had already declined past a vulnerability threshold could not be recovered. These findings provide a strong argument for a pro-active response by returning native habitats, and not just waiting for a species to be recognised as being in trouble before any action is taken. However, defining vulnerability thresholds for different species is still required.

Revegetation

Within Australia, rural landholders, business organisations and conservation and community groups, along with various levels of government participate in revegetation (Kimber *et al.* 1999; Bennett *et al.* 2000; Commonwealth of Australia 2011). This activity is occasionally supported by information generated to guide plant establishment (see Greening Australia 2008). However, historically less emphasis has been placed on incorporating the habitat needs of fauna in these restoration activities (Ryan 2000; Morrison 2001; Paton *et al.* 2004; Munro *et al.* 2007; Paton and O'Connor 2010).

Typically the benefits of revegetation for fauna are merely incidental or assumed (Bennett *et al.* 2000). Palmer *et al.* (1997) describes this situation as the “Field of Dreams Hypothesis”, implying that “if you build it, they will come”. This assumes that, 1) the floristic composition that is planted is appropriate for that location, and 2) it provides suitable habitat for native biota. Currently there is limited evidence to support the assumption that habitat return will automatically benefit declining and/or threatened species (Vesk and Mac Nally 2006; Kavanagh *et al.* 2007; Loyn *et al.* 2007). The question remains as to the habitat features that different bird species require before they can effectively colonise newly constructed woodland habitats. Critical habitat features might be the number of trees with hollows, a specific range of tree densities, the density and

types of understorey shrubs, as well as having sufficient area of suitable habitat to support individuals.

Time lags in habitat development

Time is of the essence to restore and reinstate suitable habitats for temperate woodland birds (e.g. Vesk and Mac Nally 2006). The amount of time that is required for new habitats to develop sufficiently is a limiting factor that hinders its use by many fauna (Manning *et al.* 2004; Mac Nally 2008; Maron *et al.* 2010). Several key habitat attributes require decades or even centuries to develop (Kanowski *et al.* 2003; Dolman *et al.* 2007). For example, it can take up to 150 years for significant fallen timber loads to establish, over a century for tree hollows to develop and 50 years for adequate nectar loads to be produced by flowers (Vesk *et al.* 2008). In contrast, most revegetation in Australia is only a decade or so old (Vesk and Mac Nally 2006; Mac Nally *et al.* 2010).

The length of time required to create useful habitats means even if we start now, some habitats may not be beneficial for some species for another century or so. This time lag in suitable habitat development may be too late for some species that are currently considered to be of conservation concern (Morris *et al.* 2006). In Australia for instance, around 100 vertebrate species classified as near threatened to rare use hollows, with around half of these being bird species (Gibbons and Lindenmayer 2002). These hollow dependent fauna are unlikely to have new areas made available to them that contain suitable habitat features (i.e. hollows) in the foreseeable future.

The pressing need for effective habitat reinstatement remains a priority. Habitat development however is slow and uncertain, which must be factored into any restoration program for the recovery of species (Moilanen *et al.* 2009). This is especially so for temperate woodland ecosystems (Morris *et al.* 2006). Unfortunately this issue is rarely ever considered in revegetation policy (Gibbons and Lindenmayer 2007; Mac Nally 2008). There is an urgency to understand the habitats and habitat features needed by the declining species if the ongoing revegetation works are going to deliver effective habitat for these species. One might even argue that this information is needed before we even invest in the on-ground works. Given the urgency with which new habitats need to be established, this

may require initial plantings to be subsequently modified once the habitat requirements of particular bird species are better known.

Connectivity within fragmented landscapes

Disproportionate vegetation clearance over the more arable regions has imposed environmental discontinuity, producing isolated fragments of native vegetation within the landscape (SECA 1996). The ability for fauna to effectively use habitats which are highly fragmented is an ongoing concern. Anticipated biotic responses to habitat fragmentation often have their foundation in ecological theories proposed by island biogeography (MacArthur and Wilson 1967) and metapopulation dynamics (Levins 1969). In part, these assume a limiting factor is either a reduced rate of biotic dispersal and/or an inability to successfully move through a landscape.

If a lack of physical connectedness between habitats is a limiting factor, then establishing connectivity between isolated remnants should promote accessibility by biota and enhance landscape community structure. In this context, establishing wildlife corridors within fragmented landscapes has been widely used in environmental management; with connectivity traditionally solved by establishing strips of vegetation between the remnant patches (McDonald 2001; Freudenberger *et al.* 2004). Indeed, the most common form of revegetation within agricultural districts are linear shelterbelts designed for crop and stock protection (Nicoll and Dobbie 1996; Kimber *et al.* 1999). As a corollary, some researchers suggest these linear plantings may provide useful habitat in their own right, together with functioning as potential wildlife corridors by connecting populations within disjunct remnants (e.g. Bennett 1991; Saunders and De Rebeira 1991; Bentley and Catterall 2003; Haslem and Bennett 2008). Notably though, these farm plantings do not necessarily connect two pieces of remnant vegetation, and there remains a lack of information and/or evidence that separates potential use from actual use by biota.

Some positive responses in the use of vegetation strips have been reported for a range of biota from both Australia and other countries, which include mammals (Mansergh and Scotts 1989), birds (Haas 1995) and plants (Gonzalez *et al.* 1998). However, despite being promoted as the primary method for connecting isolated populations, there is a paucity of

information as to the ecological effectiveness of strips of vegetation (Simberloff *et al.* 1992; Rosenberg and Noon 1997; Donald and Evans 2006). Many researchers are increasingly questioning the function of strip vegetation within fragmented landscapes, with concerns that relate primarily to habitat quality (e.g. strip width, vegetation component, etc.) and actual use by fauna (Nicholls and Margules 1991; Rosenberg and Noon 1997; Harrison and Bruna 1999b; Donald and Evans 2006).

Confounding this issue, woodland bird species generally differ in their response to the degree of habitat fragmentation and spatial isolation (Trzcinski *et al.* 1999; Watson *et al.* 2005). The perceived lack of habitat connectivity must be considered from the perspective of individual species (Saunders *et al.* 1991; Lindenmayer *et al.* 2000). This includes the ability of a species to disperse and the size of its natural home range (Hobbs 1992). For instance, Brooker *et al.* (1999) showed that within the highly fragmented Western Australian wheat-belt, the movements of the Blue-breasted Fairy-wren (*Malurus pulcherrimus*) was impeded by gaps in remnant stands greater than 60 m, whereas the White-browed Babbler (*Pomatostomus superciliosus*) crossed gaps at least 270 m wide. Factors other than the perceived inability of a species to move within a landscape must also be considered when assessing the vulnerability of a species to habitat fragmentation. For example, the susceptibility of the White-browed Babbler to habitat fragmentation can relate more to the social structure of this species than to the degree of habitat fragmentation alone (Cale 2003).

Species decline within remnant stands cannot always be explained by an inability to move between fragmented patches. Fraser and Stutchbury (2004) showed that connectedness does not necessarily equate to effective ecological functioning for the Scarlet Tanager (*Piranga olivacea*). In their study, although these birds were not unduly restricted by the lack of connectivity between fragmented habitats, the pairing success of those inhabiting the smaller fragments were reduced. Pairs attempting to breed in small remnants had lower reproductive success compared to birds in larger patches, which was in part presumably the result of edge effects with increased predation. In support, corridors have strongly correlated with increased predation rates on the nest contents of some woodland bird species (Major *et al.* 1999; Willson *et al.* 2001). A low likelihood of successful nesting may mean that individual birds do not see the small remnants as being suitable for breeding and so do not remain, rather than their being a lack of connectivity *per se*. This

would imply that corridors are not necessarily good quality habitats for some species that actually use them and importantly, may even potentially exacerbate the decline of some species through reduced fecundity.

The concept of wildlife corridors is too simplistic to be broadly applied for many species (Wiens 1994). This undoubtedly explains why corridors have typically not functioned as first hypothesised. There remains a critical lack of cohesion between ecological theory and empirical evidence on this issue (Harrison and Bruna 1999a). Despite serious attempts at providing habitat connectivity via corridors, there is also an inability to gauge the effects of fragmentation over multiple scales relative to ecological relationships, which hinders effective management policy (Kupfer 2006). Bailey (2007) suggested that this may be due to sheer regional habitat loss posing a greater effect on biodiversity than fragmentation alone.

One may even question if corridors are practical for the recovery of species within highly fragmented landscapes. Remnant stands across southern Australia are often small (with many less than one hectare) and/or highly modified from their natural pre-European condition (e.g. Paton *et al.* 2004; Lindenmayer *et al.* 2005). Hence remnants typically lack a sufficient ability to deal with a range of environmental stresses and/or disturbances, which further erodes their worth as a conservation resource (Bennett *et al.* 2000; Lindenmayer *et al.* 2008). Consequently, many relic patches of native vegetation are not considered to be viable in the long-term (Cofinas and Creighton 2001). Connecting fragmented habitats which are subject to ongoing degradation makes little sense when trying to achieve long-term nature conservation goals.

Broad-scale revegetation

Broad-scale revegetation has been used both within Australia and overseas in an attempt to address the negative impacts of vegetation clearance (e.g. SA Urban Forests – Million Trees Program (www.milliontrees.com.au/), Habitat 141 (<http://www.habitat141.org.au/>), Gondwana Link (<http://www.gondwanalink.org/>), Walker *et al.* (2004) and Wenhau (2004)). In this context however, much of the revegetation across southern Australia has not provided good quality habitat for threatened birds, as most of the bird species using the

revegetation are either introduced and/or of least conservation concern (as reviewed by Kimber *et al.* 1999; Ryan 2000; Munro *et al.* 2007; but see Kinross 2004). These findings are consistent with similar studies abroad (e.g. Yahner 1982; Hinsley and Bellamy 2000). The likely reasons for the lack of use by desired species includes revegetation that only covers an average of a few hectares, the use of only one or two plant species, and new plantings being planted as isolated stands (Wilson *et al.* 1995; Freudenberger *et al.* 2004; Paton *et al.* 2004; Zerger *et al.* 2006; Smith 2008). In comparison, more bird species of concern and more bird species overall are likely to use a single large patch of habitat, rather than many smaller patches which collectively cover the same land area (Lyon 1985a; Kimber *et al.* 1999; Kavanagh *et al.* 2007; Munro *et al.* 2007).

But where within agricultural landscapes are the best locations to revegetate to optimise the benefits for native species, especially those of conservation concern?

Proximity to remnant vegetation

The placement of new revegetation to gain maximum benefits for biota is of critical importance. The literature abounds with examples that remnant vegetation hosts significantly greater native biodiversity than the surrounding farming landscape (e.g. Hobbs *et al.* 2003; Loyn *et al.* 2007; Bhagwat *et al.* 2008; Brockerhoff *et al.* 2008). Remnant habitats also host many more bird species that are declining within a region (Martin *et al.* 2004; Loyn *et al.* 2007; Munro *et al.* 2011). This argues for increasing habitat in close proximity to existing native vegetation, rather than as isolated stands within an agricultural setting (e.g. Paton *et al.* 2004; Kavanagh *et al.* 2007; Lindenmayer *et al.* 2010).

The benefits of positioning new vegetation near existing remnant patches are numerous. Both the remnant and developing revegetation would benefit through an increased core to edge ratio, thus reducing the potential for edge effect induced perturbations (e.g. Luck *et al.* 1999; Major *et al.* 1999; Weldon 2006). This would also help in retaining what natural integrity is still held by the remnant stand, which is critical for maximising the conservation value of both the remnant and revegetated habitats (Prober and Thiele 2005). For example, when revegetation is in close to remnants, it increases the potential for

desired bird species to eventually use the new vegetation (Major *et al.* 2001). By revegetating in close proximity to a source bird population, the risks of dispersal into the nearby newly created habitat are also lower (Major *et al.* 2001). The obvious question then is how much habitat area is required to sustain viable populations of woodland birds?

Habitat area requirement

The literature provides strong evidence that habitat fragmentation disrupts ecological processes and that continuity of vegetation is important for biodiversity. This is based on both qualitative and quantitative evidence. For example, in a review of simulated ecological models using fragmented habitats, Andrén (1994) suggested that a species maintenance threshold may exist between 10 and 30% of native habitat cover within the landscape (based on the progressive removal of “units” from 10,000 area units). Between 10 and 30% cover, population decline and species loss reverted from a linear relationship with pure habitat loss to an exponential decline. Andrén (1994) argued that a tipping-point which amplified species loss occurred when the effects of small patch sizes and growing isolation effects became compounded with increasing loss of habitat.

Radford *et al.* (2005) supported the study of Andrén (1994) in part, by demonstrating that a positive (although non-linear) relationship existed between the percent of native vegetation in the landscape (at 100 km²) and the decline in species richness of woodland-dependent bird species across north-central Victoria. Species richness declined sharply however as vegetation cover fell below 10% within the landscape. Fahrig (2001) took the minimum habitat concept further by modelling the biotic factors that could influence sudden species decline. The study found that the species extinction threshold was influenced most by reproductive rate, followed closely by emigration rate. Importantly, no single amount of minimum habitat cover could be found that would preserve all species, as minimum area requirements varied between species. This was influenced by a species reproductive potential and dispersal attributes, which varied greatly between species. Species with low fecundity and a risky dispersal strategy (high emigration rate with low survivorship within the surrounding matrix) required the largest habitat areas to persist.

In support, it is important to note that most studies used to model species richness with habitat area generally assume that bird population density is proportional to the habitat cover within the landscape. This gives no insight as to the minimum habitat area that is required by a breeding pair / group of a certain species, nor the number of pairs / groups and hence the overall habitat area that is required to sustain a viable population of that species.

The proposed species maintenance threshold of up to 30% native vegetation cover is troubling, as some declining bird species require hundreds of hectares per breeding pair or family group (e.g. Paton *et al.* 2010). Large area requirements for individuals will require extensive areas of habitat to be present to support sufficient individuals of a species to have viable populations. With temperate woodlands across southern Australia having been cleared by around 90% (SECA 1996), the amount of remnant woodland across most agricultural districts falls well below the proposed 10% critical threshold for many bird species. Ten percent native woodland cover provides only 100 ha of habitat every 10 km² and this may only be useful when existing as a single patch. Within highly fragmented landscapes (such as agricultural districts), 10% cover may only consist of lots of small patches of vegetation, none of which meets the areal requirement for bird species.

The long-term conservation of temperate woodland bird species will in many cases require substantial revegetation to return woodland habitats. To date though, revegetation has typically been conducted over small spatial scales of less than one to several tens of hectares (e.g. Nicoll and Dobbie 1996; Paton *et al.* 2004; Mac Nally *et al.* 2010; Munro *et al.* 2011). Across the Mount Lofty Ranges in South Australia for example, vegetation plots are typically less than one hectare and are densely planted in rows with only one or a few plant species (Paton *et al.* 2004). To be of more benefit to birds, good quality revegetation is needed to cover substantially more area, and be delivered in large patches. For instance, Westphal *et al.* (2007) modelled the area requirements for 22 bird species within the Mount Lofty Ranges, with half of these being declining species (based on Paton *et al.* 2004). This study recommended that for these birds, the mean revegetation patch size should be at least 780 to 4,010 ha, in order to support a sufficient number of individuals of each species to maintain sustainable populations of those species.

Considering the Para Woodland Reserve only covers around 300 ha, this highlights the fact that when revegetated, only a few individuals of some of the declining species would be

supported (even if the habitat was of high quality for those bird species). Accordingly, there is a critical need to carefully plan revegetation activities so to maximise the conservation benefits for this property, with the added need to undertake comparable restoration on adjacent properties.

Knowledge of the area requirements of bird species is only one aspect in creating useful habitat. Also needed is the re-establishment of the right types of habitat for birds, together with an appreciation for the complementary nature of habitats that might provide resources at different times of the year, thus providing year-round resources at the regional scale.

The need to re-establish temperate woodlands over agricultural soils

The areas that were preferentially cleared of their natural habitats that now support high agricultural production and some human settlements, are typically located within high rainfall areas over better quality soils (Recher 1985b; Australian Natural Resources Atlas 2007). Much of the native vegetation that has been left is largely situated over poorer quality soils and in settings that were not conducive for agriculture. These include areas with impoverished sandy soils and landscapes that contain steep gullies and rocky ridges (e.g. Cofinas and Creighton 2001; Paton *et al.* 2004).

As a result of preferential clearing, the comparatively better quality habitats have been essentially removed (Watson 2011). To explain, the disproportionately cleared areas were specifically targeted as they stood over the more productive soil types (e.g. Burvill 1979; Paton *et al.* 2004; Kraehenbuehl 2005). Often these productive areas were cleared right to the boundary of agriculturally poorer soils, where any native vegetation was subsequently left (Thomson *et al.* 2009).

The principal reason for these clearances was to enable agricultural development. Just as the quality of soils affect human food quality, so the quality of soils also affects the quality of the habitat for a wide array of soil biota (Warkentin 1995). Indeed, a measure of soil invertebrates is often used to gauge the quality of a soil (Stork and Eggleton 1992). This is in recognition that the morphological, chemical and physical properties of soils are controlled by the biological processes of the soil biota (McKenzie *et al.* 2004).

For example, for a plant to be able to uptake nutrients via its roots, plant nutrients must be available in the inorganic (or mineral) form. For this to occur, inorganic nutrients must first become available within the soil solution via the process of mineralisation. The plant itself is unable to perform this function and must rely upon the metabolic processes of an array of soil organisms. The soil biota do this by releasing inorganic nutrients as by-products from the decomposition of organic matter, such as excrement and dead plant and animal tissues (Brady and Weil 2000). To emphasise the importance of this process, the soil biomass has been likened to “the eye of the needle through which all nutrients must pass” (Jenkinson and Ladd 1981).

Soil invertebrates further contribute to increased soil fertility by enhancing soil porosity. This helps to aerate the soil and promotes a higher soil water content by enhancing water infiltration (Warkentin 1995; Brady and Weil 2000). Soil organisms also participate in the carbon cycle, liberating trapped carbon contained within organic matter back into the atmosphere as carbon dioxide (Brady and Weil 2000).

Native plants growing within these richer soils would intuitively be more productive, with flow-on benefits for the fauna inhabiting such habitats. In support, Recher (1985a) reported both the abundance of birds and bird species richness was positively correlated with the productivity of the plant communities within the habitats in which birds reside. Watson (2011) has extended this and argued that clearance of natural woodlands over agriculturally rich soils has reduced the number of soil invertebrates within heavily cleared regions, which may be associated with the decline of many insectivorous bird species.

The return of temperate woodlands that were preferentially cleared may also contribute to the stability of regional biodiversity in some areas. To reiterate, the more productive soils have been disproportionately cleared and thus more of the productive native habitats have been lost. These lost habitats may have provided resources at different times of the year, and were likely to have supported more birds and/or bird species. Thus there is a requirement to preferentially return habitat within extensively cleared areas to return some balance to the ecosystem.

For instance, sunlight energy is rarely ever a limiting resource in Australia. Sugars in the form of nectar are thus inexpensive for plants to produce via photosynthesis, except when

water is limiting (Ford *et al.* 1979). Although remnant vegetation growing at altitude generally exists within higher rainfall zones (e.g. Recher 1985b), the plants generally grow over skeletal soils compared to the deeper soils that drape over the adjacent plains. Elevated slopes also have a greater tendency for precipitation run-off, which lowers the water-holding potential for shallow, higher altitude soils (Burt and Butcher 1985).

This is true for the remnant vegetation that grows within the elevated regions of the Mount Lofty Ranges. Here it has been shown that remnant vegetation flowers in winter and spring, to coincide with periods of more consistent rainfall (Paton and Ford 1977; Ford 1979; 1983). In contrast, the now cleared Gum and Box temperate woodlands on the more productive lower elevations traditionally flowered in spring and late summer-autumn (Ford 1979; 1983; Ford and Paton 1985). These staggered flowering periods once provided a more consistent year-round nectar supply within this region. Due to the loss of the more productive habitats however, nectar today is a limited resource in the summer to autumn period (Paton *et al.* 2004). This has been shown to adversely affect some nectarivorous bird populations and their pollinating efficiency of local native plants, leading to reduced seed-set in some plant species that flower during winter and spring (Paton 2000; Paton *et al.* 2004).

A further argument for establishing temperate woodlands is the opportunity to return certain woodland types which are currently under-represented within the landscape (Cofinas and Creighton 2001; Paton *et al.* 2004; Thomson *et al.* 2009). On the western plains and escarpments of the southern and north-central zone of the Mount Lofty Ranges, various woodland plant communities that once prevailed are now severely limited due to their extensive clearing. These include various woodlands that were dominated by at least five different Gum and Box *Eucalyptus* species, together with species such as *Allocasuarina*, *Callitris* and *Melaleuca* (Smith and Goodwins 2001; Croft and Brandle 2003). These pre-European vegetation communities grew in relatively defined areas across the Adelaide Plains (Fig. 2.1). What is not clear however is the influence such an array of different woodlands may have had on either the distributions of bird species in general, or the importance of certain woodland types for specific bird species. The question remains, are certain woodland types better for some declining bird species, compared to other woodland types within the same region?

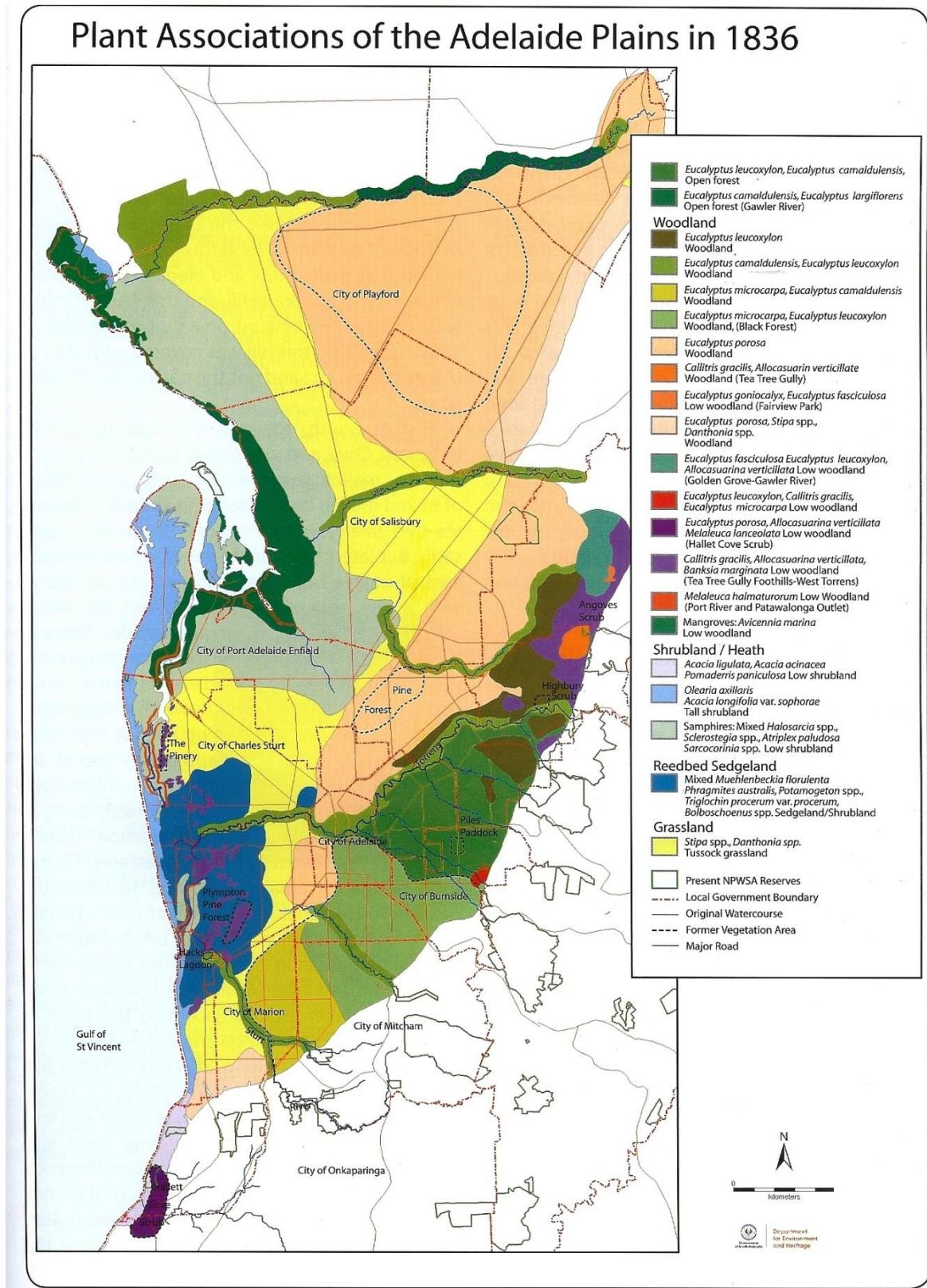


Figure 2.1: Pre-European vegetation communities across the Adelaide Plains. The vegetation ranges from open forest and woodland to shrub / heath, reedbed sedgeland and grassland (vegetation boundaries have been approximated; sourced from Krahenbuehl 1996).

Guidelines to inform revegetation activities

Revegetation works intended to deliver ecological gain in the heavily cleared regions across southern Australia have been based primarily on general guidelines. Central to most of these revegetation strategies are, 1) the need to increase the vegetation extent across landscapes, and 2) to build on existing patterns of remnant vegetation to improve their integrity, habitat value and connectivity (Wilson and Lowe 2003). In response, remnant vegetation *per se* is often used as a model to guide revegetation works (Bennett *et al.* 2000). The characteristics of remnant vegetation that are mostly used to help plan revegetation works are generally considered at the coarse-scale. This typically consists of listing the species of plants found within a particular woodland type (e.g. grassy woodland or heathy woodland), but does not consider the relative densities of the different plant species and how they are arranged in space (Hobbs 1993).

Identifying broad habitat characteristics of natural reference sites are still important in the planning stages for revegetation. For example, native ground cover plants (e.g. grasses, forbs and shrubs) that are usual in many remnant stands may not naturally colonise restored areas, requiring those species to be included in initial plantings (Munro *et al.* 2009). Information on the structural components of remnant vegetation is also critical. For instance, habitats containing understorey vegetation have been shown to host a greater bird species richness (Armstrong and Nichols 2000; Arnold 2003; Martin *et al.* 2004), while a diversity of shrub species can enhance the natural biodiversity among invertebrates (Moir *et al.* 2010).

Despite efforts to re-establish more natural systems through revegetation, these plantings have generally not proved to be useful habitat for many desired bird species (Paton *et al.* 2004; Loyn *et al.* 2007; Munro *et al.* 2007; Bhagwat *et al.* 2008; Paton and O'Connor 2010). Of the woodland bird species that do occupy revegetated habitats, many tend not to reproduce well within revegetated areas (Selwood *et al.* 2009; Mac Nally *et al.* 2010). This could suggest that either the revegetation does not provide quality habitat for those species or that habitat development is in the early phases of succession (Wilkins *et al.* 2003). Furthermore, if there are fundamental habitat deficiencies from the onset, then

these areas of new habitat as they mature may not support breeding for vulnerable bird species in the long run.

Of great concern though is the fact that few revegetation practices actually set specific conservation goals that are then measured for their success (Hobbs and Norton 1996). Indeed, the Australian National Audit Office (2001) challenged the Natural Heritage Trust for deficiencies in monitoring and evaluation of their revegetation strategies to gauge the progress of funded projects. Clear achievable goals for new revegetation should be set from the start, with these goals focused on accomplishing the characteristics that are desired of these new habitats in the future (Hobbs and Harris 2001; Bennett and Mac Nally 2004). With stated goals, revegetation can then be monitored and assessed for specific ecological outcomes, against which revegetation effectiveness can be measured (Hobbs *et al.* 1993b).

However there is a general lack of agreed standards on what and how to measure biodiversity outcomes (Catterall *et al.* 2004; Atyeo and Thackway 2009). As a result, measures of success for revegetation focus mainly on action targets, such as the number of trees that are planted, rather than performance targets like plant survivorship, self-sustainability and extent to which functional habitats have been created and are being used by specific fauna (Hobbs 2003). This has resulted in a distinct lack of information within the literature that can be used to help guide the re-establishment of functional woodlands that support desired wildlife (Yates and Hobbs 1997; Cabin 2007).

For woodland birds, this general lack of information is due to the fact that many studies simply measure the presence / absence or changes in abundances of birds within patches of vegetation. Often, bird-habitat relationships have been inferred but not statistically supported, with most bird species habitat requirements concluded from patterns of occupancy within certain habitat types (Gilmore 1985). Although some key habitat requirements have been identified for some bird species (e.g. rough-barked tree species and tree hollows for the Brown Treecreeper; e.g. Possingham and Possingham 1997; Cooper and Walters 2002), for the vast majority of species those details are lacking. The explanatory variables often used to describe habitat associations by bird species are often related to habitat attributes at the course-scale. These include floristic content, age of the vegetation and its location and configuration within the landscape (Martin *et al.* 2004;

Kavanagh *et al.* 2005; Kavanagh *et al.* 2007; Lindenmayer *et al.* 2010). This course level perspective does not provide adequate details on which to base a revegetation program, because the vegetation details such as plant densities are often not documented. As a result, there is little consensus as to what floristic and structural assemblages constitute functional and self-sustaining habitats for declining bird species (Hobbs 2003).

There remains a critical need to understand the explicit habitat needs of bird species of conservation concern, especially at the fine-scale (Lindenmayer *et al.* 2002; Paton *et al.* 2004; Prober and Thiele 2005; Lindenmayer *et al.* 2008; Mac Nally 2008; Paton *et al.* 2010). This is a glaring gap in knowledge which restricts the advance in effective habitat restoration to help conserve wildlife populations (Kimber *et al.* 1999). From inception, human choice influences many of the eventual restoration outcomes (Allison 2007), as the design at the time of planting dictates many of the habitat attributes that eventuate (Fuller *et al.* 1995). Effective revegetation must then consider the specific habitat needs of the species that are ultimately desired to use these newly created habitats (Mac Nally 2008). However, other than coarse-scale requirements such as sufficient habitat area for home ranges (e.g. Fahrig 2001; Vance *et al.* 2003), the fine-scale habitat needs of species are rarely ever considered in the design and implementation of habitat restoration (Lindenmayer *et al.* 2008). This gap partly relates to our limited knowledge of the habitat requirements of most species, both here in Australia and elsewhere (e.g. Kimber *et al.* 1999; Block *et al.* 2001; Miller and Hobbs 2007), or how faunal communities may be reassembled (Walker *et al.* 2004).

The need to understand habitat use at finer-scales becomes evident when species use habitat disproportionately within their individual home ranges (e.g. Paton *et al.* 2010), or if species require different habitat types for different life events (Mac Nally 2008). Supporting this, Loyn *et al.* (2007) found that habitat attributes within eucalypt plantations proved to be a stronger explanatory variable than the plantation size or shape in explaining the distributions of birds. Laiolo *et al.* (2004) also reported that fine-scale habitat features were correlated with predicting the composition of a bird community within the Italian Alps. These finding also agreed with Mac Nally *et al.* (2002a) who reported birds collectively are strongly influenced by habitat structure.

Responses to habitat can be species (Loyn 1987) and/or guild specific (Loyn *et al.* 2007). Different responses by species to a range of environmental cues can also dictate which species are present, even between closely related species (Villard *et al.* 1999; Mitchell *et al.* 2006; Radford and Bennett 2007). At the landscape scale, individual species can respond differently to habitat mosaics (Lindenmayer *et al.* 2002). This is presumably owing to the fact that individual species may respond differently to the same environmental condition or to different scales of that condition (Storch and Bissonette 2002). Different responses to the same environmental cue may even be experienced by bird species that are ecologically similar (Lindenmayer and Fischer 2003; Nichols and Nichols 2003). This underscores the need to consider habitat requirements of species individually, especially those species of conservation concern (Lindenmayer *et al.* 2002).

Summary

Extensive habitat clearance across southern Australia has left remnant vegetation as highly fragmented stands across the landscape. Because of this, losses of some woodland bird species and the on-going decline of many others, support predictions that more species are likely to become extinct if we fail to act. Simply maintaining residual habitats as they currently stand will not result in species conservation, demanding large-scale reinstatement of priority habitat (Saunders *et al.* 1993; Bennett *et al.* 2000; Hobbs and Harris 2001; Vesk and Mac Nally 2006).

The literature provides some broad recommendations to help guide revegetation works. These include, 1) revegetation of the disproportionately cleared lands that contain the higher quality soils, 2) plant large patches rather than corridors, 3) plant numerous species to create habitat heterogeneity, 4) plant in close proximity to stands of remnant vegetation, and 5) increase habitat area to at least 30% within the landscape. However the specifics such as plant densities and arrangements are still missing from the literature. Also missing are the specific habitat elements that are required by individual bird species.

Revegetation works to date have not produced quality habitats for many woodland bird species. A limiting step in effectively revegetating for woodland birds is the general lack of understanding of how these species relate to their environment. Most studies on birds

simply report on patterns rather than fundamental processes. As a result, the specific needs of different bird species remains poorly documented, together with how those needs may vary both temporally and spatially. The key here is to focus on the ecological requirements of bird species at different scales and identify the habitat attributes that may be missing and/or limiting within habitat areas (Ford *et al.* 2009). These habitat features can then be used to enhance existing vegetation (both remnant and revegetated) and be incorporated into future revegetation strategies to benefit vulnerable wildlife (Paton *et al.* 2004).

According to the literature, the Para Woodland Reserve is an excellent example of a high priority area for the re-establishment of temperate woodland habitat. This property, covering in excess of 300 ha, stands over relatively fertile soils that were originally cleared for farming. This Reserve is also situated within a heavily cleared agricultural zone, with the remnant vegetation on this property and within the surrounding landscape falling well short of the recommended 30% cover to adequately sustain native wildlife populations (Fig. 2.2). Fortunately, several large stands of remnant vegetation exist within a 15 km radius of this Reserve (see Chapter 1 for details). This allows the opportunity to study and report on habitat use by a number of locally declining woodland bird species. Bird species such as these are hoped to one day inhabit the newly revegetated Para Woodland Reserve.

This current Ph. D project will study habitat use by birds belonging to a range of woodland species within the nearby remnant vegetation to the Para Woodland Reserve, with an emphasis on declining species. This project was specifically designed to aid in the planning of the revegetation works for the Para Woodland Reserve, in the hope of helping to reverse the ongoing decline of a suite of temperate woodland bird species within this region. This project will consider habitat use by the birds at both the coarse-scale (i.e. woodland types at ten hectares) and the fine-scale (i.e. patches of habitat within individual home ranges). The effects of “woodland type” and “patch type” in influencing which bird species occupy and use certain habitat areas have long been recognised in bird ecology (e.g. Karr and Roth 1971; Cody 1981). The general view is that woodland birds in Australia (and elsewhere) respond more to structural features rather than floristic features (e.g. Recher 1991; Munro *et al.* 2011). This suggests that woodlands that consist of different plant species but which are structurally similar will be used by the same suite of bird species.

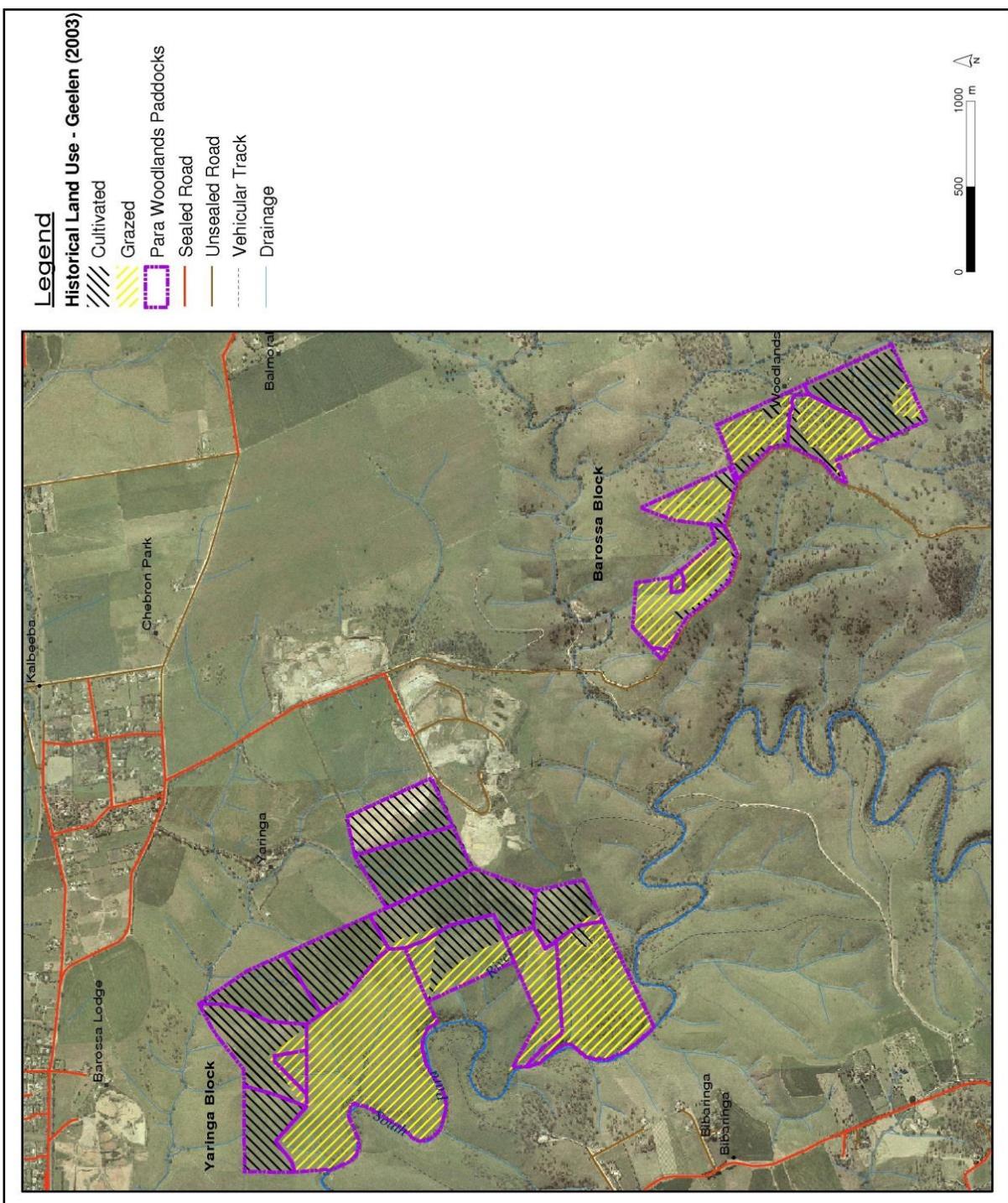


Figure 2.2: The Para Woodland Reserve (purple coloured boundaries) is situated within an agricultural zone that has largely been cleared of its native vegetation. The areas containing black coloured parallel lines indicate previously cultivated land. The areas containing yellow coloured parallel lines indicate previously grazed land. The housing to the north of this imagery is associated with the rural township of Gawler, South Australia (Sourced from Bickerton 2005).

Chapter 3

Coarse-scale habitat use by declining woodland bird species

Introduction

Associations between bird species and habitat types are generally considered at large spatial scales (e.g. landscape scale) or along broad topographic gradients (e.g. Lyon 1985b; Milledge and Recher 1985; Shields *et al.* 1985; Fisher 2001; Hewson *et al.* 2011). In ecology however, a trade-off must exist between attempting to understand how birds relate to their natural environment and using a manageable scale in which to study a habitat area, in both time and space. To this end, bird surveys designed to examine species-habitat associations usually employ point counts, timed area searches (e.g. 20 min / 2 ha) or by conducting counts along fixed width transects. These data are then used to infer which species are associated with the wider habitat area.

Armed with this information, researchers then compare the compositions of avian communities between different areas or habitat types. This is aimed at explaining the ecological separation between species based upon habitat differences, or in identifying general habitat attributes that may help explain the occupation by certain species within particular habitat settings (e.g. Catterall *et al.* 2001; Loyn 2002; McElhinny *et al.* 2006). Other applications include demonstrating seasonal differences in habitat use by birds, and comparing the bird species within both remnant and revegetated sites to gauge the effectiveness of that revegetation (e.g. Slater 1995; Loyn *et al.* 2009).

Collecting data over small spatial scales that are then used to infer the characteristics of the broader habitat comes at a price. Although sampling small areas is efficient in both time and effort, using these data to infer patterns at broader scales becomes increasingly inadequate as the level of habitat heterogeneity increases and the composition of mobile biota fluctuates across a region (Mac Nally and Horrocks 2002). This implies that in many cases, the typical scale of the sampling unit that is used for most ecological surveys may be insufficient to satisfy the purpose of the study (Watson 2003). Instead, the sampling unit

should be comparable to the habitat area in which the characterisation is being inferred, thus area-specific information requires area-specific surveying (Mac Nally and Horrocks 2002).

This current study asks the question, do declining woodland bird species within a 15 km radius of the Para Woodland Reserve use different woodland types within this region?

This coarse-scale study of habitat use by birds uses ten hectare survey areas as the sampling unit within three different woodland types; being grassy woodland, open heath woodland and closed heath woodland habitats. In many cases across the study region, a particular habitat existed in a mosaic with other habitat types, and was often limited to an area of about ten hectares. Should different groups of birds use certain woodland types over others, then this level of coarse-scale habitat heterogeneity can be used to help inform broad-scale revegetation strategies.

The two chapters that follow use these same data, but at the finer 25 m × 25 m scale. Surveying the entire site at this scale should reveal the degrees and nature of any habitat heterogeneity, which is likely to be missed by using random samples or several long transect over such a large area (see Chapter 4). The output from Chapter 4 is then used to identify if certain bird species associated with particular habitat elements within each of the surveyed ten hectare sites (see Chapter 5).

Methods

Study area

This study was conducted within the north-central zone of the southern Mount Lofty Ranges, strategically within a 15 km radius of the Para Woodland Reserve (34°37.18'S, 138°47.01'E; Fig. 3.1). The survey areas were within the Para Wirra Recreation Park (34°41.25'S, 138°50.19'E) and adjacent SA Water property (34°40.09'S, 138°51.12'E), Hale Conservation Park (34°41.18'S, 138°54.33'E), Sandy Creek Conservation Park (34°36.24'S, 138°51.20'E) and remnant woodland at Altona (34°34.57'S, 138°54.30'E; Fig. 3.1).

Although regarded as remnant vegetation, these systems show evidence of human disturbance. For instance, the Para Wirra Recreation Park had previously been used for quarrying, mining, logging, farming and sheep grazing, prior to being proclaimed a protected reserve in 1962 (Mitchell 2000). Also since settlement, increased herbivory and trampling pressure by the Western Grey Kangaroo (*Macropus fuliginosus*) has dramatically impacted the understorey within the Parks at Para Wirra (Department of Environment 2005) and Sandy Creek (David Paton unpublished, David Lee, pers. comm.) and the woodland at Altona (Doreen Kies, pers. comm.).

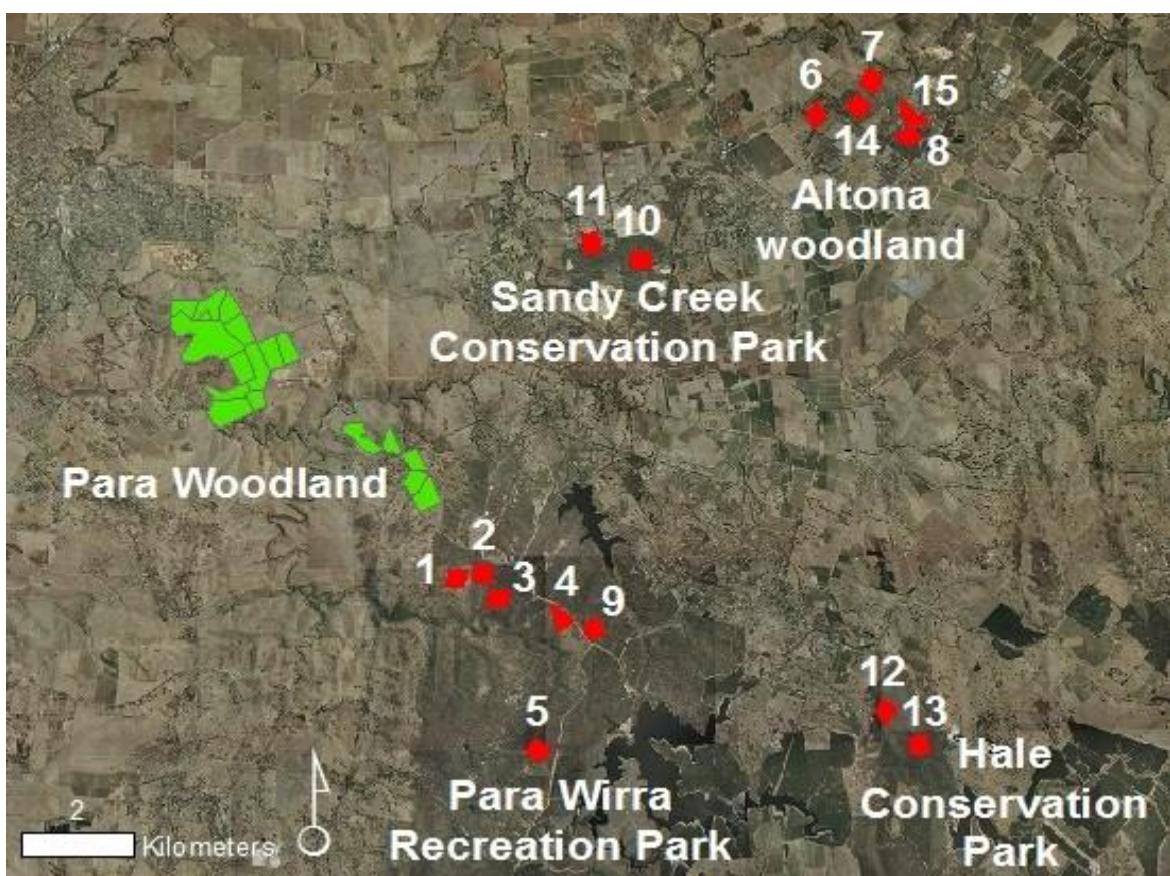


Figure 3.1: Survey sites across the north-central zone of the southern Mount Lofty Ranges (in red) within a 15 km radius of the Para Woodland Reserve (in green). Survey sites covered ten hectares each and were divided *a priori* equally between grassy woodland (Sites 1, 4, 6, 7 and 15), open heath woodland (Sites 5, 8, 10, 11 and 14) and closed heath woodland habitats (Sites 2, 3, 9, 12 and 13).

Ten hectare study sites

This study considered three habitat types, being grassy woodland, open heath woodland and closed heath woodland. Grassy woodland and open heath woodland habitats are thought to have once clothed the region that now incorporates the Para Woodland Reserve (Bickerton 2005). The remnant vegetation that is in the closest proximity to the Para Woodland Reserve however is the extensive stands of Long-leaved Box (*Eucalyptus goniocalyx*) and Stringybark closed heath woodland and open forest, which grow at higher elevations (refer Chapter 1).

To select the areas in which to conduct the surveys, I initially consulted vegetation maps of the region, which were produced by the South Australian Department of Environment, Water and Natural Resources. These maps were used to identify areas of remnant vegetation across the study region, and to detect changes in woodland types and their plant communities. I then conducted on-ground validation of potential survey sites, prior to selecting the habitat areas in which to study. This initial on-ground assessment involved confirming that the plant species present and their structures (e.g. storey heights and densities) matched the general descriptions given by the vegetation maps to be one of the three habitat types that were sought for this study. Distinction between the three habitat types was based on visual assessments of vegetation cover below four metres high. Open heath and closed heath woodland sites were categorised by the percent of heath cover, being the combined cover of shrubs and midstorey vegetation. Distinction between open heath and closed heath sites were visually determined using a threshold of 25% cover for vegetation within four metres of the ground. Grassy woodland sites lacked significant heathy shrub cover (e.g. < 2%), but varied in the amount of grass cover within each site. To avoid including a forest habitat within the selected woodland sites, the threshold for maximum estimated overstorey canopy cover was set at around 40% (Heard and Channon 1997).

The size of the survey area that was used covered ten hectares. Within this region specific “types of woodland” were often interspersed and contiguous patches of one woodland “type” was at times limited in area to around ten hectares. Also, within the Para Woodland Reserve the likely scale of revegetation works for any woodland “patch” is around the ten

hectare scale, as determined by the topography and soil types. Therefore the ten hectare scale seems to be an appropriate scale to use.

A total of 15 ten hectare sites were used in this survey, with five sites for each of the three woodland types. The shape of each ten hectare survey sites were square or as close to square as possible to incorporate a relatively homogeneous cover of either grassy woodland, open heath woodland or closed heath woodland habitat. The selected sites stand over high and low rainfall elevations. Within this region, high rainfall elevations are associated with topographic relief and are signified by extensive areas of Long-leaved Box and Stringybark systems that host the protected Parks at Para Wirra and Hale (Fig. 3.1). Areas of lower rainfall extend over the surrounding plains to the north and host the Para Woodland Reserve, together with residual Gum eucalypt systems that incorporate the Sandy Creek Conservation Park and Altona woodland (Fig. 3.1).

Where possible, survey sites within each of the three woodland types contained different plant communities. This was done for several reasons. Primarily it was to reduce any influence a dominant plant species may have had on the bird behaviours. Second, the variety of plant species incorporated a range of floristic covers that potentially may have once existed over the site of the Para Woodland Reserve (see Chapter 1). It should be noted however that Long-leaved Box and Stringybark species are not endemic to the lower lying reaches that now incorporates the Para Woodland Reserve, so these habitats will not be included when revegetating this Reserve area. Also to be noted, Peppermint Box (*E. odorata*) and Black Mallee-box (*E. porosa*) habitats were not sampled, due to their preferential clearing within this area (refer Chapter 1). This has resulted in these woodland systems now no longer being present within this region, except as small remnants that are not much greater than a hectare in area.

To help associate bird species with woodland type, study sites were also selected across a range of habitat and landscape settings. For instance, the grassy woodland sites included both high rainfall elevations (Sites 1 and 4) and low rainfall elevations (Sites 6, 7 and 15; Fig. 3.1). These sites included open woodland with few trees (Site 4) and woodland that was grazed by sheep (Site 7). Grassy Sites 1 and 4 were also embedded within surrounding closed heath woodland habitats.

For the open heath woodland sites, Site 5 was positioned within the high rainfall area, while Sites 8, 10, 11 and 14 were on the low rainfall plains (Fig. 3.1). Sites 8 and 14 were adjacent to nearby grassy woodland habitats at Altona, while Sites 10 and 11 were located some five kilometres away within the Sandy Creek Conservation Park (Fig. 3.1).

Closed heath woodland sites were all situated within the high rainfall areas (Fig. 3.1). Site 2 lay within an area considered to be the floristically richest part of the Para Wirra Recreation Park (Senior Ranger Erik Dahl, pers. comm.). Site 3 contained habitat areas which are best described as clogged heath (Crichton *et al.* 1978). Site 12 incorporated numerous undulating slopes radiating from several drainage depressions. Site 13 incorporated the eastern and western flanks on either side of a steep hill.

Habitat and vegetation surveys

Using ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006), a 25 m × 25 m grid was superimposed over each ten hectare site. Most ten hectare sites were uniform in shape (i.e. square), but some were irregular in shape to capture a relatively homogeneous cover of one of the three woodland types used in this study. For all 15 ten hectare sites and in particular for those sites with an irregular shape, only those 25 m × 25 m squares with their centroids falling within a site boundary were included in this survey. This resulted in 165 ± 0.9 , 25 m × 25 m survey squares per ten hectare site.

Each ten hectare site was characterised by collecting habitat data from every 25 m × 25 m square. Each site was surveyed at the scale of 25 m × 25 m for several reasons. Within each ten hectare site, natural variability in vegetation structure and floristics were expected. Thus, the data for each of the habitat attributes that were measured were averaged for the entire ten hectares, in an attempt to account for the expected within-site variability. The method is appropriate for the purpose of this study, as the aim is to measure bird species responses to woodland types at the coarse-scale of ten hectares.

The habitat and vegetation surveys were conducted between October and November 2008. The plant species and their abundances, together with the ground cover attributes that were contained within each of the 25 m × 25 m squares were identified using a rapid assessment

of that square. These rapid assessments were performed by initially walking the perimeter of the square to gain an overall view of its contents. Then using the perspective from the centre of that square, visual cover estimates were made based on what had been identified during the mobile assessment and from what was being observed from the central position. These estimates included the vegetation cover at three storey levels, together with a range of ground cover substrates (see below for details). From the central position of a 25 m × 25 m square, the vegetation and ground cover only needed to be assessed to a distance of 12.5 m in any one direction.

Woody plants with their trunks within the 25 m × 25 m square being surveyed were categorised as shrub (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m). Within each square, the percent cover for each of these three strata levels, together with the percent cover of grasses, weeds and herbs, were estimated individually. The percent of cover was assessed as the overall percentage of the surface area of the square. The contribution that individual plant species made to the total percent cover within each stratum was then estimated by eye as a percent of cover for each species, except for overstorey trees. As the canopies of different tree species often overlapped, the numbers belonging to each tree species were instead counted. The proportion of 25 m × 25 m survey squares in which an overstorey species was recorded within a ten hectare site was used to indicate the dominant overstorey species for each of the 15 ten hectare sites.

Also based on the mobile assessment and observations made from the centre of each square, ground cover attributes other than standing vegetation were visually estimated. These were the percent cover of bare ground (areas > 20 cm radius), light litter (litter that did not fully cover the ground surface, but was associated with any other ground cover attribute), heavy litter (litter that fully covered the ground surface), fallen timber, lichen / moss, grasses (both native and introduced), weeds (other than exotic grasses) and herbs. Litter cover was segregated into two categories to distinguish between vegetation litter that had accumulated directly beneath a tree, with vegetation litter that had been wind-blown and strewn over wider areas. The aim here was to see if either a thick or sparse litter cover may have had an influence on the areas in which certain bird species were detected within a survey site.

Bird surveys

Systematic area searches were undertaken across each ten hectare site for all terrestrial bird species. Systematic area searching involved a series of parallel transects the length of the survey site across the entire survey area. In this study, transects were approximately 50 m apart. To help in achieving this, each site was navigated using a GPS to ensure that the site was uniformly searched. A Garmin Colorado 300 GPS with a downloaded map of the 25 m × 25 m grids for each of the 15 sites was used for navigation. The transects across a site acted as a guide only, with the surveyor afforded freedom in an effort to record all the birds that were present at the time of the survey. This included returning to previously searched areas if a new bird and/or species was observed. The actual position of each individual bird or conspecific group of birds that were detected was recorded using a second hand-held GPS (Garmin 60). This involved moving to the exact location that the bird/s were first observed to record that location with a GPS waypoint. By using this method, the exact bird locations could be compared with the underlying habitat at the scale of 25 m × 25 m (Chapters 4 and 5). The bird surveys for the 15 ten hectare sites took on average 2.6 ± 0.1 hours each (\pm standard error in hours). The small variance in the survey times is attributed to the terrain within some sites being easier to negotiate than others.

The number of individual birds and their predominant behaviour was recorded (e.g. forage, call, perch, preen, chase, roost, breed (see details below), fly within site, and fly over site). The behaviours that were recorded captured what the birds were predominantly doing when first observed. For birds belonging to the same species within a group, the single most predominant behaviour displayed by that group was recorded. All behavioural observations were incidental, with no specific behaviours actively searched for. Breeding records included breeding displays, courtship feeding, mating, nest building, brooding on nest with eggs, brooding on nest with nestlings, and feeding dependent fledglings. Particular emphasis was placed on identifying the plant species that the birds used for nest placement (see Appendix B) and the resources that were used by the birds for foraging (see Appendix C). Also included was the average foraging height per waypoint position (estimated to the nearest 0.5 m).

Critical to each survey was the need to avoid recording the same bird twice as separate individuals, thus biasing the results. To minimise the potential for this occurring, a

conservative approach was adopted for when it was considered that a new bird had been encountered. For example, for species that may congregate in groups (such as the New Holland Honeyeater, *Phylidonyris novaehollandiae*), unless the whereabouts of any previously recorded birds were known, only the number of birds that were “extra” to the previous count was recorded. For birds which called frequently, such as the Rufous Whistler (*Pachycephala rufiventris*), the numbers of birds (presumed to be males) were counted based on their calling locations and then actively searched out along the survey transects to obtain GPS fixes. For sexually dimorphic species like robins, a bird was only counted if it was a different sex to a previously recorded conspecific. If the bird was the same sex as a previously recorded bird, it was only recorded if the other bird could be accounted for.

The bird surveys were conducted between September and October 2008, which coincided with habitat use by these birds during the breeding season (see Appendix B). The birds were counted between dawn and midday, with two to three ten hectare sites scored per day. It has been shown that two to three visits is generally adequate to identify the occupants within a particular site, unless the occupancy is very high or detectability is low (Field *et al.* 2005a). As several of the ten hectare sites used in this study were difficult to negotiate due to the terrain and/or vegetation, the sites were visited in rotation a total of five times. Multiple surveys that are conducted within a particular site over different days yield more species than same-day surveys (Field *et al.* 2005b). Accordingly, the five surveys for each of the 15 sites were conducted on different days.

The objective of this particular study was to document bird species that associated with at least one of the three woodland types being surveyed. As birds are known to alter their activity during the day (e.g. Bibby *et al.* 2000; Timewell and Mac Nally 2004), it was considered important to avoid introducing a potential search bias by constantly surveying the same habitat area at the same time but on different days. Accordingly, the five surveys for each of the sites were conducted on different days, starting at different times. I also started each survey from a different start location and performed the surveys in different directions. This was aimed to avoid any potential search bias if the surveys were all started at the same time of day, same start location and used the same search path to cover the ten hectare area.

Analyses

The level of significance for the tests used for this study and for all tests in the studies that follow in the ensuing chapters was set at alpha ≤ 0.05 . All values are reported as mean \pm standard error (SE) throughout this thesis.

Data for all of the habitat attributes that were measured and the bird species that were recorded within each of the ten hectare sites were analysed separately using hierarchical cluster analysis in dendograms using PC-ORD5 (McCune and Mefford 1999b). This produced two dendograms, one illustrating the associations between the ten hectare sites based upon habitat and vegetation characteristics and the other based upon the composition of the occupying bird community (excluding birds that flew through and/or over a site).

Hierarchical clustering used the space-conserving Flexible Beta linkage method (using the default β of -0.25) with Sorenson (Bray-Curtis) distance measures. This method avoids distortion of space in the vicinity of the clustered groups and has less propensity to chain clusters together, while a Beta value below zero helps produce strong definition between the clusters (McCune and Grace 2002).

The “Sites by Habitat” dendrogram used the untransformed habitat and vegetation data from the 25 m \times 25 m squares, which were averaged for each of the ten hectare sites. The data used were the 11 ground cover attributes that were estimated, together with the canopy cover estimates for shrubs, midstorey and overstorey vegetation. The untransformed data used to produce the “Sites by Birds” dendrogram used a modified “birds ha $^{-1}$ ” dataset. Conducting multiple bird surveys over the same site risks a high potential for individual birds to be recorded more than once during the survey period. To adjust for this, for each ten hectare site the average birds per hectare were calculated for each species across the five surveys. The mean and standard error from the five surveys is then reported for each species within each site. These data included all the birds that were recorded using habitat within the survey areas, but excludes those which merely flew over or through a survey site.

The cluster groups within both dendograms were initially defined qualitatively as those groups which naturally clustered together and were separated from other clusters by long

stems in the dendrogram. Non-parametric Multi-response Permutation Procedures (MRPP) with Sørensen (Bray-Curtis) distance measures (in PC-ORD5) were used to test the null hypothesis that no differences existed among the clusters that formed in these analyses. In these analyses, the test statistic T describes the heterogeneity between the cluster groups (heterogeneity increasing with decreasing T), the agreement statistic A denotes within-group homogeneity (with $A > 0.3$ indicating high homogeneity; McCune and Grace 2002), while delta p evaluated if the observed differences were significantly different from random.

Two diagnostic techniques were used to quantitatively characterise the differences between the clustered groups within the two dendograms. These were Discriminant Analysis (DA) for the Sites by Habitat dendrogram and Indicator Species Analysis (ISA) for the Sites by Birds dendrogram. For the habitat data, DA with stepwise entry of the variables was used using JMP10 (2012). By implementing stepwise entry, the univariate F -ratio can be used to indicate levels of significance as the variables are added to the model. Discriminant Analysis was used to test the accuracy of the cluster classifications to determine which (if any) of the habitat variables contributed most to the separation of the woodland types in the dendrogram (i.e. variables that discriminated well). This procedure is mathematically similar to multivariate analysis of variance, but instead uses the habitat attributes as independent variables to predict association with the *a priori* dependent groups (i.e. the three woodland types).

Regularized DA (RDA) was applied as this method is a compromise between Linear DA and Quadratic DA. Linear and Quadratic DA analyses were not considered as the first assumes the independent variables are normally distributed with the same variance and covariance, while the latter suffers in small data sets because it has insufficient data to create functionally invertible and stable covariance matrices. With the RDA method, the user sets two regularisation parameters in the estimation of the covariance matrices, with both using values between zero and one. The first parameter (Lambda, λ) controls the degree of shrinkage to mix the individual and group covariance matrices. A low Lambda should be used when variables are correlated (SAS Institute Inc. 2009). As the ground cover variables in this study are highly correlated, λ was set at zero. The second parameter (Gamma, γ) controls the deflation of the covariance across the variables. Here γ was set at

zero to maintain minimal shrinkage of the covariance across the variables (SAS Institute Inc. 2009).

The value calculated for a ten hectare site that is then used to assign that site to a cluster group comprises of a probability density, which carries a margin of error. As such, a repeated analysis on these same data will not produce the exact same value for each site. This runs the risk that a site may be assigned to a different cluster group, should the analysis be re-run using these same data. An assessment is required to assess the risk that a site had been misclassified and incorrectly assigned to the wrong cluster group. In DA, the misclassification risk is computed from the canonical correlation (one minus the canonical correlation²) and is given by the Wilks' Lambda. Wilks' Lambda portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the groups. It ranges from zero (perfect separation of groups) to one (no separation of groups) but by itself is not a measure of statistical significance. Wilks' Lambda is tested statistically using chi-square against the null hypothesis that the means of all the independent variables are equal across the groups of dependent variables, that is they have no discriminatory ability. The DA method in JMP10 uses the approximation to the *F* distribution. Accordingly, the "Approx. *F*" statistic and the associated *p*-value (Prob > *F*) that is given to the right of Wilkes' Lamdba in the output correspond to the test of interest, with the interpretation being the same as if the chi-squared approximation was used.

In DA, the standardised discriminant function coefficient has been standardised to unit variance for each habitat variable. Therefore, these coefficients can be used to directly interpret the importance of each of the habitat variables in contributing to the discriminant function, with larger values having greater ability to discriminate. These coefficients are absolute values and their signs indicate the direction of the relationship. The combination of group means, univariate *F*-ratios and the standardized discriminant function coefficient may be used to describe the most influential habitat variables behind the classification of the woodland type groups in this study (with $A > 0.3$ indicating high homogeneity; McCune and Grace 2002).

For the bird dendrogram, ISA using PC-ORD is considered ideal for identifying species that can be used to differentiate between the clustered groups (McCune and Grace 2002). This procedure calculates species indicator values, following the methods proposed by

Dufrene (1997) and described in detail by McCune and Grace (2002). This analysis first calculates the relative abundance of a species in a particular cluster relative to its abundance in all clusters. It then calculates the relative frequency of that species in each cluster. Multiplying these two indicator terms together yields an indicator value. The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species is statistically tested using a Monte Carlo test based on 1,000 randomisations. The resultant *p*-values indicate the proportion of permutations that had maximum indicator values that were equal to or higher than the maximum observed indicator values (using $\alpha \leq 0.05$).

To relate the dendrogram arrangements of woodland types based on both the measured habitat variables and occupying bird species, joint plot overlays were constructed for both in PC-ORD5. These overlays used the default cut-off coefficient of determination value of 0.20 to create vectors that indicated which measured variables (if any) best explained a portion of the variation represented by a dendrogram axis. The orientation and length of the vector indicates the direction and strength of the relationship between that variable and the distribution of the woodland sites within ordination space. To gauge the quality of the resulting Non-metric Multidimensional Scaling (NMS) ordination, an after-the-fact evaluation of the variance represented by each axis was performed. This evaluation compares the quality of the data reduction that was used to manipulate the data into ordination space, by comparing the differences in the distance between points in the ordination diagram with the distances in the original, unreduced space (with $A > 0.3$ indicating high homogeneity; McCune and Grace 2002). How well these distances compare is given as a percent for each primary axis in the ordination, with their sum being the percent of the variation that is represented by the distribution of the woodland types within the ordination plot (McCune and Mefford 1999a). The reported “stress” level is the measure of the distortion applied to produce a low-dimensional representation from a high-dimensional data matrix (McCune and Mefford 1999a). Stress of $< 5\%$ represents high reliability of the two-dimensional interpretation, stress of between 5 and 10% allows for high confidence in the interpretation without the risk of drawing false inferences, while

stress approaching 20% increases the potential to produce misleading results (Clarke 1993).

Results

Habitat assessment

The five grassy woodland sites were floristically varied with a total of five dominant tree species (Table 3.1). Notably, Peppermint Box was co-dominant with a Gum eucalypt within Site 7; and Slender Cypress-pine (*Callitris gracilis*) was co-dominant with Gum eucalypts within Site 6 (Table 3.1).

The five open heath woodland sites were also floristically varied with a total of seven dominant tree species (Table 3.2). Notably, co-dominant with Gum eucalypts were Drooping Sheoak (*Allocasuarina verticillata*) within Sites 10, 11 and 14; Golden Wattle (*Acacia pycnantha*) within Sites 8 and 14; Long-leaved Box within Site 5; Peppermint Box within Site 11; and Slender Cypress-pine within Sites 8, 10, 11 and 14 (Table 3.2).

The five closed heath woodland sites were also floristically varied with a total of eight dominant tree species (Table 3.3). Notably, Sites 2, 3 and 9 within the Para Wirra area were dominated by Box and Gum eucalypts, together with Golden Wattle within Site 3 and Slender Cypress-pine within Sites 2 and 9 (Table 3.3). The Peppermint Box within Sites 2 and 3 was a low growing “mallee form” that grew to around two metres high. In contrast, Sites 12 and 13 within the Hale Conservation Park was dominated by Long-leaved Box, Messmate Stringybark (*E. obliqua*), and Pink Gum (*E. fasciculosa*); while Site 13 was also dominated by Oyster Bay Pine (*C. rhomboidea*; Table 3.3).

Table 3.1: Five ten hectare survey sites classified *a priori* as grassy woodland within the north-central zone of the southern Mount Lofty Ranges. Sites with a relatively grassy understorey and minimal heath cover below 4 m (e.g. < 2%) were selected. The five sites were then chosen to be floristically varied from one another at the overstorey level. Site 4 was selected to represent open woodland with few trees. The values represent the proportion that each plant species was recorded within each of the 25 m × 25 m survey squares that covered the entire ten hectare survey site (average 165 ± 0.9 squares per site). The figures in bold depict the dominant plant species within each of the ten hectare sites.

| Common name | Species name | Site 1 | Site 4 | Site 6 | Site 7 | Site 15 |
|----------------------|-----------------------------------|-------------|--------|-------------|-------------|-------------|
| Drooping Sheoak | <i>Allocasuarina verticillata</i> | — | — | 0.03 | 0.01 | 0.13 |
| Golden Wattle | <i>Acacia pycnantha</i> | — | 0.01 | 0.07 | — | 0.07 |
| Peppermint Box | <i>Eucalyptus odorata</i> | — | — | — | 0.52 | 0.10 |
| Pink Gum | <i>Eucalyptus fasciculosa</i> | 0.04 | 0.04 | 0.39 | 0.01 | 0.09 |
| River Red-gum | <i>Eucalyptus camaldulensis</i> | 0.01 | 0.04 | 0.09 | — | 0.43 |
| Silver Banksia | <i>Banksia marginata</i> | — | — | — | — | 0.01 |
| Slender Cypress-pine | <i>Callitris gracilis</i> | 0.01 | — | 0.68 | — | 0.18 |
| SA Blue Gum | <i>Eucalyptus leucoxylon</i> | 0.89 | 0.08 | 0.24 | 0.29 | 0.22 |
| Swamp Wattle | <i>Acacia retinodes</i> | — | — | 0.02 | — | 0.07 |

Table 3.2: Five ten hectare survey sites classified *a priori* as open heath woodland within the north-central zone of the southern Mount Lofty Ranges. Sites with an open heath understorey were selected (< 25% cover below 4m). The five sites were then chosen to be floristically varied from one another at the overstorey level. The values represent the proportion that each plant species was recorded within each of the 25 m × 25 m survey squares that covered the entire ten hectare survey site (average 165 ± 0.9 squares per site). The figures in bold depict the dominant plant species within each of the ten hectare sites.

| Common name | Species name | Site 5 | Site 8 | Site 10 | Site 11 | Site 14 |
|----------------------|-----------------------------------|-------------|-------------|-------------|-------------|-------------|
| Drooping Sheoak | <i>Allocasuarina verticillata</i> | — | 0.14 | 0.46 | 0.60 | 0.20 |
| Golden Wattle | <i>Acacia pycnantha</i> | 0.03 | 0.27 | 0.02 | 0.02 | 0.21 |
| Long-leaved Box | <i>Eucalyptus goniocalyx</i> | 0.50 | — | — | — | — |
| Peppermint Box | <i>Eucalyptus odorata</i> | — | 0.01 | — | 0.44 | — |
| Pink Gum | <i>Eucalyptus fasciculosa</i> | 0.41 | 0.68 | 0.79 | 0.58 | 0.39 |
| River Red-gum | <i>Eucalyptus camaldulensis</i> | — | 0.01 | — | — | 0.15 |
| Silver Banksia | <i>Banksia marginata</i> | — | 0.01 | 0.02 | — | — |
| Slender Cypress-pine | <i>Callitris gracilis</i> | 0.05 | 0.98 | 0.93 | 0.93 | 0.92 |
| SA Blue Gum | <i>Eucalyptus leucoxylon</i> | 0.62 | 0.11 | — | 0.10 | 0.21 |
| Swamp Wattle | <i>Acacia retinodes</i> | 0.09 | — | — | — | 0.04 |

Table 3.3: Five ten hectare survey sites classified *a priori* as closed heath woodland within the north-central zone of the southern Mount Lofty Ranges. Sites with a closed heath understorey were selected ($> 25\%$ cover below 4m). The five sites were then chosen to be floristically varied from one another at the overstorey level. The values represent the proportion that each plant species was recorded within each of the 25 m \times 25 m survey squares that covered the entire ten hectare survey site (average 165 ± 0.9 squares per site). The figures in bold depict the dominant plant species within each of the ten hectare sites.

| Common name | Species name | Site 2 | Site 3 | Site 9 | Site 12 | Site 13 |
|----------------------|-----------------------------------|-------------|-------------|-------------|-------------|-------------|
| Drooping Sheoak | <i>Allocasuarina verticillata</i> | 0.18 | 0.15 | 0.14 | – | – |
| Golden Wattle | <i>Acacia pycnantha</i> | 0.15 | 0.20 | 0.13 | 0.07 | 0.08 |
| Long-leaved Box | <i>Eucalyptus goniocalyx</i> | 0.71 | – | 0.35 | 0.80 | 0.79 |
| Messmate Stringybark | <i>Eucalyptus obliqua</i> | – | – | – | 0.34 | 0.95 |
| Native Cherry | <i>Exocarpos cupressiformis</i> | – | 0.01 | 0.01 | 0.01 | 0.01 |
| Oyster Bay Pine | <i>Callitris rhomboidea</i> | – | – | – | 0.08 | 0.44 |
| Peppermint Box | <i>Eucalyptus odorata</i> | 0.20 | 0.48 | – | – | – |
| Pink Gum | <i>Eucalyptus fasciculosa</i> | 0.69 | 0.31 | 0.72 | 0.34 | 0.30 |
| Slender Cypress-pine | <i>Callitris gracilis</i> | 0.43 | 0.15 | 0.61 | – | – |
| SA Blue Gum | <i>Eucalyptus leucoxylon</i> | 0.17 | 0.30 | 0.21 | 0.03 | – |

The cluster analysis undertaken on habitat attributes generally confirmed the *a priori* visual classification of the 15 sites to vegetation type, producing grassy woodland, open heath woodland and closed heath woodland clusters (Fig. 3.2). There was clear separation between the clusters ($T = -5.58$) and strong within-group homogeneity ($A = 0.44$) that was unlikely to be due to chance ($\text{delta } p = < 0.0001$). However, some sites were placed in groups different from those expected. Two sites surveyed as grassy woodland formed a fourth cluster, being open woodland with few trees ($11.6 \text{ trees ha}^{-1}$; Site 4) and grazed woodland (Site 7). In addition, Site 5 (classified *a priori* as open heath woodland) contained enough other plant species to align within the closed heath woodland cluster.

Based on the habitat attributes that were measured, the grassy woodland cluster was more similar to the open heath woodland cluster, while both were less similar to the closed heath woodland cluster (Fig. 3.2). The open woodland with few trees and grazed woodland cluster was an outlier (Fig 3.2).

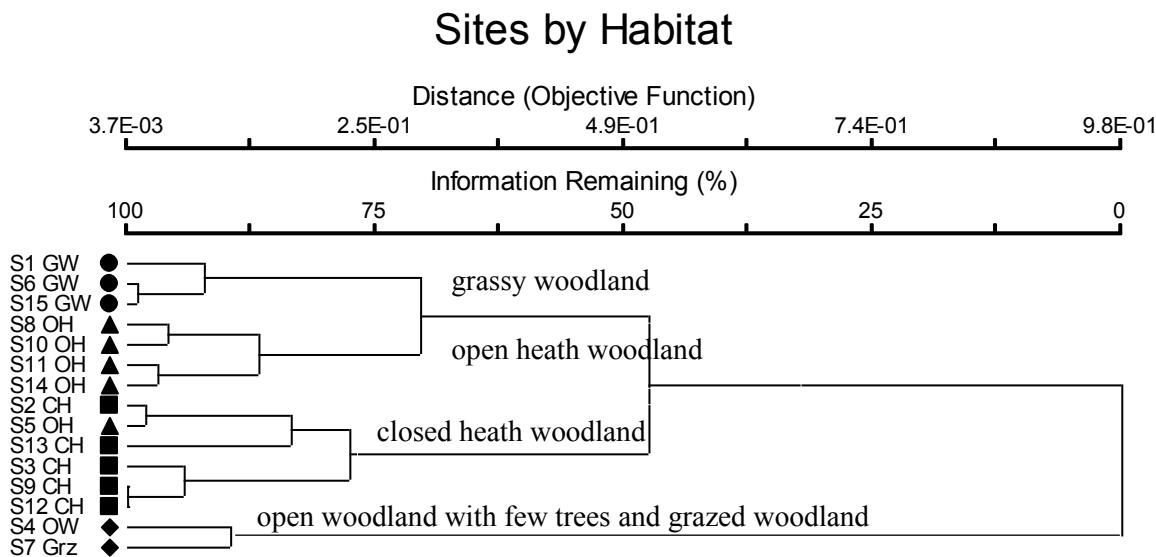


Figure 3.2: Cluster analysis dendrogram of 15 ten hectare survey sites based upon habitat attributes. Clustering revealed that Sites 4 (S4) and 7 (S7) that were classified *a priori* as grassy woodland (GW), formed a separate cluster (open woodland with few trees (OW) and grazed woodland (Grz)). Site 5 (S5) classified *a priori* as open heath woodland (OH) aligned within the closed heath woodland (CH) cluster. These data were collected from locations within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected October – November 2008.

Regularized Discrimination Analysis applied to the four clusters that were created in the Sites by Habitat dendrogram (Fig. 3.2) identified three functions, the first of which accounted for 99.96% of the variance (Table 3.4). Group associations of the 15 sites were perfectly predicted for the dendrogram clusters to which they were associated, with 100% prediction probability and no misclassifications (Wilks' $\lambda = 4.0e^{-7}$; Table 3.5). This result was highly significant ($\chi^2_{96, 0.05} = 16.29, p = 0.01$; Table 3.5). This indicates that a strong relationship existed between the woodland types and the habitat variables that were used to differentiate them.

The variables with the strongest discriminatory value were grasses and weeds for the open woodland with few trees and grazed woodland cluster (Table 3.6; Fig. 3.3). Fallen timber and lichen / moss were the strongest discriminates for the open heath woodland cluster (Table 3.6; Fig. 3.3). For the closed heath woodland cluster it was light litter, shrub cover and midstorey cover (Table 3.6; Fig. 3.3). The open woodland with few trees and grazed woodland cluster was distinct from the grassy woodland cluster due to a greater cover of grasses and weeds (Table 3.6). These two clusters also contained different combinations of dominant grass species (Table 3.7) and weed species (Table 3.8). The variables with the

least discriminatory value to help explain the separation of the four clusters were bare ground and overstorey cover (Table 3.6; Fig. 3.3).

As the sites varied from being more open woodland to containing greater heath vegetation, the percent cover and number of shrub species increased (Tables 3.6 and 3.9). This culminated in the shrub component for closed heath woodland sites being floristically richer per 25 m × 25 m square, compared to both grassy woodland and open heath woodland sites (Table 3.9). In comparison, closed heath woodland sites also had the highest midstorey cover (Table 3.6) and more plant species within the midstorey strata (Table 3.10). The midstorey component within grassy woodland and open heath woodland sites was similar, with both dominated by Golden Wattle, Kangaroo Thorn (*Acacia paradoxa*) and groves of developing Slender Cypress-pine (Tables 3.6 and 3.10).

The grassy, open heath and closed heath woodland clusters all had similar amounts of overstorey cover (around 30 – 40%; Table 3.6). The lowest overstorey cover was evident within the open woodland with few trees and grazed woodland cluster (20%; Table 3.6). All four cluster types contained between seven to nine tree species, although the species and/or their proportions were different (Table 3.11). All 15 ten hectare sites contained a different number of trees per hectare, with varying combinations of tree species that dominated within each site (Table 3.12).

The closed heath cluster contained two discrete groups (Figs 3.2 and 3.3), which were evenly divided into Sites 2, 5 and 13 (Group A) and Sites 3, 9 and 12 (Group B). The differences were in the proportions of cover at the different storey levels. Group A contained more shrub and overstorey cover, while Group B contained more midstorey cover (Fig. 3.4).

Table 3.4: Eigenvalues for the three functions identified during Regularized Discrimination Analysis (JMP10 2012). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first function accounting for 99.96% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 40078.2 | 99.96 | 99.96 | 0.999 |
| 2 | 13.6 | 0.03 | 99.99 | 0.965 |
| 3 | 3.3 | 0.01 | 100.00 | 0.875 |

Table 3.5: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the eleven habitat variables used to help interpret the woodland type group classifications. Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the woodland type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance ($\text{Prob} > F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value ($\text{Prob} > F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis the first function was significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob > F |
|-------------------|---------------|-------------|-------|-------|----|------------|
| 1 | $4.0e^{-7}$ | 16.29 | 33 | 4 | 96 | 0.01 |
| 2 | 0.02 | 1.38 | 20 | 4 | 57 | 0.42 |
| 3 | 0.24 | 1.08 | 9 | 3 | 16 | 0.53 |

Table 3.6: Average percent cover of habitat attributes within four habitat types; open woodland with few trees and grazed woodland (OGW), grassy woodland (GW), open heath woodland (OHW) and closed heath woodland (CHW). Regularized Discrimination Analysis (RDA) with stepwise entry of the eleven habitat variables was used to help interpret the woodland type group classifications. The univariate *F*-ratio can be used to indicate levels of significance as variables are added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, *F*-ratio and function coefficient. Attributes with the strongest discriminatory values are in bold. Surveys were conducted across 15 ten hectare sites within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Each site was divided into 25 m × 25 m squares that covered the entire ten hectare area and were surveyed individually (165 ± 0.9 squares per site). Values are the average percent cover for each habitat variable (mean ± SE) within the corresponding survey squares for each woodland type (dominant attributes are in bold). *n* = number of ten hectare sites. Data collected October – November 2008.

| Habitat attribute | OGW (<i>n</i> = 2) | GW (<i>n</i> = 3) | OHW (<i>n</i> = 4) | CHW (<i>n</i> = 6) | Univariate <i>F</i> -ratio | Standardized discriminant function coefficient |
|-------------------|----------------------------------|-----------------------|---------------------------------|----------------------------------|-------------------------------|---|
| Bare ground | 2.9 ± 0.3 | 6.7 ± 2.6 | 8.1 ± 0.3 | 3.3 ± 0.2 | 5.72 | 0.8 |
| Lichen / Moss | 0.9 ± 0.2 | 2.5 ± 0.2 | 5.8 ± 0.3 | 1.4 ± 0.1 | 3.99 | -71.6 |
| Light litter | 5.0 ± 3.6 | 14.4 ± 0.6 | 28.3 ± 0.7 | 41.1 ± 0.7 | 11.11 | -10.8 |
| Full litter | 0.2 ± 0.1 | 3.9 ± 0.4 | 8.2 ± 0.4 | 10.1 ± 0.6 | 1.95 | -32.4 |
| Fallen timber | 1.1 ± 0.1 | 4.6 ± 0.3 | 6.5 ± 0.2 | 2.7 ± 0.1 | 5.79 | -86.3 |
| Grasses | 60.2 ± 0.8 | 49.3 ± 0.9 | 24.3 ± 0.7 | 1.2 ± 0.1 | 47.75 | -74.3 |
| Weeds | 25.1 ± 0.9 | 6.2 ± 0.3 | 0.9 ± 0.1 | 0.1 ± 0.05 | 67.35 | -17.5 |
| Herbs | 0.8 ± 0.1 | 7.2 ± 0.3 | 7.2 ± 0.2 | 4.7 ± 0.2 | 6.31 | 74.2 |
| Shrub canopy | 3.2 ± 0.6 | 4.6 ± 0.4 | 10.7 ± 0.5 | 32.8 ± 0.7 | 9.66 | -50.4 |
| Midstorey canopy | 2.3 ± 0.4 | 19.4 ± 0.9 | 18.5 ± 0.7 | 34.1 ± 0.8 | 3.66 | -50.5 |
| Overstorey canopy | 20.0 ± 1.5 | 32.5 ± 1.2 | 43.3 ± 0.9 | 41.5 ± 0.8 | 2.64 | 13.2 |

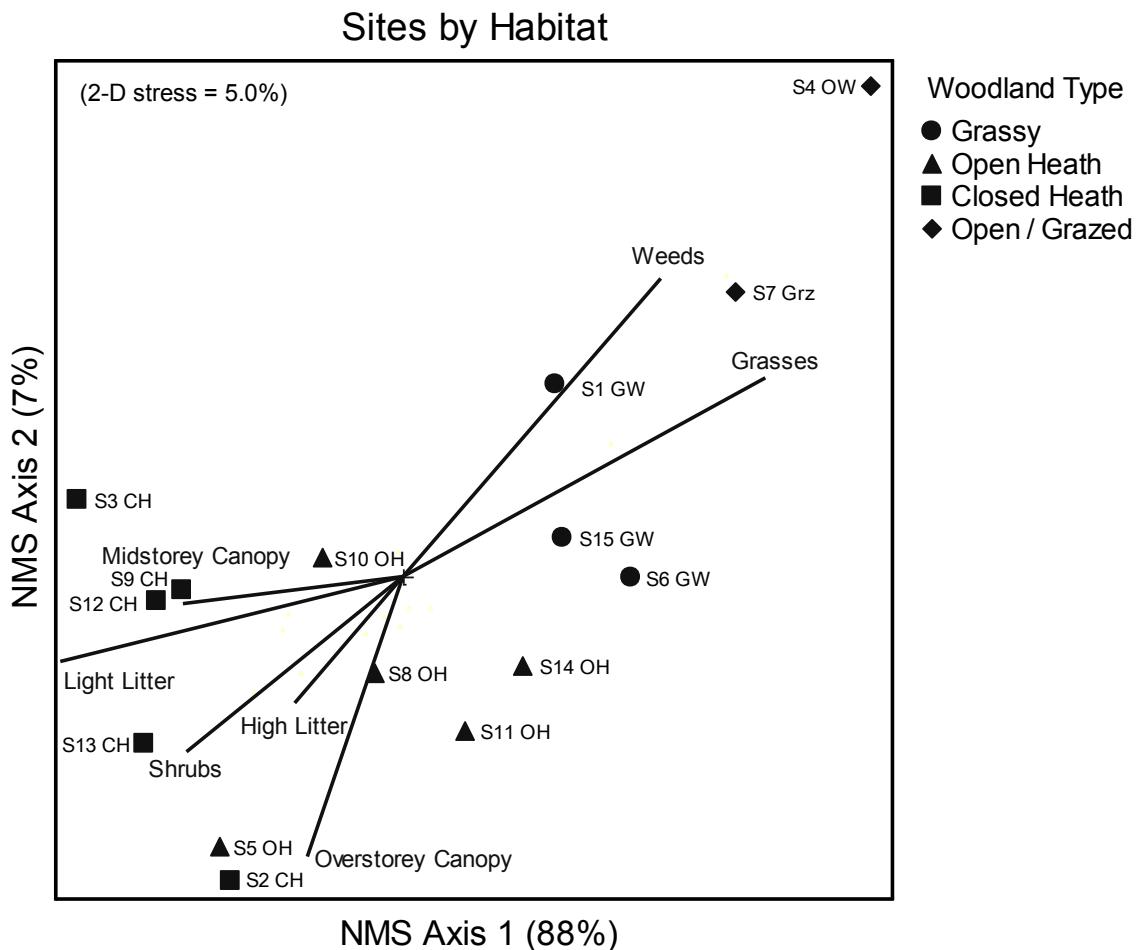


Figure 3.3: Two-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure, for vegetation features in respect to habitat types. The 15 sites are divided between grassy woodland (Grassy), open heath woodland (Open Heath), closed heath woodland (Closed Heath) and open woodland with few trees and grazed woodland (Open / Grazed). The habitat variables shown have strong correlation with the ordination. The habitat variable vector orientation indicates the direction of positive increase and the length represents the strength of association with the different woodland types. At a stress level of 5.0%, 95% of the variation is represented by the distribution of the woodland types. These data were collected from locations within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected October – November 2008.

Table 3.7: Grass species recorded within ten hectare survey sites; being open woodland with few trees and grazed woodland (OGW), grassy woodland (GW), open heath woodland (OHW) and closed heath woodland (CHW). Each ten hectare site was segregated into 25 m × 25 m squares, producing 165 ± 0.9 (mean ± SE) squares per site. Survey sites were located within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Values are the average percent cover for each species (mean ± SE) within the corresponding survey squares for each woodland type (dominant species in bold). n = number of ten hectare sites. Data collected October – November 2008.

| Grasses | | OGW (n = 2) | GW (n = 3) | OHW (n = 4) | CHW (n = 6) |
|--|--------------------------------|----------------------------------|----------------------------------|---------------------------------|------------------------|
| Common name | Species name | % cover | % cover | % cover | % cover |
| # grass species | | 17 | 15 | 12 | 5 |
| # grass species per 25 m x 25 m square | | 3.5 ± 0.1 | 3.2 ± 0.1 | 3.3 ± 0.04 | 0.5 ± 0.03 |
| Annual Veldt-grass | <i>Ehrharta longiflora</i> | 0.1 ± 0.1 | 0.1 ± 0.1 | – | – |
| Avellinia | <i>Avellinia michelii</i> | 2.3 ± 0.4 | 4.4 ± 0.4 | 7.6 ± 0.3 | 0.04 ± 0.02 |
| Barley Grass | <i>Hordeum leporinum</i> | 3.4 ± 0.5 | 0.02 ± 0.01 | – | – |
| Bearded Oat | <i>Avena barbata</i> | 16.6 ± 0.8 | 11.2 ± 0.7 | 1.5 ± 0.1 | – |
| Canary Grass | <i>Phalaris</i> spp | 0.1 ± 0.1 | 0.2 ± 0.1 | – | – |
| False Hair-grass | <i>Pentaschistis airoides</i> | 0.2 ± 0.1 | 1.6 ± 0.2 | 0.4 ± 0.1 | – |
| Fox-tail Mulga-grass | <i>Neurachne alopecuroidea</i> | 0.2 ± 0.1 | 0.3 ± 0.1 | 0.1 ± 0.03 | 0.1 ± 0.02 |
| Hares Tail Grass | <i>Lagurus ovatus</i> | 0.2 ± 0.1 | 1.8 ± 0.2 | 0.1 ± 0.04 | – |
| Large Quaking-grass | <i>Briza maxima</i> | 0.2 ± 0.1 | 8.8 ± 0.6 | 3.1 ± 0.2 | 0.1 ± 0.03 |
| Lesser Quaking-grass | <i>Briza minor</i> | 0.1 ± 0.1 | 0.1 ± 0.05 | 0.3 ± 0.1 | 0.02 ± 0.01 |
| Madrid Brome | <i>Bromus madritensis</i> | 0.4 ± 0.1 | – | – | – |
| Native Wheat-grass | <i>Elymus scaber</i> | 13.5 ± 0.7 | 0.9 ± 0.2 | – | – |
| Perennial Veldt-grass | <i>Ehrharta calycina</i> | 3.3 ± 0.4 | 7.7 ± 0.6 | 1.5 ± 0.1 | – |
| Red Brome | <i>Bromus rubens</i> | 0.4 ± 0.1 | 1.2 ± 0.3 | 0.1 ± 0.02 | – |
| Silvery Hair-grass | <i>Aira caryophyllea</i> | 6.5 ± 0.6 | 8.7 ± 0.5 | 8.7 ± 0.3 | 0.4 ± 0.1 |
| Speargrass | <i>Austrostipa</i> sp. | 2.5 ± 0.3 | 1.3 ± 0.2 | 0.5 ± 0.1 | – |
| Squirrel-tail Grass | <i>Vulpia bromoides</i> | 9.1 ± 0.7 | – | 0.1 ± 0.03 | – |

Table 3.8: Weed species recorded within ten hectare survey sites; being open woodland with few trees and grazed woodland (OGW), grassy woodland (GW), open heath woodland (OHW) and closed heath woodland (CHW). Each ten hectare site was segregated into 25 m × 25 m squares, producing 165 ± 0.9 (mean ± SE) squares per site. Survey sites were located within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Values are the average percent cover for each species (mean ± SE) within the corresponding survey squares for each woodland type (dominant species in bold). n = number of ten hectare sites. Data collected October – November 2008.

| Weeds | | OGW (n = 2) | GW (n = 3) | OHW (n = 4) | CHW (n = 6) |
|---------------------------------------|------------------------------------|----------------------------------|-----------------|-----------------|------------------|
| # weed species | | 9 | 11 | 4 | 3 |
| # weed species per 25 m x 25 m square | | 2.5 ± 0.1 | 1.5 ± 0.05 | 0.5 ± 0.03 | 0.02 ± 0.01 |
| Common name | Species name | % cover | % cover | % cover | % cover |
| Artichoke Thistle | <i>Cynara cardunculus</i> | 0.01 ± 0.01 | 0.1 ± 0.03 | – | – |
| Boneseed | <i>Chrysanthemoides monilifera</i> | – | 0.2 ± 0.1 | 0.1 ± 0.04 | – |
| Bridal Creeper | <i>Asparagus asparagoides</i> | 0.2 ± 0.1 | 0.6 ± 0.1 | 0.4 ± 0.1 | 0.01 ± 0.004 |
| Broad-leaf Cotton-bush | <i>Asclepias rotundifolia</i> | 0.5 ± 0.2 | 0.2 ± 0.04 | – | – |
| Cape Weed | <i>Arctotheca calendula</i> | 0.9 ± 0.1 | 0.02 ± 0.01 | – | – |
| Hares Foot Clover | <i>Trifolium arvense</i> | 2.2 ± 0.3 | 0.7 ± 0.1 | 0.1 ± 0.02 | – |
| Horehound | <i>Marrubium vulgare</i> | 0.6 ± 0.1 | 0.01 ± 0.01 | – | – |
| Long Storks Bills | <i>Erodium botrys</i> | 4.4 ± 0.4 | 0.1 ± 0.03 | – | – |
| Ribgrass | <i>Plantago lanceolata</i> | – | 0.7 ± 0.1 | – | – |
| Salvation Jane | <i>Echium plantagineum</i> | 15.4 ± 0.7 | 2.3 ± 0.2 | – | – |
| Small-leaved Blackberry | <i>Rubus parvifolius</i> | – | – | – | 0.04 ± 0.02 |
| Smooth Cat's-ear | <i>Hypochoeris glabra</i> | 0.3 ± 0.1 | 0.3 ± 0.1 | 0.04 ± 0.01 | – |
| Three-cornered Garlic | <i>Allium triquetrum</i> | – | – | – | 0.01 ± 0.003 |

Table 3.9: Shrub species (< 1 m) within ten hectare survey sites; being open woodland with few trees and grazed woodland (OGW), grassy woodland (GW), open heath woodland (OHW) and closed heath woodland (CHW). Each ten hectare site was segregated into 25 m × 25 m squares, producing 165 ± 0.9 (mean ± SE) squares per site. Survey sites were located within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Values are average percent cover for each species (mean ± SE) within the corresponding survey squares for each woodland type (with only species whose values were > 5% cover in any one habitat type shown). n = number of ten hectare sites. Data collected October – November 2008.

| Shrubs (< 1 m) | | OGW (n = 2) | GW (n = 3) | OHW (n = 4) | CHW (n = 6) |
|---|----------------------------------|-----------------|-----------------|----------------|----------------|
| # shrubs species | | 8 | 16 | 23 | 37 |
| # shrubs species per 25 m x 25 m square | | 0.5 ± 0.04 | 1.4 ± 0.1 | 2.9 ± 0.1 | 8.0 ± 0.1 |
| Common name | Species name | % cover | % cover | % cover | % cover |
| Common Fringe-myrtle | <i>Calytrix tetragona</i> | 0.1 ± 0.1 | 0.3 ± 0.1 | 1.9 ± 0.1 | 1.5 ± 0.1 |
| Dusty Miller | <i>Spiridium parvifolium</i> | – | – | – | 0.6 ± 0.1 |
| Erect Guinea-flower | <i>Hibbertia riparia</i> | 0.7 ± 0.3 | – | – | 1.7 ± 0.1 |
| Flame Heath | <i>Astroloma conostephioides</i> | – | 0.6 ± 0.1 | 2.6 ± 0.6 | 4.2 ± 0.2 |
| Muntries | <i>Kunzea pomifera</i> | – | 0.1 ± 0.1 | 0.7 ± 0.1 | – |
| Native Current | <i>Acrotriche depressa</i> | – | – | – | 1.3 ± 0.1 |
| Sandhill Sword-sedge | <i>Lepidosperma concavum</i> | – | – | 0.9 ± 0.1 | – |
| Silky Guinea-flower | <i>Hibbertia sericea</i> | 0.01 ± 0.01 | – | – | 0.8 ± 0.1 |
| Spiky Guinea-flower | <i>Hibbertia exutiacies</i> | – | 0.1 ± 0.05 | – | 3.1 ± 0.2 |
| Yacca | <i>Xanthorrhoea semiplana</i> | 1.5 ± 0.5 | 0.01 ± 0.01 | 0.9 ± 0.2 | 9.9 ± 0.6 |

Table 3.10: Midstorey plant species (1 – 4 m) within ten hectare survey sites; being open woodland with few trees and grazed woodland (OGW), grassy woodland (GW), open heath woodland (OHW) and closed heath woodland (CHW). Each ten hectare site was segregated into 25 m × 25 m squares, producing 165 ± 0.9 (mean ± SE) squares per site. Survey sites were located within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Values are average percent cover for each species (mean ± SE) within the corresponding survey squares for each woodland type (with only species whose values were > 5% cover in any one habitat type shown). n = number of ten hectare sites. Data collected October – November 2008.

| Midstorey (1 – 4 m) | | OGW (n = 2) | GW (n = 3) | OHW (n = 4) | CHW (n = 6) |
|--|-----------------------------------|------------------------|-----------------------|------------------------|------------------------|
| # midstorey species | | 12 | 19 | 21 | 31 |
| # midstorey species per 25 m x 25 m square | | 0.4 ± 0.04 | 1.8 ± 0.05 | 3.3 ± 0.1 | 4.4 ± 0.1 |
| Common name | Species name | % cover | % cover | % cover | % cover |
| Beaked Hakea | <i>Hakea rostrata</i> | – | – | – | 0.7 ± 0.1 |
| Broombush | <i>Melaleuca uncinata</i> | – | 0.2 ± 0.1 | 2.6 ± 0.2 | 3.1 ± 0.4 |
| Christmas Bush | <i>Bursaria spinosa</i> | – | 0.7 ± 0.1 | – | 0.6 ± 0.1 |
| Common Fringe-myrtle | <i>Calytrix tetragona</i> | – | 0.1 ± 0.03 | 1.8 ± 0.1 | 1.0 ± 0.1 |
| Downy Mintbush | <i>Prostanthera behriana</i> | – | – | – | 0.8 ± 0.1 |
| Dusty Miller | <i>Spyridium parvifolium</i> | – | – | – | 0.5 ± 0.1 |
| Flame Heath | <i>Astroloma conostephiooides</i> | – | 0.02 ± 0.02 | 0.8 ± 0.1 | 0.3 ± 0.1 |
| Golden Wattle | <i>Acacia pycnantha</i> | 0.8 ± 0.2 | 6.4 ± 0.5 | 1.3 ± 0.1 | 4.4 ± 0.2 |
| Hakea | <i>Hakea carinata</i> | – | – | – | 0.8 ± 0.1 |
| Kangaroo Thorn | <i>Acacia paradoxa</i> | 0.5 ± 0.2 | 2.8 ± 0.4 | 4.1 ± 0.5 | 2.1 ± 0.2 |
| Long-leaved Box | <i>Eucalyptus goniocalyx</i> | 0.03 ± 0.02 | – | – | 0.7 ± 0.1 |
| Oyster Bay Pine | <i>Callitris rhomboidea</i> | – | – | – | 0.7 ± 0.1 |
| Peppermint Box | <i>Eucalyptus odorata</i> | – | – | – | 2.3 ± 0.2 |
| Pink Gum | <i>Eucalyptus fasciculosa</i> | 0.04 ± 0.03 | 0.2 ± 0.1 | 0.4 ± 0.1 | 1.3 ± 0.1 |
| River Red-gum | <i>Eucalyptus camaldulensis</i> | 0.1 ± 0.1 | 0.8 ± 0.2 | – | – |
| SA Blue Gum | <i>Eucalyptus leucoxylon</i> | 0.4 ± 0.2 | 1.7 ± 0.2 | 0.04 ± 0.01 | 0.3 ± 0.1 |
| Slaty Sheoak | <i>Allocasuarina muelleriana</i> | – | – | – | 5.2 ± 0.3 |
| Slender Cypress-pine | <i>Callitris gracilis</i> | – | 2.5 ± 0.3 | 4.7 ± 0.3 | 1.8 ± 0.1 |
| Sticky Hopbush | <i>Dodonaea viscosa</i> | 0.1 ± 0.1 | 1.4 ± 0.3 | 0.9 ± 0.1 | 4.0 ± 0.3 |

Table 3.11: Proportions of overstorey tree species (> 4 m high) within ten hectare survey sites classified as open woodland with few trees and grazed woodland (OGW), grassy woodland (GW), open heath woodland (OHW) and closed heath woodland (CHW). The closed heath woodland sites are separated into those located at Para Wirra and those located at Hale. Para Wirra sites contained Gum and Box eucalypt species and Hale sites contained Box, Stringybark and some Gum eucalypt species. Each ten hectare site was segregated into $25\text{ m} \times 25\text{ m}$ squares, producing 165 ± 0.9 (mean \pm SE) squares per site. Survey sites were located within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Values are the proportion of each tree species within all of the $25\text{ m} \times 25\text{ m}$ survey squares within each of the woodland types. n = number of ten hectare sites. Data collected October – November 2008.

| Overstorey (> 4 m) | | OGW (n = 2) | GW (n = 3) | OHW (n = 4) | CHW (n = 6) | |
|--|-----------------------------------|------------------------|-----------------------|------------------------|------------------------|-----------------|
| # overstorey species | | 7 | 9 | 9 | (n = 4) | (n = 2) |
| # overstorey species per $25\text{ m} \times 25\text{ m}$ square | | 0.5 ± 0.03 | 1.3 ± 0.03 | 2.3 ± 0.03 | 2.0 ± 0.04 | 2.1 ± 0.06 |
| Common name | Species name | proportion | proportion | proportion | Para Wirra proportion | Hale proportion |
| Drooping Sheoak | <i>Allocasuarina verticillata</i> | 0.003 | 0.05 | 0.35 | 0.13 | – |
| Golden Wattle | <i>Acacia pycnantha</i> | 0.01 | 0.04 | 0.13 | 0.14 | 0.08 |
| Long-leaved Box | <i>Eucalyptus goniocalyx</i> | 0.03 | – | – | 0.39 | 0.79 |
| Messmate Stringybark | <i>Eucalyptus obliqua</i> | – | – | – | – | 0.65 |
| Native Cherry | <i>Exocarpos cupressiformis</i> | – | – | – | 0.003 | 0.01 |
| Oyster Bay Pine | <i>Callitris rhomboidea</i> | – | – | – | – | 0.26 |
| Peppermint Box | <i>Eucalyptus odorata</i> | 0.27 | 0.03 | 0.11 | 0.17 | – |
| Pink Gum | <i>Eucalyptus fasciculosa</i> | 0.02 | 0.17 | 0.61 | 0.54 | 0.32 |
| River Red-gum | <i>Eucalyptus camaldulensis</i> | 0.02 | 0.18 | 0.04 | – | – |
| SA Blue Gum | <i>Eucalyptus leucoxylon</i> | 0.19 | 0.45 | 0.10 | 0.33 | 0.02 |
| Silver Banksia | <i>Banksia marginata</i> | – | 0.004 | 0.01 | – | – |
| Slender Cypress-pine | <i>Callitris gracilis</i> | – | 0.29 | 0.94 | 0.31 | – |
| Swamp Wattle | <i>Acacia retinodes</i> | – | 0.03 | 0.01 | – | – |

Table 3.12: Overstorey trees per hectare (> 4 m high) within ten hectare survey sites. The woodland types are open woodland with few trees and grazed woodland (Open / Grazed), grassy woodland (Grassy), open heath woodland (Open heath) and closed heath woodland (Closed Heath). The codes directly below the woodland types are the site numbers used to distinguish the 15 ten hectare sites. The values are the trees per hectare for the sum of all trees as well as for individual species. Data collected October – November 2008.

| Common name | Species name | Open / Grazed | | Grassy | | | Open Heath | | | | Closed Heath | | | | | |
|---------------------------------------|-----------------------------------|----------------------|-------------|---------------|-------------|-------------|-------------------|--------------|--------------|--------------|---------------------|-------------|--------------|-------------|--------------|--------------|
| | | S4 | S7 | S1 | S6 | S15 | S8 | S10 | S11 | S14 | S2 | S3 | S5 | S9 | S12 | S13 |
| Total of all trees per hectare | | 11.6 | 22.7 | 63.2 | 71.0 | 76.7 | 165.4 | 114.3 | 355.6 | 218.2 | 115.9 | 66.3 | 142.6 | 99.0 | 110.4 | 225.4 |
| Drooping Sheoak | <i>Allocasuarina verticillata</i> | – | 0.5 | – | 0.5 | 4.3 | 4.8 | 16.2 | 30.9 | 5.1 | 12.1 | 4.9 | 2.2 | 6.6 | – | – |
| Golden Wattle | <i>Acacia pycnantha</i> | 0.2 | – | – | 1.2 | 1.3 | 7.1 | 0.3 | 0.4 | 7.9 | 3.3 | 4.4 | 1.9 | 3.2 | 1.2 | 1.8 |
| Long-leaved Box | <i>Eucalyptus goniocalyx</i> | 4.7 | – | – | – | – | – | – | – | – | 29.4 | – | 24.7 | 14.5 | 51.3 | 48.8 |
| Messmate Stringybark | <i>Eucalyptus obliqua</i> | – | – | – | – | – | – | – | – | – | – | – | – | – | 41.4 | 131.9 |
| Native Cherry | <i>Exocarpos cupressiformis</i> | – | – | – | – | – | – | – | – | – | – | 0.1 | – | 0.1 | 0.1 | 0.1 |
| Oyster Bay Pine | <i>Callitris rhomboidea</i> | – | – | – | – | – | – | – | – | – | – | – | – | – | 4.7 | 35.4 |
| Peppermint Box | <i>Eucalyptus odorata</i> | – | 16.0 | – | – | 4.6 | 0.1 | – | 61.5 | – | 7.9 | 19.8 | – | – | – | – |
| Pink Gum | <i>Eucalyptus fasciculosa</i> | 1.2 | 0.3 | 1.1 | 9.6 | 2.5 | 24.7 | 33.0 | 24.6 | 12.4 | 31.5 | 14.0 | 20.2 | 27.3 | 11.0 | 7.4 |
| River Red-gum | <i>Eucalyptus camaldulensis</i> | 2.6 | – | 0.2 | 2.2 | 41.2 | 0.1 | – | – | 5.1 | – | – | – | – | – | – |
| SA Blue Gum | <i>Eucalyptus leucoxylon</i> | 2.9 | 5.9 | 61.7 | 9.2 | 10.1 | 2.9 | – | 1.7 | 7.0 | 8.4 | 18.2 | 92.8 | 13.4 | 0.7 | – |
| Silver Banksia | <i>Banksia marginata</i> | – | – | – | – | 0.4 | 0.1 | 0.4 | – | – | – | – | – | – | – | – |
| Slender Cypress-pine | <i>Callitris gracilis</i> | – | – | 0.2 | 47.8 | 9.7 | 125.6 | 64.4 | 236.5 | 180.0 | 23.3 | 4.9 | 0.8 | 33.9 | – | – |
| Swamp Wattle | <i>Acacia retinodes</i> | – | – | – | 0.5 | 2.6 | – | – | – | 0.7 | – | – | – | – | – | – |

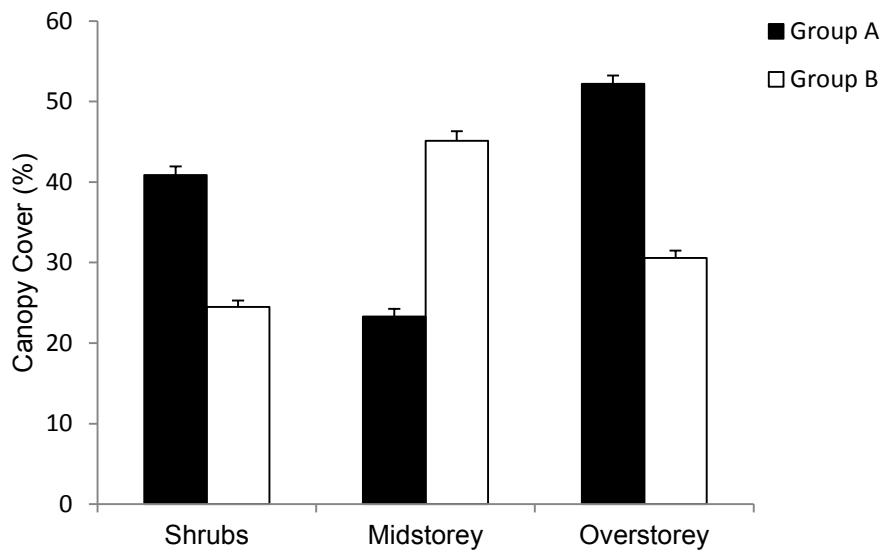


Figure 3.4: Vegetation differences within six ten hectare sites that were broadly classified *a priori* as closed heath woodland. The sites separated evenly into two groups (Group A – consisting of Sites 2, 5 and 13; and Group B – consisting of Sites 3, 9 and 12). These data are based on the average percent canopy cover for the relevant 25 m × 25 m survey squares for both Groups ($n = 501$ for each group). On average, sites in Group A contained greater cover of shrubs (< 1 m) and overstorey vegetation (> 4 m), while sites in Group B contained greater cover of midstorey vegetation (1 – 4 m). Error bars are standard error. These data were collected from locations within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected October – November 2008.

Declining bird species

Seventy nine woodland bird species were observed, with 34 regarded as declining within the Mount Lofty region (based on Paton *et al.* 2004). Grassy woodland, open heath woodland and closed heath woodland sites all hosted large bird communities, with each containing an array of declining bird species (Table 3.13). The minimal presence of the Noisy Miner (*Manorina melanocephala*) did not appear to adversely influence the outcome of this survey, as this aggressive bird species was only detected in two grassy woodland sites and in low frequency (Site 6, 4% of all birds observed and Site 7, 3% of all birds observed).

Table 3.13: Woodland birds detected within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Species in bold are regarded as declining within the Mount Lofty Ranges (based on Paton *et al.* 2004). Values are average densities per hectare of each bird species (birds ha⁻¹ ± SE). Surveys were conducted across 15 ten hectare sites within grassy woodland (GW; *n* = 5 sites), open heath woodland (OHW; *n* = 4 sites) and closed heath woodland (CHW; *n* = 6 sites). Data collected September – October 2008.

| Common name | Species name | GW (birds ha ⁻¹) | OHW (birds ha ⁻¹) | CHW (birds ha ⁻¹) |
|--------------------------------|-------------------------------------|---------------------------------|----------------------------------|----------------------------------|
| Total of all birds | | 7.24 ± 0.70 | 6.24 ± 0.46 | 6.47 ± 0.40 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 0.42 ± 0.05 | 0.04 ± 0.02 | 0.04 ± 0.02 |
| Australian Owlet-nightjar | <i>Aegotheles cristatus</i> | – | – | 0.01 ± 0.01 |
| Australian Raven | <i>Corvus coronoides</i> | 0.02 ± 0.01 | – | – |
| Black-faced Cuckoo-shrike | <i>Coracina novaehollandiae</i> | 0.08 ± 0.04 | 0.03 ± 0.01 | 0.01 ± 0.01 |
| Brown Falcon | <i>Falco berigora</i> | 0.02 ± 0.01 | – | – |
| Brown Goshawk | <i>Accipiter fasciatus</i> | 0.03 ± 0.01 | 0.02 ± 0.01 | – |
| Brown Thornbill | <i>Acanthiza pusilla</i> | – | 0.01 ± 0.01 | 0.34 ± 0.07 |
| Brown Treecreeper | <i>Climacteris picumnus</i> | 0.08 ± 0.03 | – | 0.01 ± 0.01 |
| Brown-headed Honeyeater | <i>Melithreptus brevirostris</i> | 0.04 ± 0.03 | 0.09 ± 0.04 | 0.15 ± 0.04 |
| Brush Bronzewing | <i>Phaps elegans</i> | 0.01 ± 0.01 | – | 0.03 ± 0.01 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 0.08 ± 0.05 | 0.04 ± 0.02 | 0.23 ± 0.05 |
| Collared Sparrowhawk | <i>Accipiter cirrhocephalus</i> | – | 0.01 ± 0.01 | – |
| Common Blackbird | <i>Turdus merula</i> | 0.01 ± 0.01 | 0.02 ± 0.01 | 0.02 ± 0.01 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0.15 ± 0.04 | 0.21 ± 0.06 | 0.07 ± 0.02 |
| Common Starling | <i>Sturnus vulgaris</i> | 0.10 ± 0.03 | 0.07 ± 0.03 | – |
| Crescent Honeyeater | <i>Phylidonyris pyrrhoptera</i> | – | – | 0.31 ± 0.04 |
| Crested Pigeon | <i>Ocyphaps lophotes</i> | 0.06 ± 0.03 | 0.01 ± 0.01 | – |
| Crested Shrike-tit | <i>Falcunculus frontatus</i> | 0.01 ± 0.01 | – | – |
| Crimson Rosella | <i>Platycercus elegans</i> | 0.82 ± 0.10 | 0.37 ± 0.07 | 0.23 ± 0.04 |
| Diamond Firetail | <i>Stagonopleura guttata</i> | 0.08 ± 0.03 | 0.02 ± 0.01 | – |
| Dusky Woodswallow | <i>Artamus cyanopterus</i> | 0.04 ± 0.02 | 0.01 ± 0.01 | 0.01 ± 0.01 |
| Eastern Spinebill | <i>Acanthorhynchus tenuirostris</i> | 0.01 ± 0.01 | 0.05 ± 0.02 | 0.06 ± 0.02 |
| Elegant Parrot | <i>Neophema elegans</i> | – | – | 0.01 ± 0.01 |
| European Goldfinch | <i>Carduelis carduelis</i> | 0.01 ± 0.01 | 0.03 ± 0.02 | 0.04 ± 0.02 |
| Fairy Martin | <i>Petrochelidon ariel</i> | 0.14 ± 0.08 | – | 0.03 ± 0.02 |
| Fan-tailed Cuckoo | <i>Cacomantis flabelliformis</i> | 0.01 ± 0.01 | – | 0.01 ± 0.01 |
| Galah | <i>Cacatua roseicapilla</i> | 0.33 ± 0.09 | – | 0.01 ± 0.01 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 0.01 ± 0.01 | 0.05 ± 0.02 | 0.23 ± 0.04 |

| Common name | Species name | GW (birds ha ⁻¹) | OHW (birds ha ⁻¹) | CHW (birds ha ⁻¹) |
|-----------------------------------|-------------------------------------|---------------------------------|----------------------------------|----------------------------------|
| Grey Currawong | <i>Strepera versicolor</i> | 0.06 ± 0.02 | 0.04 ± 0.02 | 0.15 ± 0.02 |
| Grey Fantail | <i>Rhipidura fuliginosa</i> | 0.02 ± 0.01 | 0.05 ± 0.02 | 0.05 ± 0.02 |
| Grey Shrike-thrush | <i>Colluricincla harmonica</i> | 0.06 ± 0.02 | 0.09 ± 0.02 | 0.21 ± 0.02 |
| Hooded Robin | <i>Melanodryas cucullata</i> | 0.02 ± 0.01 | 0.08 ± 0.03 | — |
| Horsfield's Bronze-Cuckoo | <i>Chrysococcyx basalis</i> | 0.01 ± 0.01 | 0.03 ± 0.03 | 0.05 ± 0.02 |
| House Sparrow | <i>Passer domesticus</i> | 0.04 ± 0.02 | — | — |
| Jacky Winter | <i>Microeca fascinans</i> | 0.01 ± 0.01 | 0.01 ± 0.01 | — |
| Laughing Kookaburra | <i>Dacelo novaeguineae</i> | 0.08 ± 0.02 | — | — |
| Little Raven | <i>Corvus mellori</i> | 0.05 ± 0.03 | 0.04 ± 0.02 | 0.01 ± 0.01 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 0.13 ± 0.03 | 0.30 ± 0.04 | 0.04 ± 0.01 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 0.48 ± 0.12 | 0.03 ± 0.02 | 0.07 ± 0.02 |
| Nankeen Kestrel | <i>Falco cenchroides</i> | — | 0.01 ± 0.01 | — |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 0.58 ± 0.18 | 0.26 ± 0.07 | 1.50 ± 0.22 |
| Noisy Miner | <i>Manorina melanocephala</i> | 0.14 ± 0.04 | — | — |
| Painted Button-quail | <i>Turnix varia</i> | 0.01 ± 0.01 | — | 0.01 ± 0.01 |
| Peaceful Dove | <i>Geopelia striata</i> | 0.11 ± 0.05 | 0.27 ± 0.06 | 0.02 ± 0.01 |
| Rainbow Bee-eater | <i>Merops ornatus</i> | 0.03 ± 0.02 | 0.01 ± 0.01 | — |
| Rainbow Lorikeet | <i>Trichoglossus haematodus</i> | — | 0.02 ± 0.02 | — |
| Red-browed Finch | <i>Neochmia temporalis</i> | 0.01 ± 0.01 | 0.02 ± 0.01 | 0.06 ± 0.02 |
| Red-capped Robin | <i>Petroica goodenovii</i> | — | 0.38 ± 0.06 | 0.01 ± 0.01 |
| Red-rumped Parrot | <i>Psephotus haematonotus</i> | 0.43 ± 0.08 | 0.01 ± 0.01 | 0.02 ± 0.02 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 0.45 ± 0.10 | 0.05 ± 0.03 | 0.15 ± 0.04 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 0.10 ± 0.03 | 0.43 ± 0.05 | 0.07 ± 0.02 |
| Sacred Kingfisher | <i>Todiramphus sanctus</i> | — | — | 0.01 ± 0.01 |
| Scarlet Robin | <i>Petroica boodang</i> | — | — | 0.04 ± 0.01 |
| Shining Bronze-Cuckoo | <i>Chrysococcyx lucidus</i> | — | 0.02 ± 0.01 | — |
| Silvereye | <i>Zosterops lateralis</i> | 0.02 ± 0.02 | 0.27 ± 0.04 | 0.22 ± 0.04 |
| Southern Boobook | <i>Ninox novaeseelandiae</i> | — | 0.02 ± 0.01 | 0.01 ± 0.01 |
| Spotted Pardalote | <i>Pardalotus punctatus</i> | 0.02 ± 0.01 | — | 0.01 ± 0.01 |
| Square-tailed Kite | <i>Lophoictinia isura</i> | — | — | 0.01 ± 0.01 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 0.54 ± 0.07 | 0.25 ± 0.07 | 0.45 ± 0.06 |
| Striated Thornbill | <i>Acanthiza lineata</i> | — | 0.01 ± 0.01 | 0.35 ± 0.08 |
| Sulphur-crested Cockatoo | <i>Cacatua galerita</i> | 0.01 ± 0.01 | — | — |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 0.10 ± 0.04 | 0.02 ± 0.02 | 0.35 ± 0.05 |
| Tawny Frogmouth | <i>Podargus strigoides</i> | — | — | 0.01 ± 0.01 |
| Tree Martin | <i>Hirundo nigricans</i> | 0.08 ± 0.04 | — | — |
| Varied Sittella | <i>Daphoenositta chrysoptera</i> | — | 0.07 ± 0.04 | 0.01 ± 0.01 |
| Wedge-tailed Eagle | <i>Aquila audax</i> | 0.01 ± 0.01 | — | 0.01 ± 0.01 |
| Weebill | <i>Smicrornis brevirostris</i> | 0.03 ± 0.02 | 0.11 ± 0.05 | 0.01 ± 0.01 |
| Welcome Swallow | <i>Hirundo neoxena</i> | 0.02 ± 0.01 | — | — |
| White-browed Babbler | <i>Pomatostomus superciliosus</i> | 0.28 ± 0.11 | 0.15 ± 0.05 | 0.09 ± 0.04 |
| White-browed Scrubwren | <i>Sericornis frontalis</i> | — | — | 0.23 ± 0.05 |
| White-plumed Honeyeater | <i>Lichenostomus penicillatus</i> | 0.30 ± 0.06 | 0.01 ± 0.01 | 0.01 ± 0.01 |
| White-throated Treecreeper | <i>Cormobates leucophaeus</i> | — | — | 0.04 ± 0.02 |
| White-winged Chough | <i>Corcorax melanorhamphos</i> | 0.07 ± 0.04 | 0.13 ± 0.05 | 0.05 ± 0.03 |
| White-winged Triller | <i>Lalage tricolor</i> | 0.01 ± 0.01 | 0.05 ± 0.02 | 0.02 ± 0.01 |
| Willie Wagtail | <i>Rhipidura leucophrys</i> | 0.16 ± 0.03 | 0.02 ± 0.01 | — |
| Yellow Thornbill | <i>Acanthiza nana</i> | 0.13 ± 0.03 | 1.74 ± 0.19 | 0.26 ± 0.06 |
| Yellow-faced Honeyeater | <i>Lichenostomus chrysops</i> | 0.02 ± 0.01 | 0.10 ± 0.06 | 0.03 ± 0.01 |
| Yellow-rumped Thornbill | <i>Acanthiza chrysorrhoa</i> | 0.06 ± 0.03 | 0.19 ± 0.04 | 0.01 ± 0.01 |
| Zebra Finch | <i>Taeniopygia guttata</i> | 0.01 ± 0.01 | — | — |

The cluster analysis based on bird abundances (Fig. 3.5) produced comparable clusters to those that formed in the Sites by Habitat dendrogram (refer Fig. 3.2). Again, there was clear separation between the clusters ($T = -6.76$), with strong within-group homogeneity ($A = 0.22$) that was unlikely to be due to chance (delta $p = < 0.0001$). However the relationships between the clusters in the two dendograms were completely different. In the Sites by Birds dendrogram, the open heath woodland cluster was now more similar to the closed heath woodland cluster, and both were now distinct from the grassy woodland cluster (Fig. 3.5). Also, both sites within the open grassland with few trees and grazed woodland cluster now nested within the grassy woodland cluster (Fig. 3.5). There were also some patterns in common between the two dendograms, with Site 5 (classified *a priori* as open heath woodland) again associating with the closed heath woodland cluster (Fig. 3.5).

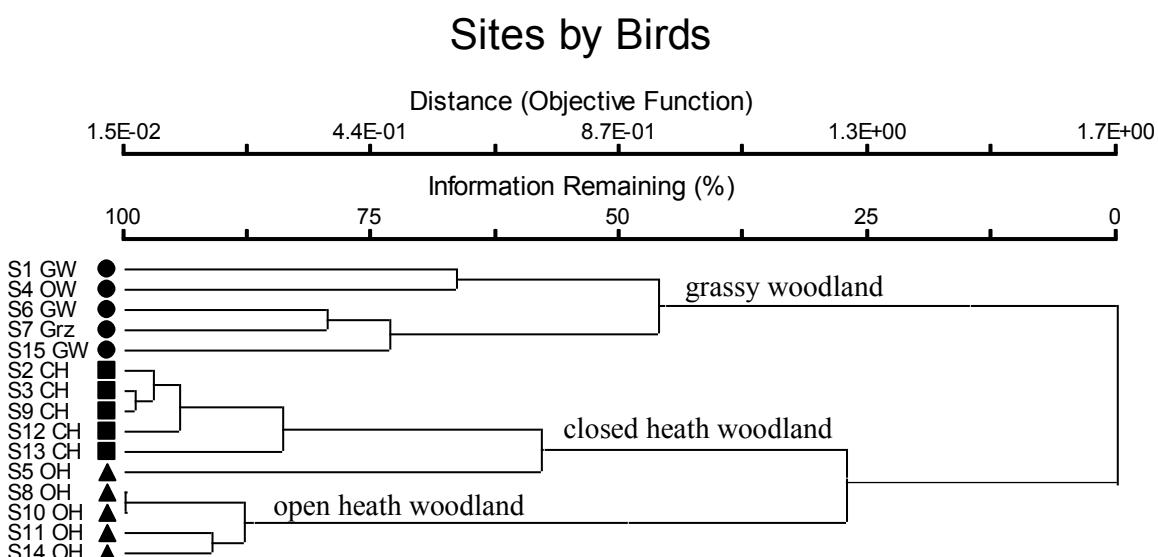


Figure 3.5: Cluster analysis dendrogram of 15 ten hectare survey sites based upon bird species associations. These data were collected from locations within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Surveys were conducted within grassy woodland (GW, OW and Grz), open heath woodland (OH) and closed heath woodland (CH). Data collected September – October 2008.

Indicator Species Analysis linked particular bird species to habitat type. Twenty-five bird species were identified as having a statistically significant high indicator value (Table 3.14). These 25 bird species displayed a specific high faithfulness to one of the three woodland types. Nine of these species are regarded as declining within the Mount Lofty

region. These were the Brown Treecreeper (*Climacteris picumnus*) and Red-rumped Parrot (*Psephotus haematonotus*) for grassy woodland sites; the Rufous Whistler (*Pachycephala rufiventris*), Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*) and Yellow Thornbill (*A. nana*) for open heath woodland sites; and the Brown Thornbill (*A. pusilla*), Buff-rumped Thornbill (*A. reguloides*), Crescent Honeyeater (*Phylidonyris pyrrhoptera*) and Scarlet Robin (*Petroica boodang*) for closed heath woodland sites (Table 3.14).

While ISA disregards species expressing low and/or infrequent detection rates, it also masked some declining species that associated with more than one habitat type. These were the Common Bronzewing (*Phaps chalcoptera*) and White-browed Babbler (*Pomatostomus superciliosus*), which had high indicator values for both grassy woodland and open heath woodland sites (Table 3.13). Also, the Silvereye (*Zosterops lateralis*) had high indicator values for both open heath woodland and closed heath woodland sites (Table 3.13).

While the closed heath woodland cluster delineated into two distinct groups based upon their physical habitat attributes (Fig. 3.2; Groups A and B in Fig. 3.4), these six closed heath sites shared more in common with regards to the bird species that were recorded using them (Fig. 3.5). Also, although the grassy woodland cluster formed with all five grassy sites, Sites 1 and 4 were distinct from the other three (Figs 3.5 and 3.6). Sites 1 and 4 were both located within the higher rainfall Para Wirra region and were both nested within closed heath habitats. The other three grassy woodland sites (Sites 2, 7 and 15) were located further away on the lower rainfall plains (refer Fig. 3.1).

Indicator Species Analysis could not distinguish any characteristic bird species that could be used to distinguish Sites 1 and 4 from Sites 2, 7 and 15. Instead, these sites delineated due to the different numbers of birds and bird species that were recorded in this study. Of the 26 woodland bird species that were observed using the two grassy sites at higher elevations, 77% of these were also observed using the grassy sites at lower elevations. However, the three grassy sites on the lower elevations hosted many more bird species in total (56 vs. 26 species) and had greater numbers of birds overall (9.3 ± 0.7 vs. 4.1 ± 0.6 birds ha^{-1}).

Table 3.14: Indicator bird species from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). Dendrogram clusters are for grassy woodland (GW; $n = 5$), open heath woodland (OHW; $n = 4$) and closed heath woodland (CHW; $n = 6$). Indicator species with a statistically significant association to a specific woodland type are shown with the corresponding maximum observed indicator value (highest percent in bold) and its significant p -value. The p -value tests the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value). Statistically significant p -values ($\alpha \leq 0.05$) identify those species whose indicator value was larger than expected by chance. Indicator values are based on the combination of relative abundance and relative frequency for each individual species within each woodland type. Indicator values indicate the degree of faithfulness and reliability that a particular species may be found within a certain woodland type. Indicator values range from zero (no indication) to 100 (perfect indication). Declining species within the Mount Lofty Ranges are in bold (based on Paton *et al.* 2004). Data collected September – October 2008.

| Common name | Species name | Dendrogram cluster | | | <i>p</i>-value |
|--------------------------------|-------------------------------------|---------------------------|------------|------------|-----------------------|
| | | GW | OHW | CHW | |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 83 | 6 | 6 | 0.0008 |
| Brown Thornbill | <i>Acanthiza pusilla</i> | 0 | 1 | 97 | 0.0006 |
| Brown Treecreeper | <i>Climacteris picumnus</i> | 58 | 0 | 1 | 0.0492 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 10 | 9 | 65 | 0.0442 |
| Crescent Honeyeater | <i>Phylidonyris pyrrhoptera</i> | 0 | 0 | 100 | 0.0006 |
| Crimson Rosella | <i>Platycercus elegans</i> | 58 | 26 | 16 | 0.0156 |
| Galah | <i>Cacatua roseicapilla</i> | 97 | 0 | 0 | 0.0006 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 2 | 17 | 79 | 0.0056 |
| Grey Currawong | <i>Strepera versicolor</i> | 15 | 12 | 59 | 0.0134 |
| Grey Shrike-thrush | <i>Colluricinclla harmonica</i> | 10 | 25 | 58 | 0.0026 |
| Laughing Kookaburra | <i>Dacelo novaeguineae</i> | 100 | 0 | 0 | 0.0008 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 28 | 65 | 7 | 0.0118 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 83 | 1 | 9 | 0.0030 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 20 | 8 | 64 | 0.0276 |
| Peaceful Dove | <i>Geopelia striata</i> | 17 | 67 | 3 | 0.0254 |
| Red-capped Robin | <i>Petroica goodenovii</i> | 0 | 99 | 0 | 0.0008 |
| Red-rumped Parrot | <i>Psephotus haematonotus</i> | 93 | 1 | 1 | 0.0084 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 70 | 8 | 23 | 0.0056 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 10 | 74 | 10 | 0.0084 |
| Scarlet Robin | <i>Petroica boodang</i> | 0 | 0 | 67 | 0.0226 |
| Striated Thornbill | <i>Acanthiza lineata</i> | 0 | 1 | 97 | 0.0018 |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 13 | 1 | 74 | 0.0064 |
| White-browed Scrubwren | <i>Sericornis frontalis</i> | 0 | 0 | 83 | 0.0104 |
| Yellow-rumped Thornbill | <i>Acanthiza chrysorrhoa</i> | 14 | 72 | 2 | 0.0072 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 6 | 82 | 10 | 0.0020 |

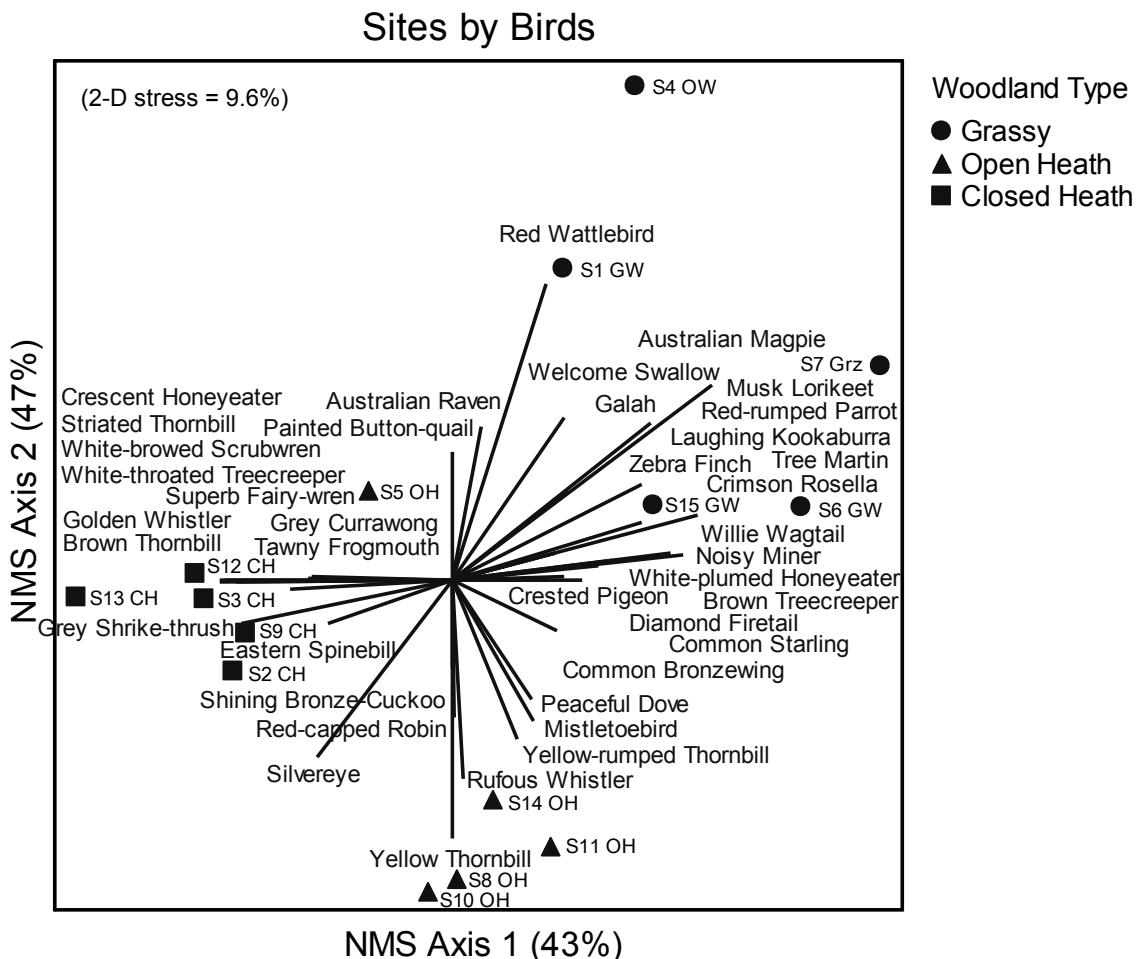


Figure 3.6: Two-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure, for declining bird species in respect to habitat types. Sites are grassy woodland (Grassy), open heath woodland (Open Heath) and closed heath woodland (Closed Heath). The bird species shown have strong correlation with the ordination. Each bird species vector orientation indicates the direction of positive increase and the length represents the strength of association with the different woodland types. The woodland types for each of the 15 ten hectare sites are indicated by the symbol in the legend. Individual sites are labelled with their site identifier (S#) and the site type. For grassy woodlands (GW) this also included open woodland with few trees (OW) and grazed woodland (Grz). Open heath woodland sites are labelled OH and closed heath woodland sites are labelled CH. At a stress level of 9.6%, 90% of the variation is represented by the distribution of the woodland types. These data were collected from locations within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected October – November 2008.

Habitat associations by declining bird species

Several ecologically similar bird species were clearly segregated by habitat structure. The Hooded Robin (*Melanodryas cucullata*) and Jacky Winter (*Microeca fascinans*) were only found within grassy woodland and open heath woodland sites, while the Scarlet Robin was only found in closed heath woodland sites (Fig. 3.7). The Rufous Whistler was generally found in more open habitats, while the Golden Whistler (*Pachycephala pectoralis*) was generally found in more closed habitats (Fig. 3.8). The Common Bronzewing was also generally found in more open habitats, while the Brush Bronzewing (*Phaps elegans*) was generally found in more closed habitats (Fig. 3.9). The Brown Treecreeper was an indicator species for grassy woodland sites, while the White-throated Treecreeper (*Cormobates leucophaeus*) was only ever detected within closed heath woodland sites (Fig. 3.10).

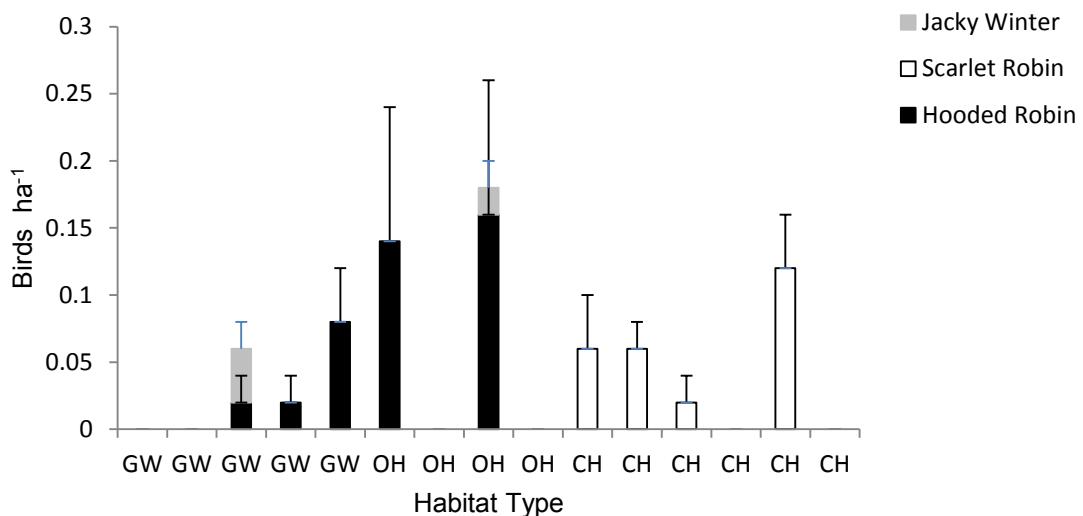


Figure 3.7: Distributions of the Hooded Robin (*Melanodryas cucullata*), Jacky Winter (*Microeca fascinans*) and Scarlet Robin (*Petroica boodang*) across three habitat types; grassy woodland (GW), open heath woodland (OH) and closed heath woodland (CH). These data indicate that the Hooded Robin and Jacky Winter were generally found in more open habitats and the Scarlet Robin in more closed habitats. Error bars are standard error (the error bars for the Jacky Winter data are colour-coded blue). Surveys were conducted within 15 ten hectare sites within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected September – October 2008.

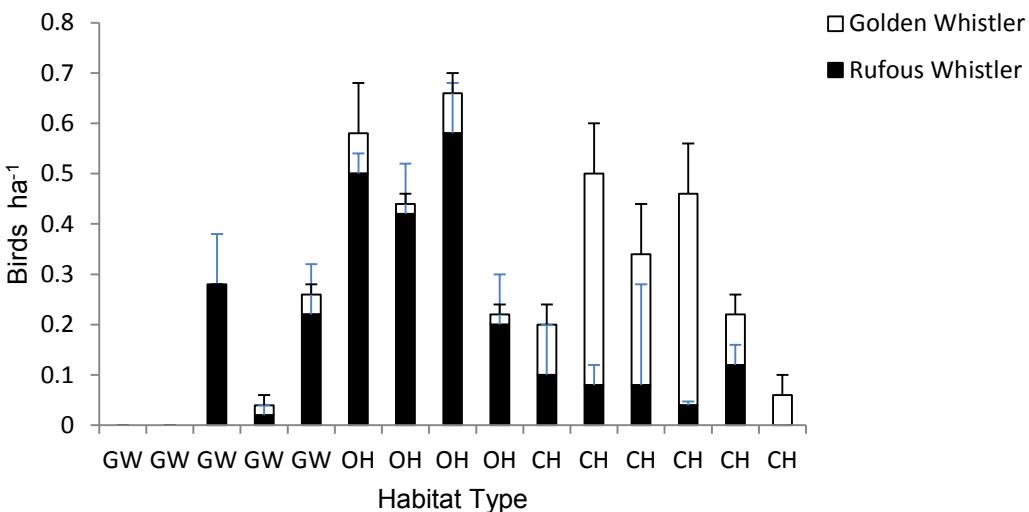


Figure 3.8: Distributions of the Rufous Whistler (*Pachycephala rufiventris*) and Golden Whistler (*Pachycephala pectoralis*) across three habitat types; grassy woodland (GW), open heath woodland (OH) and closed heath woodland (CH). These data indicate that the Rufous Whistler was generally found in more open habitats and the Golden Whistler in more closed habitats. Error bars are standard error (the error bars for the Rufous Whistler data are colour-coded blue). Surveys were conducted within 15 ten hectare sites within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected September – October 2008.

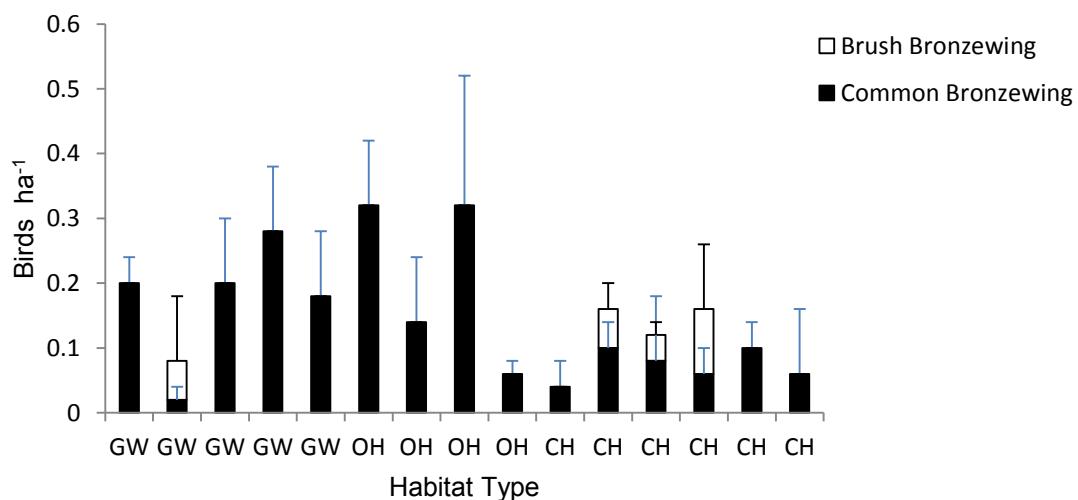


Figure 3.9: Distributions of the Common Bronzewing (*Phaps chalcoptera*) and Brush Bronzewing (*Phaps elegans*) across three habitat types; grassy woodland (GW), open heath woodland (OH) and closed heath woodland (CH). These data indicate that the Common Bronzewing was generally found in more open habitats and the Brush Bronzewing in more closed habitats. Error bars are standard error (the error bars for the Common Bronzewing data are colour-coded blue). Surveys were conducted within 15 ten hectare sites within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected September – October 2008.

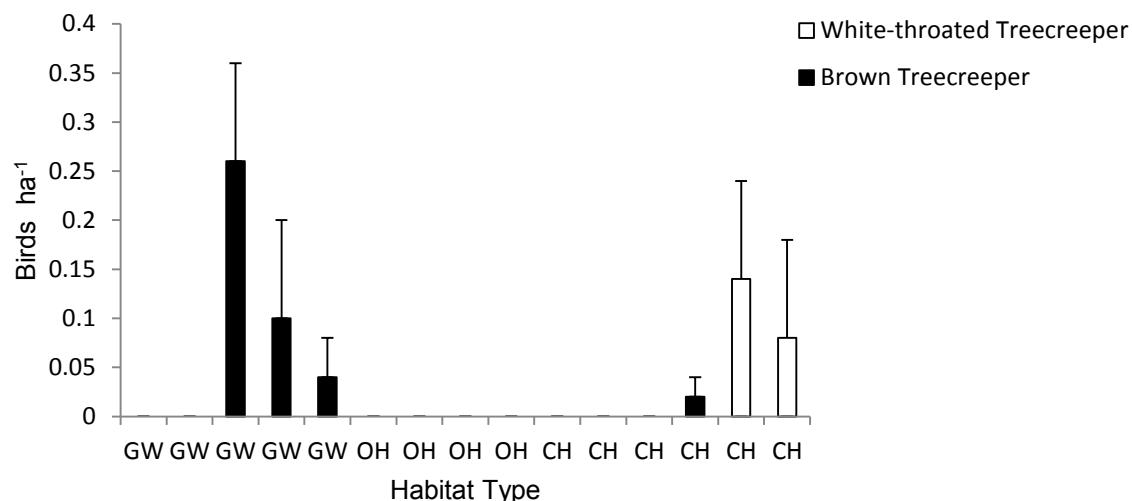


Figure 3.10: Distributions of the Brown Treecreeper (*Climacteris picumnus*) and White-throated Treecreeper (*Cormobates leucophaeus*) across three habitat types; grassy woodland (GW), open heath woodland (OH) and closed heath woodland (CH). These data indicate that the Brown Treecreeper was generally found in grassy woodland sites and the White-throated Treecreeper was only ever found in closed heath woodland sites. Error bars are standard error. Surveys were conducted within 15 ten hectare sites within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected September – October 2008.

Discussion

This study showed that there are distinct woodland systems within the north-central zone of the Mount Lofty Ranges in South Australia, which differ in structural elements and floristic composition. This study also demonstrated that woodland bird communities varied between different woodland systems (at the ten hectare scale at least). Importantly, a particular woodland type did not support the full suite of declining bird species within this region. The principal goal of the revegetation works on the Para Woodland Reserve is to maximize the number of woodland bird species (especially those in decline) which are likely to benefit from the revegetation works. As such, a mixture of different woodland types needs to be considered. The success of this (and similar) broad-scale projects that aim to re-instate woodland habitats to support vulnerable bird species, will ultimately depend on which woodland systems are returned. As a broad application, a heterogeneous planting of different woodland types should attract a wider suite of woodland bird species, including some of conservation concern.

The distinction between grassy, open heath and closed heath woodland habitats had more to do with the habitat structure below four metres than at the tree level. Research has shown that ground conditions and the habitat structure at lower storey levels can profoundly influence the species composition of a woodland bird community (Fuller 2001; Antos and Bennett 2006). In agreement, the bird communities in this study that were associated with certain woodland types appeared to be independent of landscape settings and degrees of habitat heterogeneity within the woodland clusters. For example, although open heath sites and grassy sites had more in common with regards to vegetation than closed heath sites, the bird communities were more similar in open heath and closed heath sites than grassy sites. This is despite the fact that open heath sites and most grassy sites were located within the same geographic region. In fact, three grassy sites and two open heath sites that were located within the Altona woodland were virtually adjacent to one another.

Also, the five grassy sites included areas of both high and low topographic reliefs, with the two grassy sites at higher elevations nested within closed heath habitats. Intriguingly, the bird species within the two higher elevation grassy sites had less in common with the bird species within the surrounding closed heath habitats, but were more similar to the species using the grassy woodland sites at lower elevations. Similarly, although the “grazed woodland” and “open woodland with few trees” sites were distinct from all other sites in regards to vegetation, these sites hosted similar bird species to the more natural grassy woodland sites.

The bird community that was associated with the closed heath sites also appeared to be relatively faithful to habitat type, despite landscape differences (e.g. sites in gullies vs. atop hills) and high variability in vegetation structure at lower storey levels. For example, while the six closed heath sites delineated equally into two groups based on vegetation, no such difference was evident with the closed heath bird community. These finding are not unexpected from a theoretical and empirical perspective, as a robust generalisation exists between habitat structure and the occupying bird community (e.g. MacArthur 1972; Cody 1985; Gilmore 1985; Major *et al.* 2001; Fischer *et al.* 2008).

The methods used in this study to categorise the woodland types and their bird communities at the scale of ten hectares appears to be sound. There was clear separation

of the woodland types based on both the habitat attributes and the occupying bird species. Also, a total of 25 bird species were statistically associated with one of the three woodland types. Similar studies have also reported that bird species differ significantly among vegetation types within continuous vegetation (e.g. Arnold 1988; Maron and Kennedy 2007).

Results from this study provide broad recommendations that can be integrated into future revegetation works. For instance, by integrating large patches of different woodland types across wide areas should cater for a broader array of woodland bird species. Planting such habitat heterogeneity in ten hectare blocks is manageable for broad-scale revegetation, such as that planned for the Para Woodland Reserve. Such plantings should then enhance the habitat value for multiple bird species.

However, there are limitations in using relatively small spatial scales within a larger habitat continuum. Landscapes consist of multi-scale processes and incorporate a hierarchy of species-specific and scale-dependent patch dynamics (Wu and Loucks 1995). In comparison, this study only considered habitat use within ten hectare sites for one season of one year. This may not adequately allow for dynamic processes that may influence habitat use at other times, which can affect site-level bird densities, species richness, species composition and their distribution (Maron *et al.* 2005).

To reinforce this point, in this study the Crescent Honeyeater was an indicator species for closed heath woodland at higher elevations. This resulted from this bird species only ever being recorded within the closed heath sites (and at elevation) during this study. However, during the fine-scale habitat studies conducted in the following year (in 2009; refer Chapter 8) the Crescent Honeyeater was casually observed using open heath and grassy woodland habitats within the lower lying regions. These open heath and grassy woodlands contained numerous ten hectare survey sites that were surveyed in 2008, and no Crescent Honeyeaters were ever detected during this time. This expansion in geographic location into new habitat types may have been as a result of altered resource availabilities, brought on by the drought conditions that prevailed during the time of this project.

The results of this study did not appear to be unduly compromised by the presence of the Noisy Miner. This aggressive bird species has been shown to adversely affect the presence

of small bodied birds through competitive exclusion (e.g. Grey *et al.* 1997; Major *et al.* 2001; Piper and Catterall 2003). Studies into the occurrence of Noisy Miners typically report that these birds occupy habitats that are largely dominated by various *Eucalyptus* species. Many studies then suggest that the presence of a midstorey stratum (e.g. *Acacia* spp) restricts the occurrence of the Noisy Miner (Lindenmayer *et al.* 2010). Hastings and Beattie (2006) concluded that together with a shrubby understorey, *Eucalyptus* woodlands require at least 15% acacia cover to restrict the influence of the Noisy Miner. Their recommendation was to use *Acacia* species with bipinnate leaves (leaves that divide into smaller leaflets) to reduce the impacts of Noisy Miners on other woodland bird species. Similarly, Maron and Kennedy (2007) reported that Cypress Pine was also a potentially important refuge plant for smaller birds against Noisy Miner attacks.

During this study, I personally observed this to be the case while conducting fine-scale habitat use studies within Site 6 (open grassy woodland). This site was one of two sites in which Noisy Miners were recorded. The plant community within this site contained significant numbers of Slender Cypress-pine and Kangaroo Thorn. Many bird species including the White-browed Babbler and Brown Treecreeper were observed to quickly retreat into these pines and/or Kangaroo Thorn when attacked by Noisy Miners. The reason why Noisy Miners were not a problem in this current study was presumably owing to the fact the sites generally contained *Callitris* and/or *Acacia* species. This type of habitat structure either deterred its use by the Noisy Miner and/or provided adequate refuge protection for other bird species when Noisy Miners were present.

A further noteworthy finding from this study was that only 6 – 7 birds ha⁻¹ were recorded within the three woodland types. These densities are consistent with several previous woodland studies across southern Australia; including 6 – 7 birds ha⁻¹ within *Eucalyptus wandoo* woodland in south-west Western Australia (Arnold 1988) and 7 birds ha⁻¹ within eucalypt remnants in Tasmania (MacDonald and Kirkpatrick 2003). These abundances contrast however with around 23 birds ha⁻¹ in remnants in New South Wales (Briggs *et al.* 2007). At 6 – 7 birds ha⁻¹ though, revegetating the 300 or so hectares of the Para Woodland site would equate to a further 2,000 or so birds occupying this region. This in part would also depend on the area requirements of the individual bird species and the total amount of habitat that was required to support functional populations of these species (see

Chapter 6 for the minimum home range estimates of birds belonging to the nine declining woodland species within the north-central zone of the southern Mount Lofty Ranges).

Importantly though, simply returning habitat of a certain woodland “type” and expecting a certain “bird community” to inhabit, errs towards the “build it and they will come” analogy (Palmer *et al.* 1997; Bennett *et al.* 2000; McAlpine *et al.* 2007). Information on broad-scale habitat attributes helps guide restoration policy, but it only serves to predict general possible trends in faunal response. Critical information regarding species-specific habitat requirements are still missing, together with the scales and/or frequency with which key habitat features are required to enable certain species to occupy particular habitats. The revegetation model should be aimed at also capturing finer-scale ecological requirements. The next two chapters begin to explore this issue, by first examining the degree of structural habitat heterogeneity within each of the 15 ten hectare sites, and then in identifying individual bird species that were statistically associated with particular habitat elements within a ten hectare site.

Chapter 4

Structural habitat heterogeneity within ten hectare sites

Introduction

Structural complexity and floristic richness have long been associated with the abundance and diversity of woodland bird species (MacArthur and MacArthur 1961; Recher 1969; Gilmore 1985; Loyn 1985; Recher 1985a). A suite of habitat attributes have been used to characterise the structure of vegetation stands and provide measures of heterogeneity, which are assumed to be indicative of biodiversity (see McElhinny 2002). Attributes which have been shown to be of benefit to woodland birds both in Australia and elsewhere include the density and diversity of plant material at different heights (Gilmore 1985), various aspects of the foliage (Braithwaite *et al.* 1989; Woinarski *et al.* 1997), shrub cover and shrub species richness (Recher *et al.* 1985b; Seddon *et al.* 2001), density of tree hollows (Fanning 1995; Seddon *et al.* 2001), presence of decorticating bark (Pearce 1996) and ground cover attributes such as litter and woody debris (Laven and Mac Nally 1998; Recher and Davis 1998).

Vegetation complexity has also been positively correlated with species richness for breeding birds, together with a higher number of nest sites that they contain (Recher 1991; Twedt *et al.* 2002). Some more specific habitat attributes have also been directly related to increases in bird species richness. For example, perching substrates one metre from the ground have been shown to increase the number of insectivorous bird species that forage for some part near the ground level (Arnold 2003).

To demonstrate habitat associations by birds, habitat attributes are usually measured in terms of gross characteristics, with the data for each variable reduced to a single value for analysis (e.g. the average or an index value; Van Den Meersschaert and Vandekerckhove 1998; Hewson *et al.* 2011). However, this simplification of the characteristics of a stand of vegetation cannot reveal the underlying nature of habitat heterogeneity within the system being studied. Consequently this does little to inform how birds may be distributed within a habitat, or how they may be using that habitat area.

The measure of habitat heterogeneity should instead be designed to capture changes across spatial scales larger than one to two hectares (Benton *et al.* 2003). Most woodland consists of mosaics of different habitat types; with each habitat type being different from the others. Certain aspects of this heterogeneity may be important in determining which bird species are present (Fisher 2000; Radford and Bennett 2007). For example, it is the habitat features that are contained in patches within larger habitat areas that ultimately determine how most species are distributed (Tilman 1999; Mac Nally *et al.* 2000b; Fisher 2001; Mac Nally *et al.* 2001; Gibbons and Lindenmayer 2002). Accordingly, sampling areas must be sufficiently large in order to capture important habitat heterogeneity when aiming to explain bird species associations with habitats.

This current study asks the question, do the 15 ten hectare sites that were introduced in Chapter 3, contain significant habitat heterogeneity at the ten hectare scale? If so, what is the nature and distribution of this heterogeneity across the different ten hectare sites? This study will assess how variable the vegetation is across the 165 ± 0.9 , 25 m \times 25 m survey squares within each of the ten hectare sites. Answers to these questions can be used to help guide revegetation works, with regards to planting key structural features and incorporating heterogeneity akin to more natural woodland. Chapter 5 then investigates bird species associations with specific habitat elements within each of the ten hectare sites.

Methods

The methods employed to collect the vegetation data are described in Chapter 3. In this chapter, the data used were the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) that were collected at the scale of 25 m \times 25 m across the entirety of each ten hectare site. These data were used to produce a dendrogram for each site in PC-ORD5 (McCune and Mefford 1999b), using the Flexible Beta linkage method (default β of -0.25) with Sorenson (Bray-Curtis) distance measures.

For each ten hectare site, the dendrogram consisted of hierarchical clusters of cells that contained strong within-group homogeneity in their respective habitat attributes. During the computation of the dendrogram, the clusters are progressively formed from left to right. The more distant the relationship is between the individual clusters, the greater the differences become between the clustered groups. A natural break point occurs where long

“stems” are created among the clusters, producing “natural” cluster groups. Provisional cluster groups were initially identified by ruling a single vertical line through the first natural break point (method referred to as “pruning”). The point where this vertical line intersects with a “stem” identifies each new cluster group. The aim is to produce a small number of cluster groups that yield strong within-group homogeneity. Applying a step-wise approach and moving from left to right in the direction of data usage, I initially assessed the within-group homogeneity at the first major natural break point. This was done by visually comparing the percent cover for each habitat attribute against that of the other cluster groups. If these cluster groups had formed due to only minor variations in the percent cover of the habitat variables, I continued pruning the dendrogram to the right until defined within-group homogeneity became evident for a ten hectare site. The cut-off thresholds that were used to delineate between cluster groups were: > 30% cover for grasses, > 10% cover for weeds, > 20% cover for shrubs, and > 30% cover for both midstorey and overstorey vegetation.

Non-parametric Multi-response Permutation Procedures (MRPP) with Sørensen (Bray-Curtis) distance measures in PC-ORD5 were then used to test the null hypothesis that no differences exist among the cluster groups of 25 m × 25 m cells that had been created. Regularized Discriminant Analysis with stepwise entry of the variables (using JMP10 2012) was also used to test the accuracy of the cluster classifications and determination which (if any) of the environmental variables discriminate well (this analysis is described in Chapter 3). For these analyses, a Lambda value of one was used. A high Lambda should be used when variables are uncorrelated (SAS Institute Inc. 2009). I considered that the vegetation that was measured at the different storey levels were unlikely to be strongly spatially correlated. The Gamma value was retained at zero to maintain minimal shrinkage of the covariance across the variables (SAS Institute Inc. 2009). The misclassification risk (as described in Chapter 3) was used to assess the chance that a 25 m × 25 m cell had been incorrectly assigned to the wrong cluster group.

Results

The dendograms created for the 15 sites averaged 4.1 ± 0.3 cluster groups each (Table 4.1), based upon the habitat attributes that were measured (from here these clusters are referred to as “vegetation clusters”). Cluster analysis for each of the 15 sites revealed that there was clear separation between the vegetation clusters (low T values) and strong within-group homogeneity (average A of 0.31) that was unlikely to be due to chance (very highly significant delta p values; Table 4.1).

Regularized Discrimination Analysis calculated that on average, $9.0 \pm 0.7\%$ of the 25 m × 25 m squares were misclassified within each of the ten hectare sites (Table 4.2). The grid of 25 m × 25 m squares had been superimposed in Arc-GIS across each of the ten hectare sites, purely as a means to guide the bird and vegetation surveys (refer Chapter 3 for details). As the 25 m × 25 m grids were neither designed nor intended to delineate structural habitat differences at this scale, the misclassification rate of less than 10% was considered to be acceptable for the purpose of this study.

Table 4.1: Non-parametric Multi-response Permutation Procedures (MRPP) with Sørensen (Bray-Curtis) distance measures were used to test the null hypothesis that no differences existed among the hierarchical clusters that were identified for each of the 15 sites. The test statistic T describes heterogeneity between the groups (heterogeneity increasing with decreasing T), agreement statistic A denotes within-group homogeneity (with $A > 0.3$ indicating high homogeneity within the detected clusters; McCune and Grace 2002), and delta p evaluating if the observed differences were significantly different from random. Habitat attributes were measured at a scale of 25 m × 25 m across the entire area within each of the 15 ten hectare sites. The attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The cluster groups within each site contain *a priori* within-group homogeneity.

| Site | # of clusters | T | A | delta p |
|------|---------------|--------|------|-----------|
| 1 | 6 | -56.67 | 0.38 | < 0.0001 |
| 2 | 5 | -58.99 | 0.36 | < 0.0001 |
| 3 | 3 | -60.17 | 0.27 | < 0.0001 |
| 4 | 3 | -35.18 | 0.18 | < 0.0001 |
| 5 | 3 | -67.99 | 0.31 | < 0.0001 |
| 6 | 3 | -59.01 | 0.22 | < 0.0001 |
| 7 | 4 | -59.69 | 0.37 | < 0.0001 |
| 8 | 4 | -62.09 | 0.28 | < 0.0001 |
| 9 | 3 | -71.79 | 0.26 | < 0.0001 |
| 10 | 3 | -63.34 | 0.23 | < 0.0001 |
| 11 | 3 | -57.07 | 0.22 | < 0.0001 |
| 12 | 5 | -63.94 | 0.45 | < 0.0001 |
| 13 | 6 | -67.89 | 0.44 | < 0.0001 |
| 14 | 4 | -61.51 | 0.32 | < 0.0001 |
| 15 | 4 | -62.65 | 0.30 | < 0.0001 |

Table 4.2: Number (and percent) of 25 m × 25 m survey squares that were misclassified within each ten hectare site, using Regularized Discriminant Analysis (JMP10 2012). The entire areas of all 15 ten hectare sites were surveyed at the scale of 25 m × 25 m (165 ± 0.9 squares per site). Habitat attributes measured within each 25 m × 25 m square were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. Classifications for the 25 m × 25 m squares were based on their respective association within the hierarchical clusters of the dendrogram for each site, using PC-ORD5 (McCune and Mefford 1999b).

| Site | # 25 m × 25 m squares | # of clusters | # misclassified | % misclassified |
|------|-----------------------|---------------|-----------------|-----------------|
| 1 | 166 | 6 | 18 | 10.8 |
| 2 | 163 | 5 | 15 | 9.2 |
| 3 | 162 | 3 | 20 | 12.4 |
| 4 | 157 | 3 | 7 | 4.5 |
| 5 | 169 | 3 | 11 | 6.5 |
| 6 | 165 | 3 | 19 | 11.5 |
| 7 | 170 | 4 | 15 | 8.8 |
| 8 | 164 | 4 | 15 | 9.1 |
| 9 | 168 | 3 | 8 | 4.8 |
| 10 | 169 | 3 | 15 | 8.9 |
| 11 | 166 | 3 | 21 | 12.7 |
| 12 | 161 | 5 | 9 | 5.6 |
| 13 | 169 | 6 | 16 | 9.5 |
| 14 | 163 | 4 | 14 | 8.6 |
| 15 | 165 | 4 | 19 | 11.5 |

Site 1 – grassy woodland

The cluster analysis undertaken on the habitat attributes for Site 1 identified six vegetation clusters and an outlier (Fig. 4.1). The outlier resulted from a single 25 m × 25 m survey square which contained very low percent covers of all the habitat variables that were measured. Regularized Discrimination Analysis for the vegetation clusters identified five functions, with the first three accounting for 99% of the variance (Table 4.3). Group association for the 25 m × 25 m squares within these three functions were well predicted (based on Wilks' Lambda), with all three being very highly significant (Table 4.4).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. The habitat attributes with the strongest discriminatory value in descending order were overstorey cover, grasses and midstorey cover (Table 4.5; Fig. 4.2). The least discriminatory variables in ascending order were shrubs and weeds (Table 4.5). The habitat structure within each of the 25 m × 25 m squares appeared to be intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.3).

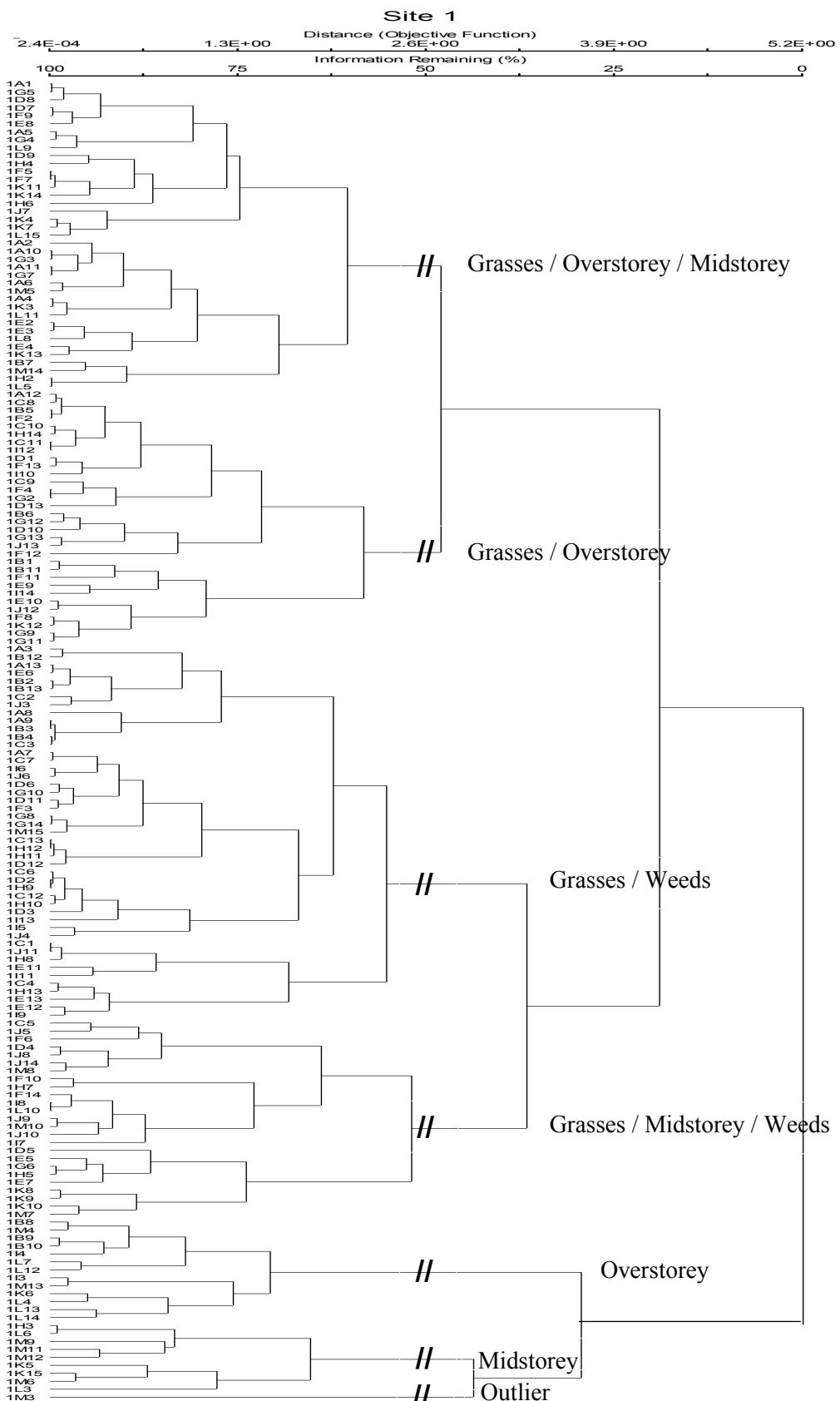


Figure 4.1: Hierarchical clustering of 166, 25 m × 25 m squares used to survey the entire ten hectares of Site 1 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained. The outlier contained low percent cover of all the habitat attributes that were measured.

Table 4.3: Eigenvalues for the five functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 1 (grassy woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first three functions accounting for 99.03% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 4.05 | 55.12 | 55.12 | 0.895 |
| 2 | 2.18 | 29.70 | 84.82 | 0.828 |
| 3 | 1.04 | 14.21 | 99.03 | 0.715 |
| 4 | 0.06 | 0.78 | 99.81 | 0.233 |
| 5 | 0.01 | 0.19 | 100.00 | 0.116 |

Table 4.4: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 1 (grassy woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance ($\text{Prob} > F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value ($\text{Prob} > F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis the first three functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob > F |
|-------------------|---------------|-------------|-------|-------|-------|------------|
| 1 | 0.028 | 29.78 | 30 | 622 | 18009 | < 0.0001 |
| 2 | 0.143 | 20.63 | 20 | 518 | 9823 | < 0.0001 |
| 3 | 0.456 | 11.96 | 12 | 416 | 4565 | < 0.0001 |
| 4 | 0.933 | 1.86 | 6 | 316 | 1575 | 0.087 |
| 5 | 0.986 | 1.09 | 2 | 159 | 158 | 0.339 |

Table 4.5: Average percent cover of habitat attributes within the six vegetation clusters (and an outlier) for Site 1 (grassy woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 166$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the six statistically distinct vegetation clusters. Each vegetation cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes it contained. The vegetation clusters are 1. grasses, overstorey cover and midstorey cover (GOM), 2. grasses and overstorey cover (GO), 3. grasses and weeds (GW), 4. grasses, midstorey cover and weeds (GMW), 5. overstorey cover (O), 6. midstorey cover (M), and 7. outlier. Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the vegetation cluster classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each vegetation cluster (dominant attributes are in bold). n = the number of 25 m × 25 m squares per vegetation cluster. This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.

| Habitat attribute | GOM (n = 39) | GO (n = 32) | GW (n = 47) | GMW (n = 25) | O (n = 13) | M (n = 9) | Outlier (n = 1) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|-----------------|----------------|----------------|-----------------|---------------|--------------|--------------------|--------------------|--|
| Grasses | 63.1 | 62.9 | 66.6 | 60.8 | 20.3 | 18.9 | 15.0 | 52.93 | 0.68 |
| Weeds | 5.3 | 7.5 | 12.8 | 14.0 | 2.3 | 3.8 | 0.5 | 6.82 | 0.53 |
| Shrubs | 4.8 | 5.8 | 2.5 | 4.8 | 6.4 | 9.8 | 2.5 | 2.86 | 0.06 |
| Midstorey | 46.4 | 11.2 | 8.2 | 44.8 | 20.8 | 63.8 | 0.5 | 44.51 | 0.12 |
| Overstorey | 50.3 | 56.3 | 13.1 | 9.2 | 75.4 | 28.1 | 0.0 | 63.15 | -0.79 |

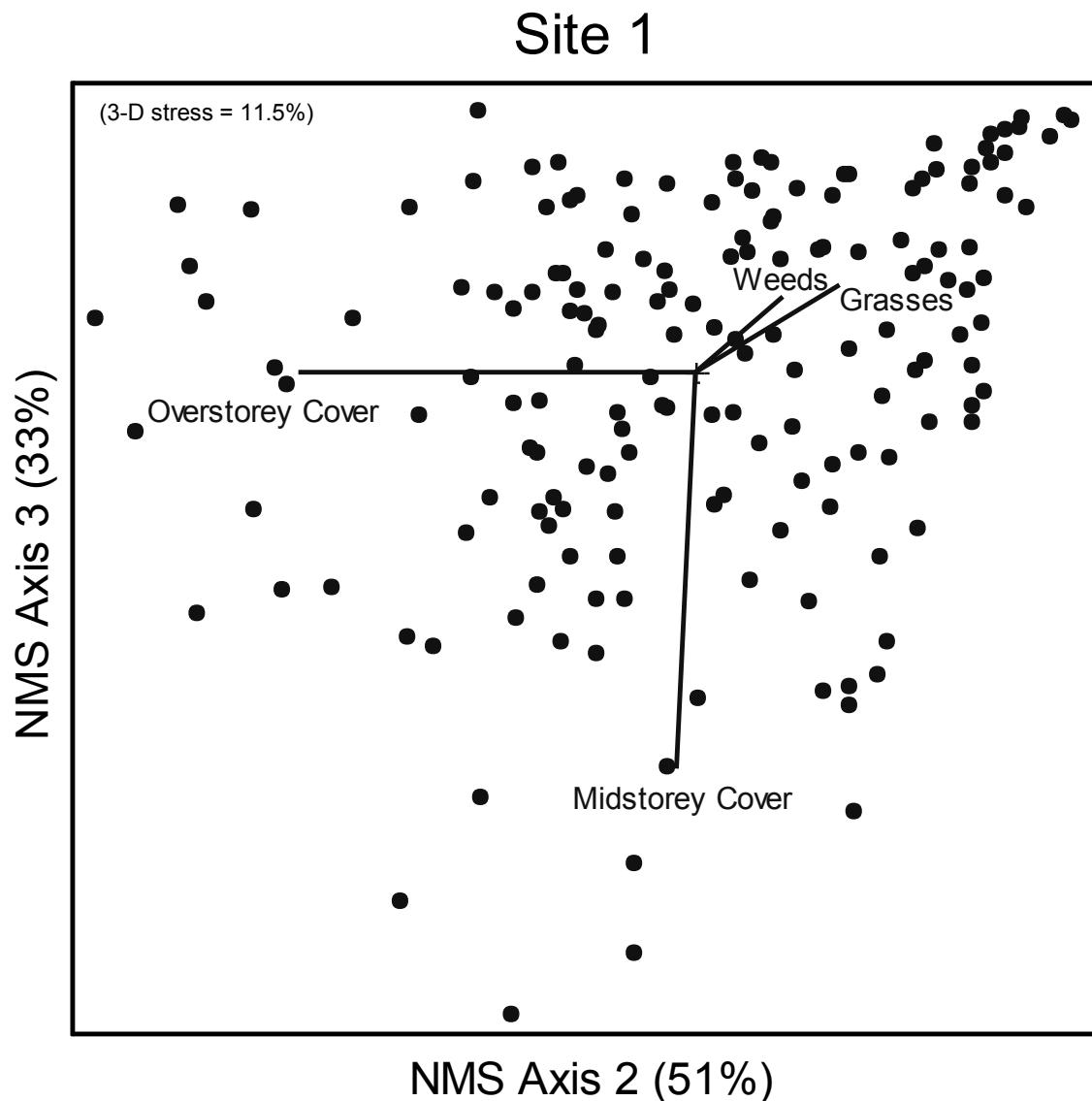


Figure 4.2: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 1 (grassy woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 166, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 11.5%, Axes 2 and 3 accounts for 84% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.

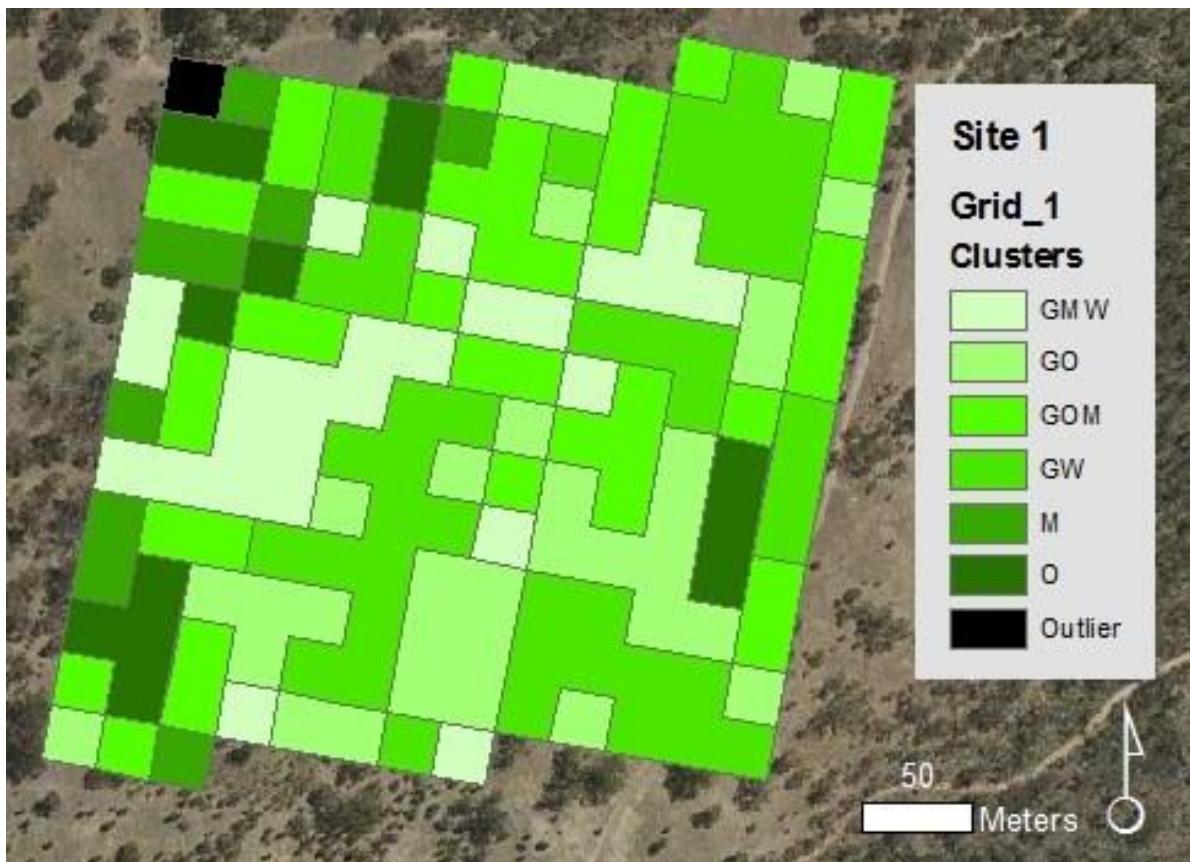


Figure 4.3: Habitat heterogeneity within Site 1 (grassy woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 166$) is colour-coded for the dendrogram cluster it aligned with. The six clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses, midstorey cover and weeds (GMW), 2. grasses and overstorey cover (GO), 3. grasses, overstorey cover and midstorey cover (GOM), 4. grasses and weeds (GW), 5. midstorey cover (M), 6. overstorey cover (O), and 7. outlier. This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.

Site 2 – closed heath woodland

The cluster analysis undertaken on the habitat attributes for Site 2 identified five vegetation clusters (Fig. 4.4). Regularized Discrimination Analysis for the vegetation clusters identified four functions, with the first three accounting for 99% of the variance (Table 4.6). Group association for the 25 m × 25 m squares within these three functions were well predicted (based on Wilks' Lambda), with all three being very highly significant (Table 4.7).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were midstorey cover, shrubs and overstorey cover (Table 4.8; Fig. 4.5). The least discriminatory variables in ascending order were weeds and grasses (Table 4.8). The habitat structure within each of the 25 m × 25 m squares appeared to be intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.6).

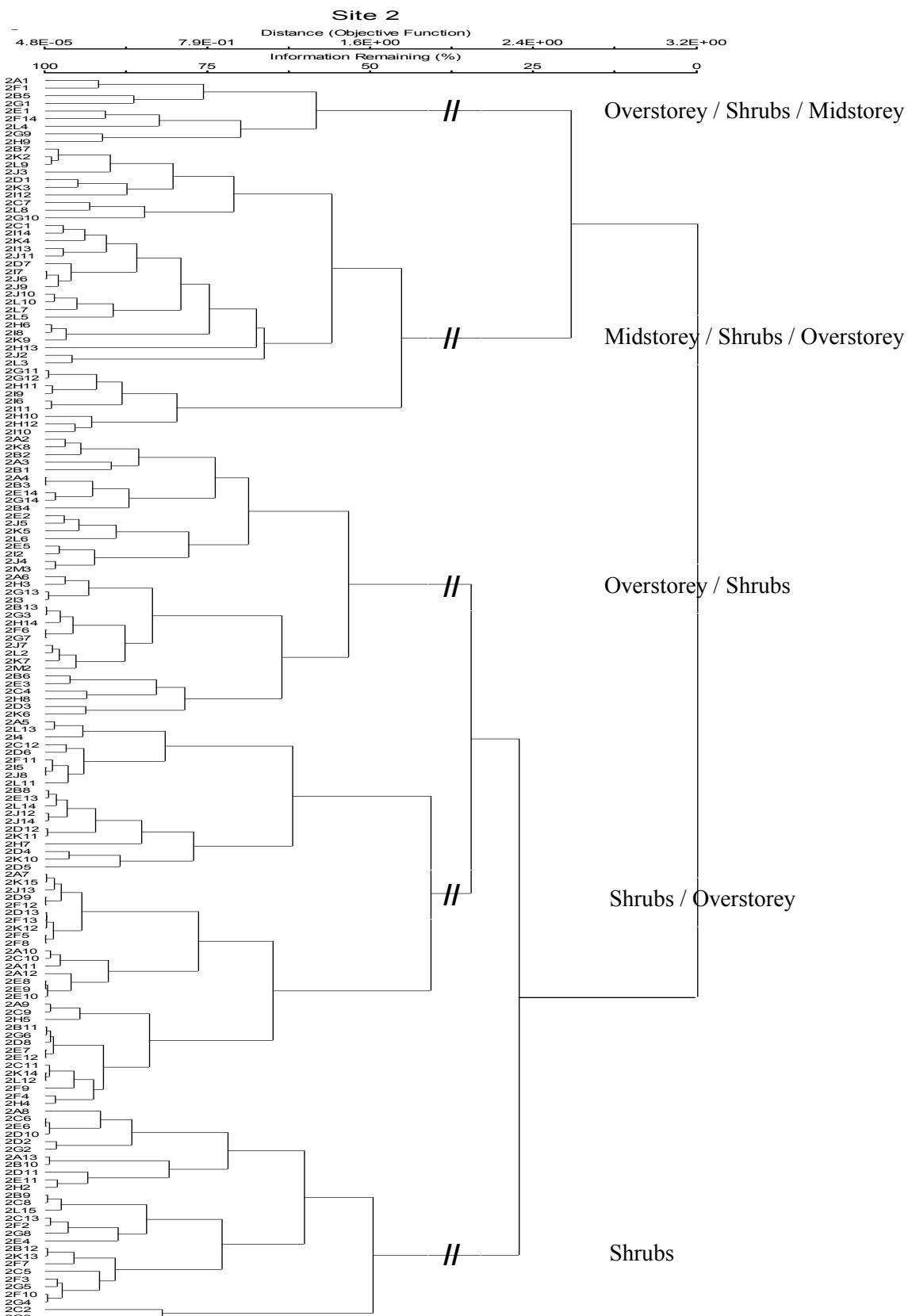


Figure 4.4: Hierarchical clustering of 163, 25 m × 25 m squares used to survey the entire ten hectares of Site 2 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.6: Eigenvalues for the four functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 2 (closed heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first three functions accounting for 99.54% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 5.31 | 74.19 | 74.19 | 0.917 |
| 2 | 1.21 | 16.89 | 91.08 | 0.740 |
| 3 | 0.61 | 8.46 | 99.54 | 0.614 |
| 4 | 0.03 | 0.46 | 100.00 | 0.177 |

Table 4.7: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 2 (closed heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob $> F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value (Prob $> F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis the first three functions were very highly significant and the fourth was significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob $> F$ |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.043 | 53.17 | 16 | 474 | 7095 | < 0.0001 |
| 2 | 0.273 | 29.72 | 9 | 380 | 3032 | < 0.0001 |
| 3 | 0.603 | 22.56 | 4 | 314 | 939 | < 0.0001 |
| 4 | 0.969 | 5.14 | 1 | 158 | - | 0.025 |

Table 4.8: Average percent cover of habitat attributes within the five vegetation clusters for Site 2 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 163$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the five statistically distinct vegetation clusters. Each vegetation cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The vegetation clusters are 1. overstorey cover, shrubs and midstorey cover (OSM), 2. midstorey cover, shrubs and overstorey cover (MSO), 3. overstorey cover and shrubs (OS), 4. shrubs and overstorey cover (SO), and 5. shrubs (S). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the vegetation cluster classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each vegetation cluster (dominant attributes are in bold). n = the number of 25 m × 25 m squares per vegetation cluster. This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.

| Habitat attribute | OSM ($n = 9$) | MSO ($n = 38$) | OS ($n = 37$) | SO ($n = 51$) | S ($n = 28$) | Univariate F -ratio | Standardized discriminant function coefficient |
|-------------------|--------------------|---------------------|--------------------|--------------------|-------------------|-----------------------|--|
| Grasses | 1.1 | 0.6 | 1.5 | 0.3 | 0.6 | 12.53 | 0.04 |
| Weeds | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.00 | n/a |
| Shrubs | 47.5 | 45.4 | 47.1 | 70.1 | 63.5 | 39.52 | 0.14 |
| Midstorey | 40.4 | 70.8 | 13.0 | 20.1 | 10.3 | 199.47 | 1.08 |
| Overstorey | 49.3 | 40.1 | 67.3 | 55.1 | 28.6 | 25.05 | 0.25 |

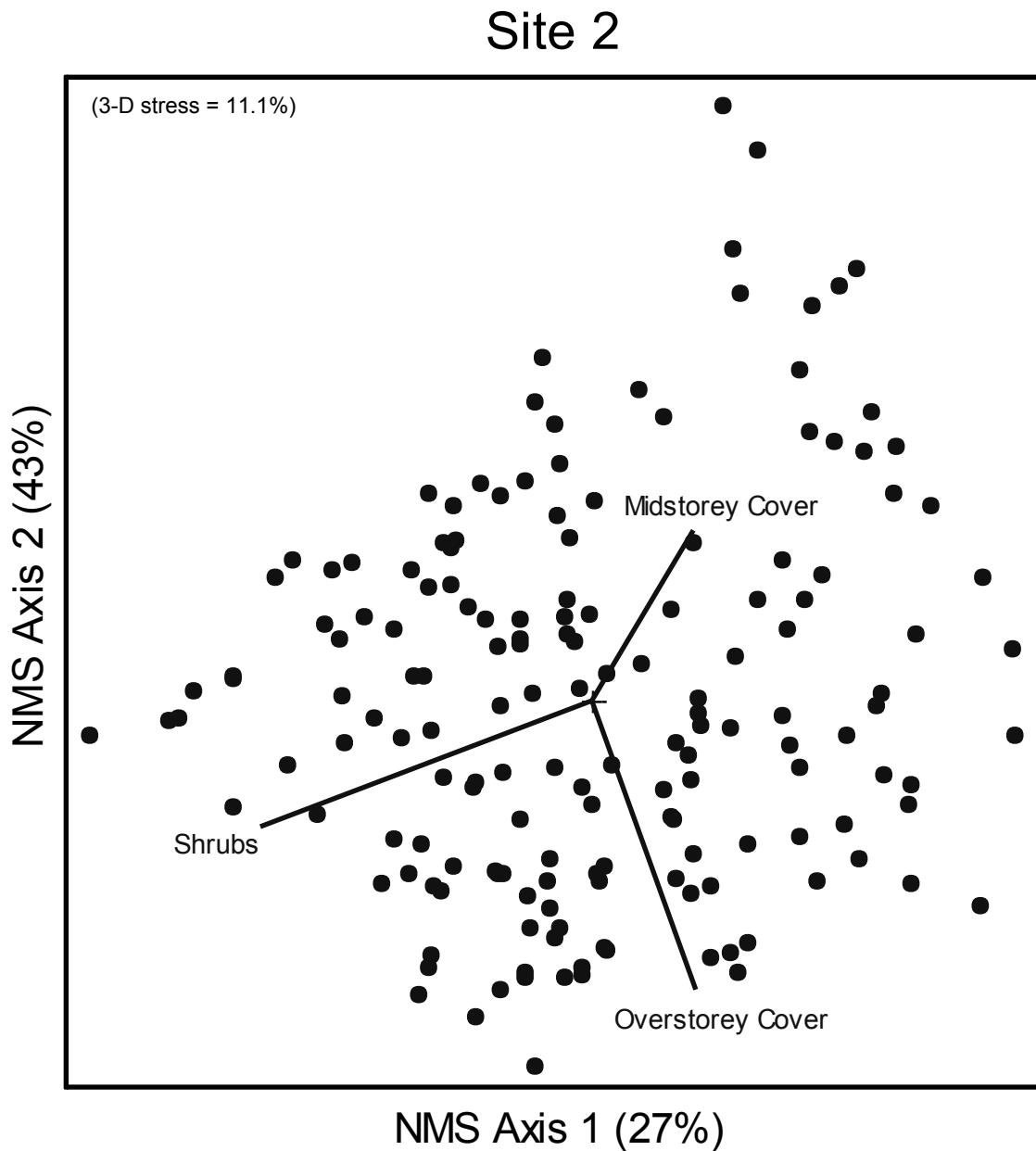


Figure 4.5: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 2 (closed heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 163, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 11.1%, Axes 1 and 2 accounts for 70% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.

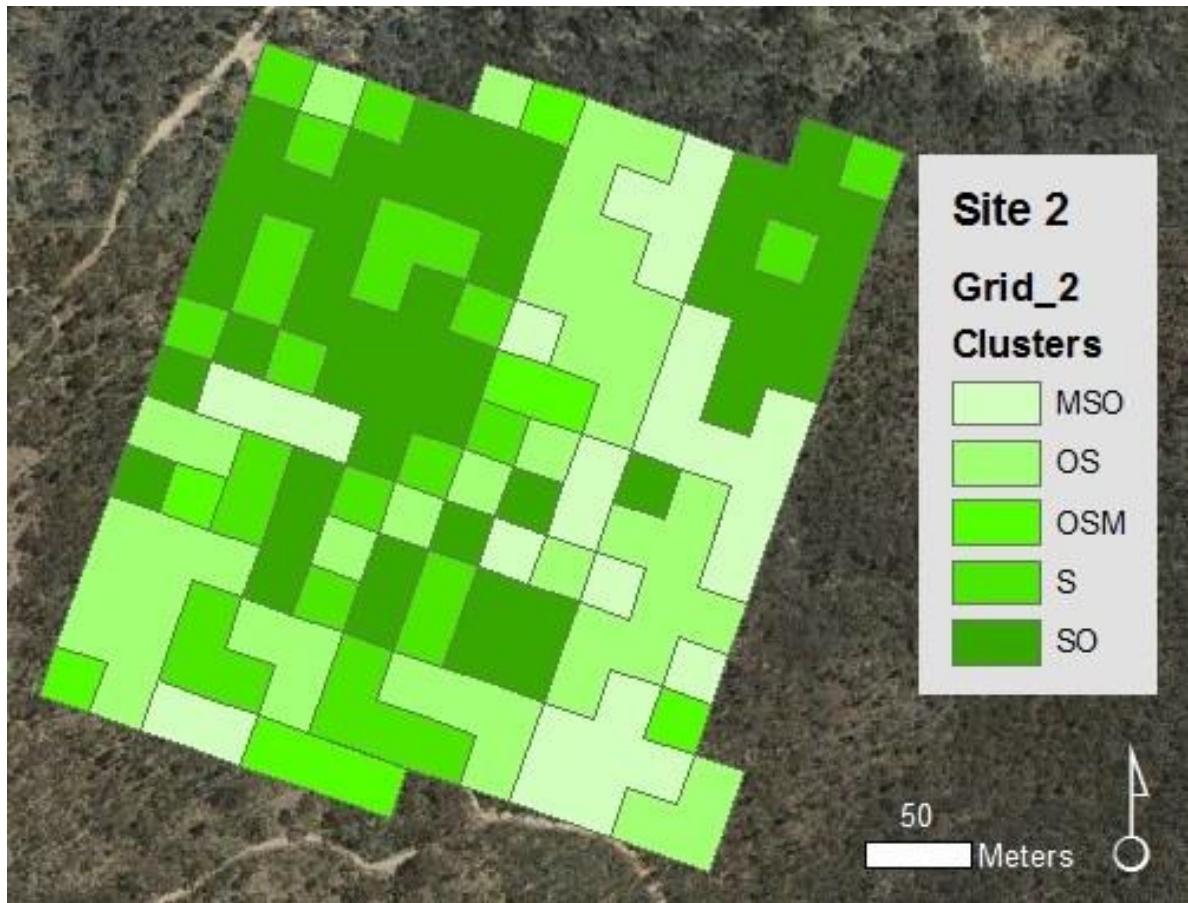


Figure 4.6: Habitat heterogeneity within Site 2 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 163$) is colour-coded for the dendrogram cluster it aligned with. The five clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover, shrubs and overstorey cover (MSO), 2. overstorey cover and shrubs (OS), 3. overstorey cover, shrubs and midstorey cover (OSM), 4. shrubs (S), and 5. shrubs and overstorey cover (SO). This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.

Site 3 – closed heath woodland

The cluster analysis undertaken on the habitat attributes for Site 3 identified three vegetation clusters (Fig. 4.7). Regularized Discrimination Analysis for the vegetation clusters identified two functions, with the first accounting for 76% of the variance (Table 4.9). Group association for the 25 m × 25 m squares within these two functions were well predicted (based on Wilks' Lambda), with both being very highly significant (Table 4.10).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were overstorey cover, midstorey cover and shrubs (Table 4.11; Fig. 4.8). The least discriminatory variables in ascending order were grasses and weeds (Table 4.11). The habitat structure within each of the 25 m × 25 m squares appeared to range from spatially aggregated with similar habitat features to intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.9).

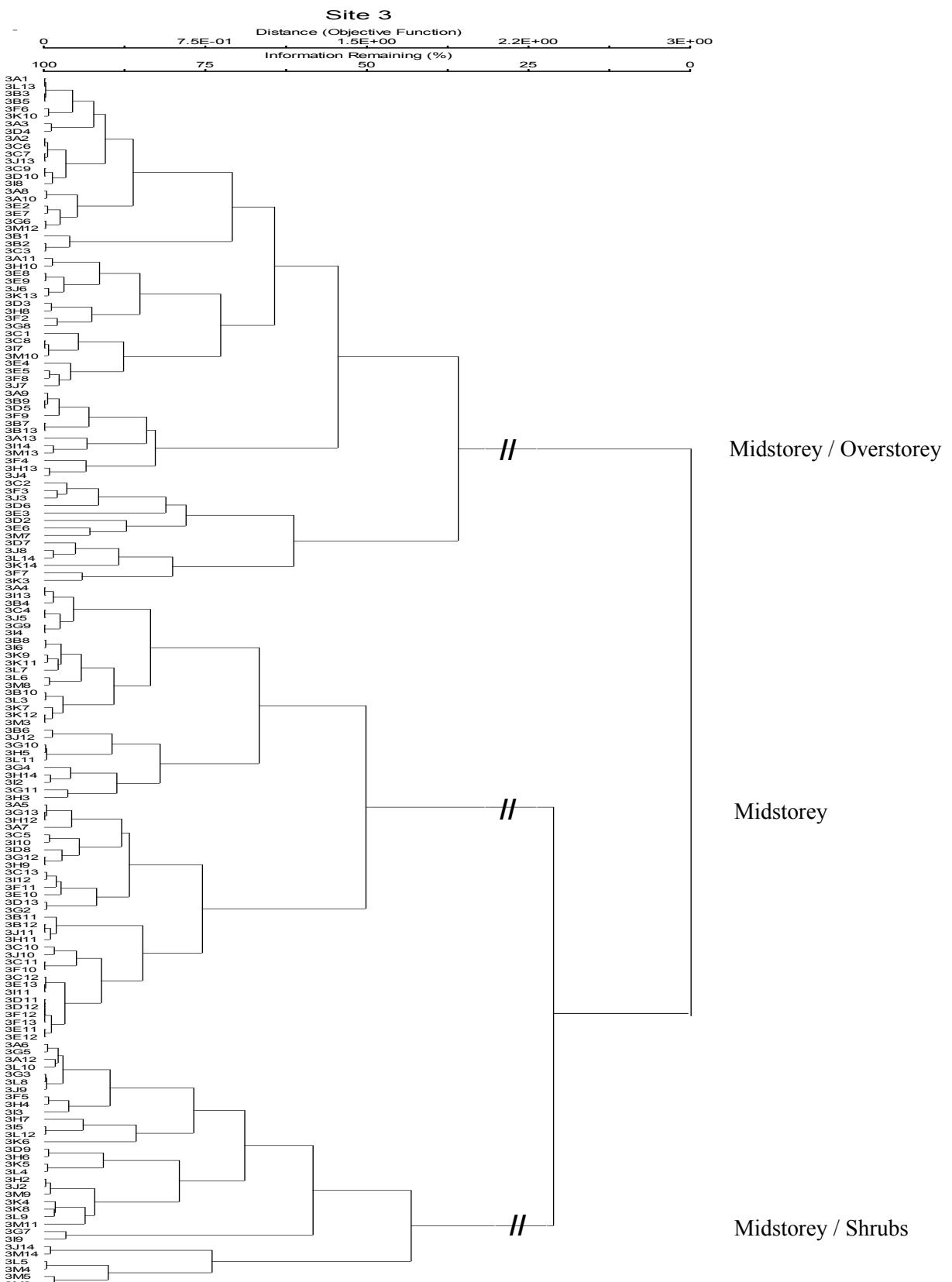


Figure 4.7: Hierarchical clustering of 162, 25 m × 25 m squares used to survey the entire ten hectares of Site 3 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.9: Eigenvalues for the two functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 3 (closed heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first function accounting for 75.72% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 1.90 | 75.72 | 75.72 | 0.809 |
| 2 | 0.61 | 24.28 | 100.00 | 0.615 |

Table 4.10: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 3 (closed heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob $> F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value (Prob $> F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis both functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob $> F$ |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.214 | 35.99 | 10 | 310 | 2781 | < 0.0001 |
| 2 | 0.621 | 23.75 | 4 | 156 | 465 | < 0.0001 |

Table 4.11: Average percent cover of habitat attributes within the three cluster types for Site 3 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 162$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the three statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover and overstorey cover (MO), 2. midstorey cover (M), and 3. midstorey cover and shrubs (MS). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.

| Habitat attribute | MO ($n = 68$) | M ($n = 61$) | MS ($n = 33$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|--------------------|-------------------|--------------------|--------------------|--|
| Grasses | 1.3 | 0.6 | 0.5 | 3.60 | 0.03 |
| Weeds | 0.0 | 0.0 | 0.0 | 0.83 | -0.08 |
| Shrubs | 20.9 | 23.1 | 40.8 | 20.67 | -0.28 |
| Midstorey | 60.7 | 80.8 | 52.4 | 52.65 | -0.24 |
| Overstorey | 44.6 | 10.7 | 18.5 | 132.21 | -0.93 |

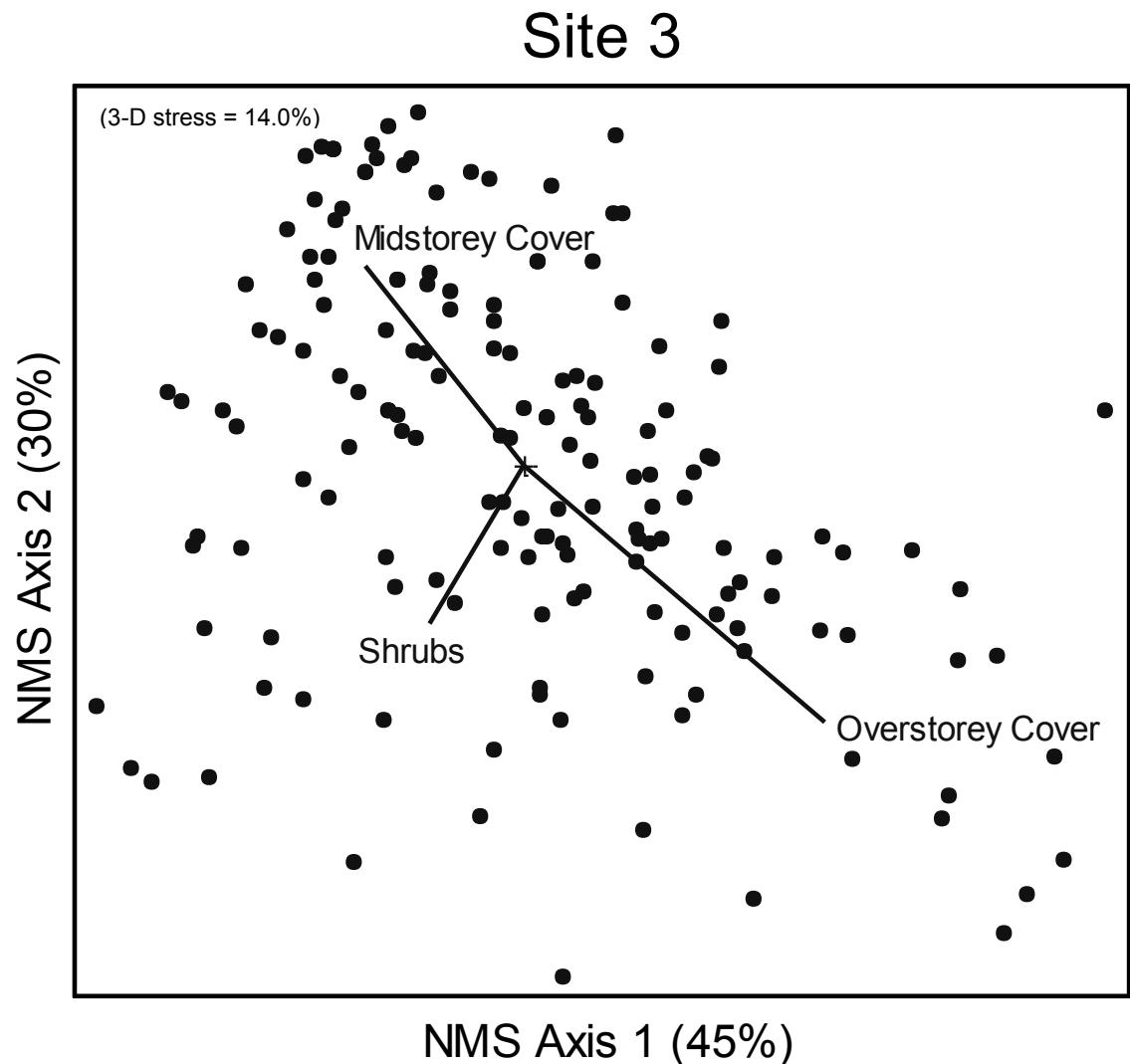


Figure 4.8: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 3 (closed heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 162, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 14.0%, Axes 1 and 2 accounts for 75% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.



Figure 4.9: Habitat heterogeneity within Site 3 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 162$) is colour-coded for the dendrogram cluster it aligned with. The three clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover (M), 2. midstorey cover and overstorey cover (MO), and 3. midstorey cover and shrubs (MS). This site is located within the Para Wirra Recreation Park, South Australia. Data collected November 2008.

Site 4 – open woodland with few trees

The cluster analysis undertaken on the habitat attributes for Site 4 identified three vegetation clusters (Fig. 4.10). Regularized Discrimination Analysis for the vegetation clusters identified two functions, with the first accounting for 76% of the variance (Table 4.12). Group association for of the 25 m × 25 m squares within these two functions were well predicted (based on Wilks' Lambda), with both being very highly significant (Table 4.13).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were shrubs, overstorey cover and grasses (Table 4.14; Fig. 4.11). The least discriminatory variables in ascending order were midstorey cover and weeds (Table 4.14). The habitat structure across this ten hectare site was very open with sparse shrubs, midstorey and overstorey vegetation (Fig. 4.12).

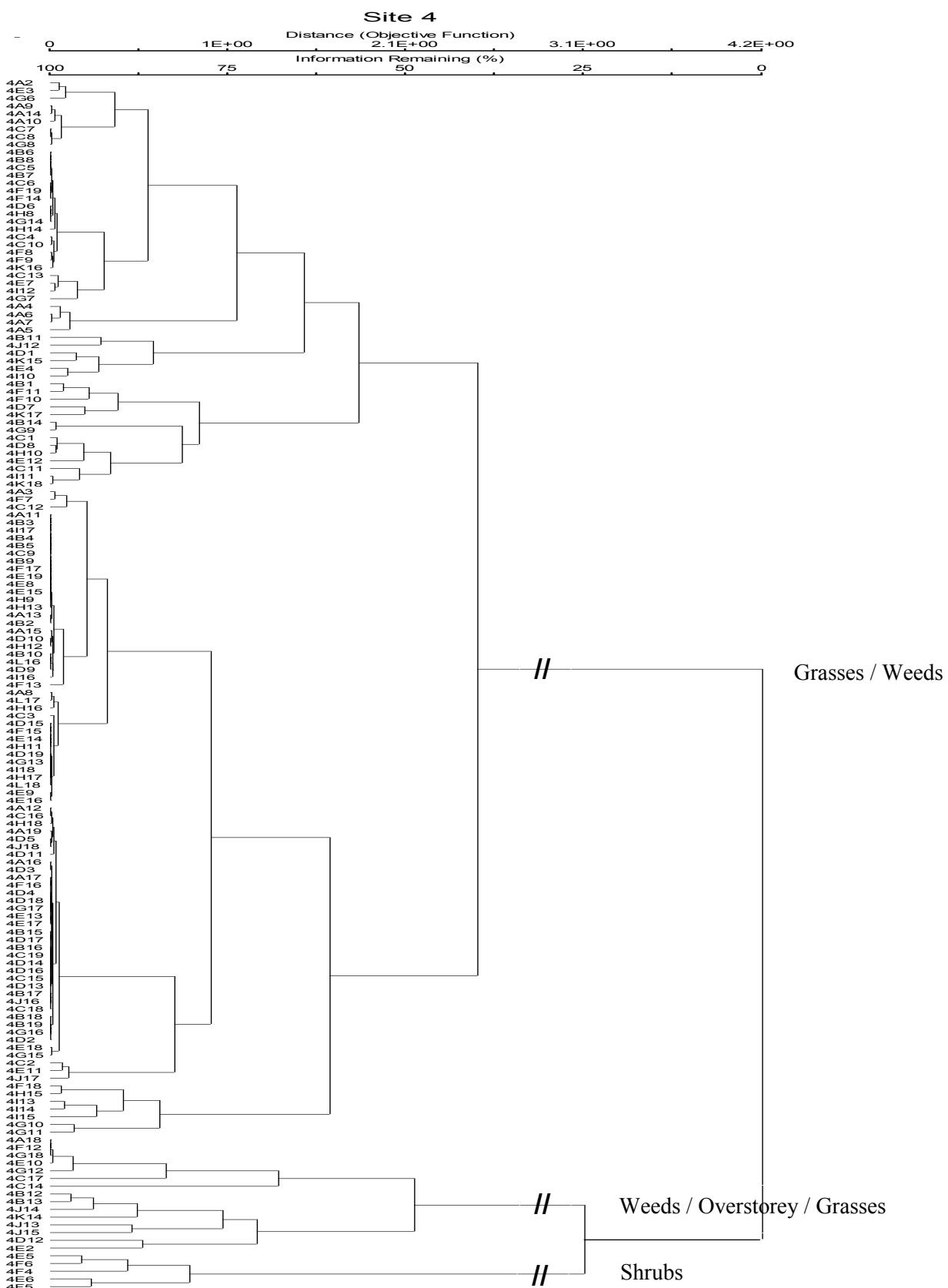


Figure 4.10: Hierarchical clustering of 157, 25 m × 25 m squares used to survey the entire ten hectares of Site 4 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.12: Eigenvalues for the two functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 4 (open woodland with few trees). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first function accounting for 76.38% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 2.47 | 76.38 | 76.38 | 0.844 |
| 2 | 0.76 | 23.62 | 100.00 | 0.658 |

Table 4.13: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 4 (open woodland with few trees). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob $> F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value (Prob $> F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis both functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob $> F$ |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.163 | 44.22 | 10 | 300 | 2691 | < 0.0001 |
| 2 | 0.567 | 28.83 | 4 | 151 | 450 | < 0.0001 |

Table 4.14: Average percent cover of habitat attributes within the three cluster types for Site 4 (open woodland with few trees). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 157$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the three statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses and weeds (GW), 2. weeds, overstorey cover and grasses (WOG), and 3. shrubs (S). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Para Wirra Recreation Park, South Australia. Data collected October 2008.

| Habitat attribute | GW ($n = 137$) | WOG ($n = 15$) | S ($n = 5$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|---------------------|---------------------|------------------|--------------------|--|
| Grasses | 63.7 | 32.7 | 18.4 | 67.15 | 0.20 |
| Weeds | 29.9 | 36.5 | 7.2 | 6.67 | -0.34 |
| Shrubs | 2.5 | 19.3 | 64.4 | 137.01 | -1.05 |
| Midstorey | 1.5 | 6.7 | 14.0 | 23.09 | -0.21 |
| Overstorey | 3.8 | 36.0 | 0.4 | 34.39 | -0.24 |

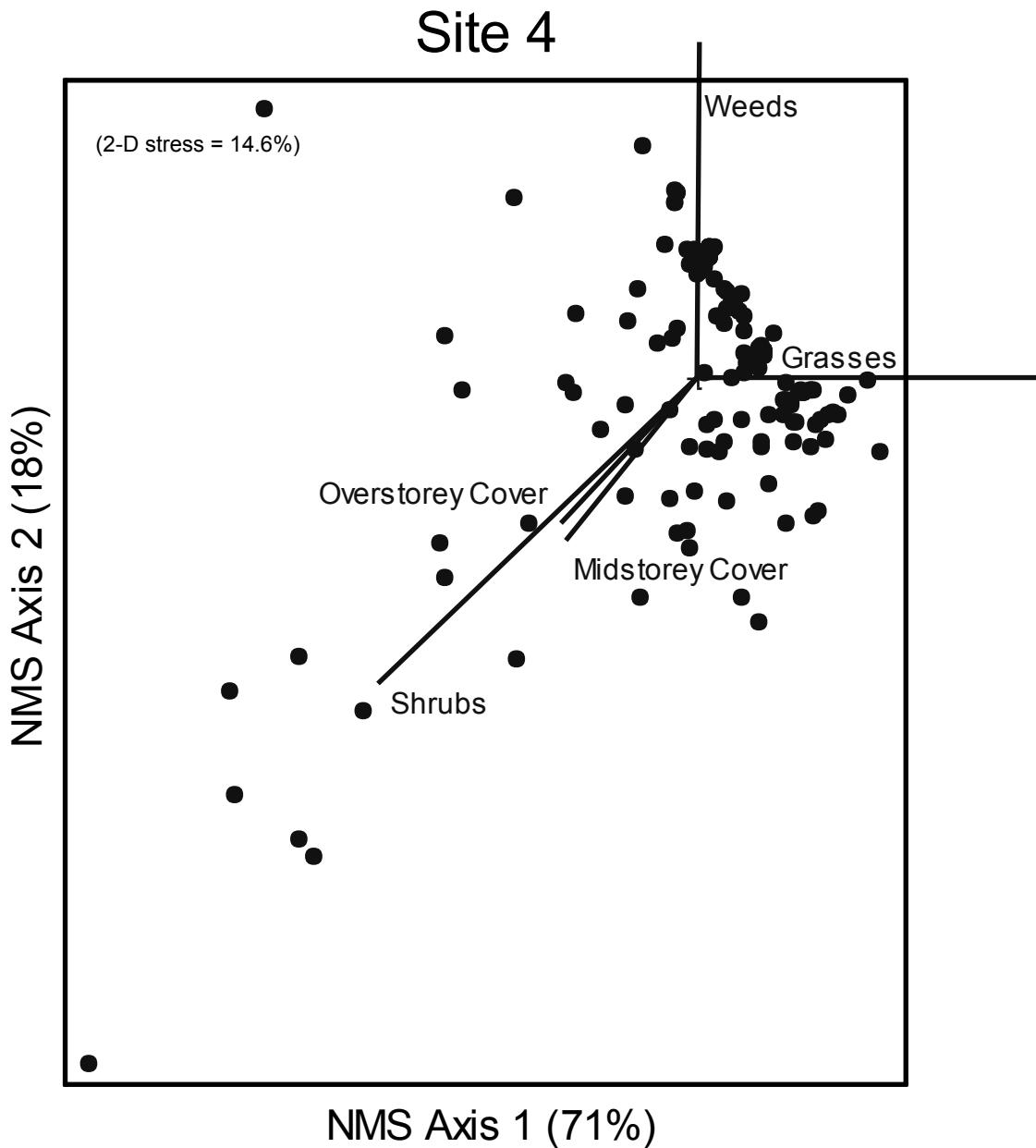


Figure 4.11: Two-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 4 (open woodland with few trees). The data points (black circles) are the 157, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 14.6%, Axes 1 and 2 accounts for 89% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Para Wirra Recreation Park, South Australia. Data collected October 2008.

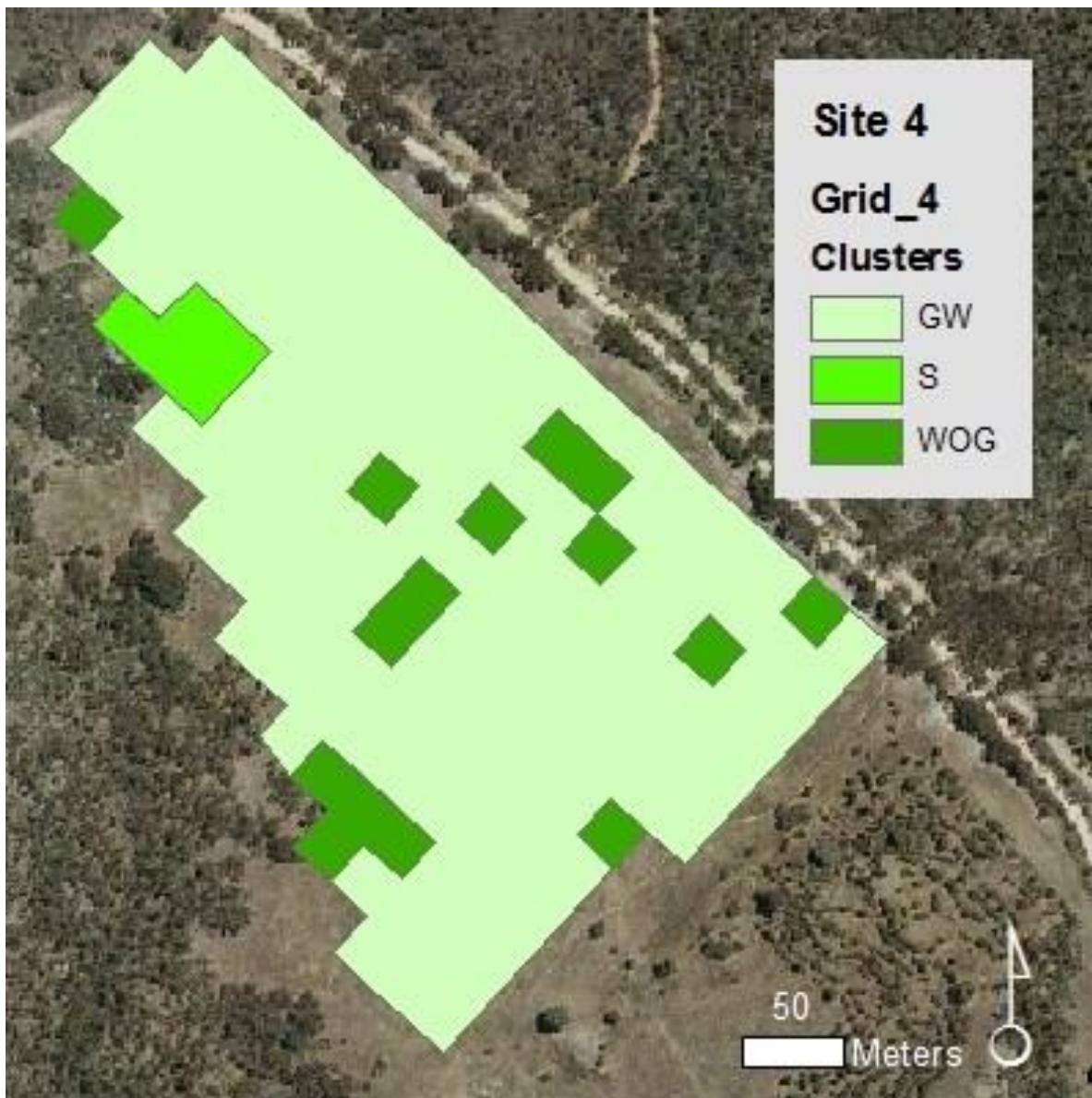


Figure 4.12: Habitat heterogeneity within Site 4 (open woodland with few trees). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 157$) is colour-coded for the dendrogram cluster it aligned with. The three clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses and weeds (GW), 2. shrubs (S), and 3. weeds, overstorey cover and grasses (WOG). This site is located within the Para Wirra Recreation Park, South Australia. Data collected October 2008.

Site 5 – closed heath woodland

The cluster analysis undertaken on the habitat attributes for Site 5 identified three vegetation clusters (Fig. 4.13). Regularized Discrimination Analysis for the vegetation clusters identified two functions, with the first accounting for 70% of the variance (Table 4.15). Group association for the 25 m × 25 m squares within these two functions were well predicted (based on Wilks' Lambda), with both being very highly significant (Table 4.16).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were midstorey cover, shrubs and overstorey cover (Table 4.17; Fig. 4. 14). The least discriminatory variables in ascending order were weeds and grasses (Table 4.17). The habitat structure within each of the 25 m × 25 m squares appeared to range from spatially aggregated with similar habitat features to intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.15).

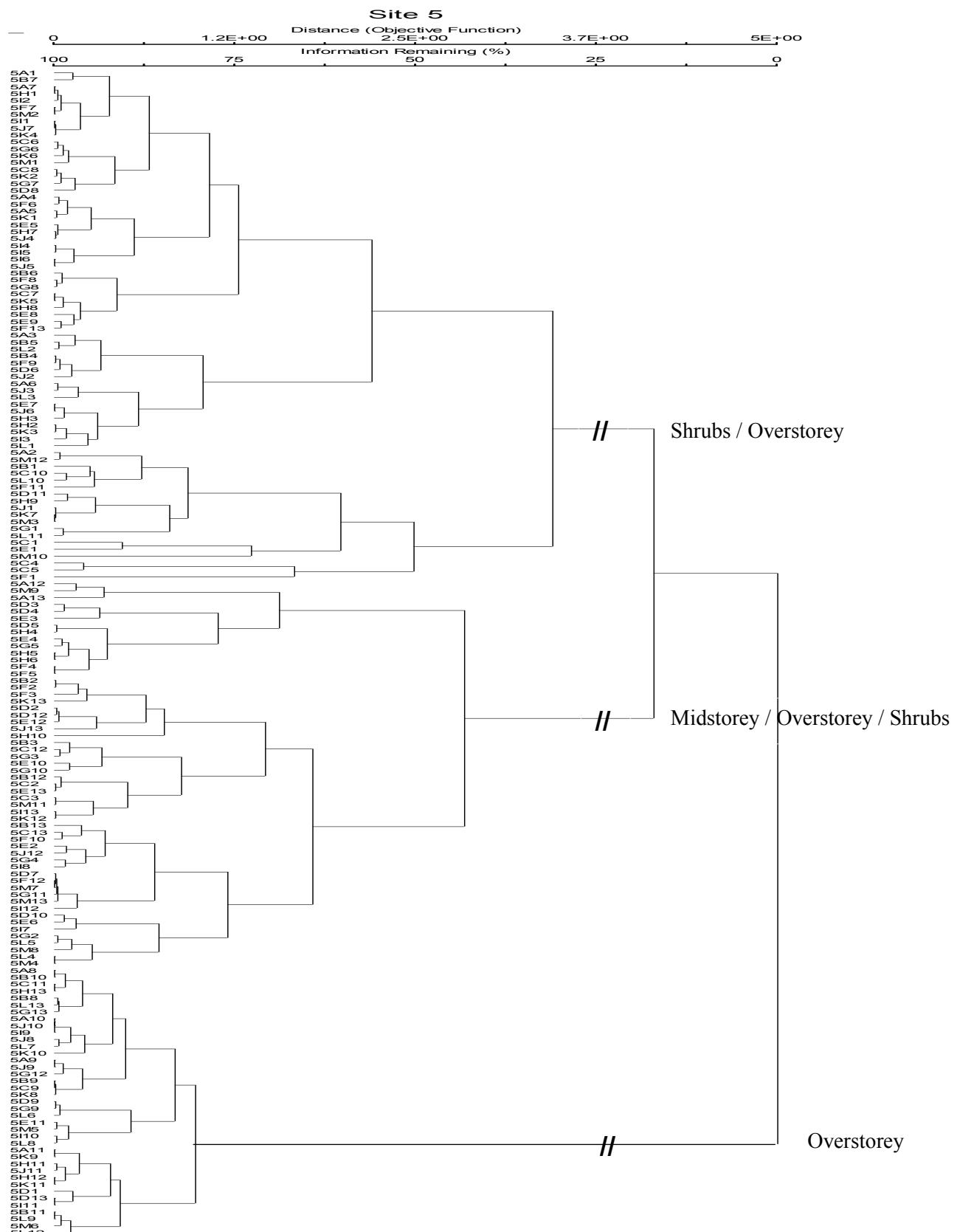


Figure 4.13: Hierarchical clustering of 169, 25 m × 25 m squares used to survey the entire ten hectares of Site 5 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.15: Eigenvalues for the two functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 5 (closed heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first function accounting for 70.13% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 2.49 | 70.13 | 70.13 | 0.845 |
| 2 | 1.06 | 29.87 | 100.00 | 0.718 |

Table 4.16: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 5 (closed heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob $> F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value (Prob $> F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis both functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob $> F$ |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.138 | 68.79 | 8 | 326 | 2275 | < 0.0001 |
| 2 | 0.484 | 58.19 | 3 | 164 | 326 | < 0.0001 |

Table 4.17: Average percent cover of habitat attributes within the three cluster types for Site 5 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 169$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the three statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. shrubs and overstorey cover (SO), 2. midstorey cover, overstorey cover and shrubs (MOS), and 3. overstorey cover (O). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Para Wirra Recreation Park, South Australia. Data collected October 2008.

| Habitat attribute | SO ($n = 74$) | MOS ($n = 56$) | O ($n = 39$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|--------------------|---------------------|-------------------|--------------------|--|
| Grasses | 0.7 | 0.8 | 0.3 | 0.49 | -0.22 |
| Weeds | 0.0 | 0.0 | 0.0 | 0.00 | n/a |
| Shrubs | 56.4 | 30.3 | 16.0 | 91.60 | -0.90 |
| Midstorey | 15.3 | 47.3 | 18.2 | 98.39 | 0.98 |
| Overstorey | 43.3 | 33.4 | 76.7 | 60.54 | 0.25 |

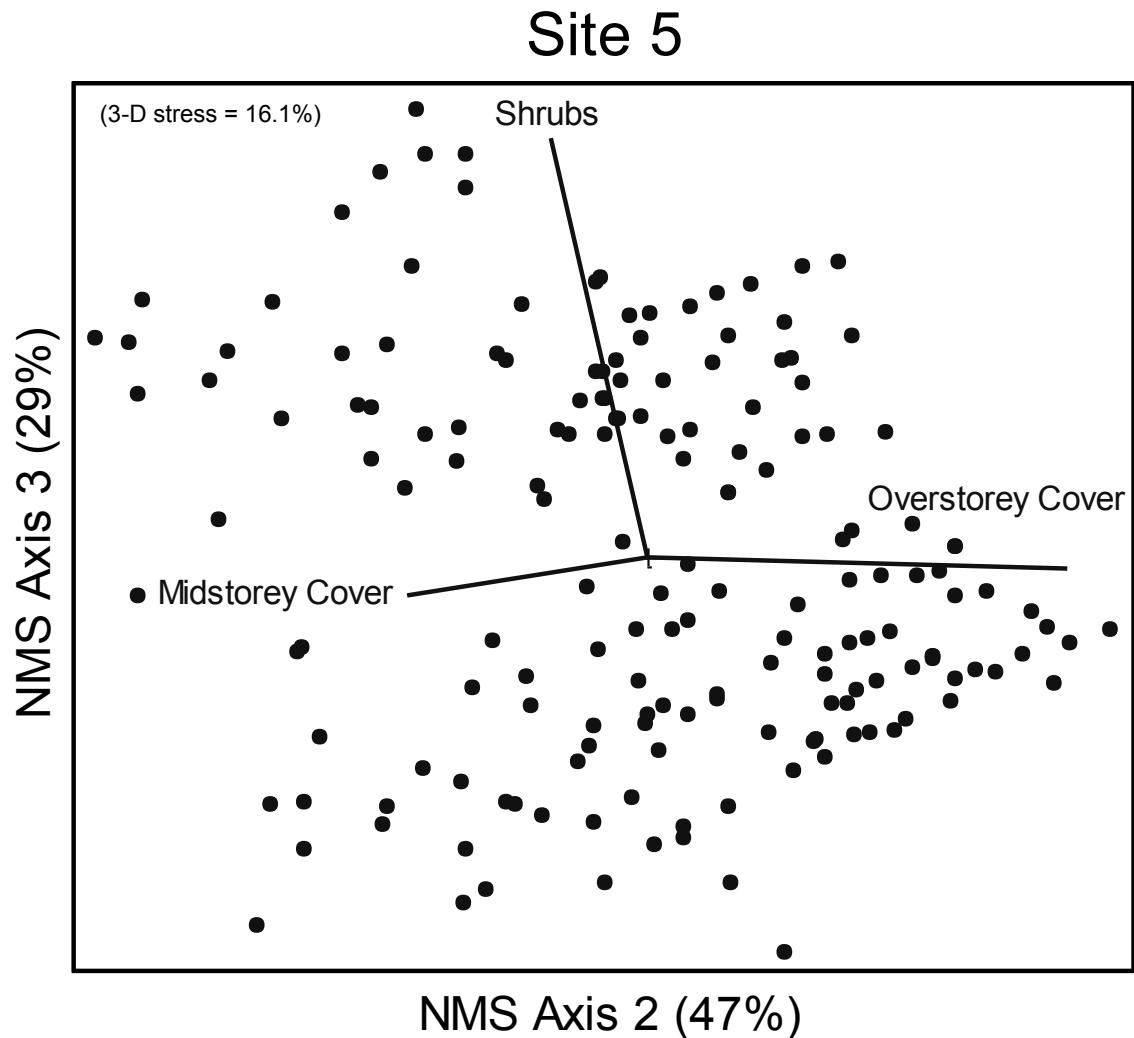


Figure 4.14: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 5 (classified *a priori* as open heath woodland but the vegetation structure had more in common with that of closed heath woodland sites; refer Figure 3.2). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 169, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 16.1%, Axes 2 and 3 accounts for 76% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Para Wirra Recreation Park, South Australia. Data collected October 2008.

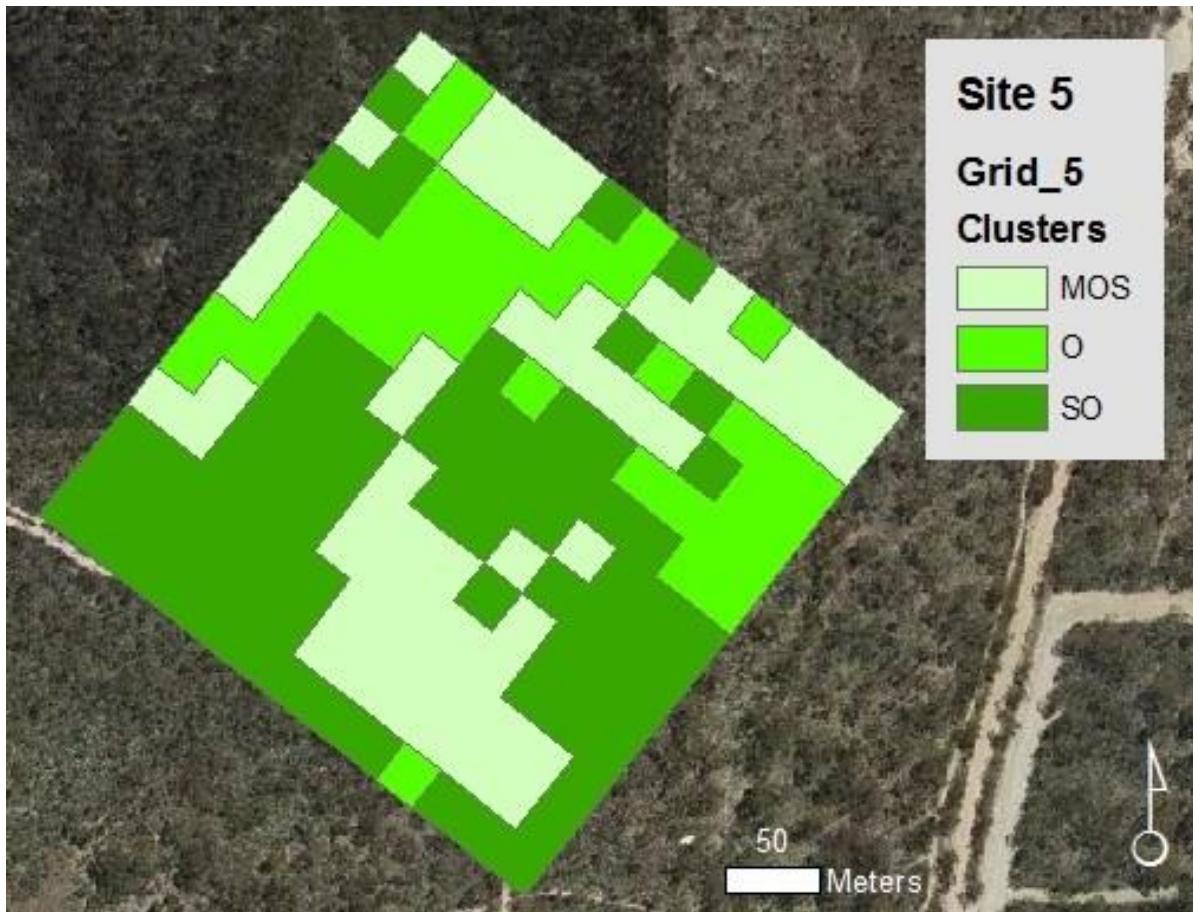


Figure 4.15: Habitat heterogeneity within Site 5 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 169$) is colour-coded for the dendrogram cluster it aligned with. The three clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover, overstorey cover and shrubs (MOS), 2. overstorey cover (O), and shrubs and overstorey cover (SO). This site is located within the Para Wirra Recreation Park, South Australia. Data collected October 2008.

Site 6 – grassy woodland

The cluster analysis undertaken on the habitat attributes for Site 6 identified three vegetation clusters and two outliers (Fig. 4.16). The first outlier contained seven 25 m × 25 m survey squares which all had very low percent covers of the habitat variables that were measured. These seven survey squares corresponded with an area containing a small, abandoned and un-rehabilitated sand mine. The second outlier resulted from a single 25 m × 25 m survey square which for this site contained an unusually high cover of weeds (58%) and shrubs (20%). Regularized Discrimination Analysis for the vegetation clusters identified three functions, with all three accounting for the variance (Table 4.18). Group association for the 25 m × 25 m squares within these three functions were well predicted (based on Wilks' Lambda), with all three being very highly significant (Table 4.19).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were midstorey cover, overstorey cover and grasses (Table 4.20; Fig. 4.17). The least discriminatory variables in ascending order were weeds and shrubs (Table 4.20). The habitat structure within each of the 25 m × 25 m squares appeared to be intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.18).

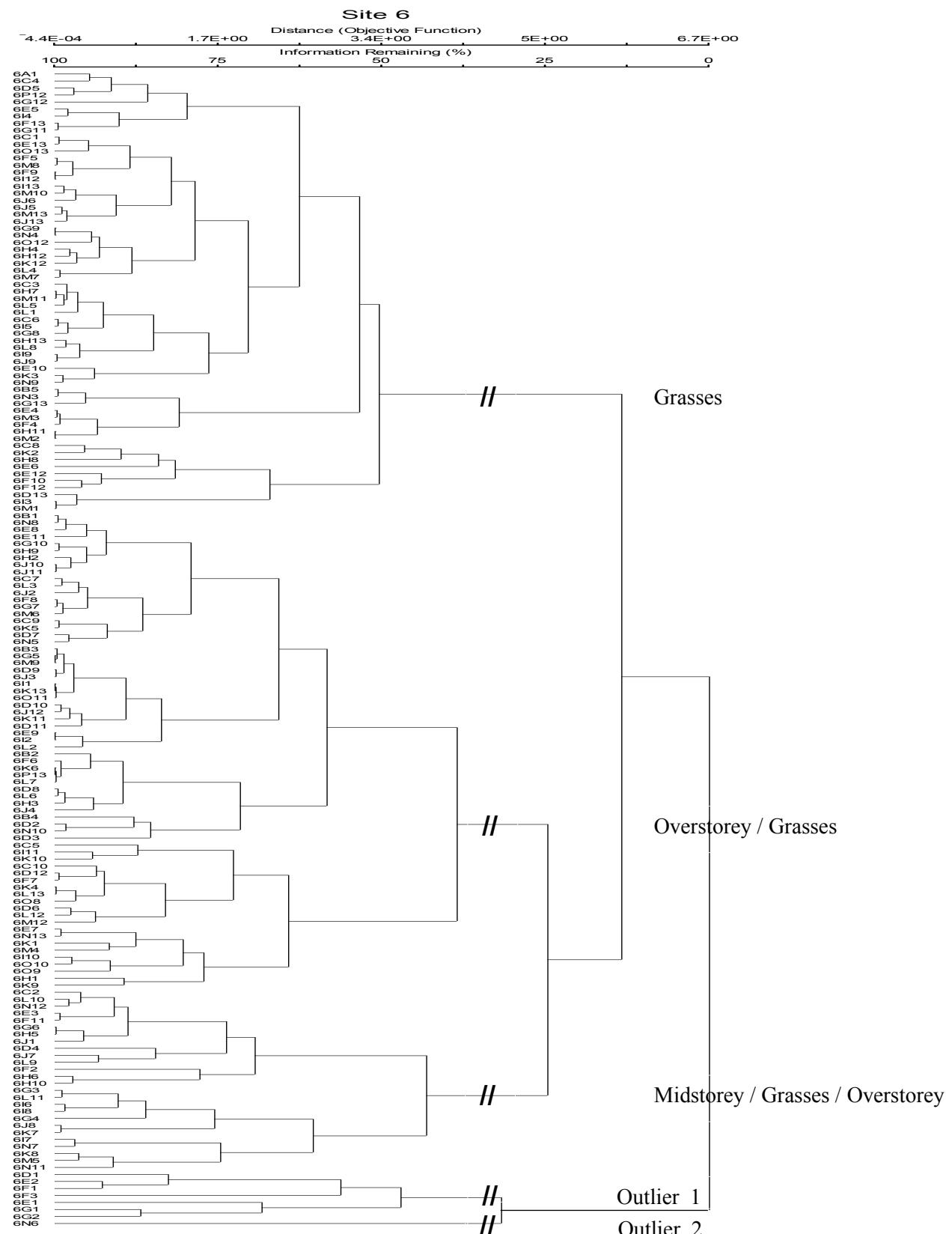


Figure 4.16: Hierarchical clustering of 165, 25 m × 25 m squares used to survey the entire ten hectares of Site 6 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained. Outlier 1 contained low percent cover of all the habitat attributes that were measured. Outlier 2 contained higher weed and shrub cover than average.

Table 4.18: Eigenvalues for the three functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 6 (grassy woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with all two functions accounting for 80.96% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 1.64 | 49.64 | 49.64 | 0.788 |
| 2 | 1.04 | 31.32 | 80.96 | 0.713 |
| 3 | 0.63 | 19.04 | 100.00 | 0.622 |

Table 4.19: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 6 (grassy woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated *F*-value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob > *F*). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the *F* distribution. Therefore, the "Approx. *F*" statistic and *p*-value (Prob > *F*) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis all three functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. <i>F</i> | NumDF | DenDF | df | Prob > <i>F</i> |
|-------------------|---------------|------------------|-------|-------|------|-----------------|
| 1 | 0.114 | 34.55 | 15 | 434 | 6062 | < 0.0001 |
| 2 | 0.302 | 32.44 | 8 | 316 | 2205 | < 0.0001 |
| 3 | 0.614 | 33.37 | 3 | 159 | 316 | < 0.0001 |

Table 4.20: Average percent cover of habitat attributes within the three cluster types (and two outliers) for Site 6 (grassy woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 165$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the three statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses (G), 2. overstorey cover and grasses (OG), 3. midstorey cover, grasses and overstorey cover (MGO), 4. outlier 1, and 5. outlier 2. Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Altona woodland area, South Australia. Data collected October 2008.

| Habitat attribute | G (n = 63) | OG (n = 68) | MGO (n = 26) | Outlier 1 (n = 7) | Outlier 2 (n = 1) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|---------------|----------------|-----------------|----------------------|----------------------|--------------------|--|
| Grasses | 54.7 | 44.7 | 34.3 | 13.6 | 0.0 | 34.36 | -0.31 |
| Weeds | 4.1 | 2.1 | 2.9 | 0.9 | 58.0 | 3.68 | 0.15 |
| Shrubs | 0.6 | 0.2 | 0.2 | 2.6 | 20.0 | 15.48 | -0.17 |
| Midstorey | 6.0 | 10.9 | 44.4 | 10.7 | 5.0 | 79.36 | 0.92 |
| Overstorey | 11.4 | 44.9 | 29.6 | 7.1 | 15.0 | 52.11 | 0.06 |

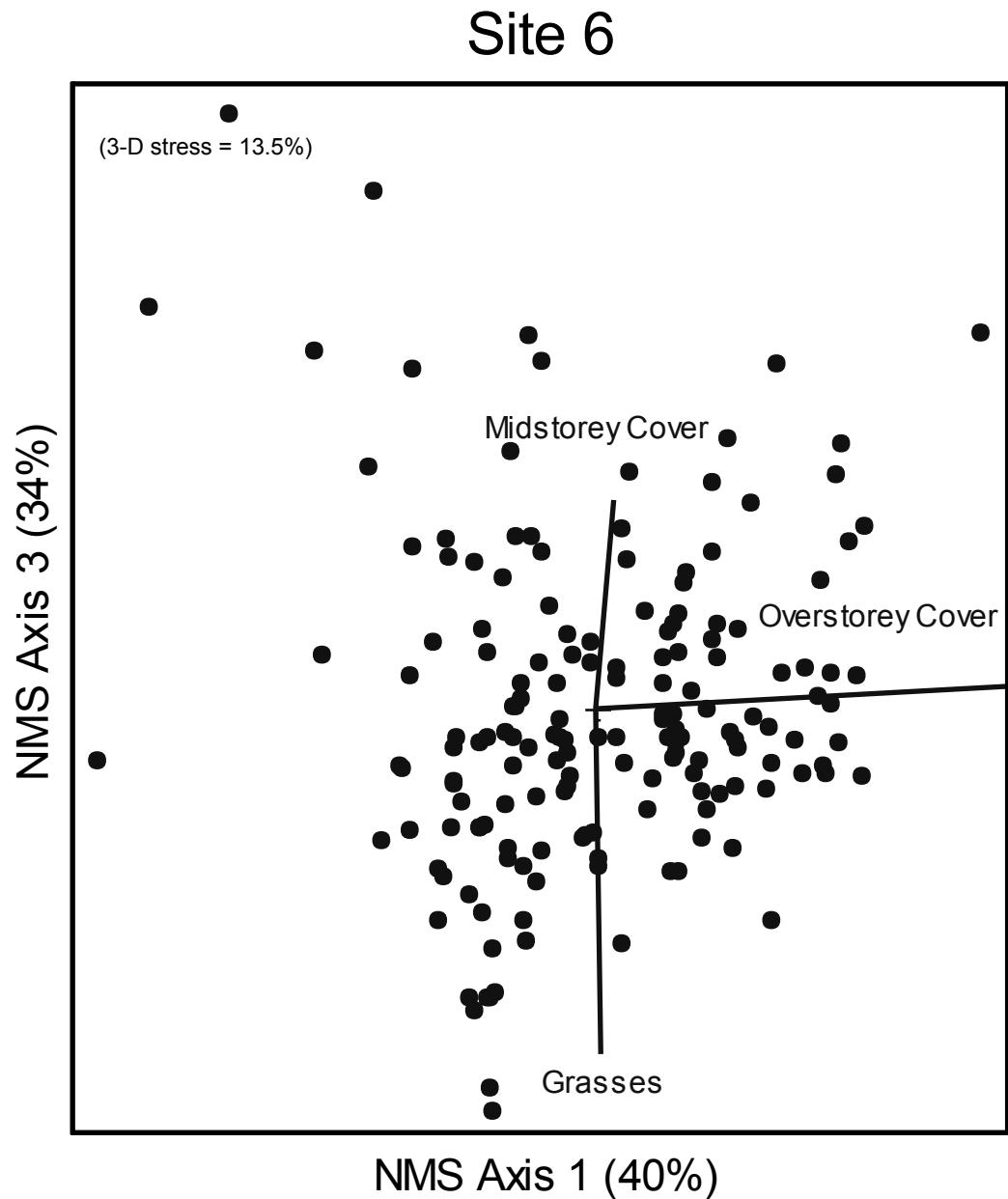


Figure 4.17: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 6 (grassy woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 165, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 13.5%, Axes 1 and 3 accounts for 74% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Altona woodland area, South Australia. Data collected October 2008.

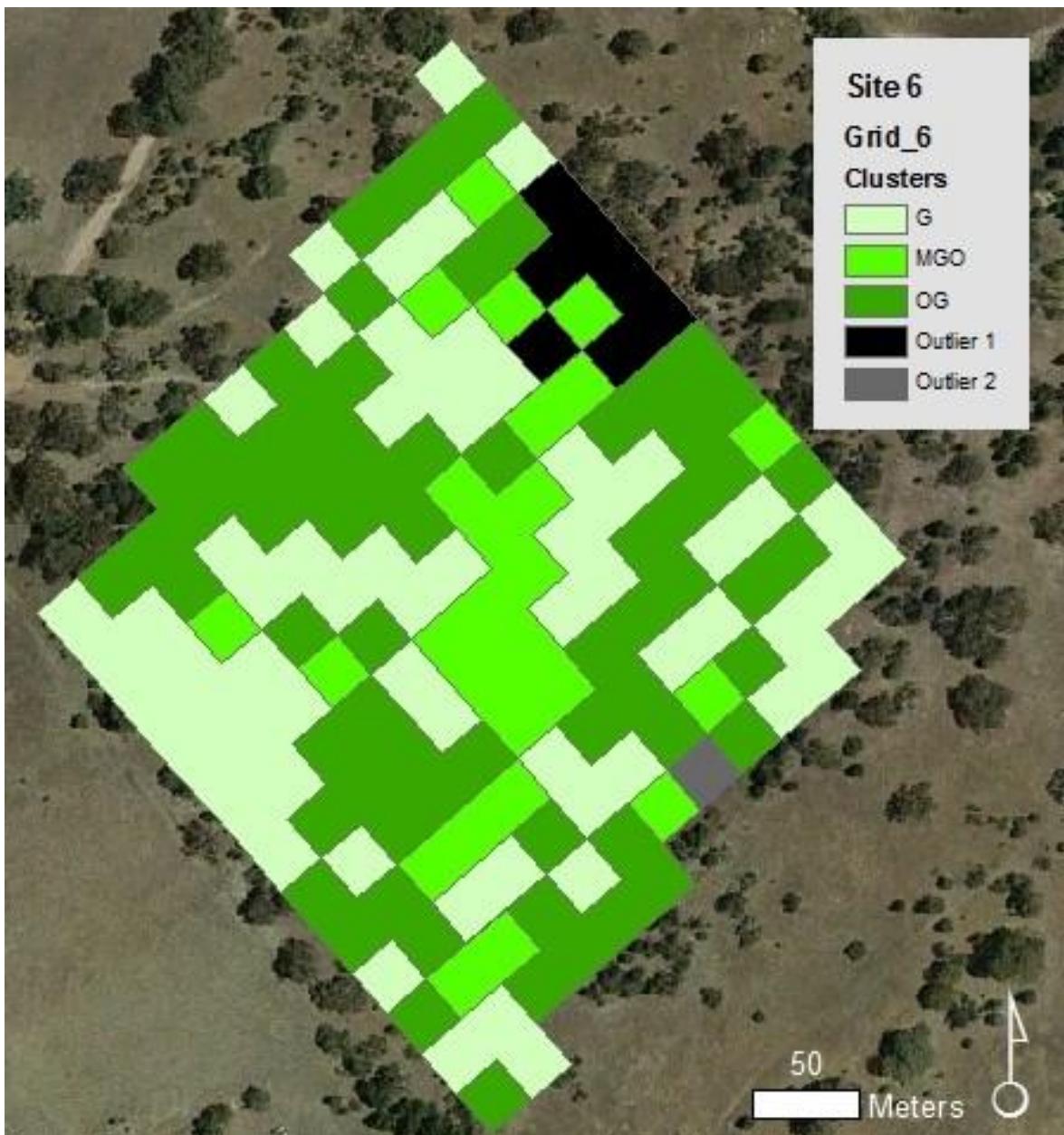


Figure 4.18: Habitat heterogeneity within Site 6 (grassy woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 165$) is colour-coded for the dendrogram cluster it aligned with. The three clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses (G), 2. midstorey cover, grasses and overstorey cover (MGO), and 3. overstorey cover and grasses (OG). This site is located within the Altona woodland area, South Australia. Data collected October 2008.

Site 7 – grazed woodland

The cluster analysis undertaken on the habitat attributes for Site 7 identified four vegetation clusters (Fig. 4.19). Regularized Discrimination Analysis for the vegetation clusters identified three functions, with the first two accounting for 92% of the variance (Table 4.21). Group association for the 25 m × 25 m squares within these three functions were well predicted (based on Wilks' Lambda), with all three being very highly significant (Table 4.22).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were midstorey cover, weeds and overstorey cover (Table 4.23; Fig. 4.20). The least discriminatory variables in ascending order were shrubs and grasses (Table 4.23). The habitat structure within each of the 25 m × 25 m squares appeared to be intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.21).

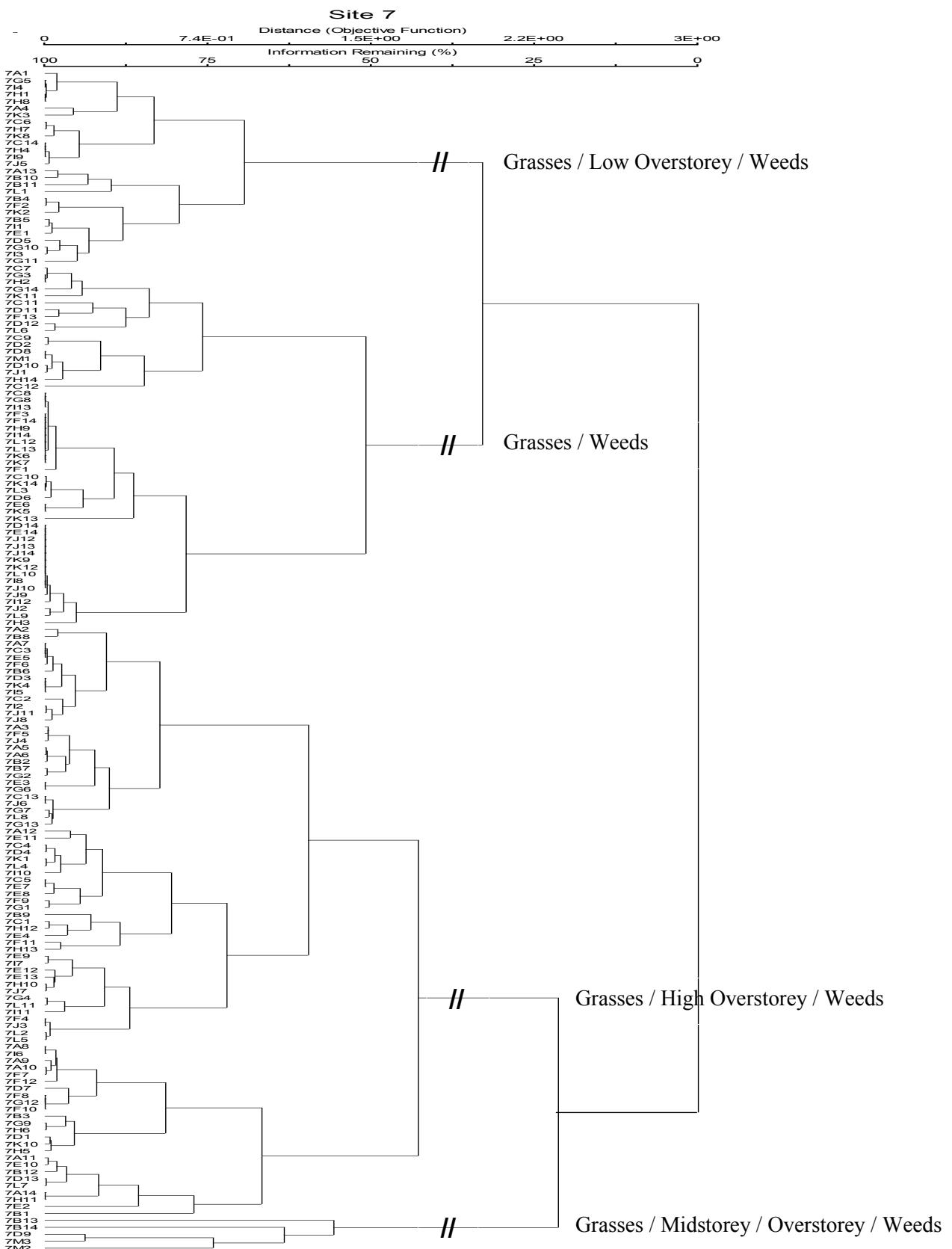


Figure 4.19: Hierarchical clustering of 170, 25 m × 25 m squares used to survey the entire ten hectares of Site 7 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.21: Eigenvalues for the three functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 7 (grazed woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first two functions accounting for 92.35% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 3.92 | 50.79 | 50.79 | 0.893 |
| 2 | 3.21 | 41.56 | 92.35 | 0.873 |
| 3 | 0.59 | 7.65 | 100.00 | 0.609 |

Table 4.22: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 7 (grazed woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated *F*-value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob > *F*). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the *F* distribution. Therefore, the "Approx. *F*" statistic and *p*-value (Prob > *F*) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis all three functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. <i>F</i> | NumDF | DenDF | df | Prob > <i>F</i> |
|-------------------|---------------|------------------|-------|-------|------|-----------------|
| 1 | 0.030 | 75.98 | 15 | 448 | 6258 | < 0.0001 |
| 2 | 0.149 | 64.68 | 8 | 326 | 2275 | < 0.0001 |
| 3 | 0.629 | 32.29 | 3 | 164 | 326 | < 0.0001 |

Table 4.23: Average percent cover of habitat attributes within the four cluster types for Site 7 (grazed woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 170$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the four statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses, low overstorey cover and weeds (GLOW), 2. grasses and weeds (GW), 3. grasses, high overstorey cover and weeds (GHOW), and 4. grasses, midstorey cover, overstorey cover and weeds (GMOW). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Altona woodland area, South Australia. Data collected October – November 2008.

| Habitat attribute | GLOW ($n = 28$) | GW ($n = 52$) | GHOW ($n = 85$) | GMOW ($n = 5$) | Univariate F -ratio | Standardized discriminant function coefficient |
|-------------------|----------------------|--------------------|----------------------|---------------------|-----------------------|--|
| Grasses | 61.1 | 54.3 | 62.0 | 56.2 | 14.40 | -0.03 |
| Weeds | 20.8 | 39.0 | 11.9 | 13.4 | 136.28 | 0.08 |
| Shrubs | 0.6 | 0.2 | 0.6 | 5.4 | 30.66 | 0.18 |
| Midstorey | 2.1 | 1.2 | 0.7 | 46.0 | 193.17 | 0.96 |
| Overstorey | 32.1 | 10.5 | 52.8 | 40.0 | 82.61 | -0.45 |

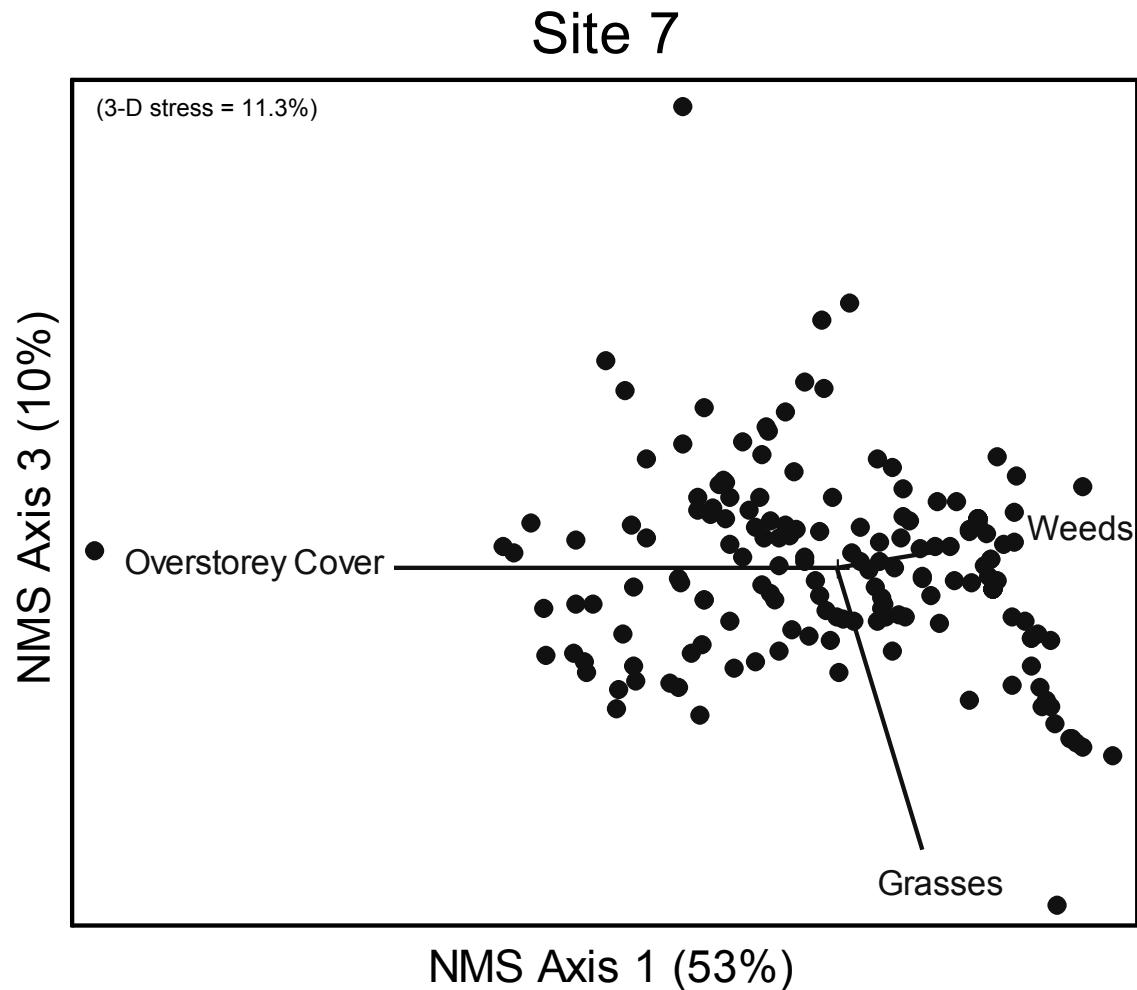


Figure 4.20: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 7 (grazed woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 170, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 11.3%, Axes 1 and 3 accounts for 63% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Altona woodland area, South Australia. Data collected October – November 2008.

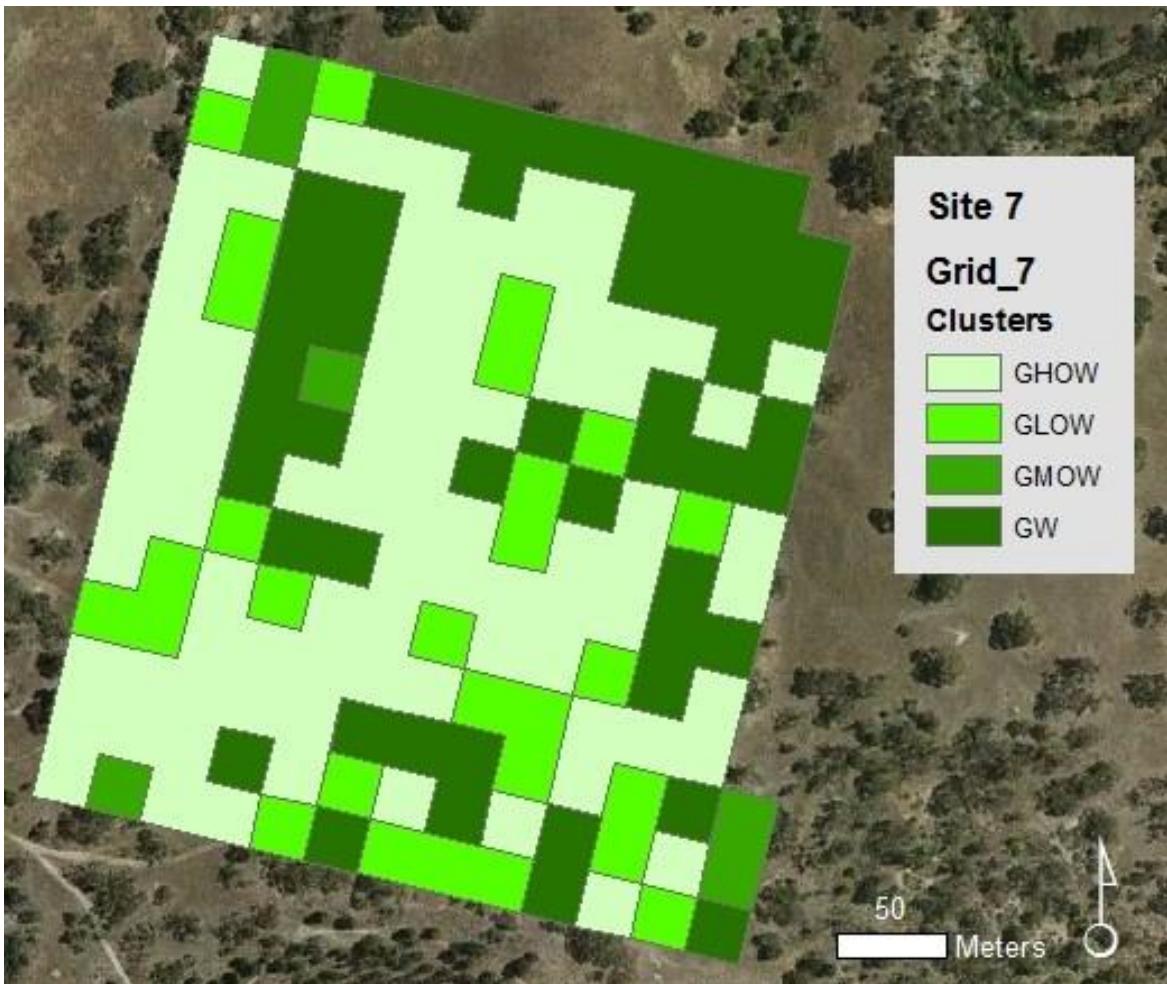


Figure 4.21: Habitat heterogeneity within Site 7 (grazed woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 170$) is colour-coded for the dendrogram cluster it aligned with. The four clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses, high overstorey cover and weeds (GHOW), 2. grasses, low overstorey cover and weeds (GLOW), 3. grasses, midstorey cover, overstorey cover and weeds (GMOW), and 4. grasses and weeds (GW). This site is located within the Altona woodland area, South Australia. Data collected October – November 2008.

Site 8 – open heath woodland

The cluster analysis undertaken on the habitat attributes for Site 8 identified four vegetation clusters (Fig. 4.22). Regularized Discrimination Analysis for the vegetation clusters identified three functions, with the first two accounting for 99% of the variance (Table 4.24). Group association for the 25 m × 25 m squares within all three functions were well predicted (based on Wilks' Lambda), with the first two being very highly significant and the third being significant (Table 4.25).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were overstorey cover, midstorey cover and shrubs (Table 4.26; Fig. 4.23). The least discriminatory variables in ascending order were weeds and grasses (Table 4.26). The habitat structure within each of the 25 m × 25 m squares appeared to be intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.24).

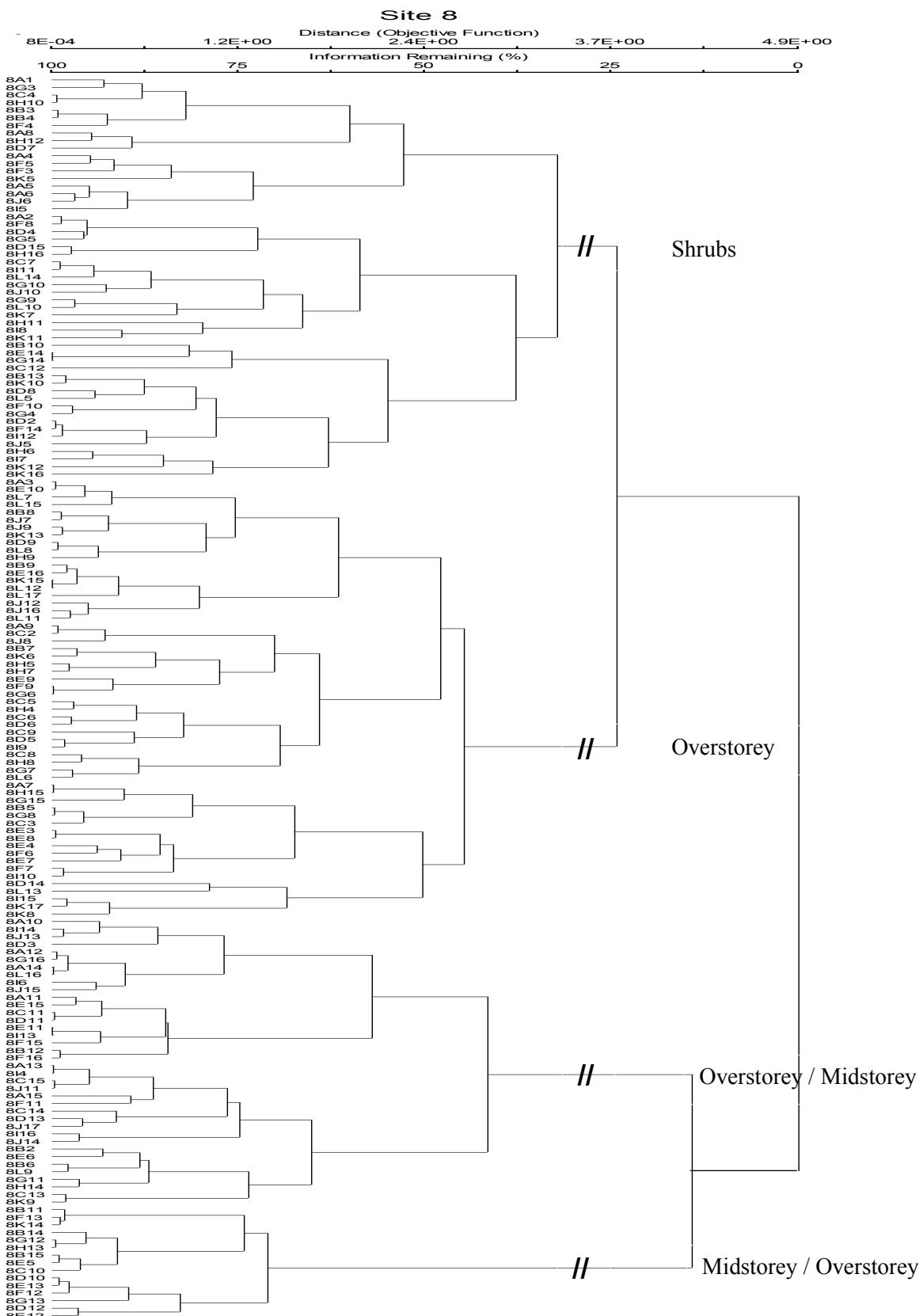


Figure 4.22: Hierarchical clustering of 164, 25 m × 25 m squares used to survey the entire ten hectares of Site 8 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.24: Eigenvalues for the three functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 8 (open heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first two functions accounting for 99.25% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 5.69 | 83.50 | 83.50 | 0.922 |
| 2 | 1.07 | 15.75 | 99.25 | 0.719 |
| 3 | 0.05 | 0.75 | 100.00 | 0.220 |

Table 4.25: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 8 (open heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated *F*-value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob > *F*). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the *F* distribution. Therefore, the "Approx. *F*" statistic and *p*-value (Prob > *F*) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis the first two functions were very highly significant and the third was significant.

| Test of functions | Wilks' Lambda | Approx. <i>F</i> | NumDF | DenDF | df | Prob > <i>F</i> |
|-------------------|---------------|------------------|-------|-------|------|-----------------|
| 1 | 0.069 | 47.11 | 15 | 431 | 6020 | < 0.0001 |
| 2 | 0.459 | 18.69 | 8 | 314 | 2191 | < 0.0001 |
| 3 | 0.952 | 2.68 | 3 | 158 | 314 | 0.0489 |

Table 4.26: Average percent cover of habitat attributes within the four cluster types for Site 8 (open heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 164$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the four statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. shrubs (S), 2. overstorey cover (O), 3. overstorey cover and midstorey cover (OM), and 4. midstorey cover and overstorey cover (MO). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Altona woodland area, South Australia. Data collected October 2008.

| Habitat attribute | S ($n = 53$) | O ($n = 58$) | OM ($n = 38$) | MO ($n = 15$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|-------------------|-------------------|--------------------|--------------------|--------------------|--|
| Grasses | 17.2 | 20.3 | 17.6 | 10.7 | 4.82 | 0.22 |
| Weeds | 0.6 | 1.3 | 2.1 | 0.8 | 4.44 | -0.09 |
| Shrubs | 25.2 | 15.5 | 8.6 | 6.8 | 20.74 | -0.15 |
| Midstorey | 18.9 | 16.5 | 33.6 | 74.7 | 57.17 | 0.13 |
| Overstorey | 21.2 | 41.7 | 75.9 | 32.3 | 227.71 | 1.01 |

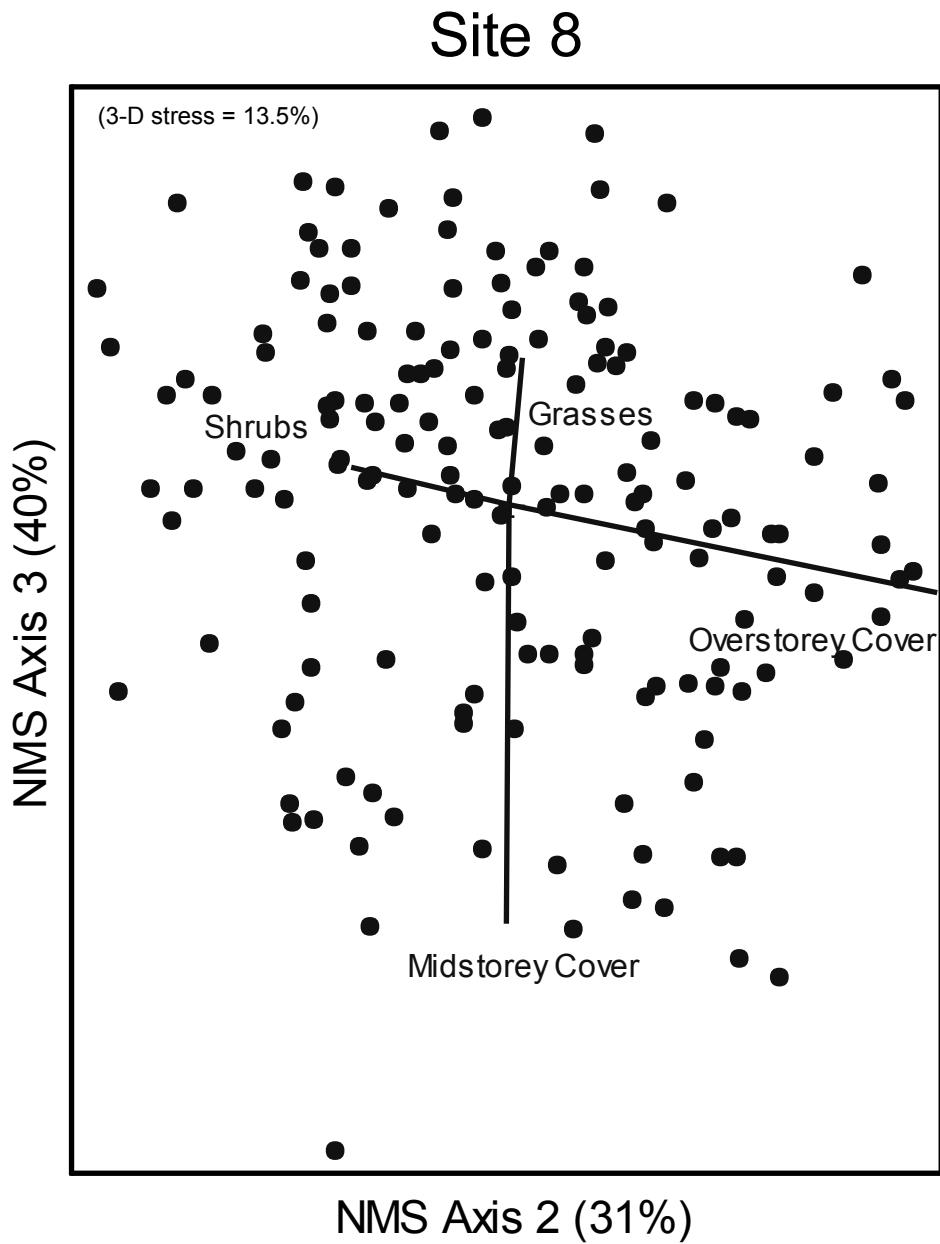


Figure 4.23: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 8 (open heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 168, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 13.5%, Axes 2 and 3 accounts for 71% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Altona woodland area, South Australia. Data collected October 2008.



Figure 4.24: Habitat heterogeneity within Site 8 (open heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 164$) is colour-coded for the dendrogram cluster it aligned with. The four clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover and overstorey cover (MO), 2. overstorey cover (O), 3. overstorey cover and midstorey cover (OM), and 4. shrubs (S). This site is located within the Altona woodland area, South Australia. Data collected October 2008.

Site 9 – closed heath woodland

The cluster analysis undertaken on the habitat attributes for Site 9 identified three vegetation clusters (Fig. 4.25). Regularized Discrimination Analysis for the vegetation clusters identified two functions, with the first accounting for 64% of the variance (Table 4.27). Group association for the 25 m × 25 m squares within these two functions were well predicted (based on Wilks' Lambda), with both being very highly significant (Table 4.28).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were shrubs, midstorey cover and overstorey cover (Table 4.29; Fig. 4.26). The least discriminatory variables in ascending order were grasses and weeds (Table 4.29). The habitat structure within each of the 25 m × 25 m squares appeared to range from spatially aggregated with similar habitat features to intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.27).

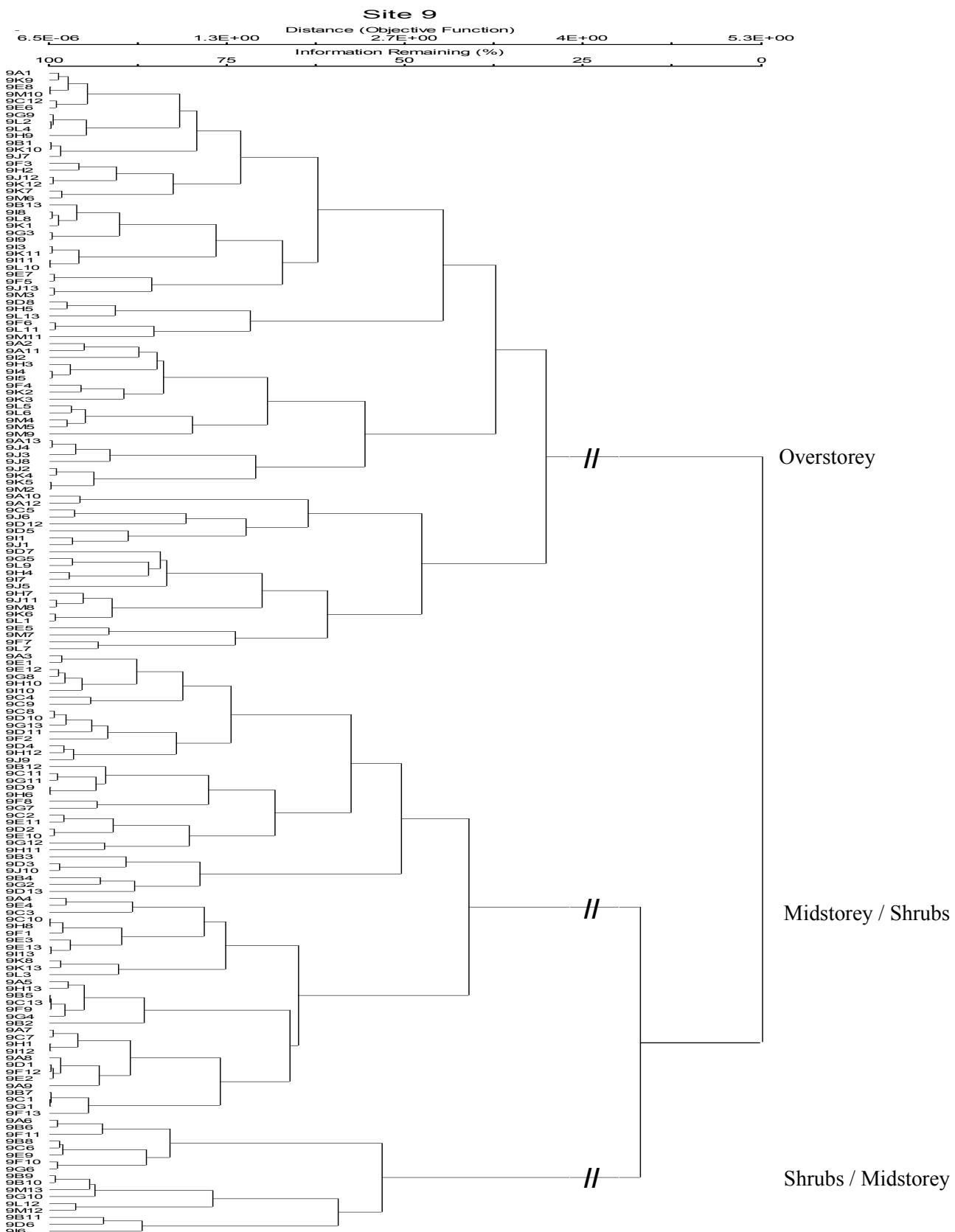


Figure 4.25: Hierarchical clustering of 168, 25 m × 25 m squares used to survey the entire ten hectares of Site 9 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.27: Eigenvalues for the two functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 9 (closed heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first function accounting for 63.51% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 3.09 | 63.51 | 63.51 | 0.869 |
| 2 | 1.78 | 36.49 | 100.00 | 0.800 |

Table 4.28: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 9 (closed heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob > F). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value (Prob > F) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis both functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob > F |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.088 | 76.47 | 10 | 322 | 2889 | < 0.0001 |
| 2 | 0.359 | 72.08 | 4 | 162 | 483 | < 0.0001 |

Table 4.29: Average percent cover of habitat attributes within the three cluster types for Site 9 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 168$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the three statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. overstorey cover (O), 2. midstorey cover and shrubs (MS), and 3. shrubs and midstorey cover (SM). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Para Wirra area, South Australia. Data collected October 2008.

| Habitat attribute | O ($n = 84$) | MS ($n = 67$) | SM ($n = 17$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|-------------------|--------------------|--------------------|--------------------|--|
| Grasses | 5.1 | 3.4 | 0.5 | 4.45 | 0.08 |
| Weeds | 0.0 | 0.0 | 0.0 | 0.75 | 0.10 |
| Shrubs | 12.6 | 20.1 | 68.2 | 224.12 | 0.90 |
| Midstorey | 21.4 | 56.7 | 31.8 | 125.84 | 0.33 |
| Overstorey | 41.5 | 24.9 | 15.1 | 32.02 | -0.33 |

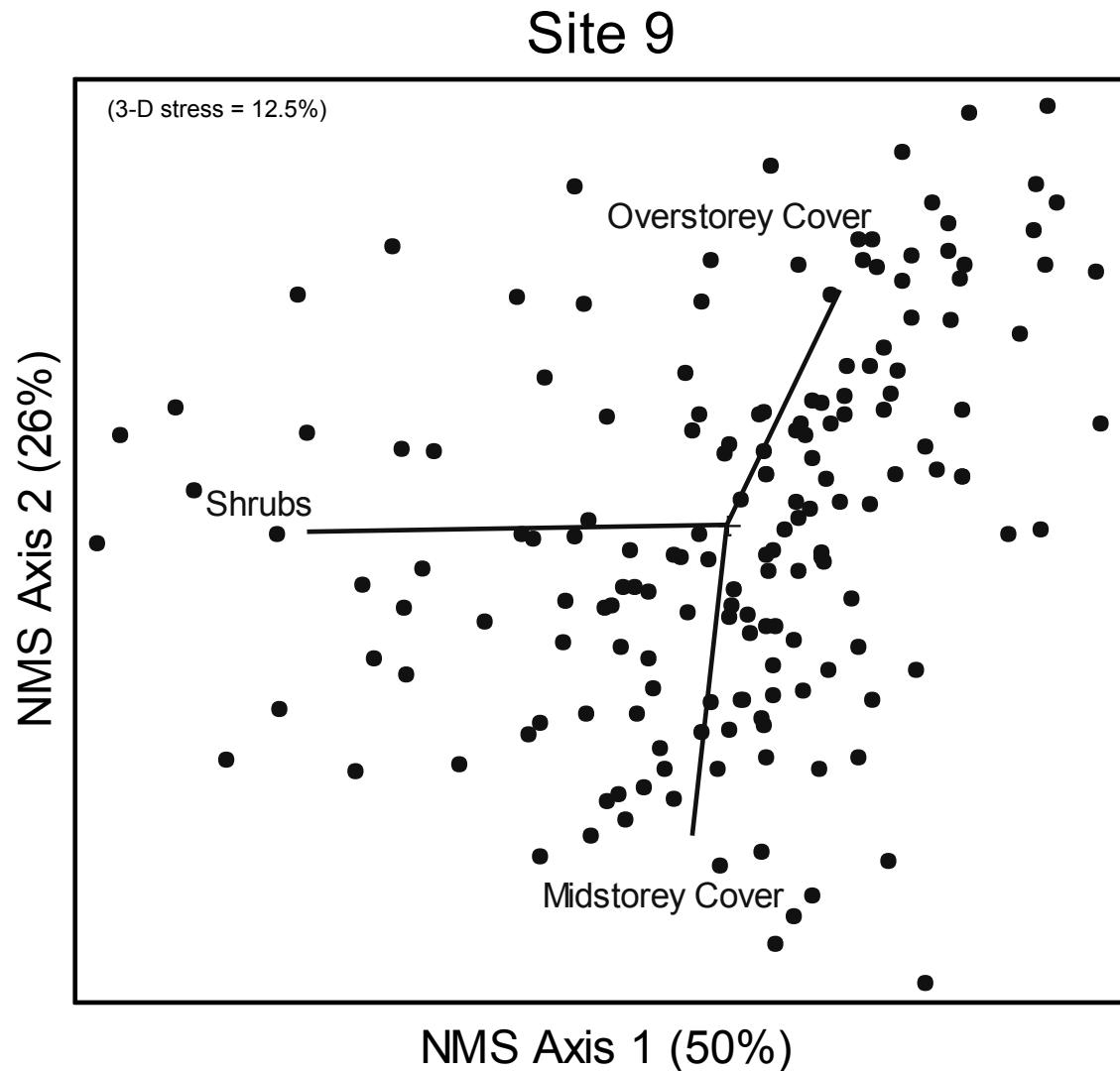


Figure 4.26: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 9 (closed heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 168, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 12.5%, Axes 1 and 2 accounts for 76% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Para Wirra area, South Australia. Data collected October 2008.

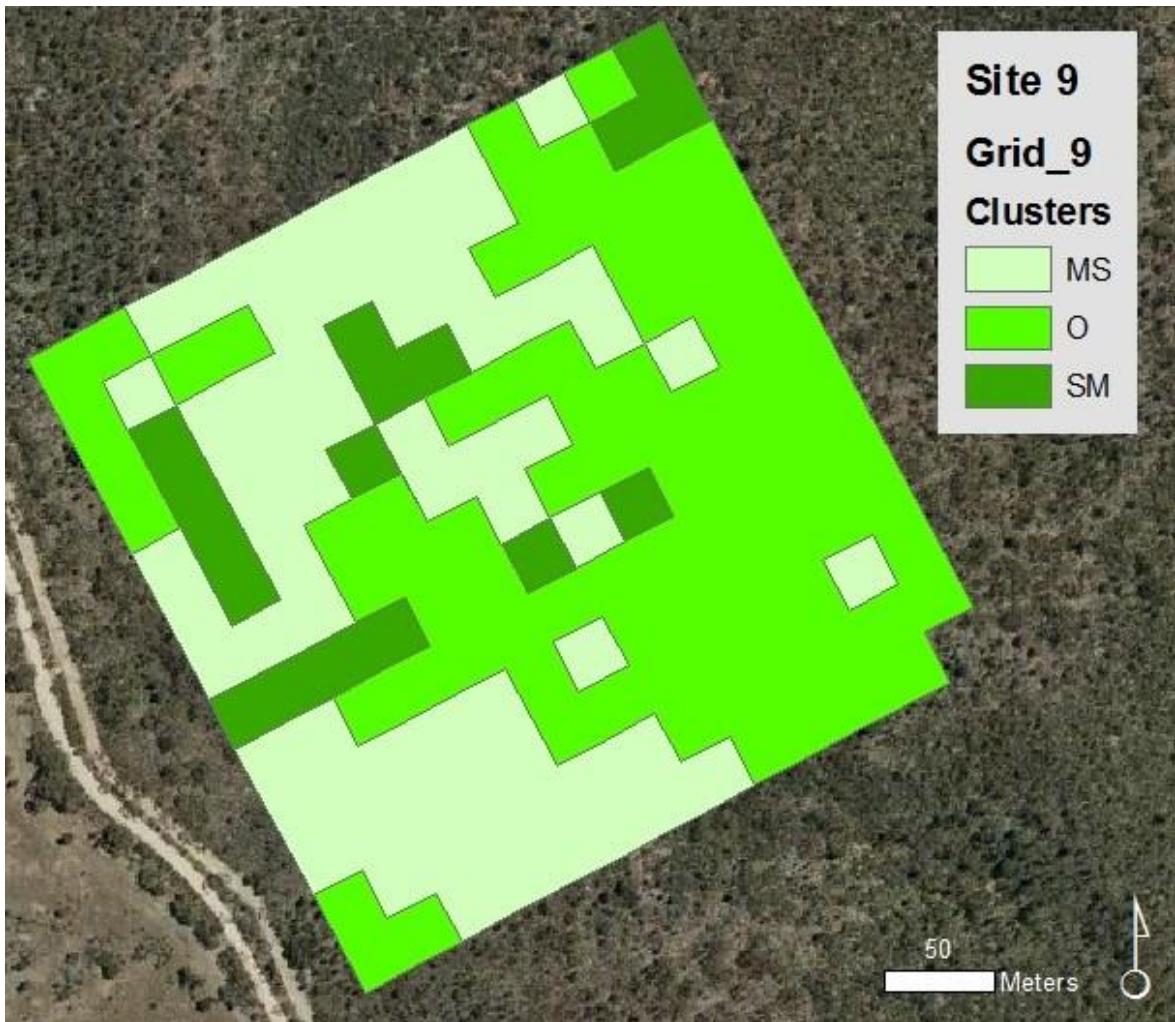


Figure 4.27: Habitat heterogeneity within Site 9 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 168$) is colour-coded for the dendrogram cluster it aligned with. The three clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover and shrubs (MS), 2. overstorey cover (O), and 3. shrubs and midstorey cover (SM). This site is located within the Para Wirra area, South Australia. Data collected October 2008.

Site 10 – open heath woodland

The cluster analysis undertaken on the habitat attributes for Site 10 identified three vegetation clusters (Fig. 4.28). Regularized Discrimination Analysis for the vegetation clusters identified two functions, with the first accounting for 58% of the variance (Table 4.30). Group association for the 25 m × 25 m squares within these two functions were well predicted (based on Wilks' Lambda), with both being very highly significant (Table 4.31).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were overstorey cover, midstorey cover and grasses (Table 4.32; Fig. 4.29). The least discriminatory variables in ascending order were weeds and shrubs (Table 4.32). The habitat structure within each of the 25 m × 25 m squares appeared to be intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.30).

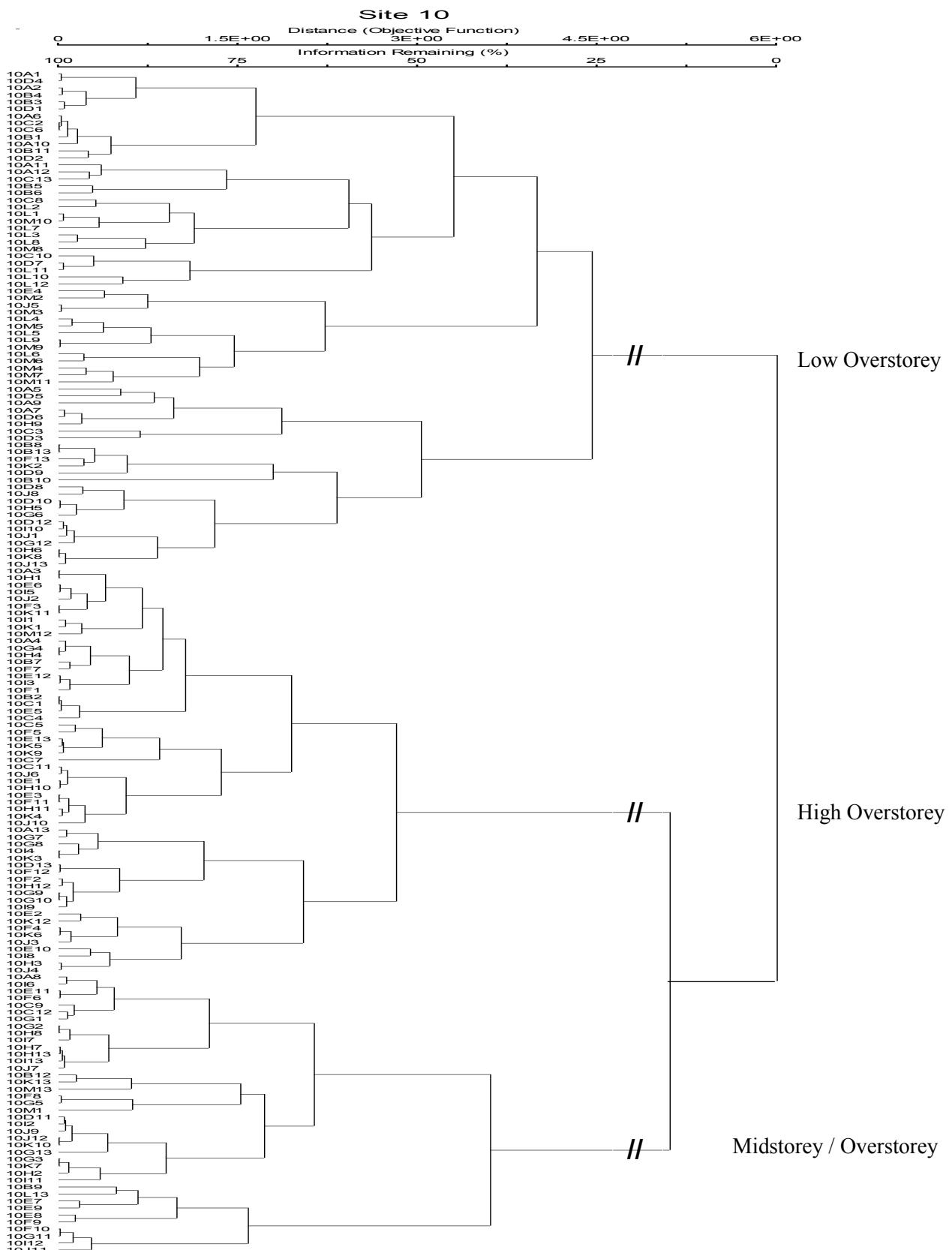


Figure 4.28: Hierarchical clustering of 169, 25 m × 25 m squares used to survey the entire ten hectares of Site 10 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.30: Eigenvalues for the two functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 10 (open heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first function accounting for 57.93% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 1.70 | 57.93 | 57.93 | 0.794 |
| 2 | 1.24 | 42.07 | 100.00 | 0.743 |

Table 4.31: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 10 (open heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob $> F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value (Prob $> F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis both functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob $> F$ |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.166 | 47.20 | 10 | 324 | 2907 | < 0.0001 |
| 2 | 0.447 | 50.33 | 4 | 163 | 486 | < 0.0001 |

Table 4.32: Average percent cover of habitat attributes within the three cluster types for Site 10 (open heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 169$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the three statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. low overstorey cover (LO), 2. high overstorey cover (HO), and 3. midstorey cover and overstorey cover (MO). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Sandy Creek Conservation Park, South Australia. Data collected October 2008.

| Habitat attribute | LO ($n = 71$) | HO ($n = 58$) | MO ($n = 40$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|--------------------|--------------------|--------------------|--------------------|--|
| Grasses | 13.5 | 9.0 | 9.9 | 7.48 | -0.34 |
| Weeds | 0.3 | 0.7 | 0.5 | 1.76 | 0.01 |
| Shrubs | 13.4 | 7.1 | 8.2 | 8.72 | -0.24 |
| Midstorey | 11.4 | 16.8 | 41.6 | 88.84 | 0.16 |
| Overstorey | 20.7 | 53.7 | 28.6 | 115.69 | 0.95 |

Site 10

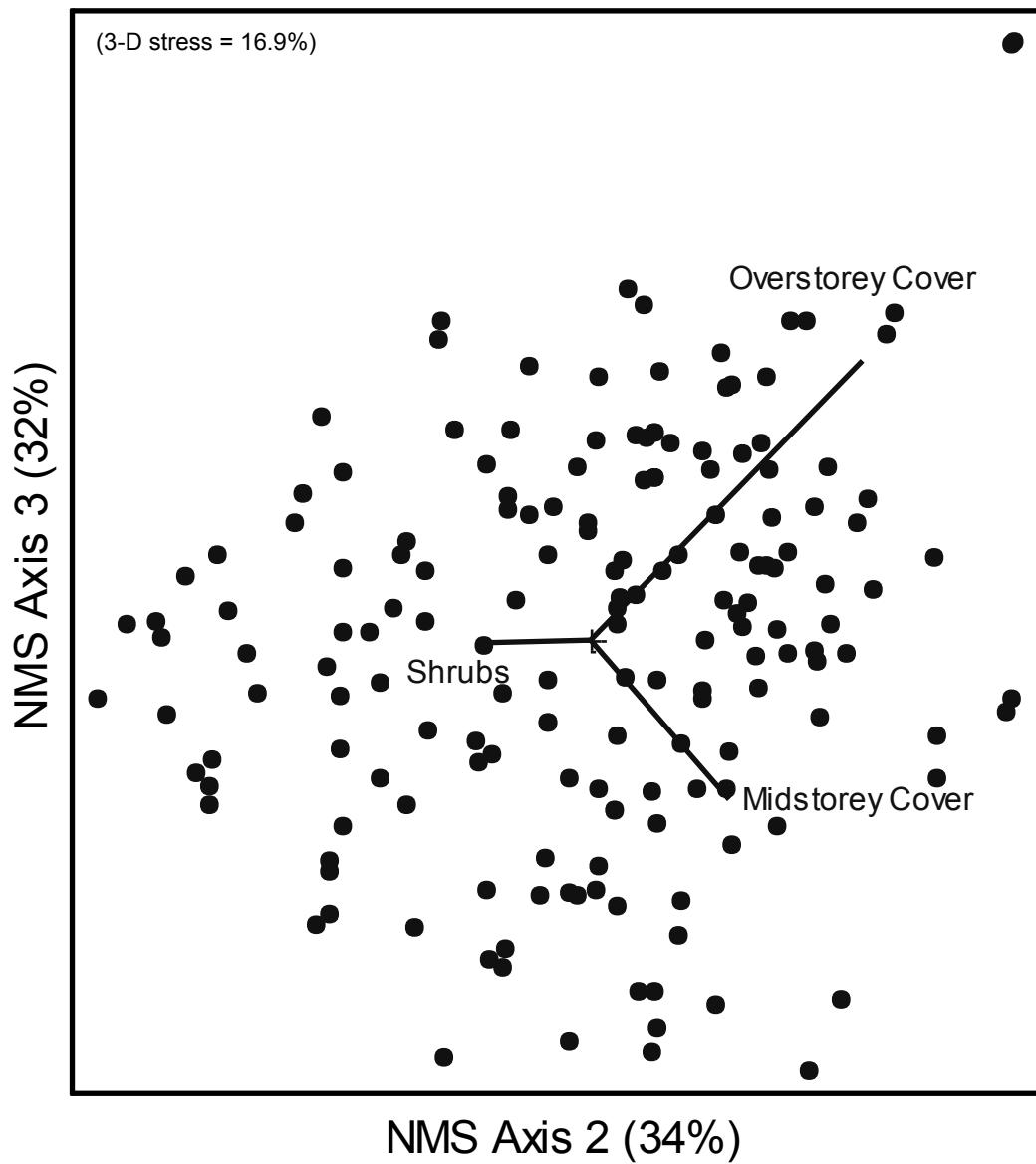


Figure 4.29: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 10 (open heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 169, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 16.9%, Axes 2 and 3 accounts for 66% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Sandy Creek Conservation Park, South Australia. Data collected October 2008.



Figure 4.30: Habitat heterogeneity within Site 10 (open heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 169$) is colour-coded for the dendrogram cluster it aligned with. The three clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. high overstorey cover (HO), 2. low overstorey cover (LO), and 3. midstorey cover and overstorey cover (MO). This site is located within the Sandy Creek Conservation Park, South Australia. Data collected October 2008.

Site 11 – open heath woodland

The cluster analysis undertaken on the habitat attributes for Site 11 identified three vegetation clusters (Fig. 4.31). Regularized Discrimination Analysis for the vegetation clusters identified two functions, with the first accounting for 69% of the variance (Table 4.33). Group association for the 25 m × 25 m squares within these two functions were well predicted (based on Wilks' Lambda), with both being very highly significant (Table 4.34).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were grasses, overstorey cover and midstorey cover (Table 4.35; Fig. 4.32). The least discriminatory variables in ascending order were weeds and shrubs (Table 4.35). The habitat structure within each of the 25 m × 25 m squares appeared to range from spatially aggregated with similar habitat features to intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.33).

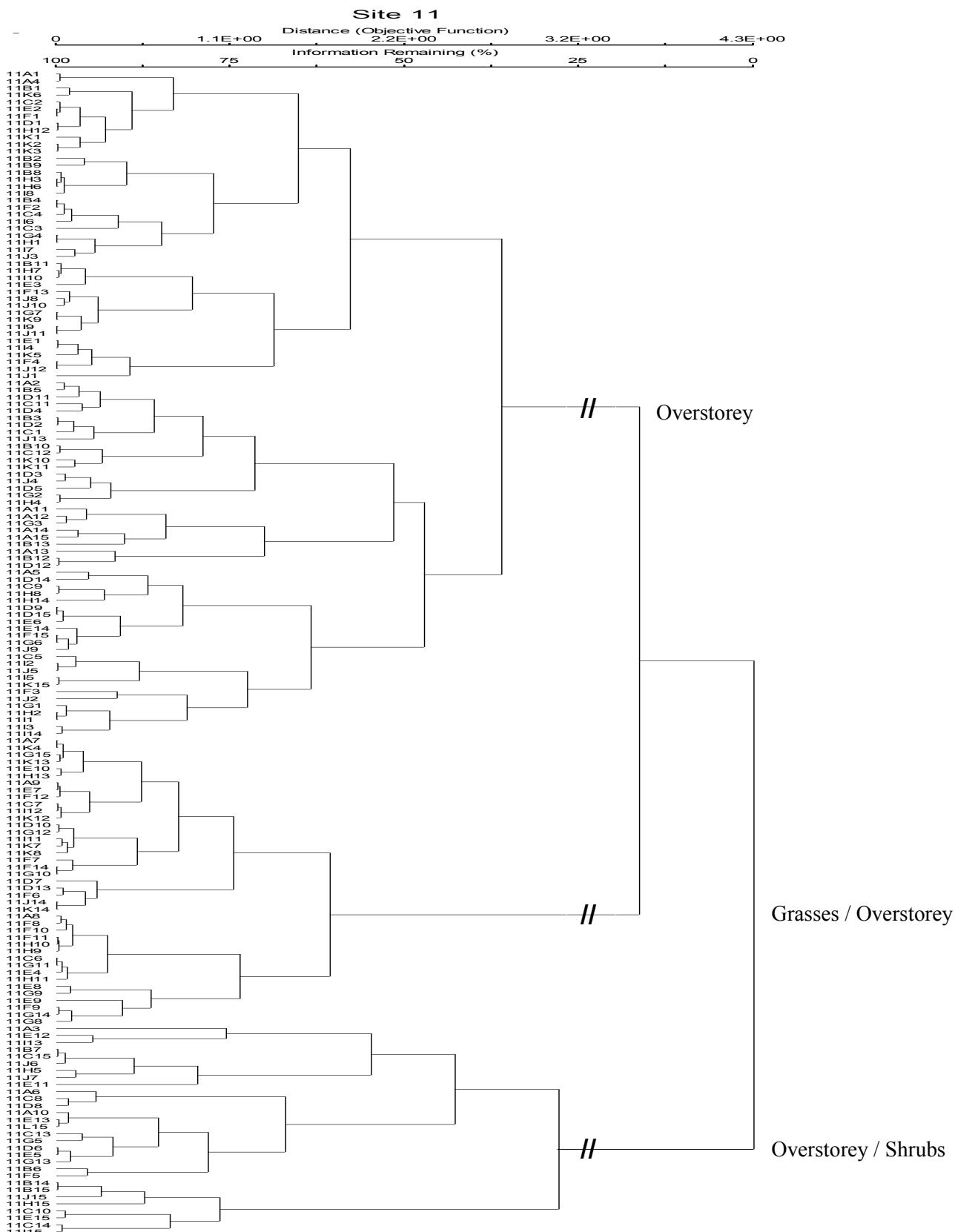


Figure 4.31: Hierarchical clustering of 166, 25 m × 25 m squares used to survey the entire ten hectares of Site 11 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.33: Eigenvalues for the two functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 11 (open heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first function accounting for 69.13% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 1.99 | 69.13 | 69.13 | 0.816 |
| 2 | 0.89 | 30.87 | 100.00 | 0.529 |

Table 4.34: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 11 (open heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob $> F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value (Prob $> F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis both functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob $> F$ |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.177 | 43.77 | 10 | 318 | 2853 | < 0.0001 |
| 2 | 0.529 | 35.55 | 4 | 160 | 477 | < 0.0001 |

Table 4.35: Average percent cover of habitat attributes within the three cluster types for Site 11 (open heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 166$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the three statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. overstorey cover (O), 2. grasses and overstorey cover (GO), and 3. overstorey cover and shrubs (OS). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Sandy Creek Conservation Park, South Australia. Data collected October 2008.

| Habitat attribute | O ($n = 95$) | GO ($n = 41$) | OS ($n = 30$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|-------------------|--------------------|--------------------|--------------------|--|
| Grasses | 20.4 | 56.3 | 23.6 | 161.23 | 0.98 |
| Weeds | 0.9 | 1.0 | 0.8 | 0.189 | 0.02 |
| Shrubs | 15.4 | 6.7 | 21.0 | 13.16 | -0.05 |
| Midstorey | 15.7 | 7.0 | 9.3 | 12.64 | -0.05 |
| Overstorey | 56.5 | 51.2 | 25.7 | 52.56 | -0.02 |

Site 11

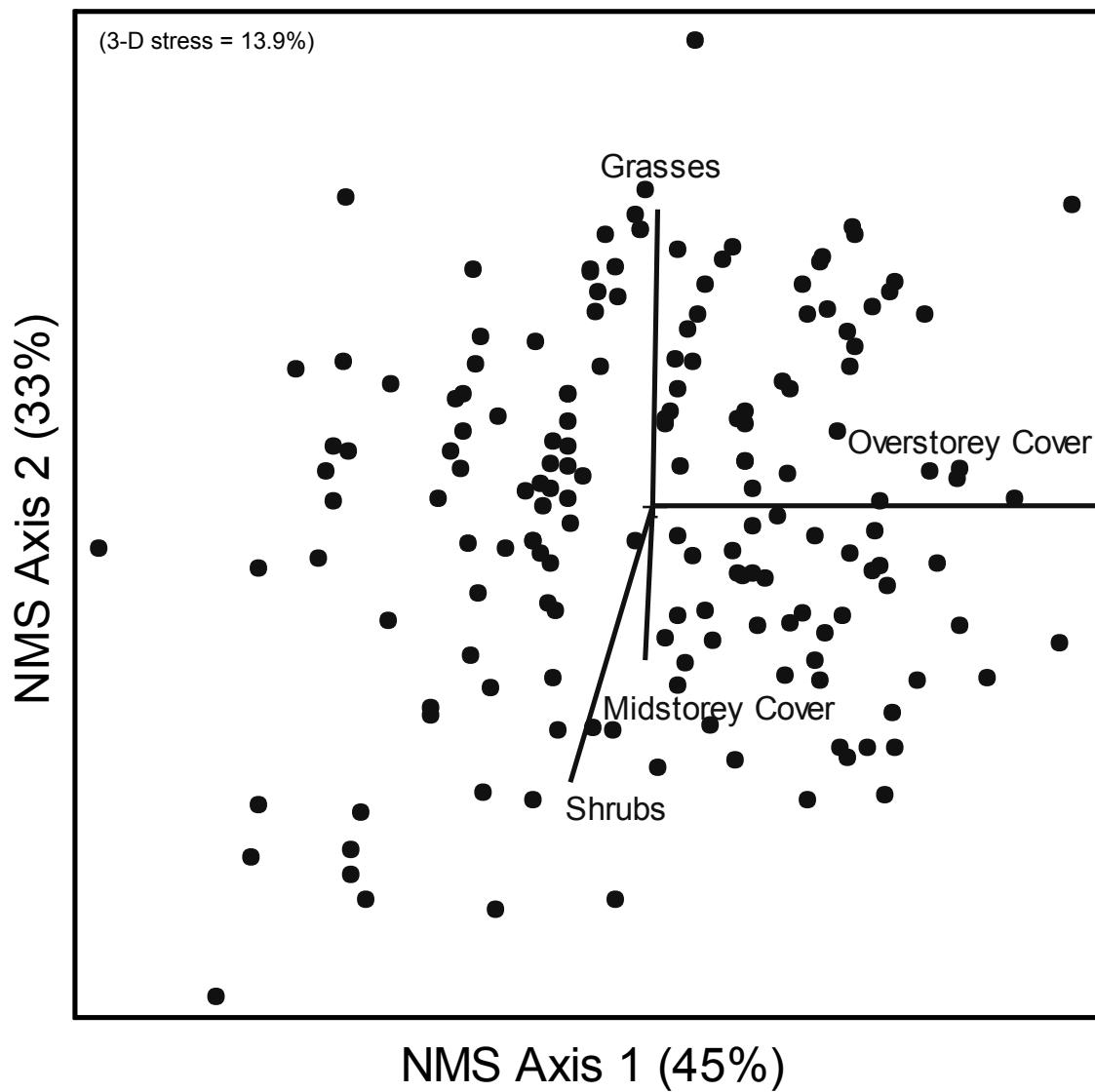


Figure 4.32: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 11 (open heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 166, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 13.9%, Axes 1 and 2 accounts for 78% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Sandy Creek Conservation Park, South Australia. Data collected October 2008.

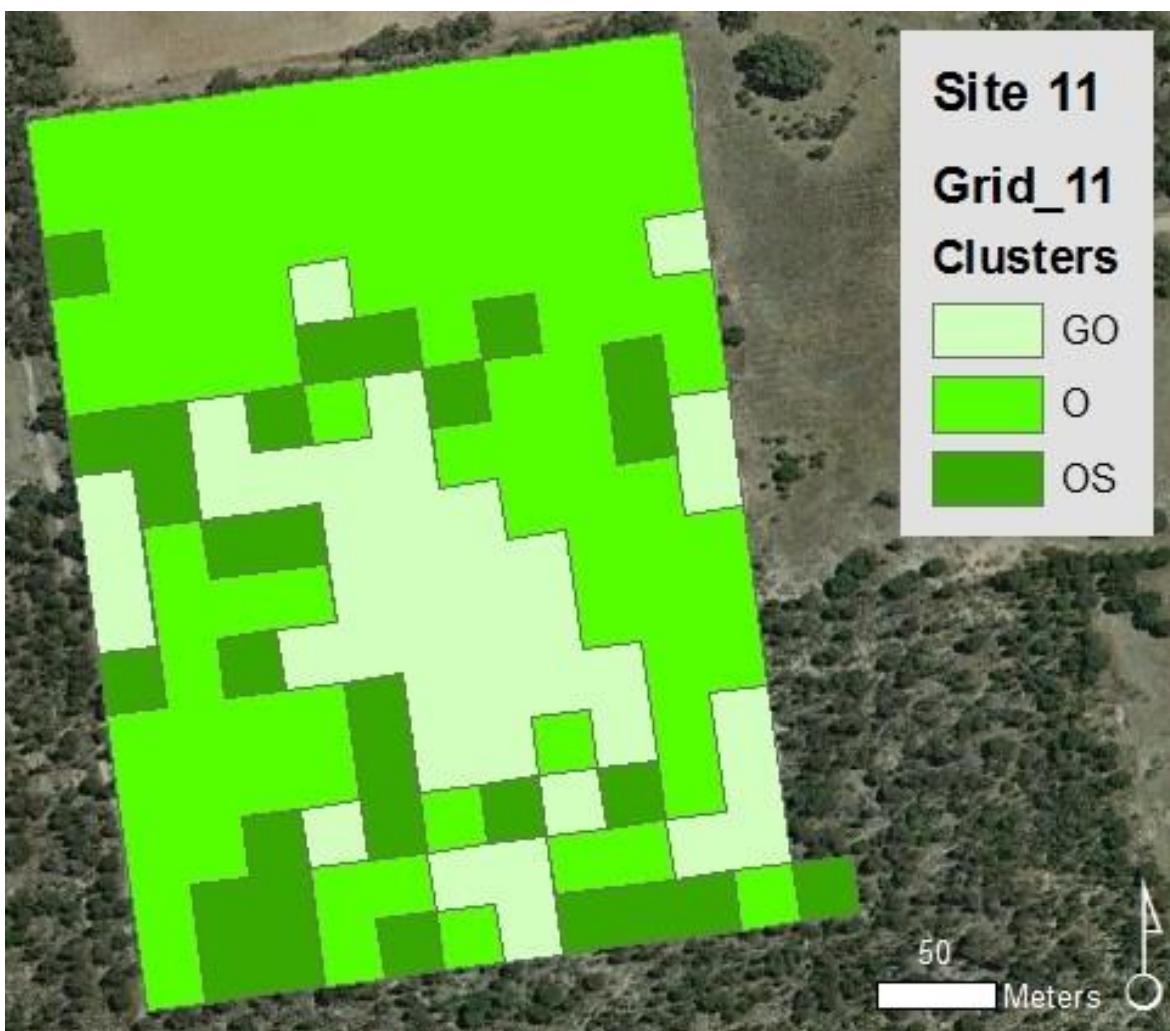


Figure 4.33: Habitat heterogeneity within Site 11 (open heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 166$) is colour-coded for the dendrogram cluster it aligned with. The three clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses and overstorey cover (GO), overstorey cover (O), and 3. overstorey cover and shrubs (OS). This site is located within the Sandy Creek Conservation Park, South Australia. Data collected October 2008.

Site 12 – closed heath woodland

The cluster analysis undertaken on the habitat attributes for Site 12 identified five vegetation clusters (Fig. 4.34). Regularized Discrimination Analysis for the vegetation clusters identified four functions, with the first three accounting for 99% of the variance (Table 4.36). Group association for the 25 m × 25 m squares within these three functions were well predicted (based on Wilks' Lambda), with all three being very highly significant (Table 4.37).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were overstorey cover, shrubs and midstorey cover (Table 4.38; Fig. 4.35). The least discriminatory variables in ascending order were grasses and weeds (Table 4.38). The habitat structure within each of the 25 m × 25 m squares appeared to range from intermixed to spatially aggregated with similar habitat features across this ten hectare site, based on the attributes that were measured (Fig. 4.36).

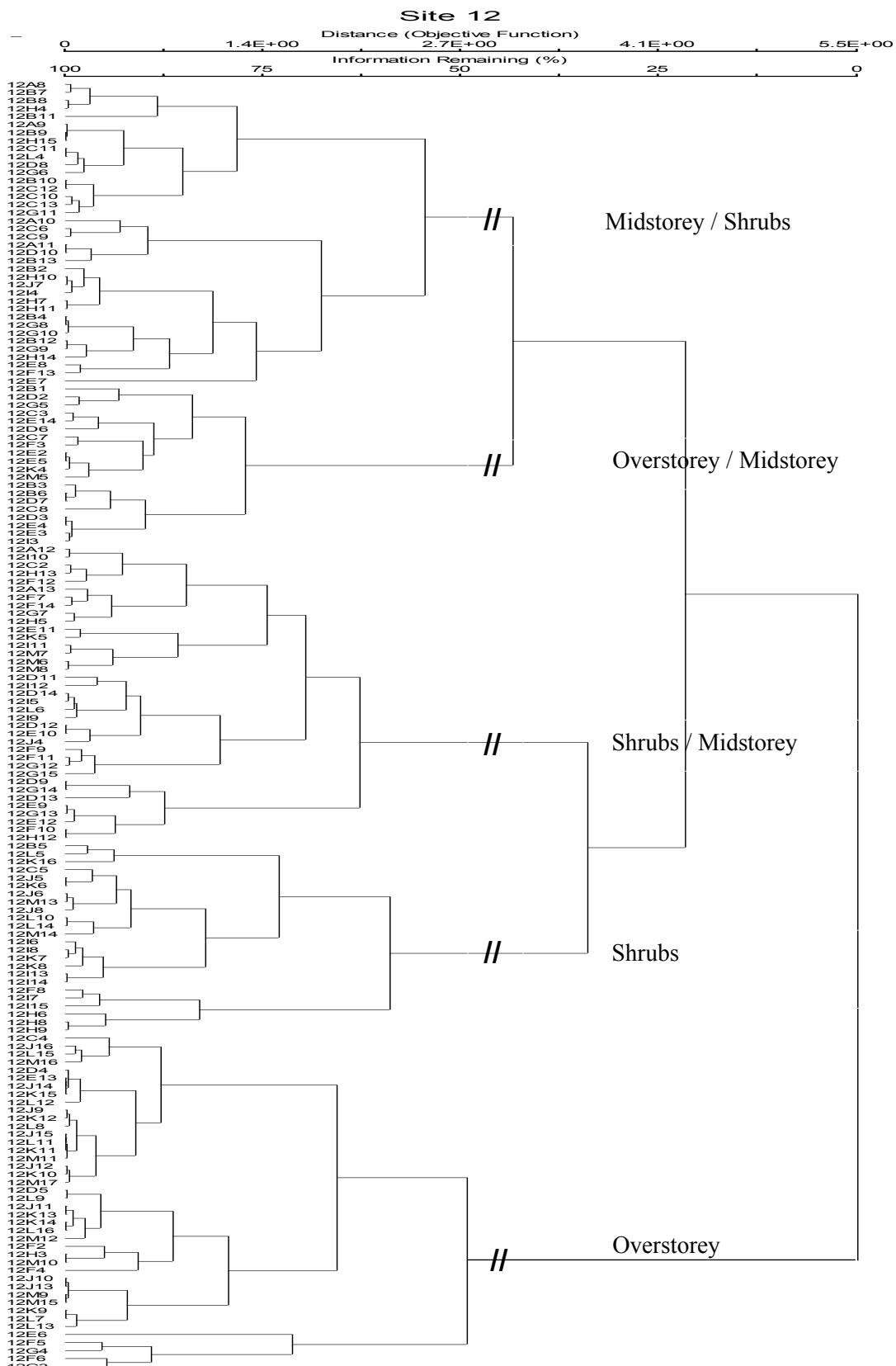


Figure 4.34: Hierarchical clustering of 161, 25 m × 25 m squares used to survey the entire ten hectares of Site 12 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.36: Eigenvalues for the four functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 12 (closed heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first three functions accounting for 99.60% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 3.89 | 57.35 | 57.35 | 0.892 |
| 2 | 2.03 | 29.92 | 87.27 | 0.818 |
| 3 | 0.84 | 12.33 | 99.60 | 0.675 |
| 4 | 0.03 | 0.40 | 100.00 | 0.163 |

Table 4.37: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 12 (closed heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob $> F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value (Prob $> F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis the first three functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob $> F$ |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.036 | 43.64 | 20 | 505 | 9576 | < 0.0001 |
| 2 | 0.175 | 31.45 | 12 | 405 | 4444 | < 0.0001 |
| 3 | 0.530 | 19.16 | 6 | 380 | 1895 | < 0.0001 |
| 4 | 0.973 | 2.13 | 2 | 155 | 154 | 0.123 |

Table 4.38: Average percent cover of habitat attributes within the five cluster types for Site 12 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 161$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the five statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover and shrubs (MS), 2. overstorey cover and midstorey cover (OM), 3. shrubs and midstorey cover (SM), 4. shrubs (S), and 5. overstorey cover (O). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Hale Conservation Park, South Australia. Data collected November 2008.

| Habitat attribute | MS ($n = 38$) | OM ($n = 20$) | SM ($n = 37$) | S ($n = 24$) | O ($n = 42$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|--------------------|--------------------|--------------------|-------------------|-------------------|--------------------|--|
| Grasses | 0.1 | 0.1 | 0.1 | 0.3 | 2.0 | 1.21 | 0.07 |
| Weeds | 0.0 | 0.2 | 0.0 | 0.0 | 2.6 | 4.07 | -0.10 |
| Shrubs | 21.1 | 16.8 | 51.7 | 24.8 | 14.9 | 82.16 | 0.41 |
| Midstorey | 60.1 | 50.3 | 32.8 | 13.0 | 10.0 | 84.51 | 0.51 |
| Overstorey | 9.4 | 51.5 | 23.8 | 25.4 | 57.9 | 105.80 | -0.67 |

Site 12

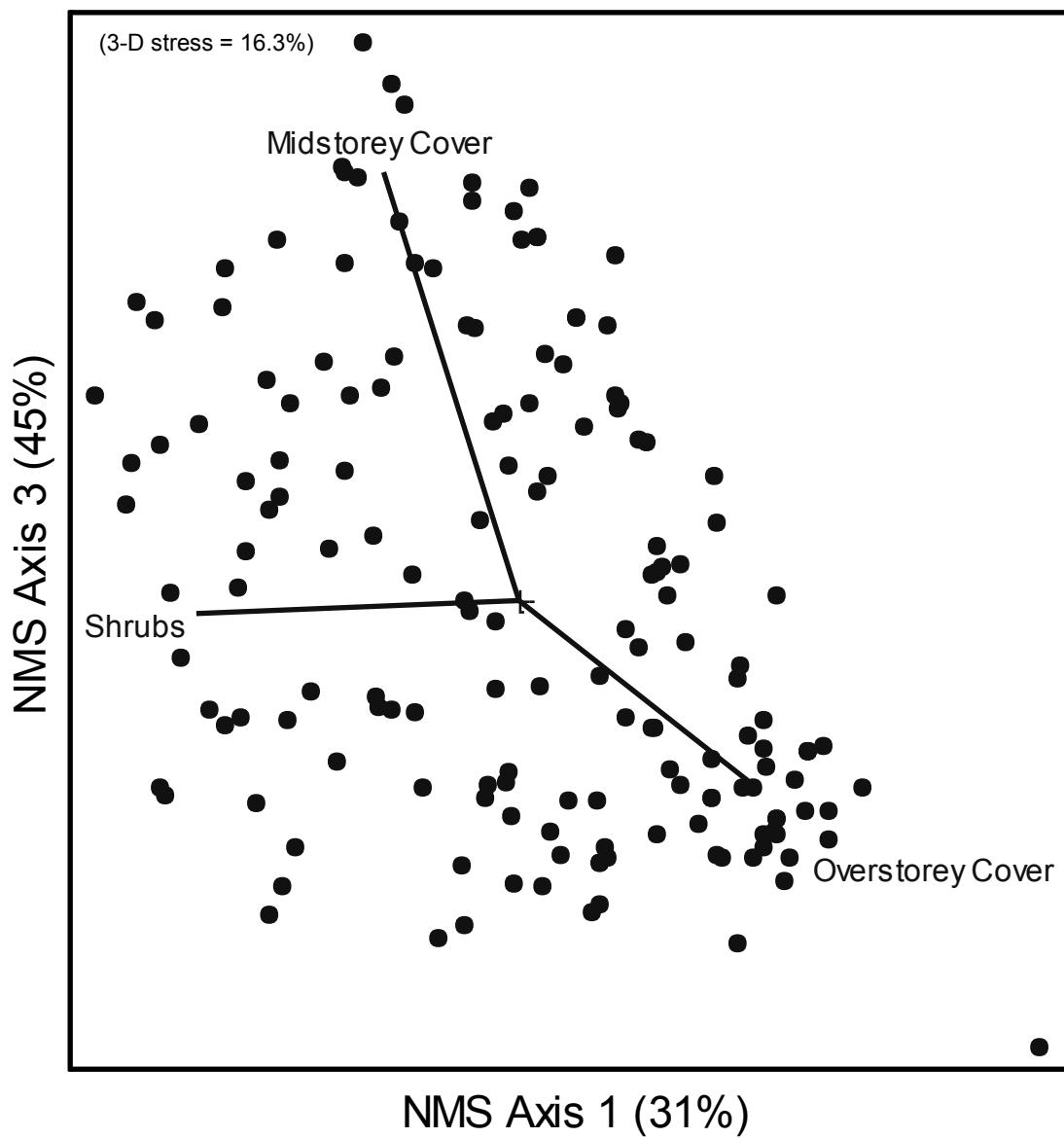


Figure 4.35: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 12 (closed heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 161, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 16.3%, Axes 1 and 3 accounts for 76% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Hale Conservation Park, South Australia. Data collected November 2008.

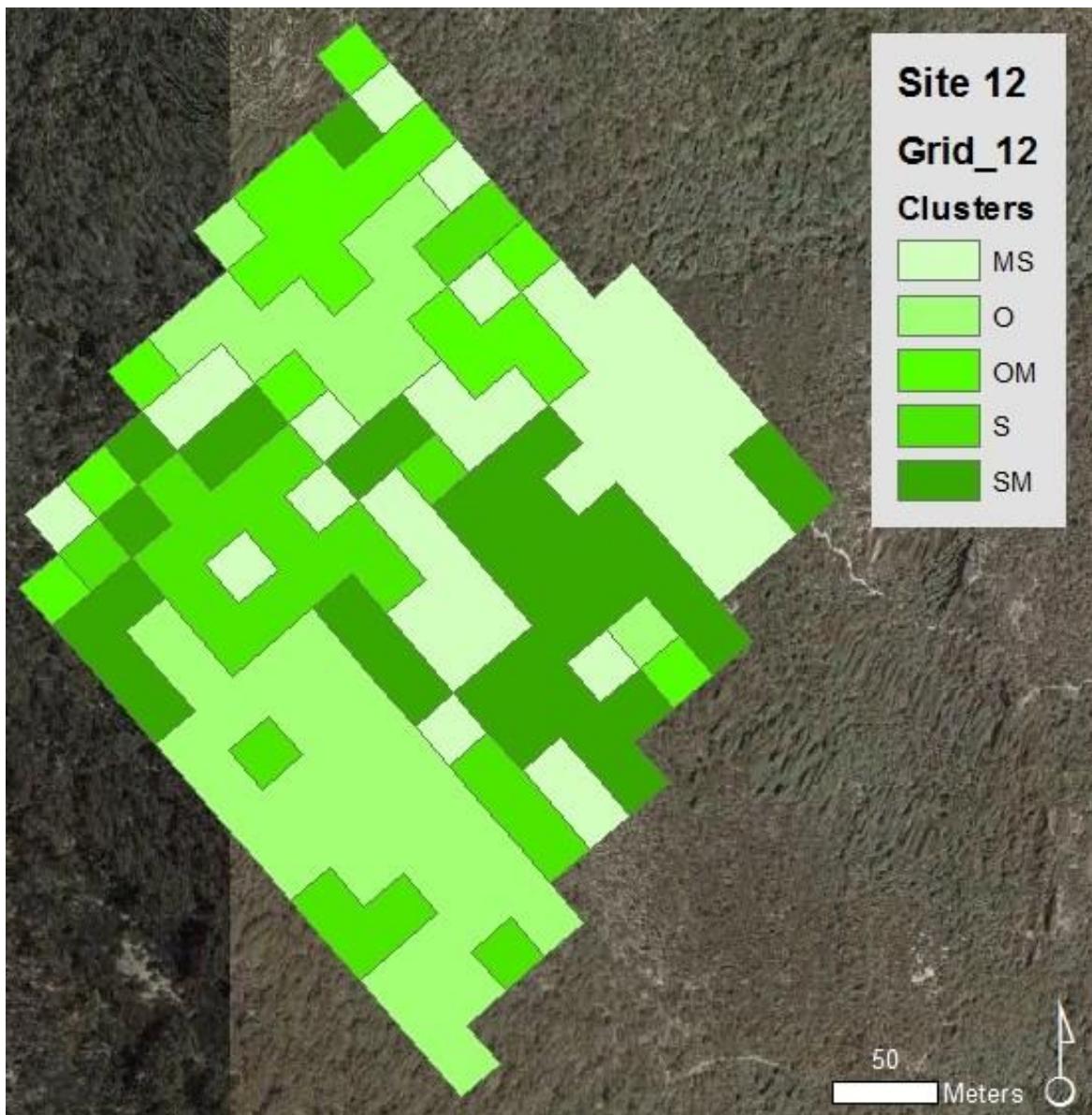


Figure 4.36: Habitat heterogeneity within Site 12 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 161$) is colour-coded for the dendrogram cluster it aligned with. The five clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover and shrubs (MS), 2. overstorey cover (O), 3. overstorey cover and midstorey cover (OM), 4. shrubs (S), and 5. shrubs and midstorey cover (SM). This site is located within the Hale Conservation Park, South Australia. Data collected November 2008.

Site 13 – closed heath woodland

The cluster analysis undertaken on the habitat attributes for Site 13 identified six vegetation clusters (Fig. 4.37). Regularized Discrimination Analysis for the vegetation clusters identified five functions, with the first three accounting for 99% of the variance (Table 4.39). Group association for the 25 m × 25 m squares within these three functions were well predicted (based on Wilks' Lambda), with all three being very highly significant (Table 4.40).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were overstorey cover, shrubs and midstorey cover (Table 4.41; Fig. 4.38). The least discriminatory variables in ascending order were grasses and weeds (Table 4.41). The habitat structure within each of the 25 m × 25 m squares appeared to be intermixed across this ten hectare site, based on the attributes that were measured (Fig. 4.39).

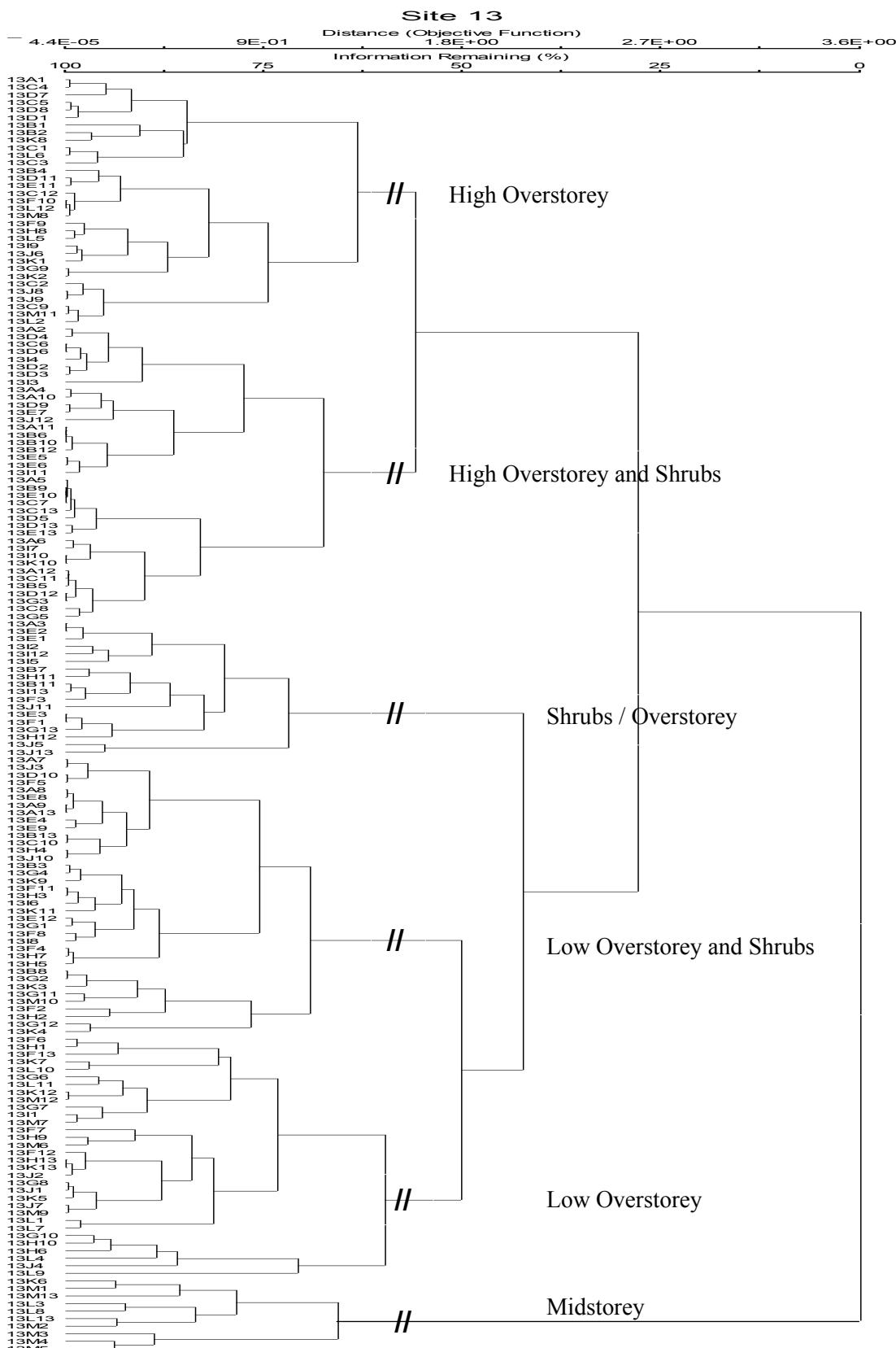


Figure 4.37: Hierarchical clustering of 169, 25 m × 25 m squares used to survey the entire ten hectares of Site 13 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.39: Eigenvalues for the five functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 13 (closed heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first three functions accounting for 99.89% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|--------------------|---------------------|--------------------|-----------------------|
| 1 | 5.15 | 57.58 | 57.58 | 0.915 |
| 2 | 3.26 | 36.40 | 93.98 | 0.875 |
| 3 | 0.53 | 5.91 | 99.89 | 0.588 |
| 4 | 0.01 | 0.10 | 99.99 | 0.096 |
| 5 | 4.7e ⁻⁵ | 0.01 | 100.00 | 0.007 |

Table 4.40: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 13 (closed heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated *F*-value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob > *F*). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the *F* distribution. Therefore, the "Approx. *F*" statistic and *p*-value (Prob > *F*) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis the first three functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. <i>F</i> | NumDF | DenDF | df | Prob > <i>F</i> |
|-------------------|---------------|------------------|-------|-------|-------|-----------------|
| 1 | 0.025 | 40.43 | 25 | 592 | 14184 | < 0.0001 |
| 2 | 0.152 | 26.06 | 16 | 489 | 7320 | < 0.0001 |
| 3 | 0.648 | 8.51 | 9 | 392 | 3128 | < 0.0001 |
| 4 | 0.991 | 0.38 | 4 | 324 | 969 | 0.823 |
| 5 | 0.999 | 0.01 | 1 | 163 | - | 0.929 |

Table 4.41: Average percent cover of habitat attributes within the six cluster types for Site 13 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 169$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the six statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. high overstorey cover (HO), 2. high overstorey cover and shrubs (HOS), 3. shrubs and overstorey cover (SO), 4. low overstorey cover and shrubs (LOS), 5. low overstorey cover (LO), and 6. midstorey cover (M). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Hale Conservation Park, South Australia. Data collected November 2008.

| Habitat attribute | HO ($n = 33$) | HOS ($n = 39$) | SO ($n = 18$) | LOS ($n = 37$) | LO ($n = 32$) | M ($n = 10$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|--------------------|---------------------|--------------------|---------------------|--------------------|-------------------|--------------------|--|
| Grasses | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.3 | 6.57 | -0.09 |
| Weeds | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 | 3.41 | -0.17 |
| Shrubs | 13.2 | 42.8 | 61.4 | 32.5 | 14.4 | 7.0 | 117.38 | 0.69 |
| Midstorey | 14.6 | 11.5 | 15.1 | 9.2 | 13.0 | 42.0 | 17.94 | -0.27 |
| Overstorey | 78.0 | 80.9 | 40.8 | 57.3 | 43.8 | 17.5 | 135.66 | 0.71 |

Site 13

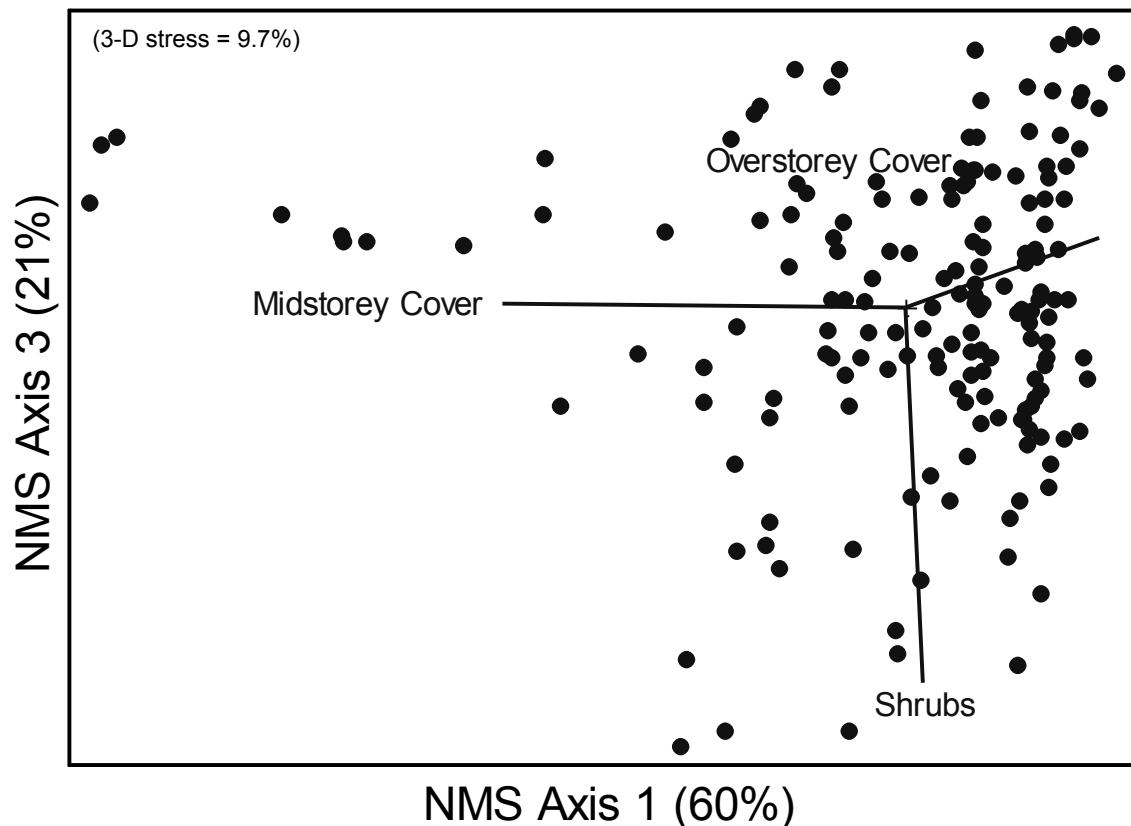


Figure 4.38: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 13 (closed heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 169, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 9.7%, Axes 1 and 3 accounts for 81% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Hale Conservation Park, South Australia. Data collected November 2008.



Figure 4.39: Habitat heterogeneity within Site 13 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 169$) is colour-coded for the dendrogram cluster it aligned with. The six clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. high overstorey cover (HO), 2. high overstorey cover and shrubs (HOS), 3. low overstorey cover (LO), 4. low overstorey cover and shrubs (LOS), 5. midstorey cover (M), and 6. shrubs and overstorey cover (SO). This site is located within the Hale Conservation Park, South Australia. Data collected November 2008.

Site 14 – open heath woodland

The cluster analysis undertaken on the habitat attributes for Site 14 identified four vegetation clusters (Fig. 4.40). Regularized Discrimination Analysis for the vegetation clusters identified three functions, with all three functions accounting for the variance (Table 4.42). Group association for the 25 m × 25 m squares within these three functions were well predicted (based on Wilks' Lambda), with all three being very highly significant (Table 4.43).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were overstorey cover, midstorey cover and grasses (Table 4.44; Fig. 4.41). The least discriminatory variables in ascending order were weeds and shrubs (Table 4.44). The habitat structure within each of the 25 m × 25 m squares appeared to range from intermixed to spatially aggregated with similar habitat features across this ten hectare site, based on the attributes that were measured (Fig. 4.42).

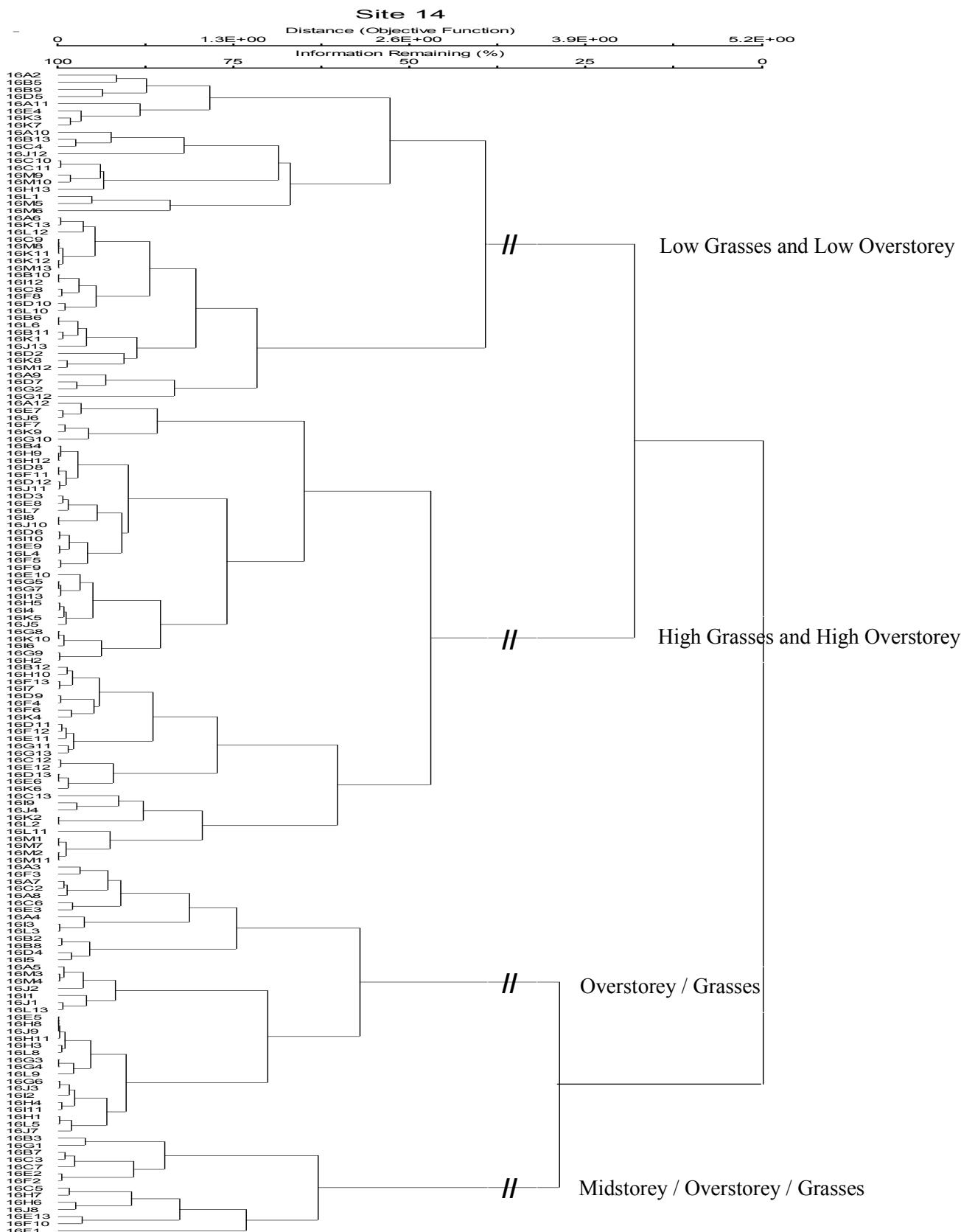


Figure 4.40: Hierarchical clustering of 163, 25 m × 25 m squares used to survey the entire ten hectares of Site 14 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained.

Table 4.42: Eigenvalues for the three functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 14 (open heath woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with all three functions accounting for the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 1.74 | 46.09 | 46.09 | 0.797 |
| 2 | 1.17 | 31.12 | 77.21 | 0.735 |
| 3 | 0.86 | 22.79 | 100.00 | 0.680 |

Table 4.43: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 14 (open heath woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated *F*-value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance (Prob > *F*). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the *F* distribution. Therefore, the "Approx. *F*" statistic and *p*-value (Prob > *F*) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis all three functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. <i>F</i> | NumDF | DenDF | df | Prob > <i>F</i> |
|-------------------|---------------|------------------|-------|-------|------|-----------------|
| 1 | 0.090 | 39.70 | 15 | 428 | 5978 | < 0.0001 |
| 2 | 0.247 | 39.45 | 8 | 312 | 2177 | < 0.0001 |
| 3 | 0.537 | 45.03 | 3 | 157 | 312 | < 0.0001 |

Table 4.44: Average percent cover of habitat attributes within the four cluster types for Site 14 (open heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 163$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the four statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. low grasses and low overstorey cover (LGLO), 2. high grasses and high overstorey cover (HGHO), 3. overstorey cover and grasses (OG), and 4. midstorey cover, overstorey cover and grasses (MOG). Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Altona woodland area, South Australia. Data collected October 2008.

| Habitat attribute | LGLO ($n = 46$) | HGHO ($n = 65$) | OG ($n = 38$) | MOG ($n = 14$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|----------------------|----------------------|--------------------|---------------------|--------------------|--|
| Grasses | 29.5 | 54.1 | 27.3 | 31.6 | 59.25 | -0.61 |
| Weeds | 0.7 | 0.8 | 1.1 | 1.1 | 0.85 | 0.16 |
| Shrubs | 4.3 | 1.6 | 0.9 | 0.4 | 11.30 | -0.04 |
| Midstorey | 7.8 | 8.6 | 16.7 | 61.1 | 62.81 | 0.22 |
| Overstorey | 29.3 | 42.7 | 81.3 | 38.9 | 66.54 | 0.77 |

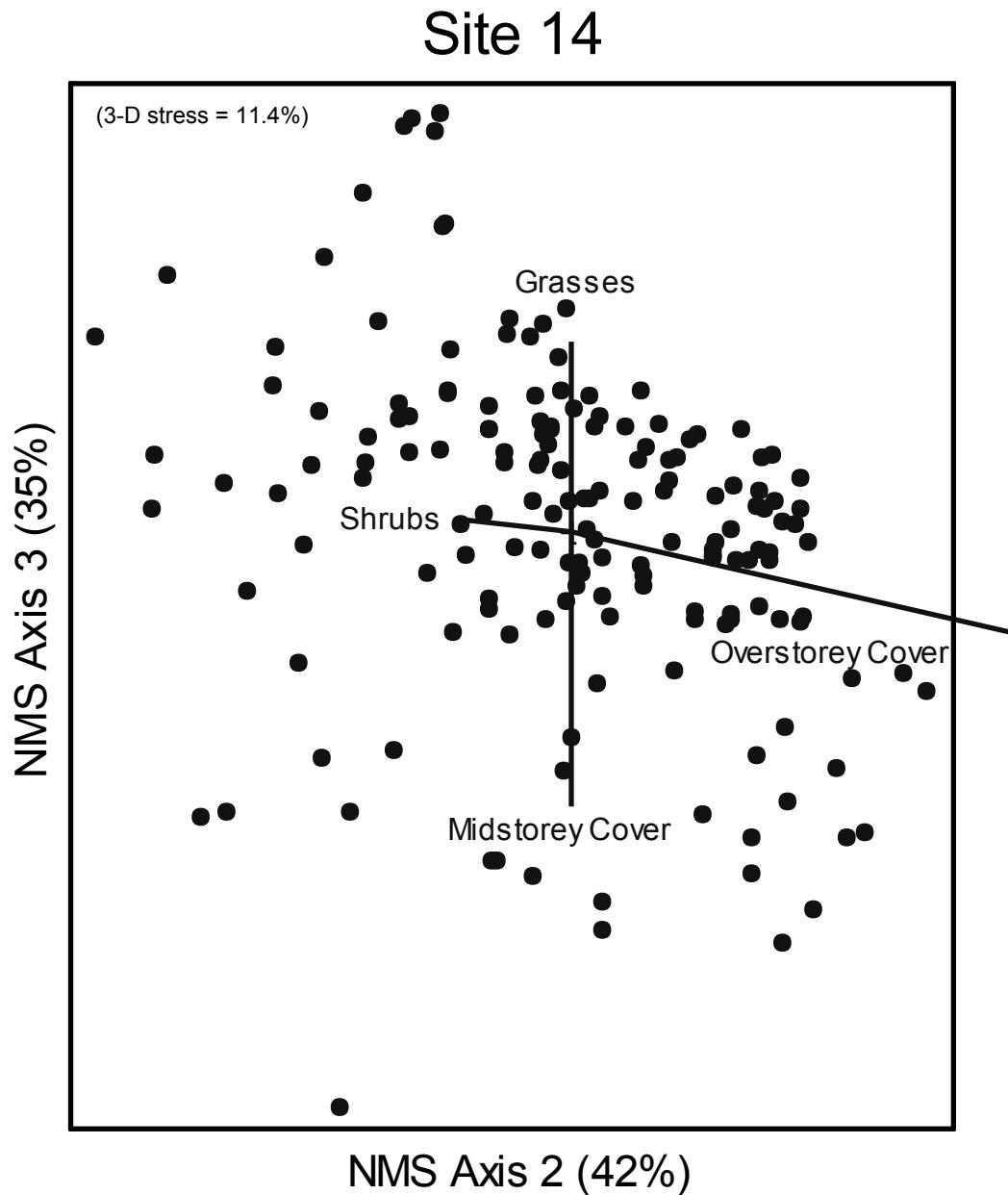


Figure 4.41: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 14 (open heath woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 163, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 11.4%, Axes 2 and 3 accounts for 77% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Altona woodland area, South Australia. Data collected October 2008.

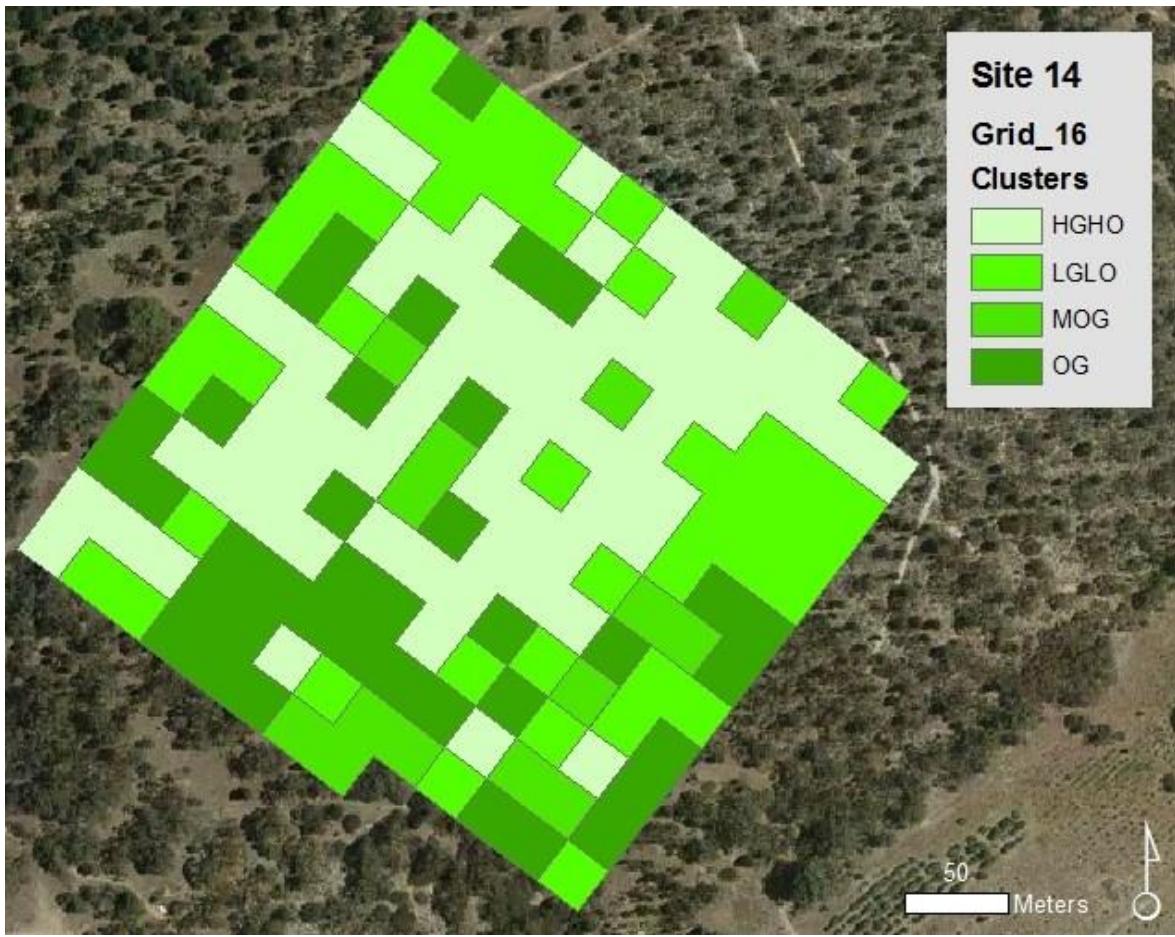


Figure 4.42: Habitat heterogeneity within Site 14 (open heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 163$) is colour-coded for the dendrogram cluster it aligned with. The four clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. high grasses and high overstorey cover (HGO), 2. low grasses and low overstorey cover (LGLO), 3. midstorey cover, overstorey cover and grasses (MOG), and 4. overstorey cover and grasses (OG). This site is located within the Altona woodland area, South Australia. Data collected October 2008.

Site 15 – grassy woodland

The cluster analysis undertaken on the habitat attributes for Site 15 identified four vegetation clusters and an outlier (Fig. 4.43). The outlier resulted from two 25 m × 25 m survey squares which contained very low percent covers of all the habitat variables that were measured. Regularized Discrimination Analysis for the vegetation clusters identified four functions, with the first three accounting for 98% of the variance (Table 4.45). Group association for the 25 m × 25 m squares within all four functions were well predicted (based on Wilks' Lambda), with all four being very highly significant (Table 4.46).

These results indicate a strong relationship existed between the vegetation clusters and the habitat variables that were used to identify them. Habitat attributes with the strongest discriminatory value in descending order were overstorey cover, shrubs and midstorey cover (Table 4.47; Fig. 4.44). The least discriminatory variables in ascending order were weeds and grasses (Table 4.47). The habitat structure within each of the 25 m × 25 m squares appeared to range from intermixed to spatially aggregated with similar habitat features across this ten hectare site, based on the attributes that were measured (Fig. 4.45).

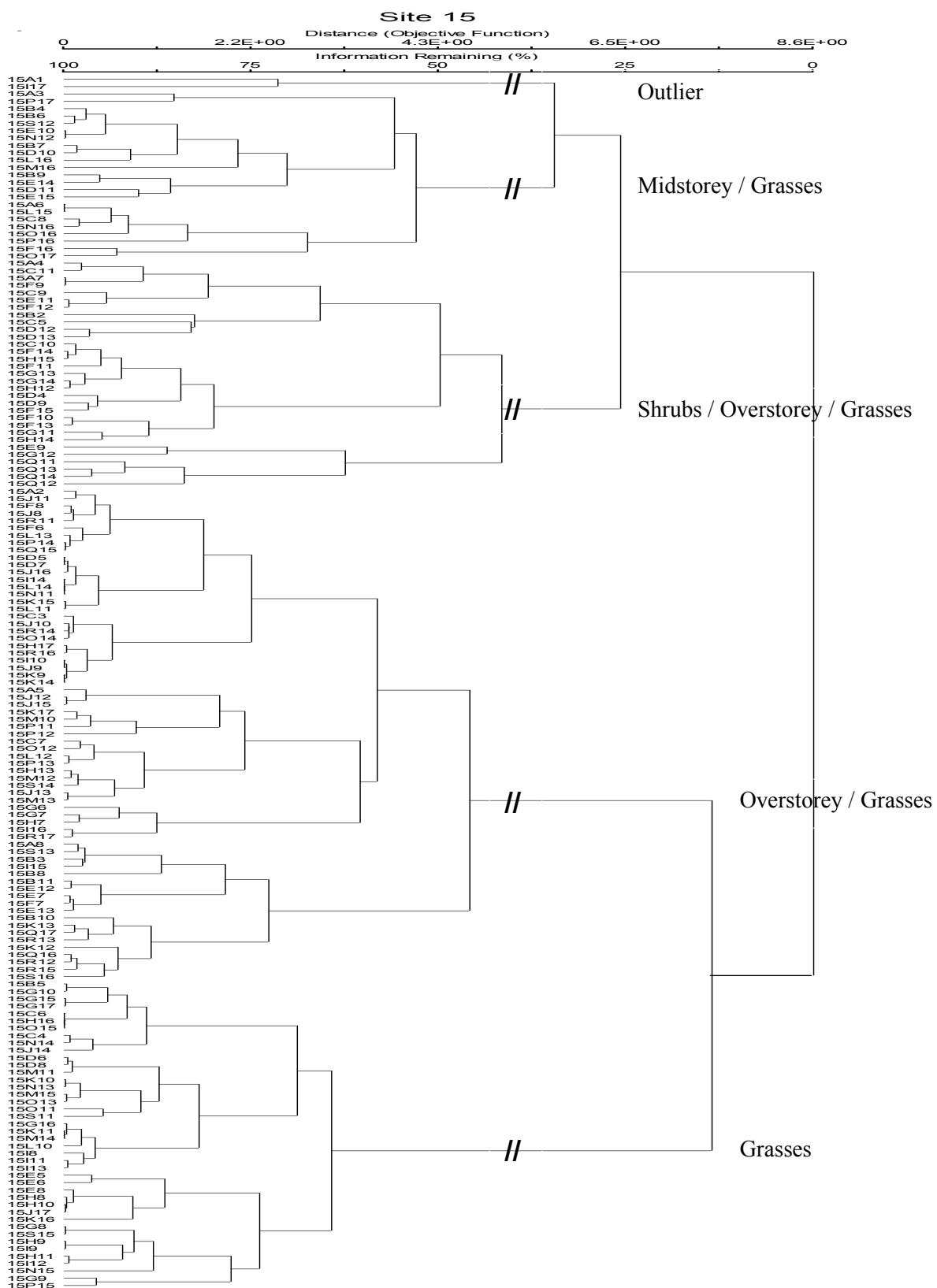


Figure 4.43: Hierarchical clustering of 165, 25 m × 25 m squares used to survey the entire ten hectares of Site 15 (the 25 m × 25 m cell codes indicate the site number and grid reference). Habitat attributes measured were the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation. The vegetation clusters are named after the dominant attributes they contained. The outlier resulted from two survey squares which contained very low percent covers of all the habitat variables measured.

Table 4.45: Eigenvalues for the four functions identified during Regularized Discrimination Analysis (JMP10 2012) for Site 15 (grassy woodland). The Function is the number of discriminant functions revealed by the analysis. The magnitudes of the Eigenvalues indicate the functions' discriminatory capability. Percent of variance relates to the proportion of the discriminatory ability of the variables found in a given function. Cumulative percent is the cumulative proportion of discriminatory ability. Canonical Correlation is the correlation of the predictor variables and the groupings. The proportion of variance represented by these axes totalled 100%, with the first three functions accounting for 97.93% of the variance.

| Function | Eigenvalue | Percent of variance | Cumulative percent | Canonical Correlation |
|----------|------------|---------------------|--------------------|-----------------------|
| 1 | 2.35 | 52.43 | 52.43 | 0.837 |
| 2 | 1.13 | 25.16 | 77.59 | 0.728 |
| 3 | 0.91 | 20.34 | 97.93 | 0.690 |
| 4 | 0.09 | 2.07 | 100.00 | 0.292 |

Table 4.46: Wilks' Lambda computations from Regularized Discrimination Analysis (JMP10 2012) with stepwise entry of the five habitat variables used to help interpret the cluster type classifications for Site 15 (grassy woodland). Test of functions are the number of functions applied to test the null hypothesis that the canonical correlations associated with the functions are all equal to zero. Wilks' Lambda is the multivariate statistic which portrays the proportion of the total variance in the discriminant scores that is not explained by the differences among the cluster type groups. Associated with Wilks' Lambda is the approximated F -value, degrees of freedom for the numerator (NumDF) and denominator (DenDF) (used to calculate the total degrees of freedom; $df = (\text{NumDF} - 1) * (\text{DenDF} - 1)$) and the statistical significance ($\text{Prob} > F$). Wilks' Lambda values are statistically tested using the chi-square approximation to test the null hypothesis that the means of all of the independent variables are equal across the groups of the dependent variables and have no discriminatory ability. The method used in JMP10 (2012) uses the approximation to the F distribution. Therefore, the "Approx. F " statistic and p -value ($\text{Prob} > F$) given to the right of Wilks Lambda correspond to the test of interest. The interpretation is the same as if the chi-squared approximation was used. In this analysis all four functions were very highly significant.

| Test of functions | Wilks' Lambda | Approx. F | NumDF | DenDF | df | Prob > F |
|-------------------|---------------|-------------|-------|-------|------|------------|
| 1 | 0.067 | 32.58 | 20 | 518 | 9823 | < 0.0001 |
| 2 | 0.225 | 26.22 | 12 | 416 | 4565 | < 0.0001 |
| 3 | 0.479 | 23.45 | 6 | 316 | 1575 | < 0.0001 |
| 4 | 0.915 | 7.38 | 2 | 159 | 158 | < 0.0001 |

Table 4.47: Average percent cover of habitat attributes within the four cluster types (and an outlier) for Site 15 (grassy woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m ($n = 165$). The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. The dendrogram that was produced based on the habitat attributes for each survey square produced the four statistically distinct clusters. Each cluster was individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. midstorey cover and grasses (MG), 2. shrubs, overstorey cover and grasses (SOG), 3. overstorey cover and grasses (OG), 4. grasses (G), and 5. outlier. Regularized Discrimination Analysis (RDA) with stepwise entry of the five habitat variables was used to help interpret the cluster type group classifications. The univariate F -ratio was used to indicate levels of significance as variables were added to the RDA model. The standardized discriminant function coefficient has been standardised to unit variance, so it can be used to directly interpret each variable's importance in contributing to the discriminant function. These coefficients are absolute values and their signs indicate the direction of the relationship. The most influential habitat variables may be deduced from the combination of the group mean, F -ratio and the standardized discriminant function coefficient. Attributes with the strongest discriminatory values are in bold. Values are the average percent cover for each habitat attribute (mean ± SE) within the corresponding survey squares for each cluster group (dominant attributes are in bold). n = the number of 25 m × 25 m squares per cluster. This site is located within the Altona woodland area, South Australia. Data collected October 2008.

| Habitat attribute | MG ($n = 23$) | SOG ($n = 31$) | OG ($n = 67$) | G ($n = 42$) | Outlier ($n = 2$) | Univariate F-ratio | Standardized discriminant function coefficient |
|-------------------|--------------------|---------------------|--------------------|-------------------|------------------------|--------------------|--|
| Grasses | 28.7 | 29.2 | 47.4 | 61.5 | 15.0 | 23.00 | 0.36 |
| Weeds | 5.0 | 2.7 | 8.4 | 7.7 | 0.5 | 5.83 | 0.19 |
| Shrubs | 8.0 | 30.2 | 2.6 | 3.2 | 2.5 | 55.63 | -0.60 |
| Midstorey | 43.0 | 12.4 | 17.8 | 5.4 | 0.5 | 26.49 | -0.02 |
| Overstorey | 11.1 | 29.7 | 60.1 | 11.8 | 0.0 | 63.19 | 0.80 |

Site 15

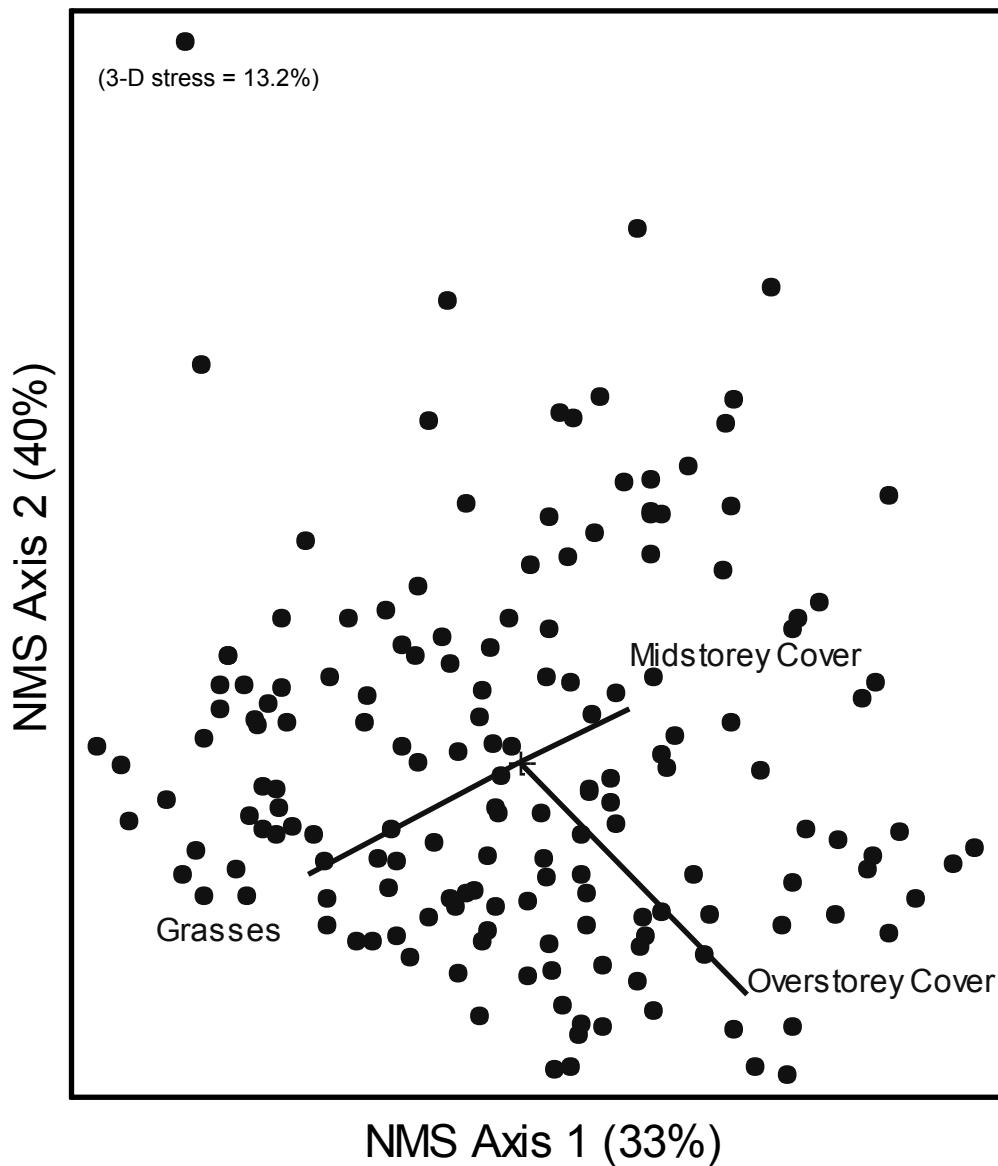


Figure 4.44: Three-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features within Site 15 (grassy woodland). Shown is the recommended 3-D solution displayed as a convenient 2-D representation. The data points (black circles) are the 165, 25 m × 25 m survey squares contained within this site. Within each 25 m × 25 m square the percent cover of grasses, weeds, shrubs (< 1 m), midstorey (1 – 4 m) and overstorey (> 4 m) were scored. The floristic variables shown have strong correlations with this ordination. The vector orientation of each floristic variable indicates the direction of positive increase and the length represents the strength of association within the data space of 25 m × 25 m squares. At a stress level of 13.2%, Axes 1 and 2 accounts for 73% of the variation that is represented by the distribution of the 25 m × 25 m squares. This site is located within the Altona woodland area, South Australia. Data collected October 2008.



Figure 4.45: Habitat heterogeneity within Site 15 (grassy woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured within each 25 m × 25 m survey square. Each survey square ($n = 165$) is colour-coded for the dendrogram cluster it aligned with. The four clusters are individually labelled to indicate the descending order of cover for the dominant habitat attributes they contained. The clusters are 1. grasses (G), 2. midstorey cover and grasses (MG), 3. overstorey cover and grasses (OG), and 4. shrubs, overstorey cover and grasses (SOG). This site is located within the Altona woodland area, South Australia. Data collected October 2008.

Discussion

Each of the 15 ten hectare sites contained statistically significant amounts of habitat heterogeneity at the scale of 25 m × 25 m (average 4.0 ± 0.3 vegetation clusters per site). The distribution of this heterogeneity ranged from being intermixed to spatial aggregations of the 25 m × 25 m squares that contained similar habitat features. The key finding was that significant structural heterogeneity existed within a relatively homogeneous type of woodland at the ten hectare scale (homogeneous as identified in Chapter 3). A further key finding was that each of these three woodland types contained an array of different combinations of heterogeneity. For example as a collective, the five grassy woodland sites contained 14 different types of vegetation clusters, the five open heath woodland sites contained eight different types of vegetation clusters, and the five closed heath woodland sites contained 12 different types of vegetation clusters.

This level of information is often missed by ecological surveys that measure at spatial scales that are much smaller than the habitat areas for which they are supposed to be representing (Mac Nally and Horrocks 2002; Watson 2003). These results also vindicate the labour intensive use of the ten hectare survey areas that were used to infer the habitat associations by bird species in Chapter 4. Clearly, by opting for much smaller “representative” samples within broad heterogeneous habitats, only serves to dramatically over-simplify the results and may miss important ecological information. Due to the significant habitat heterogeneity at the ten hectare scale, the choice of where spatially small sampling areas are placed could also dramatically influence the results, even within seemingly homogeneous habitats. Also, by simply sampling a random selection of plots in an area does not inform about how the different vegetation clusters are distributed across the site and that is part of the heterogeneity in itself. A random selection of points or places may define the bird-habitat associations, but it does not inform on the arrangements of different types of habitat in space.

In support, the need for a more holistic approach when measuring vegetation conditions has prompted calls for habitats to be assessed at much larger spatial scales than what is commonly practiced (Gibbons *et al.* 2006; Gibbons and Freudenberger 2006). These studies argue against the popular and more convenient forms of rapid on-ground

assessment, which are used to gauge the general condition of vegetation and habitats. Instead, more information is required on the condition of vegetation at broader scales, ranging from individual sites to a more regional context. This current study showed the level of local, within ten hectare, habitat heterogeneity that could fail to be captured by using a few smaller sampling areas and simply assuming that they were representative of the surrounding habitat area.

Site characteristics and environmental gradients can impose strong influences on the composition of the plant community at local scales. These include changes in the geology, soil fertility and moisture levels, together with site aspect, exposure and micro-topography (Kimber *et al.* 1999; Society for Ecological Restoration International Science & Policy Working Group 2004). Environmental cues such as these help to produce the high floristic and structural diversity that is common of remnant vegetation. Habitat heterogeneity, and in particular structural heterogeneity, is important for providing a range of key resources (e.g. foraging substrates and nesting sites) for a variety of bird species (e.g. Recher 1991; Munro *et al.* 2011).

In recognition for the importance of habitat heterogeneity through floristic diversity and structural complexity, it has been recommended that remnant vegetation should provide a template to help guide plantings that are intended for conservation gain (e.g. Kimber *et al.* 1999). At the fine-scale, it has been argued that revegetation should emulate remnant habitats, by matching plant species to landforms, establishing a range of storey levels, and in providing patches of both dense and sparse vegetation, together with small clearings (e.g. Bennett *et al.* 2000; Munro and Lindenmayer 2011).

However, these are broad guidelines only and do not offer specific strategies that can be used to inform revegetation works. For example, this level of information does not provide sufficient guidance for the fine-scale work that is performed by revegetation practitioners, such as the spatial configuration and/or densities that are required for new plantings (e.g. Ellis 2000; Corr 2003; Clarke *et al.* 2010). Decisions on how to apply these broad concepts rests very much with those directly involved with revegetation. For instance, the published literature has been described as offering rhetoric but not practical advice (Tony Randall, Programs Manager, Goolwa to Wellington Local Action Planning Association Inc., pers. comm.). When planning spacing and arrangements for eucalypt species, this

Association relies more on personal experience and expert advice rather than the published literature (Tony Randall, pers. comm.).

In agreement, a representative from Trees for Life considers that for revegetation guidance, the level of knowledge within the literature is not the same as personal experience (David Hein, Direct Seeding Manager, pers. comm.). Interestingly, the Trees for Life revegetation policy takes into account obvious landscape changes when deciding on which plant species to use and where. However, this simply identifies the habitat type or association that needs to be reconstructed and then there is a tendency that exactly the same mix of plants be used within each area within the patch being revegetated. This leads to a rather homogeneous and repetitive habitat structure being re-established, where the habitat that is established in one 25 m × 25 m plot is simply repeated for the other plots across the site. In contrast, this current study revealed strong heterogeneity existed within landscapes that appeared to be relatively homogeneous (e.g. comparatively flat with a consistent sandy soil surface throughout). The heterogeneity within the plant communities at the ten hectare scale revealed by this study, presumably resulted from more subtle environmental cues, which were beyond the scope of this study to identify. Nevertheless, this heterogeneity needs to be incorporated into planting programs, where adjacent 25 m × 25 m areas are deliberately planted to capture the range of habitat clusters and habitat features that are found in natural woodlands.

This current study provides some guidance as to how new plantings may be spatially arranged, for the design and re-construction of woodlands in general and specifically for the Para Woodland Reserve. The heterogeneity that has been shown relates to structural characteristics; however there will be further heterogeneity within these plots because of changes in the actual species of plants that contributed to the different layers of vegetation across a ten hectare site. The aim of this study though was to demonstrate that even at a basic structural level there was significant heterogeneity.

What is not clear is what influence this local, fine-scale heterogeneity has on woodland birds. Should the scale of this heterogeneity have no measurable influence on birds, then there is no strong argument for this level of heterogeneity to be actively replicated in re-vegetation works. However, if various bird species are responding to vegetation components at finer-scales than ten hectares, then these finer-scales need to be considered.

This call for heterogeneity at fine-scales would not see the various habitat features pooled and spread across a site, but rather planted as discrete patches (e.g. at the 25 m × 25 m scale). This would result in discrete patches of one structural type being intermingled with discrete patches of numerous other structural types of habitat. In the next chapter, I will explore how woodland birds are responding to the heterogeneity within the 15 ten hectare patches of woodland that I have chosen to study.

Chapter 5

Habitat associations by bird species within ten hectare sites

Introduction

Within regions that have undergone agricultural intensification, increasing habitat heterogeneity is considered to be a key in restoring and sustaining the habitat needs of temperate woodland bird species (Benton *et al.* 2003). The correlation between bird species richness and increasing habitat heterogeneity has long been recognised (e.g. Roth 1976; Freemark and Merriam 1986). It is also logical that structural habitat heterogeneity is strongly influenced by floristic diversity. That is, different plant species will differ in the types of habitat features that they provide (e.g. bark type, leaf shape, foliage density, etc.). Some debate exists however regarding the relative importance of plant species versus vegetation structure and habitat configuration (Rotenberry 1985; Muller *et al.* 2010). Nevertheless, the measure of habitat heterogeneity rests very much with the grain of resolution in which a habitat area is measured (Mac Nally and Watson 1997).

This project to date has demonstrated that at the scale of ten hectares, distinct woodland bird communities occupied different, relatively homogeneous woodland types (Chapter 3). It has also been shown that at finer-scales, these woodlands contained strong habitat heterogeneity that was based on different habitat structures and their spatial arrangements (Chapter 4). It should be noted however that the heterogeneity shown thus far is structural, but there may be further floristic heterogeneity that has not been documented. The importance of habitat heterogeneity for birds within the Mount Lofty Ranges may be best demonstrated by revealing those species that occupied a ten hectare area (from Chapter 3), that can be correlated with a distinct vegetation arrangement within that ten hectare site (from Chapter 4). Each of these ten hectare sites had contained considerable structural habitat heterogeneity that was mapped at the scale of 25 m × 25 m square.

In this chapter I ask the question, do birds associate with particular habitat structures within a broadly defined woodland type? To address this, I compared the distributional

data of the birds with the distributional data of the different vegetation clusters found within each of the ten hectare sites. This was done to determine if any bird species showed a strong predilection to use certain habitat patches (or clusters) within the ten hectare sites to a greater or lesser extent than expected by chance. Should individual bird species be disproportionately associated with particular habitat structures, then this reinforces the need to integrate similar heterogeneity into revegetation practice.

Methods

The methods of collecting and processing the vegetation and bird data that are used in this study have been described in Chapters 3 and 4. In summary, during the bird surveys the exact locations that the birds were first observed were recorded with a GPS waypoint. For the habitat survey, each ten hectare site was subdivided into 25 m × 25 m squares, where within each square the habitat was scored. For the study in this chapter, this enabled the locations of each bird species to be compared directly with the habitat within the 25 m × 25 m squares within which the birds were detected. The descriptions used in this chapter for the vegetation clusters (from Chapter 4) have been reduced to their acronyms. For example, in this chapter a vegetation cluster with the code OSM contained a high percent of overstorey cover, shrub cover and midstorey cover, in that order of dominance.

Indicator Species Analysis (ISA) was used to identify which bird species (if any) displayed high fidelity with any of the vegetation clusters within each of the ten hectare sites (see Chapter 3 for a detailed description of ISA, including the associated statistics and interpretation of the *p*-values). Indicator Species Analysis provided a statistical measure of the likelihood of each bird species being associated with a specific vegetation cluster. This is based on the relative abundance and the relative frequency that a particular bird species was recorded within habitat associated with each of the vegetation clusters within a particular ten hectare site. Any statistically significant associations between a bird species and a vegetation cluster can then be explored against the observations made during this study and the known ecology of that bird species.

These analyses were inspired by the strong within-site habitat heterogeneity that was revealed in Chapter 4. The fine-scale resolution (i.e. 25 m × 25 m) over such a large area (i.e. 10 ha) may have required more than the five bird surveys that were conducted at each site, to better match bird species with habitat structures at this spatial scale. However in this chapter, I take those initial datasets and explore them for significant associations between bird species and habitat features, aware that some associations may not be detectable. As such, the significant alpha level used for this study was set at ≤ 0.05 prior to rounding to two decimal places. This was aimed at also capturing any marginally significant associations between the bird species and the vegetation clusters within each site.

Results

Eleven of the 15 ten hectare sites that were surveyed contained at least one bird species that was statistically associated with a vegetation cluster (Tables 5.1 to 5.11). The habitat differences between the vegetation clusters within some ten hectare sites also helped to define the habitat associations by co-inhabiting bird species. For example in Site 5 (closed heath woodland), the pounce foraging Scarlet Robin (*Petroica boodang*) was statistically associated with, and was only ever detected within, vegetation cluster MOS (Table 5.1). The MOS cluster was the only cluster within this ten hectare site that contained significant amounts of midstorey vegetation (Chapter 4, Table 4.17). The vegetation was dominated by Slaty Sheoak (*Allocasuarina muelleriana*) and Golden Wattle (*Acacia pycnantha*) within the midstorey strata (Fig. 5.1). Many of the Golden Wattles were bowed with age, providing low vertical perching substrates. Scarlet Robins were observed to be using this plant species to scan the thick ground litter below prior to executing pounce manoeuvres. These perching substrates were also used by a pair of Scarlet Robins for courtship feeding.

Also in Site 5, the Superb Fairy-wren (*Malurus cyaneus*) was statistically associated with, and was only ever detected within, vegetation cluster SO (Table 5.1). This cluster contained the most shrub cover of any cluster within this ten hectare site (Chapter 4, Table 4.17). Superb Fairy-wrens recorded within this site were always detected using the shrub layer. Similarly, Superb Fairy-wrens detected within Site 7 (grazed woodland) were statistically associated with, and were only ever detected within, vegetation cluster GMOW

(Table 5.2). Within this ten hectare site, cluster GMOW was the only cluster that provided vegetation cover below four metres. Superb Fairy-wrens within this site were always found to be associated with Kangaroo Thorn (*Acacia paradoxa*).

Table 5.1: Indicator values for indicator bird species for Site 5 (closed heath woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.13) contained three statistically distinct vegetation clusters; being 1. shrubs and overstorey cover (SO), 2. midstorey cover, overstorey cover and shrubs (MOS), and 3. overstorey cover (O). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | SO | MOS | O | <i>p</i> -value |
|-------------------------------|-------------------------------------|-----------|-----------|-----------|-----------------|
| White-winged Chough | <i>Corcorax melanorhamphos</i> | 0 | 0 | 15 | 0.001 |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 15 | 0 | 0 | 0.003 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 1 | 0 | 11 | 0.019 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0 | 6 | 0 | 0.034 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 0 | 1 | 8 | 0.035 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 7 | 22 | 2 | 0.048 |
| Scarlet Robin | <i>Petroica boodang</i> | 0 | 6 | 0 | 0.052 |
| Silveryeye | <i>Zosterops lateralis</i> | 0 | 4 | 0 | 0.112 |
| Tawny Frogmouth | <i>Podargus strigoides</i> | 0 | 0 | 4 | 0.144 |
| Common Blackbird | <i>Turdus merula</i> | 0 | 3 | 0 | 0.176 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 3 | 10 | 2 | 0.183 |
| White-winged Triller | <i>Lalage tricolor</i> | 1 | 0 | 5 | 0.191 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 0 | 4 | 0 | 0.261 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 0 | 0 | 3 | 0.429 |
| Horsfield's Bronze-Cuckoo | <i>Chrysococcyx basalis</i> | 0 | 0 | 3 | 0.437 |
| Brown-headed Honeyeater | <i>Melithreptus brevirostris</i> | 2 | 0 | 4 | 0.467 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 2 | 1 | 0 | 0.538 |
| Red-browed Finch | <i>Neochmia temporalis</i> | 0 | 1 | 0 | 0.547 |
| Painted Button-quail | <i>Turnix varia</i> | 0 | 1 | 0 | 0.551 |
| Peaceful Dove | <i>Geopelia striata</i> | 0 | 1 | 0 | 0.562 |
| Grey Shrike-thrush | <i>Colluricincla harmonica</i> | 0 | 3 | 1 | 0.622 |
| Brown Thornbill | <i>Acanthiza pusilla</i> | 3 | 0 | 0 | 0.641 |
| Crimson Rosella | <i>Platycercus elegans</i> | 3 | 1 | 3 | 0.729 |
| Grey Currawong | <i>Strepera versicolor</i> | 0 | 2 | 2 | 0.741 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 0 | 2 | 0 | 0.746 |
| Dusky Woodswallow | <i>Artamus cyanopterus</i> | 0 | 2 | 0 | 0.753 |
| Crescent Honeyeater | <i>Phylidonyris pyrrhoptera</i> | 2 | 3 | 0 | 0.774 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 1 | 0 | 2 | 0.856 |
| White-browed Scrubwren | <i>Sericornis frontalis</i> | 2 | 2 | 0 | 0.889 |
| White-browed Babbler | <i>Pomatostomus superciliosus</i> | 2 | 3 | 1 | 0.930 |
| Brush Bronzewing | <i>Phaps elegans</i> | 2 | 1 | 0 | 1.000 |
| Eastern Spinebill | <i>Acanthorhynchus tenuirostris</i> | 1 | 1 | 0 | 1.000 |
| Fan-tailed Cuckoo | <i>Cacomantis flabelliformis</i> | 1 | 0 | 0 | 1.000 |
| Spotted Pardalote | <i>Pardalotus punctatus</i> | 1 | 1 | 0 | 1.000 |
| Striated Thornbill | <i>Acanthiza lineata</i> | 1 | 0 | 0 | 1.000 |

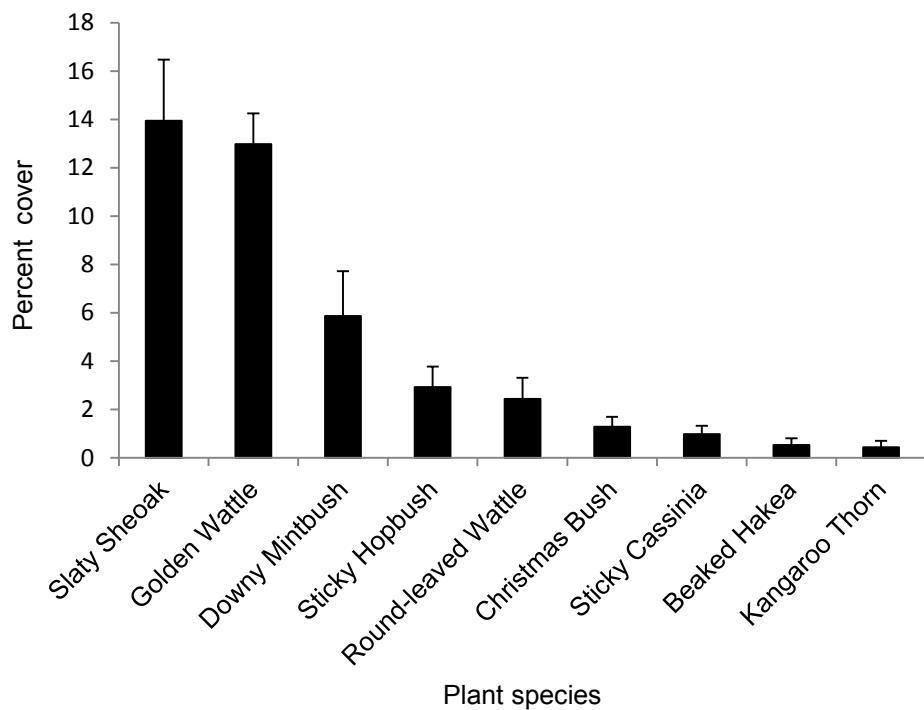


Figure 5.1: Midstorey plant species within Site 5 (closed heath woodland). The plant species are: Slaty Sheoak (*Allocasuarina muelleriana*), Golden Wattle (*Acacia pycnantha*), Downy Mintbush (*Prostanthera behriana*), Sticky Hopbush (*Dodonaea viscosa*), Round-leaved Wattle (*Ac. acinacea*), Christmas Bush (*Bursaria spinosa*), Sticky Cassinia (*Cassinia uncata*), Beaked Hakea (*Hakea rostrata*) and Kangaroo Thorn (*Ac. paradoxa*).

Table 5.2: Indicator values for indicator bird species for Site 7 (grazed woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.19) contained four statistically distinct vegetation clusters; being were 1. grasses, low overstorey cover and weeds (GLOW), 2. grasses and weeds (GW), 3. grasses, high overstorey cover and weeds (GHOW), and 4. grasses, midstorey cover, overstorey cover and weeds (GMOW). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant p -values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | GLOW | GW | GHOW | GMOW | <i>p</i> -value |
|--------------------------|-------------------------------------|------|----|------|-----------|-----------------|
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 0 | 0 | 0 | 38 | 0.001 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 0 | 0 | 0 | 13 | 0.065 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0 | 13 | 2 | 0 | 0.114 |
| White-plumed Honeyeater | <i>Lichenostomus penicillatus</i> | 0 | 2 | 2 | 14 | 0.134 |
| Peaceful Dove | <i>Geopelia striata</i> | 9 | 0 | 0 | 0 | 0.138 |
| Zebra Finch | <i>Taeniopygia guttata</i> | 9 | 0 | 0 | 0 | 0.139 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 0 | 0 | 11 | 0 | 0.201 |
| Grey Currawong | <i>Strepera versicolor</i> | 0 | 4 | 0 | 0 | 0.305 |
| Wedge-tailed Eagle | <i>Aquila audax</i> | 0 | 4 | 0 | 0 | 0.314 |
| Common Starling | <i>Sturnus vulgaris</i> | 1 | 0 | 2 | 8 | 0.353 |
| Willie Wagtail | <i>Rhipidura leucophrys</i> | 7 | 1 | 1 | 9 | 0.371 |
| Spotted Pardalote | <i>Pardalotus punctatus</i> | 0 | 4 | 0 | 0 | 0.523 |
| Red-rumped Parrot | <i>Psephotus haematonotus</i> | 0 | 1 | 6 | 0 | 0.547 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 7 | 5 | 2 | 0 | 0.575 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 0 | 5 | 2 | 0 | 0.589 |
| Tree Martin | <i>Hirundo nigricans</i> | 0 | 5 | 1 | 0 | 0.660 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 0 | 0 | 7 | 8 | 0.730 |
| Noisy Miner | <i>Manorina melanocephala</i> | 0 | 1 | 5 | 0 | 0.734 |
| Galah | <i>Cacatua roseicapilla</i> | 0 | 0 | 3 | 0 | 0.827 |
| Crimson Rosella | <i>Platycercus elegans</i> | 7 | 6 | 5 | 0 | 0.830 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 4 | 1 | 4 | 3 | 0.987 |
| Brown Treecreeper | <i>Climacteris picumnus</i> | 0 | 0 | 10 | 0 | 1.000 |
| Crested Pigeon | <i>Ocyphaps lophotes</i> | 0 | 0 | 2 | 0 | 1.000 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 0 | 0 | 1 | 0 | 1.000 |
| Hooded Robin | <i>Melanodryas cucullata</i> | 0 | 0 | 1 | 0 | 1.000 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 0 | 0 | 1 | 0 | 1.000 |
| Weebill | <i>Smicromys brevirostris</i> | 0 | 0 | 1 | 0 | 1.000 |
| Welcome Swallow | <i>Hirundo neoxena</i> | 0 | 0 | 1 | 0 | 1.000 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 0 | 0 | 1 | 0 | 1.000 |

Again in Site 5, the White-winged Chough (*Corcorax melanorhamphos*) was statistically associated with, and was only ever detected within, vegetation cluster O (Table 5.1).

Compared to the other clusters within this ten hectare site, cluster O contained the lowest shrub cover and was also low in midstorey cover (Chapter 4, Table 4.17). White-winged Choughs also attended a nest built high in a South Australian Blue Gum (*Eucalyptus leucoxylon*; refer Appendix B) within a 25 m × 25 m square that was associated with the O vegetation cluster, and foraged among the accumulated ground litter below the nest.

The availability of different foraging resources also helped to define habitat associations by some co-inhabiting bird species. In Site 5, the Red Wattlebird (*Anthochaera carunculata*) was statistically associated with vegetation cluster O, whereas within this same ten hectare site the New Holland Honeyeater (*Phylidonyris novaehollandiae*) was statistically associated with vegetation cluster MOS (Table 5.1). Cluster O was dominated by overstorey cover and contained little vegetation at lower storey levels, while cluster MOS was strongly represented by vegetation cover across all three storey levels (Chapter 4, Table 4.17). During this survey, Red Wattlebirds were always recorded foraging within the overstorey strata, whilst New Holland Honeyeaters were recorded foraging across the full spectrum of storey heights (Table 5.3).

Similarly in Site 4 (open woodland with few trees), the Red Wattlebird and Musk Lorikeet (*Glossopsitta concinna*) were statistically associated with patches that contained significant amounts of overstorey vegetation (cluster WOG), while the New Holland Honeyeater was statistically associated with patches that contained significant amounts of shrubby vegetation (cluster S; Table 5.4). Observations within this site again showed Red Wattlebirds and Musk Lorikeets foraged within the overstorey strata, while New Holland Honeyeaters foraged within the shrub level (refer Table 5.3).

Likewise, the Yellow-faced Honeyeater (*Lichenostomus chrysops*) was statistically associated with vegetation cluster M within Site 1 (grassy woodland; Table 5.5). Cluster M was dominated by Golden Wattles which were in flower, which the Yellow-faced Honeyeaters were observed to be forage amongst the inflorescence (refer Table 5.3).

Table 5.3: Plant species used for foraging by a selection of nectarivorous bird species within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. The codes used are ML for Musk Lorikeet (*Glossopsitta concinna*), NHH for New Holland Honeyeater (*Phylidonyris novaehollandiae*), RWB for Red Wattlebird (*Anthochaera carunculata*) and YFH for Yellow-faced Honeyeater (*Lichenostomus chrysops*). Ticks are used to identify the plant species that each bird species used. Data collected September – October 2008.

| Common name | Species name | ML | NHH | RWB | YFH |
|---------------------------|-----------------------------------|----|-----|-----|-----|
| Shrubs (< 1 m) | | | | | |
| Common Fringe-myrtle | <i>Calytrix tetragona</i> | ✓ | | | |
| Downy Mintbush | <i>Prostanthera behriana</i> | ✓ | | | |
| Flame Heath | <i>Astrolobma conostephioides</i> | ✓ | | | |
| Showy Parrot Pea | <i>Dillwynia sericea</i> | ✓ | | | |
| Yacca | <i>Xanthorrhoea semiplana</i> | ✓ | | | |
| Midstorey (1 – 4 m) | | | | | |
| Beaked Hakea | <i>Hakea rostrata</i> | ✓ | | | |
| Downy Mintbush | <i>Prostanthera behriana</i> | ✓ | | | |
| Golden Wattle | <i>Acacia pycnantha</i> | ✓ | | ✓ | |
| Hakea | <i>Hakea carinata</i> | ✓ | | ✓ | |
| Kangaroo Thorn | <i>Acacia paradoxa</i> | ✓ | | | |
| Silver Banksia | <i>Banksia marginata</i> | ✓ | | | |
| Slaty Sheoak | <i>Allocasuarina muelleriana</i> | ✓ | | | |
| Overstorey (> 4 m) | | | | | |
| Long-leaved Box | <i>Eucalyptus goniocalyx</i> | ✓ | ✓ | ✓ | |
| Oyster Bay Pine | <i>Callitris rhomboidea</i> | ✓ | | | |
| Peppermint Box | <i>Eucalyptus odorata</i> | ✓ | | | |
| Pink Gum | <i>Eucalyptus fasciculosa</i> | | ✓ | ✓ | ✓ |
| Radiata Pine | <i>Pinus radiata</i> | | | | ✓ |
| Slender Cypress-pine | <i>Callitris gracilis</i> | | ✓ | | |
| South Australian Blue Gum | <i>Eucalyptus leucoxylon</i> | ✓ | ✓ | ✓ | ✓ |

Table 5.4: Indicator values for indicator bird species for Site 4 (open woodland with few trees) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.10) contained three statistically distinct vegetation clusters; being 1. grasses and weeds (GW), 2. weeds, overstorey cover and grasses (WOG), and 3. shrubs (S). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | GW | WOG | S | <i>p</i> -value |
|------------------------|-------------------------------------|----|-----------|-----------|-----------------|
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 0 | 0 | 73 | 0.001 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 2 | 77 | 2 | 0.006 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 0 | 28 | 0 | 0.037 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0 | 0 | 25 | 0.084 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 2 | 26 | 0 | 0.243 |
| Galah | <i>Cacatua roseicapilla</i> | 2 | 14 | 0 | 0.502 |
| Australian Raven | <i>Corvus coronoides</i> | 0 | 9 | 0 | 0.678 |
| Crimson Rosella | <i>Platycercus elegans</i> | 10 | 7 | 0 | 0.908 |
| Brush Bronzewing | <i>Phaps elegans</i> | 0 | 3 | 0 | 1.000 |
| Little Raven | <i>Corvus mellori</i> | 0 | 3 | 0 | 1.000 |
| Painted Button-quail | <i>Turnix varia</i> | 0 | 3 | 0 | 1.000 |
| Red-rumped Parrot | <i>Psephotus haematonotus</i> | 0 | 3 | 0 | 1.000 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 0 | 3 | 0 | 1.000 |
| Welcome Swallow | <i>Hirundo neoxena</i> | 0 | 6 | 0 | 1.000 |

Table 5.5: Indicator values for indicator bird species for Site 1 (grassy woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.1) contained six statistically distinct vegetation clusters; being 1. grasses, overstorey cover and midstorey cover (GOM), 2. grasses and overstorey cover (GO), 3. grasses and weeds (GW), 4. grasses, midstorey cover and weeds (GMW), 5. overstorey cover (O), and 6. midstorey cover (M). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | GOM | GO | GW | GMW | O | M | <i>p</i> -value |
|--------------------------------|----------------------------------|-----|----|----|-----------|-----------|-----------|-----------------|
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0 | 1 | 0 | 0 | 18 | 0 | 0.008 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 3 | 2 | 0 | 17 | 0 | 1 | 0.030 |
| Yellow-faced Honeyeater | <i>Lichenostomus chrysops</i> | 0 | 0 | 0 | 1 | 1 | 12 | 0.059 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 0 | 0 | 0 | 0 | 0 | 10 | 0.081 |
| Little Raven | <i>Corvus mellori</i> | 0 | 0 | 2 | 0 | 10 | 0 | 0.117 |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 12 | 0 | 1 | 4 | 1 | 2 | 0.124 |
| Fan-tailed Cuckoo | <i>Cacomantis flabelliformis</i> | 7 | 0 | 0 | 0 | 0 | 0 | 0.185 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 2 | 1 | 1 | 1 | 2 | 10 | 0.190 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 0 | 0 | 0 | 7 | 7 | 0 | 0.219 |
| Black-faced Cuckoo-shrike | <i>Coracina novaehollandiae</i> | 0 | 0 | 0 | 0 | 7 | 0 | 0.333 |
| Crimson Rosella | <i>Platycercus elegans</i> | 0 | 5 | 9 | 4 | 3 | 0 | 0.438 |
| Galah | <i>Cacatua roseicapilla</i> | 0 | 0 | 5 | 0 | 3 | 0 | 0.459 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 0 | 2 | 4 | 0 | 2 | 0 | 0.527 |
| Red-rumped Parrot | <i>Psephotus haematonotus</i> | 5 | 0 | 0 | 0 | 2 | 0 | 0.662 |
| Grey Currawong | <i>Strepera versicolor</i> | 1 | 4 | 0 | 2 | 0 | 0 | 0.709 |
| White-winged Chough | <i>Corcorax melanorhamphos</i> | 4 | 2 | 1 | 0 | 1 | 0 | 0.761 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 0 | 3 | 2 | 0 | 3 | 0 | 0.814 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 5 | 5 | 4 | 3 | 3 | 5 | 0.999 |
| Brown Thornbill | <i>Acanthiza pusilla</i> | 0 | 4 | 0 | 0 | 0 | 0 | 1.000 |
| Grey Shrike-thrush | <i>Colluricinclla harmonica</i> | 4 | 0 | 0 | 0 | 0 | 0 | 1.000 |
| Laughing Kookaburra | <i>Dacelo novaeguineae</i> | 0 | 4 | 0 | 0 | 0 | 0 | 1.000 |

The structural distinction between vegetation clusters within some ten hectare sites helped to define multiple habitat used by some co-inhabiting bird species. For example in Site 11 (open heath woodland), the Red-browed Finch (*Neochmia temporalis*), Rufous Whistler (*Pachycephala rufiventris*), Silvreye (*Zosterops lateralis*) and Striated Pardalote

(*Pardalotus striatus*) were all statistically associated with vegetation cluster OS (Table 5.6). Compared to the other two vegetation clusters within this ten hectare site, cluster OS was more open with 50% less overstorey cover (at 25%; Chapter 4, Table 4.35). Cluster OS was also the only cluster that contained significant habitat heterogeneity above the cut-off threshold of > 20% for shrubs, with shrub cover at 21% (Chapter 4, Table 4.35).

Using habitat that was associated with more open vegetation within Site 11 (cluster OS), Red-browed Finches were regularly observed to be foraging on the ground for fallen seeds of Slender Cypress-pine (*Callitris gracilis*) and Drooping Sheoak (*Allocasuarina verticillata*; refer Appendix C). This is despite the fact that these two plant species were readily available throughout this entire ten hectare site (Fig. 5.2).

Similarly in Site 11, the Rufous Whistler was associated with the more open but floristically more heterogeneous habitat. In comparison, the Golden Whistler (*Pachycephala pectoralis*) was statistically associated with vegetation cluster MOG within Site 14 (open heath woodland; Table 5.7). Cluster MOG was the only cluster within this ten hectare site to contain closed heath vegetation, with 61% cover within the midstorey strata (Chapter 4, Table 4.44). The midstorey was dominated by Kangaroo Thorn and immature Slender Cypress-pine, both of which Golden Whistlers were observed to be using.

As with the Rufous Whistler, the Silvireye also appeared to respond to the more heterogeneous OS cluster within Site 11 (Table 5.6). Silvereyes were observed to forage at various heights, which included fallen timber and various shrub and overstorey plant species.

The only bird species that was statistically associated with a vegetation cluster other than cluster OS within Site 11 was the Yellow Thornbill (*Acanthiza nana*) with cluster GO (Table 5.6). Cluster GO contained a much higher average number of Slender Cypress-pine per 25 m × 25 m survey square (Fig. 5.2). Anecdotally, within this ten hectare site Yellow Thornbills were recorded using Slender Cypress-pine for foraging 81% of the time ($n = 48$ records).

Table 5.6: Indicator values for indicator bird species for Site 11 (open heath woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.31) contained three statistically distinct vegetation clusters; being 1. overstorey cover (O), 2. grasses and overstorey cover (GO), and 3. overstorey cover and shrubs (OS). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | O | GO | OS | <i>p</i> -value |
|---------------------------|------------------------------------|----|-----------|-----------|-----------------|
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 1 | 0 | 23 | 0.001 |
| Red-browed Finch | <i>Neochmia temporalis</i> | 0 | 0 | 10 | 0.028 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 2 | 0 | 12 | 0.035 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 10 | 27 | 5 | 0.039 |
| Silvereye | <i>Zosterops lateralis</i> | 1 | 2 | 12 | 0.051 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 0 | 0 | 6 | 0.137 |
| Crimson Rosella | <i>Platycercus elegans</i> | 4 | 0 | 10 | 0.154 |
| Peaceful Dove | <i>Geopelia striata</i> | 7 | 0 | 0 | 0.166 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 2 | 8 | 0 | 0.170 |
| Red-capped Robin | <i>Petroica goodenovii</i> | 0 | 8 | 4 | 0.208 |
| Weebill | <i>Smicromis brevirostris</i> | 8 | 1 | 1 | 0.304 |
| White-winged Chough | <i>Corcorax melanorhamphos</i> | 6 | 0 | 2 | 0.339 |
| White-plumed Honeyeater | <i>Lichenostomus penicillataus</i> | 0 | 3 | 0 | 0.396 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 0 | 3 | 0 | 0.409 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 4 | 0 | 0 | 0.552 |
| Common Starling | <i>Sturnus vulgaris</i> | 2 | 0 | 0 | 0.650 |
| Black-faced Cuckoo-shrike | <i>Coracina novaehollandiae</i> | 2 | 0 | 0 | 0.679 |
| Red-rumped Parrot | <i>Psephotus haematonotus</i> | 2 | 0 | 0 | 0.679 |
| Little Raven | <i>Corvus mellori</i> | 2 | 0 | 0 | 0.680 |
| Brown-headed Honeyeater | <i>Melithreptus brevirostris</i> | 1 | 2 | 0 | 0.776 |
| Yellow-rumped Thornbill | <i>Acanthiza chrysorrhoa</i> | 2 | 6 | 6 | 0.818 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 3 | 4 | 1 | 0.923 |
| Grey Currawong | <i>Strepera versicolor</i> | 1 | 1 | 0 | 1.000 |
| Grey Shrike-thrush | <i>Colluricinclla harmonica</i> | 0 | 2 | 0 | 1.000 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 0 | 2 | 0 | 1.000 |
| Rainbow Lorikeet | <i>Trichoglossus haematodus</i> | 1 | 0 | 0 | 1.000 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 0 | 2 | 0 | 1.000 |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 1 | 1 | 0 | 1.000 |

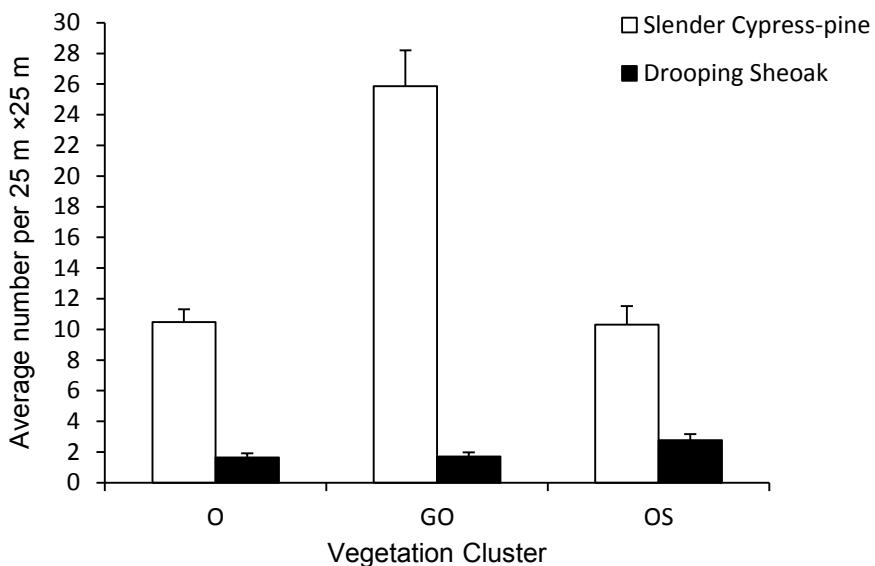


Figure 5.2: Average number of Slender Cypress-pine (*Callitris gracilis*) and Drooping Sheoak (*Allocasuarina verticillata*) in three vegetation clusters within a ten hectare site of open heath woodland (Site 11). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. There were 166, 25 m × 25 m squares surveyed within this site. Among the habitat attributes that were measured within each 25 m × 25 m survey square was the number of Slender Cypress-pine and Drooping Sheoak above four metres high (i.e. overstorey vegetation). The dendrogram produced for this site using PC-ORD5 (McCune and Mefford 1999b) contained three vegetation clusters. These vegetation clusters were confirmed to be statistically distinct from one another using Non-parametric Multi-response Permutation Procedures (MRPP) with Sørensen (Bray-Curtis) distance measures using PC-ORD5. MRPP was used to test the null hypothesis that no differences existed among the vegetation clusters that were identified. This analysis returned a delta p value of < 0.0001. The vegetation clusters are labelled to indicate the descending order of percent cover for the most dominant habitat attributes. Vegetation cluster O was high in overstorey cover, vegetation cluster GO was high in grass cover and overstorey cover and vegetation cluster OS was high in overstorey cover and shrub cover. Vegetation cluster O contained 95, 25 m × 25 m survey squares, vegetation cluster GO contained 41, 25 m × 25 m survey squares and vegetation cluster OS contained 30, 25 m × 25 m survey squares. Error bars are Standard Error. Data collected October – November 2008.

Table 5.7: Indicator values for indicator bird species for Site 14 (open heath woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.40) contained four statistically distinct vegetation clusters; being 1. low grasses and low overstorey cover (LGLO), 2. high grasses and high overstorey cover (HGHO), 3. overstorey cover and grasses (OG), and 4. midstorey cover, overstorey cover and grasses (MOG). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | LGLO | HGHO | OG | MOG | <i>p</i> -value |
|---------------------------|-------------------------------------|------|------|----|-----------|-----------------|
| Golden Whistler | <i>Pachycephala pectoralis</i> | 0 | 0 | 0 | 16 | 0.003 |
| Yellow-rumped Thornbill | <i>Acanthiza chrysorrhoa</i> | 7 | 0 | 0 | 0 | 0.105 |
| Shining Bronze-Cuckoo | <i>Chrysococcyx lucidus</i> | 0 | 0 | 0 | 6 | 0.106 |
| Varied Sittella | <i>Daphoenositta chrysoptera</i> | 0 | 0 | 0 | 6 | 0.110 |
| Red-capped Robin | <i>Petroica goodenovii</i> | 9 | 2 | 0 | 3 | 0.206 |
| Common Blackbird | <i>Turdus merula</i> | 0 | 1 | 0 | 5 | 0.209 |
| Grey Shrike-thrush | <i>Colluricinclla harmonica</i> | 0 | 0 | 1 | 4 | 0.266 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0 | 0 | 1 | 4 | 0.279 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 3 | 3 | 10 | 2 | 0.293 |
| White-winged Triller | <i>Lalage tricolor</i> | 0 | 0 | 3 | 5 | 0.300 |
| Black-faced Cuckoo-shrike | <i>Coracina novaehollandiae</i> | 0 | 0 | 3 | 0 | 0.344 |
| Hooded Robin | <i>Melanodryas cucullata</i> | 4 | 0 | 0 | 0 | 0.363 |
| Southern Boobook | <i>Ninox novaeseelandiae</i> | 0 | 4 | 0 | 0 | 0.363 |
| Striated Thornbill | <i>Acanthiza lineata</i> | 0 | 0 | 3 | 0 | 0.364 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 0 | 0 | 4 | 2 | 0.370 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 3 | 11 | 7 | 3 | 0.377 |
| Crimson Rosella | <i>Platycercus elegans</i> | 7 | 2 | 5 | 1 | 0.394 |
| Silvereye | <i>Zosterops lateralis</i> | 3 | 5 | 1 | 0 | 0.491 |
| Yellow-faced Honeyeater | <i>Lichenostomus chrysops</i> | 0 | 1 | 4 | 0 | 0.520 |
| Brown-headed Honeyeater | <i>Melithreptus brevirostris</i> | 0 | 1 | 0 | 2 | 0.612 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 0 | 2 | 0 | 0 | 0.660 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 0 | 2 | 0 | 0 | 0.669 |
| Crested Shrike-tit | <i>Falcunculus frontatus</i> | 0 | 2 | 0 | 0 | 0.671 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 1 | 0 | 2 | 3 | 0.675 |
| Jacky Winter | <i>Microeca fascinans</i> | 0 | 2 | 0 | 0 | 0.681 |
| White-browed Babbler | <i>Pomatostomus superciliosus</i> | 2 | 0 | 1 | 0 | 0.690 |
| Red-rumped Parrot | <i>Psephotus haematonotus</i> | 2 | 0 | 3 | 2 | 0.705 |
| Horsfield's Bronze-Cuckoo | <i>Chrysococcyx basalis</i> | 0 | 2 | 0 | 0 | 0.727 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 4 | 1 | 2 | 5 | 0.728 |
| Grey Currawong | <i>Strepera versicolor</i> | 2 | 2 | 0 | 0 | 0.783 |
| Peaceful Dove | <i>Geopelia striata</i> | 1 | 1 | 3 | 1 | 0.809 |
| Eastern Spinebill | <i>Acanthorhynchus tenuirostris</i> | 2 | 2 | 5 | 0 | 0.862 |
| Brown Thornbill | <i>Acanthiza pusilla</i> | 2 | 0 | 0 | 0 | 1.000 |
| Grey Fantail | <i>Rhipidura fuliginosa</i> | 1 | 1 | 0 | 0 | 1.000 |

| Common name | Species name | LGLO | HGHO | OG | MOG | p-value |
|--------------------|--------------------------------|-------------|-------------|-----------|------------|----------------|
| Red Wattlebird | <i>Anthochaera carunculata</i> | 2 | 0 | 0 | 0 | 1.000 |
| Weebill | <i>Smicrornis brevirostris</i> | 2 | 0 | 0 | 0 | 1.000 |
| Willie Wagtail | <i>Rhipidura leucophrys</i> | 2 | 0 | 0 | 0 | 1.000 |

In this study, the Striated Pardalote foraged almost exclusively on a variety of eucalypt species within the overstorey strata. The Striated Pardalote was statistically associated with a vegetation cluster within four of the 15 ten hectare sites that were surveyed. These were vegetation cluster MO within Site 3 (closed heath woodland; Table 5.8), vegetation cluster SM within Site 9 (closed heath woodland; Table 5.9), vegetation cluster OS within Site 11 (open heath woodland; Table 5.6) and vegetation cluster OS2 within Site 13 (closed heath woodland; Table 5.10). The Striated Pardalote was only ever statistically associated once with the vegetation cluster that contained the highest overstorey canopy cover for a particular ten hectare site; being Site 3 which was low overall in overstorey cover (maximum of 45%). The other vegetation clusters that the Striated Pardalote was statistically associated with ranged in overstorey cover of between 15 and 57% (refer Chapter 4, Tables 4.11, 4.29, 4.35 and 4.41).

Table 5.8: Indicator values for indicator bird species for Site 3 (closed heath woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.7) contained three statistically distinct vegetation clusters; being 1. midstorey cover and overstorey cover (MO), 2. midstorey cover (M), and 3. midstorey cover and shrubs (MS). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | MO | M | MS | <i>p</i> -value |
|---------------------------|-------------------------------------|-----------|---|----|-----------------|
| Striated Pardalote | <i>Pardalotus striatus</i> | 12 | 0 | 1 | 0.055 |
| Brown Thornbill | <i>Acanthiza pusilla</i> | 0 | 8 | 0 | 0.060 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 3 | 7 | 18 | 0.101 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 0 | 0 | 4 | 0.162 |
| Scarlet Robin | <i>Petroica boodang</i> | 4 | 0 | 0 | 0.216 |
| Striated Thornbill | <i>Acanthiza lineata</i> | 2 | 0 | 7 | 0.251 |
| Crescent Honeyeater | <i>Phylidonyris pyrrhoptera</i> | 2 | 9 | 3 | 0.315 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 2 | 2 | 2 | 0.345 |
| Grey Fantail | <i>Rhipidura fuliginosa</i> | 0 | 4 | 0 | 0.374 |
| White-browed Babbler | <i>Pomatostomus superciliosus</i> | 0 | 4 | 0 | 0.381 |
| Brush Bronzewing | <i>Phaps elegans</i> | 0 | 0 | 3 | 0.466 |
| Horsfield's Bronze-Cuckoo | <i>Chrysococcyx basalis</i> | 0 | 0 | 3 | 0.472 |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 2 | 5 | 0 | 0.497 |
| Fan-tailed Cuckoo | <i>Cacomantis flabelliformis</i> | 0 | 2 | 0 | 0.498 |
| Crimson Rosella | <i>Platycercus elegans</i> | 1 | 0 | 3 | 0.515 |
| Red-browed Finch | <i>Neochmia temporalis</i> | 3 | 0 | 0 | 0.630 |
| Grey Shrike-thrush | <i>Colluricinclla harmonica</i> | 3 | 1 | 5 | 0.694 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 3 | 1 | 0 | 0.714 |
| Silvereye | <i>Zosterops lateralis</i> | 3 | 1 | 1 | 0.755 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 6 | 2 | 6 | 0.779 |
| White-browed Scrubwren | <i>Sericornis frontalis</i> | 2 | 4 | 3 | 0.859 |
| Brown-headed Honeyeater | <i>Melithreptus brevirostris</i> | 1 | 0 | 0 | 1.000 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 1 | 1 | 0 | 1.000 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 1 | 1 | 0 | 1.000 |
| Grey Currawong | <i>Strepera versicolor</i> | 1 | 2 | 2 | 1.000 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 1 | 1 | 0 | 1.000 |
| Painted Button-quail | <i>Turnix varia</i> | 1 | 0 | 0 | 1.000 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 1 | 1 | 0 | 1.000 |

Table 5.9: Indicator values for indicator bird species for Site 9 (closed heath woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.25) contained three statistically distinct vegetation clusters; being 1. overstorey cover (O), 2. midstorey cover and shrubs (MS), and 3. shrubs and midstorey cover (SM). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | O | MS | SM | <i>p</i> -value |
|---------------------------|-------------------------------------|---|----|-----------|-----------------|
| Striated Pardalote | <i>Pardalotus striatus</i> | 0 | 2 | 10 | 0.057 |
| White-browed Scrubwren | <i>Sericornis frontalis</i> | 0 | 3 | 8 | 0.115 |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 2 | 2 | 10 | 0.125 |
| Brown-headed Honeyeater | <i>Melithreptus brevirostris</i> | 0 | 6 | 0 | 0.141 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 0 | 1 | 3 | 0.142 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 9 | 1 | 1 | 0.172 |
| Striated Thornbill | <i>Acanthiza lineata</i> | 4 | 0 | 0 | 0.204 |
| Yellow-faced Honeyeater | <i>Lichenostomus chrysops</i> | 4 | 0 | 0 | 0.210 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 7 | 1 | 1 | 0.258 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 1 | 3 | 7 | 0.287 |
| Brush Bronzewing | <i>Phaps elegans</i> | 0 | 4 | 0 | 0.291 |
| Silvereye | <i>Zosterops lateralis</i> | 6 | 1 | 1 | 0.299 |
| Crimson Rosella | <i>Platycercus elegans</i> | 3 | 0 | 0 | 0.429 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 3 | 12 | 9 | 0.434 |
| Crescent Honeyeater | <i>Phylidonyris pyrrhoptera</i> | 3 | 0 | 0 | 0.443 |
| Grey Shrike-thrush | <i>Colluricincla harmonica</i> | 1 | 5 | 0 | 0.486 |
| Yellow-rumped Thornbill | <i>Acanthiza chrysorrhoa</i> | 3 | 0 | 0 | 0.598 |
| Scarlet Robin | <i>Petroica boodang</i> | 3 | 0 | 0 | 0.601 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 3 | 0 | 0 | 0.608 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 3 | 0 | 0 | 0.609 |
| Black-faced Cuckoo-shrike | <i>Coracina novaehollandiae</i> | 3 | 0 | 0 | 0.614 |
| Grey Currawong | <i>Strepera versicolor</i> | 2 | 0 | 3 | 0.775 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 0 | 1 | 3 | 0.784 |
| Brown Thornbill | <i>Acanthiza pusilla</i> | 1 | 2 | 2 | 0.990 |
| Brown Treecreeper | <i>Climacteris picumnus</i> | 1 | 0 | 0 | 1.000 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 1 | 1 | 0 | 1.000 |
| Eastern Spinebill | <i>Acanthorhynchus tenuirostris</i> | 0 | 1 | 0 | 1.000 |
| Fairy Martin | <i>Petrochelidon ariel</i> | 1 | 0 | 0 | 1.000 |
| Horsfield's Bronze-Cuckoo | <i>Chrysococcyx basalis</i> | 0 | 2 | 2 | 1.000 |
| Little Raven | <i>Corvus mellori</i> | 1 | 0 | 0 | 1.000 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 2 | 0 | 0 | 1.000 |
| Australian Owlet-nightjar | <i>Aegotheles cristatus</i> | 0 | 1 | 0 | 1.000 |
| Red-browed Finch | <i>Neochmia temporalis</i> | 0 | 1 | 0 | 1.000 |
| Southern Boobook | <i>Ninox novaeseelandiae</i> | 1 | 0 | 0 | 1.000 |
| Square-tailed Kite | <i>Lophoictinia isura</i> | 1 | 1 | 0 | 1.000 |

| Common name | Species name | O | MS | SM | p-value |
|----------------------|----------------------------------|---|----|----|---------|
| Varied Sittella | <i>Daphoenositta chrysoptera</i> | 0 | 1 | 1 | 1.000 |
| White-winged Chough | <i>Corcorax melanorhamphos</i> | 1 | 0 | 0 | 1.000 |
| White-winged Triller | <i>Lalage tricolor</i> | 0 | 1 | 0 | 1.000 |

Table 5.10: Indicator values for indicator bird species for Site 13 (closed heath woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.37) contained six statistically distinct vegetation clusters; being 1. high overstorey cover (HO), 2. high overstorey cover and shrubs (HOS), 3. shrubs and overstorey cover (SO), 4. low overstorey cover and shrubs (LOS), 5. low overstorey cover (LO), and 6. midstorey cover (M). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant p-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | HO | HOS | SO | LOS | LO | M | p-value |
|----------------------------|-------------------------------------|----|-----|----|-----------|----|----|--------------|
| Striated Pardalote | <i>Pardalotus striatus</i> | 0 | 1 | 0 | 26 | 0 | 0 | 0.013 |
| Crimson Rosella | <i>Platycercus elegans</i> | 0 | 2 | 10 | 0 | 0 | 0 | 0.079 |
| European Goldfinch | <i>Carduelis carduelis</i> | 0 | 0 | 17 | 0 | 0 | 0 | 0.102 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0 | 1 | 0 | 0 | 0 | 0 | 0.129 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 0 | 0 | 0 | 0 | 1 | 0 | 0.149 |
| Grey Currawong | <i>Strepera versicolor</i> | 1 | 1 | 0 | 0 | 0 | 9 | 0.164 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 1 | 1 | 0 | 0 | 0 | 10 | 0.193 |
| Silvereye | <i>Zosterops lateralis</i> | 0 | 0 | 9 | 1 | 2 | 0 | 0.233 |
| Red-capped Robin | <i>Petroica goodenovii</i> | 4 | 0 | 0 | 0 | 0 | 0 | 0.300 |
| Spotted Pardalote | <i>Pardalotus punctatus</i> | 4 | 0 | 0 | 0 | 0 | 0 | 0.303 |
| Eastern Spinebill | <i>Acanthorhynchus tenuirostris</i> | 1 | 0 | 0 | 0 | 0 | 8 | 0.337 |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 9 | 3 | 10 | 1 | 0 | 0 | 0.364 |
| White-throated Treecreeper | <i>Cormobates leucophaeus</i> | 0 | 6 | 0 | 0 | 0 | 0 | 0.399 |
| Brown Thornbill | <i>Acanthiza pusilla</i> | 4 | 0 | 2 | 1 | 7 | 10 | 0.404 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 0 | 1 | 0 | 0 | 0 | 6 | 0.428 |
| Striated Thornbill | <i>Acanthiza lineata</i> | 11 | 3 | 0 | 10 | 4 | 0 | 0.444 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 6 | 0 | 0 | 0 | 6 | 0 | 0.447 |
| Grey Shrike-thrush | <i>Colluricinclla harmonica</i> | 0 | 1 | 6 | 1 | 1 | 0 | 0.485 |
| White-browed Scrubwren | <i>Sericornis frontalis</i> | 1 | 1 | 0 | 1 | 2 | 5 | 0.729 |
| Black-faced Cuckoo-shrike | <i>Coracina novaehollandiae</i> | 0 | 0 | 0 | 4 | 0 | 0 | 0.735 |
| Yellow-faced Honeyeater | <i>Lichenostomus chrysops</i> | 2 | 1 | 3 | 2 | 0 | 0 | 0.742 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 0 | 2 | 0 | 1 | 0 | 0 | 0.899 |
| Crescent Honeyeater | <i>Phylidonyris pyrrhoptera</i> | 4 | 3 | 4 | 1 | 4 | 0 | 0.923 |
| Tawny Frogmouth | <i>Podargus strigoides</i> | 0 | 3 | 0 | 0 | 0 | 0 | 1.000 |

The Buff-rumped Thornbill (*A. reguloides*) was regularly detected within comparably more open habitats, relative to the habitat structure within those sites. These more open habitats however also contained vegetation across all strata levels. In this study, Buff-rumped Thornbills used numerous plant species as foraging substrates at all storey heights (Table 5.11). In Site 1, the Buff-rumped Thornbill was statistically associated with areas containing the lowest overstorey cover, but with higher midstorey and weed cover (cluster GMW, Table 5.5). Site 1 contained infestations of exotic Boneseed (*Chrysanthemoides monilifera*) which acted as a shrub layer. A pair of Buff-rumped Thornbills within this vegetation cluster raised chicks within a hollow of a South Australian Blue Gum (refer Appendix B).

In Site 2 (closed heath woodland) the Buff-rumped Thornbill was again statistically associated with areas that contained vegetation at all strata levels (vegetation cluster OSM, Table 5.12). Buff-rumped Thornbills again used numerous plant species as foraging substrates across all storey heights (Table 5.11). Cluster OSM contained only nine, 25 m × 25 m survey squares, compared to the structurally similar MSO vegetation cluster within this same ten hectare site, which contained 38, 25 m × 25 m survey squares. The difference between these two clusters was that cluster MSO contained 30% more vegetation cover at the midstorey level (Chapter 4, Table 4.8). Cluster MSO also possessed more habitat clogging plant species, dominated by Yacca (*Xanthorrhoea semiplana*), Slaty Sheoak (*Allocasuarina muelleriana*), Sticky Hopbush (*Dodonaea viscosa*) and Broombush (*Melaleuca uncinata*; Table 5.13).

Table 5.11: Plant species used for foraging by the Buff-rumped Thornbill (*Acanthiza reguloides*) within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. These data indicate this bird species used numerous plant species as a foraging substrate across all strata levels. Sites 1 and 2 covered ten hectares each and were both surveyed a total of five times. Site 1 was classified as grassy woodland and Site 2 classified as closed heath woodland. Data collected September – October 2008.

| Common name | Species name | Site |
|------------------------------|-------------------------------------|------|
| Shrubs (< 1 m) | | |
| Boneseed | <i>Chrysanthemoides monilifera</i> | 1, 2 |
| Erect Guinea-flower | <i>Hibbertia riparia</i> | 2 |
| Flame Heath | <i>Astroloma conostephioides</i> | 2 |
| Green Rock-fern | <i>Cheilanthes austrotenuifolia</i> | 2 |
| Spiky Guinea-flower | <i>Hibbertia exutiacies</i> | 2 |
| Yacca | <i>Xanthorrhoea semiplana</i> | 2 |
| Midstorey (1 – 4 m) | | |
| Broombush | <i>Melaleuca uncinata</i> | 2 |
| Christmas Bush | <i>Bursaria spinosa</i> | 2 |
| Golden Wattle | <i>Acacia pycnantha</i> | 1, 2 |
| Hakea | <i>Hakea carinata</i> | 2 |
| Kangaroo Thorn | <i>Acacia paradoxa</i> | 2 |
| Mount Lofty Grass-tree | <i>Xanthorrhoea quadrangulata</i> | 2 |
| Olive | <i>Olea europaea</i> | 2 |
| Slaty Sheoak | <i>Allocasuarina muelleriana</i> | 2 |
| Sticky Cassinia | <i>Cassinia uncata</i> | 2 |
| Sticky Hopbush | <i>Dodonaea viscosa</i> | 2 |
| Strangle Vine | <i>Cassytha melantha</i> | 2 |
| Overstorey (> 4 m) | | |
| Box Mistletoe | <i>Amyema miquelii</i> | 2 |
| Drooping Sheoak | <i>Allocasuarina verticillata</i> | 2 |
| Harlequin Mistletoe | <i>Lysiana exocarpi</i> | 2 |
| Long-leaved Box | <i>Eucalyptus goniocalyx</i> | 2 |
| Peppermint Box | <i>Eucalyptus odorata</i> | 2 |
| Pink Gum | <i>Eucalyptus fasciculosa</i> | 2 |
| Slender Cypress-pine | <i>Callitris gracilis</i> | 1, 2 |
| South Australian Blue Gum | <i>Eucalyptus leucoxylon</i> | 1, 2 |

Table 5.12: Indicator values for indicator bird species for Site 2 (closed heath woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.4) contained five statistically distinct vegetation clusters; being 1. overstorey cover, shrubs and midstorey cover (OSM), 2. midstorey cover, shrubs and overstorey cover (MSO), 3. overstorey cover and shrubs (OS), 4. shrubs and overstorey cover (SO), and 5. shrubs (S). The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | OSM | MSO | OS | SO | S | <i>p</i> -value |
|---------------------------|-------------------------------------|-----------|-----|----|-----------|----|-----------------|
| Crimson Rosella | <i>Platycercus elegans</i> | 0 | 0 | 0 | 12 | 0 | 0.044 |
| Buff-rumped Thornbill | <i>Acanthiza reguloides</i> | 13 | 1 | 3 | 0 | 1 | 0.059 |
| Grey Currawong | <i>Strepera versicolor</i> | 0 | 0 | 3 | 0 | 10 | 0.133 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 0 | 1 | 10 | 2 | 0 | 0.164 |
| Horsfield's Bronze-Cuckoo | <i>Chrysococcyx basalis</i> | 7 | 0 | 1 | 0 | 0 | 0.185 |
| Crescent Honeyeater | <i>Phylidonyris pyrrhoptera</i> | 2 | 0 | 1 | 3 | 8 | 0.215 |
| Eastern Spinebill | <i>Acanthorhynchus tenuirostris</i> | 6 | 0 | 0 | 2 | 0 | 0.240 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 0 | 9 | 4 | 3 | 0 | 0.303 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 6 | 1 | 0 | 0 | 2 | 0.325 |
| Striated Thornbill | <i>Acanthiza lineata</i> | 0 | 0 | 6 | 2 | 1 | 0.351 |
| Brown Thornbill | <i>Acanthiza pusilla</i> | 5 | 1 | 1 | 0 | 1 | 0.354 |
| Australian Owlet-nightjar | <i>Aegotheles cristatus</i> | 0 | 3 | 0 | 0 | 0 | 0.441 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 0 | 0 | 0 | 5 | 0 | 0.447 |
| Silvereye | <i>Zosterops lateralis</i> | 4 | 1 | 0 | 0 | 0 | 0.480 |
| Superb Fairy-wren | <i>Malurus cyaneus</i> | 7 | 4 | 1 | 2 | 2 | 0.495 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 0 | 2 | 1 | 1 | 5 | 0.544 |
| Golden Whistler | <i>Pachycephala pectoralis</i> | 0 | 3 | 2 | 0 | 0 | 0.577 |
| Peaceful Dove | <i>Geopelia striata</i> | 0 | 1 | 4 | 0 | 0 | 0.590 |
| Fan-tailed Cuckoo | <i>Cacomantis flabelliformis</i> | 0 | 1 | 1 | 0 | 0 | 0.835 |
| Grey Shrike-thrush | <i>Colluricinclla harmonica</i> | 0 | 2 | 0 | 2 | 3 | 0.836 |
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0 | 1 | 1 | 0 | 0 | 0.840 |
| Varied Sittella | <i>Daphoenositta chrysoptera</i> | 0 | 1 | 1 | 0 | 0 | 0.845 |
| Grey Fantail | <i>Rhipidura fuliginosa</i> | 0 | 1 | 3 | 0 | 0 | 0.873 |
| Brown-headed Honeyeater | <i>Melithreptus brevirostris</i> | 0 | 0 | 2 | 0 | 0 | 1.000 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 0 | 0 | 0 | 2 | 0 | 1.000 |
| Scarlet Robin | <i>Petroica boodang</i> | 0 | 0 | 1 | 1 | 0 | 1.000 |
| Weebill | <i>Smicromis brevirostris</i> | 0 | 0 | 0 | 2 | 0 | 1.000 |
| Yellow-rumped Thornbill | <i>Acanthiza chrysorrhoa</i> | 0 | 0 | 0 | 2 | 0 | 1.000 |

Table 5.13: Percent cover of plant species below four metres within Site 2 (closed heath woodland). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m. The percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured for each 25 m × 25 m survey square. The dendrogram based on habitat structure (refer Chapter 4, Fig. 4.4) included two statistically distinct vegetation clusters that were separated by differences in the percent cover over three storey levels. These were 1. overstorey cover, shrubs and midstorey cover (OSM) and 2. midstorey cover, shrubs and overstorey cover (MSO). The vegetation clusters are labelled to indicate the descending order of cover for the dominant habitat attributes. Vegetation cluster OSM contained 49.3% overstorey cover, 40.4% midstorey cover and 47.5% shrub cover. Vegetation cluster MSO contained 40.1% overstorey cover, 70.8% midstorey cover and 45.4% shrub cover. The Buff-rumped Thornbill (*Acanthiza reguloides*) was statistically associated with vegetation cluster OSM, which contained lower cover of the plant species which can physically clog habitats (in bold). Values are percent cover. Data collected September – October 2008.

| Common name | Species name | OSM | MSO |
|----------------------------|---|-----|-------------|
| Shrubs (< 1 m) | | | |
| Common Fringe-myrtle | <i>Calytrix tetragona</i> | 3.2 | 2.2 |
| Downy Mintbush | <i>Prostanthera behriana</i> | 0.0 | 0.1 |
| Dusty Miller | <i>Spyridium parvifolium</i> | 1.4 | 2.4 |
| Erect Guinea-flower | <i>Hibbertia riparia</i> | 5.3 | 5.6 |
| Flame Heath | <i>Astroloma conostephoides</i> | 2.6 | 6.7 |
| Heath Tea-tree | <i>Leptospermum myrsinoides</i> | 0.0 | 0.2 |
| Lavender Grevillea | <i>Grevillea lavandulacea</i> | 0.0 | 0.2 |
| Leafless Bitter-pea | <i>Daviesia brevifolia</i> | 0.0 | 0.2 |
| Narrow-leaf Wax-flower | <i>Philotheeca angustifolia</i> | 0.0 | 0.1 |
| Native Current | <i>Acrotriche depressa</i> | 2.1 | 2.0 |
| Showy Parrot Pea | <i>Dillwynia sericea</i> | 0.8 | 0.5 |
| Silky Guinea-flower | <i>Hibbertia sericea</i> | 1.1 | 0.4 |
| Spiky Guinea-flower | <i>Hibbertia exutiacies</i> | 3.3 | 5.2 |
| Twiggy Guinea-flower | <i>Hibbertia virgata</i> | 0.0 | 0.7 |
| Yacca | <i>Xanthorrhoea semiplana</i> | 0.4 | 9.0 |
| Midstorey (1 – 4 m) | | | |
| Common Fringe-myrtle | <i>Calytrix tetragona</i> | 1.7 | 2.2 |
| Beaked Hakea | <i>Hakea rostrata</i> | 1.7 | 1.6 |
| Broombush | <i>Melaleuca uncinata</i> | 0.6 | 6.1 |
| Christmas Bush | <i>Bursaria spinosa</i> | 5.1 | 2.2 |
| Downy Mintbush | <i>Prostanthera behriana</i> | 2.2 | 2.2 |
| Dusty Miller | <i>Spyridium parvifolium</i> | 2.4 | 2.7 |
| Golden Wattle | <i>Acacia pycnantha</i> | 3.1 | 5.0 |
| Hakea | <i>Hakea carinata</i> | 1.6 | 1.4 |
| Kangaroo Thorn | <i>Acacia paradoxa</i> | 0.4 | 1.1 |
| Round-leaved Wattle | <i>Acacia acinacea</i> | 0.0 | 1.4 |
| Slaty Sheoak | <i>Allocasuarina muelleriana</i> | 4.4 | 15.9 |
| Slender Cypress-pine | <i>Callitris gracilis</i> | 2.4 | 4.2 |
| Sticky Cassinia | <i>Cassinia uncata</i> | 3.6 | 0.0 |
| Sticky Hopbush | <i>Dodonaea viscosa</i> | 0.2 | 8.2 |

Other bird species were also statistically associated with more open habitats that contrasted with the remaining habitat within a ten hectare site. The Common Bronzewing (*Phaps chalcoptera*) was statistically associated with vegetation cluster O within Site 1 (Table 5.5). In Site 1, cluster O contained low grass cover compared with the average across this

grassy ten hectare site (Chapter 4, Table 4.5). The Common Bronzewings that were detected within this site were regularly observed to be foraging on the ground.

Some bird species were statistically associated with habitat elements that provided a strong contrast with that of the remaining habitat within a ten hectare site. In Site 6 (grassy woodland), the Crested Pigeon (*Ocyphaps lophotes*), Dusky Woodswallow (*Artamus cyanopterus*) and Yellow-rumped Thornbill (*A. chrysorrhoa*) were all statistically associated with habitat which aligned as an outlier within the habitat dendrogram for this site (Table 5.14). This outlier contained seven 25 m × 25 m survey squares which all had very low percent covers of the habitat variables that were measured (Chapter 4, Fig. 4.16). These seven survey squares corresponded with an abandoned but un-rehabilitated sand mine. The confounding variable here was that this site regularly held surface water which various bird species used for drinking and bathing.

Table 5.14: Indicator values for indicator bird species for Site 6 (grassy woodland) calculated from Indicator Species Analysis in PC-ORD5 (McCune and Mefford 1999b). The entire area within this ten hectare site was assessed at a scale of 25 m × 25 m, within which the percent cover of grasses, weeds, shrubs, midstorey and overstorey vegetation were measured. The resultant dendrogram based on habitat structure (refer Chapter 4, Fig. 4.16) contained four statistically distinct vegetation clusters; being 1. grasses (G), 2. overstorey cover and grasses (OG), 3. midstorey cover, grasses and overstorey cover (MGO), and 4. outlier 1. The vegetation clusters are labelled to indicate the descending order of the percent covers for the most dominant habitat attributes. Indicator Species Analysis was used to identify bird species that associated with a particular vegetation cluster within this site. This analysis first calculated the relative abundance of a bird species in a particular cluster relative to its abundance in all clusters within this site. It then calculated the relative frequency of that bird species in each cluster within this site. Multiplying these two indicator terms together yields the indicator value. Indicator values range from zero (no indication) to 100 (perfect indication). The strength of this procedure is that both indicator terms need to be high in order to produce a high indicator value. Testing the null hypothesis that the highest indicator value for a species is no greater than would be expected by chance (i.e. the species has no indicator value), the indicator value for each species was statistically tested using a Monte Carlo test based on 1,000 randomisations. Statistically significant *p*-values ($\alpha \leq 0.05$ prior to rounding to two decimal places) identified those bird species whose indicator value for a particular vegetation cluster was larger than expected by chance (species and figures in bold). Bird species not in bold either showed no pattern to their distribution with the vegetation clusters and/or were not detected sufficiently to permit valid analysis. Data collected September – October 2008.

| Common name | Species name | G | OG | MGO | Outlier 1 | <i>p</i> -value |
|-------------------------|-----------------------------------|---|----|-----|-----------|-----------------|
| Crested Pigeon | <i>Ocyphaps lophotes</i> | 1 | 0 | 0 | 18 | 0.014 |
| Dusky Woodswallow | <i>Artamus cyanopterus</i> | 0 | 1 | 0 | 16 | 0.017 |
| Yellow-rumped Thornbill | <i>Acanthiza chrysorrhoa</i> | 0 | 0 | 0 | 10 | 0.046 |
| Noisy Miner | <i>Manorina melanocephala</i> | 0 | 1 | 1 | 7 | 0.130 |
| Red Wattlebird | <i>Anthochaera carunculata</i> | 1 | 1 | 7 | 0 | 0.185 |
| White-winged Triller | <i>Lalage tricolor</i> | 0 | 0 | 6 | 0 | 0.216 |
| Grey Shrike-thrush | <i>Colluricincla harmonica</i> | 0 | 1 | 1 | 6 | 0.233 |
| Musk Lorikeet | <i>Glossopsitta concinna</i> | 1 | 6 | 0 | 0 | 0.242 |
| Willie Wagtail | <i>Rhipidura leucophrys</i> | 1 | 0 | 1 | 6 | 0.292 |
| White-browed Babbler | <i>Pomatostomus superciliosus</i> | 7 | 1 | 3 | 0 | 0.296 |

| Common name | Species name | G | OG | MGO | Outlier 1 | p-value |
|---------------------------|-------------------------------------|---|----|-----|-----------|---------|
| Common Bronzewing | <i>Phaps chalcoptera</i> | 0 | 1 | 5 | 0 | 0.310 |
| Crimson Rosella | <i>Platycercus elegans</i> | 4 | 8 | 0 | 0 | 0.313 |
| House Sparrow | <i>Passer domesticus</i> | 0 | 0 | 3 | 0 | 0.356 |
| Jacky Winter | <i>Microeca fascinans</i> | 0 | 0 | 3 | 0 | 0.359 |
| Peaceful Dove | <i>Geopelia striata</i> | 1 | 1 | 0 | 5 | 0.363 |
| White-plumed Honeyeater | <i>Lichenostomus penicillatus</i> | 0 | 6 | 3 | 3 | 0.474 |
| Hooded Robin | <i>Melanodryas cucullata</i> | 2 | 0 | 0 | 0 | 0.508 |
| Brown Goshawk | <i>Accipiter fasciatus</i> | 2 | 0 | 0 | 0 | 0.511 |
| Sulphur-crested Cockatoo | <i>Cacatua galerita</i> | 2 | 0 | 0 | 0 | 0.517 |
| Galah | <i>Cacatua roseicapilla</i> | 4 | 1 | 0 | 0 | 0.552 |
| Striated Pardalote | <i>Pardalotus striatus</i> | 5 | 2 | 3 | 0 | 0.559 |
| Common Blackbird | <i>Turdus merula</i> | 0 | 1 | 2 | 0 | 0.586 |
| Mistletoebird | <i>Dicaeum hirundinaceum</i> | 0 | 2 | 4 | 0 | 0.684 |
| Rainbow Bee-eater | <i>Merops ornatus</i> | 0 | 2 | 0 | 0 | 0.689 |
| Black-faced Cuckoo-shrike | <i>Coracina novaehollandiae</i> | 0 | 2 | 0 | 0 | 0.689 |
| Red-rumped Parrot | <i>Psephotus haematonotus</i> | 0 | 3 | 1 | 4 | 0.695 |
| Brown Treecreeper | <i>Climacteris picumnus</i> | 0 | 3 | 2 | 0 | 0.752 |
| Diamond Firetail | <i>Stagonopleura guttata</i> | 4 | 1 | 3 | 0 | 0.760 |
| Grey Fantail | <i>Rhipidura fuliginosa</i> | 0 | 2 | 2 | 0 | 0.774 |
| Laughing Kookaburra | <i>Dacelo novaeguineae</i> | 0 | 2 | 2 | 0 | 0.777 |
| New Holland Honeyeater | <i>Phylidonyris novaehollandiae</i> | 7 | 6 | 3 | 1 | 0.784 |
| Rufous Whistler | <i>Pachycephala rufiventris</i> | 3 | 1 | 2 | 4 | 0.814 |
| Australian Magpie | <i>Gymnorhina tibicen</i> | 2 | 2 | 0 | 0 | 0.841 |
| Yellow Thornbill | <i>Acanthiza nana</i> | 1 | 2 | 0 | 0 | 0.883 |
| Brown Falcon | <i>Falco berigora</i> | 1 | 1 | 0 | 0 | 1.000 |
| European Goldfinch | <i>Carduelis carduelis</i> | 0 | 1 | 0 | 0 | 1.000 |
| Little Raven | <i>Corvus mellori</i> | 0 | 1 | 0 | 0 | 1.000 |
| Spotted Pardalote | <i>Pardalotus punctatus</i> | 0 | 1 | 0 | 0 | 1.000 |
| Common Starling | <i>Sturnus vulgaris</i> | 0 | 1 | 0 | 0 | 1.000 |

Discussion

At least one bird species was statistically associated with a vegetation cluster across many of the ten hectare sites. This is a significant result considering that the density estimates for these sites were around $6 - 7$ birds ha^{-1} (see Chapter 3). At these densities, for ISA to associate a bird species with a vegetation cluster, that species needed to be reliably found within $25 \text{ m} \times 25 \text{ m}$ survey squares that were associated with that particular vegetation cluster type. The key finding of this chapter is that different habitat clusters within a site are used by different bird species, and so it makes sense to provide different habitat clusters within revegetation. Whether the scale of this heterogeneity should be at the scale of $25 \text{ m} \times 25 \text{ m}$, or 1 ha, or larger has not been determined, but using the measured habitat heterogeneity at the scale of $25 \text{ m} \times 25 \text{ m}$ may be a good starting point for planting programs.

Importantly, this heterogeneity must cater for multiple bird species. This requires that an array of discrete habitat patches should be provided, with each patch “type” catering for the specific habitat needs of one or a few bird species. For example, by providing flowering plants at different storey heights and over different areas, enables habitat use to be proportioned between some nectarivorous bird species. The Red Wattlebird competitively excludes most other honeyeaters (and other small birds) from rich nectar resources, especially within eucalypts (Ford 1981; Ford and Paton 1982; Davis and Recher 1993). The canopy foraging Musk Lorikeet is also an antagonistic species (Hutchins *et al.* 1985), which helps explain how these two bird species manage to co-forage in this study. The aggressive behaviour of bird species such as these also helps to explain the apparent habitat partitioning between these species and the smaller New Holland Honeyeater. Such inter-species interactions may also help drive the New Holland Honeyeater’s preference for habitats that contain a dense shrub layer (Recher 1971; Morris and Burton 1989).

Foraging resources at various storey heights are also used by the Yellow-faced Honeyeater (Paton and Ford 1977; Keast 1985), which was also apparent in this study. This further highlights the need for an array of floristic resources to be available for nectarivorous bird species. This would provide foraging opportunities at different storey levels and as this study has shown, these resources should be distributed patchily across an area.

Several bird species responded positively to spatial habitat heterogeneity. The Superb Fairy-wren is typically found within habitats that contain a mosaic of small clearings that are interspersed with low dense vegetation (Bell 1980b; Nias 1984; Rowley and Russell 1990). The importance of this type of habitat structure was evident in this study, with the Superb Fairy-wren using Kangaroo Thorn to occupy in an otherwise inhospitable grazed woodland habitat.

Conversely, the White-winged Chough occupied habitat areas that contained minimal vegetation cover at low storey levels. This bird species typically avoids habitats that contain a dense shrub layer (Loyn 1985; Gosper 1992; Traill *et al.* 1996). The absence of a shrub layer presumably enables unhindered access to leaf litter, which is this bird species favoured foraging substrate in which they glean and probe for invertebrate prey (Ford *et al.* 1986; Antos and Bennett 2006; Antos *et al.* 2008).

Areas of more open habitat also seemed to be appreciated by the Red-browed Finch. The availability of foraging resources also appeared to be secondary to habitat structure. The Red-browed Finch is often associated with clearings within woodlands and along habitat edges (Recher 1975; Lainie 2001; Kavanagh and Stanton 2003). The importance of an accessible ground layer for this bird species has also been demonstrated by the recorded decrease of this species in areas where grass cover has increased (Stagoll *et al.* 2010).

Similarly, the granivorous Common Bronzewing prefers more open areas that contain no more than an open understorey of shrubs and more bare ground than grass cover (Pawsey 1966; Nichols 1984; Fulton *et al.* 2008; Kutt and Martin 2010). The negative influence high grass cover imposes on habitat use by the Common Bronzewing was also inferred by this bird species in Site 1. Strong evidence exists that high exotic grass cover can impede habitat use by many woodland bird species that use the ground to forage (Maron and Lill 2005; Ford *et al.* 2009). The impost of exotic grasses is also considered to be a contributing factor behind the decline of many woodland birds across southern Australia (Ford *et al.* 2001).

A number of other bird species responded to vertical habitat complexity. In this study, the Rufous Whistler was found in habitats that contained various storey levels of open vegetation. This is a common observation for this bird species (Johnson and Mighell 1999; Recher and Holmes 2000). As such, multiple layers of vegetation (i.e. multiple storey levels) is likely to be a minimum habitat requirement that is needed to support both male and female Rufous Whistlers. Conversely, in this study the Golden Whistler was associated with more closed habitats. This contrasting pattern in habitat associations between these ecologically similar bird species was also evident in Chapter 3. These results are consistent with what is already known about the habitat preferences for both species (e.g. Johnson and Mighell 1999; Lindenmayer *et al.* 2002). These species are able to coexist in areas that offer suitable habitat for each, such as habitat transition zones (e.g. Loyn 1985). However this current study showed that these two Whistler species also respond to habitat heterogeneity at much smaller spatial scales.

Similarly, the Silvereye in this study occupied habitat areas that contained a significant shrub layer. This bird species occurs in most types of woodlands, especially those that contain a shrubby understorey (Egan *et al.* 1997; Davis and Recher 2002; Arnold 2003;

Tzaros 2005). Significant vertical habitat structure seems to be important for the Silvreye, as this bird species is a flexible forager of fruits, invertebrates and nectar across all storey levels, including the ground (Tullis *et al.* 1982; Recher and Holmes 1985; Ford *et al.* 1986; Wheal 1996). In agreement, foraging across the full spectrum of storey heights by this species was evident in this study.

The Buff-rumped Thornbill also occupied habitats that contained various storey levels with open vegetation. This observation is also well supported within the literature (Leach 1988; Reilly 1991b; Traill *et al.* 1996; Possingham and Possingham 1997). Such habitat settings allow this bird species to use the ground and all storey layers in which to forage (Leach and Hines 1987; Wood 1995).

The Striated Pardalote occurs mainly in eucalypt habitats (Leach 1988; Gosper 1992; Chan 1995). However, this bird species is usually recorded within more open habitats than closed habitats (Woinarski *et al.* 1989; Gosper 1992; Chan 1995; Slater 1995). In agreement, within the Para Woodland region the Striated Pardalote associated with patches containing more open eucalypt overstorey compared to the immediately surrounding habitat. The habitat preference of this bird species for more open canopy structures may be influenced by several factors. Sites with lower canopy cover may contain fewer but more mature trees and thus contain many more hollows. This bird species uses hollows in which to breed (Woinarski and Rounsevell 1983). Fewer trees but with more open canopy structures would also aid the foraging technique of this bird species. From a perched position on a twig, these birds glean lerps and other foods from leaf surfaces (Woinarski 1985). This requires clear visibility and relatively unimpeded access to leaves, which a more open canopy with less dense foliage would provide.

One bird species in this study responded to a physical component of the habitat. The Scarlet Robin is often associated with habitats containing open areas at lower storey levels (Coventry 1989; Robinson 1990; Er 1997; Possingham and Possingham 1997). In this current study however, the Scarlet Robin associated with a vegetation cluster that contained the highest cover of low vegetation within that ten hectare site. The reason for this apparently contradictory habitat association was that the habitat response had more to do with the plant species that were present than the foliage cover within the midstorey strata. Mature Golden Wattle provided low perching substrates which this species used to

forage from. Low, vertical perching substrates are a critical habitat requirement for this (and several other) bird species (Ford *et al.* 1986; Cale 1994; Recher and Davis 1998). One bird species in this study displayed a greater response to a particular plant species. The Yellow Thornbill was a usual suspect found foraging within Slender Cypress-pine during this study. In the ten hectare site where this bird species was associated with a vegetation cluster, that cluster contained more than twice the number of Slender Cypress-pine than the other two clusters within that site. The Yellow Thornbill is often associated with vegetation that possesses bifurcated (or “feathery”) foliage (Pizzi and Knight 2003) and the birds tend to concentrate their foraging towards species of *Acacia*, *Allocasuarina* and *Callitris* (Blakers *et al.* 1984). The probable reason why the Yellow Thornbill preferentially target plant species such as the Slender Cypress-pine has to do with the availability of invertebrate prey. Plant species with short and dense foliage (like *Callitris* spp and *Exocarpus* spp) provide a greater range of micro-habitats than plant species with more open growing foliage (e.g. *Eucalyptus* spp). This can equate to the former plant species hosting a significantly higher arthropod biomass, which birds such as the Yellow Thornbill forage upon (Watson *et al.* 2011).

When breeding, many insectivorous birds return to their dependent young carrying one prey item at a time. For optimal foraging return and energy efficiency, this requires nest sites to be located close to preferred foraging habitats. In this way, parenting birds return to a centrally located nest (i.e. central place foraging; see Stephens and Krebs 1986). Central place foraging by breeding birds has been widely reported within the literature (e.g. Bryant and Turner 1982; Hegner 1982; Kacelnik 1984; Kacelnik and Cuthill 1990; Naef-Daenzer 2000). Anecdotal evidence for preferentially selecting optimal breeding habitat was observed by a pair of colour-banded Red-caped Robins (*Petroica goodenovii*) by Northeast (2007). This pair had constructed a nest directly adjacent to an area that had previously been identified as a hot-spot for foraging within this pair’s home range during the preceding winter. This hot-spot coincided with a high percent of bare ground and fallen timber and with a low percent of grass cover.

It makes intuitive sense that birds should select habitat areas that compliment both their nesting requirements and foraging style. Habitat elements sought by breeding birds also help to galvanise the importance of those attributes for individual species. My observations on the birds using the 15 ten hectare sites near Para Woodland provide further

evidence of breeding birds preferentially selecting areas that suit their individual habitat needs. For example, the White-winged Chough with high litter cover and lack of low vegetation, the Buff-rumped Thornbill with multiple open levels of vegetation, and to some extent the Scarlet Robin with abundant low, vertical perching substrates. This level of fine-scale information is required to help produce functional revegetated habitats for targeted bird species.

In addition to showing that some birds used specific vegetation clusters within the ten hectare sites, there were also others that showed no statistically significant patterns, inferring that they used a diversity of vegetation clusters. Such a pattern is consistent with the conclusions drawn in Chapter 3, that some bird species are able to use a variety of habitats. These species seemingly have different habitat needs from the theoretical consideration of niche theory (see Vandermeer 1972). However there may still be patterns present that have not been detected, due to the sampling effort being modest.

In summary, the finding that some bird species within the ten hectare sites were closely associated with only one of several vegetation clusters, reinforces the conclusion made in Chapter 3 that it is not sufficient to merely plant an “open grassy woodland” habitat and assume that bird species typical of such environments are capable of successfully inhabiting that area. This highlights the need to incorporate the habitat requirements of individual bird species into the design of any new revegetation. Unfortunately though, the specific habitat needs of species at fine-scales (e.g. at the patch scale) are poorly known and/or understood, especially for species of conservation concern (Lindenmayer *et al.* 2002; Paton *et al.* 2004).

The next three chapters investigate habitat use by individuals of the nine declining woodland species within the north-central zone of the southern Mount Lofty Ranges (the bird species are identified in Chapter 1). Knowledge of the different habitat types that different bird species use improves the guidelines for revegetation, but we also need to know how much habitat area they require. To determine the minimum home ranges for individual birds of these species, I determine their area use by tracking individual birds over a number of days and report the findings in Chapter 6. Chapter 7 investigates the variable nature of habitat use by these birds within the confines of their individual home ranges, adding support to the findings of this chapter. An understanding of the variable

habitat needs of birds helps promote the effective return of revegetated habitat for those birds. Chapter 8 then identifies the habitat features that were associated with high use areas within the home ranges of birds that belonged to nine declining woodland species, to further define the habitat needs of those birds.

Chapter 6

Area requirements of woodland birds in the Mount Lofty region

Introduction

The Para Woodland Reserve is a 321 ha retired farm that has been set aside for restoration. The restoration goal is to re-establish a woodland habitat that supports a suite of declining woodland bird species. Fortunately, a range of woodland bird species still exist within nearby remnant vegetation (and within a 15 km radius of this Reserve). These birds however show some preferences for different woodland types (Chapter 3), suggesting that some species will be more likely to colonise certain woodland systems than others. The remnant woodlands within this region also do not consist of homogeneous vegetation but instead display considerable spatial heterogeneity (Chapter 4). A number of bird species that occupy these remnants appear to select patches of habitat that contain certain habitat features within these various habitat types (Chapter 5). Thus the variety of bird species that will ultimately use revegetated woodland will be influenced by the diversity of woodland habitats that are re-established and by the heterogeneity within each habitat type. Put simply, re-establishing a single homogenous woodland across the Para Woodland Reserve will not provide additional habitat that all woodland bird species will use.

Based on these findings, the recommendations for restoring woodlands on this Reserve (and elsewhere) would be to create a diversity of woodland types, with considerable habitat heterogeneity within each. A simple rule would be to vary the density or cover of the different vegetation layers within the woodland at small spatial scales (e.g. 25 m × 25 m) and to also vary the key plant species providing those structural features. No revegetation program takes such an approach of deliberately varying the densities and ultimately the cover of plants at this spatial scale.

An equally important factor in re-establishing habitat for woodland birds is an understanding of the spatial requirements of individual pairs and/or cohesive groups of birds. To date my work has only shown that woodland birds use different woodland

habitats and habitat features. Not only do those features need to be provided, they also need to be provided in adequate amounts to be able to support those individual pairs and/or cohesive groups. For this purpose, I will explore the short-term areal requirements for a range of woodland bird species in this chapter. Although the birds were surveyed intensively throughout the day, pairs and cohesive groups were surveyed on average for three days each, so the estimates represent minimum area requirements only. More survey days and/or by surveying at different times of the year are likely to result in these estimates increasing. Although the term “home range” will be used to describe these area estimates, it should be noted that the spatial areas used by these birds will likely only form part of their total home ranges. Thus, these estimates represent the birds’ minimum home range requirements. In the subsequent chapters I will show variable habitat use by birds within individual home ranges (Chapter 7), highlight the importance of fine-scale habitat heterogeneity from the perspective of an individual pair and/or cohesive group of birds (Chapter 8), and document how this fine-scale heterogeneity helps enable individual birds from multiple species the ability to coexist within a common patch of woodland (Chapter 9).

Methods

Surveys were conducted at Para Wirra, Sandy Creek and Altona, within the north-central zone of the southern Mount Lofty Ranges (see Chapter 3 for location details). The study sites were named after their respective locations. These were, Para Wirra Site A (SA Water property), Para Wirra Site B (Para Wirra Recreation Park), Sandy Creek Sites A and B (both within the Sandy Creek Conservation Park) and the privately owned woodlands at Altona (identified here as Altona Chatterton, Altona Kies, Altona Landcare and Altona Torbreck). Site selection was based on earlier reconnaissance of these areas and observations that were made during the ten hectare surveys (Chapter 3). The individual bird species that were chosen for this study were reliably found within the areas selected.

This study used data that were intensely collected for sustained periods on individual birds almost continuously from dawn to dusk for one or more days (details to follow; also see Chapter 7). Nine declining woodland bird species (based on Paton *et al.* 2004) were studied. These species were the Brown Treecreeper (*Climacteris picumnus*), Buff-rumped

Thornbill (*Acanthiza reguloides*), Hooded Robin (*Melanodryas cucullata*), Rufous Whistler (*Pachycephala rufiventris*), White-browed Babbler (*Pomatostomus superciliosus*), White-winged Chough (*Corcorax melanorhamphos*), Willie Wagtail (*Rhipidura leucophrys*), Yellow-rumped Thornbill (*A. chrysorrhoa*) and Yellow Thornbill (*A. nana*).

This study explored the extent of the habitat areas that were used by individuals of selected species. It did not particularly matter if the same individual bird within the same cohesive group was surveyed or not, as the emphasis of this study was at the level of pairs and/or cohesive groups, not of individual birds *per se*. Having said that, there were only small numbers of these birds within the study areas, so keeping track of individuals was not difficult. Thus, there was no reason to individually mark the birds (e.g. colour-banding). From personal experience with the Hooded Robin, capturing and handling birds can adversely affect how closely you are then able to work with those birds. Surveying a colour-banded focal bird within a group of birds is also not necessarily the best option. Species such as the Brown Treecreeper, White-browed Babbler and White-winged Chough forage on the ground in relatively cohesive groups. Should the focal bird of any of these species be separated from the observer by other conspecific birds, disturbing those birds in an attempt to maintain visual contact with the focal bird will disturb the entire group. As the survey design required close contact with the birds to be maintained for long periods of time, the option not to unduly interfere with the birds by capturing and colour-banding was considered to be justified.

An assumption of this survey was that the same pairs and/or groups of birds were surveyed within the same habitat areas over multiple days. With the exception of the Rufous Whistler, the bird species surveyed tended to permanently reside within woodland habitats within this region. The Rufous Whistler is known to migrate away from much of the southern Mount Lofty Ranges during winter, as they do across south-eastern Australia in general (e.g. Ford 2011). It has already been noted in Chapter 1 that the Brown Treecreeper and White-browed Babbler hold permanent territories. There is also no evidence of large-scale movements by the other six bird species, which are generally described as being sedentary or resident within the habitats that they occupy (Blakers *et al.* 1984). The Rufous Whistler returned to these habitats to breed during the time of this survey and in so doing, actively defended nesting territories. Breeding home range

boundaries were readily defined through interactions with conspecific neighbours.

Plumage idiosyncrasies were used to identify individual birds, to ensure that the birds belonging to the same pair were being surveyed.

Only the birds of one species were surveyed within a single home range per day, with surveys conducted between dawn and dusk. The manner in which the birds moved through their habitat dictated how many survey days were required to assess their minimum home range area requirements. For example, the Yellow Thornbill moved rapidly between trees over relatively small areas, so only one survey day was considered to be necessary per minimum home range estimate. Conversely, the foraging behaviour of the White-browed Babbler was not conducive for these birds to cover their much larger minimum home range within a single day. For this and similar species (e.g. White-winged Chough), at least two or three days were required per home range.

The birds were surveyed from a distance that did not appear to unduly influence their behaviour, whilst maintaining adequate visual contact with the birds. This ranged from 10 m for the canopy foraging Yellow Thornbill to around 30 m for more flighty species that foraged predominantly on the ground (particularly the White-browed Babbler and White-winged Chough). When surveying a cohesive flock, I attempted to follow an individual bird for as long as possible, but as flock birds often intermingled it was usually impossible to identify individual un-marked birds for long periods. Instead, I typically surveyed a single focal bird at each bird-stop location (see next paragraph for the definition of a bird-stop location). The locations the focal birds used within their individual home ranges at each bird-stop location were recorded using a hand-held GPS. A single waypoint was used to mark the point of predominant behaviour by the focal bird at each bird-stop location (refer Chapter 8 for full details on the bird behaviour data that were recorded for each bird-stop location). The times the focal bird arrived and departed each bird-stop location were also recorded using a wrist watch.

A bird-stop location was defined as an individual plant that the focal bird used for more than one minute. At times the foliage of two or more plants merged (e.g. a clump of Kangaroo Thorn (*Acacia paradoxa*), various overstorey tree canopies, etc.). In these situations, if the focal bird moved directly between plants that were physically touching, it was considered to be a single bird-stop location. When the focal bird moved along the

ground, each location that bird stopped for more than one minute was considered to be a bird-stop (e.g. to forage, preen, dust-bathe, etc.). When the focal bird moved along the ground without notably stopping, waypoints were assigned every 30 m.

A total of 28 minimum home ranges were surveyed in 2009 (details to follow). This included one home range for a pair of Hooded Robins that was surveyed twice; once during fine weather and once during a storm. At least one home range for each species was then re-surveyed in 2010. This again included the same home range for a pair of Hooded Robins that was surveyed twice; once during fine weather and once during inclement weather. Also, both Yellow Thornbill home ranges at Sandy Creek were re-surveyed in 2010. One home range for a pair of Willie Wagtails was re-surveyed in 2011. The 2010 and 2011 repeated surveys were designed to investigate any differences in inter-annual habitat use (compared to 2009) within the individual home ranges of the birds. These data are presented in Chapter 7 (on variable habitat use by the birds), however the minimum home range extents from these surveys have been included in this study.

Data for each minimum home range were analysed spatially via Minimum Convex Polygon (MCP) analysis using ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006). This was done to determine the minimum home range sizes for each pair of birds and each cohesive group of birds that were studied. Minimum Convex Polygon analysis is the most frequently used method for assessing home ranges (Harris *et al.* 1990). As such, the results from this study can be directly compared with that of many other home range studies. Understanding the extent of habitat area that is required by pairs and cohesive groups of a species is of fundamental importance when planning restoration activities to return useful amounts of habitats for species of conservation concern.

I assessed the likelihood that the measured minimum home ranges were good area estimates by using asymptote analysis (see Laver and Kelly 2008). This was achieved by plotting the cumulative area with the observation time as an XY scatterplot. In this scatterplot, as the fitted line approaches an asymptote, confidence is increasingly gained that the home range area that is being estimated is approaching the actual home range size of the organism that is being studied.

Justification of the methods used

Estimates of species area requirements have long been a fundamental tool for ecological research (e.g. Jennrich and Turner 1969; Ford and Krumme 1979). To quantify the area that is used by a species, MCP analysis based on a number of location points is recommended (IUCN Red List Categories 1994). This method uses the minimum convex polygon to enclose all the locational fixes of an animal. In such a polygon, every line segment between two vertices lies on the MCP boundary and every internal angle is less than or is equal to 180 degrees.

Unlike a concave polygon (which has at least one interior angle greater than 180 degrees), a convex polygon can overestimate the size of a home range (Burgman and Fox 2003). This occurs when line segments include peripheral fixes that incorporate large unused areas within the MCP (Harris *et al.* 1990; Burgman and Fox 2003). Here though, Burgman and Fox (2003) consider that sampling bias such as this is noticeably reduced by consistently taking location fixes throughout a survey period. In support, many researchers consider relatively large data sets are required to accurately capture home range sizes and disproportionate habitat use by species (Harris *et al.* 1990; White and Garrott 1990; Arthur and Schwartz 1999; Girard *et al.* 2002).

This is at odds however with the common statistical requirement that data that is used for home range analyses and patterns of habitat use should avoid autocorrelation (Harris *et al.* 1990). Autocorrelation is analogous to pseudoreplication in statistics, where non-statistically independent replicates are used to test treatment effects (Hurlbert 1984). For example, small data sets that are temporally autocorrelated can result in under-estimates of actual home range sizes (Swihart and Slade 1985).

More commonly though, home range studies using tracking data need to be highly reduced in order to avoid violating the statistical assumption of independence between successive location fixes. Jansen (1999) found that consecutive data points needed to be separated by at least 1.5 hours to avoid autocorrelation. Jansen (1999) also estimated that a minimum of 30 independent locations were required to estimate home range sizes and define patterns of habitat use. In agreement, Kenward (2001) also suggests a minimum of 30 independent fixes are required to estimate the area requirements of an individual animal.

However, severely reducing data through re-sampling to achieve statistical independence has resulted in significant underestimates of home range sizes and the actual daily movements for mobile biota (Reynolds and Laundre 1990; Rooney *et al.* 1998; Kernohan *et al.* 2001; Dahle and Swenson 2003). Furthermore, sub-sampling data sets may not reliably reduce the effect of autocorrelation in any case (Andersen and Rongstad 1989). Nevertheless, these truncated data sets are often used to test hypotheses from which conclusions are then drawn and recommendations made regarding the habitat needs of species (Solla *et al.* 1999). This should raise serious concerns as to the robustness of some conclusions that are drawn from these ecologically incomplete data sets. For example, despite attempts to manipulate data sets to conform to statistical requirements, the accuracy and precision of home range estimates are often improved with linked time-series data, especially for biota that move freely within their home range (Solla *et al.* 1999).

For the purpose of this study (and the studies in the chapters that follow), it makes no sense to treat the within-home range data as being spatially or temporally independent from each other (Otis and White 1999). Significantly reducing these datasets would also unnecessarily sacrifice biologically important and meaningful information that is critical for the purpose of this study and the studies that follow. The behaviour of biota within their respective home ranges is inherently autocorrelated in the first place (Solla *et al.* 1999). Attempting to reduce time-series behavioural records to a small set of statistically independent observations, cannot sufficiently represent what is essentially the non-independent use of habitats (Lair 1987).

Collecting time series data over sustained periods also improves the potential for capturing biologically significant information, while producing better estimates of animal movements and area use within a home range (Solla *et al.* 1999). Data sets that provide an unbiased temporal coverage of behaviours during a study period are also not necessarily limited in their interpretation due to the influence of autocorrelation (Katajisto and Moilanen 2006). Besides, regularly collecting data over long study periods can reduce or even prevent the effects of autocorrelation (Robertson *et al.* 1998; Rooney *et al.* 1998; Fieberg 2007). Furthermore, as these studies are concerned with the area use and time partitioning within individual home ranges, the lack of independence among the observations does not violate the assumptions of home range analysis and area use by the birds (Solla *et al.* 1999). As such, the methods used in this study allows for the confident reporting and interpretation of the bird behaviors that were observed.

Results

Minimum home range estimates for the birds that were surveyed varied from 1.2 ha for a pair of Willie Wagtails to 196.4 ha for a cohesive group of White-winged Choughs (Table 6.1). Surveys periods ranged from one day each for Yellow Thornbill minimum home range estimates to around seven days for White-winged Chough minimum home range estimates (Table 6.2). Excluding the White-winged Chough data because of the extremely large minimum home range sizes (e.g. 196 ha), the estimated minimum home range sizes for the other (smaller) bird species averaged 9.0 ± 1.1 ha.

For the Brown Treecreeper, around two survey days were required for the minimum home range asymptote to be achieved (Figs 6.1 to 6.4). The maximum home range estimate was 20.3 ha (Table 6.1), which contained five birds (Table 6.2).

For the Buff-rumped Thornbill, around two survey days were required for the minimum home range asymptote to be achieved (Figs 6.5 to 6.7). The maximum home range estimate was 9.7 ha (Table 6.1), which contained 10 birds (Table 6.2). A breeding pair at Para Wirra Site A occupied the smallest home range estimate of 3.0 ha (Table 6.2).

For the Hooded Robin, around two survey days were required for the minimum home range asymptote to be achieved (Figs 6.8 to 6.12). The maximum home range estimate was 26.4 ha (Table 6.1) for a pair of birds (Table 6.2). This survey was conducted over four days, with the first two days in fine and sunny conditions and the second two days during a storm. A breeding pair of Hooded Robins at Sandy Creek regularly tended to their nest for the first three survey days and then appeared to lose interest in the nest on the fourth survey day. This increased the minimum home range estimate from around 3 ha for the first three days to at least 7 ha by the fourth day. Because of this, the home range asymptote was not achieved for this home range (Fig. 6.13).

For the Rufous Whistler, one to three survey days were required for the minimum home range asymptote to be achieved (Figs 6.14 to 6.16). The maximum home range estimate was 14.2 ha (Table 6.1) for a pair of birds at Sandy Creek (Table 6.2). However, the minimum home range asymptote for this home range was not achieved (Fig. 6.17).

For the White-browed Babbler, around two to three survey days were required for the minimum home range asymptote to be achieved (Figs 6.18 to 6.21). The maximum home range estimate was 36.4 ha (Table 6.1), which contained five birds (Table 6.2).

For the White-winged Chough, around four to five survey days were required for the minimum home range asymptote to be achieved (Figs 6.22 to 6.24). The maximum home range estimate was 196.4 ha (Table 6.1), which contained approximately 40 birds (Table 6.2).

For the Willie Wagtail, around one survey day was required for the minimum home range asymptote to be achieved (Figs 6.25 to 6.30). The maximum home range estimate was 11.8 ha (Table 6.1) for a pair of birds (Table 6.2).

For the Yellow-rumped Thornbill, around one to two survey days were required for the minimum home range asymptote to be achieved (Figs 6.31 to 6.33). The maximum home range estimate was 24.5 ha (Table 6.1), which contained 15 birds (Table 6.2).

For the Yellow Thornbill, one survey day was required for the minimum home range asymptote to be achieved (Figs 6.34 to 6.41). The maximum home range estimate was 9.4 ha (Table 6.1), which contained 3 birds (Table 6.2).

Table 6.1: Minimum home range estimates for birds of the nine woodland species within the north-central zone of the southern Mount Lofty Ranges, South Australia. Home ranges were based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). Shown are the high and low home range estimates for each species (in hectares) and the average home range area for each species (Mean) with the associated standard error (SE) and the number of home ranges that were analysed (*n*). Data collected April 2009 – February 2011.

| Bird species | Range (ha) | | Mean (ha) | SE | <i>n</i> |
|-------------------------|------------|-------|--------------|------|----------|
| | Low | High | | | |
| Brown Treecreeper | 5.6 | 20.3 | 10.2 | 3.4 | 4 |
| Buff-rumped Thornbill | 3.0 | 9.7 | 6.7 | 2.0 | 3 |
| Hooded Robin | 7.2 | 26.4 | 16.8 | 3.9 | 4 |
| Rufous Whistler | 2.3 | 14.2 | 7.7 | 2.5 | 4 |
| White-browed Babbler | 6.6 | 36.4 | 18.8 | 6.3 | 4 |
| White-winged Chough | 14.0 | 196.4 | 120.7 | 54.9 | 3 |
| Willie Wagtail | 1.2 | 11.8 | 5.4 | 1.8 | 6 |
| Yellow-rumped Thornbill | 14.3 | 24.5 | 18.1 | 3.2 | 3 |
| Yellow Thornbill | 3.0 | 9.4 | 6.3 | 0.7 | 8 |

Table 6.2: Minimum home range data for birds of nine woodland species at various locations within the north-central zone of the southern Mount Lofty Ranges, South Australia. Shown are the number of days per survey, the number of birds of each species within each minimum home range, and the number of GPS waypoints used for Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006).

| Species | Location | Date | # days | # birds | # waypoints | MCP (ha) |
|-------------------------|----------------------|----------------------------------|-----------|------------|----------------|-------------|
| Brown Treecreeper | Altona Chatterton | June / July 2009 | 3 | 4 | 577 | 5.6 |
| Brown Treecreeper | Altona Kies | April 2009 | 3 | 5 | 482 | 20.3 |
| Brown Treecreeper | Altona Torbreck | June / July 2009 | 3 | 5 | 576 | 8.7 |
| Brown Treecreeper | Altona Torbreck | June 2010 | 2 | 3 | 283 | 6.3 |
| Buff-rumped Thornbill | Para Wirra - Site A | August 2009 | 3 | 2 | 462 | 3.0 |
| Buff-rumped Thornbill | Para Wirra - Site B | July 2009 | 3 | 10 | 496 | 9.7 |
| Buff-rumped Thornbill | Para Wirra - Site B | June / July 2010 | 3 | 5 | 431 | 7.3 |
| Hooded Robin | Altona Chatterton | July 2009 | 2 | 2 | 529 | 19.2 |
| Hooded Robin | Altona Chatterton | July 2009 (storm) ¹ | 2 | 2 | 607 | 12.1 |
| Hooded Robin | Altona Chatterton | June 2010 | 2 | 2 | 278 | 8.5 |
| Hooded Robin | Altona Chatterton | June 2010 (showers) ¹ | 1 | 2 | 277 | 8.1 |
| Hooded Robin | Altona Kies | May 2009 | 5 | 3 | 510 | 17.1 |
| Hooded Robin | Sandy Creek | August 2009 | 4 | 2 | 467 | 7.2 |
| Rufous Whistler | Altona Kies | September 2009 | 2 | 2 | 305 | 2.3 |
| Rufous Whistler | Altona Landcare | September 2009 | 3 | 2 | 279 | 5.9 |
| Rufous Whistler | Sandy Creek | August 2009 | 3 | 2 | 491 | 14.2 |
| Rufous Whistler | Sandy Creek | September 2010 | 3 | 2 | 467 | 8.5 |
| White-browed Babbler | Altona Chatterton | May / June 2009 | 7 | 5 | 975 | 36.4 |
| White-browed Babbler | Altona Kies | April / May 2009 | 6 | 15 | 847 | 14.4 |
| White-browed Babbler | Altona Kies | June 2010 | 5 | 12 | 825 | 17.9 |
| White-browed Babbler | Sandy Creek | August 2009 | 3 | 2 | 377 | 6.6 |
| White-winged Chough | Para Wirra | August 2009 | 5 | 6 | 508 | 14.0 |
| White-winged Chough | Sandy Creek | June 2009 | 8 | 40 | 1052 | 196.4 |
| White-winged Chough | Sandy Creek | June / July 2010 | 7 | 35 | 794 | 151.8 |
| Willie Wagtail | Altona Chatterton | July 2009 | 3 | 2 | 406 | 11.8 |
| Willie Wagtail | Altona Chatterton | February 2011 | 1 | 2 | 79 | 1.2 |
| Willie Wagtail | Altona Kies - Site A | May 2009 | 2 | 2 | 244 | 3.9 |
| Willie Wagtail | Altona Kies - Site B | May 2009 | 2 | 2 | 357 | 3.0 |
| Willie Wagtail | Altona Kies - Site B | June 2010 | 2 | 2 | 414 | 2.6 |
| Willie Wagtail | Sandy Creek | August 2009 | 2 | 1 | 304 | 9.6 |
| Yellow-rumped Thornbill | Altona Chatterton | May 2009 | 3 | 15 | 584 | 24.5 |
| Yellow-rumped Thornbill | Altona Chatterton | June 2010 | 3 | 12 | 454 | 15.6 |
| Yellow-rumped Thornbill | Altona Kies | May 2009 | 2 | 5 | 355 | 14.3 |
| Yellow Thornbill | Altona Chatterton | June 2009 | 1 | 2 | 275 | 6.5 |
| Yellow Thornbill | Altona Kies | May 2009 | 1 | 3 | 213 | 4.0 |
| Yellow Thornbill | Para Wirra - Site A | August 2009 | 1 | 4 | 255 | 3.0 |
| Yellow Thornbill | Para Wirra - Site B | July 2009 | 2 | 3 | 334 | 7.3 |
| Yellow Thornbill | Sandy Creek - Site A | June 2009 | 1 | 3 | 269 | 9.4 |
| Yellow Thornbill | Sandy Creek - Site A | June 2010 | 1 | 10 | 346 | 6.6 |
| Yellow Thornbill | Sandy Creek - Site B | June 2009 | 1 | 3 | 261 | 6.8 |
| Yellow Thornbill | Sandy Creek - Site B | June 2010 | 1 | 4 | 314 | 7.1 |

¹For the Hooded Robin at Altona Chatterton, the combined MCP for July 2009 was 26.4 ha and the combined MCP for June 2010 was 16.6 ha.

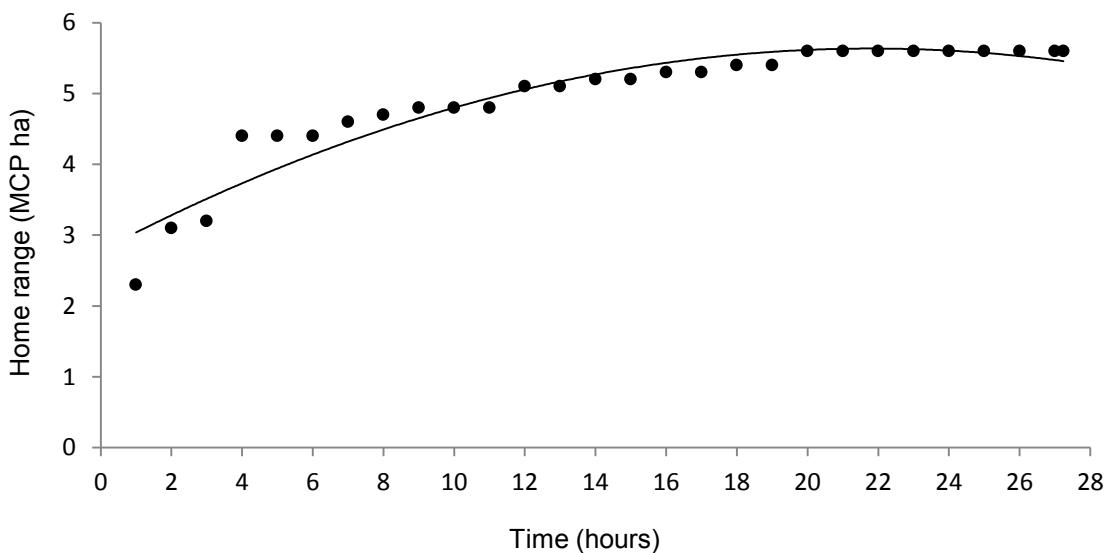


Figure 6.1: Minimum home range estimate for the Brown Treecreeper (*Climacteris picumnus*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on four birds at Altona Chatterton. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 577 waypoint locations being recorded. The MCP estimated a home range of 5.6 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June – July 2009.

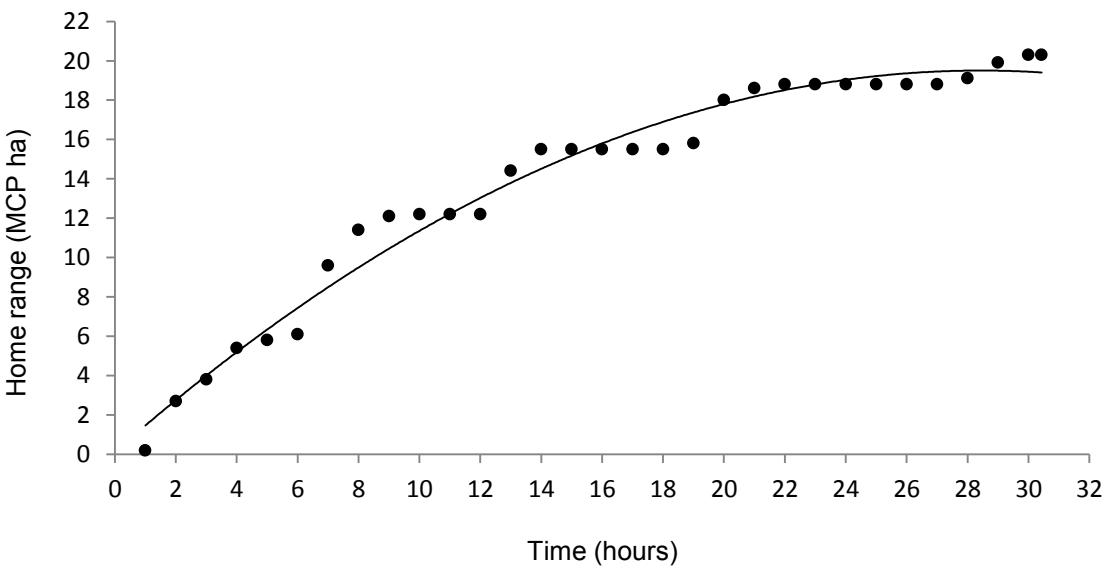


Figure 6.2: Minimum home range estimate for the Brown Treecreeper (*Climacteris picumnus*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on five birds at Altona Kies. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 482 waypoint locations being recorded. The MCP estimated a home range of 20.3 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected April 2009.

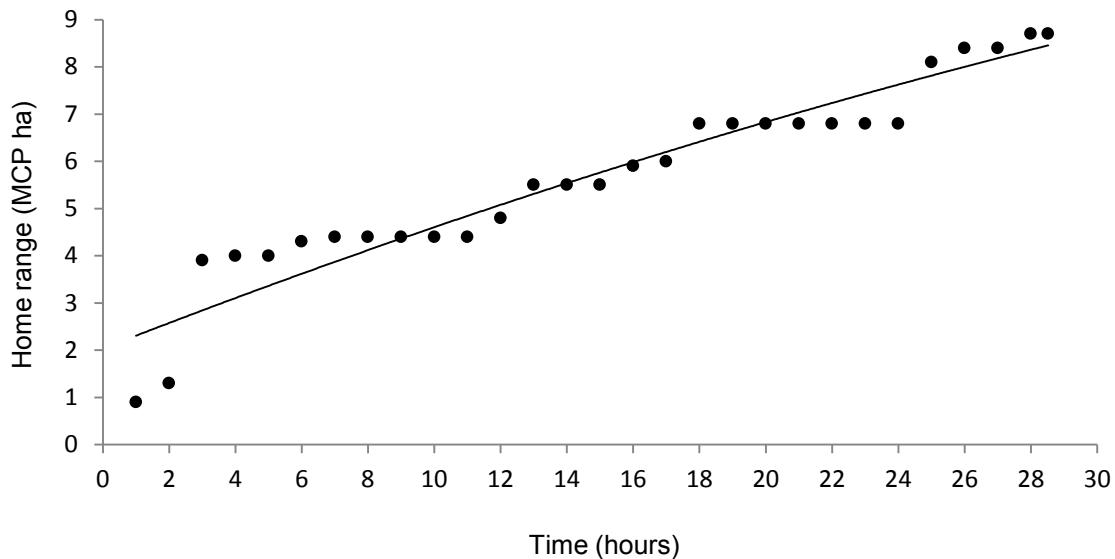


Figure 6.3: Minimum home range estimate for the Brown Treecreeper (*Climacteris picumnus*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on five birds at Altona Torbreck. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 576 waypoint locations being recorded. The MCP estimated a home range of 8.7 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. As indicated by the polynomial trendline, the minimum home range estimate was not achieved for these birds. Data collected June – July 2009.

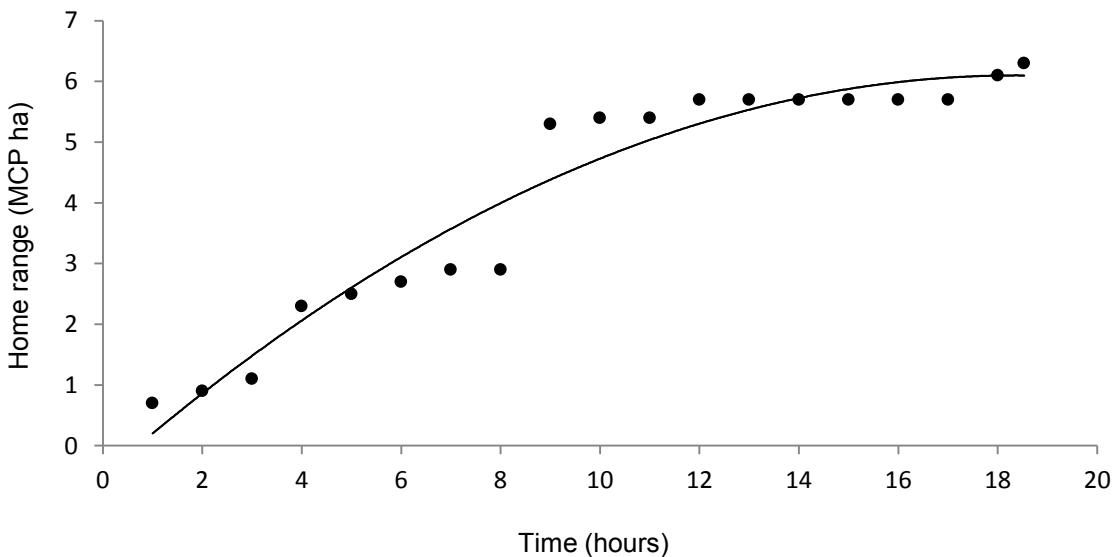


Figure 6.4: Minimum home range estimate for the Brown Treecreeper (*Climacteris picumnus*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on three birds at Altona Torbreck. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 283 waypoint locations being recorded. The MCP estimated a home range of 6.3 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected July 2010.

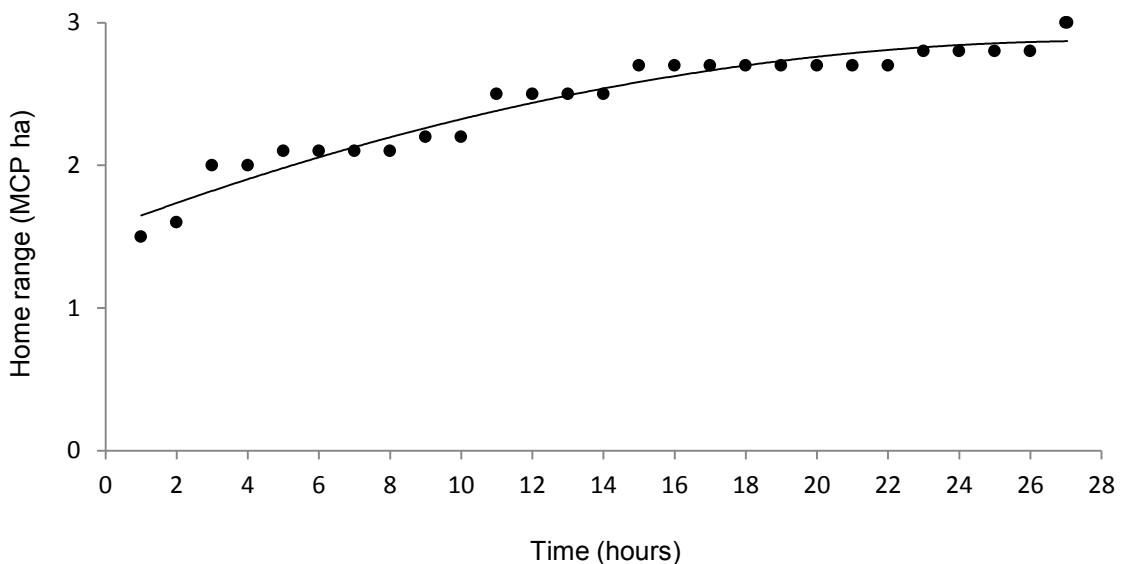


Figure 6.5: Minimum home range estimate for the Buff-rumped Thornbill (*Acanthiza reguloides*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on two birds at Para Wirra Site A. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 462 waypoint locations being recorded. The MCP estimated a home range of 3.0 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected August 2009.

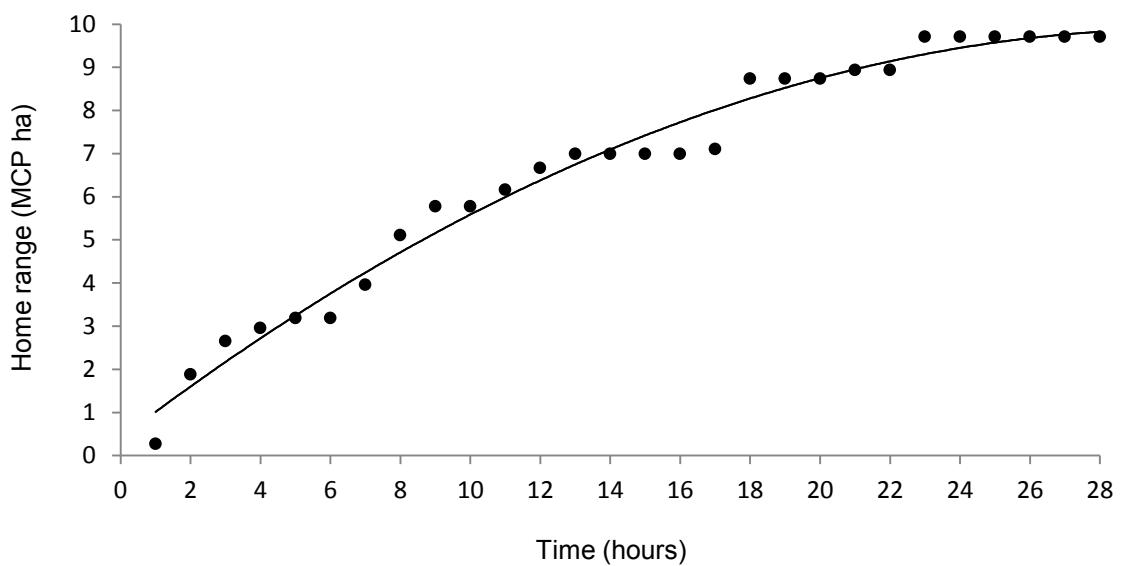


Figure 6.6: Minimum home range estimate for the Buff-rumped Thornbill (*Acanthiza reguloides*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on 10 birds at Para Wirra Site B. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 496 waypoint locations being recorded. The MCP estimated a home range of 9.7 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected July 2009.

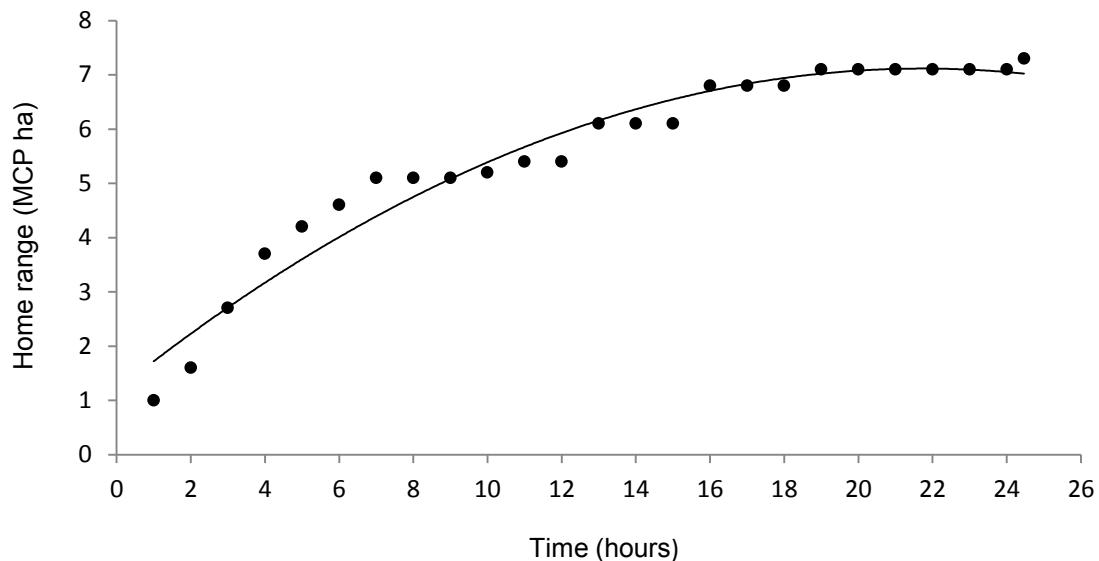


Figure 6.7: Minimum home range estimate for the Buff-rumped Thornbill (*Acanthiza reguloides*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on five birds at Para Wirra Site B. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 431 waypoint locations being recorded. The MCP estimated a home range of 7.3 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June – July 2010.

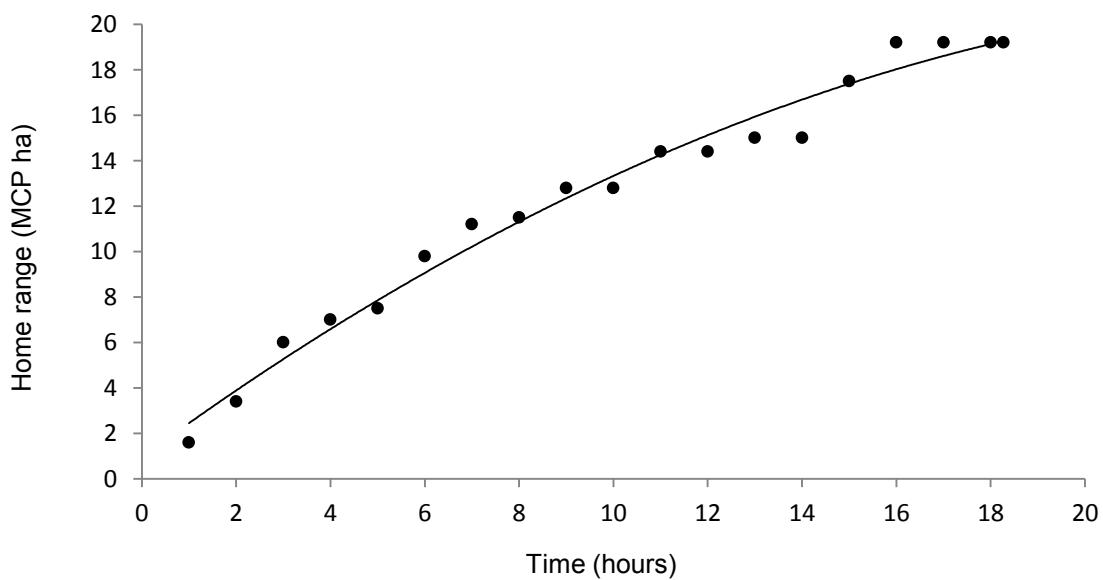


Figure 6.8: Minimum home range estimate for the Hooded Robin (*Melanodryas cucullata*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on two birds at Altona Chatterton. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 529 waypoint locations being recorded. The MCP estimated a home range of 19.2 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected July 2009.

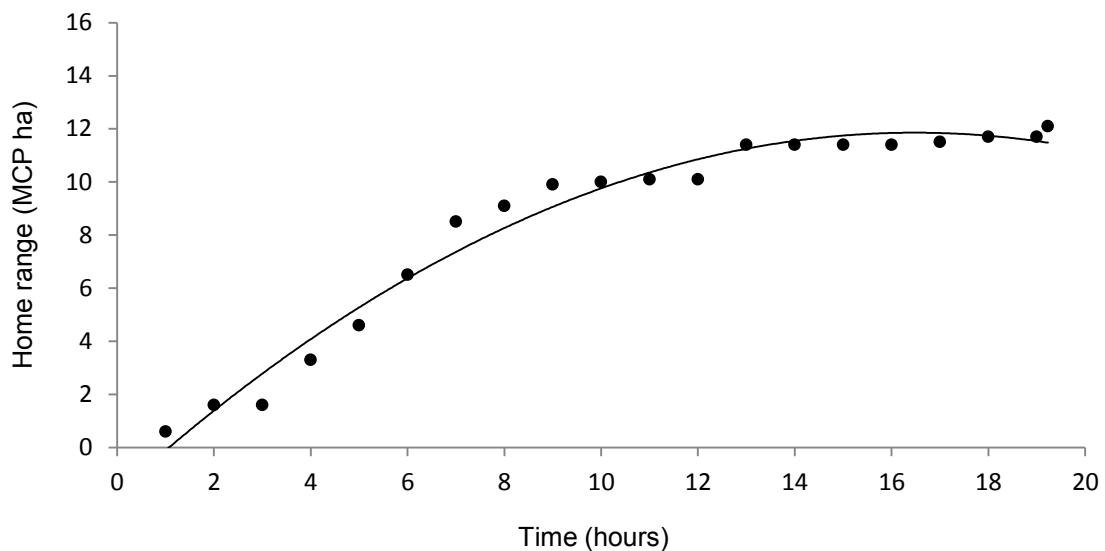


Figure 6.9: Minimum home range estimate for the Hooded Robin (*Melanodryas cucullata*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on two birds at Altona Chatterton (during a storm). The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 607 waypoint locations being recorded. The MCP estimated a home range of 12.1 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected July 2009.

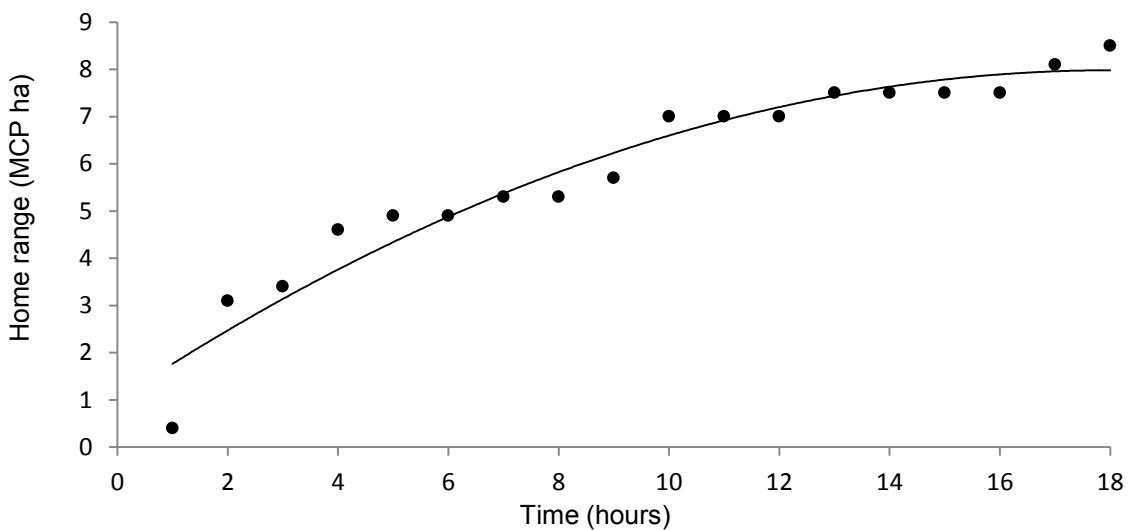


Figure 6.10: Minimum home range estimate for the Hooded Robin (*Melanodryas cucullata*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on two birds at Altona Chatterton. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 278 waypoint locations being recorded. The MCP estimated a home range of 8.5 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2010.

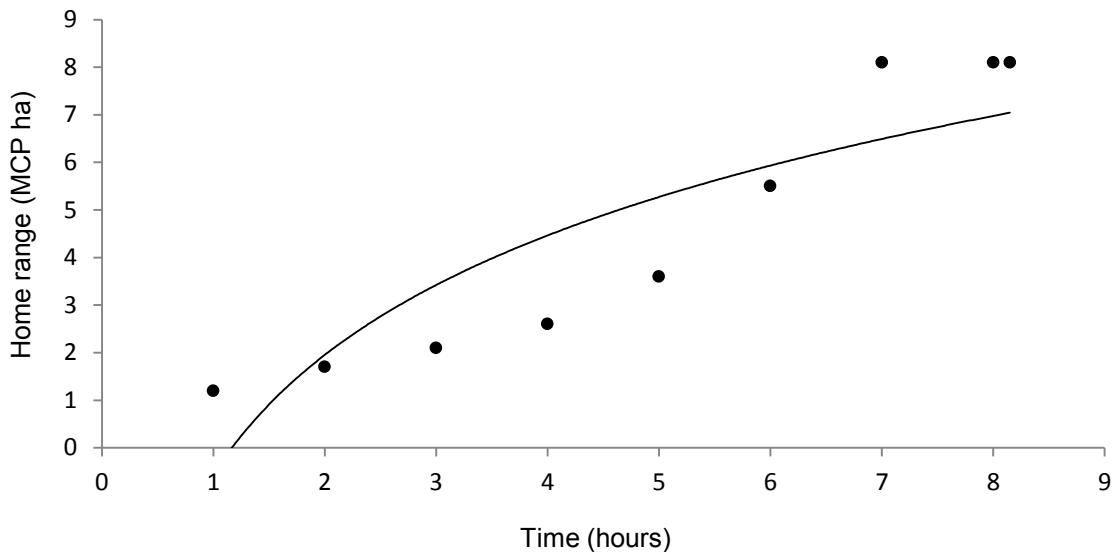


Figure 6.11: Minimum home range estimate for the Hooded Robin (*Melanodryas cucullata*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on two birds at Altona Chatterton (overcast conditions with persistent light showers). The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 277 waypoint locations being recorded. The MCP estimated a home range of 8.1 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2010.

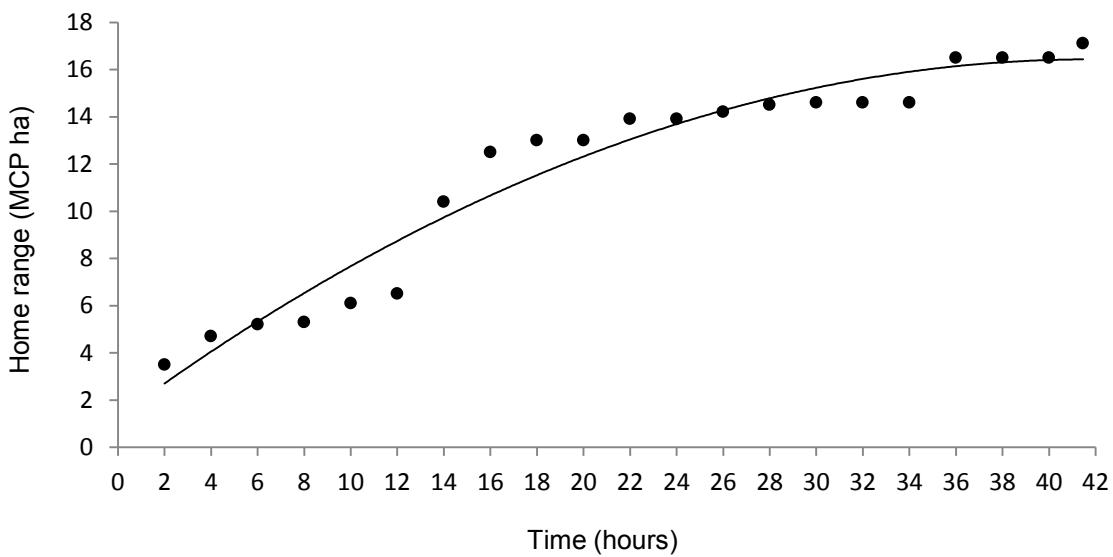


Figure 6.12: Minimum home range estimate for the Hooded Robin (*Melanodryas cucullata*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over five days on three birds at Altona Kies. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 510 waypoint locations being recorded. The MCP estimated a home range of 17.1 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected May 2009.

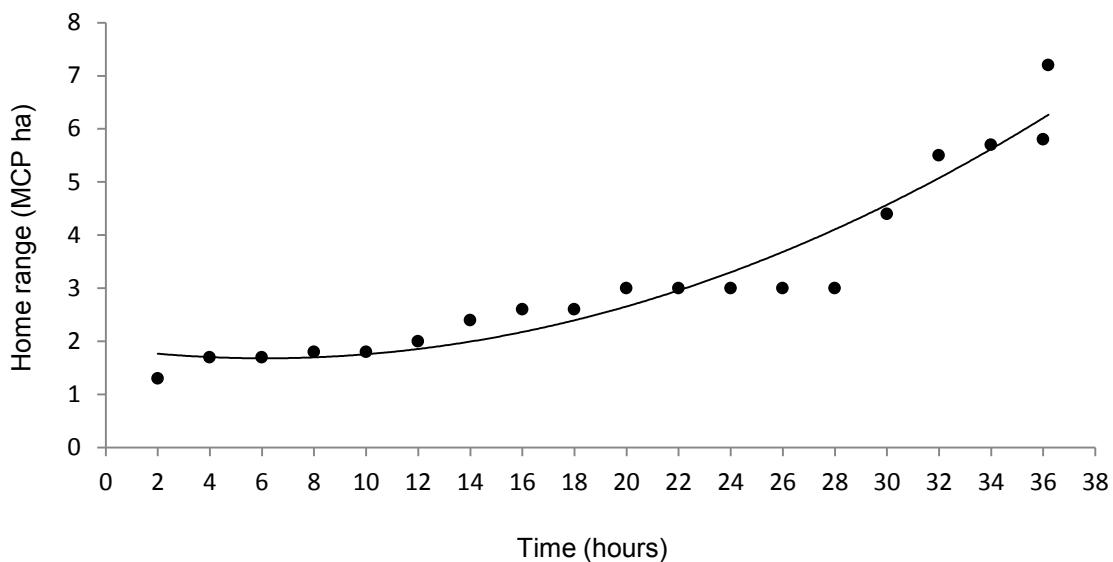


Figure 6.13: Minimum home range estimate for the Hooded Robin (*Melanodryas cucullata*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over four days on two birds at Sandy Creek. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 467 waypoint locations being recorded. The MCP estimated a home range of 7.2 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. As indicated by the polynomial trendline, the minimum home range estimate was not achieved for these birds. Data collected August 2009.

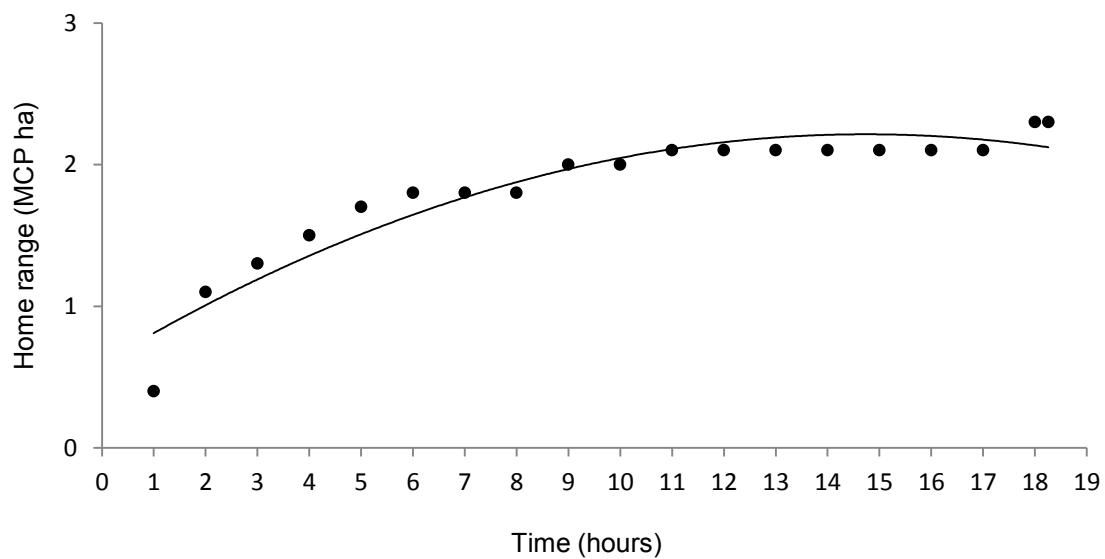


Figure 6.14: Minimum home range estimate for the Rufous Whistler (*Pachycephala rufiventris*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on two birds at Altona Kies. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 305 waypoint locations being recorded. The MCP estimated a home range of 2.3 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected September 2009.

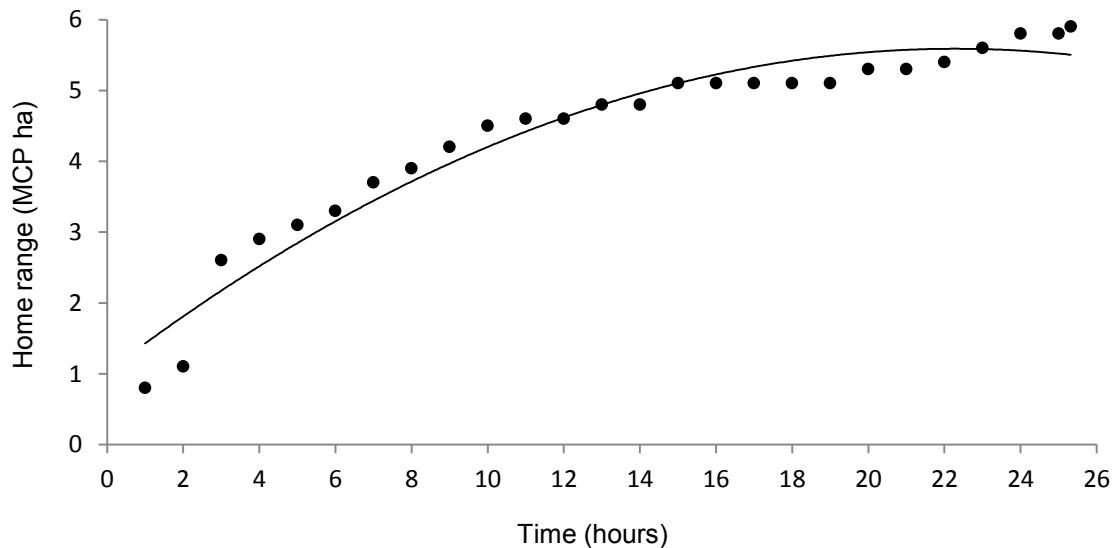


Figure 6.15: Minimum home range estimate for the Rufous Whistler (*Pachycephala rufiventris*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on two birds at Altona Landcare. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 297 waypoint locations being recorded. The MCP estimated a home range of 5.9 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected September 2009.

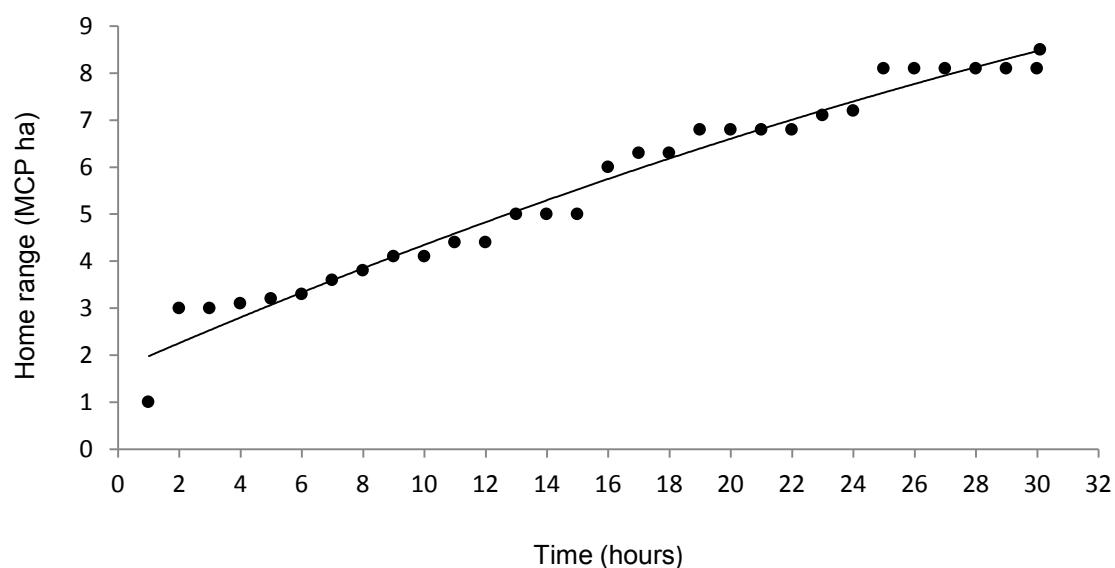


Figure 6.16: Minimum home range estimate for the Rufous Whistler (*Pachycephala rufiventris*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on two birds at Sandy Creek. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 491 waypoint locations being recorded. The MCP estimated a home range of 14.2 ha during the survey. As indicated by the polynomial trendline, the minimum home range estimate was not achieved for these birds. Data collected August 2009.

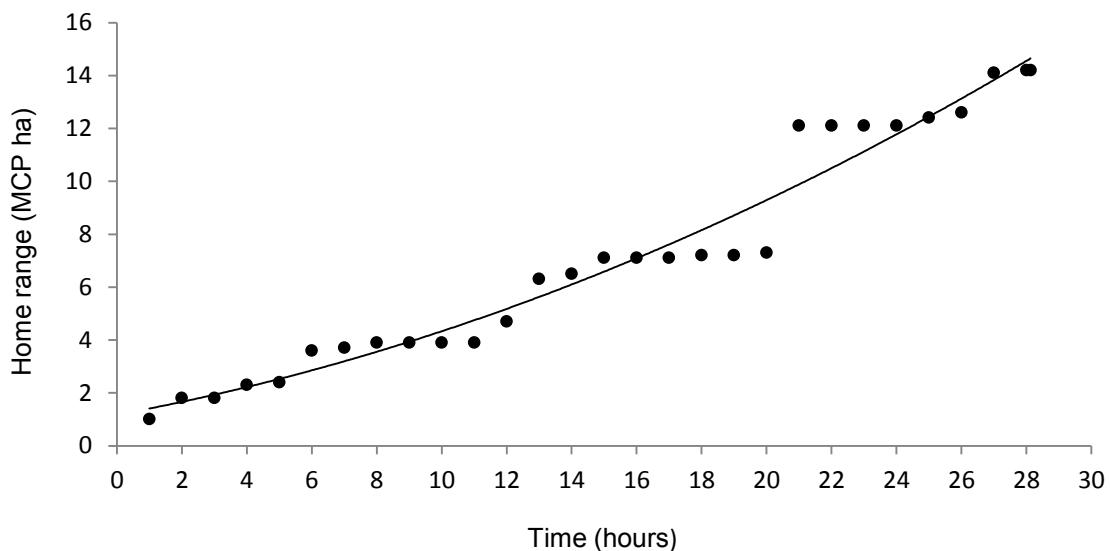


Figure 6.17: Minimum home range estimate for the Rufous Whistler (*Pachycephala rufiventris*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on two birds at Sandy Creek. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 467 waypoint locations being recorded. The MCP estimated a home range of 8.5 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. As indicated by the polynomial trendline, the minimum home range estimate was not achieved for these birds. Data collected September 2010.

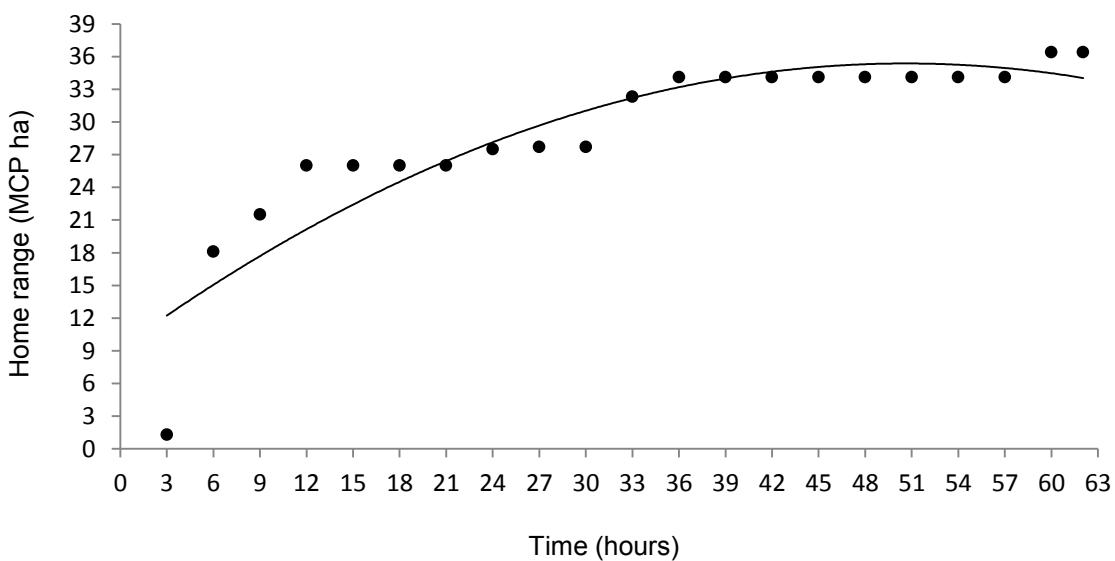


Figure 6.18: Minimum home range estimate for the White-browed Babbler (*Pomatostomus superciliosus*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over seven days on five birds at Altona Chatterton. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 975 waypoint locations being recorded. The MCP estimated a home range of 36.4 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected May – June 2009.

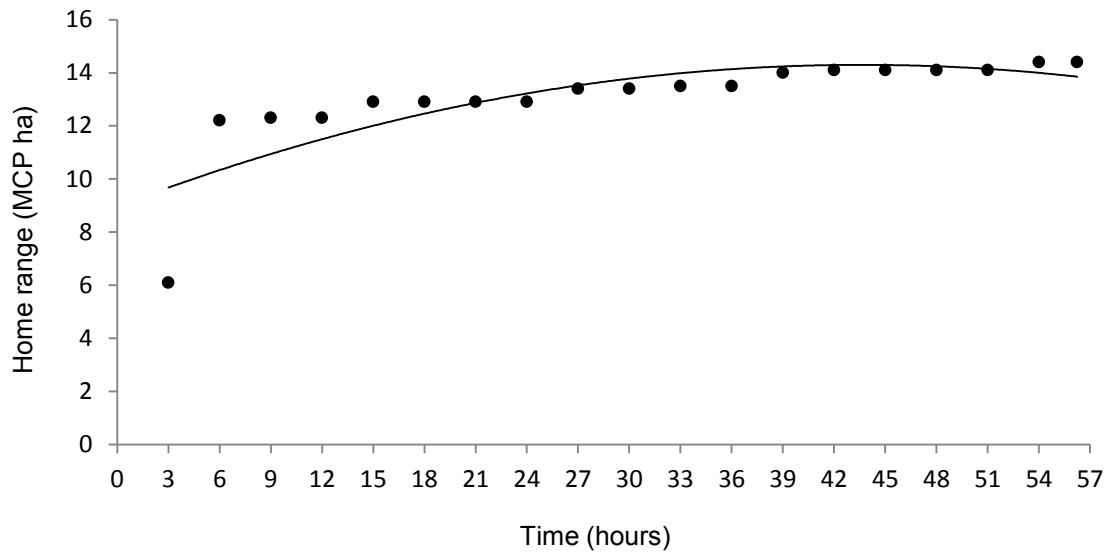


Figure 6.19: Minimum home range estimate for the White-browed Babbler (*Pomatostomus superciliosus*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over six days on 15 birds at Altona Kies. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 847 waypoint locations being recorded. The MCP estimated a home range of 14.4 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected April – May 2009.

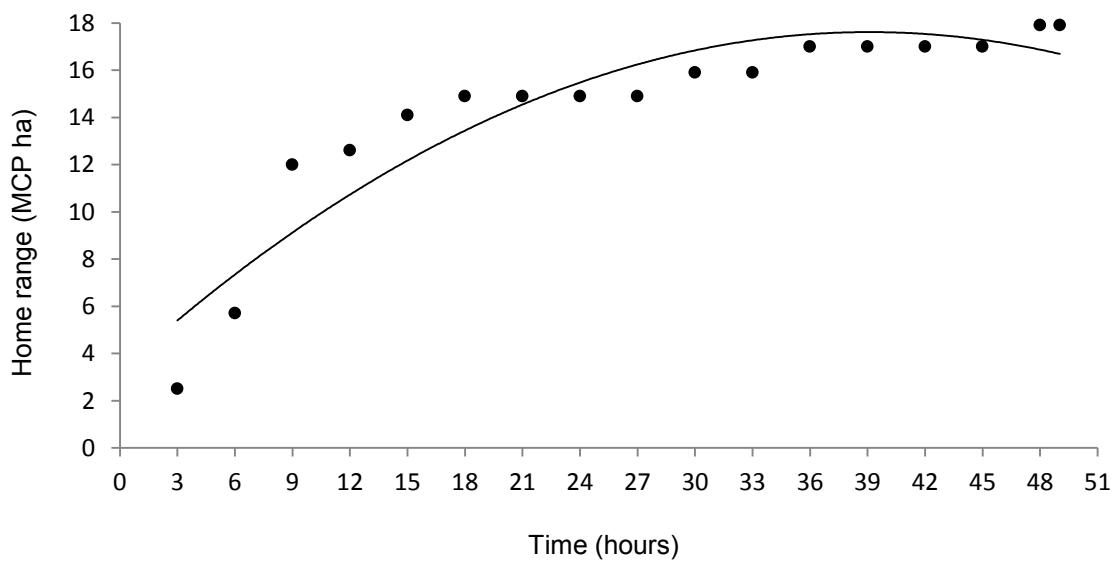


Figure 6.20: Minimum home range estimate for the White-browed Babbler (*Pomatostomus superciliosus*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over five days on 12 birds at Altona Kies. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 825 waypoint locations being recorded. The MCP estimated a home range of 17.9 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2010.

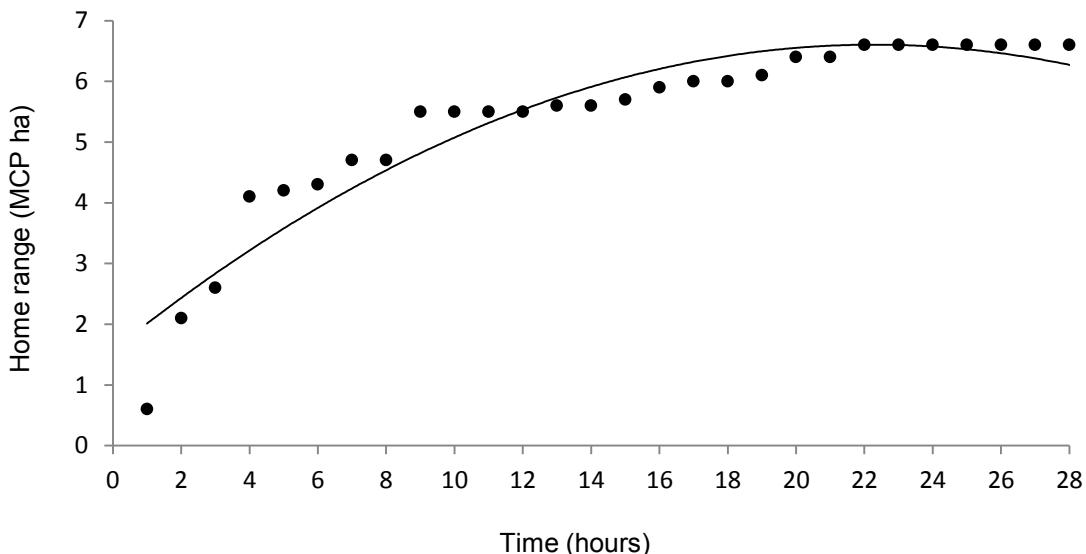


Figure 6.21: Minimum home range estimate for the White-browed Babbler (*Pomatostomus superciliosus*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on two birds at Sandy Creek. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 377 waypoint locations being recorded. The MCP estimated a home range of 6.6 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected August 2009.

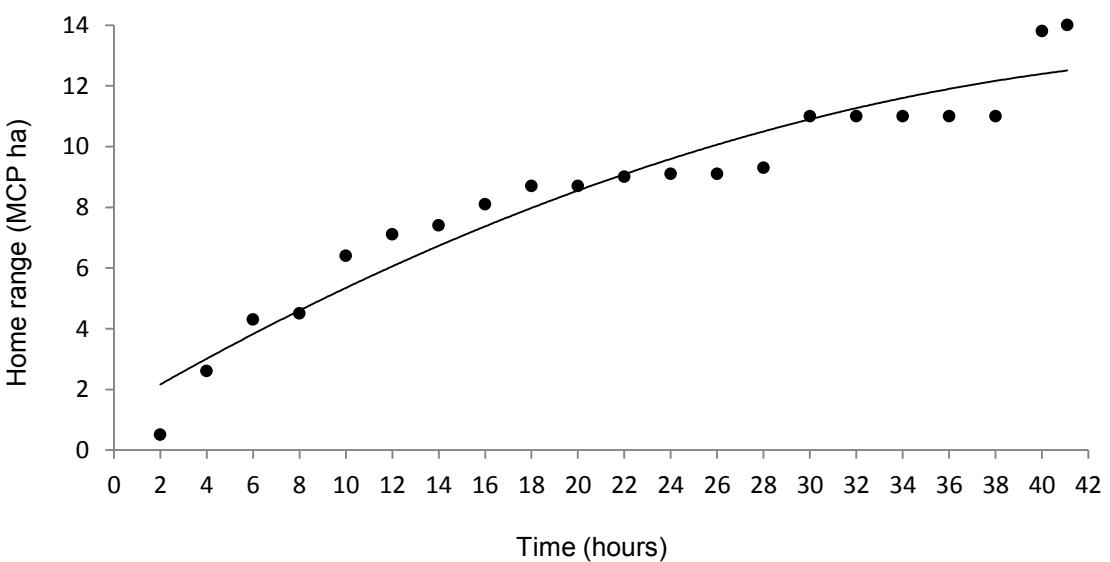


Figure 6.22: Minimum home range estimate for the White-winged Chough (*Corcorax melanorhamphos*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over five days on six birds at Para Wirra. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 508 waypoint locations being recorded. The MCP estimated a home range of 14.0 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected August 2009.

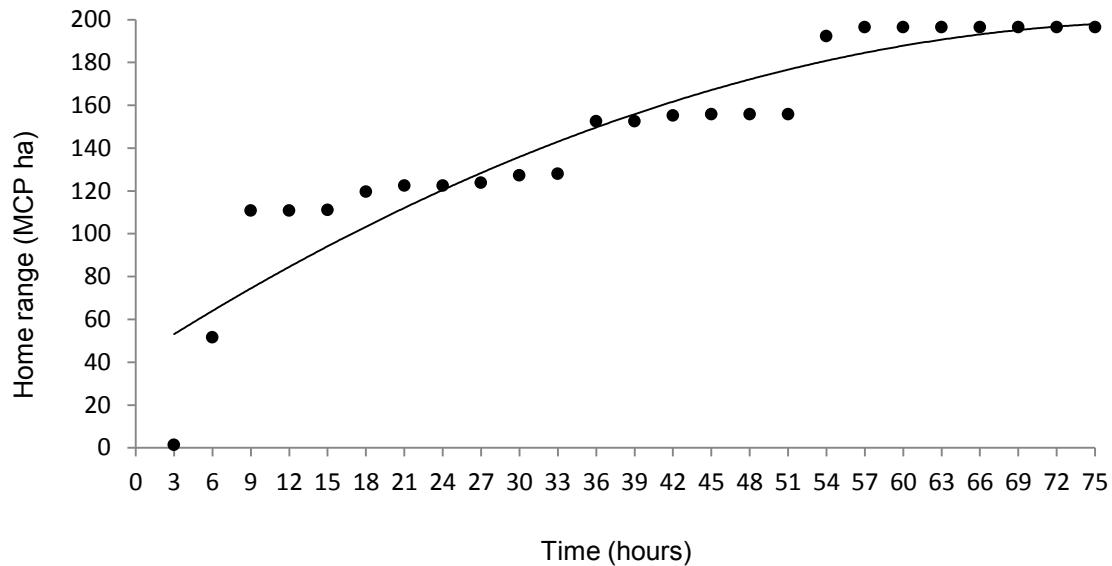


Figure 6.23: Minimum home range estimate for the White-winged Chough (*Corcorax melanorhamphos*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over eight days on 40 birds at Sandy Creek. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 1,052 waypoint locations being recorded. The MCP estimated a home range of 196.4 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2009.

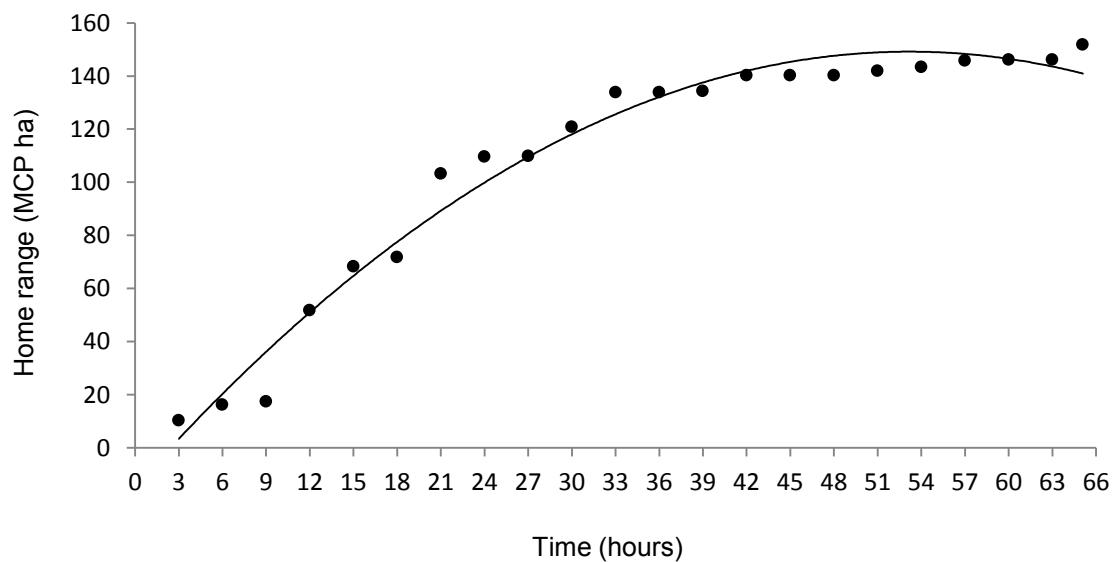


Figure 6.24: Minimum home range estimate for the White-winged Chough (*Corcorax melanorhamphos*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over seven days on 35 birds at Sandy Creek. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 794 waypoint locations being recorded. The MCP estimated a home range of 151.8 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June – July 2010.

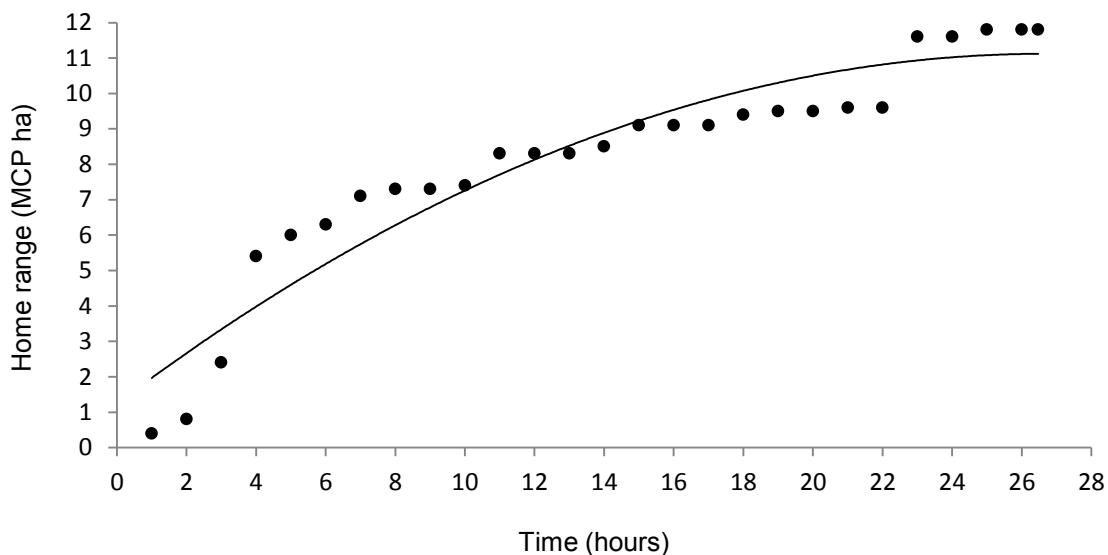


Figure 6.25: Minimum home range estimate for the Willie Wagtail (*Rhipidura leucophrys*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on two birds at Altona Chatterton. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 406 waypoint locations being recorded. The MCP estimated a home range of 11.8 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected July 2009.

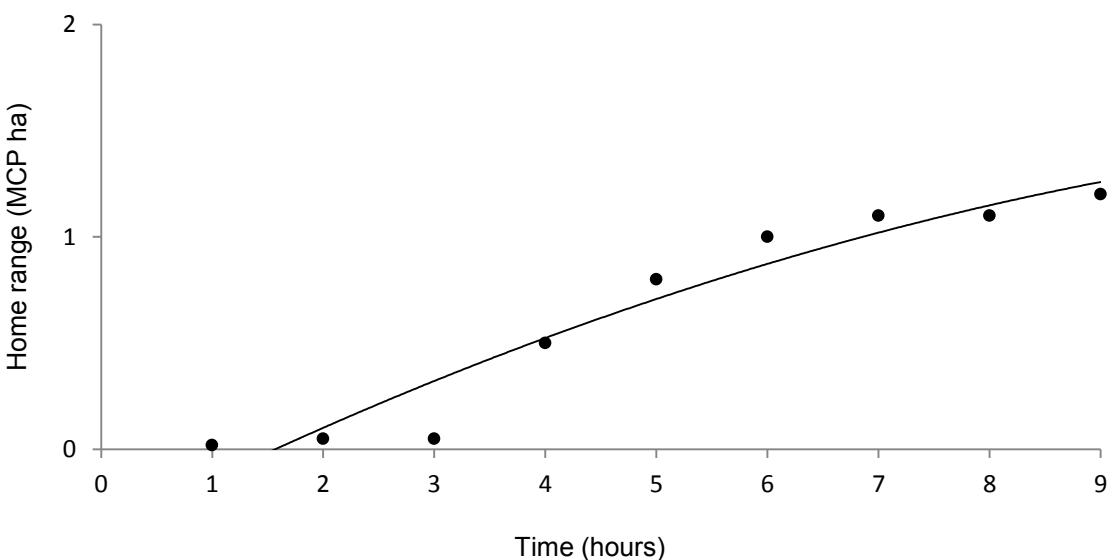


Figure 6.26: Minimum home range estimate for the Willie Wagtail (*Rhipidura leucophrys*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on two birds at Altona Chatterton. The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 79 waypoint locations being recorded. The MCP estimated a home range of 1.2 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected February 2011.

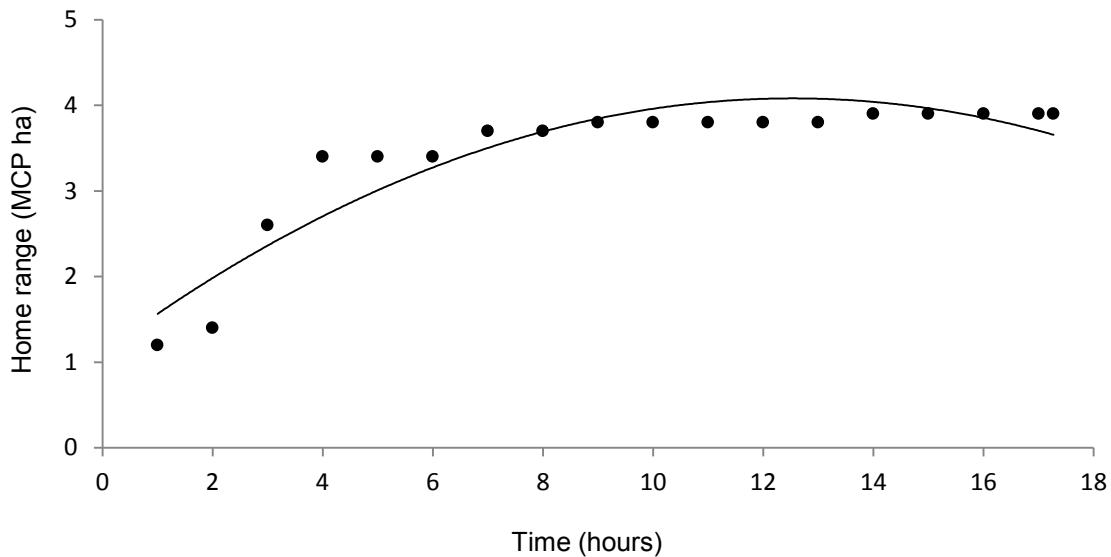


Figure 6.27: Minimum home range estimate for the Willie Wagtail (*Rhipidura leucophrys*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on two birds at Altona Kies Site A. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 244 waypoint locations being recorded. The MCP estimated a home range of 3.9 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected May 2009.

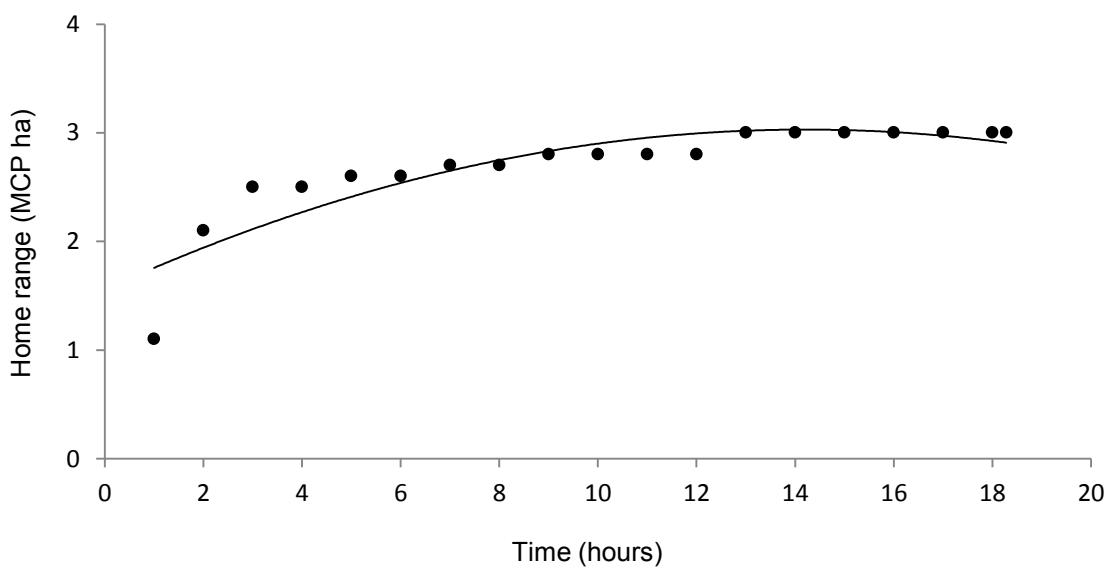


Figure 6.28: Minimum home range estimate for the Willie Wagtail (*Rhipidura leucophrys*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on two birds at Altona Kies Site B. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 357 waypoint locations being recorded. The MCP estimated a home range of 3.0 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected May 2009.

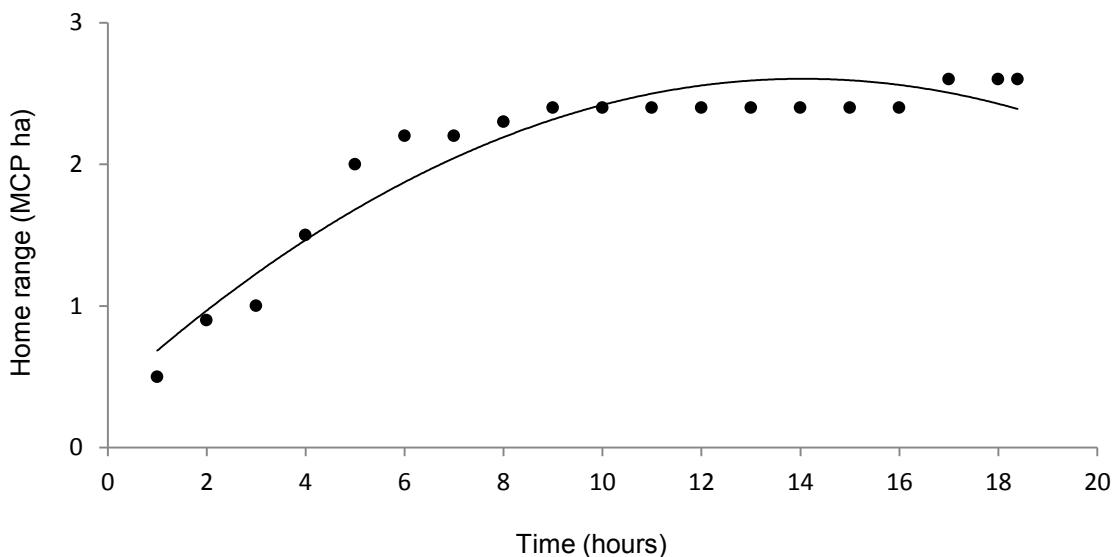


Figure 6.29: Minimum home range estimate for the Willie Wagtail (*Rhipidura leucophrys*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on two birds at Altona Kies Site B. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 414 waypoint locations being recorded. The MCP estimated a home range of 2.6 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2010.

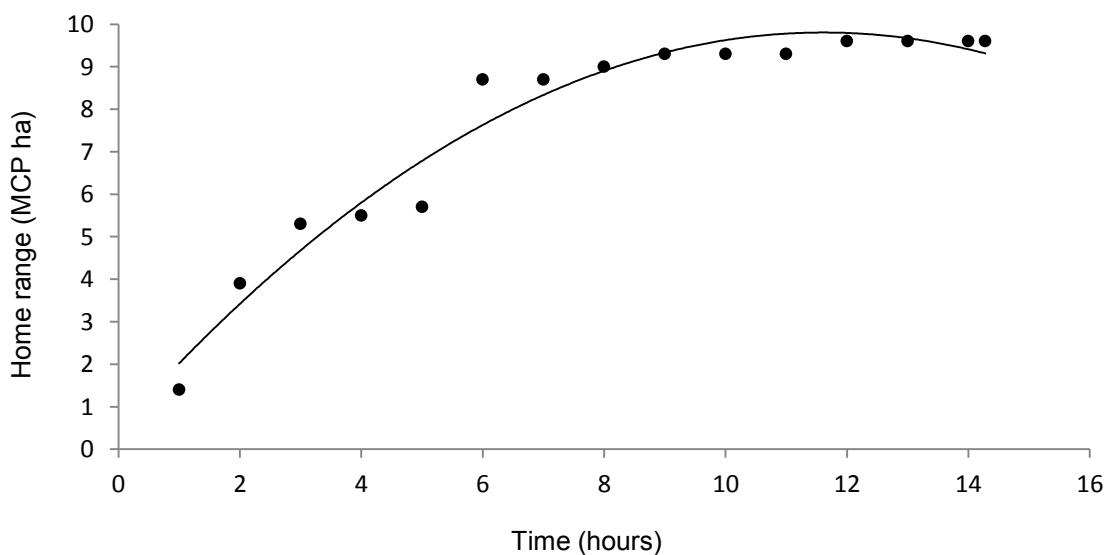


Figure 6.30: Minimum home range estimate for the Willie Wagtail (*Rhipidura leucophrys*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on one bird at Sandy Creek. The survey was conducted for sustained periods throughout each day to record the locations the bird stopped as it moved around its habitat. This resulted in 304 waypoint locations being recorded. The MCP estimated a home range of 9.6 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected August 2009.

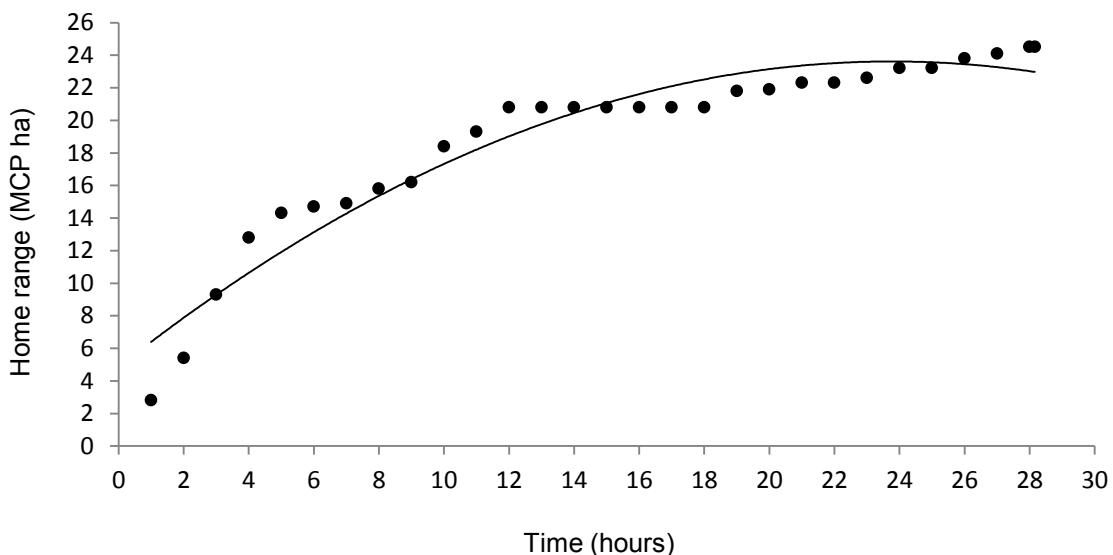


Figure 6.31: Minimum home range estimate for the Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on 15 birds at Altona Chatterton. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 584 waypoint locations being recorded. The MCP estimated a home range of 24.5 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected May 2009.

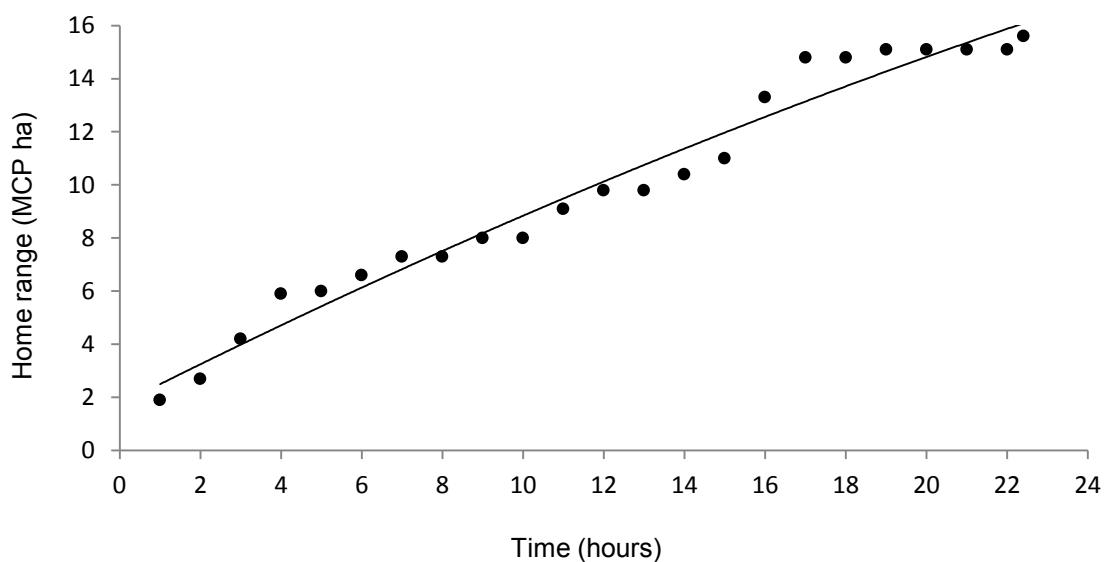


Figure 6.32: Minimum home range estimate for the Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over three days on 12 birds at Altona Chatterton. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 454 waypoint locations being recorded. The MCP estimated a home range of 15.6 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. As indicated by the polynomial trendline, the minimum home range estimate was not achieved for these birds. Data collected June 2010.

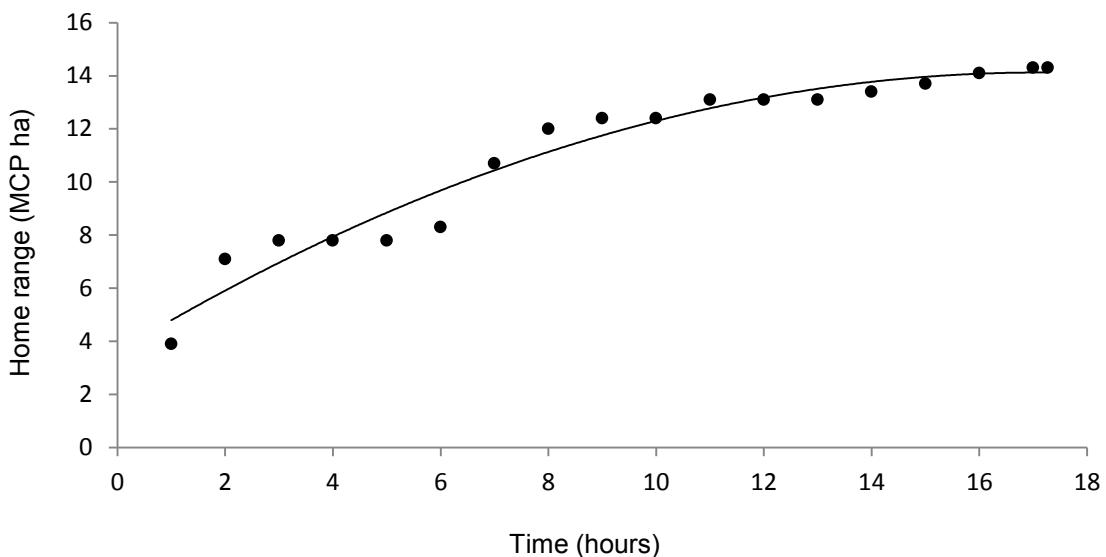


Figure 6.33: Minimum home range estimate for the Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on five birds at Altona Kies. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 355 waypoint locations being recorded. The MCP estimated a home range of 14.3 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected May 2009.

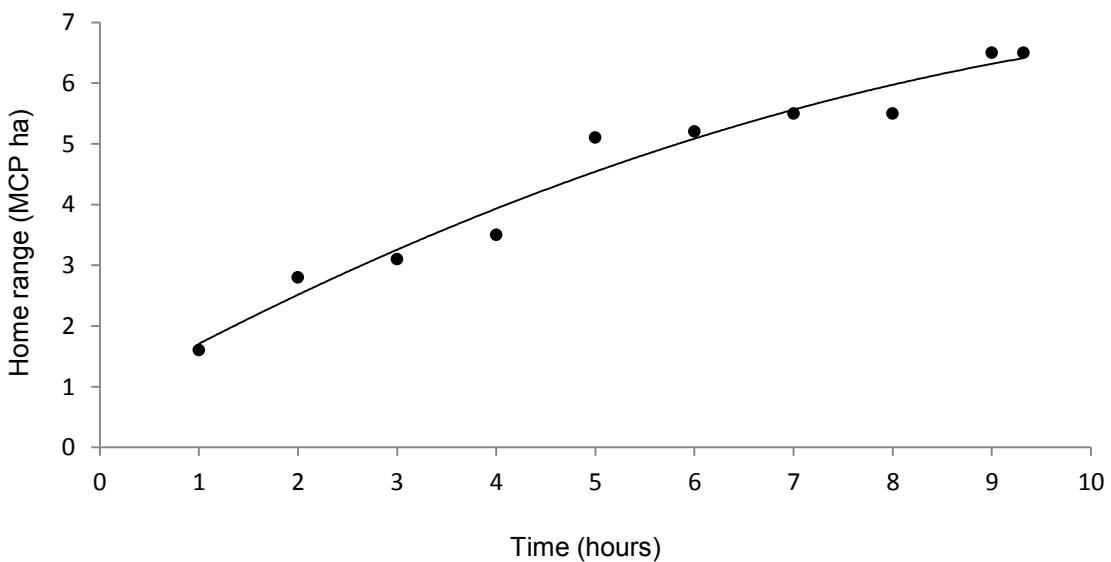


Figure 6.34: Minimum home range estimate for the Yellow Thornbill (*Acanthiza nana*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on two birds at Altona Chatterton. The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 275 waypoint locations being recorded. The MCP estimated a home range of 6.5 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2009.

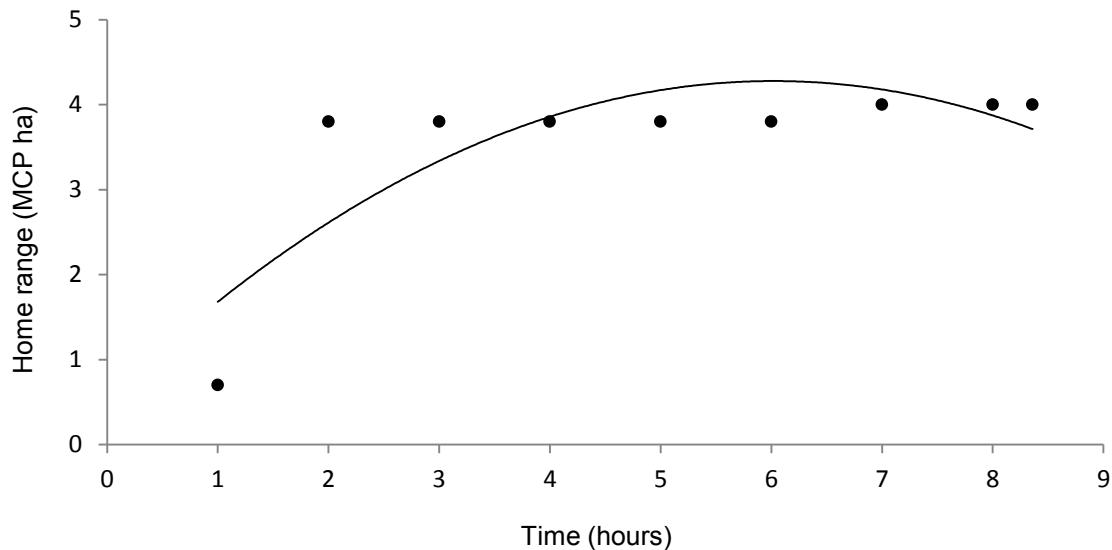


Figure 6.35: Minimum home range estimate for the Yellow Thornbill (*Acanthiza nana*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on three birds at Altona Kies. The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 213 waypoint locations being recorded. The MCP estimated a home range of 4.0 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected May 2009.

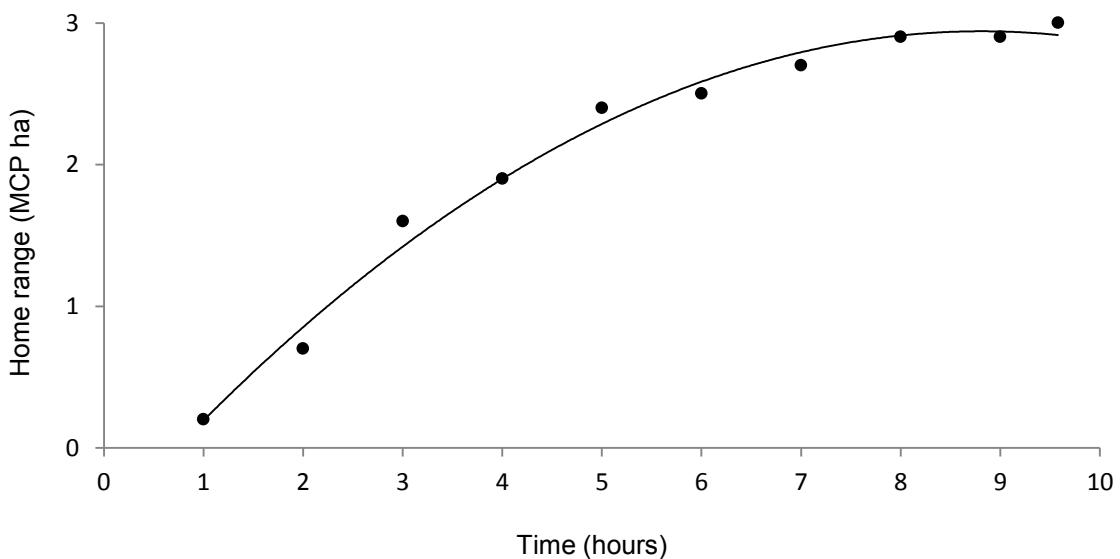


Figure 6.36: Minimum home range estimate for the Yellow Thornbill (*Acanthiza nana*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on four birds at Para Wirra Site A. The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 255 waypoint locations being recorded. The MCP estimated a home range of 3.0 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected August 2009.

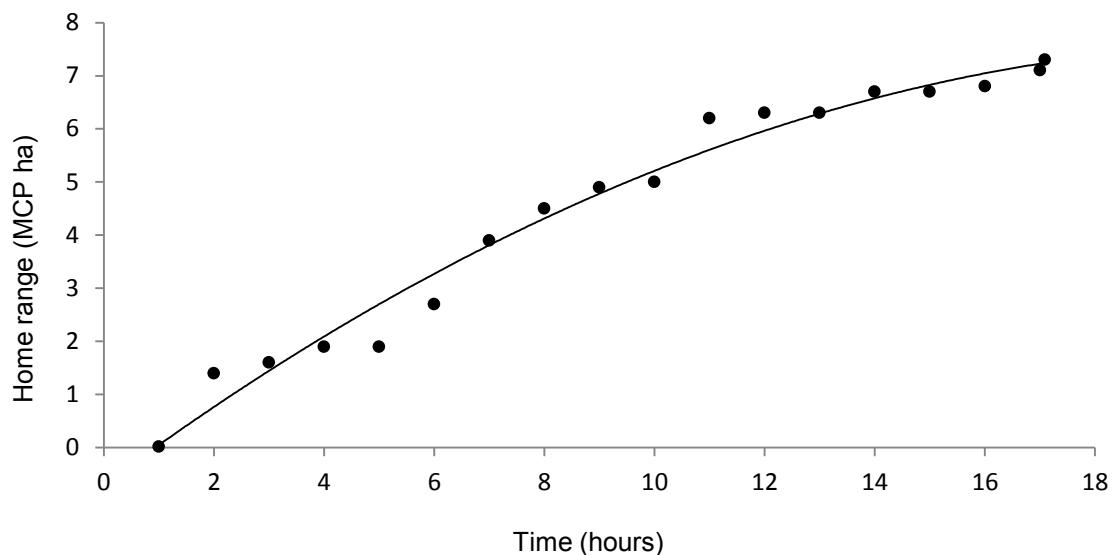


Figure 6.37: Minimum home range estimate for the Yellow Thornbill (*Acanthiza nana*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected over two days on three birds at Para Wirra Site B. The survey was conducted for sustained periods throughout each day to record the locations the birds stopped as they moved around their habitat. This resulted in 334 waypoint locations being recorded. The MCP estimated a home range of 7.3 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected July 2009.

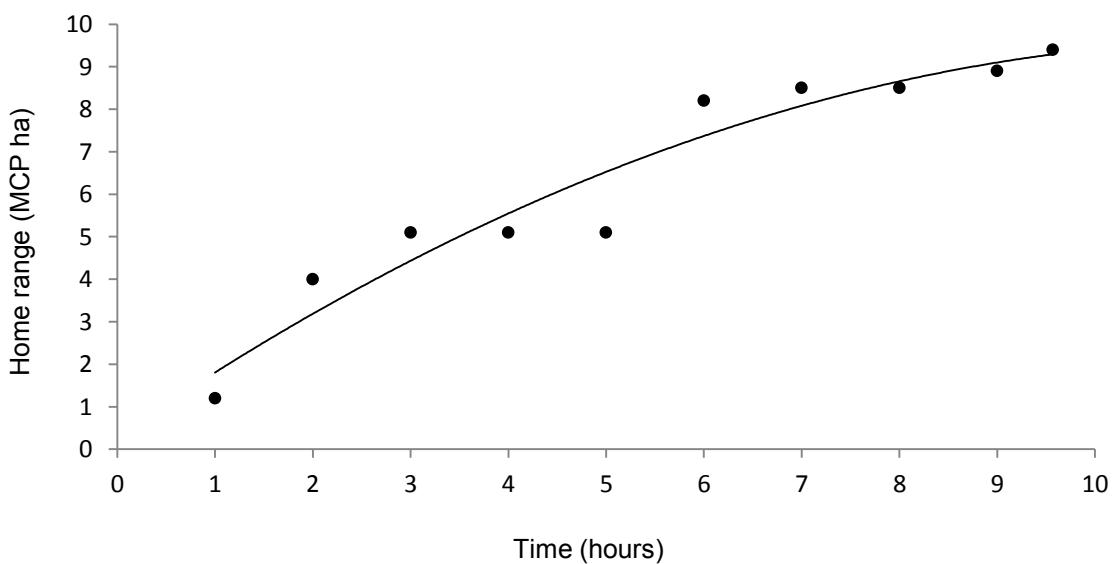


Figure 6.38: Minimum home range estimate for the Yellow Thornbill (*Acanthiza nana*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on three birds at Sandy Creek Site A. The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 269 waypoint locations being recorded. The MCP estimated a home range of 9.4 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2009.

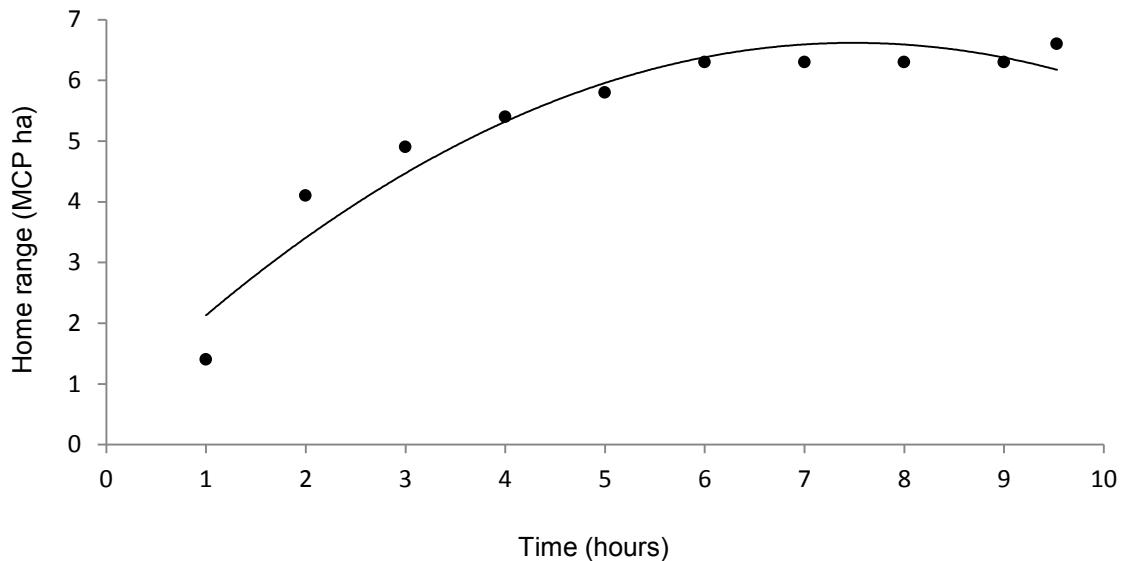


Figure 6.39: Minimum home range estimate for the Yellow Thornbill (*Acanthiza nana*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on 10 birds at Sandy Creek Site A. The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 346 waypoint locations being recorded. The MCP estimated a home range of 6.6 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2010.

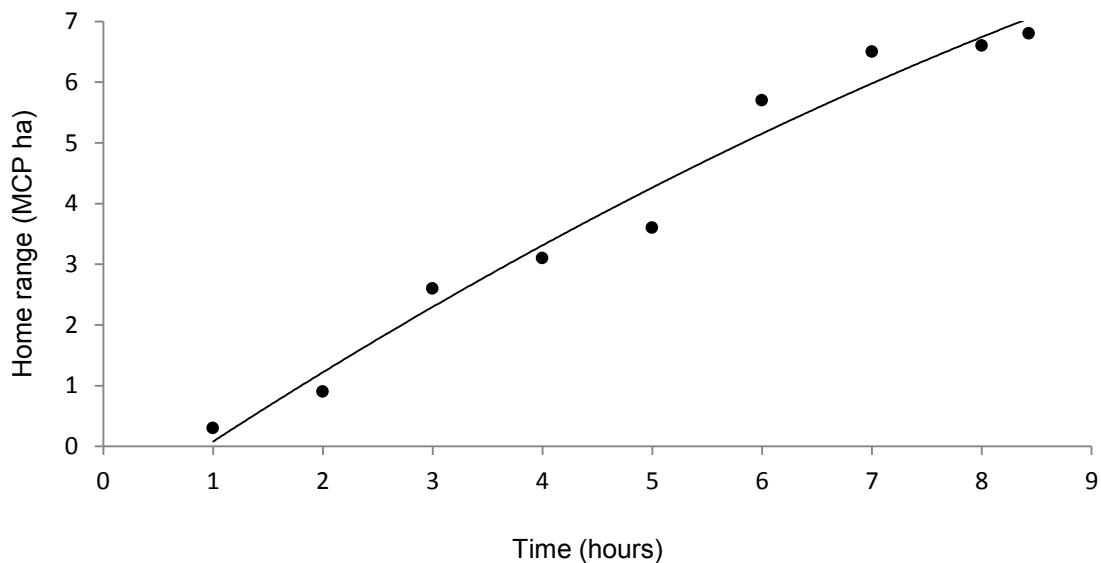


Figure 6.40: Minimum home range estimate for the Yellow Thornbill (*Acanthiza nana*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on three birds at Sandy Creek Site B. The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 261 waypoint locations being recorded. The MCP estimated a home range of 6.8 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. As indicated by the polynomial trendline, the minimum home range estimate was not achieved for these birds. Data collected June 2009.

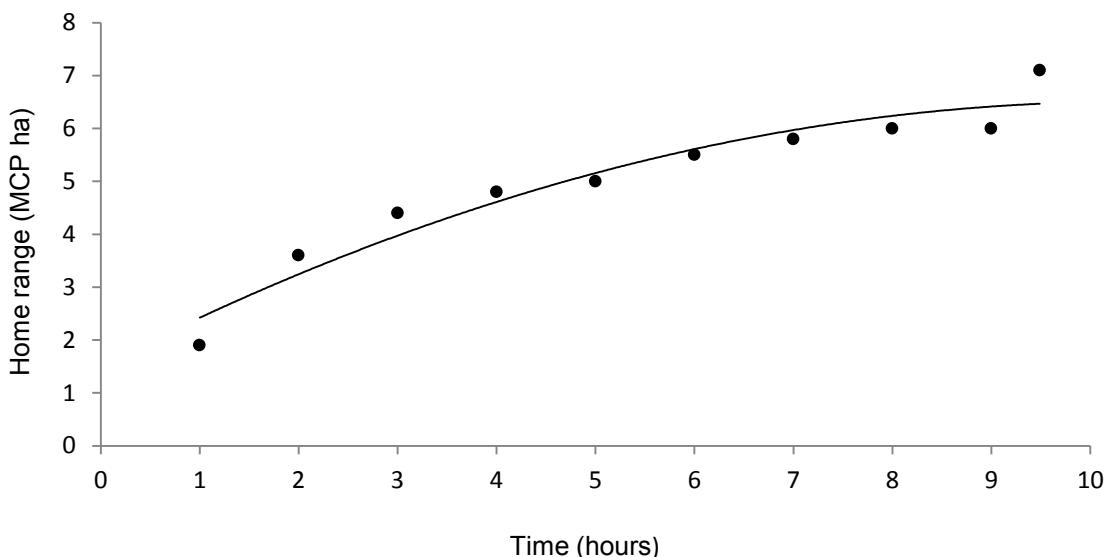


Figure 6.41: Minimum home range estimate for the Yellow Thornbill (*Acanthiza nana*) based on Minimum Convex Polygon (MCP) analysis using Arc-GIS (ESRI 1999 - 2006). These data were collected for one day on four birds at Sandy Creek Site B. The survey was conducted for sustained periods throughout the day to record the locations the birds stopped as they moved around their habitat. This resulted in 314 waypoint locations being recorded. The MCP estimated a home range of 7.1 ha during the survey. The polynomial trendline approximates the asymptote to gauge the reliability of the home range area measured. Data collected June 2010.

Discussion

Knowledge of a species home range requirement is highly relevant for revegetation strategies. An important and key finding of this study was the spatial requirements for individuals of several woodland bird species being quite large, ranging from several hectares to tens of hectares, during relatively short periods of intensive study. These short-term patterns of area use are likely to underestimate the area requirements of these birds over longer time periods however, which magnifies the results of this study. Seemingly, effective revegetation for birds belonging to several woodland species in this study should cover tens of hectares (e.g. for Hooded Robin, White-browed Babbler and Yellow-rumped Thornbill) to hundreds of hectares (e.g. for White-winged Chough). Unfortunately, many revegetation practices in Australia are commonly restricted to small holdings of only a hectare or so (Kimber *et al.* 1999; Paton *et al.* 2004).

The minimum home range estimates for several bird species were larger than expected (e.g. Rufous Whistler, White-browed Babbler and Willie Wagtail – details to follow).

Notably, home range sizes of woodland birds have been positively correlated with the number of birds that are contained within a home range (e.g. Noske 1991; Taylor 2003). This was not necessarily the case however in this study (e.g. White-browed Babbler).

Small groups of Brown Treecreepers hold all-purpose territories throughout the year. Like many species though, the spatial areas that they use between the breeding and non-breeding seasons differ. Breeding home ranges for this species have been estimated to be between 1.1 and 10.7 ha (Doerr and Doerr 2001; Cooper *et al.* 2002; Doerr and Doerr 2006). Noske (1991) also reported breeding pairs of Brown Treecreepers occupying home ranges spanning 6.8 – 7.3 ha, although quartets occupied larger areas of 19.6 – 20.5 ha. Home range estimates during the non-breeding season are generally larger and incorporate the breeding home range (Cooper *et al.* 2002). Year-round area estimates are not usually significantly larger than their breeding territories, however an extreme example is a colour-banded pair recorded using 50 ha in north-eastern New South Wales (Noske 1979).

These studies suggest the typical home range for the Brown Treecreeper should cover 3 – 6 ha and that home ranges over 10 ha may infer lower quality habitats (e.g. Cooper and Walters 2002; Doerr and Doerr 2006). This bird species is strongly influenced by specific habitat features, namely the density of tree hollows (Cooper and Walters 2002; Barker 2007) and fallen timber loads (Mac Nally *et al.* 2002b; Mac Nally 2006; Bennett *et al.* 2013). The non-breeding minimum home range estimates in this current study for small groups of Brown Treecreepers are consistent with the literature. However the birds at Altona Kies were using a relatively large area of 20.3 ha, which may suggest low habitat quality for these birds (see Chapter 8 for further discussion).

The Brown Treecreeper relies on habitat components that are only found within a mature habitat. This species then is unlikely to colonise re-established woodlands until adequate amounts of fallen timber have accumulated and the trees are sufficiently large to carry suitable hollows. For this species the prospect of providing additional habitat in the short to medium term appears limited, even if nest boxes are used to provide hollows as the trees mature.

Buff-rumped Thornbills live in conspecific groups and occupy a group territory that is segregated into several smaller territories by breeding pairs in late winter and early spring

(Blakers *et al.* 1984). During the non-breeding season, clans of 10 – 20 birds hold territories of around 13 – 15 ha (Bell 1982; 1985). During the breeding season, these same birds defend nesting territories of 0.25 – 3.5 ha that incorporates part of their annual home range (Bell and Ford 1986). The minimum home ranges found within the remnant woodlands around the Para Woodland Reserve are consistent with these other studies that were conducted in northern New South Wales.

The Hooded Robin is socially monogamous and is stoutly territorial towards intruding conspecifics, defending both breeding and feeding territories (Fitri and Ford 1997; 1998; Northeast 2007). Hooded Robin breeding home ranges have been estimated to be around 6 ha, expanding to approximately 12 – 30 ha at other times (Bell 1984; Fitri and Ford 1997; Northeast 2007). The larger home range estimates usually result when these birds have no immediate conspecific neighbours (Bell 1984). These findings are again consistent with the minimum home range estimates for the birds of this species in this study that used the remnant woodlands near the Para Woodland Reserve. However at Altona Chatterton, a pair of Hooded Robins used different areas on days with different weather conditions (Chapter 7). This suggests that combinations of habitats or habitat features are needed to meet these birds' territory requirement. This issue is explored more fully in Chapter 7.

In autumn, most Rufous Whistlers migrate north from southern breeding areas, only to return again in spring to breed as socially monogamous pairs (Frith 1969; Morris *et al.* 1981; Blakers *et al.* 1984; Paton *et al.* 1994). The same birds reportedly return to the same or nearby breeding territories that they used in the previous year (Marchant 1982; Bell and Ford 1987; Bridges 1994b; 1994a). These breeding territories are relatively small, typically 0.3 – 4.2 ha (Bridges 1994b; 1994a). Breeding home range boundaries can be clearly defined through territorial disputes with conspecific neighbours, as breeding territories often abut one another (Bell and Ford 1987; Bridges 1994b; 1994a). Home range areas outside of the breeding season can be significantly larger, although for most individuals these are not known because the birds depart their breeding areas. One pair that did not undertake a winter migration in Western Australia used around 32 ha during the non-breeding season, which incorporated their 3.2 ha breeding territory (Erickson 1950).

The pair of Rufous Whistlers surveyed at Sandy Creek however maintained larger breeding home ranges than commonly reported within the literature. This home range was surveyed for three days each in 2009 and again in 2010. The minimum home range from both surveys were 14.2 and 8.5 ha in area respectively; however for both surveys the minimum home range estimates were not achieved for these birds (i.e. the graphed asymptote was not achieved). In both years the surveyed pair was surrounded by other conspecific pairs. However most pairs stayed some distance apart, with the surveyed pair only interacting with the birds of one other adjoining home range (the same home range in both years) during my observations. This suggests that the surrounding home ranges at Sandy Creek were equally as large. In comparison, the 2.3 ha minimum home range that was surveyed at Altona Kies was closely surrounded by other conspecific pairs. The 5.9 ha minimum home range that was surveyed at Altona Landcare was spatially restricted, being on the edge of a woodland habitat. The Sandy Creek and Altona Kies home ranges were floristically similar. Both home ranges were dominated by overstorey Slender Cypress-pine (*Callitris gracilis*) and Pink Gum (*Eucalyptus fasciculosa*). Unlike Altona Kies though, the Sandy Creek site contained vegetation at lower storey levels. Pairs within both home ranges were constructing nests in readiness to breed, suggesting that both areas were of sufficient quality to allow for breeding. The dissimilarity in the minimum home range sizes between the two sites may have resulted from differences in habitat productivity levels and resultant differences in invertebrate prey biomass, although this hypothesis requires testing.

White-browed Babblers occur in groups of up to 15 or so birds (Ashton 1996; Taylor 2002). The social structure of this species is complex, with permanent home ranges often overlapping significantly with neighbouring and usually amicable conspecific groups (Brooker 1998; Recher and Davis 1998; Cale 1999; Taylor 2003). Non-breeding home ranges have been estimated to cover between 6.8 and 11.2 ha in other parts of Australia (Cale 1999; 2002; Taylor 2002). Within these studies there is evidence that group home ranges expand by 0.9 – 1.1 ha per bird (Taylor 2003) and that a 6.2 ha increase in habitat area is likely to accommodate an additional group of White-browed Babblers (Cale 2003). This current study however reports larger than expected non-breeding minimum home range estimates for the birds that were observed. Also, the largest estimate (36.4 ha) was occupied by only five birds. This suggests that the remnant habitats currently being used by these White-browed Babblers may be low quality for these birds.

During the breeding season, White-winged Choughs associate in small groups and occupy nesting territories of 10 – 50 ha (Rowley 1978; Heinsohn *et al.* 1988; Lewis 1997). During the non-breeding season these same groups may temporarily coalesce with each other and their home ranges usually overlap appreciably across much wider areas. Group home ranges may expand to as much as 1,000 ha (Rowley 1978; 1982; Heinsohn *et al.* 1988). The same breeding groups then return to their usual nesting territories in the following breeding season (Heinsohn 1992). For the majority of the time, groups of White-winged Choughs do not actively defend foraging territories (Heinsohn 1988). Seemingly aggressive interactions are common between conspecifics however, but these amount to no more than elaborate displays where there are no clear winners (Rowley 1978; Heinsohn 1988; this study). In this study, the estimated area requirements for the groups of White-winged Choughs during the breeding (14 ha) and non-breeding seasons (152 and 196 ha) were consistent with the literature.

The Willie Wagtail usually occurs singly, in pairs or in small family groups (Gibson 1977; Carpenter and Matthew 1997; Hornsby 1997). Home ranges are actively defended throughout the year, which reportedly cover 1.0 – 3.3 ha (Plumb 1948; McFarland 1984; Cameron 1985; Goodey and Lill 1993). The minimum home range estimates for the Willie Wagtails in this study ranged up to around 12 ha, which far exceeded what is expected for this species. Again these larger minimum home ranges may reflect poorer quality habitat for these birds.

Small flocks of Yellow-rumped Thornbills (3 – 30 birds) reportedly wander locally; having been recorded up to 1 km from banding sites (Green 1989; Saunders and Ingram 1995). The minimum home range estimate in this study of up to 25 ha would not seem excessive for these birds; however it is important to note that this was achieved in just three days.

The Yellow Thornbill has been described as resident but locally mobile (Morris *et al.* 1981; Paton *et al.* 1994). The estimated home range for this species is poorly known. Based on this current study however, around 1.6 ha of suitable habitat is required per bird for each bird within a cohesive group. This equates to 0.6 birds ha⁻¹, which compares with other studies. For example, 1 bird ha⁻¹ in Keast (1985a), 0.02 – 0.56 birds ha⁻¹ in Woinarski (1984) and 0.11 – 0.69 birds ha⁻¹ in Wood (1995). A density of 0.5 – 1 bird per hectare would equate to individual birds requiring 1 – 2 ha of suitable habitat.

Based on previous studies, the habitat areas required by numerous woodland bird species have been approximated. These estimates range from as little as 3 – 4 ha for the Buff-rumped Thornbill, Willie Wagtail and Yellow-rumped Thornbill to over 100 ha for the Hooded Robin (Watson *et al.* 2001). Based on this current study, these lower estimates do not appear to provide adequate habitat area for these species. Results from this study would suggest that much larger patches of several hundreds of hectares are needed to be re-established to sustain viable populations of these species. This would provide habitat areas that would better accommodate multiple home ranges for the majority of woodland bird species. Where large areas are not available, or when funding to do on-ground work is limited, then the revegetation works should be planted in the vicinity of remnant habitat. Larger habitat areas and proximity to remnant vegetation both correlate with improved habitat outcomes for many species, including some of conservation concern (e.g. Fisher and Goldney 1998; Kavanagh *et al.* 2007). In support, and as previously noted in Chapter 2, Westphal *et al.* (2007) formulated that for 22 bird species within the Mount Lofty Ranges, revegetation patches need to range in size from an average 780 to 4,010 ha.

The planned revegetation for the 321 ha Para Woodland Reserve goes some way in achieving a substantially revegetated area, but clearly even more needs to be done. Based on the needs of individual birds or bird units (i.e. pairs and/or cohesive groups), then at best the maximum number of home ranges that could be established on this Reserve is typically 1 – 2 to 10 – 20, depending on the species. This assumes that the short-term minimum home ranges that were measured in this study equate to the overall areas that are needed throughout the year by these birds (which is probably not the case). Also assumed is that all the bird species can be accommodated within the habitat areas that are to be produced on the Reserve. It is unlikely though that this will be the case, given that some species use different habitat types more than others (Chapters 3 and 5). Although some species can co-exist within the same habitat area, not all species can and/or will. This topic will be explored more fully in the following chapters, specifically investigating the variable nature of habitat use by birds of the nine declining species (Chapter 7), the fine-scale (within patch) habitat needs of individual birds from these species (Chapter 8), and how fine-scale habitat heterogeneity influences habitat use by co-inhabiting birds of these different species (Chapter 9).

In regards to the large home range requirements of bird species, of the remaining 10% or so pre-European native vegetation that still exists across the Mount Lofty region, only an estimated five percent is considered to be intact (Bushcare 2003). This suggests that not only is there a lack of habitat, it is packaged in many small areas that are unlikely to sustainably support many declining woodland bird species. The 30% requirement for intact natural habitats (see Chapter 2) is not met across the Mount Lofty region, so further species losses and collapse of viable populations can be expected (Andrén 1994; Fahrig 1997; Radford *et al.* 2005; Lowe 2010). Reaching a minimum target of 30% habitat cover may help to alleviate the predicted species losses within this region.

As a final point, the higher than expected minimum home range estimates in this study may have resulted from the drought conditions that prevailed during the course of this project. Drought imposes environmental stresses which impact upon bird life (Barnard 1917; Traill *et al.* 1996; Woinarski and Catterall 2004). For insectivorous bird species, one ramification of low (or no) rainfall may be a lower and/or altered invertebrate prey biomass. Drought has been shown to suppress the emergence of some invertebrate groups (e.g. Alvarez *et al.* 1999). Consequently larger habitat areas may be required by birds during these trying times, in order for them to attain sufficient quantities of invertebrate prey. Nevertheless, drought conditions are a natural and expected phenomenon across southern Australia. Should such perturbations result in increased area requirements for birds, then this must be factored into their life-time habitat needs. This again suggests that much more habitat area is needed per pair or functional group than has previously been reported. If true, the recommended 30% cover of natural habitat areas across the Mount Lofty region may be an underestimate of what is required to adequately support viable populations of species over the long-term.

Chapter 7

Variable habitat use within individual home ranges

Introduction

Bird ecologists have a long history of attempting to understand the principles of habitat selection by species (e.g. Cody 1981). Here various spatial scales need to be considered, ranging from the landscape to the micro-patch level. Habitat selection is then described as a hierarchical series of decisions that are made by birds (Johnson 1980; Hutto 1985).

Initially, a geographic location is selected at the macro-scale. This is akin to bird species preferentially selecting one or two woodland types within a region, as indicated in Chapter 3. Birds then choose particular habitat areas. In agreement, habitat selections by birds within broad habitat types were revealed in Chapter 5. Birds then ultimately select (and use) micro-habitats that contain desirable habitat features that contrast with the habitat areas in which they are embedded. Disproportionate use of habitats then result when individual species respond differently to various habitat elements at these finer-scales (Hutto 1985; Block and Brennan 1993).

However, habitat selection can be strongly influenced by an array of processes and events at various spatial and/or temporal scales, as perceived by the organism (Kotliar and Wiens 1990; Orians and Wittenberger 1991). Spatially, habitat use will vary based on the changing needs of individual birds and bird species. These include the need for protective cover, to seek optimum foraging habitats, and to source favoured nesting locations to reproduce (Mysterud and Ims 1998; Mackey and Lindenmayer 2001). Variability in habitat use can also be related to both long-term and short-term temporal scales. These include responses to predictable processes like seasonality and stochastic events such as prevailing weather conditions (Benton *et al.* 2003).

A challenge then exists when researchers call for greater information on the fine-scale habitat use by species (e.g. Lindenmayer *et al.* 2002; Paton *et al.* 2004; Prober and Thiele 2005; Lindenmayer *et al.* 2008; Mac Nally 2008; Paton *et al.* 2010). The conundrum is

that habitat attributes that are deemed to be important for certain species may only be important during the times the birds were being surveyed. This is especially so considering much of what is known of species-habitat relationships is biased towards fine-weather data collection. Furthermore, predictive modelling of species distributions use biased field observations to infer which environmental variables help to predict the presence of species within habitats (Guisan and Zimmermann 2000). The problem here is that most data sets used to construct these models result from sampling that is restricted to particular times of the day and are conducted over spatially limited scales, so interpretations of the fitted models can be decidedly restricted (Guisan and Thuiller 2005). Most data sets for bird-habitat relationships also only record species presence-absence or presence-only data. This type of information merely reports on general patterns, not the underlying processes that may help explain the actual habitat associations by species. This can result in considerable uncertainty when attempting to predict the distribution of a species (e.g. Brotons *et al.* 2004; Wilson *et al.* 2005).

This study asks the question, do birds demonstrate variable habitat use within the confines of their individual home ranges? Specifically, the study will focus on identified areas of concentrated habitat use (i.e. “hot-spots”) using sustained periods of observation. A hot-spot is defined as a patch of habitat within a home range that was used disproportionately more than other areas within that same home range (i.e. “cold-spots”). Variability in habitat use by birds will be considered over various temporal scales; namely during a single day, over multiple days and between years. Any significant variability in habitat use revealed by this study has direct ramifications for survey designs, especially those that seek to reveal fine-scale habitat needs of species and to infer the preferred habitats of species using distribution models.

Methods

Nine woodland bird species were used for this study (refer to Chapter 1 for the criteria that was used to select the species that were surveyed). The bird species were the Brown Treecreeper (*Climacteris picumnus*), Buff-rumped Thornbill (*Acanthiza reguloides*), Hooded Robin (*Melanodryas cucullata*), Rufous Whistler (*Pachycephala rufiventris*), White-browed Babbler (*Pomatostomus superciliosus*), White-winged Chough (*Corcorax*

melanorhamphos), Willie Wagtail (*Rhipidura leucophrys*), Yellow-rumped Thornbill (*A. chrysorrhoa*) and Yellow Thornbill (*A. nana*).

The collection method for the GPS waypoint data used for this study has been previously detailed in Chapter 6 (to determine minimum home range sizes). The data collection method was also designed to identify habitat use by individuals of these nine bird species within the confines of their home ranges. Each bird survey was conducted for sustained periods between dawn and dusk for one or more days. This allowed the subject birds ample time to repeatedly visit more favoured areas of their habitat. As such, the information is not biased towards the detection of more obvious bird behaviours. Instead, these data are an accurate account of what the birds actually did during the extended periods that they were being observed.

The bird surveys were performed between April and September 2009, with 11 of these home ranges being re-surveyed in 2010 / 2011. Typically, the bird surveys were conducted during the “winter period”, as birds tend to forage for larger proportions of daylight hours during this period, with days being shorter and resources more limited, compared to the “summer” period (Recher *et al.* 1983). Limited foraging resources also produce minimal overlap in habitat use between species, which may not necessarily be the case during the warmer months (Recher 1989). Surveys were usually conducted during fine sunny weather, which had more to do with the extremely mild winter due to prevailing drought than personal choice. However, opportunistic surveys of habitat use by a pair of Hooded Robins were made during rare periods of wintery weather in 2009 and again in 2010.

The waypoint data that were collected for each home range were used to identify disproportionate habitat use by the inhabiting birds. Disproportionate use of habitats was identified via Point Density analysis using ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006). Point Density was chosen as it specifically calculates the density of point features, without applying a smoothing factor (smoothly tapered surfaces depicting density of points within fixed areas). Smoothing factors are linked with kernel-density estimations that are commonly used for home range estimations and to define probabilities of area use within home ranges (e.g. Aebischer *et al.* 1993; Shawe-Taylor and Cristianini 2004). However, the recommended (although arbitrary) use of least square cross-validation to calculate the

smoothing factor for this type of analysis is not ideal, as it can produce highly variable results (Hemson *et al.* 2005; Horne and Garton 2006; Row and Blouin-Demers 2006).

The application of Point Density analysis can also have its limitations however. In this study, there was a need to consider both the spatial and temporal arrangements of the waypoint data. For example as detailed in Chapter 6, the survey method involved recording the location of each individual tree as that tree was being used by the bird/s being surveyed, together with the time the bird/s spent using that tree. Now imagine a habitat setting where an isolated mature South Australian Blue Gum (*Eucalyptus leucoxylon*) grew adjacent to a stand of mature Slender Cypress-pine (*Callitris gracilis*). Consider a pair of Hooded Robins occupying this site. The foraging strategy of the Hooded Robin involves moving between the various low lateral branches, as this species scans the ground for invertebrate prey. In this scenario, a cluster of GPS waypoints would be created around the location of the group of Pines, as the birds repeatedly moved between these closely growing trees. In comparison, only a relatively few waypoints would be assigned to the location of the Blue Gum, as the birds would merely move along the extensive branches of this single tree to scan new areas of the ground below. The resultant Point Density analysis based on the spread of these waypoint locations would depict the stand of Pines as being a hot-spot compared to the Blue Gum (i.e. more waypoints would be assigned to the stand of Pines). However in this real-life scenario, the Robins spent much more time foraging from the branches of the Blue Gum that broadly extended over the ground litter below.

The depiction that the stand of Pines represented a hot-spot was merely the product of the growing arrangements of these two tree species and the manner in which the waypoints were assigned in this study. This would create a false-positive error (i.e. reporting a positive finding that actually did not exist), that the Pines were being used more than the Blue Gum. Conversely, Point Density analysis based solely on the time the bird/s spent per waypoint may produce equally skewed results. For example, a surveyed Brown Treecreeper that rested for over an hour in a location that was not used again by this species during the survey.

To limit the adverse influence either of these situations would impose on the analyses, the waypoint location data and the time spent per waypoint data were combined for each home

range. To effectively combine the spatial dataset with the temporal dataset required both datasets to be first reclassified to a common measurement. This was achieved using the Spatial Analysis Tool in ArcMap. Weighted Overlay in ArcMap was then used to combine these two reclassified datasets. An influence ratio of 50:50 for the two reclassified datasets was used to produce the single output. The assumption here was that the waypoint location and the time spent per waypoint were equally important when defining differential habitat use by the birds. The default pixel size of 5 m and circular neighbourhood radius of 20 m (used to calculate density values) were applied.

In this study, Natural Breaks (Jenks) classification in ArcMap was used to identify natural class breaks inherent in the data. This procedure is based on Jenk's Natural Breaks algorithm (Jenks 1967) and emphasises major clusters of information in the data set. It does this by assigning data to classes using an iterative algorithm that minimises the variation within classes while maximising the variation between classes. Class breaks are identified where maximum differences exist between groups containing similar values.

This study used two Natural Breaks (Jenks) to identify two classes, representing areas of high habitat use (hot-spots) and areas of low habitat use (cold-spots). This produced output rasters that revealed clear variations in habitat use within the home ranges of the birds that were observed. A measure of usage frequency within territory boundaries instils more confidence that all parts of a habitat area is available to its inhabitants and that some areas are preferentially used over others (Jones 2001).

The Point Density calculations for each home range were then viewed as histograms, which in all cases revealed highly right skewed distributions (e.g. Fig. 7.1). These distributions confirmed that these data were biased towards many lower values that represented cold-spots, with only a relatively few higher values that represented hot-spots.

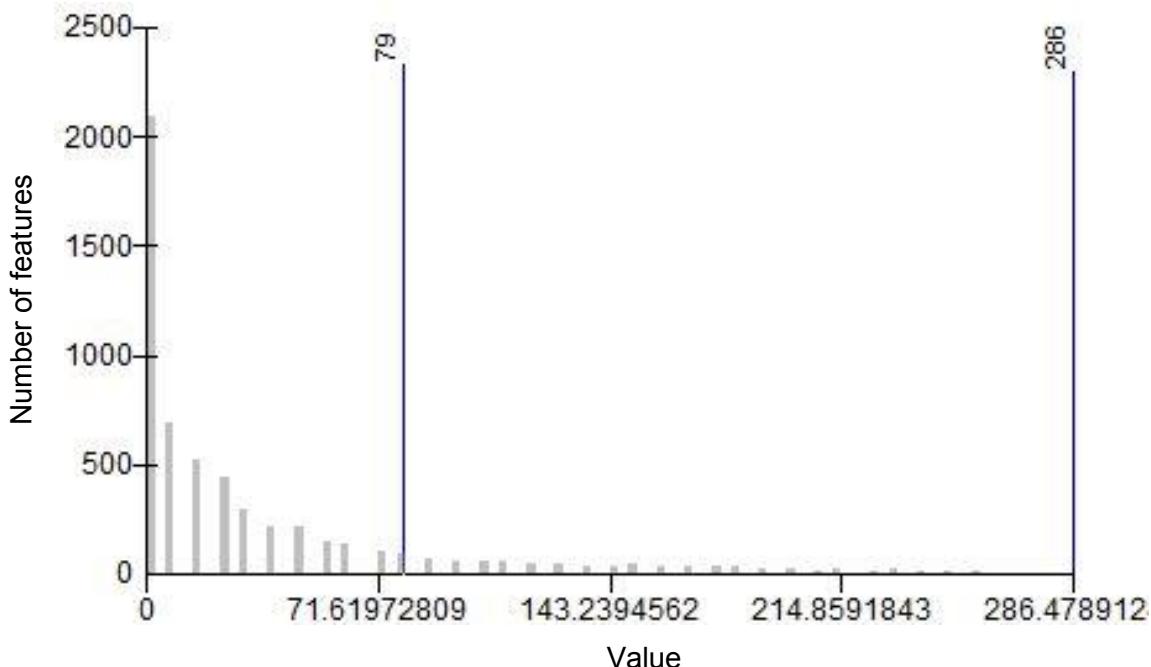


Figure 7.1: Histogram of habitat use by Rufous Whistlers (*Pachycephala rufiventris*) within the Sandy Creek Conservation Park, using ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006). These data are derived from Point Density analysis that was based on the equal combination of two reclassified datasets. These datasets were for the spatial waypoint location data and the temporal dwell times per waypoint data. Natural Breaks (Jenks) using two classes classified two natural groupings that was inherent in the data. Natural Breaks classes identify where maximum differences exist between groups containing similar values. The Class Break value at 79 (purple line) distinguishes areas of low habitat use or “cold-spots” to the left from the highly skewed areas of high habitat use or “hot-spots” to the right. The Class Break value at 286 represents the highest value and the end of the data series. The values along the x-axis are a computation of the reclassified and then combined data for the waypoint locations and dwell times per waypoint. These values were computed to classify and group these data only and the values have no relevance to the original source data.

The methods employed to measure and assess the habitat characteristics within hot-spots and cold-spots are discussed in full in the next chapter. Relevant to this chapter though, each hot-spot and cold-spot was assessed at the scale of $30\text{ m} \times 30\text{ m}$ (referred to from here as “plots”). Selections of these data are presented in this chapter to assist in qualifying and quantifying variable habitat use by the bird species that were observed.

The Hooded Robin data relating to habitat use in opposing weather conditions (fine weather versus inclement weather) were not normally distributed. For the comparative analysis, the total overstorey abundance data (see next chapter for details) required Log₁₀ transformation to be able to be compared using two-tailed *t*-tests. The tree species

abundance data defied transformation, so these data were compared using the Kruskal-Wallis tests and the corresponding chi-square statistic (χ^2).

Buff-rumped Thornbill foraging heights were compared for the two annual surveys. A chi-square analysis was used to test for agreements and differences in the foraging heights between the two survey periods at the shrub (< 1 m), midstorey (1 – 4 m) and overstorey levels (> 4 m). Yates correction for continuity was applied when there was only one degree of freedom (Zar 2010). Any statistical differences in the foraging heights displayed by this species between the two survey periods was then identified by subdividing the chi-square goodness of fit (Zar 2010). This procedure identified which storey level/s contained significant disagreement between the observed and expected frequencies.

As detailed in Chapter 6, at least one home range of a pair or cohesive group for each species of the nine species that were intensively surveyed in 2009, were re-surveyed in 2010 / 2011. This amounted to 11 home ranges being re-surveyed. Due to time constraints imposed on this project, only the spatial and temporal uses of habitats were recorded, and not the corresponding habitat survey (which was performed during the 2009 surveys).

These data were collected to investigate the extent to which the high use areas identified during the 2009 survey were being used by the birds during the 2010 / 2011 re-surveys. Comparing the spatial use of habitat areas within a home range between different survey days and between different survey periods would seem at first to be conceptually easy. However, ArcMap which was used to generate these data does not have the capacity to perform this function. Instead, this analysis required the use of Hawth's tools in ArcMap to generate a set of random points within each of the relevant Minimum Convex Polygons (MCP). An enforced minimum distance of 20 m between points avoided spatial autocorrelation, allowing for eight points per hectare to be used within each home range. Using the Sample tool in ArcMap, sample values from an equal number of actual waypoint locations (compared to the number of random points used) were taken from both the 2009 and 2010 / 2011 datasets for all 11 re-surveyed home ranges. These were then compared with the values of the random set of points for the corresponding home range. The resultant values were subjected to regression analysis, with the Coefficient of Determination (R^2) used as the indicator of agreement of habitat use between the 2009 and 2010 / 2011 surveys. This procedure was also used when comparing habitat use over

multiple days within the same survey period for the Hooded Robin and White-browed Babbler.

Results

The average time the birds were surveyed per day was $9:06 \pm 0:01$ h:min. During this time, survey data were only ever recorded whilst visual contact of the surveyed birds was being maintained, which averaged $70 \pm 3\%$ of the total survey time (Table 7.1). Generally most survey time was lost while following the birds after they had moved location. Survey time was also lost within home ranges that contained numerous barbed-wire fences (e.g. Altona Kies and Sandy Creek; Table 7.1). In the Para Wirra region, time was lost negotiating the steep terrain and dense vegetation, while ensuring not to flush the birds that were being surveyed (Table 7.1). Generally the surveyed birds were easy to follow throughout the day and were readily re-located after visual contact had been lost after the birds had moved location.

The time all birds spent per bird-stop location averaged $2:21 \pm 0:09$ min:sec (from 18,359 waypoints over 117 survey days). This ranged from an average $1:35 \pm 0:05$ min:sec for the Yellow Thornbills that were surveyed (from 2,267 waypoints over 9 survey days) to an average $4:22 \pm 0:09$ min:sec for the White-winged Choughs that were surveyed (from 2,354 waypoints over 20 survey days; Table 7.2). The time the birds spent at each bird-stop location may have influenced the detectability of some less conspicuous species. For example, individual Hooded Robins at Altona Kies spent on average $1:35 \pm 0:05$ min:sec per bird-stop location (from 510 waypoints over 5 survey days; Table 7.1). During the ten hectare survey conducted on this property in 2008 (Chapter 3), only one male Hooded Robin was detected, being on the fourth of five surveys. However the 2009 species-specific survey in this area identified a family group of three Hooded Robins that occupied a minimum home range estimated at 17.1 ha. This home range incorporated 7.9 ha (or 79%) of the original ten hectare survey site (Fig. 7.2).

Table 7.1: Survey efficiency as a percent of time visual contact was maintained and the activities of birds recorded throughout the day. Surveys were conducted over multiple days between dawn and dusk. Surveys averaged around nine hours per day, in which the birds were surveyed around 70% of the time.

| Bird species | Location | Date (month/year) | Survey time (h:min) | Time per waypoint (min:sec) | Survey efficiency (%) |
|-------------------------|-----------------------------|----------------------|---------------------------|-----------------------------------|-----------------------------|
| Brown Treecreeper | Altona Chatterton | 6-7/2009 | 27:24 | 2:18 | 81 |
| Brown Treecreeper | Altona Kies | 4/2009 | 30:43 | 2:51 | 75 |
| Brown Treecreeper | Altona Torbreck | 6-7/2009 | 28:52 | 2:18 | 77 |
| Brown Treecreeper | Altona Torbreck | 6/2010 | 18:53 | 3:15 | 81 |
| Buff-rumped Thornbill | Para Wirra - Site A | 8/2009 | 27:04 | 1:43 | 49 |
| Buff-rumped Thornbill | Para Wirra - Site B | 7/2009 | 28:09 | 1:40 | 49 |
| Buff-rumped Thornbill | Para Wirra - Site B | 6-7/2010 | 24:47 | 1:55 | 55 |
| Hooded Robin | Altona Chatterton | 7/2009 | 18:27 | 1:45 | 84 |
| Hooded Robin | Altona Chatterton - storm | 7/2009 | 19:23 | 1:22 | 71 |
| Hooded Robin | Altona Chatterton | 6/2010 | 18:00 | 2:34 | 66 |
| Hooded Robin | Altona Chatterton - showers | 6/2010 | 8:15 | 1:20 | 61 |
| Hooded Robin | Altona Kies | 5/2009 | 41:46 | 1:35 | 32 |
| Hooded Robin | Sandy Creek | 8/2009 | 36:20 | 1:22 | 30 |
| Rufous Whistler | Altona Kies | 9/2009 | 18:26 | 2:10 | 60 |
| Rufous Whistler | Altona Landcare | 9/2009 | 25:32 | 2:25 | 44 |
| Rufous Whistler | Sandy Creek | 8/2009 | 28:13 | 1:52 | 54 |
| Rufous Whistler | Sandy Creek | 10/2010 | 30:10 | 2:03 | 53 |
| White-browed Babbler | Altona Chatterton | 5-6/2009 | 62:04 | 3:26 | 90 |
| White-browed Babbler | Altona Kies | 4-5/2009 | 56:24 | 3:33 | 89 |
| White-browed Babbler | Altona Kies | 6/2010 | 49:03 | 3:23 | 95 |
| White-browed Babbler | Sandy Creek | 8/2009 | 28:06 | 2:16 | 51 |
| White-winged Chough | Para Wirra | 8/2009 | 41:09 | 4:27 | 91 |
| White-winged Chough | Sandy Creek | 6/2009 | 75:36 | 4:04 | 94 |
| White-winged Chough | Sandy Creek | 6-7/2010 | 65:09 | 4:34 | 92 |
| Willie Wagtail | Altona Chatterton | 7/2009 | 26:47 | 2:53 | 73 |
| Willie Wagtail | Altona Chatterton | 2/2011 | 9:01 | 4:31 | 66 |
| Willie Wagtail | Altona Kies - Site A | 5/2009 | 17:27 | 3:11 | 74 |
| Willie Wagtail | Altona Kies - Site B | 5/2009 | 18:28 | 2:21 | 76 |
| Willie Wagtail | Altona Kies - Site B | 6/2010 | 18:39 | 2:24 | 89 |
| Willie Wagtail | Sandy Creek | 8/2009 | 14:28 | 1:31 | 53 |
| Yellow-rumped Thornbill | Altona Chatterton | 5/2009 | 28:16 | 2:04 | 71 |
| Yellow-rumped Thornbill | Altona Chatterton | 6/2010 | 22:41 | 2:09 | 72 |
| Yellow-rumped Thornbill | Altona Kies | 5/2009 | 17:27 | 2:37 | 84 |
| Yellow Thornbill | Altona Chatterton | 6/2009 | 9:32 | 1:45 | 84 |
| Yellow Thornbill | Altona Kies | 5/2009 | 8:36 | 2:07 | 87 |
| Yellow Thornbill | Para Wirra - Site A | 8/2009 | 9:58 | 1:34 | 59 |
| Yellow Thornbill | Para Wirra - Site B | 7/2009 | 17:09 | 1:29 | 48 |
| Yellow Thornbill | Sandy Creek - Site A | 6/2009 | 9:57 | 1:24 | 63 |
| Yellow Thornbill | Sandy Creek - Site A | 6/2010 | 9:53 | 1:24 | 82 |
| Yellow Thornbill | Sandy Creek - Site B | 6/2009 | 8:43 | 1:31 | 75 |
| Yellow Thornbill | Sandy Creek - Site B | 6/2010 | 9:49 | 1:22 | 73 |

Table 7.2: Average time spent per waypoint location for individuals of nine woodland bird species within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. The species are arranged in increasing order from those which spent the least time per waypoint location to those that spent the most time per waypoint location. Data collected April 2009 – February 2011.

| Bird species | Time spent per waypoint (min:sec) | # waypoints | # days | # home ranges |
|-------------------------|-----------------------------------|-------------|--------|---------------|
| Yellow Thornbill | 1:35 ± 0:05 | 2267 | 9 | 8 |
| Hooded Robin | 1:40 ± 0:12 | 2668 | 16 | 6 |
| Buff-rumped Thornbill | 1:46 ± 0:05 | 1389 | 9 | 3 |
| Rufous Whistler | 2:07 ± 0:07 | 1542 | 11 | 4 |
| Yellow-rumped Thornbill | 2:17 ± 0:10 | 1393 | 8 | 3 |
| Brown Treecreeper | 2:40 ± 0:14 | 1918 | 11 | 4 |
| Willie Wagtail | 2:48 ± 0:25 | 1804 | 12 | 6 |
| White-browed Babbler | 3:10 ± 0:18 | 3024 | 21 | 4 |
| White-winged Chough | 4:22 ± 0:09 | 2354 | 20 | 3 |

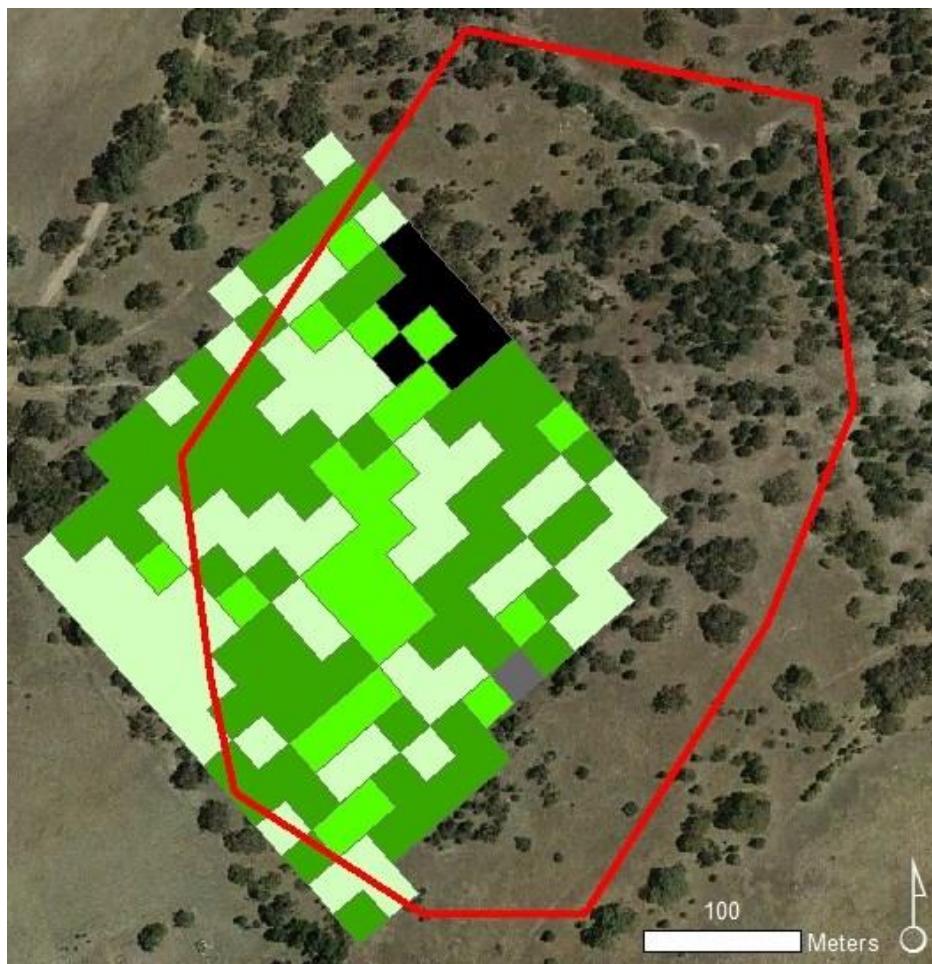


Figure 7.2: Minimum home range (red coloured Minimum Convex Polygon) for a family group of three Hooded Robins (*Melanodryas cucullata*) at Altona Kies. The coloured chequered area is the ten hectare site that was previously surveyed five times to identify the bird species within this open grassy woodland habitat. The coloured chequered areas represent different habitat structures, as identified in Chapter 5. Despite this home range covering 79% of the ten hectare survey site, only the male bird was detected (once) during the ten hectare surveys. This record occurred on the fourth of five surveys.

Some birds used different parts of their habitat at different times of the day. For example, over two consecutive days at Altona Chatterton, a pair of Hooded Robins used distinctly different habitat areas during the course of the day (Fig. 7.3). The Hooded Robin is fiercely territorial and will not tolerate conspecifics intruding into their territory. This pair of Hooded Robins at Altona Chatterton did not interact with any other conspecifics during this survey, so it was confidently assumed that the same birds were being surveyed on both days. The extent of spatial overlap in the habitat areas used by this pair ranged from 0.8% to 1.1% between three temporal periods; being 1) dawn to early morning, 2) mid-morning to early afternoon, and 3) late afternoon to dusk (Table 7.3).

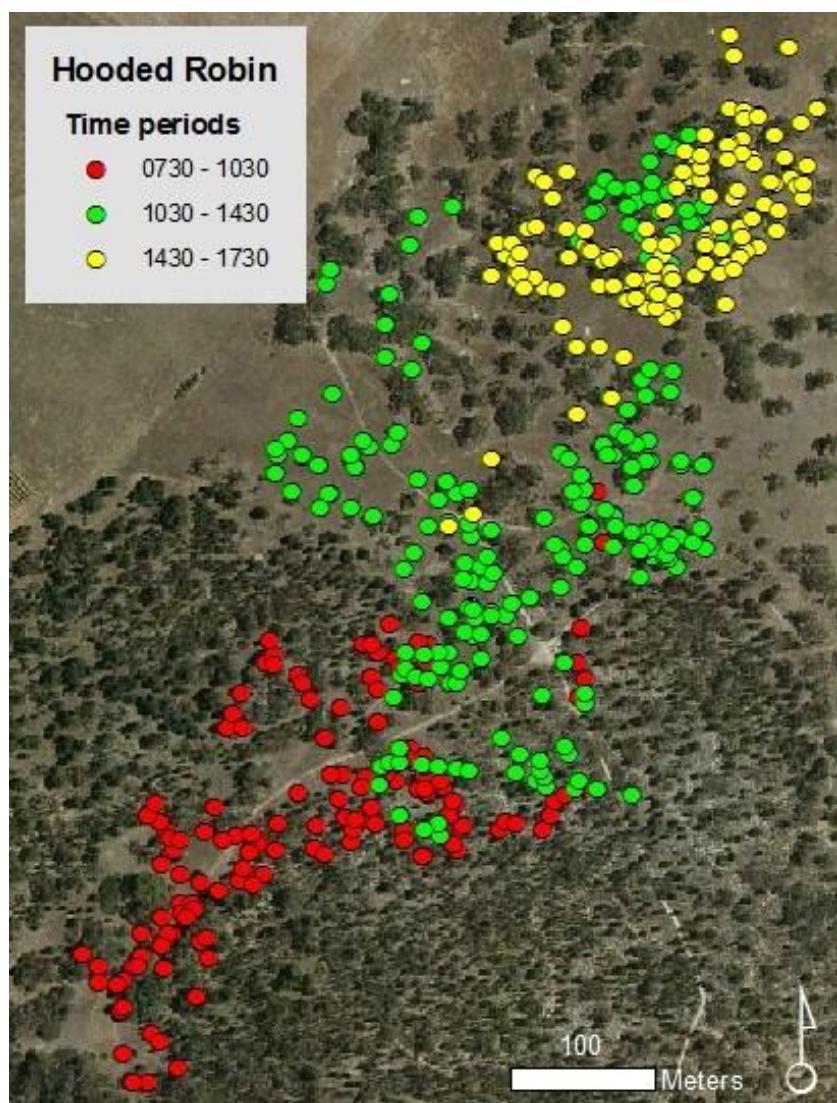


Figure 7.3: A pair of Hooded Robins (*Melanodryas cucullata*) surveyed at Altona Chatterton over two consecutive days, displaying both temporal and spatial disproportionate habitat use. Each point represents a GPS location where the birds perched ($n = 529$ waypoints). Each point is colour-coded to represent the time the data was recorded, which has been divided into three time increments throughout the day (military time used). Data collected July 2009.

Table 7.3: Quantitative analysis of temporal variability in habitat use within the minimum home range of a pair of Hooded Robins (*Melanodryas cucullata*) at Altona Chatterton. These data were collected over two consecutive days and have been proportioned into approximate thirds between the hours of 0700 and 1730. For this analysis, the area used during the 0730-1030 period was compared against the combined areas that were used during the 1030-1430 and 1430-1730 periods; the area used during the 1030-1430 period was compared against the combined areas that were used during the 0730-1030 and 1430-1730 periods; and, the area used during the 1430-1730 period was compared against the combined areas that were used during the 0730-1030 and 1030-1430 periods. Comparing the spatial use of habitat areas between these three temporal periods would seem to be conceptually easy. However, ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006) which was used to generate these data (refer Figure 7.3) does not have the capacity to perform this function. Instead, this analysis required the use of Hawth's tools using ArcMap to generate a set of random points across these birds' home range. An enforced minimum distance of 20 m between points avoided spatial autocorrelation, allowing for 153 random points to be generated. Using the Sample tool in ArcMap, sample values were then taken from 153 actual waypoint locations. These actual values were then compared with the values of the random set of points. The resultant values were subjected to regression analysis, where the R^2 values were calculated for the agreement in habitat use between each time increment. The extents of spatial overlap in habitat use between these three time periods are given by the Coefficient of Determination (R^2). The extent of spatial overlap in the habitat areas used by this pair ranged from 0.8 to 1.1% between dawn to early morning (0730 hrs - 1030 hrs), mid-morning to early afternoon (1030 hrs - 1430 hrs) and late afternoon to dusk (1430 hrs - 1730 hrs). Data collected July 2009.

| Time periods (military time) | | |
|------------------------------|-------------|-------------|
| 0730 - 1030 | 1030 - 1430 | 1430 - 1730 |
| R^2 | R^2 | R^2 |
| 0.008 | 0.008 | 0.011 |

Some birds also used different habitat areas on different days. For instance, a cohesive group of five White-browed Babblers were surveyed throughout the day for seven days as they left at dawn and returned at dusk to the same roost-nest location. The area of habitat used by these birds over these seven days totalled 36.4 ha, however the birds tended to use different areas on each day (Fig. 7.4, Table 7.4). The extent of spatial overlap in the habitat areas that were used from day to day varied from around 16% ($R^2 = 0.159$) to entirely new habitat areas being used ($R^2 = 0.000$; Table 7.4).

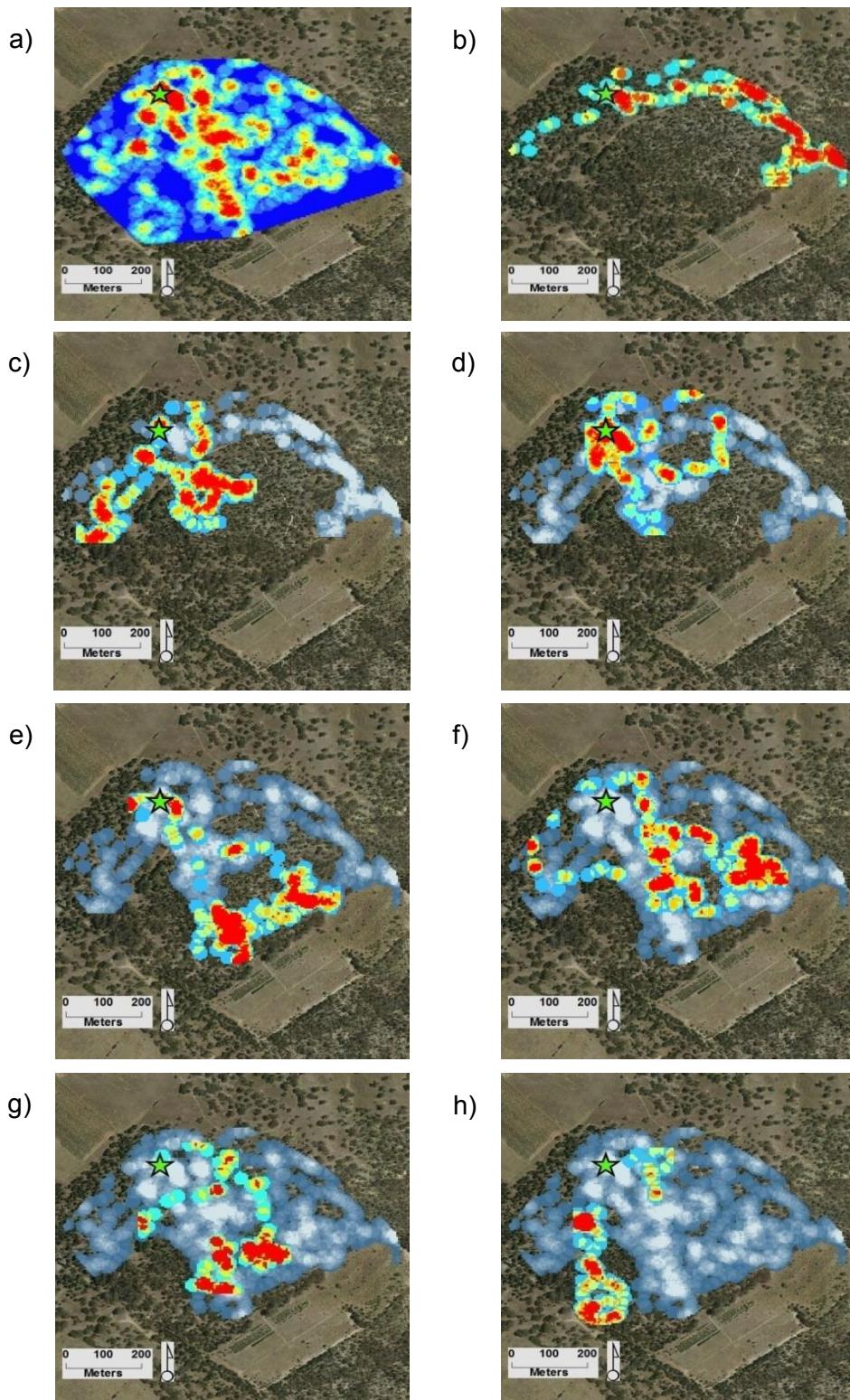


Figure 7.4: Use of habitat by a cohesive group of five White-browed Babblers (*Pomatostomus superciliosus*) at Altona Chatterton. Figure 7.4a shows the total area that was used (home range estimate 36.4 ha) and the disproportionate use of habitat areas over seven days. In these Figures, the red colour depicts areas of high habitat use and the light blue colour depicts areas of low habitat use. The dark blue colour in Figure 7.4a also depicts areas that these birds were not recorded using within the confines of the Minimum Convex Polygon. Figures 7.4b-h also shows daily habitat use (red and light blue areas) for each respective day that overlay the accumulative habitat areas that had been used on previous days (grey coloured areas). The roost-nest location used by these birds is indicated with a green coloured star. Data collected May – June 2009.

Table 7.4: Quantitative analysis of spatial variability in habitat use within the minimum home range of a cohesive group of five White-browed Babblers (*Pomatostomus superciliosus*) at Altona Chatterton. These data were collected over seven days within a home range estimated at 36.4 ha. The Minimum Convex Polygon (MCP) for each day indicated the size of the areas that were used by these birds on each survey day. The number of waypoints indicates the amount of data that was used to construct each MCP. Comparing the spatial use of habitat areas between different days would seem to be conceptually easy. However, ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006) which was used to generate these data (refer Figure 7.4) does not have the capacity to perform this function. Instead, this analysis required the use of Hawth's tools using ArcMap to generate a set of random points for each survey day as the accumulated area used by these birds increased. An enforced minimum distance of 20 m between points avoided spatial autocorrelation. Using the Sample tool in ArcMap, sample values were taken from the actual waypoint locations for each subsequent day, relative to the number of random points. These actual values were then compared with the values of the random set of points. The resultant values were subjected to regression analysis, where the R^2 values were calculated for the agreement in habitat use for each survey day. The Coefficient of Determination (R^2) value represents the percent of overlap in the habitat areas that were used each day, with that used on all previous survey days. Data collected May – June 2009.

| Day | MCP (ha) | # waypoints | R^2 | # random points | Figure reference (page 257) |
|-----|----------|-------------|-------|-----------------|-----------------------------|
| 1 | 12.1 | 148 | - | - | 7.4b |
| 2 | 9.7 | 146 | 0.000 | 174 | 7.4c |
| 3 | 10.2 | 149 | 0.115 | 174 | 7.4d |
| 4 | 11.5 | 161 | 0.008 | 279 | 7.4e |
| 5 | 16.4 | 183 | 0.049 | 279 | 7.4f |
| 6 | 6.9 | 90 | 0.015 | 279 | 7.4g |
| 7 | 5.5 | 91 | 0.159 | 291 | 7.4h |

The natural perturbation of wildfire adversely influenced habitat use by individuals of the Brown Treecreeper. At Altona Chatterton, a wildfire in late December 2009 burnt around 65% of a previously surveyed home range for a cohesive group of four Brown Treecreepers (Fig. 7.5a). Across this home range the fire destroyed several hollow-bearing trees, including the principal roost tree used during the June 2009 survey – a dead Pink Gum (*E. fasciculosa*) possessing four hollows. This home range was subsequently re-surveyed three times for a day each over the next 14 months. The June 2010 survey revealed three Brown Treecreepers using this area, with habitat use favouring unburnt habitats (Fig. 7.5b). Notably, during this survey the areas of high use were dominated by closed heath habitat; namely thick patches of Flame Heath (*Astroloma conostephoides*) and Common Fringe-myrtle (*Calytrix tetragona*) and thickets of Kangaroo Thorn (*Acacia paradoxa*; red coloured areas in Fig. 7.5b). During this June 2010 survey, the birds often foraged on the ground among the shrubby vegetation. The November 2010 survey detected a breeding pair of Brown Treecreepers with three fledged offspring. Again much of the habitat used by these birds favoured unburnt areas (Fig. 7.5c). The natal hollow used by this pair belonged to a live Pink Gum within an unburnt area that offered four hollows. The February 2011 survey revealed three Brown Treecreepers that persistently favoured the unburnt habitat surrounding the fire scarred habitat (Fig. 7.5d).

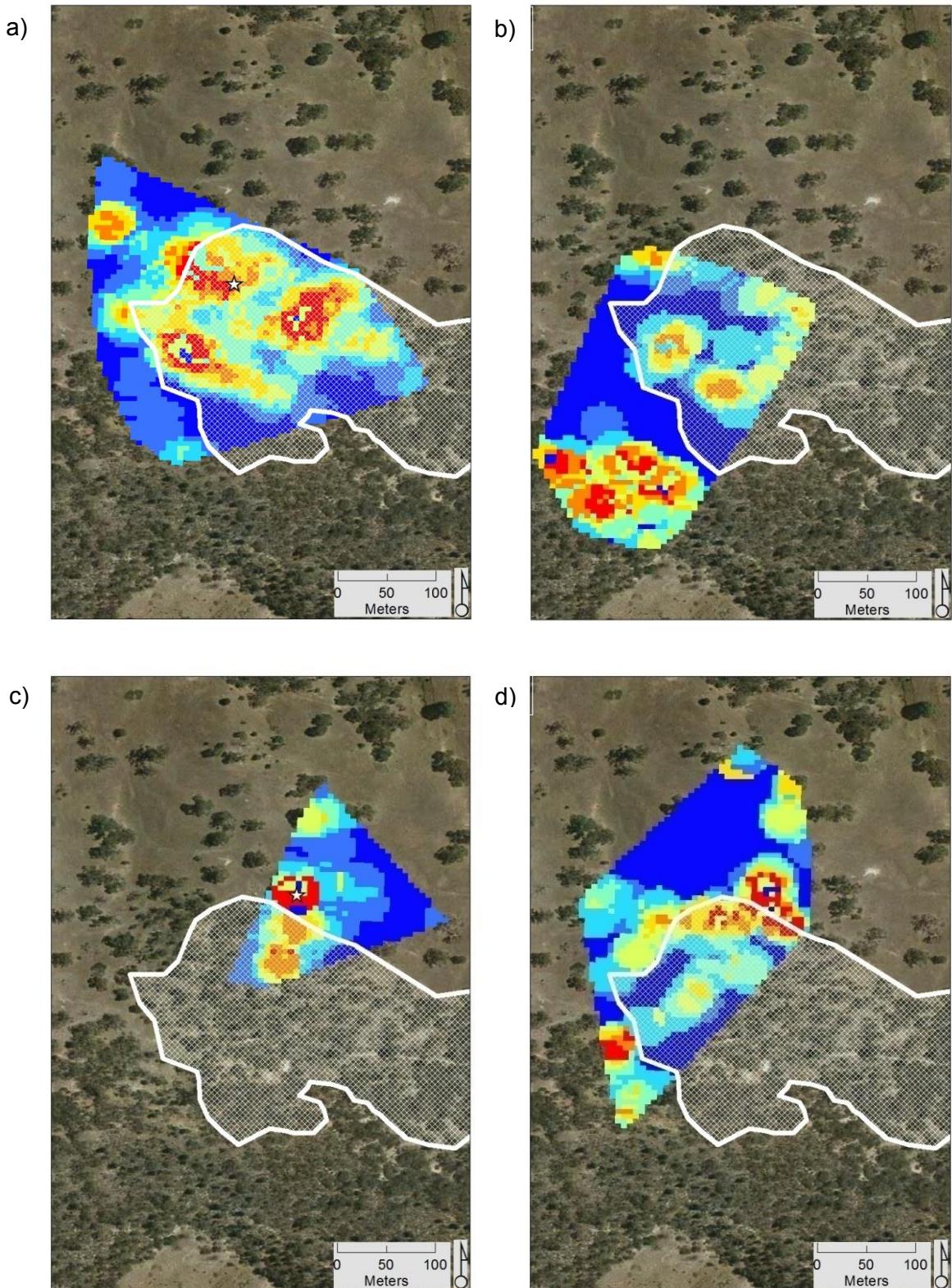


Figure 7.5: Brown Treecreepers at Altona Chatterton, comparing amalgamated Point Densities of waypoints and time spent per waypoint for (a) area used in June/July 2009 pre-fire and their roost-nest location (white star), (b) area used in June 2010 post-fire, (c) area used in November 2010 and their breeding hollow location (white star) and (d) area used in February 2011. The overlying white crosshatch indicates the area that was burnt by wildfire in late December 2009 after the initial survey was conducted in June 2009. The fire scar on Figure 7.5a is for reference purposes only.

Prevailing weather conditions also influenced which habitat areas were used by some bird species. A pair of Hooded Robins at Altona Chatterton was surveyed for two consecutive days in mild and sunny conditions in July 2009. Within the habitat area that these birds used (MCP 19.2 ha), eight hot-spots were identified within which nine 30 m × 30 m plots were surveyed. Three days later, this pair was re-surveyed for two consecutive days but this time in stormy conditions with high wind and persistent rain, including periods of small hail. During the storm, these birds used an additional 7.2 ha of habitat area. Only 0.6% of the same habitat areas were used between the two contrasting weather surveys (Table 7.5). Six new hot-spots were identified during the storm, within which a further nine 30 m × 30 m plots were surveyed. Comparing the hot-spot plots generated during the fine weather with those generated during the storm revealed that statistically, these birds used more open areas during fine weather but areas with a denser tree layer during the storm ($t_{16} = 2.82, P = 0.012$; Fig. 7.6a). Slender Cypress-pine was statistically more prominent in hot-spot plots during the storm than during fine weather ($\chi^2_{0.05,1} = 8.95, P = 0.003$; Fig. 7.6b).

This same Hooded Robin home range at Altona Chatterton was re-surveyed in June the following year (2010), where a pair of Hooded Robins was again found to inhabit. Surveys during the first two consecutive days were again conducted in fine and mild conditions, with the pair using an ecotone between woodland and stock-grazed farmland for most of the day. Although the habitat area these birds used during the fine conditions in 2010 nested within the habitat area that was used during the fine weather survey in 2009, the birds in 2010 used a much smaller area in comparison (refer Chapter 6). This resulted in only 6.2% of the same habitat areas being used between these two survey periods (Table 7.5). Similar to the 2009 survey, 10 days later in 2010 these birds were re-surveyed on an overcast day with persistent light showers. As with the habitat areas that were used by Hooded Robins during the storm in 2009, during the inclement weather survey in 2010 the birds remained within the woodland and did not venture out into the more exposed habitats. Only 9.6% of the same habitat areas were used during these contrasting weather conditions in 2010 (Table 7.5). The wintery weather in 2010 was not on the same scale as the storm in 2009 however, and although on both occasions the birds remained within the woodland during these inclement periods, only 6.3% of the same habitat areas were used (Table 7.5).

A number of other species also expressed low spatial correlation in habitat uses between the initial survey in 2009 and the re-survey in 2010. The Yellow-rumped Thornbill surveyed at Altona Chatterton only used 1.5% of the same habitat areas (Table 7.5). The Yellow Thornbill surveyed at Sandy Creek Site A used only 1.9% of the same habitat areas, however the Yellow Thornbill surveyed at Sandy Creek Site B used 24% of the same habitat areas (Table 7.5).

The White-winged Chough surveyed at Sandy Creek only used 13% of the same habitat areas between the two survey periods (Table 7.5). This became obvious during these surveys, as although all surveys were commenced within the confines of the Park, the surveyed birds ventured out and spent more time within the surrounding farmland (35%) and associated vegetated creeks (17%) in the 2010 survey compared to the 2009 survey. On all days during both surveys however the surveyed birds always returned to the Park to roost.

The Willie Wagtail surveyed at Altona Kies Site B only used 17% of the same habitat areas between the 2009 and 2010 surveys (Table 7.5). The home range at Altona Chatterton also appeared to have been abandoned during the one-off 2011 summer survey (Table 7.5). Numerous searches throughout the day were performed within the boundary of this 2009 winter home range, with no Willie Wagtails detected. This site was dominated by annual grasses at the time of the summer survey. This period also coincided with a locust plague, with large quantities of locust in their hopper stage. I have witnessed a Willie Wagtail foraging on juvenile crickets, so it was assumed that appropriate foraging resources were not a limiting factor that impeded habitat use within this area. Up to seven Willie Wagtails were located just north of the Altona Chatterton winter home range site however. These birds actively foraged within the canopy of the tree-lined North Para River for the majority of the day.

During surveys on the Willie Wagtail in both 2009 and 2010, the birds were also recorded using the overstorey strata in which to forage. This occurred within sites that contained high amounts of overstorey vegetation (e.g. Altona Kies Site B and Sandy Creek). These foraging records occurred when there was either dim or no sunlight illuminating the ground in the early morning and late afternoon (e.g. Fig. 7.7). At these times, the birds foraged among and just above the sunlit canopy. These areas used by the Willie Wagtails

at dawn and dusk compared to the areas used at other times of the day were not in different parts of their home range, but were often adjacent or neareby.

Several species expressed higher spatial correlation in the habitat areas that they used between the 2009 and 2010 surveys. The Rufous Whistlers surveyed at Sandy Creek used 21% of the same habitat areas between the two survey periods (Table 7.5). Similarly, the Buff-rumped Thornbills surveyed at Para Wirra Site B used 22% of the same habitat areas between the two survey periods (Table 7.5). However, the Buff-rumped Thornbills within this home range used this habitat areas differently between these two surveys ($\chi^2_{0.05,2} = 5.99$, $P = < 0.001$). The defining difference was less foraging within the midstorey vegetation in 2010 compared to 2009 ($\chi^2_{0.05,1} = 51.4$, $P = < 0.001$; Fig. 7.8). These birds foraged on the ground and within the shrub layer at similar intensities in both years, however they foraged within the overstorey strata more in 2010 than in 2009 (Fig. 7.8).

The two bird species that hold permanent all-purpose territories had the highest spatial correlation in the habitat areas that they used between the two survey periods. These were the Brown Treecreeper at Altona Torbreck (33%) and the White-browed Babbler at Altona Kies (43%; Table 7.5).

Table 7.5: Quantitative analyses of annual variability in habitat use within individual minimum home ranges of nine bird species within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Initial surveys within individual home ranges were conducted in 2009. Point Density analysis in ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006) was used to define patches of both high and low habitat use by the birds within their home range. These same home ranges were then re-surveyed in 2010 / 2011, where again Point Density analysis was used to define patches of variable habitat use by the birds. For Hooded Robins (*Melanodryas cucullata*) at Altona Chatterton, surveys in 2009 and 2010 were conducted during both fine and inclement weather. Comparing the spatial use of habitat areas within a home range between annual surveys would seem to be conceptually easy. However, ArcMap which was used to generate these data does not have the capacity to perform this function. Instead, this analysis required the use of Hawth's tools in ArcMap to generate a set of random points across each home range. An enforced minimum distance of 20 m between points avoided spatial autocorrelation. Using the Sample tool in ArcMap, sample values were taken from the actual waypoint locations for each home range, relative to the subsequent number of random points used for that home range. These actual values were then compared with the values of the random set of points. The resultant values were subjected to regression analysis, where the R^2 values were calculated for the agreement in habitat use between the 2009 and 2010 / 2011 surveys for each home range. The extent of spatial overlap in the degree of habitat use within each home range between the 2009 and 2010 / 2011 surveys is given by the Coefficient of Determination (R^2).

| Bird species | Location | Initial survey | Re-survey | R^2 | # random points |
|-------------------------|-------------------------------------|----------------|-----------|-------|-----------------|
| Brown Treecreeper | Altona Torbreck | 6-7/2009 | 6/2010 | 0.33 | 70 |
| Buff-rumped Thornbill | Para Wirra Site B | 7/2009 | 6-7/2010 | 0.22 | 77 |
| Hooded Robin | Altona Chatterton (fine / storm) | 7/2009 | 7/2009 | 0.006 | 210 |
| Hooded Robin | Altona Chatterton (both fine) | 7/2009 | 6/2010 | 0.062 | 210 |
| Hooded Robin | Altona Chatterton (fine / showers) | 6/2010 | 6/2010 | 0.096 | 210 |
| Hooded Robin | Altona Chatterton (storm / showers) | 7/2009 | 6/2010 | 0.063 | 210 |
| Rufous Whistler | Sandy Creek | 8/2009 | 9/2010 | 0.21 | 112 |
| White-browed Babbler | Altona Kies | 4-5/2009 | 6/2010 | 0.43 | 115 |
| White-winged Chough | Sandy Creek | 6/2009 | 6-7/2010 | 0.13 | 1,256 |
| Willie Wagtail | Altona Kies Site B | 5/2009 | 6/2010 | 0.17 | 24 |
| Willie Wagtail | Altona Chatterton | 7/2009 | 2/2011 | 0.00 | n/a |
| Yellow-rumped Thornbill | Altona Chatterton | 5/2009 | 6/2010 | 0.015 | 196 |
| Yellow Thornbill | Sandy Creek Site A | 6/2009 | 6/2010 | 0.019 | 75 |
| Yellow Thornbill | Sandy Creek Site B | 6/2009 | 6/2010 | 0.24 | 54 |

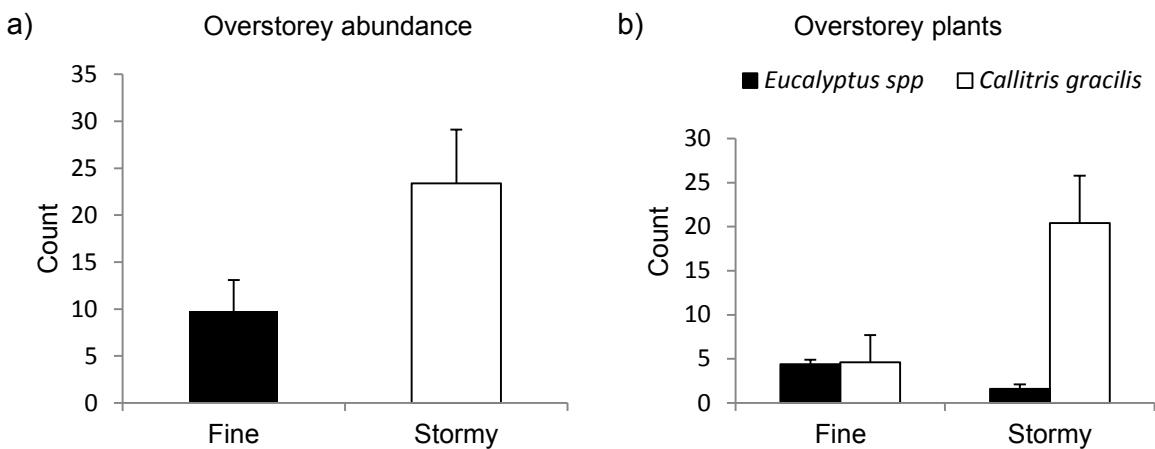


Figure 7.6: Habitat use by Hooded Robins (*Melanodryas cucullata*) at Altona Chatterton, comparing two days of fine and mild weather with two days of stormy conditions. These data compare nine, 30 m x 30 m plots within high use areas (i.e. hot-spots) during the fine weather with nine, 30 m x 30 m plots within high use areas during the inclement weather. During the storm the habitat areas these birds used contained (a) greater overstorey abundance which was (b) dominated by Slender Cypress-pine (*Callitris gracilis*). Error bars are standard error. Data collected July 2009.

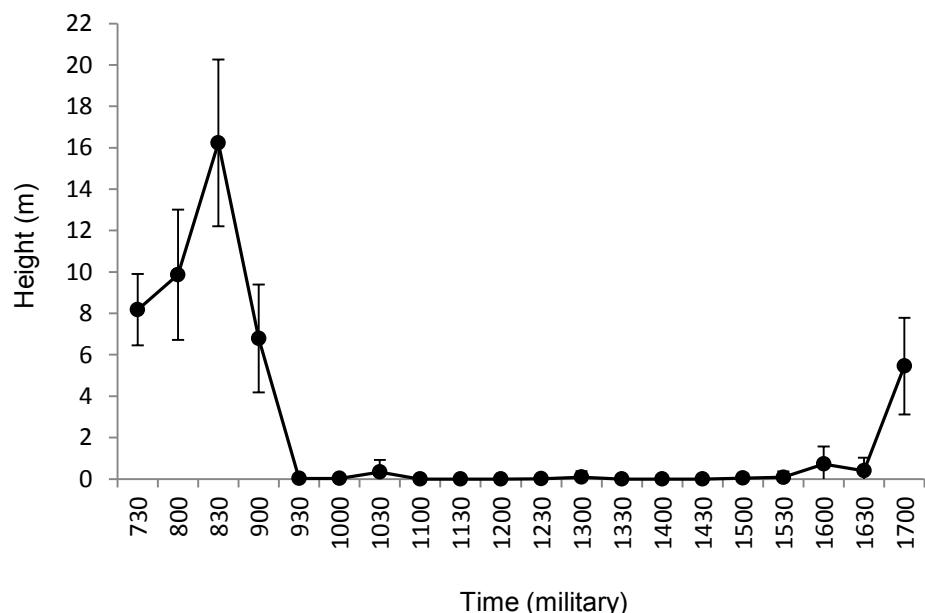


Figure 7.7: Foraging by a pair of Willie Wagtails (*Rhipidura leucophrys*) at Altona Kies Site B. The birds foraged among the overstorey canopy in the early morning and late afternoon on both days. The data points are the average heights the birds foraged at between dawn and dusk (30 minute increments in military time). Error bars are Standard Error. Data collected over two days in June 2010.

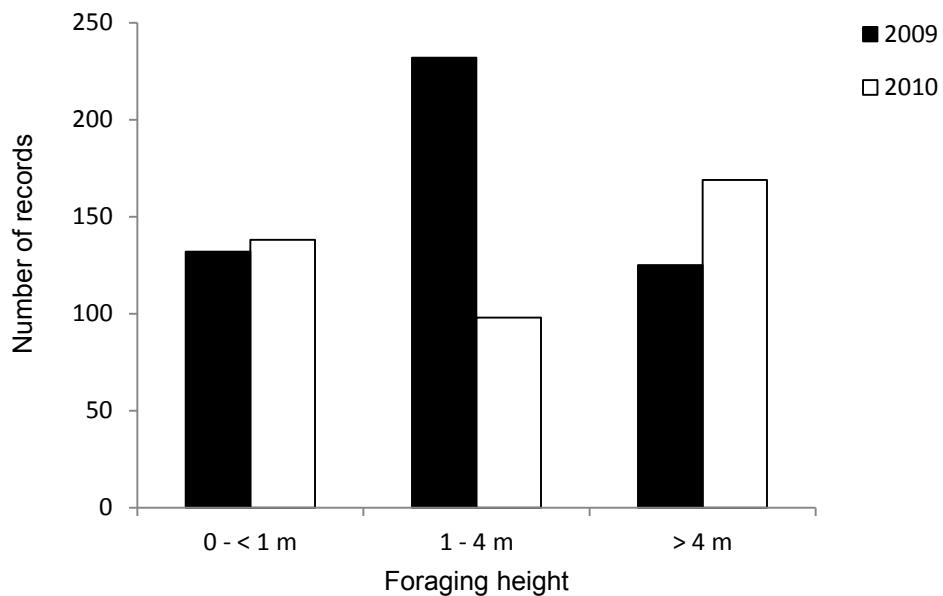


Figure 7.8: Foraging by Buff-rumped Thornbills (*Acanthiza reguloides*) within Para Wirra Site B. Two surveys were conducted, one in July 2009 and the other in June / July 2010. Foraging by this bird species has been segregated by storey level; being the ground and shrub layer (0 - < 1 m), the midstorey layer (1 - 4 m) and the overstorey layer (> 4 m). Foraging within the midstorey strata by this species was significantly different between the 2009 and 2010 surveys (using the subdivided chi-square goodness of fit; $\chi^2_{0.05,1} = 51.4$, $P = < 0.001$).

Discussion

A key finding of this study was the high variability in habitat use by a number of individuals from a range of bird species in both time and space. This is a unique insight into habitat use by members of these woodland birds. Most previous studies that associate species to habitat simply treat birds as either being present or absent within a habitat area, which largely ignores spatial and temporal variability in habitat use by the birds (Westphal *et al.* 2003b; Westphal *et al.* 2007).

The importance of habitat heterogeneity within woodland habitats from a bird's perspective was again emphasised by this study, and supports the findings in Chapters 4 and 5. The behaviour of the birds indicated that birds use different habitats from time to time, either because the habitat conditions changed over time (e.g. extent of grasses) or the birds needed different habitat features depending on the prevailing conditions (e.g. Willie Wagtails foraging on or above the tree canopies at dawn and dusk; Slender Cypress-pines providing shelter from adverse weather conditions for a pair of Hooded Robins). In

response, revegetation should not simply consist of planting stands of trees and shrubs all at average densities, but instead the re-established woodlands need to have plantings that consist of different densities of plants, which should also contain different combinations of plant species. This would help cater for the variable habitat needs of birds, as illustrated by this study.

The findings of this study also have direct ramifications on surveys that are designed to identify the fine-scale habitat needs of species and to infer species' broad-scale habitat associations. Many bird studies use the popular twenty-minute / two-hectare field survey method to identify the species that occupied a survey area. From this information, inferences are then drawn as to species-habitat associations and to gauge the conservation status of individual species across the broader landscape (e.g. Barrett *et al.* 2003; Possingham *et al.* 2004). Results from this current study would suggest however that the twenty-minute time period may be inadequate to effectively record the presence of species that potentially occupy a particular site. Based on the times each species in this study spent at a bird-stop location, only five to 15 (average nine) bird-stops would occur within a twenty-minute period. Not surprisingly, it has been suggested that most bird species within a twenty-minute period can be recorded within the first five minutes (Bonthoux and Balent 2012). Realistically though, a twenty-minute period sometime in the morning will not be adequate to assess the relative abundance of birds and/or their use of habitat within a two-hectare area.

Coupled to this, most minimum home range estimates for individuals of the nine bird species that were surveyed far exceed two hectares (refer Chapter 6). Consequently, two hectare survey areas only amount to a small sub-section of the total habitat area used by the individuals that actually use the two hectares being surveyed. Field *et al.* (2002) alluded to the fact that home range sizes and habitat use by some species may not suit the twenty-minute / two-hectare method, especially if multiple surveys are conducted on the same day. These authors conceded however that quantitative evidence on species' home ranges and rates of movements were not available. My observations on the individuals of nine species of declining birds using some of the woodland areas in which the studies of Field *et al.* (2002) were conducted, revealed that they all had minimum home range areas that were much larger than two hectares, and that the rates in which the birds moved within their respective home ranges would limit their detectability within a twenty-minute period.

These finding suggest that the common practice of assessing the relative index of species abundances using the twenty-minute / two-hectare sampling technique may not be adequate at capturing many of the species that actually use that site. Importantly, this study also revealed that the combination of limited temporal periods and small spatial scales (i.e. 20 min / 2 ha) compared to the actual extent of area that was used by individuals of the species that were studied, together with the manner in which those birds move within their minimum home range areas, could easily result in the presence of some species not being detected at all. Indeed, increasing the survey time from twenty minutes to one hour can improve species richness and/or abundance estimates that better reflect the actual numbers that are present (e.g. Possingham *et al.* 2006).

Many studies also employ three repeat visits to a survey location to improve species estimates (e.g. Possingham *et al.* 2004; Westphal *et al.* 2007). This current study also showed that less conspicuous bird species (e.g. the Hooded Robin) may not always be detected during the first three surveys. This potentially results in a false-negative error, where a species is recorded as being absent when in fact it was present. This finding supports Tyre *et al.* (2003), who suggested that up to six repeated surveys are required to improve the detectability of some species. A problem then exists when using species presence-absence data from a limited number of repeated surveys, which are then used to compare between sites and to infer habitat preferences by species. The habitat areas in which a species is recorded as being absent during these multiple surveys within the same season under favourable weather conditions, may in fact prove to be functional habitat for members of that species, thus potentially confounding the results.

As previously mentioned, an important finding of this study was the high variability in habitat use by numerous individuals of several bird species at fine-scales. Importantly, this study demonstrated that high variability in habitat use can occur during a single day, can occur over multiple days and can differ inter-annually. A further key finding was that hot-spots could change from one day to the next, depending on how the birds used their habitat. Habitat use can also be strongly influenced by sustained perturbations such as the effects of wildfire and stochastic events like extreme weather conditions. Clearly, the times in which surveys are conducted, on which days surveys are performed and under what weather conditions, can strongly influence how birds are perceived to use (or not use) habitat areas.

That said, most information concerning habitat use by birds are limited to small data sets (Katajisto and Moilanen 2006) that apart from a few exceptions (e.g. Kinross 2004), traditionally do not span entire days (e.g. Fisher 2001) and rarely if ever are conducted during inclement weather (e.g. Luck *et al.* 1999; Mac Nally 2000; Antos *et al.* 2008). This frequently applied “fine weather” approach to sampling runs a high risk of missing many less common but equally important habitat associations, such as the more enclosed habitat that was used by the pair of Hooded Robins during the storm. The behaviour of the individual Hooded Robins on days of inclement weather is consistent with the birds seeking out more sheltered habitat under these conditions. Strong variability in habitat use by individuals of various bird species, as shown by this study, emphasises the need to amend how surveys that aim to define the habitat associations of bird species are conducted. The aim here would be to generate a broader understanding of the different habitat associations by birds of individual species. This would include conducting surveys that are not so biased towards particular times of the day and to days with only fine weather. Surveys need to be performed under variable conditions, to better understand the holistic habitat needs of species.

In contrast, information on habitat use by bird species is generally limited to relatively small areas of habitat that are nested within broader habitat areas that are yet to be surveyed (Ferrier *et al.* 2002b). Furthermore, the collective data that is obtained from these small scale surveys offer little or no information on the fine-scale spatial distributions of species (Ferrier *et al.* 2002b). Clearly, the lack of detailed information regarding the use of habitats by species, let alone the variable nature of habitat use, limits the usefulness of this previously collected data in effectively predicting the habitat preferences of species.

Nevertheless, these ecologically incomplete data sets are used for Species Distribution Models (SDM) that seek to identify suitable habitat areas for particular species within the landscape (Wintle *et al.* 2005). The deficient nature of these species-habitat data sets has given rise to predictive habitat distribution models and associated species detection probability estimates (Guisan and Zimmermann 2000; MacKenzie *et al.* 2002). However due to data limitations, expert opinion is recommended at different stages of the model building process to help produce ecologically meaningful models (Wintle *et al.* 2005). As a result though, the correlation between modelled predictions and statistical significance remains unclear (Wintle *et al.* 2005). The literature also offers little guidance as to the

recommended minimum coverage of habitat area that should be used (Wintle *et al.* 2005). The choice of sample size and which environmental variables are used can also produce varied and unpredictable results (Stockwell and Peterson 2002). There also remains the issue of producing statistically independent data that is required to run these models versus the loss of biologically meaningful information, in the pursuit of avoiding statistically undesirable autocorrelation (as discussed in Chapter 6).

An implicit assumption of the data that is used for SDM is that species occupy preferred habitat areas, yet not enough information exists on most species to produce robust single species models (Wintle *et al.* 2005). More so, the untested assumption of occupancy equalling suitability may be frequently violated, as implied by the Brown Treecreepers in this study. The fact that these birds uncharacteristically used closed heath habitat post-fire exposes a potential limitation of this and other studies that seek to identify suitable habitats for fauna. Habitat use by these birds had obviously been displaced due to the effects of the fire; with the birds seemingly “making do” with what habitat remained that was still suitable. Similarly, remnant habitats do not necessarily represent the types of habitats that have been disproportionately cleared, and it is these disproportionately cleared habitats that we should be aiming to re-establish (Paton 2000; Cofinas and Creighton 2001; Paton *et al.* 2004; Mac Nally *et al.* 2009; Thomson *et al.* 2009). Yet our understanding of many species habitat associations are based on their current use of residual habitats left standing since clearance. The habitats which are occupied by today’s birds may not be reminiscent of the habitats that these species once preferentially used, which have since succumbed to clearance. Species may only be “making do” with the best of what habitat is currently available. Hence, the presence of a species within a habitat area does not necessarily infer quality habitat for that species.

For instance, Treecreepers in general display sensitivity to burnt habitats, at least in the initial post-fire years. This has been shown for the Brown Treecreeper in mixed *Callitris* and *Eucalyptus* woodland (Turner 1987), for the White-throated Treecreeper (*Cormobates leucophaeus*) in forests (Recher *et al.* 1985a), for the Black Treecreeper (*Climacteris picumnus melanota*) on Cape York Peninsula (Garnett and Crowley 1995) and for the Black-tailed Treecreeper (*Climacteris melanura*) in northern tropical woodlands and open forest (Woinarski 1990). It may take several years for these Treecreepers to re-colonise habitats that have been inflicted by major wildfire (e.g. Turner 1987; Reilly 1991a). The

relatively temporary nature of this habitat displacement cannot be adequately explained by the loss of hollows to fire. The creation of new hollows to replace those destroyed usually takes much longer than a few years (Gibbons and Lindenmayer 2002). The bias towards unburnt habitats displayed by individuals of Brown Treecreepers in this and previous studies may be due instead to the influence fire has on their invertebrate prey, which typically consists of ants (e.g. Noske 1991). Habitats disturbed by fire directly impact the communal structure of ants (Andersen 1991), but this also depends on the habitat type, the severity of the fire and rate of post-fire habitat recovery (Hoffmann and Andersen 2003). The typical ant species that were preyed upon by individuals of Brown Treecreepers during this study (i.e. *Iridomyrmex*, *Camponotus* and *Pheidole* spp; refer Appendix C) generally respond quickly following fire disturbance and attain similar abundances to unburnt habitats within a few years (Gill *et al.* 1999; Hoffmann and Andersen 2003). The return of Treecreepers to previously burned habitats only after several years have passed correlates with the return in abundance of their invertebrate prey species.

Examples such as that above highlight the need to understand more fully how a species interrelates with its environment. The simplistic assumption that habitat use correlates with habitat quality also gives no consideration for the long-term survivorship and/or reproductive potential of species within a particular habitat area (Van Horne 1983). For example, the birds may belong to a small, unviable population which is in decline (Caughley 1994). This would occur if species are confined to a “sink” habitat, where mortality exceeds reproduction and populations can only be maintained via immigration from a “source” population located elsewhere (Pulliam 1988).

Some researchers also uncritically accept a linear association exists between an organism and the environmental variables that are used to predict species-habitat relationships (Austin 2002). This seems to ignore the fact that nature is too highly complex and heterogeneous for modelled solutions to incorporate high degrees of variable habitat use, as indicated by this study. Also, the coarse-scale nature of much of the data that is used in modelling is generally insufficient for predicting the distributions of species at fine-scales. Austin and Van Niel (2011) argued in support for fine-scale variation in habitats to be incorporated into SDM, to better understand the ecological relationships of species with their habitat.

It is then critically important to understand the range of habitat features that can be used by a species. Equally important is an understanding of the mechanisms that drive variable habitat use. This insight can then be incorporated into predictive modelling and be entrenched into the thinking of revegetation practitioners who seek to provide quality habitats for species of concern. The critical point here is that there is an insufficient amount of knowledge regarding the habitat needs of individual species. This explains the emphasis given to community level research that incorporates a multitude of limited data sets, which includes at best a restricted amount of information regarding the habitat needs of declining and/or rare species (e.g. Ferrier *et al.* 2002a).

This study has also highlighted that not every part of a minimum home range will necessarily be used during a single day. Soderquist and Gibbons (2007) also demonstrated this with the Powerful Owl (*Ninox strenua*), which on average used only 5 – 12% of its home range during a single night. As a consequence, a hot-spot identified on one day may not necessarily be “hot” on other days. This is where an understanding of habitat use by particular species comes to the fore. In regards to variable hot-spots that were used by individuals of White-browed Babblers at Altona Chatterton, this had more to do with how these birds moved within their home range. For example, the relatively short distances the White-browed Babblers travelled (hopped) between locations were not conducive for these birds to cover every part of their minimum home range during a single day. In contrast, species like the Hooded Robin, which regularly flies long distances between locations, can easily cover its similarly large home range during a single day (e.g. Northeast 2007).

Together with the behavioural aspects of species, habitat features must also be considered when explaining variable habitat use. Take for instance the different areas that were used by the pair of Hooded Robins at different times of the day. The reason for this pattern in habitat use was predictably due to the reliance these birds have on the visual detection of active invertebrate prey. The principal heat source for many ectothermic invertebrates is solar input (May 1979). Intuitively, invertebrate prey should be more lively (and thus more easily detected) within thermally warmed open habitats, especially during the winter months (Recher *et al.* 1983). My interpretation as to why the individual Hooded Robins in this study used different habitat areas at different times of the day has to do with direct sunlight exposure and the physical nature of the Altona Chatterton site. The protected woodland to the south that these birds first used lay on the eastern flank of a slight rise. As

the sun rose in the morning, thermal radiation warmed the leaf-litter that had accumulated on the ground. The increase in ambient temperature would have stimulated the invertebrates that reside among this litter. In the afternoon, shadows were cast over this same protected woodland. By this time, the pair of Hooded Robins was using habitat further north within the more open stock-grazed woodland. Within this area, large portions of these undulating paddocks were bathed in sunlight until dusk, where it was suspected that more active invertebrate prey could be found.

The importance of conducting surveys during variable weather conditions and over different seasons was also highlighted by this study, with habitat used by the Hooded Robins that were studied during the storm and the Willie Wagtails that were studied during the summer period. The general use of habitats by these birds suggests they are both open woodland specialist that predominantly forage at or near the ground level, with both prone to using clearings and habitat edges (Hindwood and McGill 1950; Purcell 1993; Priday 2010). These habitat associations typically emanate from fine-weather surveys however, which are often conducted during the cooler winter months. The principal reason given as to why most bird surveys are conducted in consistently fine conditions is to reduce variability when statistically comparing between sites. However the bird-habitat associations emanating from such studies should only be used to describe how those birds used habitat during those times, and not be extrapolated to infer general habitat use and preferences.

A prime example of this is the habitat that was used by the Hooded Robins in this study during the storm. The reason why these Hooded Robins sought more sheltered habitat was strikingly obvious at the time. Strong winds and driving rain made for very uncomfortable conditions, from which stands of Slender Cypress-pine offered reasonable protection. These birds foraged throughout the day and made many successful pounces. My impression was that either the invertebrates were being rain-washed from the surrounding vegetation and/or were being flooded out from their subterranean habitat. When it hailed though, both the birds and the observer took refuge within large Pines. Hail storms are a known cause of injury and death among birds (e.g. Lincoln 1931; Sarasola *et al.* 2005; Saunders *et al.* 2011). Having immediate access to appropriate vegetation cover during these times limits the potential for birds to suffer injuries caused by hail stones.

Clearly the habitat requirements of birds must be catered for to satisfy a broad range of their life-time needs. Thus, the information that is needed to better inform on the habitat requirements of species must also include survey data at other times. For instance as shown by the Willie Wagtails in this study, the ground layer can be subjected to changes in exotic annual vegetation, influencing seasonal habitat use by these birds. The Altona Chatterton site previously surveyed for individuals of Willie Wagtails was re-surveyed in the summer of 2011. At that time, this entire sheep-grazed woodland reverted to a field of annual grasses up to 600 mm high. The Willie Wagtail appeared to have abandoned this area during this period, as the grass cover would have impeded their typical ground foraging technique. This was despite the abundance of likely invertebrate prey within this immediate area. This observation agrees with Gilmore (1985), who claimed that the availability of suitable foraging substrates seems to be more important than the availability of a foraging resource for most woodland birds.

While vegetation at height provided an alternative foraging substrate away from the ground-covering summer grasses, in winter numerous Willie Wagtails also foraged among the overstorey canopy early and late in the day when the ground was not bathed in sunlight. This was probably due to more active invertebrate prey occupying these solar-warmed micro-habitats, as previously discussed. Hence, the tree layer is an important habitat attribute in supporting variable habitat use by these birds, in respect to both daily and seasonal changes. In support, Webb-Pullman and Elgar (1998) also showed how the Willie Wagtail can adjust its foraging technique to correspond with patterns of insect activity during diurnal cycles. It is not uncommon for this bird species to be recorded foraging at the canopy level, although studies show this species predominantly forages at much lower heights year round (Cameron 1985; Ford *et al.* 1986; Brooker *et al.* 1990; Recher and Davis 1998).

The Buff-rumped Thornbills that were surveyed displayed low spatial correlation in habitat use between years, but this may have also been influenced by seasonal change. Seasonal shifts in foraging often see this species foraging at higher levels during the warmer months compared to the cooler months (Bell 1985; Recher 1989). Bell (1985) suggested the reason why the Buff-rumped Thornbill foraged higher in spring than in autumn and winter may be due to changes in the social structure of this species. During the cooler months the birds within the clan forage and roost as a group. These birds then separate out across their

same winter territory into pairs or smaller groups to breed in spring. Flocking by birds is strongly suggested to decrease an individual's probability of being taken by a predator (Caraco *et al.* 1980; Goldman 1980). The break-up of the clan for the breeding season is a likely reason why this species seeks the relative safety of an elevated foraging position (Johnson 1980; Bell 1985). When there are more birds in a flock, the flock is more likely to detect the presence of a predator. Foraging on the ground is more risky than foraging within a canopy of foliage, but the risks are reduced when the birds forage as a flock lower to the ground.

In this current study, surveys on the Buff-rumped Thornbills were conducted in June and July of 2009 and 2010. In both survey periods the birds were building nests. The timing of these surveys obviously coincided with the commencement of their breeding season and the likely transition period from mostly near-ground foraging to mostly arboreal foraging. This would help explain why the two inter-annual surveys returned different results. These findings also support those in Chapter 5, where habitat use by the Buff-rumped Thornbill correlated with habitats that contained various storey levels but with an open vegetation cover.

Variable habitat use by some species was not so easily accounted for. For example, the low spatial correlation in habitat use between inter-annual periods for the Yellow-rumped Thornbills that were surveyed may be due to this species apparent flexible use of various micro-habitats (Maron and Lill 2005) and perceived tolerances to patch size, isolation and structural complexity (Freudenberger 1999). This highlights a further need to study habitat use by species at fine-scales, to help identify important habitat attributes for individual species.

As a final point, marked variations from one day to the next in what constituted a hot-spot may raise concerns regarding the reliability that a particular hot-spot area represents favoured habitat for those birds concerned. This is especially so considering the relatively small number of days that were used as input in this study for most individuals of the species that were studied. Limited sampling may have influenced which areas were deemed to be hot-spots overall. This can be explained by the fact that many birds did not use all parts of their home range equally each day. This is especially so for birds that occupied large home ranges and moved relatively short distances between bird-stop

locations (e.g. White-browed Babbler and White-winged Chough). Species such as these required a greater number of survey days to produce sufficient spatial overlap in habitat use to identify overall preferred habitat areas during the survey period. This was also the case when estimating the sizes of the minimum home ranges for individuals of these species (Chapter 6). The fact remains though that some habitat areas were preferentially used on some days and not on others. This does not weaken the argument that a hot-spot on one day that was not hot overall was not important for those birds using that habitat. If anything, such areas may be referred to as being “warm-spots”.

The next chapter directly compares the habitat differences between the hot-spots and cold-spots that were identified in this study. The critical point made here is that only habitat areas that were on average “hot” were only ever compared with areas that were always cold. Any areas that were preferentially used on one day but were not preferentially used overall were excluded from these analyses. As such, only the polar extremes of high use habitat areas and low use habitat areas were being compared.

Chapter 8

Fine-scale habitat use by birds

Introduction

Natural habitats are typically heterogeneous both floristically and structurally, with the heterogeneity creating an array of different habitat features. Individual bird species often respond to these specific habitat elements within the habitat matrices in which the features are found (Johnson 1980; Hutto 1985; Flesch and Steidl 2010). Different bird species have also been found to use different environmental features within the same habitats. This has been shown for bird species in general (Wooller and Calver 1981; Frith 1984; Ford *et al.* 1986), for species belonging to the same taxonomic family (Recher and Majer 1994) and between species of the same genus (Robinson 1992; Serrano and Astrain 2005). Habitat partitioning by woodland birds at the micro-habitat level was also reported by Fisher (2001), where species separated along a naturally occurring regeneration gradient within eucalypt woodland. Betts *et al.* (2006) has subsequently suggested that bird species require adequate quantities of their preferred micro-habitats within the landscape in which they reside. Their study demonstrated that within fragmented habitats, some woodland birds expanded their home ranges when confronted with landscapes that contained a low percentage of suitable cover. This presumably was to incorporate a higher percentage of preferred patches of habitat for those species.

Use of habitats by individual species is influenced by their physiological (Walsberg 1985), morphological (Winkler 1985) and behavioural (Klopfer 1985) adaptations. Habitat selection by birds is also considered to be more adaptive than learnt. Martin (1998) tested this assumption by studying the choice of nest site locations by birds and concluded that habitat choice may in part be driven by natural selection. Nests located in preferred micro-habitats compared to non-preferred micro-habitats produced better reproductive outcomes for the seven bird species that were studied. Evidence of natal sites influencing habitat selection by birds was also documented by Lack (1971).

Disproportionate use of habitats intuitively manifest as core areas within individual home ranges (Kaufmann 1962; Samuel *et al.* 1985). This clumping pattern of disproportionate habitat use by birds was shown by Paton *et al.* (2010), where it has been suggested that aggregations of observations indicate repeated activities, such as foraging and breeding (Swihart and Slade 1985). These patches of high use habitat (or “hot-spots”) within home ranges should then correlate with important habitat attributes that support these activities, more so than the corresponding patches of low use habitat (or “cold-spots”) within the same home range (Brown *et al.* 1995). If true, features of habitat at fine-scales should help explain patterns of spatial use by individual bird species (Brown *et al.* 1995).

To identify habitat attributes that are important for species, comparing habitat that is used by birds versus habitat that is not used is far more informative than comparing habitat that is used by birds with habitat that is merely available. The latter however is far more common within the literature (Jones 2001). Similarly, the literature in general offers information on broad-scale habitat associations for bird species that is then used to help guide restoration policy, but this only serves to predict general patterns of habitat use by the birds. Critical information regarding species-specific habitat requirements is still missing, together with the scales and/or frequency with which they are required within a habitat. Discriminating species-specific habitat requirements critical to their persistence within the landscape should fine tune habitat relationships and better inform restoration programs (Mac Nally 2008). To date though, this knowledge is clearly limited for most woodland bird species (Reid 1999; Block *et al.* 2001; Miller and Hobbs 2007).

Recent intense studies of fine-scale habitat use by several declining woodland bird species within the Mount Lofty Ranges have delivered some encouraging results, in respect to the birds’ fine-scale habitat requirements. For example, Barker (2007) identified that areas of high habitat use by the Brown Treecreeper (*Climacteris picumnus*) contained abundant tree hollows within remnant vegetation. In this study, the lack of hollows in adjacent 30 year-old revegetated woodland limited the use of these areas by Brown Treecreepers. Northeast (2007) reported that areas of high habitat use by the Hooded Robin (*Melanodryas cucullata*) contained consistently high amounts of litter cover. This study also revealed that Hooded Robins made single flights in excess of 50 m to move directly between these hot-spots within their home ranges. Richards (2008) found that high use habitat areas for the Chestnut-rumped Thornbill (*Acanthiza uropygialis*) were positively associated with the

presence of understorey and other low vegetation. This correlated with the predominant foraging height for these birds of between 0.3 and 2.8 m from the ground. Hunt (2011) revealed that high use habitat areas for the Rufous Whistler (*Pachycephala rufiventris*) contained significantly more vegetation at all strata levels.

This current study asks the question, do birds use habitat areas evenly within their home range? If birds do not use their habitat evenly, can comparisons between hot-spots and cold-spots in habitat use be explained by certain underlying habitat features? Importantly, do any fine-scale habitat associations make ecological sense for those species concerned? Should disproportionate use of habitats correlate with particular habitat elements, these features can then be incorporated into the design of new revegetation (Brown *et al.* 1995). This is hoped to improve the habitat quality for species of conservation concern. This approach is also an improvement on simply using remnant vegetation as a template, in the hope that it contains the necessary habitat elements that are required by certain species (see Bennett *et al.* 2000). To date, plantings that have been modelled on the broad habitat characteristics of remnant vegetation have not proved to be useful habitats for many species of conservation concern (Loyn *et al.* 2007; Bhagwat *et al.* 2008).

Methods

Bird surveys and data analyses

This study used data that were collected for individuals of nine declining woodland bird species within the north-central zone of the southern Mount Lofty Ranges, South Australia. The species were the Brown Treecreeper, Buff-rumped Thornbill (*Acanthiza reguloides*), Hooded Robin, Rufous Whistler, White-browed Babbler (*Pomatostomus superciliosus*), White-winged Chough (*Corcorax melanorhamphos*), Willie Wagtail (*Rhipidura leucophrys*), Yellow-rumped Thornbill (*A. chrysorrhoa*) and Yellow Thornbill (*A. nana*). The analyses in this chapter however are generally based on seasonal data (i.e. winter in 2009), so they only provide a basis for the use of specific habitats at one time of the year.

To collect data on habitat use by these birds within their minimum home ranges, I tracked individual birds almost continuously from dawn to dusk for one or more days (see Chapters 6 and 7 for details). This involved sequentially recording the behaviour of the

tracked birds and the position of the birds using a hand-held GPS. To do this, I streamlined the data collection by only recording the predominant behaviour of the focal bird at each bird-stop location (i.e. for each GPS waypoint), the plant species being used at that location, the typical height used by the focal bird, and the times the focal bird arrived and departed that location (recorded by the time on a wrist watch). Only one GPS waypoint was used for each bird-stop location. I defined a shift in location as being a move by the focal bird between individual trees or other vegetation, or every 30 m if the focal bird was moving along the ground (see Chapter 6 for full details on how a bird-stop location was defined).

To streamline the data collection at each bird-stop location, no attempt was made to distinguish between different foraging activities by the focal bird (i.e. a foraging bird was simply recorded as “foraging”). Also, if a bird foraged within a tree for one minute, then preened for one minute, and then continued foraging for one minute before moving to another tree, that bird was recorded as foraging at that location (i.e. only one bird behaviour was recorded for each bird-stop location). Only one height was recorded per bird-stop location, being the height at which the bird spent the longest time. For example, if the focal bird within a tree spent thirty seconds at two metres high, then one minute at three metres high and then 15 seconds at four metres high, the average height that was recorded was three metres. The average height estimate for each bird-stop location was constantly being amended as the focal bird was being observed, together with recording all other necessary information. By doing this, when the focal bird moved from a bird-stop location, all that was required was to record the time the bird vacated that location, proceed to that point to record the location with a GPS, and then immediately follow the bird/s. This reduced the chances of losing the birds and increased the time the birds were actually in view (and consequently being surveyed).

From the data above, I could provide a measure of the amount of time the birds spent at each location within a home range, together with the frequency with which those locations were being revisited by the birds. The percent of time allocated to foraging and average foraging heights were also calculated for each bird species. For the Hooded Robin, I also tallied the numbers of different foraging manoeuvres that the birds performed (e.g. pounce, glean, hawk). I also kept the data separate for males and females of each surveyed pair for the species where the sexes are easily distinguished (e.g. Hooded Robin, Rufous Whistler).

This was done to determine if there were any sex-related differences in habitat use by individuals of those species, such as differences in foraging heights. For sex-specific foraging height comparisons, I used chi-square analysis. When doing this, to assess if the data from the multiple minimum home ranges for individuals of a particular species could be combined and tested against the same null hypothesis, heterogeneity chi-square testing for goodness of fit was applied (Zar 2010). Should the null hypothesis that the data are homogeneous be rejected, it would not be appropriate to combine these datasets. In this case, the foraging height comparisons between the sexes were made for each individual home range. Chi-square was used to test for agreements and differences in the foraging heights between the sexes at the shrub (< 1 m), midstorey ($1 - 4$ m) and overstorey levels (> 4 m). Yates correction for continuity was applied when there was only one degree of freedom (Zar 2010). Any statistical differences in the foraging heights displayed between the sexes of individuals of a particular species was then identified by subdividing the chi-square goodness of fit (Zar 2010). This procedure identified which storey level/s contained significant disagreement between the observed and expected frequencies in the heights in which the birds were recorded to be foraging.

Plot sizes for habitat surveys

The primary objective of this study was to identify patches of habitat that the birds preferentially used within their individual minimum home ranges. The Point Density analysis (using ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006)) that was used to infer disproportionate habitat use by the birds, relied upon both the aggregations of GPS waypoints and the time spent per waypoint location (i.e. two combined datasets; see Chapter 7 for a detailed description of the Point Density analysis that was used). The sizes and orientations of the hot-spots deemed by the Point Density output were variable. The minimum areas of a hot-spot covered approximately $30\text{ m} \times 30\text{ m}$, but some extended to $100\text{ m} \times 60\text{ m}$ (see Chapter 7).

Given this, individual $30\text{ m} \times 30\text{ m}$ survey plots were placed in the centre over each hot-spot location. This scale is also consistent with the $30\text{ m} \times 30\text{ m}$ quadrat size used for native vegetation surveys in South Australia by the Department of Environment, Water and Natural Resources (Heard and Channon 1997). Multiple $30\text{ m} \times 30\text{ m}$ plots were assigned

to larger hot-spots where they could be accommodated, in an effort to survey as much hot-spot area as possible.

The same number of $30\text{ m} \times 30\text{ m}$ plots depicting cold-spots were also evaluated within each minimum home range area. Within home ranges where multiple $30\text{ m} \times 30\text{ m}$ plots were assigned to incorporate larger hot-spots, an equivalent number of multiple $30\text{ m} \times 30\text{ m}$ plots covering equally large cold-spots were also used. This was aimed at counteracting any bias of having multiple $30\text{ m} \times 30\text{ m}$ plots within only one of the plot types. Cold-spot survey locations were selected from low use habitat areas identified from the Point Density outputs (see Chapter 7). The rule that was applied when selecting cold-spot locations was that each $30\text{ m} \times 30\text{ m}$ cold-spot plot must contain at least one waypoint. This method differentiated the cold-spot habitat areas from the areas that were not used by the surveyed birds within the confines of their individual minimum home ranges. This way the habitat areas being compared (i.e. hot-spots vs. cold-spots) were identified as being available to the surveyed birds, which is more informative than simply comparing between used and unused habitat areas (Jones 2001). Habitat attributes influencing bird behaviours were assumed to be contained within the polar extremes of the Point Density output (Brown *et al.* 1995), defined here as hot-spots and cold-spots.

Habitat surveys and data analyses

Using ArcMap, the GPS co-ordinates for each corner of all $30\text{ m} \times 30\text{ m}$ plots were recorded to guide the boundary placements for the survey sites during the field surveys. In the field, survey boundaries were marked with measuring tapes. The $30\text{ m} \times 30\text{ m}$ plot was then subdivided into $5\text{ m} \times 5\text{ m}$ squares using more measuring tapes, producing 36, $5\text{ m} \times 5\text{ m}$ squares per plot.

I then visually estimated within each $5\text{ m} \times 5\text{ m}$ square the percent of any of the following that covered the ground – understorey and/or midstorey plants that physically obscured the ground, bare ground (area $> 20\text{ cm}$ radius), light litter (litter that did not fully cover the ground surface, but was associated with any other ground cover attribute), heavy litter (litter that fully covered the ground surface), fallen timber, lichen / moss, grasses (both native and introduced), weeds (other than exotic grasses) and herbs. See Chapter 3 for an

explanation as to why litter was segregated into two categories. Also included was any shrub cover below one metre (estimated percent cover within the 5 m × 5 m square). This was then proportioned against the plant species that were present. Data for each of these habitat attributes were averaged across the 36, 5 m × 5 m cells to produce a single value for each 30 m × 30 m plot.

To calculate the spatial density of shrubs below one metre, four 30 m line-intercept transects were performed within each 30 m × 30 m plot at 10 m intervals across the square (i.e. at 0 m, 10 m, 20 m and 30 m; adapted from Brower *et al.* 1998). Along each transect, measurements recorded the distances all vertical projections of shrubs intercepted with the measuring tape, to produce a linear coverage index (calculating the degree of floristic cover based upon the vertical projections of crown spread intercepts), expressed as:

$$IC_i = \frac{I_i}{L}$$

where I_i is the sum of intercept lengths per transect and L is the line-transect length (30 m).

The sum of crown spread intercepts were used to estimate the percent cover for shrubs within each 30 m × 30 m plot, expressed as:

$$\left(\frac{\sum_{i=0}^n (A+B+C+D)}{Z} \right) \times 100$$

where A is the sum of crown-spread intercepts along the 0 m line-intercept, B is that along the 10 m line-intercept, C is that along the 20 m line-intercept, D is that along the 30 m line-intercept, and Z is the total distance of line-intercept lengths (i.e. 120 m).

As in Chapter 3, vegetation taller than one metre was delineated as midstorey (1 – 4 m) and overstorey (> 4 m), however in this survey each plant was assessed individually. Only vegetation falling within the vertical confines of the plot boundary was measured, regardless of where the trunks for those plants were located. This meant that all vegetation falling outside of the plot boundary was ignored for a plant that had its trunk within the 30 m × 30 m plot, while vegetation that fell within the plot boundary was counted, even if the trunk for that plant was outside the plot. The species, its height, minimum canopy height,

canopy radii and number of visible tree hollows were recorded for each plant. Plant height was estimated by standing some metres away from the plant and counting the increments of a hand-held pencil against a predetermined height marked on that plant (refer Heard and Channon (1997), page 24). The minimum canopy height was calculated from the average of four measurements taken from each cardinal point of the compass, to nearest 0.5 m. The canopy radii was calculated from the average of four measurements taken radiating from the trunk in each cardinal point of the compass, to nearest 0.5 m.

These data were used to calculate:

- 1) Total abundance (sum of all plants at both the midstorey and overstorey levels within each 30 m × 30 m plot);
- 2) Abundance of each individual plant species at both the midstorey and overstorey levels within each 30 m × 30 m plot;
- 3) Relative abundance of each plant species (the number for each species as a percent of the total number of all plants, at both the midstorey and overstorey levels within each 30 m × 30 m plot);
- 4) Sum of vertical overstorey canopy heights (metres) for each species within each 30 m × 30 m plot used to calculate the total vertical canopy cover for each species within each plot, expressed as:

$$\sum_{i=0}^n (M_a - M_i)$$

where M_a is the maximum canopy height for a plant, M_i is the average minimum canopy height for that plant, and n is the number of plants of that particular species within a 30 m × 30 m plot;

- 5) Canopy cover of overstorey trees (total percent cover within each 30 m × 30 m plot), expressed as:

$$\left(\frac{\sum_{i=0}^n \pi \times (R)^2}{T} \right) \times 100$$

where R is the average canopy radius for a plant, T is the total area surveyed (i.e. 900 m^2), and n is the number of trees (including trees that overhung the plot) within the $30 \text{ m} \times 30 \text{ m}$ plot; and

- 6) Canopy cover per species within each $30 \text{ m} \times 30 \text{ m}$ plot (for both midstorey and overstorey plants), expressed as:

$$\left(\frac{\sum_{i=0}^n \pi \times (r)^2}{T} \right) \times 100$$

where r is the average canopy radius for a plant, T is the total area surveyed (i.e. 900 m^2), and n is the number of plants of that particular species within a $30 \text{ m} \times 30 \text{ m}$ plot.

Combined bird and habitat data analyses

The sequential recording of bird movements throughout the day produced data that were by nature temporally and spatially autocorrelated. However, data collected in this manner over multiple days should dilute or even eliminate the effects of autocorrelation (e.g. Robertson *et al.* 1998; Rooney *et al.* 1998; Fieberg 2007). More importantly, the resultant habitat data used to compare the habitat attributes between hot-spots and cold-spots were often plagued by zero values, which usually confounded the assumption of data normality and restricted any data transformations. Violations of the assumptions of normality and equal variance severely restricted the use of parametric statistics.

I instead used Classification Tree Analysis (CTA) using the rpart functions in R (Hornkil 2012) to show the habitat differences between hot-spots and cold-spots for each of the nine bird species that were studied. This procedure was used to analyse combined datasets for individuals of each species, which incorporated multiple sites. This non-parametric analysis makes no assumptions about the frequency distribution of the data and is robust against correlations among variables, non-Gaussian distributions, non-linear relationships, high-order interactions and missing and/or outlying values (Moore *et al.* 1991; De'ath and Fabricius 2000). The recursive structure of this model is also ideal for uncovering complex dependencies among predictor variables (McCune and Grace 2002). It can also capture important relationships which make ecological sense that may not be as evident when using conventional linear models in statistics (McCune and Grace 2002). The CTA-

model is also strong in its ability to detect interactions between factors (in this study, the habitat variables), is highly intuitive and is a valuable tool for modelling species distributions (Vayssières *et al.* 2000).

Classification Tree Analysis partitions the data set recursively to produce a hierarchical tree-like classification. The CTA algorithm works by selecting one habitat variable at a time and repetitiously partitioning paired values from the data series for that variable. At each partition the model computes an estimate of the within-group heterogeneity, using a log-likelihood function (McCune and Grace 2002). This search culminates with two mutually exclusive groups which are as homogeneous as possible within a grouping (De'ath and Fabricius 2000; McCune and Grace 2002). The recursive partitioning then continues in a similar manner until the best variable is identified for the first branch of the tree. The recursive algorithm is stopped when the tree reaches a maximum homogeneity or when no further improvements can be made based on the data that are provided (Therneau and Atkinson 1997). Each node on a branch of the tree represents the most homogeneous data split. A distinctive feature of the CTA-model is that only the single best predictor (i.e. habitat variable) is identified as an indicator variable at each node. Importantly, as the tree is built the same habitat variable may appear multiple times, highlighting the non-linear nature of this modelling process.

The first argument in the rpart tree-building function to produce a classification tree is the model formula. To explore the data, the categorical variables “hot-spots” and “cold-spots” were modelled against the habitat attributes that were measured individually for both patch types. The type of splitting rule used for the classification trees was “method = class”.

Two advisory parameters control how the data are then split between the nodes.

“Minsplit” controls the minimum number of observations in a node before a split is computed and the complexity parameter (cp) controls the number of splits (Therneau and Atkinson 1997). The default settings for minsplit (20) and cp (0.01) were retained as together they produce a near optimal tree which effectively comes as the most favoured “pre-pruned” solution (Therneau and Atkinson 1997). For example, the cost-complexity pruning table generated from the Hooded Robin data in this study depicts three trees of increasing complexity (circles with error bars) that are plotted against a decreasing relative error rate (Fig. 8.1). The dotted horizontal line represents one standard error above the minimum error rate. By convention, the error value corresponding to the first tree that falls

below this dotted line is used to prune the tree (McCune and Grace 2002). In this example, the optimal classification tree comes pre-pruned at a cp of 0.039 (Fig. 8.1).

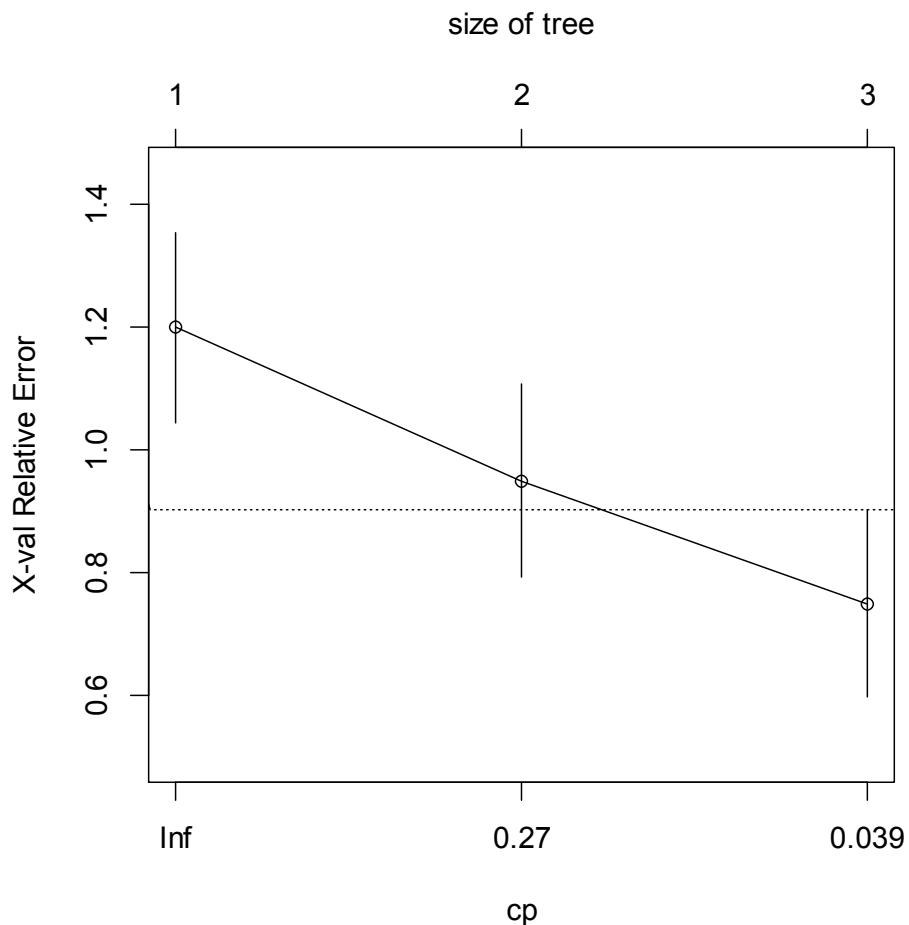


Figure 8.1: Cost-complexity pruning curve generated for a classification tree using the rpart functions in R (Hornkil 2012). This output shows the relative error rate plotted against three trees of increasing complexity (circles with error bars). The dotted horizontal line represents one standard error above the minimum error rate. Relative error is calculated by cross-validation. The complexity parameter (cp) controls the number of splits that were used to help produce the trees.

Tree models using lower settings of the minsplit and cp default values always result in statistical over-fitting as the error is fitted along with the data (Therneau and Atkinson 1997). This leads to additional branches in the tree where false and/or chance relationships appear to be significant (McCune and Grace 2002). To remedy this, “pruning” of the tree via cross-validation of subsets of the data is used to simplify the model to one that reduces tree complexity, while holding classification error to a minimum (Moore *et al.* 1991). Cross-validation is efficient at pruning out “pure noise” in the data; however this procedure

is not optimal as it produces a very conservative result which often eliminates some pertinent data (Therneau and Atkinson 1997).

To indicate the misclassification rate for hot-spots and cold-spots in the classification trees in this study, the “use.n = TRUE, all = TRUE” command was applied. This assigns the number of hot-spots and cold-spots that corresponded to each and every node on the classification tree. Each node on the tree also contains the habitat variable and the splitting criteria that were used to create that node.

In order to effectively read the classification tree, move to the left branch when the stated splitting condition is true. The order in which the variables appear on the tree can be interpreted like a dichotomous key. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spots from cold-spots. Variables appearing near the top of the tree explain more of the variation between hot-spots and cold-spots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathways that define either hot-spots or cold-spots. The tree is arranged with the branches that contain the largest average class being positioned to the right.

Results

Disproportionate habitat use within home ranges

The birds that were surveyed of the nine species all displayed disproportionate habitat used within their minimum home range areas. From the Point Density analyses, minimum home range areas typically contained 3.6 ± 0.4 hot-spots of varying sizes, within which the habitats were surveyed at the scale of $30\text{ m} \times 30\text{ m}$ (Table 8.1). Comparisons of habitat within hot-spot plots versus the same number of equally sized cold-spot plots for the birds of each species revealed differences in the habitat features that were contained within the two plot types.

Table 8.1: The number of 30 m × 30 m survey plots associated with sites interpreted to be high use habitat areas (i.e. hot-spots) within individual minimum home ranges of nine woodland bird species. Hot-spots were identified via Point Density analysis using ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006). The input data for the Point Density analysis used a combination of two sets of data to produce the Point Density output. These datasets were the GPS waypoint location data and the time spent per waypoint data. To combine these spatial and temporal datasets together, each dataset had to first be reclassified to a common measurement. This was achieved using the Spatial Analysis Tool in ArcMap. The two reclassified datasets were then combined using Weighted Overlay in ArcMap. These data were collected from locations within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected April – September 2009.

| Species | Location | # Hot-spots | # 30 m × 30 m plots |
|-------------------------|----------------------|-------------|---------------------|
| Brown Treecreeper | Altona Chatterton | 4 | 4 |
| Brown Treecreeper | Altona Kies | 7 | 8 |
| Brown Treecreeper | Altona Torbreck | 1 | 4 |
| Buff-rumped Thornbill | Para Wirra - Site A | 2 | 3 |
| Buff-rumped Thornbill | Para Wirra - Site B | 2 | 4 |
| Hooded Robin | Altona Chatterton | 8 | 9 |
| Hooded Robin | Altona Kies | 4 | 7 |
| Hooded Robin | Sandy Creek | 3 | 4 |
| Rufous Whistler | Altona Kies | 4 | 4 |
| Rufous Whistler | Altona Landcare | 3 | 5 |
| Rufous Whistler | Sandy Creek | 3 | 4 |
| White-browed Babbler | Altona Chatterton | 7 | 12 |
| White-browed Babbler | Altona Kies | 4 | 5 |
| White-browed Babbler | Sandy Creek | 1 | 2 |
| White-winged Chough | Para Wirra | 3 | 8 |
| White-winged Chough | Sandy Creek | 10 | 23 |
| Willie Wagtail | Altona Chatterton | 1 | 4 |
| Willie Wagtail | Altona Kies - Site A | 5 | 5 |
| Willie Wagtail | Altona Kies - Site B | 2 | 2 |
| Willie Wagtail | Sandy Creek | 3 | 5 |
| Yellow-rumped Thornbill | Altona Chatterton | 4 | 6 |
| Yellow-rumped Thornbill | Altona Kies | 3 | 5 |
| Yellow Thornbill | Altona Chatterton | 2 | 2 |
| Yellow Thornbill | Altona Kies | 5 | 5 |
| Yellow Thornbill | Para Wirra - Site A | 2 | 2 |
| Yellow Thornbill | Para Wirra - Site B | 3 | 4 |
| Yellow Thornbill | Sandy Creek - Site A | 3 | 5 |
| Yellow Thornbill | Sandy Creek - Site B | 3 | 3 |

Brown Treecreeper

Hot-spot plots for the Brown Treecreepers that were surveyed contained three or more tree hollows (Fig. 8.2). Conversely, cold-spot plots contained few tree hollows and were higher in the density of midstorey vegetation (Fig. 8.2). Open vegetation near the ground level correlated with these birds collectively averaging 56% of their foraging time on the ground and on fallen timber (33 h 17 min from $n = 623$ records). The remaining 44% of their foraging time was spent on the trunks and branches of vegetation (25 h 43 min from $n = 822$ records). In all three minimum home ranges, rough-barked tree species were used as a foraging substrate the most; namely Drooping Sheoak (*Allocasuarina verticillata*), Peppermint Box (*Eucalyptus odorata*) and Slender Cypress-pine (*Callitris gracilis*; Table 8.2).

In this study, the Brown Treecreepers used hollows in Peppermint Box, Pink Gum (*E. fasciculosa*) and South Australian Blue Gum (*E. leucoxylon*), as well as open fissures in mature Slender Cypress-pine. Hollows were a source of refuge for alarmed birds and were used exclusively for roosting. Birds roosted close together in areas with high hollow densities, but at Altona Kies where hollows were sparser, the birds separated at dusk and roosted solitarily. Survey plots at Altona Kies collectively contained 33 tree hollows, equating to 1.6 hollows ha^{-1} within this minimum home range area. Survey plots at Altona Torbreck collectively contained 36 tree hollows, equating to 4.1 hollows ha^{-1} within this minimum home range area. Survey plots at Altona Chatterton collectively contained 15 tree hollows, equating to 2.7 hollows ha^{-1} within this minimum home range area.

Classification Tree for Brown Treecreeper

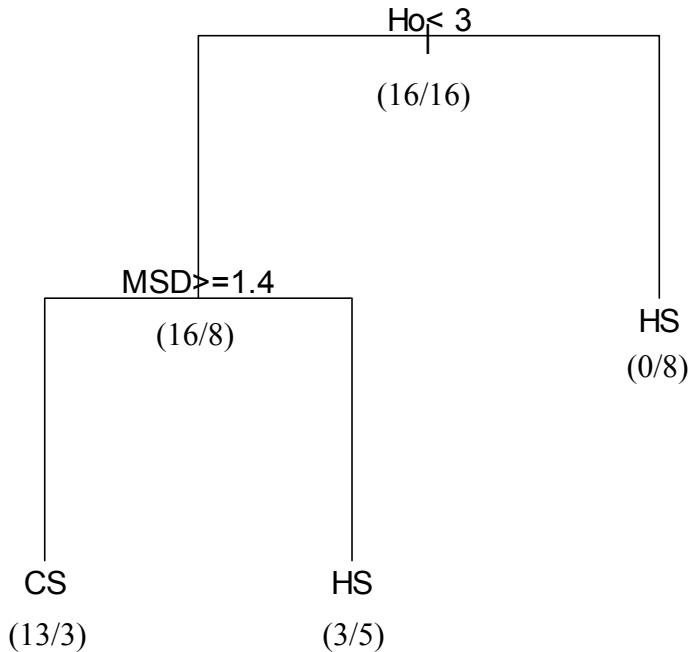


Figure 8.2: Brown Treecreeper (*Climacteris picumnus*) classification tree for two site types on two habitat variables (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). The site types are 30 m × 30 m survey plots measured for hot-spots (HS, $n = 16$) and 30 m × 30 m survey plots measured for cold-spots (CS, $n = 16$). The habitat variables are the number of tree hollows (Ho) and midstorey density (MSD). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spot plots from cold-spot plots. Variables appearing near the top of the tree explain more of the variation between hot-spot plots and cold-spot plots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either a hot-spot plot or a cold-spot plot. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #CS / #HS). Surveys were conducted at Altona Chatterton, Altona Kies and Altona Torbreck, within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected April – July 2009.

Table 8.2: Foraging substrates and plant species used by individuals of Brown Treecreepers (*Climacteris picumnus*) at three sites within the Altona woodland. The values are the percent of time spent foraging on particular substrates and various plant species, together with the observation times for each site on which the percent values were calculated (in hours:minutes). *n* = number of foraging records. These data show that these birds foraged on the ground and on fallen timber for more than half of the time spent foraging within each of the three sites. The time spent foraging on vegetation was also biased towards rough-barked plant species within all three sites (highest percent of time spent foraging in bold). Surveys were conducted within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected April – July 2009.

| Foraging substrates | Altona Chatterton | | Altona Kies | | Altona Torbreck | |
|--|--------------------------|----------|----------------------|----------|------------------------|----------|
| | % of time (h:min) | <i>n</i> | % of time (h:min) | <i>n</i> | % of time (h:min) | <i>n</i> |
| Ground / Fallen timber | 47 (9:04) | 180 | 67 (13:15) | 222 | 55 (10:58) | 221 |
| Rough-barked plant species | | | | | | |
| Drooping Sheoak (<i>Allocasuarina verticillata</i>) | 29 (5:40) | 198 | 1 (0:03) | 2 | 3 (0:40) | 27 |
| Golden Wattle (<i>Acacia pycnantha</i>) | 1 (0:11) | 8 | – | | 4 (0:50) | 36 |
| Olive (<i>Olea europaea</i>) | 2 (0:19) | 10 | – | | – | |
| Peppermint Box (<i>Eucalyptus odorata</i>) | 4 (0:49) | 26 | 0.3 (0:04) | 1 | 16 (3:09) | 91 |
| Slender Cypress-pine (<i>Callitris gracilis</i>) | 1 (0:14) | 8 | 18 (3:33) | 104 | 2 (0:28) | 17 |
| Sticky Hop-bush (<i>Dodonaea viscosa</i>) | 1 (0:10) | 8 | – | | 0.1 (0:01) | 1 |
| Swamp Wattle (<i>Acacia retinodes</i>) | – | | 0.3 (0:03) | 3 | 2 (0:22) | 18 |
| Of all trees, total foraging on rough-barked species | 72 (7:22) | 258 | 54 (3:36) | 110 | 62 (5:30) | 190 |
| Smooth-barked plant species | | | | | | |
| Pink Gum (<i>Eucalyptus fasciculosa</i>) | 12 (2:16) | 77 | 6 (1:38) | 45 | 13 (2:37) | 75 |
| River Red Gum (<i>Eucalyptus camaldulensis</i>) | – | | 1 (0:09) | 2 | 1 (0:09) | 5 |
| SA Blue Gum (<i>Eucalyptus leucoxylon</i>) | 3 (0:32) | 8 | 6 (1:17) | 31 | 3 (0:38) | 22 |
| Of all trees, total foraging on smooth-barked species | 28 (2:48) | 85 | 46 (3:04) | 78 | 38 (3:23) | 102 |

Buff-rumped Thornbill

No classification tree could be produced from the habitat data that were collected pertaining to the Buff-rumped Thornbills that were surveyed. These birds spent 56% of the time foraging on the ground and within the shrub layer (14 h 11 min from $n = 459$ records), 29% of the time foraging within the midstorey strata (7 h 19 min from $n = 297$ records) and 15% of the time foraging within the overstorey strata (3 h 41 min from $n = 164$ records).

Hooded Robin

Both male and female Hooded Robins predominantly foraged by pouncing on prey that were located on the ground (97% of the time from 1,899 foraging records) from low perching substrates (1.4 ± 0.02 m). There was no foraging height disparity between the sexes, and it was not uncommon for both birds of a pair to forage from the same perching substrate. Compared to cold-spot plots, hot-spot plots for these birds contained a greater cover of high litter and less cover of midstorey vegetation (Fig. 8.3). Midstorey vegetation within cold-spot plots were dominated by Kangaroo Thorn (*Acacia paradoxa*) at Altona and Scarlet Bottlebrush (*Callistemon rugulosus*) at Sandy Creek.

Classification Tree for Hooded Robin

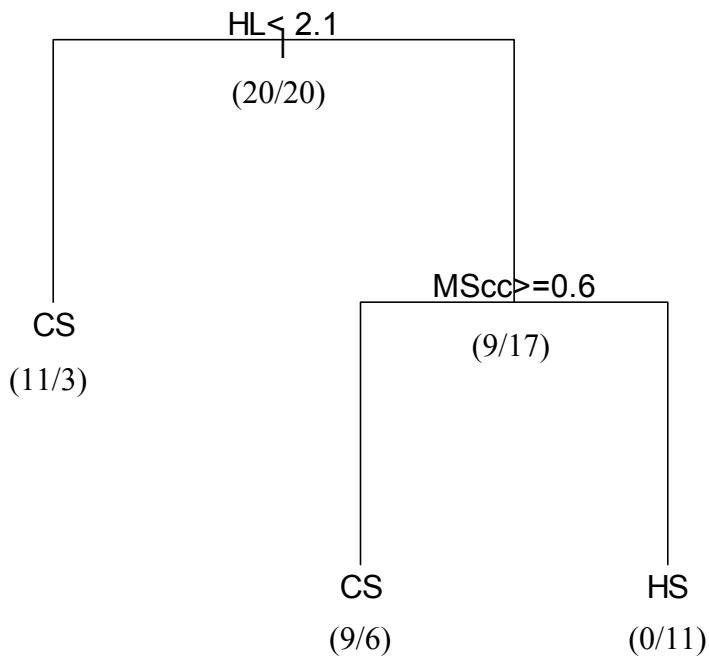


Figure 8.3: Hooded Robin (*Melanodryas cucullata*) classification tree for two site types on two habitat variables (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). The site types are 30 m × 30 m survey plots measured for hot-spots (HS, $n = 20$) and 30 m × 30 m survey plots measured for cold-spots (CS, $n = 20$). The habitat variables are the percent of high litter cover (HL) and midstorey canopy cover (MScc). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spot plots from cold-spot plots. Variables appearing near the top of the tree explain more of the variation between hot-spot plots and cold-spot plots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either a hot-spot plot or a cold-spot plot. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #CS / #HS). Surveys were conducted at Altona Chatterton, Altona Kies and Sandy Creek, within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected May – August 2009.

Rufous Whistler

Relatively few Rufous Whistlers remained within the survey region over winter. Consequently, surveys were conducted in late August and September 2009, to coincide with the annual return of this species back into the study region to breed. In this survey, male birds were recorded foraging for a total of 18 h 32 min and females for a total of 9 h 13 min. The time bias towards the male birds resulted from males being more conspicuous (vocal) and thus more easily tracked. Every opportunity was taken to survey female birds. However, at Altona Kies observations on the female were opportunistic so as to not adversely disturb her nest building activities.

A disparity existed in the foraging heights between the sexes of the Rufous Whistlers that were surveyed, with females foraging at lower heights than their male counterparts. The foraging data for the three surveyed pairs indicated that the time spent foraging by females peaked at around one metre and for males foraging peaked at around six metres (Fig. 8.4). The null hypothesis using heterogeneity chi-square testing for goodness of fit was rejected (heterogeneity chi-square = 110.1 vs. $\chi^2_{0.05,5} = 11.07$). Accordingly, the foraging data for individual Rufous Whistler minimum home ranges were analysed separately. Insufficient data were recorded between the sexes at Altona Kies to analyse gender specific habitat use. At Altona Landcare, the number of occasions the female used the ground and shrubs ($\chi^2_{0.05,1} = 25.4, P = < 0.0001$) and midstorey vegetation ($\chi^2_{0.05,1} = 11.7, P = 0.0006$) was statistically greater than that used by the male. At Sandy Creek, the female was also statistically more often found foraging on the ground and in the shrub layer than the male ($\chi^2_{0.05,1} = 6.86, P = 0.009$).

Independent Point Density analyses for the male and female Rufous Whistlers that were surveyed indicated that some hot-spots were exclusive to a particular gender. However, there was insufficient data on exclusive habitat use by females in general for habitat use between the sexes to be analysed separately. Instead, habitat use was pooled and analysed to identify the habitat areas that both sexes frequented the most. This analysis showed that more than half of the hot-spot plots contained greater canopy cover at the midstorey level (Fig. 8.5).

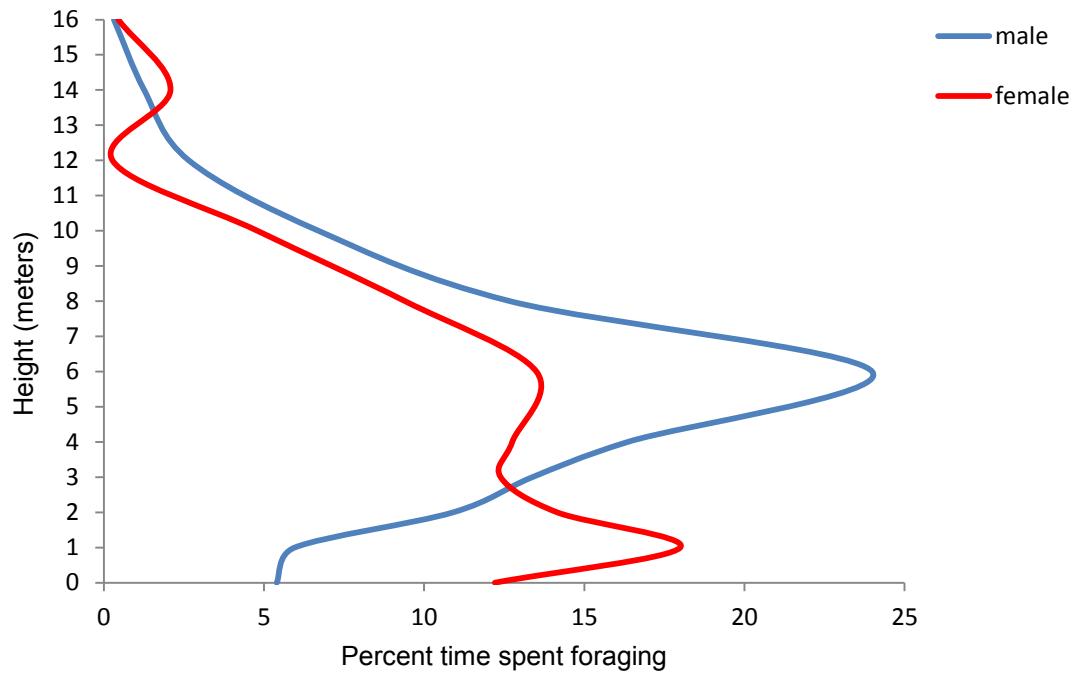


Figure 8.4: Comparison of foraging heights between male and female Rufous Whistlers (*Pachycephala rufiventris*), using combined data from sites at Altona Kies, Altona Landcare and Sandy Creek. A pair of birds was surveyed at each location. Male birds were recorded foraging for a total of 18 h 32 min and females for a total of 9 h 13 min. These data represent the percent of time that males and females were recorded foraging at one metre height increments. The height data have been categorised, with the taller heights containing data that ranges from the previous value to the stated value (e.g. the 1 m height increment includes height data that ranged from 0.1 – 1.0 m). These data indicate that the time spent foraging by females peaked at around 1 m (red line) and for males foraging peaked at around 6 m (blue line). Data collected late August – September 2009.

Classification Tree for Rufous Whistler

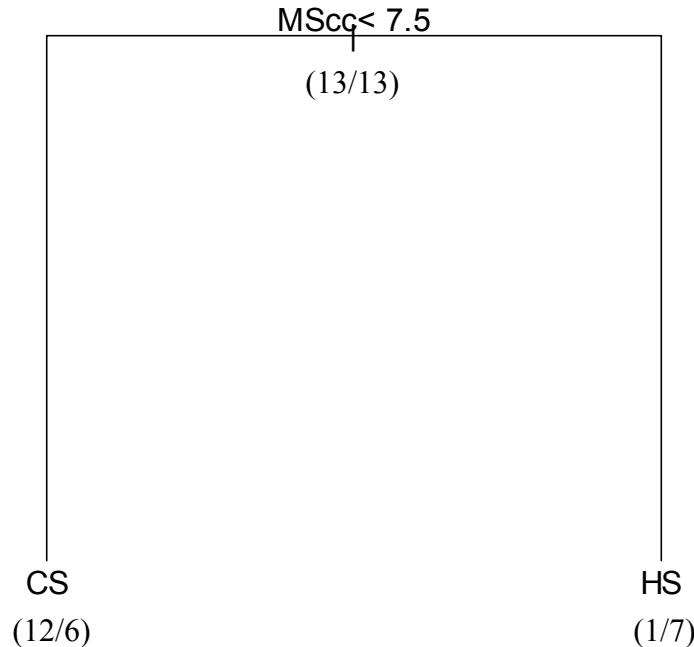


Figure 8.5: Rufous Whistler (*Pachycephala rufiventris*) classification tree for two site types on a single habitat variable (using the `rpart` function in R (Hornkil 2012) with `minsplit = 20` and complexity parameter = 0.01). The site types are 30 m × 30 m survey plots measured for hot-spots (HS, $n = 13$) and 30 m × 30 m survey plots measured for cold-spots (CS, $n = 13$). The habitat variable is the percent of canopy cover at the midstorey level (MScc). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spot plots from cold-spot plots. Variables appearing near the top of the tree explain more of the variation between hot-spot plots and cold-spot plots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either a hot-spot plot or a cold-spot plot. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #CS / #HS). Surveys were conducted at Altona Kies, Altona Landcare and Sandy Creek, within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected late August – September 2009.

White-browed Babbler

The White-browed Babblers that were surveyed regularly used Kangaroo Thorn and Slender Cypress-pine for roost-nest placement and to seek refuge when alarmed. These habitat features were also strongly represented in the hot-spot plots associated with these birds (Fig. 8.6). This information was then used to predict habitat quality for the White-browed Babbler at another nearby location. Unlike the sites surveyed within the Altona woodland, only a few White-browed Babblers were detected within the Sandy Creek Conservation Park. To assess whether Sandy Creek has extensive areas suitable for the White-browed Babbler, the habitat here was compared to habitat at Altona, where numerous groups of White-browed Babblers resided. For Sandy Creek, the hot-spot and cold-spot plot data ($n = 46$) previously generated from the White-winged Chough surveys within this Park were used. The White-winged Chough data were desirable, as this species was wide ranging across this Park and the habitat areas that they covered rarely overlapped with that used by the few White-browed Babblers detected here. This then provided a measure of habitat that may be suitable for the White-browed Babbler at Sandy Creek. For the Altona woodland, data were pooled for the hot-spot and cold-spot plots generated from the White-browed Babblers that were surveyed at Altona Chatterton and Altona Kies ($n = 34$). The number of sites used for Sandy Creek was reduced to 34 to create an equal sample size for comparison between the two sites. A table of random numbers was used to select the plots to be analysed (Zar 2010). The Sandy Creek plots contained no Kangaroo Thorn and did not contain patches of high density Slender Cypress-pine (Fig. 8.7).

Classification Tree for White-browed Babbler

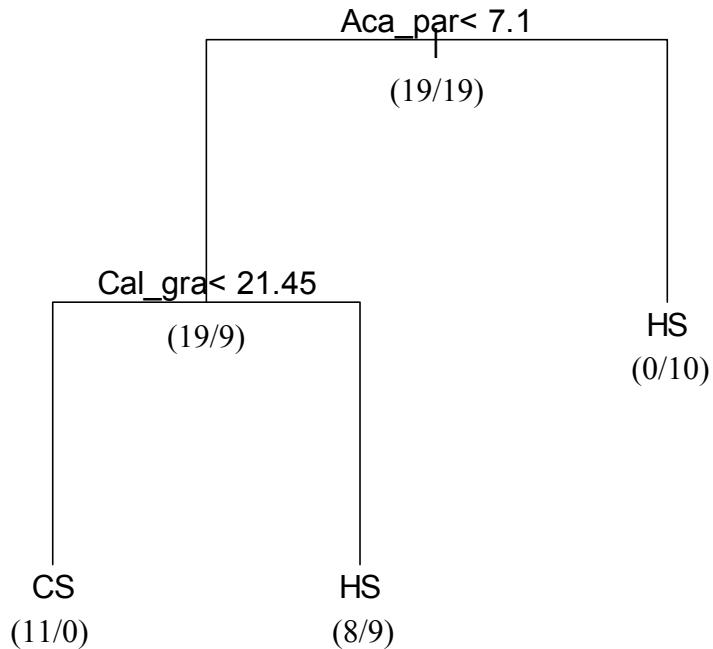


Figure 8.6: White-browed Babbler (*Pomatostomus superciliosus*) classification tree for two site types on two habitat variables (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). The site types are 30 m × 30 m survey plots measured for hot-spots (HS, $n = 19$) and 30 m × 30 m survey plots measured for cold-spots (CS, $n = 19$). The habitat variables are the percent canopy cover of Kangaroo Thorn (*Acacia paradoxa*; Aca_par) and the percent canopy cover of Slender Cypress-pine (*Callitris gracilis*; Cal_gra). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spot plots from cold-spot plots. Variables appearing near the top of the tree explain more of the variation between hot-spot plots and cold-spot plots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either a hot-spot plot or a cold-spot plot. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #CS / #HS). Surveys were conducted at Altona Chatterton, Altona Kies and Sandy Creek, within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected April – August 2009.

Altona vs Sandy Creek for White-browed Babbler

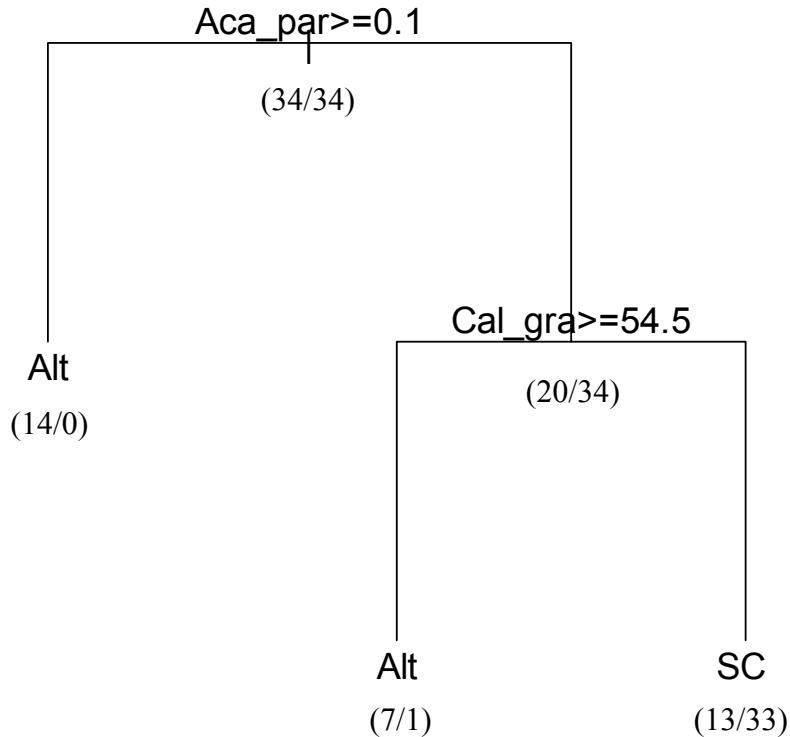


Figure 8.7: Graphical location classification tree for two site types on two habitat variables (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). These data pertain to habitat use by the White-browed Babbler (*Pomatostomus superciliosus*). The site types are 30 m × 30 m survey plots measured within the Altona woodland (Alt, n = 34) and 30 m × 30 m survey plots measured within the Sandy Creek Conservation Park (SC, n = 34). The habitat variables are the percent canopy cover of Kangaroo Thorn (*Acacia paradoxa*; Aca_par) and the percent canopy cover of Slender Cypress-pine (*Callitris gracilis*; Cal_gra). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating the habitat at Altona from the habitat at Sandy Creek. Variables appearing near the top of the tree explain more of the variation between the sites at Altona and the sites at Sandy Creek than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either the Altona site or the Sandy Creek site. The numbers in brackets specify the number of 30 m × 30 m plots at each node (read as #Alt / #SC). Surveys were conducted within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected April – August 2009.

White-winged Chough

Compared to cold-spot plots, hot-spot plots for the White-winged Choughs that were surveyed were low in understorey vegetation and high in litter cover (Fig. 8.8). The cold-spot plots that were high in understorey cover were dominated by ground-obscuring plant species, such as Spiky Guinea-flower (*Hibbertia exutiacies*) at Para Wirra and Sandhill Sword-sedge (*Lepidosperma concavum*) and Yacca (*Xanthorrhoea semiplana*) at Sandy Creek. When the White-winged Choughs foraged along vegetation edges and in more open habitats, they were often attacked by Australian Magpies (*Gymnorhina tibicen*). These attacks usually resulted in the White-winged Choughs seeking cover within nearby vegetation. Despite the rate and ferocity of these attacks however, individuals of White-winged Choughs repeatedly visited these locations often enough for some to be considered as hot-spots by the Point Density analysis.

The White-winged Chough was reliably found within the Sandy Creek Conservation Park during both the 2009 and 2010 bird surveys, inferring residency. Some five kilometres to the north-east in the Altona woodland, birds of this species were only ever encountered twice (seven birds in early and late May 2009), suggesting itinerant use of this nearby woodland. The group at Altona was assumed to be the same on both occasions, as one bird had a noticeable limp. To assess for any habitat differences between these two sites relative to habitat use by the White-winged Chough, representative samples of habitat type for both locations were used to compare them. For Sandy Creek, hot-spot and cold-spot plot data previously collected for the White-winged Chough within this Park were considered ($n = 46$). The transient White-winged Choughs at Altona used habitat at both Altona Kies and Altona Chatterton. Both of these sites at Altona hosted widespread groups of White-browed Babblers. Accordingly, hot-spot and cold-spot plot data previously collected for White-browed Babblers at both Altona sites ($n = 34$) were used as representative samples of the habitat type at Altona. This then provided a measure of habitat that may be suitable for the White-winged Chough at Altona. The number of sites used for Sandy Creek was reduced to 34 to create an equal sample size for comparison between the Sandy Creek and Altona sites. A table of random numbers was used to select the plots to be analysed (Zar 2010). The defining differences between these two locations were that the Sandy Creek plots contained a higher percent of *Eucalyptus* spp canopy cover and more high litter cover. Compared to Altona plots, Sandy Creek plots then also had a

greater contrast in habitat structure by having more open areas (as inferred by a lower canopy cover of *Eucalyptus* spp and lower accumulated litter on the ground (Fig. 8.9)).

Classification Tree for White-winged Chough

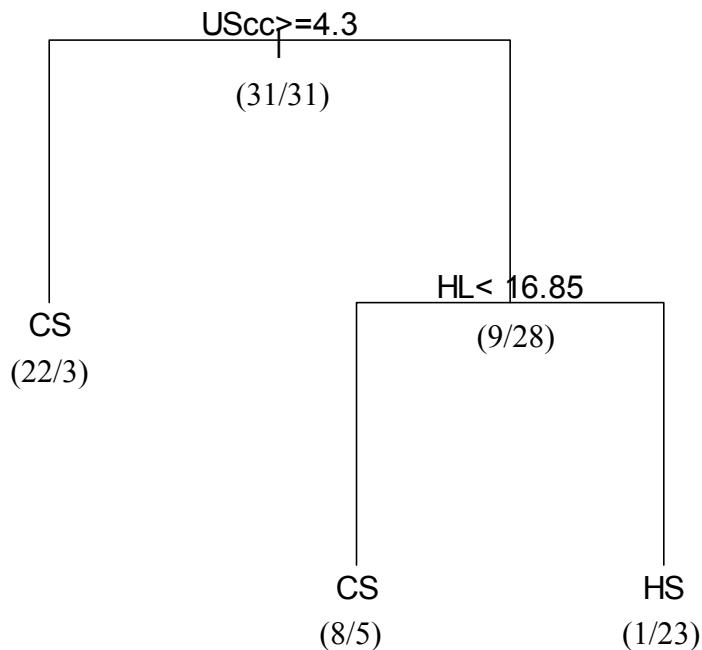


Figure 8.8: White-winged Chough (*Corcorax melanorhamphos*) classification tree for two site types on two habitat variables (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). The site types are 30 m × 30 m survey plots measured for hot-spots (HS, $n = 31$) and 30 m × 30 m survey plots measured for cold-spots (CS, $n = 31$). The habitat variables are the percent of understorey canopy cover (UScc) and the percent of high litter cover (HL). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spot plots from cold-spot plots. Variables appearing near the top of the tree explain more of the variation between hot-spot plots and cold-spot plots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either a hot-spot plot or a cold-spot plot. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #CS / #HS). Surveys were conducted at Para Wirra and Sandy Creek, within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected June – August 2009.

Sandy Creek vs Altona for White-winged Chough

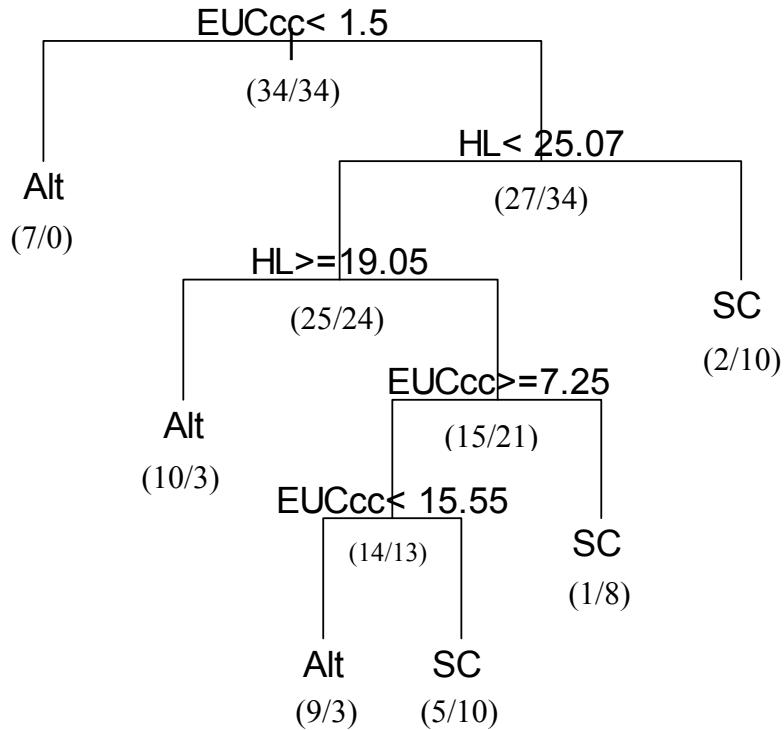


Figure 8.9: Graphical location classification tree for two site types on two habitat variables (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). All but one of the branches in this tree was above one standard error of the minimum error rate in the cost-complexity pruning table. This tree therefore is a robust representation of the habitat differences between the two sites. These data pertain to habitat use by the White-winged Chough (*Corcorax melanorhamphos*). The site types are 30 m × 30 m survey plots measured with the Altona woodland (Alt, $n = 34$) and 30 m × 30 m survey plots measured within the Sandy Creek Conservation Park (SC, $n = 34$). The habitat variables are the percent of *Eucalyptus* spp canopy cover (EUCcc) and percent of high litter cover (HL). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating the habitat at Sandy Creek from the habitat at Altona. Variables appearing near the top of the tree explain more of the variation between the sites at Sandy Creek and the sites at Altona than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either the Sandy Creek site or the Altona site. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #Alt / #SC). Surveys were conducted within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected April – August 2009

Willie Wagtail

The hot-spot plots for the Willie Wagtails that were surveyed contained densities of between 20 – 54% in overstorey vegetation (Fig. 8.10). Conversely, cold-spot plots contained densities of either less than 20% or greater than 54% in overstorey vegetation (Fig. 8.10). In this study, Willie Wagtails either actively foraged on the ground or conducted static searches from low vegetation. For example, at Altona Chatterton the birds actively searched the ground 69% of the time (12 h 20 min from $n = 244$ records) and conducted static searches 31% of the time (5 h 34 min from $n = 113$ records) from low perching substrates (0.7 ± 0.03 m). At Altona Kies Site B the birds actively searched the ground 70% of the time (8 h 20 min from $n = 210$ records) and conducted static searches 30% of the time (3 h 29 min from $n = 77$ records) from low perching substrates (1.1 ± 0.1 m). At Sandy Creek the solitary bird actively searched the ground 69% of the time (6 h 28 min from $n = 254$ records) and conducted static searches 26% of the time (2 h 27 min from $n = 90$ records) from low perching substrates (0.9 ± 0.08 m). This bird also gleaned and hawked in the early morning high (6.4 ± 0.3 m) among the sunlit canopy of Peppermint Box for 5% of the time (28 min from $n = 17$ records). However at Altona Kies Site A, foraging was dominated by the birds conducting static searches 79% of the time (6 h 40 min from $n = 100$ records) from low perching substrates (1.3 ± 0.1 m) and then actively searching the ground 21% of the time (1 h 47 min from $n = 45$ records). Altona Kies Site A was mostly dominated by a dense ground cover of Perennial Veldt-grass (*Ehrharta calycina*).

Classification Tree for Willie Wagtail

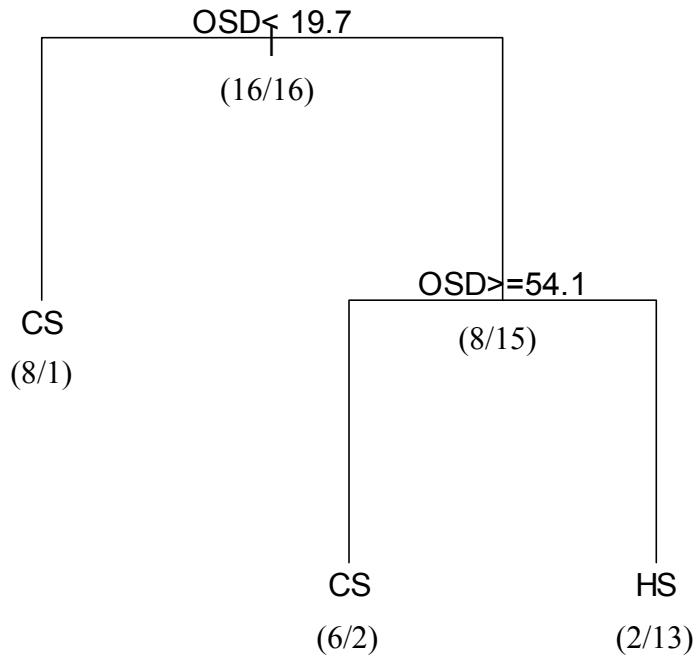


Figure 8.10: Willie Wagtail (*Rhipidura leucophrys*) classification tree for two site types on a single habitat variable (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). The site types are 30 m × 30 m survey plots measured for hot-spots (HS, $n = 16$) and 30 m × 30 m survey plots measured for cold-spots (CS, $n = 16$). The habitat variable is overstorey density (OSD), which occupies two branches on the classification tree. To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spot plots from cold-spot plots. Variables appearing near the top of the tree explain more of the variation between hot-spot plots and cold-spot plots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either a hot-spot plot or a cold-spot plot. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #CS / #HS). Surveys were conducted at Altona Chatterton, Altona Kies (two home ranges) and Sandy Creek, within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected late May – August 2009.

Yellow-rumped Thornbill

The Yellow-rumped Thornbills that were surveyed were easily spooked when foraging on the ground and readily sought cover when alarmed, typically retreating into Slender Cypress-pine. These birds also used Slender Cypress-pine as a foraging substrate. For example, although the birds at Altona Kies foraged on the ground 100% of the time (15 h 56 min from $n = 357$ records), the birds at Altona Chatterton foraged on the ground 70% of the time (13 h 8 min from $n = 367$ records) and within vegetation 30% of the time (5 h 40 min from $n = 177$ records). Of the plant species that were used for foraging by these birds, foraging effort was biased towards Slender Cypress-pine 40% of the time (5 h 17 min from $n = 88$ vegetation foraging records) and Peppermint Box 24% of the time (1 h 20 min from $n = 26$ vegetation foraging records). For the Altona Chatterton birds, foraging heights within vegetation ranged between one and eight metres (average 3.3 ± 0.1 m). As an indicator of an important habitat attribute for these birds, hot-spot plots were higher in the percent of canopy cover of Slender Cypress-pine (Fig. 8.11).

Classification Tree for Yellow-rumped Thornbill

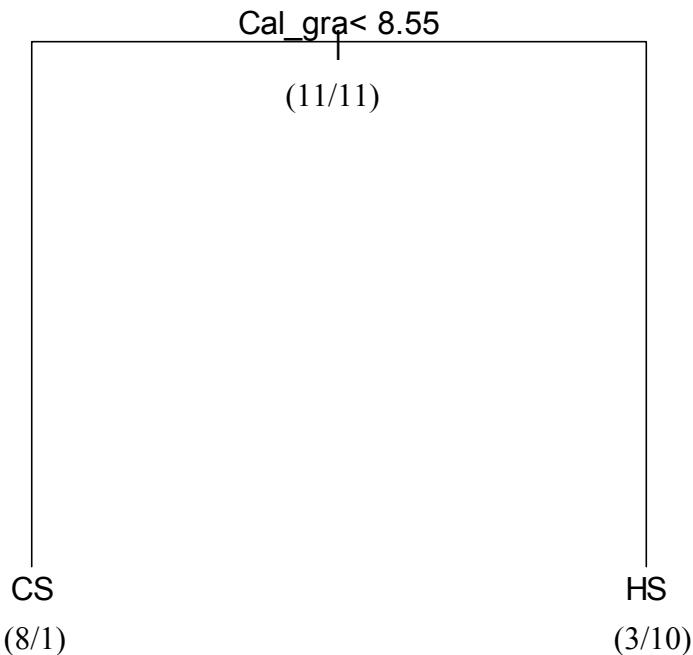


Figure 8.11: Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*) classification tree for two site types on a single habitat variable (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). The site types are 30 m × 30 m survey plots measured for hot-spots (HS, $n = 11$) and 30 m × 30 m survey plots measured for cold-spots (CS, $n = 11$). The habitat variable is the percent canopy cover of Slender Cypress-pine (*Callitris gracilis*; Cal_gra). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spot plots from cold-spot plots. Variables appearing near the top of the tree explain more of the variation between hot-spot plots and cold-spot plots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either a hot-spot plot or a cold-spot plot. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #CS / #HS). Surveys were conducted at Altona Chatterton and Altona Kies, within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected May 2009.

Yellow Thornbill

Hot-spot plots for the Yellow Thornbills that were surveyed principally contained greater canopy cover of Slender Cypress-pine and secondarily contained higher canopy cover of other tree species, such as Pink Gum (Fig. 8.12). These birds showed a strong affiliation for using Slender Cypress-pine as a foraging substrate, collectively foraging on this plant species 73% of the time (from a total 61 h 25 min of recorded foraging time; Table 8.3). For these birds, foraging in Slender Cypress-pine was prominent at all of the sites where this tree species occurred (Table 8.3), irrespective of the other plant species that were present.

Classification Tree for Yellow Thornbill

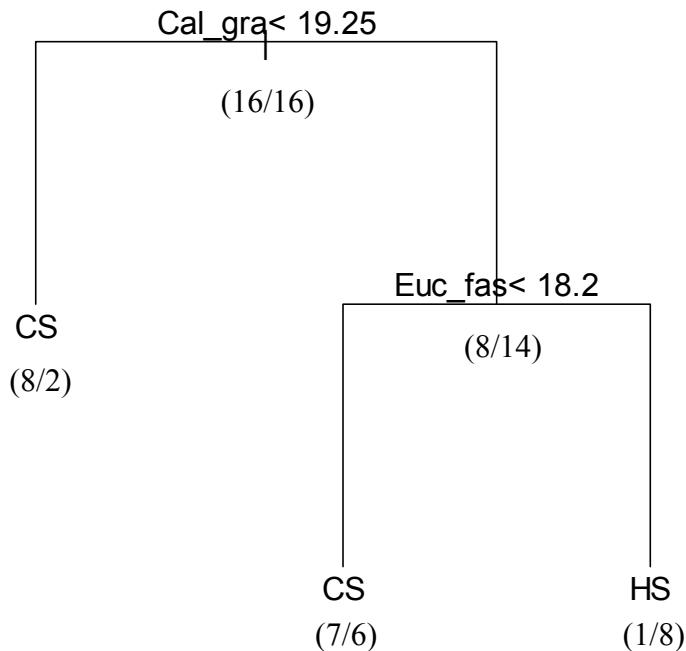


Figure 8.12: Yellow Thornbill (*Acanthiza nana*) classification tree for two site types on two habitat variables (using the rpart function in R (Hornkil 2012) with minsplit = 20 and complexity parameter = 0.01). The site types are 30 m × 30 m survey plots measured for hot-spots (HS, $n = 16$) and 30 m × 30 m survey plots measured for cold-spots (CS, $n = 16$). The habitat variables are the percent canopy cover of Slender Cypress-pine (*Callitris gracilis*; Cal_gra) and the percent canopy cover of Pink Gum (*Eucalyptus fasciculosa*; Euc_fas). To read this classification tree, move to the left branch when the stated condition is true. The variable appearing at the top of the tree (the root) represents the habitat attribute that was optimal in separating hot-spot plots from cold-spot plots. Variables appearing near the top of the tree explain more of the variation between hot-spot plots and cold-spot plots than those invoked lower on the tree. The terminal nodes of each branch (the leaves) indicate the interaction pathway that define either a hot-spot plot or a cold-spot plot. The numbers in brackets specify the number of hot-spot plots and cold-spot plots at each node (read as #CS / #HS). Surveys were conducted at Altona Chatterton, Altona Kies, Para Wirra (two home ranges) and Sandy Creek (two home ranges), within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected April – August 2009.

Table 8.3: Percent of time spent foraging in different plant species by individuals of Yellow Thornbills (*Acanthiza nana*) at locations within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Values represent percent time (*n*) spent foraging. n/a = tree species not present within the survey area. *n* = total time the birds were recorded as foraging in hours and minutes. Data collected April – August 2009 and repeated for Sandy Creek sites in June 2010.

| Sites | <i>Callitris gracilis</i> | <i>Allocasuarina verticillata</i> | <i>Eucalyptus odorata</i> | <i>Eucalyptus fasciculosa</i> | <i>Eucalyptus leucoxylon</i> | <i>Acacia pycnantha</i> | <i>n</i> |
|--------------------|---------------------------|-----------------------------------|---------------------------|-------------------------------|------------------------------|-------------------------|--------------|
| Altona Kies | 93 | 0 | n/a | 4 | 1 | 0.3 | 11:25 |
| Altona Chatterton | 87 | 0 | n/a | 2 | 2 | 6 | 8:00 |
| Sandy Creek A 2009 | 61 | 9 | 25 | 4 | 1 | 0 | 6:16 |
| Sandy Creek A 2010 | 63 | 3 | 33 | 1 | 1 | 0 | 8:02 |
| Sandy Creek B 2009 | 79 | 7 | 0 | 12 | 0.3 | 0.5 | 6:34 |
| Sandy Creek B 2010 | 84 | 4 | 2 | 6 | 0 | 1 | 7:01 |
| Para Wirra A | 67 | 15 | n/a | 4 | 1 | 2 | 5:51 |
| Parra Wirra B | 49 | 28 | n/a | 3 | 1 | 13 | 8:16 |
| Overall use | 73 | 8.3 | 7.5 | 4.5 | 0.9 | 2.9 | 61:25 |

Discussion

This study quantitatively identified that habitat use by individuals of different bird species occurred disproportionately within their minimum home ranges. This study also identified a number of habitat attributes at the fine-scale (i.e. 30 m × 30 m) that correlated with the habitat areas that were either preferentially used or avoided by the birds that were surveyed. This type of information regarding the fine-scale habitat use by birds can be used to enhance the knowledge of species-specific habitat relationships. Such knowledge can then be used to help inform how habitats may be reconstructed through revegetation to be of more benefit to birds. Integrating this level of knowledge into revegetation practice should then provide more useful habitats for target species that are currently of conservation concern.

For example, this study agrees that the Brown Treecreeper is renowned for their use of habitats that contain rough-barked tree species with an open grassy understorey (e.g. Chan 1995; Traill *et al.* 1996; Possingham and Possingham 1997). Rough-barked tree species that are predominantly used as a foraging substrate by these birds are dominated by Stringybarks and Boxes, but also include a variety of other rough-barked species. Plants that these birds use in other parts of Australia include Broad-leaved Stringybark (*E. caliginosa*), Silver Top Stringybark (*E. laevopinea*), Apple Box (*E. bridgesiana*), Fuzzy

Box (*E. conica*) and Yellow Box (*E. melliodora*), together with Narrow-leaved Ironbark (*E. crebra*) and Forest Oak (*A. torulosa*; Noske 1979; 1982b; Ford *et al.* 1986).

This current study also supports previous studies on the Brown Treecreeper both locally (Rogers 1998; Barker 2007) and interstate (Noske 1991), that report this species spends as much time foraging on the ground as they do using trees. The importance of an accessible ground layer suitable for foraging is best highlighted by the occurrence of this bird species within moderately grazed habitats (James 2003; Martin and Possingham 2005). Such habitat settings have also correlated with improved reproductive success for the Brown Treecreeper (Doerr *et al.* 2006). In comparison, altering land use that removes livestock grazing without alternative weed management has resulted in some local extinctions of the Brown Treecreeper (Ford *et al.* 2009).

Ultimately though, hollow density may be a critical element that directly influences habitat suitability for the Brown Treecreeper (Cooper and Walters 2002; Barker 2007). The birds in this study used hollows for nightly roosting, as a refuge when alarmed and for breeding, which agrees with what is already known of this bird species in the literature (e.g. Batey 1907; Noske 1977; Noske 1982a). Information in this current study may also allude to what could amount to a critical threshold in the number of tree hollows that are available at the patch scale of 30 m × 30 m. Hot-spot plots contained at least three hollows, which reflected the three to five birds that occupied the surveyed home ranges. Seemingly, patches containing a much lower number of hollows than the number of birds within a home range may not be highly regarded by these birds. This may infer that numerous suitable hollows or crevices are likely to be required per bird throughout a cohesive group's home range. Luck (2002) reported this finding for the Rufous Treecreeper (*C. rufa*) in Western Australia, with the home territories of this bird species positively related to the densities of tree hollows.

In anecdotal support for the importance of hollows in defining suitable habitat areas for the Brown Treecreeper, this study suggests that these birds' minimum home range may vary with hollow density. The largest minimum home range estimate was for five birds at Altona Kies (20.3 ha; Chapter 6) which contained the lowest recorded density of hollows (1.6 hollows ha⁻¹). Brown Treecreeper home ranges greater than 10 ha are considered by some researchers to infer lower quality habitat for this species (e.g. Cooper and Walters

2002; Doerr and Doerr 2006). The need for a sufficient number of appropriate hollows required by these birds is compounded by the fact that an otherwise highly cohesive group of Brown Treecreepers can disband at dusk and roost solitarily. Noske (1977) reported that although a pair of Brown Treecreepers foraged closely throughout the day, their individual roost trees were about 300 m apart. A similar situation was also observed during this study at Altona Kies, where hollows and crevices used by these birds were only available in isolation and positioned some distance apart. Of interest, Noske (1977) also reported that if a Brown Treecreeper was flushed from its roost, it never returned to that hollow that night. This implies that the flushed bird roosted in an alternative hollow located elsewhere within its home range.

Although no specific habitat attributes could be identified to help explain habitat use by the Buff-rumped Thornbills that were studied, the foraging height data showed that these birds used vegetation at all strata levels. This is consistent with the results in Chapter 5, which showed that habitat use by this species was correlated with habitat areas containing various storey levels with open vegetation. In support, it has long been recognised that this species prefers woodlands or open forests that contain a sparse to open understorey of grasses and/or heath (e.g. Conole 1981; Pyke 1985; Leach 1988; Traill *et al.* 1996; Possingham and Possingham 1997). Accordingly, planted habitats to suit the Buff-rumped Thornbill would provide an open understorey among a patchy distribution of midstorey and overstorey vegetation. This level of spatial heterogeneity would then help to provide habitats that tailor to the year-round requirements for this bird species, as discussed in Chapter 7.

The Hooded Robins in this study chiefly associated with woodland habitat patches that contained high amounts of litter cover. Importantly, habitat areas containing both high litter cover with practically no vegetation at lower storey levels were used the most by these birds. My interpretation for this habitat association is that the birds' view of the ground from their perched vantage points was not obstructed by dense vegetation growing at lower storey levels. This finding complements previous studies on the Hooded Robin that associate this species with specific ground conditions and habitat settings. For example, Recher *et al.* (2002) found that ground substrates were dominated by a mosaic of bare ground, litter and low ground vegetation within a three metre radius of perch locations where this species was foraging. Within the revegetated Monarto Woodland in South

Australia, 6 m × 6 m plots within hot-spots for this bird species correlated with reliable quantities of high litter cover (Northeast 2007). This agrees with Antos *et al.* (2008), who argued that litter is the foraging substrate of choice for many bird species that use the ground to forage. Hooded Robins have also been positively associated with habitat areas that contain very open vegetation, which includes vegetation edges (e.g. Priday 2010). To create quality habitats for the Hooded Robin, revegetation practitioners should provide patches of more open vegetation and/or sharp boundaries between planting densities. Although as highlighted in Chapter 7, the provision of some denser patches of vegetation that provide sheltered areas during poor weather also needs to be considered.

The foraging height disparity between the sexes of the Rufous Whistler has previously been reported (Recher and Davis 1997b; Recher and Holmes 2000; Hunt 2011; but see Mac Nally *et al.* 2000b). As in this current study, females typically foraged at lower levels (including the ground) than their male counterparts. This phenomenon has also been shown for other birds, such as *Setophaga* warblers in North America (Morse 1980). The study of Morse (1980) echoes many similar studies on other species in Australia and elsewhere, that suggest males forage higher so to be closer to singing perches while the females forage lower to be closer to their nest sites (e.g. Holmes 1986; Moysey 1997). This does not explain the height differences between the sexes in this current study though, as the two Rufous Whistler nests that were identified were both located approximately ten metres high within Slender Cypress-pine (Appendix B).

In the Altona and Sandy Creek study sites, hot-spots for pairs of Rufous Whistlers were associated with vegetation at the midstorey level. Intriguingly, the midstorey level (e.g. between 1 – 4 m) is the transition zone between the lower foraging heights preferred by the females and the higher foraging heights preferred by the males. To identify important habitat attributes that may differ between male and female birds, more intense surveys of individuals of both sexes are required. Based on the present information, hot-spots for females should correlate with vegetation nearer to the ground level while hot-spots for males should correlate with much taller vegetation. The differences between the sexes in their use of vegetation at different heights (and the lack of exclusive separation) probably explains why the Rufous Whistler is capable of inhabiting a wide variety of treed areas that contain a patchy shrubby understorey (Blakers *et al.* 1984; Boles 1988; Lindenmayer *et al.* 2003).

The White-browed Babblers in this study inhabited areas that contained patches of Kangaroo Thorn and both low and high densities of Slender Cypress-pine. These birds were observed to forage on the ground principally within less wooded areas, but used denser stands of suitably-structured vegetation for refuge and for the construction of roost-nests. Floristic structure appears critical for these birds, with the availability of plants that possess a dense outer foliage but with an internal open branch network, as being more important than simply a measure of the total vegetation that is available (Lynch *et al.* 1995). Plant species within the Mount Lofty region that possess this type of structure include the Kangaroo Thorn and Slender Cypress-pine. These plant species must therefore be considered when planning revegetation for the Para Woodland Reserve (and elsewhere) that tailors to the habitat needs of the White-browed Babbler.

A network of varying tree densities required by the White-browed Babbler was also implied by Antos *et al.* (2008). These researchers reported many ground-foraging birds respond positively to patches containing lower density vegetation when compared to random sites, with the notable exception being the White-browed Babbler. The White-browed Babbler required both open areas and patches of dense vegetation. In agreement, the Sandy Creek Conservation Park that hosted low populations of these birds was deficient in patches of suitably dense vegetation. Diversity in vegetation densities preferred by this species was also implied in Chapter 3, with the White-browed Babbler strongly associated with both open grassy woodland and open heath woodland habitats at the ten hectare scale. Clearly though, both the plant species and their planting arrangements must be considered to produce suitable habitat for this bird species.

Understorey plants that obscured the ground appeared to be a limiting factor for habitat use by the White-winged Choughs in this study. In the absence of these shrub types, these birds preferentially used habitat areas that contained high litter cover. Litter has been shown to be a favoured foraging substrate for this bird species in other parts of Australia (Ford *et al.* 1986; Antos *et al.* 2008). Hence, ground obscuring vegetation either physically hinders the movements of these birds and/or imposes a visual barrier that may impede cohesion between these highly sociable birds whilst foraging. In support, the literature reports that this bird species typically inhabits open eucalypt woodlands containing a mixture of open areas with low grasses and herbs (Fisher 2001) and litter-strewn ground

(Ford *et al.* 1986), or dry eucalypt forests with a negligible to open shrub layer (Recher 1975).

White-winged Choughs also venture into farmland that adjoins woodland (Rowley 1978; Loyn 1980; this study). This is despite this species often incurring the wrath of Australian Magpies within these more exposed environments. Magpie attacks towards White-winged Choughs are numerous and aggressive, ranging from mere swoops to physical assault (Rowley 1978; this study). Despite this challenge, the birds in this study defiantly returned to forage within these more open areas, presumably owing to their preference for the bulbs of bulbaceous weeds and fallen stones / fruit of the introduced Olive (*Olea europaea*; refer Appendix C). These introduced weedy plant species are presumably more common within agricultural settings than adjacent protected woodlands (e.g. Spennemann *et al.* 2000). Hence, areas where the White-winged Chough is uncommon (e.g. the Altona woodland) may involve aspects of both the woodland itself (e.g. the degree of, and accessibility to, high litter cover) and the nature of any surrounding agricultural matrix. It could also be speculated that in the absence of Australian Magpies, White-winged Choughs might show more extensive use of these altered agricultural areas.

Habitat areas preferentially used by the Willie Wagtails in this study complement the dual foraging strategy of this species of active searching combined with static searches. This foraging strategy by the Willie Wagtail has been well documented (e.g. Jackson and Elgar 1993; Er 1997; Recher and Davis 1997a; Rose 1999). Accordingly, an overstorey cover that ranges between 20 to 50% provides both perching substrates and sufficient open areas in which this species may forage. Importantly, adequate perching substrates also enabled the birds in this study to occupy habitat areas where access to the ground had been compromised due to the presence of introduced grasses (also see Chapter 7).

The Yellow-rumped Thornbill is commonly associated with more open habitats, such as grazing land that is adjacent to woodland habitat (Bell and Ford 1986; Pizzi and Knight 2003). However within these environments this species is rarely ever found far from some form of protective vegetation cover (e.g. Leach and Hines 1987). Consistent with this, Yellow-rumped Thornbills when foraging on the ground at Altona took every opportunity to retreat into nearby vegetation when alarmed. The plant species these birds used were often Slender Cypress-pine and Kangaroo Thorn. Both of these plant species offer low

growing dense foliage that provides immediate cover for foraging birds at the ground level, while secondarily offering potential foraging substrates (Davill 2001; this study). This strong habitat association between the Yellow-rumped Thornbill and Slender Cypress-pine was clearly demonstrated for the birds in the Altona area in this study.

The Yellow Thornbill minimum home ranges in this study were located within a variety of habitats that consisted of different plant communities (i.e. different mixes of a variety of eucalypts and other trees). Within these home ranges however, these birds predominantly foraged on the same plant species; namely Slender Cypress-pine. This is supported by the birds in this study being statistically associated with habitat patches that contained the highest aggregations of Slender Cypress-pine, when compared to the surrounding habitat, at the ten hectare scale (Chapter 5). These results suggest that Slender Cypress-pine is a crucial plant species for the Yellow Thornbill within the region surrounding the Para Woodland Reserve. In other parts of Australia, Yellow Thornbills also do not use plant species equally, instead preferring to forage in plant species with suitable (often dense) foliage. For example, the Yellow Thornbill often uses *Acacia*, *Allocasuarina*, *Callitris* and *Melaleuca* spp, which are either co-dominant with or dominated by *Eucalyptus* spp (Roberts 1979; Morris *et al.* 1981; Templeton 1992; Cooper and McAllan 1995; Ashton 1996). The *Eucalyptus* spp that the Yellow Thornbill does preferentially forage upon include Narrow-leaved Ironbark (Recher 1989), Green-leaf Mallee (*E. phenax*) and Black Mallee-box (*E. porosa*; Allan 2004). These eucalypt species (together with the aforementioned non-eucalypt species) exhibit tightly clustered foliage due to leaf shape and growing arrangements. This produces a high density of foliage in which this bird species forages on the inner portion (e.g. Allan 2004).

The species-specific habitat needs of birds that have been identified in this study can be used to enhance the effectiveness of revegetation for these bird species. This is an improvement on simply adopting the broad habitat features of remnant vegetation as a template to help guide revegetation activities (Bennett *et al.* 2000). With the need to create more heterogeneous revegetated habitats, the survey design using 30 m × 30 m plots in this study provided a scale to this habitat heterogeneity. At this scale, a wide variety of habitat features were interpreted differently by individuals of the nine bird species that were observed. Significantly, the habitat attributes that were associated with both high use and low use habitat areas for the birds in this study made ecological sense with regards to

the respective species. This helped to explain the habitat associations by these birds. Associations such as the Brown Treecreeper with hollows and the White-browed Babbler with appropriate vegetation structure, also helps to explain why these bird species hold permanent year-round territories (as noted in Chapter 1), and clarifies the strong within-patch fidelity the birds in this study showed in their inter-annual use of habitats (Chapter 7).

Some habitat features were simultaneously interpreted as providing good quality habitat by birds of some species, yet being of poorer quality habitat by other co-occurring birds of other species. For example, patches containing vegetation cover at lower storey levels were preferred by individuals of Rufous Whistlers and White-browed Babblers, yet these patches were avoided by individuals of Hooded Robins and White-winged Choughs. Since different species have different habitat needs, one should at least try to incorporate a range of habitat features at finer-scales. In practice, instead of planting the same floristically and structurally diverse habitat in each 30 m × 30 m area across a revegetation site, the actual floristic and structural diversity of each of the 30 m × 30 m plots needs to change across that area. Also, with the high variability shown in the locations of hot-spots due to seasonality and perturbations such as fire and adverse weather conditions (Chapter 7), significant habitat heterogeneity is important in supporting the lifetime habitat needs for a range of bird species (Benton *et al.* 2003).

Knowledge of species micro-habitat needs can also be used to enhance the habitat value within anthropogenically altered landscapes. For example, incorporating small bare gaps within arable farmland within the United Kingdom has dramatically increased the numbers of Skylark (*Alauda arvensis*), a species on the International Union for Conservation of Nature (IUCN) “Red List” (Royal Society for the Protection of Birds (RSPB) – <http://www.rspb.org.uk/>). Integrating the specific habitat needs of target species should be used to produce better quality revegetated habitats, together with improving degraded remnant habitats that are commonly found within the highly fragmented agricultural landscapes across southern Australia (Yates and Hobbs 1997; Paton *et al.* 2000; Ford *et al.* 2001).

Chapter 9

Habitat use by co-inhabiting bird species

Introduction

The mechanisms behind ecological separation between co-inhabiting bird species has long intrigued and challenged avian ecologists. Within landscapes, multiple species simultaneously perceive regions as containing different proportions of suitable habitats (Betts *et al.* 2006). Individual bird species should then respond differently to a range of habitat structures and floristic variables (Hewson *et al.* 2011). Early work by Lack (1971) identified three principal means by which co-inhabiting species reduce inter-species competition; these being differences in range, habitat and foraging.

For species that co-inhabit, the three factors proposed by Lack (1971) are highly interrelated. For example, range differences refer to sympatric separation, where species that share the same geographic location generally do not interact. This is often due to habitat partitioning via behavioural differences, with these differences being driven by habitat variability. In support, micro-habitat features have long been shown to be the principal determinant in explaining the separation of species that overlap in habitat used at the broad-scale, but with little or no overlap at the fine-scale (Cody and Walter 1976; Loyn 2002). In recognition of the influence fine-scale habitat heterogeneity has on the distribution of fauna species, species diversity at the landscape level (gamma diversity) has been described as consisting of two elements; alpha diversity (the mean species diversity at fine-scales) and beta diversity (habitat differences at fine-scales; Whittaker 1972).

Habitat selection by birds at fine-scales is ultimately influenced by the different habitat needs of individual species. This is clearly evident by the array of morphological, physiological and behavioural adaptations displayed by birds in relation to the way they source and handle food (Klopfer 1985; Walsberg 1985; Winkler 1985). Notably, foraging strategy is the foremost determinant that separates habitat use by congeneric bird species

(species that belong to the same taxonomic genus; Lack 1971). It follows that spatial variation in habitat use by birds should exist between species that occupy the same habitat area, especially if habitat selection is based on a mosaic of micro-habitat patches (Wiens 1989a).

In this chapter I ask the question, are discrete patches of habitat which are preferentially used by birds of one species, different from those which are preferentially used by birds of other species that occupy the same habitat area? This study builds on the findings of the previous three chapters. Within patches of remnant vegetation within the Para Woodland area, there was usually a suite of bird species present that coexisted. Many of the birds within these patches were shown to have minimum home ranges where individuals disproportionately used different parts of their home ranges (Chapters 6 and 7) and that these areas of high use correlated with specific habitat features (Chapters 7 and 8).

The design of this study was inspired by Ewer (1968, p 65), who argued that “it is not uncommon to find that while the home ranges (of species) overlap, the core areas (for individual species) do not”. A sense of scale then beckons as to the appropriate area to use in which to test this hypothesis. MacArthur (1972) concluded that a patch of habitat just large enough to hold an adequate number of species would be optimal. Differences in micro-habitat choice between species are also best observed against a continuous environmental gradient, such as a single patch of habitat (Wiens 1989a). This way, aggregations of high area use by individual species should be in direct response to the array of habitat features contained within that area (Wiens 1989a; Brown *et al.* 1995).

Armed with the *a priori* knowledge of the fine-scale habitat needs of individuals of various species (e.g. Chapter 8), knowledge of the habitat attributes that influence habitat use by birds and their spatial arrangements can be used to directly inform on important habitat features and their suggested configurations. Such information is highly relevant for, and directly transferrable to, revegetation practice. For instance, the framework for revegetation can determine the eventual makeup of its avian inhabitants (Allison 2007). So to recruit desired species, we must first tailor revegetation that caters for the specific habitat needs of individual species. This would also help in addressing the reported gap found with many revegetation attempts, between desired outcomes and realised ecological

gain (Kimber *et al.* 1999; Vesk and Mac Nally 2006; Kavanagh *et al.* 2007; Loyn *et al.* 2007).

Methods

Study sites and bird species

Multiple co-inhabiting birds belonging to numerous declining species were surveyed in four principal locations within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. These sites were located within the Altona woodland (identified here as Altona Chatterton and Altona Kies), the Para Wirra Recreation Park and the Sandy Creek Conservation Park (Table 9.1; see Chapters 1 and 6 for location and site name details). Within these locations, multiple co-inhabiting birds belonging to different species were intensively surveyed for sustained periods over multiple days, to identify high use habitat areas (hot-spots) that were used by individuals of each species. The species were the Brown Treecreeper (*Climacteris picumnus*), Buff-rumped Thornbill (*Acanthiza reguloides*), Hooded Robin (*Melanodryas cucullata*), Rufous Whistler (*Pachycephala rufiventris*), White-browed Babbler (*Pomatostomus superciliosus*), White-winged Chough (*Corcorax melanorhamphos*), Willie Wagtail (*Rhipidura leucophrys*), Yellow-rumped Thornbill (*A. chrysorrhoa*) and Yellow Thornbill (*A. nana*; Table 9.1).

The Altona Chatterton site (21.6 ha) was generally open grassy woodland that was dominated by Pink Gum (*Eucalyptus fasciculosa*) and Slender Cypress-pine (*Callitris gracilis*). The southern region of this site contained occasional thickets of Kangaroo Thorn (*Acacia paradoxa*). The northern end graded to more open grassy woodland with Peppermint Box (*E. odorata*).

The Altona Kies site (16.4 ha) was open grassy woodland that was dominated by Pink Gum and Slender Cypress-pine, with occasional thickets of Kangaroo Thorn throughout.

The Para Wirra site (9.4 ha) was on a slope with more varied woodlands. Up slope to the north was a forested stand that was dominated by Slender Cypress-pine and Drooping Sheoak (*Allocasuarina verticillata*), with low density Pink Gum and closed heath

understorey. Down slope, immediately to the south, the vegetation changed to Pink Gum and Long-leaved Box (*E. goniocalyx*) woodland, with a spatially aggregated midstorey and open heath understorey. Further south again a drainage depression fanned out, which was dominated by South Australian Blue Gum (*E. leucoxylon*) overstorey and Golden Wattle (*A. pycnantha*) midstorey. To the west, open grassy woodland extended into areas that once sustained agriculture.

The Sandy Creek site (10.7 ha) contained both open grassy woodland and open heath woodland. The overstorey was dominated by Pink Gum and Slender Cypress-pine throughout. Areas of open heath habitat were dominated by Scarlet Bottlebrush (*Callistemon rugulosus*), Common Fringe-myrtle (*Calytrix tetragona*) and Flame Heath (*Astroloba conostephioides*) at both the midstorey and shrub levels.

Table 9.1: Survey locations and the co-inhabiting bird species that were observed within the north-central zone of the southern Mount Lofty Ranges near the Para Woodland Reserve, South Australia. Data collected April – September 2009.

| Altona Chatterton | Altona Kies | Para Wirra | Sandy Creek |
|-------------------------|----------------------|-------------------------|----------------------|
| Hooded Robin | Brown Treecreeper | Buff-rumped Thornbill | Hooded Robin |
| White-browed Babbler | Hooded Robin | White-winged Chough | Rufous Whistler |
| Willie Wagtail | Rufous Whistler | Yellow Thornbill | White-browed Babbler |
| Yellow-rumped Thornbill | White-browed Babbler | | White-winged Chough |
| | Willie Wagtail | | Willie Wagtail |
| | | Yellow-rumped Thornbill | |
| | | Yellow Thornbill | |

Bird surveys and data analyses

The birds were observed and their positions and behaviour recorded almost continuously between dawn and dusk, for one or more days (refer Chapters 6, 7 and 8). These data were used to define minimum home range boundaries via Minimum Convex Polygon (MCP) analysis using ArcMap in ArcGIS 9.2 (ESRI 1999 - 2006; Chapter 6). Areas of high

habitat use (i.e. hot-spots) used by the birds within individual minimum home ranges were identified via Point Density analysis using ArcMap (Chapter 7). The habitat features that were statistically associated with these hot-spots within the home ranges of the birds that were surveyed for each of the nine species were identified via Classification Tree analysis using the rpart functions in R (Hornkil 2012; Chapter 8).

Each site was fully surveyed by walking the entire area and searching for the inhabiting target bird species. Some groups of the same species were present but not surveyed within some survey areas (namely, the Hooded Robin and White-browed Babbler at Altona Chatterton; the Rufous Whistler and Willie Wagtail at Altona Kies; and the Rufous Whistler at Sandy Creek). However, these unsurveyed birds were generally restricted to the peripheral areas within the survey location and usually were never within the core areas being analysed. Their presence therefore was not considered to have adversely influenced the interpretation of the ensuing results.

Natural Breaks (Jenks) using two classes was used to classify two natural groupings that was inherent in the data for the birds of each of the species that were surveyed within each of the four locations (see Chapter 7). Again, Natural Breaks classes identified where maximum differences existed within a dataset, and defined where the major difference was by segregating these data into two distinct groups. Each group of data contained strong within-group homogeneity that contrasted with the data values in the second group. This analysis was used to define hot-spot areas from cold-spot areas. The Class Break depicting the boundary between the hot-spot data and the cold-spot data was defined by the vertical purple line in the resulting graphed output (refer Chapter 7, Fig. 7.1).

In this chapter, the Class Break values for the birds belonging to each species that were surveyed for each location were used to reclassify the bird data. This was done to remove the groups of data that contained the lowest values (i.e. the cold-spot data), to leave only the groups of data that contained the highest values (i.e. the hot-spot data) for every individual bird species within each of the four surveyed sites. The segregated hot-spot data for each species were then layered on imagery of the four respective study areas, using ArcMap. These images were then used to measure and calculate the percent of spatial overlap between both the home ranges (i.e. MCP's) and hot-spot patches for each of the

bird species, relative to that of all the other co-inhabiting bird species that were surveyed within each site.

To investigate what influence fine-scale habitat heterogeneity had on the birds of each species that occupied a common habitat area, the significant fine-scale habitat attributes that were deemed to be important for each of the co-inhabiting bird species (from Chapter 8) were grouped for each of the four surveyed locations. Using PC-ORD5 (McCune and Mefford 1999b), these data were then subjected to Nonmetric Multidimensional Scaling (NMS) ordination, employing the recommended Sørensen (Bray-Curtis) distance measure for community type data (McCune and Mefford 1999a). Nonmetric Multidimensional Scaling was chosen as it makes few assumptions about the data (e.g. data do not need to be normally distributed), uses rank distances so the relationships between the distance measures of bird species and habitat variables are linearised, and is not confounded by unequal gradient strengths between the variables that are used as input (McCune and Grace 2002). This helps to graphically produce biologically meaningful representations of community relationships without the impost of statistical assumptions (Clarke 1993).

The “slow and thorough” autopilot mode in NMS was chosen to select the most appropriate dimensionality, with supporting statistics. Where such analysis recommends two-dimensional solutions (i.e. two axes), there are often strong gradients along at least one of the axes in the NMS ordination (McCune and Grace 2002). Points within the NMS ordination represent the bird species, with the distances between the points approximating the relative similarities between each species and the habitat attributes that were associated with those bird’s respective hot-spots. Species distributions within the NMS space were overlaid with the habitat vectors for each ordination. Each vector’s orientation and length indicated the direction and strength of the relationship between each habitat variable and the point distributions (i.e. the birds). These helped delineate habitat use by individual bird species and their relationships with other co-occurring bird species.

The significance and interpretation of the reported NMS stress levels has previously been explained in Chapter 3. An NMS stress of between 5 and 10% allows for high confidence in the interpretation without the risk of drawing false inferences (Clarke 1993). Linear relationships between the ordination scores and habitat variables are expressed by the correlation coefficient on each axis of the NMS ordination. The sum of these two

correlation coefficients denotes the percent of the variation that is represented by the distribution of the bird species within the ordination plot (McCune and Mefford 1999a).

Results

Altona Chatterton

Within the Altona Chatterton site (Fig. 9.1a), the bird species surveyed were the Hooded Robin during both fine weather (MCP 19.2 ha) and stormy weather (MCP 12.1 ha), White-browed Babbler (MCP 36.4 ha), Willie Wagtail (MCP 11.8 ha) and Yellow-rumped Thornbill (MCP 24.5 ha; Fig. 9.1b). While overlap in the MCP areas used by these four bird species averaged $40.2 \pm 6.3\%$, there was minimal overlap between the hot-spots for the individuals belonging to each species ($2.5 \pm 0.8\%$ overlap; Table 9.2).

With a stress value of 7.9%, the two-dimensional NMS ordination plot for the birds surveyed at Altona Chatterton revealed strong habitat associations, with the percent of the variation represented by the distribution of the birds being 79% (Fig. 9.2). The White-browed Babbler (46.0 ± 19.7 trees per hot-spot plot) and Hooded Robin during the storm (23.4 ± 5.7 trees per hot-spot plot) frequented patches containing higher overstorey density (Fig. 9.2). The Hooded Robin during the storm, White-browed Babbler and Yellow-rumped Thornbill associated more strongly with higher Slender Cypress-pine canopy cover (Fig. 9.2). The latter two bird species also associated more strongly with the presence of Kangaroo Thorn (Fig. 9.2). The White-browed Babbler also associated with a number of patches that contained more open habitat, but which was also associated with Kangaroo Thorn (Fig. 9.2). The Hooded Robin during fine weather (9.7 ± 3.4 trees per hot-spot plot) and Willie Wagtail (4.0 ± 1.1 trees per hot-spot plot) used patches with lower overstorey density (Fig. 9.2).

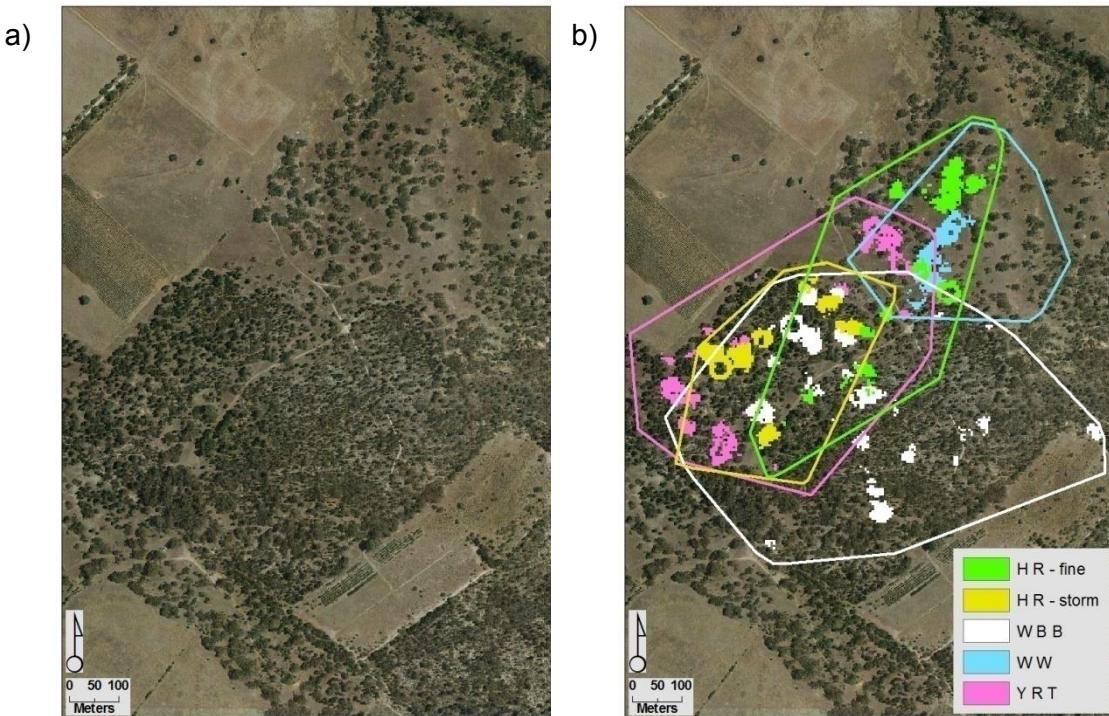


Figure 9.1: Survey site at Altona Chatterton (a), with (b) patches of high habitat use (i.e. hot-spots) for the Hooded Robin (*Melanodryas cucullata*; HR) during both fine and stormy weather, White-browed Babbler (*Pomatostomus superciliosus*; WBB), Willie Wagtail (*Rhipidura leucophrys*; WW) and Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*; YRT). The coloured lines show the Minimum Convex Polygons (MCP) that enclosed all points for each species. The MCP for the Hooded Robins in fine weather is shown as green (MCP 19.2 ha). The MCP for the Hooded Robin in stormy weather is shown in yellow (MCP 12.1 ha). The MCP for the White-browed Babbler is shown in white (MCP 36.4 ha). The MCP for the Willie Wagtail is shown in blue (MCP 11.8 ha). The MCP for the Yellow-rumped Thornbill is shown in pink (MCP 24.5 ha). Data collected May – July 2009.

Table 9.2: Percent habitat overlap between four co-inhabiting bird species at Altona Chatterton. The percent overlap in MCP areas (habitat area used by each species during the survey as derived from Minimum Convex Polygon (MCP) analysis) and HS (the amount of habitat overlap between the areas of high habitat use, or hot-spots, used by each species) are shown for each combination of species. The species were the Hooded Robin (*Melanodryas cucullata*) during both fine weather (HR – fine) and stormy weather (HR – storm), White-browed Babbler (*Pomatostomus superciliosus*; WBB), Willie Wagtail (*Rhipidura leucophrys*; WW) and Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*; YRT). Rows show the extent (as a percent) to which both a species home range and associated hot-spots overlap with that of other co-inhabiting species (refer Fig. 9.1b). Data collected May – July 2009.

| Species surveyed | HR – fine | | HR – storm | | WBB | | WW | | YRT | |
|-------------------------|------------------|-----|-------------------|-----|------------|-----|-----------|-----|------------|-----|
| | MCP | HS | MCP | HS | MCP | HS | MCP | HS | MCP | HS |
| Hooded Robin – fine | - | - | 32 | 1.0 | 50 | 1.8 | 31 | 9.3 | 60 | 0.0 |
| Hooded Robin – storm | 55 | 1.0 | - | - | 97 | 8.7 | 2 | 0.0 | 100 | 0.0 |
| White-browed Babbler | 29 | 1.6 | 33 | 6.4 | - | - | 4 | 0.0 | 43 | 2.1 |
| Willie Wagtail | 56 | 13 | 2 | 0.0 | 12 | 0.0 | - | - | 23 | 2.7 |
| Yellow-rumped Thornbill | 51 | 0.0 | 49 | 0.0 | 63 | 1.7 | 11 | 1.4 | - | - |

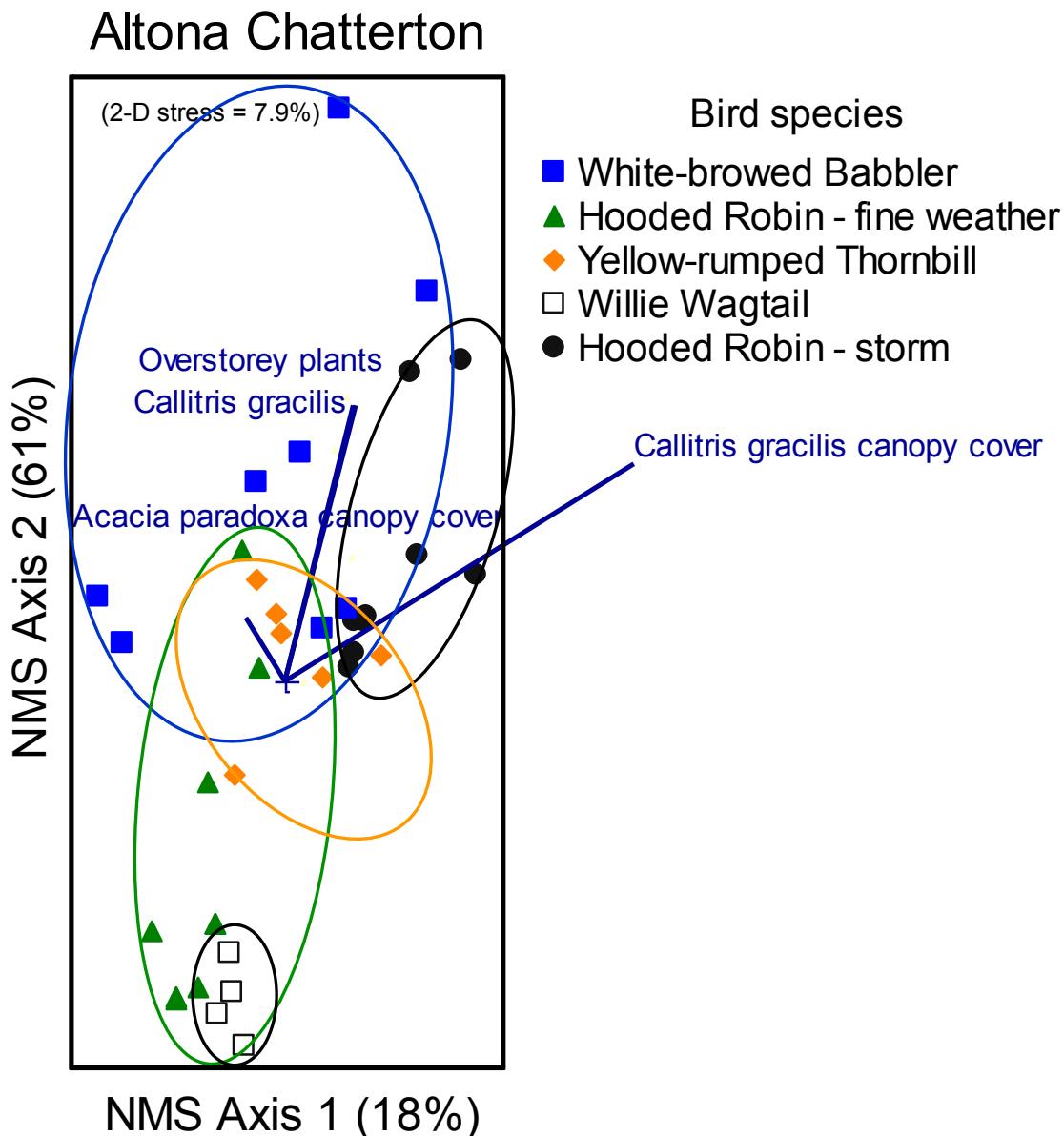


Figure 9.2: Two-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features in the respective hot-spots for four bird species surveyed at Altona Chatterton. Species are the Hooded Robin (*Melanodryas cucullata*) surveyed during both fine and stormy weather, White-browed Babbler (*Pomatostomus superciliosus*), Willie Wagtail (*Rhipidura leucophrys*) and Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*). Superimposed are the habitat variables that were associated with areas of high habitat use (i.e. 30 m × 30 m plots within hot-spots) that helped to delineate differences between the species at this site. These are the number of overstorey plants of all tree species combined, number of Slender Cypress-pine (*Callitris gracilis*), Slender Cypress-pine canopy cover and Kangaroo Thorn (*Acacia paradoxa*) canopy cover. The orientation of the vectors indicates the direction of positive increase and the length represents the strength of association. Each colour-coded symbol represents a 30 m × 30 m plot within a hot-spot for a particular bird species. The colour-coded ellipses represent ordination space envelopes for each bird species and are used here for visual aids to assist pattern interpretation, but are not statistically based. At a stress level of 7.9%, 79% of the variation within the ordination plot is represented by the distribution of the bird species. Data collected May – July 2009.

Altona Kies

Within the Altona Kies site (Fig. 9.3a), the bird species surveyed were the Brown Treecreeper (MCP 20.3 ha), Hooded Robin (MCP 17.1 ha), Rufous Whistler (MCP 2.3 ha), White-browed Babbler (MCP 14.4 ha), two pairs of Willie Wagtail (MCP 3.9 ha and 3.0 ha), Yellow-rumped Thornbill (MCP 14.3 ha) and Yellow Thornbill (MCP 4.0 ha; Fig. 9.3b). While overlap in the MCP areas used by individuals of these seven bird species averaged $48.8 \pm 4.5\%$, there was minimal overlap between the hot-spots for the individuals belonging to each species ($7.7 \pm 1.8\%$ overlap; Table 9.3).

With a stress value of 10.3%, the two-dimensional NMS ordination plot for the birds surveyed at Altona Kies revealed strong habitat associations, with the percent of the variation represented by the distribution of the birds being 93% (Fig. 9.4). The Rufous Whistler (54.8 ± 17 trees per hot-spot plot) and White-browed Babbler (22.3 ± 2.8 trees per hot-spot plot) frequented patches with higher overstorey density that was dominated by Slender Cypress-pine (Fig. 9.4). Yellow Thornbill hot-spots were closely associated with the percent of Slender Cypress-pine canopy cover ($36.9 \pm 12.6\%$ per hot-spot plot; Fig 9.4). The White-browed Babbler associated with Slender Cypress-pine and Kangaroo Thorn canopy covers (Fig. 9.4). The Yellow-rumped Thornbill had an association with the canopy cover of both the Slender Cypress-pine and Kangaroo Thorn, and also used patches of more open habitats (Fig. 9.4). The Brown Treecreeper held the strongest association with the presence of tree hollows (Fig. 9.4). A Brown Treecreeper hot-spot was also present at the base of the vectors (Fig. 9.4), which coincided with an open area that contained a quantity of fallen trees. Within this location, the birds foraged on and around these fallen timbers. The Hooded Robin (7.0 ± 1.1 trees per hot-spot plot) and Willie Wagtail (7.1 ± 1.2 trees per hot-spot plot) were typically found within patches containing lower overstorey density (Fig. 9.4).

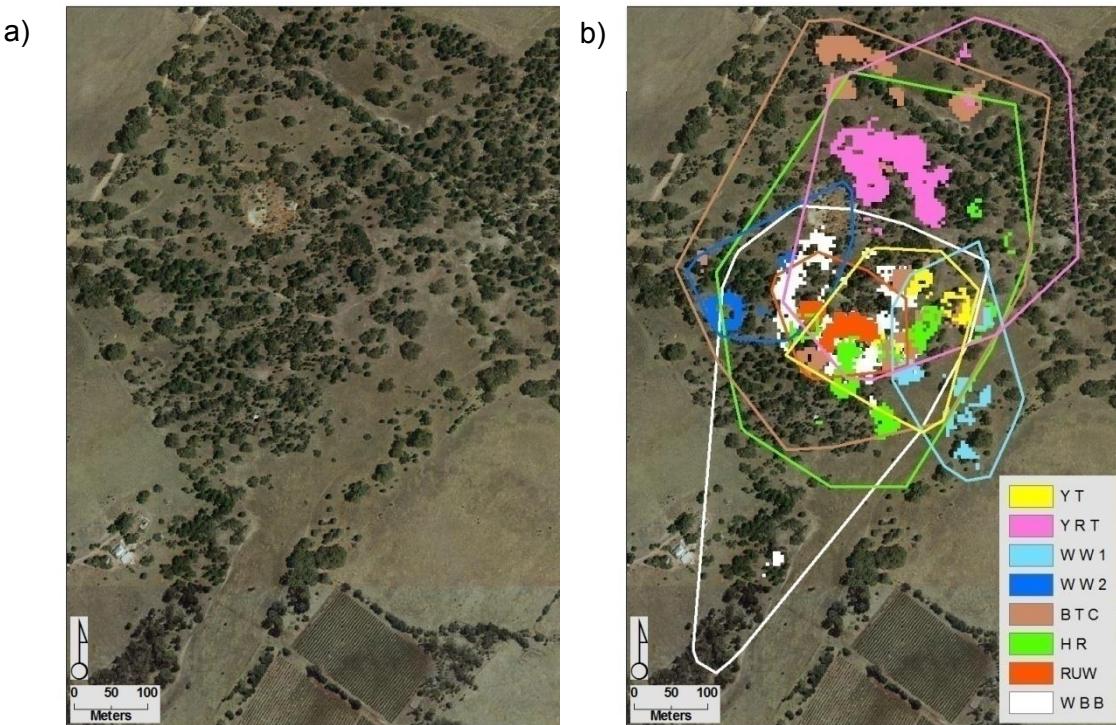


Figure 9.3: Survey site at Altona Kies (a), with (b) patches of high habitat use (i.e. hot-spots) for the Yellow Thornbill (*Acanthiza nana*; YT), Yellow-rumped Thornbill (*A. chrysorrhoa*; YRT), two pairs of Willie Wagtail (*Rhipidura leucophrys*; WW1 & WW2), Brown Treecreeper (*Climacteris picumnus*; BTC), Hooded Robin (*Melanodryas cucullata*; HR), Rufous Whistler (*Pachycephala rufiventris*; RUW) and White-browed Babbler (*Pomatostomus superciliosus*; WBB). The coloured lines show the Minimum Convex Polygons (MCP) that enclosed all points for each species. The MCP for the Yellow Thornbill is shown in yellow (MCP 4.0 ha). The MCP for the Yellow-rumped Thornbill is shown in pink (MCP 14.3 ha). The MCP for the Willie Wagtail (pair 1) is shown in light blue (MCP 3.9 ha). The MCP for the Willie Wagtail (pair 2) is shown in dark blue (MCP 3.0 ha). The MCP for the Brown Treecreeper is shown in brown (MCP 20.3 ha). The MCP for the Hooded Robin is shown in green (MCP 17.1 ha). The MCP for the Rufous Whistler is shown in red (MCP 2.3 ha). The MCP for the White-browed Babbler is shown in white (MCP 14.4 ha). Data collected April – July 2009.

Table 9.3: Percent habitat overlap between seven co-inhabiting bird species at Altona Kies. The percent overlap in MCP areas (habitat area used by each species during the survey as derived from Minimum Convex Polygon (MCP) analysis) and HS (the amount of habitat overlap between the areas of high habitat use, or hot-spots, used by each species) are shown for each combination of species. The species were the Brown Treecreeper (*Climacteris picumnus*; BTC), Hooded Robin (*Melanodryas cucullata*; HR), Rufous Whistler (*Pachycephala rufiventris*; RUW), White-browed Babbler (*Pomatostomus superciliosus*; WBB), two pairs of Willie Wagtail (*Rhipidura leucophrys*; WW1 & WW2), Yellow-rumped Thornbill (*Acanthiza chrysorrhoa*; YRT) and Yellow Thornbill (*A. nana*; YT). Rows show the extent (as a percent) to which both a species home range and associated hot-spots overlap with that of other co-inhabiting species (refer Fig. 9.3b). Data collected April – July 2009.

| Species surveyed | BTC | | HR | | RUW | | WBB | | WW1 | | WW2 | | YRT | | YT | |
|-------------------------|-----|------|-----|------|-----|------|-----|------|-----|------|-----|-----|-----|-----|-----|-----|
| | MCP | HS | MCP | HS | MCP | HS | MCP | HS |
| Brown Treecreeper | - | - | 78 | 15.6 | 12 | 3.0 | 44 | 26.7 | 12 | 9.1 | 15 | 0.0 | 58 | 2.2 | 20 | 7.4 |
| Hooded Robin | 91 | 18.5 | - | - | 14 | 9.5 | 58 | 40.0 | 14 | 12.6 | 15 | 0.0 | 61 | 0.0 | 24 | 1.3 |
| Rufous Whistler | 100 | 8.0 | 100 | 21.5 | - | - | 100 | 67.2 | 9 | 0.0 | 28 | 0.0 | 84 | 0.0 | 72 | 0.4 |
| White-browed Babbler | 44 | 17.6 | 58 | 20.1 | 100 | 16.5 | - | - | 50 | 2.5 | 75 | 1.5 | 32 | 0.0 | 96 | 2.1 |
| Willie Wagtail pair #1 | 65 | 27.1 | 64 | 31.9 | 5 | 0.0 | 47 | 11.2 | - | - | 0 | 0.0 | 43 | 0.0 | 48 | 0.0 |
| Willie Wagtail pair #2 | 100 | 0.0 | 84 | 0.0 | 21 | 0.0 | 76 | 9.7 | 0 | 0.0 | - | - | 38 | 0.0 | 0 | 0.0 |
| Yellow-rumped Thornbill | 82 | 2.3 | 72 | 0.0 | 12 | 0.0 | 31 | 0.0 | 12 | 0.0 | 8 | 0.0 | - | - | 20 | 0.0 |
| Yellow Thornbill | 99 | 39.8 | 100 | 5.9 | 38 | 0.9 | 92 | 0.2 | 45 | 0.0 | 0 | 0.0 | 69 | 0.0 | - | - |

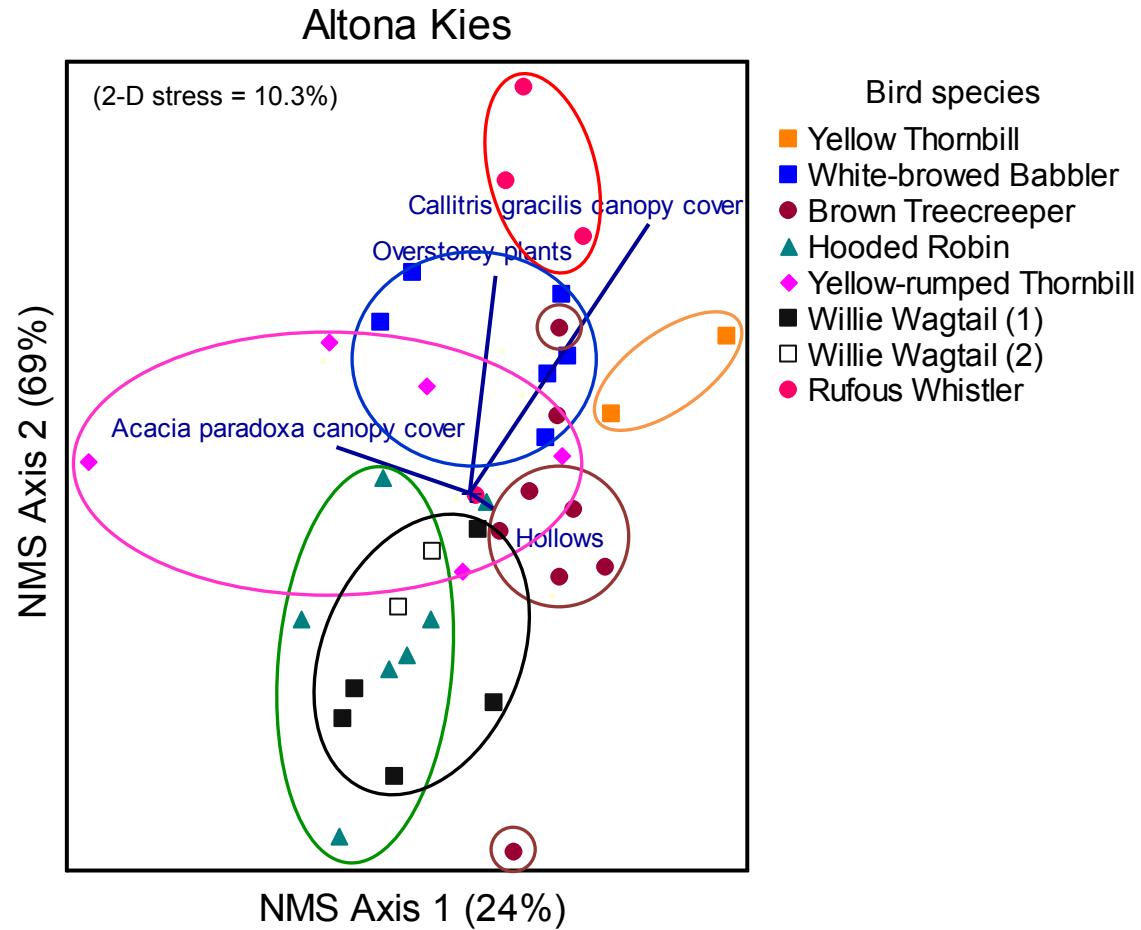


Figure 9.4: Two-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features in the respective hot-spots for seven bird species surveyed at Altona Kies. Species are the Yellow Thornbill (*Acanthiza nana*), White-browed Babbler (*Pomatostomus superciliosus*), Brown Treecreeper (*Climacteris picumnus*), Hooded Robin (*Melanodryas cucullata*), Yellow-rumped Thornbill (*A. chrysorrhoa*), Willie Wagtail (*Rhipidura leucophrys*; two pairs) and Rufous Whistler (*Pachycephala rufiventris*). Superimposed are the habitat variables that were associated with areas of high habitat use (i.e. 30 m × 30 m plots within hot-spots) that helped to delineate differences between the species at this site. These are the number of overstorey plants, Slender Cypress-pine (*Callitris gracilis*) canopy cover, Kangaroo Thorn (*Acacia paradoxa*) canopy cover and the number of tree hollows. The orientation of the vectors indicates the direction of positive increase and the length represents the strength of association. Each colour-coded symbol represents a 30 m × 30 m plot within a hot-spot for a particular bird species. The colour-coded ellipses represent ordination space envelopes for each bird species and are used here for visual aids to assist pattern interpretation, but are not statistically based. At a stress level of 10.3%, 93% of the variation within the ordination plot is represented by the distribution of the bird species. Data collected April – July 2009.

Para Wirra Recreation Park

Within the Para Wirra site (Fig. 9.5a), the bird species surveyed were the Buff-rumped Thornbill (MCP 9.7 ha), White-winged Chough (14.0 ha) and Yellow Thornbill (MCP 7.3 ha; Fig. 9.5b). While overlap in the MCP areas used by these three bird species averaged $68.3 \pm 9.5\%$, there was minimal overlap between the hot-spots for the individuals belonging to each species ($5.7 \pm 2.9\%$ overlap; Table 9.4).

With a stress value of 8.2%, the two-dimensional NMS ordination plot for the birds surveyed at Para Wirra revealed strong habitat associations, with the percent of the variation represented by the distribution of the birds being 92% (Fig. 9.6). The Yellow Thornbill largely associated with overstorey stands of Slender Cypress-pine and Drooping Sheoak (Fig. 9.6). That said, one record for Yellow Thornbill related more with midstorey plants, where birds frequented a dense patch of flowering Golden Wattle (Fig. 9.6). The Buff-rumped Thornbill was linked to patches containing both understorey and midstorey vegetation (Fig. 9.6). During the 2009 bird survey at this location, the Buff-rumped Thornbill foraged within the midstorey vegetation 44% of the time (5 h 51 min from $n = 232$ records) and although foraging was directed towards Golden Wattle 49% of that time (2 h 50 min from $n = 139$ records), this also included Sticky Cassinia (*Cassinia uncata*) 28% of that time (1 h 38 min from $n = 37$ records) and Christmas Bush (*Bursaria spinosa*) 18% of that time (1 h 03 min from $n = 44$ records). White-winged Chough hot-spots were associated with South Australian Blue Gum and Golden Wattle in patches that contained little or no understorey (Fig. 9.6).

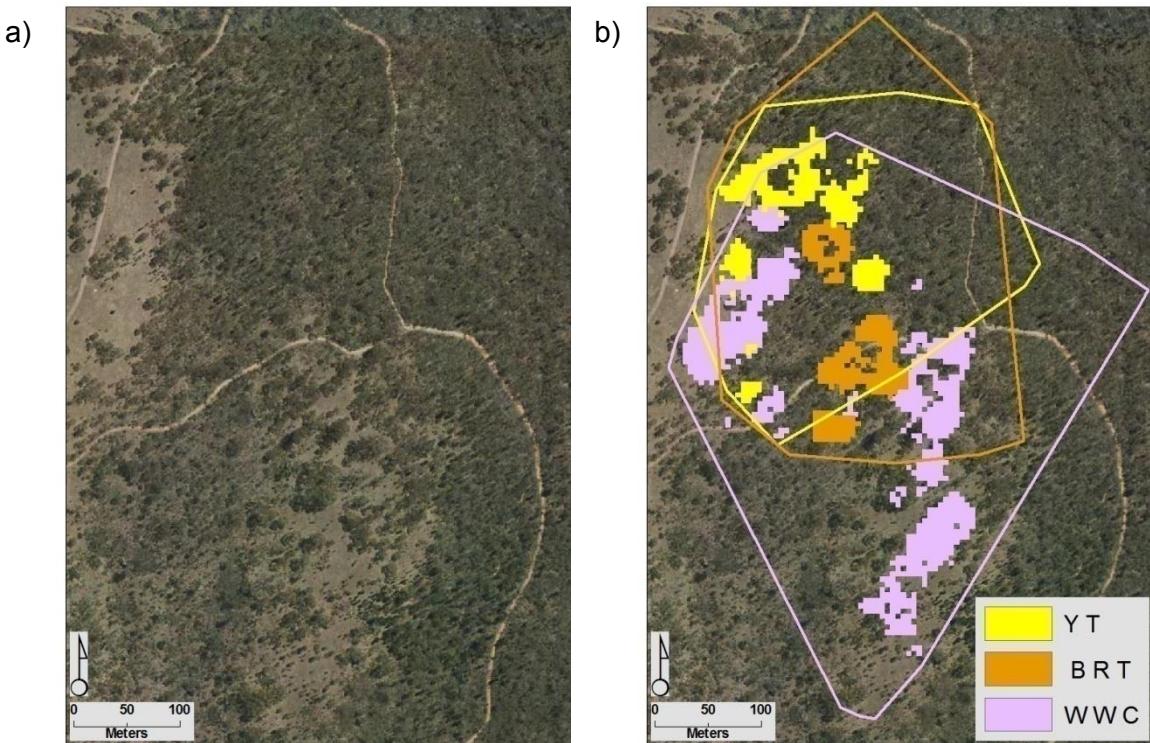


Figure 9.5: Survey site within the Para Wirra Recreation Park (a), with (b) patches of high habitat use (i.e. hot-spots) for the Yellow Thornbill (*Acanthiza nana*; YT), Buff-rumped Thornbill (*A. reguloides*; BRT) and White-winged Chough (*Corcorax melanorhamphos*; WWC). The coloured lines show the Minimum Convex Polygons (MCP) that enclosed all points for each species. The MCP for the Yellow Thornbill is shown in yellow (MCP 7.3 ha). The MCP for the Buff-rumped Thornbill is shown in orange (MCP 9.7 ha). The MCP for the White-winged Chough is shown in lilac (MCP 14.0 ha). Data collected July – August 2009.

Table 9.4: Percent habitat overlap between three co-inhabiting bird species within the Para Wirra Recreation Park. The percent overlap in MCP areas (habitat area used by each species during the survey as derived from Minimum Convex Polygon (MCP) analysis) and HS (the amount of habitat overlap between the areas of high habitat use, or hot-spots, used by each species) are shown for each combination of species. The species were the Buff-rumped Thornbill (*Acanthiza reguloides*; BRT), White-winged Chough (*Corcorax melanorhamphos*; WWC) and Yellow Thornbill (*A. nana*; YT). Rows show the extent (as a percent) to which both a species home range and associated hot-spots overlap with that of other co-inhabiting species (refer Fig. 9.5b). Data collected July – August 2009.

| Species surveyed | BRT | | WWC | | YT | |
|-----------------------|-----|-----|-----|------|-----|-----|
| | MCP | HS | MCP | HS | MCP | HS |
| Buff-rumped Thornbill | - | - | 58 | 19.5 | 71 | 2.2 |
| White-winged Chough | 50 | 1.5 | - | - | 40 | 4.1 |
| Yellow Thornbill | 95 | 0.2 | 96 | 6.5 | - | - |

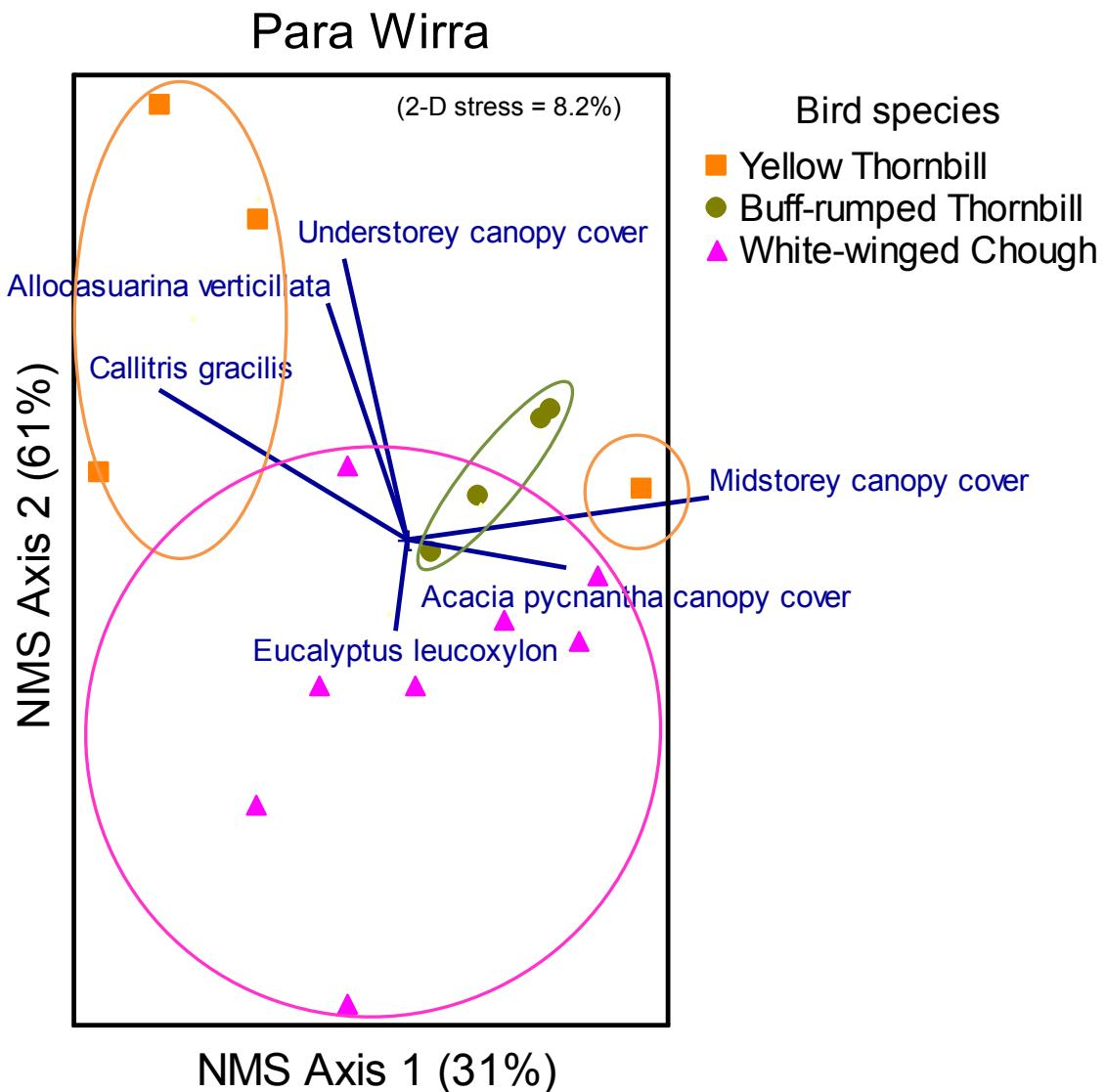


Figure 9.6: Two-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features in the respective hot-spots for three bird species surveyed within the Para Wirra Recreation Park. Species are the Yellow Thornbill (*Acanthiza nana*), Buff-rumped Thornbill (*A. reguloides*) and White-winged Chough (*Corcorax melanorhamphos*). Superimposed are the habitat variables that were associated with areas of high habitat use (i.e. 30 m × 30 m plots within hot-spots) that helped to delineate differences between the species at this site. These are the number of Slender Cypress-pine (*Callitris gracilis*), number of Drooping Sheoak (*Allocasuarina verticillata*), number of South Australian Blue Gum (*Eucalyptus leucoxylon*), midstorey canopy cover, Golden Wattle (*Acacia pycnantha*) canopy cover and understorey canopy cover. The orientation of the vectors indicates the direction of positive increase and the length represents the strength of association. Each colour-coded symbol represents a 30 m × 30 m plot within a hot-spot for a particular bird species. The colour-coded ellipses represent ordination space envelopes for each bird species and are used here for visual aids to assist pattern interpretation, but are not statistically based. At a stress level of 8.2%, 92% of the variation within the ordination plot is represented by the distribution of the bird species. Data collected July – August 2009.

Sandy Creek

Within the Sandy Creek site (Fig. 9.7a), the bird species surveyed were the Hooded Robin (MCP 7.2 ha), Rufous Whistler (MCP 14.2 ha), White-browed Babbler (MCP 6.6 ha), White-winged Chough (MCP 196.4 ha) and a solitary Willie Wagtail (MCP 9.6 ha; Fig. 9.7b). While overlap in the MCP areas used by these five bird species averaged $47.3 \pm 8.7\%$, there was minimal overlap between the hot-spots for the individuals belonging to each species ($4.9 \pm 1.6\%$ overlap; Table 9.5).

With a stress value of 6.6%, the two-dimensional NMS ordination plot for the birds surveyed at Sandy Creek revealed strong habitat associations, with the percent of the variation represented by the distribution of the birds being 94% (Fig. 9.8). The White-winged Chough was commonly found within patches that contained minimal understorey but with considerably more high litter cover, typically under *Eucalyptus* spp (Fig. 9.8). The White-winged Chough was also associated with very open habitat (Fig. 9.8), which was related to the sheep-grazed farmland that adjoined this Park. The Rufous Whistler aligned with patches that contained foliage across all storey heights, with patches of high habitat use correlating with the highest combined presence of Slender Cypress-pine and understorey canopy covers (Fig. 9.8). In contrast, the Hooded Robin associated with more open patches with minimal understorey (Fig. 9.8). The White-browed Babbler appeared to be influenced by habitat heterogeneity, responding to patches containing both high *Eucalyptus* spp and Slender Cypress-pine canopy cover, intermixed with more open habitats (Fig. 9.8). The Willie Wagtail used patches containing both high and low vegetation cover, together with patches with very low overstorey abundances (Fig. 9.8).

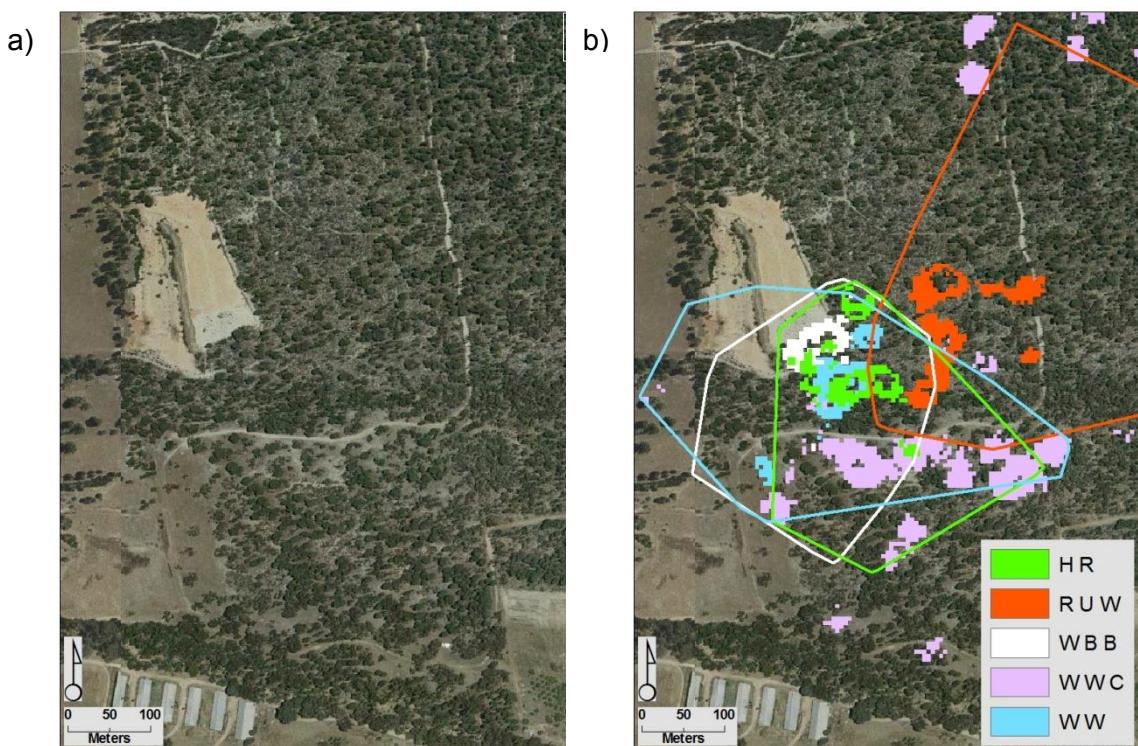


Figure 9.7: Survey site at Sandy Creek (a), with (b) patches of high habitat use (i.e. hot-spots) for the Hooded Robin (*Melanodryas cucullata*; HR), Rufous Whistler (*Pachycephala rufiventris*; RUW), White-browed Babbler (*Pomatostomus superciliosus*; WBB), White-winged Chough (*Corcorax melanorhamphos*; WWC) and a solitary Willie Wagtail (*Rhipidura leucophrys*; WW). The coloured lines show the Minimum Convex Polygons (MCP) that enclosed all points for each species. The MCP for the Hooded Robin is shown in green (MCP 7.2 ha). The MCP for the Rufous Whistler is shown in red (MCP 14.2 ha). The MCP for the White-browed Babbler is shown in (MCP 6.6 ha). The MCP for the White-winged Chough is not evident due to the WWC home range exceeding past the scale of the map shown (MCP 196.4 ha). The MCP for the Willie Wagtail is shown in blue (MCP 9.6 ha). Data collected June – September 2009.

Table 9.5: Percent habitat overlap between five co-inhabiting bird species at Sandy Creek. The percent overlap in MCP areas (habitat area used by each species during the survey as derived from Minimum Convex Polygon (MCP) analysis) and HS (the amount of habitat overlap between the areas of high habitat use, or hot-spots, used by each species) are shown for each combination of species. The species were the Hooded Robin (*Melanodryas cucullata*; HR), Rufous Whistler (*Pachycephala rufiventris*; RUW), White-browed Babbler (*Pomatostomus superciliosus*; WBB), White-winged Chough (*Corcorax melanorhamphos*; WWC) and a solitary Willie Wagtail (*Rhipidura leucophrys*; WW). Rows show the extent (as a percent) to which both a species home range and associated hot-spots overlap with that of other co-inhabiting species (refer Fig. 9.7b). Data collected June – September 2009.

| Species surveyed | HR | | RUW | | WBB | | WWC | | WW | |
|----------------------|-----|------|-----|------|-----|------|-----|------|-----|------|
| | MCP | HS |
| Hooded Robin | - | - | 23 | 0.0 | 67 | 20.2 | 4 | 10.3 | 82 | 16.2 |
| Rufous Whistler | 11 | 0.0 | - | - | 7 | 0.0 | 9 | 0.2 | 14 | 0.0 |
| White-browed Babbler | 70 | 21.6 | 15 | 0.0 | - | - | 4 | 0.0 | 91 | 7.6 |
| White-winged Chough | 100 | 0.1 | 100 | 0.01 | 100 | 0.0 | - | - | 100 | 1.2 |
| Willie Wagtail | 60 | 11.2 | 20 | 0.0 | 62 | 4.9 | 6 | 4.5 | - | - |

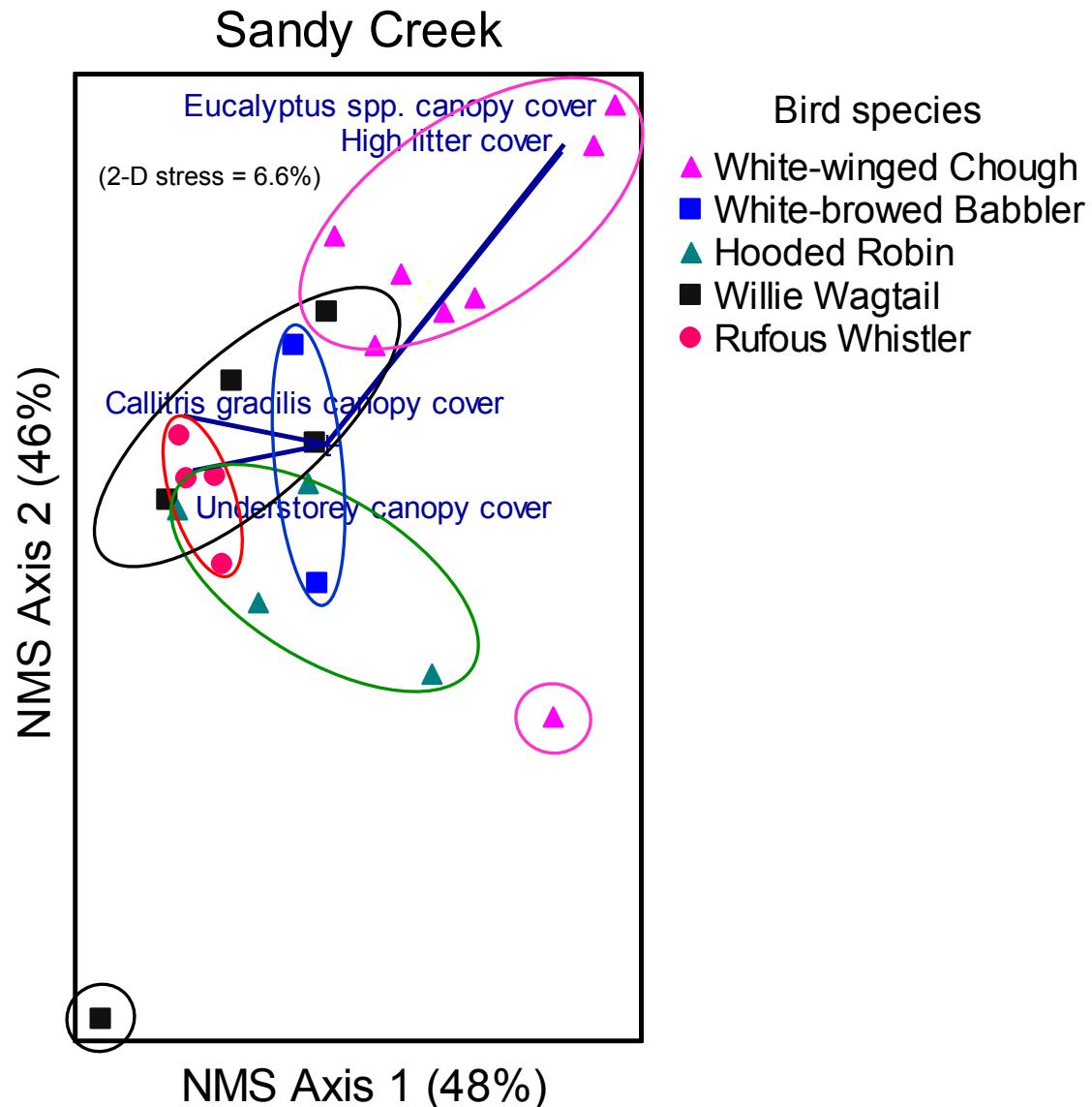


Figure 9.8: Two-dimensional Nonmetric Multidimensional Scaling (NMS) ordination with Sørensen (Bray-Curtis) distance measure for vegetation features in the respective hot-spots for five bird species surveyed at Sandy Creek. Species are the White-winged Chough (*Corcorax melanorhamphos*), White-browed Babbler (*Pomatostomus superciliosus*), Hooded Robin (*Melanodryas cucullata*), Willie Wagtail (*Rhipidura leucophrys*; a solitary bird) and Rufous Whistler (*Pachycephala rufiventris*). Superimposed are the habitat variables that were associated with areas of high habitat use (i.e. 30 m × 30 m plots within hot-spots) that helped to delineate differences between the species at this site. These are *Eucalyptus* spp canopy cover, high litter cover, Slender Cypress-pine (*Callitris gracilis*) canopy cover and understorey canopy cover. The orientation of the vectors indicates the direction of positive increase and the length represents the strength of association. Each colour-coded symbol represents a 30 m × 30 m plot within a hot-spot for a particular bird species. The colour-coded ellipses represent ordination space envelopes for each bird species and are used here for visual aids to assist pattern interpretation, but are not statistically based. At a stress level of 6.6%, 94% of the variation within the ordination plot is represented by the distribution of the bird species. Data collected June – September 2009.

Discussion

This chapter demonstrated that within all four study areas, discrete patches of habitat that were preferentially used by birds of one species were distinct from other discrete patches of habitat that were preferentially used by birds of other co-inhabiting species. The fact that strong MCP overlap existed between birds of co-inhabiting species, yet their respective hot-spots did not, also demonstrated that habitat heterogeneity at fine-scales is a strong mechanism that allowed birds belonging to different species the ability to co-exist within the one habitat area. This is not a new phenomenon, for although community structure appears to be relatively stable at the regional scale, high variability in habitat use at local scales has been reported from a wide variety of systems (e.g. Wiens 1989b). These findings are also consistent with the niche overlap hypothesis (May and Mac Arthur 1972), which implies that the extent to which co-inhabiting species separate spatially can be directly related to the variability of habitat attributes that each species is responding to. The niche overlap hypothesis predicts that co-inhabiting species that feed on similar foods (e.g. invertebrates) should separate spatially (i.e. use different areas), or if they overlap extensively in the areas they use, these species should feed on different food resources. In this study, there was minimal habitat overlap in the core areas where birds belonging to different species spent most of their time. For those individuals that belonged to different species where there was higher overlap in habitat use, the species differed greatly in body size and behaviour, so each were likely to be exploiting different food resources and/or using different substrates.

This study also helps explain how some birds that belong to different species are able to co-inhabit when they appear to broadly overlap in the habitat areas that they use, other than through shared habitat affinities. A further key finding was that the habitat attributes that were identified to be influencing habitat use by individual birds belonging to the various species across the study region (refer Chapter 8), also helped to delineate the spatial use of habitats between birds when these species co-inhabited at small spatial scales. In essence, within a habitat area, some patches that constituted a cold-spot for birds of one species were being simultaneously perceived as being of better quality habitat by birds of another co-inhabiting species. These findings are highly relevant for the design of new revegetation

at the fine-scale (e.g. 30 m × 30 m), to be of benefit for multiple co-inhabiting bird species.

Habitat partitioning between ecologically similar species can also be achieved in a number of other ways. For example, some co-inhabiting species proportion the same resource. The Brown Treecreeper has been recorded as sharing the same territory as the White-throated Treecreeper (*Cormobates leucophaeus*; Keast 1957; Wheeler 1966). These Treecreepers primarily feed on ants, with both bird species having been observed foraging on the same tree at the same time (Noske 1979). However the niches of these two species differ, with the Brown Treecreeper foraging for extended periods on the ground and fallen timber and primarily using tree trunks and major limbs as foraging substrates. In comparison, the White-throated Treecreeper rarely forages on the ground and forages on most woody parts of the tree (Noske 1979). The other co-occurring bark-foraging bird species within the Mount Lofty region is the Varied Sittella (*Daphoenositta chrysoptera*). Although this species rarely consumes ants, their search for invertebrate prey does not impose on that of the Treecreepers, as their foraging is directed primarily towards dead branches and in excavating flaky bark (Noske 1985).

Some ecologically similar bird species also proportion the same range of resources differently. The Monarto Woodland in South Australia is an approximately 35 year-old revegetation. In this Woodland, Northeast (2007) investigated habitat use by co-inhabiting pairs of colour-banded Hooded Robins and Red-capped Robins (*Petroica goodenovii*). In this area, the Hooded Robin was a habitat specialist while the Red-capped Robin was more of a habitat generalist. Importantly, this study also demonstrated that within NMS ordination space, the habitat attributes that were used by the Hooded Robins in the Monarto revegetation were primarily nested within, but distinct from, those used by the Red-capped Robins that occupied the same habitat area.

In comparison, some closely related bird species are able to co-inhabit by using different vegetation types within the same storey level. The Yellow Thornbill and Striated Thornbill (*A. lineata*) are both typically arboreal foliage gleaners within the overstorey strata. As indicated in Chapters 5, 8 and this chapter, the Yellow Thornbill used habitat areas that predominantly contained vegetation with bifurcated foliage. Conversely, the Striated Thornbill is mainly found within eucalypt forests and woodlands (Driscoll 1977; Loin

1985; Gosper 1992). The different habitat association by these two Thornbill species was apparent within the Para Wirra site during this study. The Yellow Thornbill was never encountered within the southern part of this study area, where various gum eucalypts and open canopies were prominent. Instead, the Yellow Thornbill associated with the native Pine and Drooping Sheoak that were only evident within this study area in the northern part. Habitat use by the Striated Thornbill was not documented in the Para Woodland region because it was not one of the target bird species. However as the Yellow Thornbill dropped out as the eucalypts began to dominate, the Striated Thornbill began making an appearance.

Clearly, habitat variability at the micro-scale (measured here at 30 m × 30 m) promotes occupation by multiple co-inhabiting bird species. This reinforces the findings and recommendations made throughout this thesis, that revegetation practitioners and land managers need to emulate nature by creating a repeated mosaic of different habitat patches, which include the range of specific individual habitat needs of a suite of pre-determined bird species. Important habitat attributes also need to be functionally connected and replicated across sufficiently large plantings, to provide the appropriate amount of fine-scale habitat attributes required to support and to be of benefit to populations of individual bird species. The minimum home range estimates for the bird species in Chapter 6 would require that such plantings need to incorporate areas in the range of tens to hundreds of hectares.

Identifying the important habitat needs of target bird species and their suggested spatial arrangements (e.g. Chapter 8 and this chapter), then provides restoration practitioners with the information that is required to effectively incorporate these attributes into reconstructed ecosystems. Doing this should increase the likelihood that new revegetation will provide useful habitats for species that are most at risk within modified landscapes, and so contribute to the long-term viability of their populations.

Caution is warranted however to not adopt an overly simplistic view of species habitat requirements. The habitat attributes and the scales at which they were measured in areas around the Para Woodland Reserve may not have captured other important aspects of the environment (Clarke 1993). This chapter and Chapters 5, 7 and 8 dealt with habitat use by birds at small spatial scales. This was not intended to suggest that this scale was the

optimal scale for characterising the level of spatial heterogeneity that was important to the birds. However, for the bird species that were studied in areas near the Para Woodland Reserve, the birds' hot-spots were typically in the vicinity of 30 m × 30 m in area. This is important information when aiming to inform on patch-scale restoration to meet the habitat requirements of birds. The findings in this chapter and those of the previous chapters, clearly demonstrate how information on fine-scale habitat use by woodland birds can be used to help inform patch-scale revegetation for woodland habitats.

Birds respond to their environment at multiple scales, and more so than many other organisms (Kotliar and Wiens 1990; Luck 2002; Wu 2004), so there is also a critical need to understand their habitat requirements at multiple scales. For instance, the habitat matrices in which hot-spots were embedded may also provide useful habitat, albeit not to the same degree that hot-spots did during these surveys. Also, the hot-spots considered in this study may not always remain so, and the same could also apply to cold-spots. Low use habitat areas may have been under-utilised because other areas were being used more during this study. Indeed, the so called cold-spots retained some level of use by the birds and may even become important at other times. This was clearly illustrated by the pair of Hooded Robins switching to use more enclosed patches of habitat during stormy weather (Chapter 7 and this chapter).

Nevertheless, the key recommendation from the observations in this study on woodland birds is that revegetation programs should consciously aim to create a mosaic of different-sized patches of vegetation that varies in structural and floristic elements, so as to produce heterogeneous habitats that can support suites of bird species. In the case of declining woodland birds, each patch could be tailored to the individual habitat needs of a pre-determined species, with a suite of different patches incorporated and replicated across an area that is being revegetated. High levels of habitat heterogeneity also permit high degrees of temporal and spatial flexibility in habitat use. This not only allows for different bird species to co-exist, but also allows individual birds the ability to change their habitat use should the need arise (e.g. in response to a change in ambient conditions).

Chapter 10

Summary and concluding comments

Synopses of chapters

This project was specifically designed to inform on habitat use by a suite of locally declining temperate woodland bird species, to assist in the revegetation activities on the Para Woodland Reserve, located within the north-central zone of the southern Mount Lofty Ranges (MLR) in South Australia. Chapter 1 highlighted the general plight of woodland birds within the MLR. A broad cross-section of these bird species is currently in decline, unlike some of their eastern State counterparts. Factors driving these declines include the speed at which broad-scale habitat clearance took place by the early European settlers across this region. The region as a whole is also geographically disconnected from similar bird populations located elsewhere in Australia. This limits the ability for external sources to supplement and replenish the bird species that are experiencing decline within the MLR, leading to the MLR being referred to as a “canary landscape” (Szabo *et al.* 2011) and an “urban ark” (Tait and Daniels 2005; Tait *et al.* 2005b). The dire state of woodland birds within the MLR should sound an alarm for the future of similar bird populations located in other woodland systems across southern Australia.

A review of the literature in Chapter 2 found that most ecological plantings that are intended to enhance the habitat value for fauna species are modelled on coarse-scale habitat characteristics of remnant vegetation. However, a critical missing component within the literature surrounds the paucity of information relating to the habitat requirements of bird species at finer-scales. This gap in knowledge is a limiting factor in creating useful revegetated habitats for a range of bird species that are currently of conservation concern. Whilst information regarding the habitat needs of fauna species at multiple scales is important, also important is the placement of revegetation within the landscape. The Para Woodland Reserve is a prime example of a property that when

revegetated, has the potential to provide both high quality habitats for woodland birds and enhance the collective value of woodland habitats within this region.

Chapter 3 considered coarse-scale habitat associations (at 10 ha) by woodland birds across grassy woodland, open heath woodland and closed heath woodland habitats. The key finding was that different woodland types supported a distinct assemblage of bird species. Importantly, this study also showed that birds responded strongly to woodland type, seemingly regardless of where that woodland was located within the landscape or with what landscape setting it was associated with. Also revealed, was that a particular woodland type did not support a full suite of declining bird species within the close proximity to the Para Woodland Reserve. The key recommendation from this chapter was to integrate a mixture of different woodland types into revegetation practice, to improve the habitat quality for a broader suite of declining bird species within this region.

Chapters 4 and 5 used the data that were collected at the ten hectare scale to ask questions relating to both the extent of structural habitat heterogeneity within each of the 15 study sites and associations by particular bird species with that heterogeneity. These data showed that relatively homogeneous woodlands at ten hectares contained significant amounts of habitat heterogeneity. The point was then raised as to the inadequacy of surveying at small spatial scales, which are then used to infer species-habitat associations for the wider surrounding habitat. Importantly, numerous bird species were statistically associated with elements of structural habitat heterogeneity within many of the ten hectare sites. These results highlighted the importance of incorporating this scale of habitat heterogeneity into revegetation design and that such heterogeneity should also incorporate the habitat needs of individual bird species. This information builds the case for the critical need for fine-scale studies of habitat use by woodland birds and for those details to be included in the revegetation toolbox.

Chapter 6 showed that the minimum home range requirements of several declining bird species within the north-central zone of the southern Mount Lofty Ranges were much larger than expected. Revegetation efforts must then reflect this habitat requirement of these birds by providing sufficient habitat area. Based on the finding of this study, to be of benefit revegetation should cover tens if not hundreds of hectares, in order to support viable populations of these bird species.

Chapter 7 exposed high degrees of variable habitat use by individuals of a number of different bird species within their respective minimum home range areas. This variable use of habitat occurred both temporally and spatially, and was influenced by a range of factors. These results brought into question the effectiveness of data sets that are temporarily biased and spatially limited, which are then used to infer species-habitat associations and to predict the distribution of species across broader habitat areas. In combination with the large area requirements of birds from Chapter 6, the reported movement rates of birds within their home ranges was also used to challenge the usefulness of the popular twenty-minute / two-hectare bird survey method. These results imply that bird surveys that are aimed at calculating species estimates and defining species-habitat relationships should not be so biased towards particular times of the day, should incorporate wider areas of habitat, and should be conducted under variable weather conditions.

Chapter 8 explored the fine-scale use of habitat within individual home ranges of birds belonging to the nine declining bird species within the north-central zone of the southern Mount Lofty Ranges. Numerous habitat attributes were identified in areas that were associated with both high use habitat areas (i.e. hot-spots) and low use habitat areas (i.e. cold-spots). These habitat features were used to help explain the habitat associations for the respective birds. Identified attributes that either promote or hinder habitat use by birds must be considered when planning revegetation and restoration activities for target bird species.

Chapter 9 advanced the knowledge gained in the previous chapters by comparing habitat use by individuals of co-inhabiting bird species. This study showed that highly used patches of habitat that were associated with birds of one species were distinct from those that were used by other co-inhabiting birds belonging to other species that occupied a common habitat area. Critically, the same habitat attributes that were identified in Chapter 8 as influencing habitat use by birds of a particular species, clearly differentiated habitat use between birds of other co-inhabiting species. This supported the argument for revegetation to contain a repeated mosaic of different habitat patches, with each patch “type” catering to the individual habitat needs of one (or a few) pre-determined bird species.

Area requirements versus habitat quality

Two major themes that emanated from this study were the large minimum home range requirements of birds of the species that were studied and the quality of that habitat from the perspective of the birds of each species. In relation to minimum home range spatial requirements, Smallwood (2001) highlighted the importance of adequate habitat area, by showing how spatially limited habitats that contained seemingly appropriate structural elements can fail to support functional bird populations. The question then is what constitutes an appropriately sized area for the majority of bird species? Clearly, the provision of wildlife corridors through linear strips of vegetation (see Chapter 2) and/or small “stepping-stone” patches of habitat will not produce the sheer habitat area that is required by most bird species. The minimum home range estimates for the birds in this study within in the remnant woodlands in the vicinity of the Para Woodland Reserve, would suggest that within highly fragmented landscapes (such as the Mount Lofty Ranges), an addition of many tens to hundreds of ecologically-intact hectares are required to stabilize populations of numerous declining bird species to prevent their local extinction. Ecologically-intact habitat would consist of either continuous habitat areas or large patches of contiguous habitat that the birds can easily move between.

Alarmingly, of the remaining 10% or so of the pre-European vegetation across the Mount Lofty Ranges, only an estimated half is considered to be well connected (Bushcare 2003). This suggests that not only is there a lack of natural habitat, it is packaged in many small areas that are unlikely to sustainably support declining woodland bird species within this region. The 30% requirement for natural intact habitat to cater for the needs of fauna and flora is severely compromised across the Mount Lofty Ranges, so further species losses under the extinction debt hypothesis can be expected (Ford and Howe 1980; Paton *et al.* 2004; Possingham *et al.* 2006; Szabo *et al.* 2011). This minimum benchmark of native habitat cover would seem a long way off for regions such as the Mount Lofty Ranges; which is also echoed across other agricultural zones throughout southern Australia. Reaching this target however needs to be embedded in our thinking if the full payment of the extinction debt is to be avoided.

Birds with adequately large habitat area ultimately respond to habitat quality (Mac Nally *et al.* 2002a; Laiolo *et al.* 2004; Loyn *et al.* 2007; Munro *et al.* 2011). Revegetation must

then be aimed at incorporating the ecological requirements of species of concern that may otherwise be missing and/or limited within the landscape (e.g. Taylor 2000). This highlights the need to consider the individual habitat needs of species. However, there is a limited understanding of what habitat attributes most species require, or the scales at which they are required (Walker *et al.* 2004). As a result, many well-meaning restoration projects have failed to deliver for the species we are now trying to save (Kimber *et al.* 1999; Vesk and Mac Nally 2006; Kavanagh *et al.* 2007; Loyn *et al.* 2007; Munro *et al.* 2007).

A good starting point in addressing habitat quality for birds is to consider the densities at which revegetation is planted. Many revegetation efforts are accused of planting at densities greater than historical habitats. Based on published information from early explorers and settlers, some authors argue that many traditional woodlands were savannah-like, with scattered trees among grassland (e.g. Rolls 1981; Flannery 1994; Ryan *et al.* 1996). These records show that much of the open woodlands of eastern Australian supported densities of between five to 50 trees per hectare, and commonly between 10 to 25 trees per hectare (Walker *et al.* 1993; Croft *et al.* 1997; Martin 2005; Lunt *et al.* 2006; Vesk and Mac Nally 2006; Gibbons *et al.* 2010). Indeed, the Victorian Government's guidelines for establishing native vegetation suggests a survival target of 50 overstorey trees per hectare when revegetating woodland habitats (Department of Sustainability and Environment 2006).

However, at the patch and landscape scale, many revegetation projects are notorious for planting overstorey species at very high initial densities, with many hundreds of trees planted per hectare. For example, up to 220 trees ha⁻¹ in Martin *et al.* (2004) and up to 500 stems ha⁻¹ in Michael *et al.* (2011). This is presumably owing to the fact that revegetation effectiveness is typically measured by the number of trees that are planted (Hobbs 2003). Planting at such high densities may also be to account for expected high mortalities of the planted trees. Although a problem exists when planting at high initial density in that it can adversely affects the eventual growth form of the mature tree. At high density, as developing plants compete for sunlight they direct their energy into vertical growth, severely limiting the development of low, lateral branches (Paton *et al.* 2004). Low perching substrates are a critical habitat component that is required by numerous insectivorous bird species that forage for some part on or near the ground (Arnold 2003). Of major concern, many bird species that forage on or near the ground are over-

represented by declining species (Robinson 1994). Densely planted vegetation can also restrict canopy development that may subsequently limit its usefulness as a foraging substrate for some foliage-gleaning birds (Arnold 2003). High planting densities can also potentially reduce nectar production due to competition with neighbouring plants (Paton 2008) and limit the resilience of some plant species to environmental stresses (e.g. Horner *et al.* 2009).

In comparison, revegetated areas with plants that are positioned further apart (e.g. 4 – 7 m) have produced habitats that are more useful for woodland bird species, including some that are currently regarded as declining (Arnold 2003; Martin *et al.* 2004; Paton *et al.* 2004). In support, Antos *et al.* (2008) showed that a low density of trees and shrubs was important for multiple bird species and that patches containing high tree densities tended not to be used by birds. Schemske and Brokaw (1981) also showed that the richness of bird species that use the understorey level was greater in small clearings where trees had fallen, when compared to the surrounding denser habitat.

Other authors argue for an alternative view of the purely “savannah-like” interpretation of pre-European woodlands. They suggest traditional woodlands contained far greater heterogeneity in stem numbers, ranging from very open areas to dense scrub and forest (e.g. Benson and Redpath 1997; Vesk and Mac Nally 2006; Gibbons *et al.* 2008). This interpretation of spatial habitat heterogeneity is supported by the findings in Chapter 4. Such floristic variability can be influenced by underlying environmental gradients (Gibbons *et al.* 2010), which must also be considered when planning habitat restoration.

Greater variability in plant densities (e.g. in clumps rather than uniform distribution) is required to produce spatial heterogeneity, creating patches containing both high and low vegetation cover (Harrington 1999). This would help cater for the array of habitat needs that are required by a suite of bird species, enabling multiple bird species to co-inhabit within a common habitat area (as demonstrated in Chapter 9). Ongoing management can then further enhance the habitat value for birds. For example, Mac Nally and Horrocks (2007) demonstrated the positive benefit achieved for birds by artificially manipulating fallen timber loads. Loyn *et al.* (2007) also showed how insectivorous species which forage within open patches among vegetation can find benefit from fallen branches left behind following vegetation thinning operations.

Throughout this Ph. D project, I argue strongly for the creation of large and complex habitat areas that would benefit a suite of woodland birds. However, the inadequate understanding within the literature of the habitat needs of individual bird species is a severely limiting factor for the effective return of new habitats. Likewise, the spatial arrangements of habitat features also need to be understood in order to create functional habitats for multiple species. The niche requirements for the birds of each species in this project were not provided for all in one concentrated area, but as more diffuse patches of higher quality habitat within a matrix of heterogeneous woodland. Intermixed diffuse patches that were under-utilised by birds of one species were often important habitat areas for another co-inhabiting bird of other species. In this manner, individuals of different bird species that overlapped significantly in the areas that they occupied largely used separate patches of habitat within that shared space.

Inter-species relationships

This study typically considered individual bird species in isolation from other co-occurring bird species. However, habitat use by one species can be directly influenced by the presence of others. The negative influence that the Noisy Miner (*Manorina melanocephala*) imposes on habitat use by other birds was raised in Chapter 3. Noisy Miners were not prominent within the study areas that were used for this project, so this was not a contributing factor behind the reported response of woodland birds to habitat features. The mere presence of some other bird species may also have a dramatic influence on habitat use by other birds. The formation of multi-species foraging flocks dominated by small bodied insectivorous birds is well known (e.g. Sedgwick 1949; Bell 1980a). Bell (1980a) found that birds smaller than the White-winged Chough (*Corcorax melanorhamphos*) generally participated within mixed-species feeding flocks, which tended to centre on the activities of Thornbills (*Acanthiza* spp) and/or Fairy-wrens (*Malurus* spp). The fact that numerous Thornbill species and the Superb Fairy-wren (*M. cyaneus*) have been shown to be in decline within the Mount Lofty Ranges (Szabo *et al.* 2011; and see Chapter 1), implies the potential breakdown of the community structure of these co-foraging species. This situation perhaps adds further pressure towards the ongoing decline of some bird species within the Mount Lofty region.

The influences of mixed-species group behaviour can also alter the typical behaviours of certain bird species. For example, some granivorous birds feed exclusively on invertebrates when associated with a foraging flock that is dominated by insectivorous birds (Greig-Smith 1978). Likewise, in similar company nectarivorous species also forage predominantly on invertebrates (Bell 1980a). These inter-species interactions may confound our understanding of the typical habitat needs of certain birds. In the light of such variable habitat use, it may even be naive to suggest that habitat use by a particular species can be generalised. This strengthens the argument raised in this thesis for revegetation to include a repeated montage of individual habitat patches, with each patch “type” catering to the known habitat needs of individual species. Doing this, also allows individual species the opportunity to use a wider variety of habitat elements. The need for a variety of habitat settings at small spatial scales was clearly demonstrated in this project, with the highly variable use of habitats by birds over relatively short time scales (Chapter 7).

Recommendations for future bird studies

Concerns were raised a number of times throughout this thesis as to the suspected design inadequacies of bird surveys that aim to capture important species-habitat relationships. These concerns essentially revolved around time biases, the small spatial scales at which surveys are conducted, and ecologically truncated data sets. The argument here is that the results from these studies are sensitive to where and when the observations were made. Accordingly, conclusions from such studies regarding the relationship of a species with the environment should be restricted to generalisations regarding habitat use at the scale at which it was measured, and not extended beyond the boundary of the sampling area or sampling period (Wiens 1989a). In support, Smith *et al.* (2008) reported that modelled solutions that were based on survey areas that were smaller than the home ranges of the birds they studied, severely under-estimated the habitat associations for the Black-throated Blue Warbler (*Dendroica caerulescens*) and Ovenbird (*Seiurus aurocapillus*) in American hardwood-conifer forests.

This current study only achieved a glimpse into the potential habitat associations for the bird species that were surveyed, yet it often yielded high variability in habitat use. A

concern remains that we are missing opportunities to expose such variability due to the short-term nature of most bird surveys, and many that are only conducted under constantly favourable weather conditions. This can only lead to overstating the results, by extrapolating the general findings from these ecologically limited data sets that do not capture the true extent of actual habitat use by birds under a range of conditions and habitat settings. This is best tackled through exhaustive studies over various temporal and spatial scales, and by focusing on habitat use under a variety of conditions. By achieving a greater understanding of the extent and variable nature of habitat use by birds, together with the range of alternative habitat needs of bird species, we can realistically produce more useful environments that specifically cater for multi-species objectives. More importantly, there must be strong links between such advances in knowledge and its integration into land management practices and policy (Bennett and Watson 2011).

At the qualitative level, Wiens (1989a, p. 284) suggested that “the general habitat associations of many species are known to any good birdwatcher” and that “the work of ecologists might be regarded as expressing (or sometimes obscuring) that knowledge in detailed quantitative analyses.” The issue of obscuring species-habitat associations stems from the fact that “most bird data sets contain only presence-absence records that are attempted to be correlated with measures of habitat features” (Wiens 1989a, p. 284). This author is warning that statistical analyses should be regarded as merely a tool in helping to define bird-habitat associations. This view would also condone the use of a more “natural history” approach to research, where observations are reported that have sound ecological backing, but which may not necessarily come with statistical support. In contrast, the current popularity in ecological modelling relies on a miscellaneous collection of unrelated data (i.e. “statistically independent”). Regrettably, in the pursuit for statistical independence, much relevant ecological data are excluded from many of these analyses. The take home message here is that the resultant outputs from these modelled solutions are only as good as the data that were used to generate them. Perhaps this is a reason why it is said that ecological theory and models of species distributions are often weakly linked (Elith and Leathwick 2009).

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Appendices

Appendix A: Photographs of the 15 ten hectare survey sites



Site 1 – grassy woodland (Para Wirra Recreation Park; 34°39.46'S, 138°49.19'E). This site was embedded within closed heath woodland and adjoins Site 2 (closed heath woodland).



Site 2 – closed heath woodland (Para Wirra Recreation Park; 34°39.38'S, 138°49.42'E). This site is floristically the richest part of this Park (Senior Ranger Erik Dahl, pers. comm.). This site lies between Site 1 (grassy woodland) and Site 3 (closed heath woodland).



Site 3 – closed heath woodland (Para Wirra Recreation Park; 34°39.57'S, 138°49.55'E). This site contained clogged heath (Crichton *et al.* 1978) and adjoins Site 2 (closed heath woodland).



Site 4 – open grassland with few trees (Para Wirra Recreation Park; 34°40.06'S, 138°50.39'E). This site was embedded within closed heath woodland and was chosen to embody a site having open woodland with few trees. Floristically this site shared most in common with Site 7 (grazed woodland) but hosted a bird community more similar to Site 1 (grassy woodland, also within Para Wirra). Sites 1 and 4 shared more birds in common with other grassy sites than sites with heath.



Site 5 – open heath woodland (Para Wirra Recreation Park; 34°41.25'S, 138°50.17'E). This site was originally selected *a priori* as open heath woodland but contained enough other plant species to be more similar to the other five closed heath woodland sites. Importantly, this site also hosted a bird community that was similar to the other closed heath woodland sites.



Site 6 – grassy woodland (Altona Kies; 34°35.04'S, 138°53.43'E). Private property protected by a Heritage Agreement. This site adjoins Site 14 (open heath woodland).



Site 7 – grazed woodland (Altona Chatterton; 34°34.41'S; 138°54.18'E). Floristically this site shared most in common with Site 4 (open woodland with few trees) but hosted a bird community that was similar to the other two grassy woodland sites that were located nearby (Sites 6 and 15). This site adjoins Site 14 (open heath woodland).



Site 8 – open heath woodland (Altona Landcare - Grundy Block; 34°35.14'S, 138°54.51'E). This site adjoins Site 15 (grassy woodland), with both areas managed by the Williamstown-Lyndoch Landcare Group Inc.



Site 9 – closed heath woodland (Para Wirra - SA Water property; 34°40.19'S, 138°51.00'E).



Site 10 – open heath woodland (Sandy Creek Conservation Park; 34°36.29'S, 138°51.37'E).



Site 11 – open heath woodland (Sandy Creek Conservation Park; 34°36.22'S, 138°51.04'E).



Site 12 – closed heath woodland (Hale Conservation Park; 34°41.14'S, 138°54.24'E). This site incorporated numerous undulating flanks that radiated from several drainage depressions.



Site 13 – closed heath woodland (Hale Conservation Park; 34°41.30'S, 138°54.41'E). This site incorporated the eastern and western flanks on either side of a steep crest.



Site 14 – open heath woodland (Altona Chatterton; 34°34.53'S, 138°54.11'E). Private property protected by a Heritage Agreement. This site lies between Site 6 (grassy woodland) and Site 7 (grazed woodland).



Site 15 – grassy woodland (Altona Landcare; 34°35.03'S, 138°54.49'E). This site adjoins Site 8 (open heath woodland), with both areas managed by the Williamstown-Lyndoch Landcare Group Inc.

Bird breeding and foraging data

Appendix B: Bird species detected breeding during the surveys

| Nesting substrates and associated plant species | Australian Magpie | Brown Falcon | Brown Treecreeper | Brush Bronzewing | Buff-rumped Thornbill | Collared Sparrowhawk | Common Bronzewing | Crescent Honeyeater | Galah | Grey Currawong | Grey Shrike-thrush | Hooded Robin | Musk Lorikeet | Rainbow Bee-eater | Red-rumped Parrot | Red Wattlebird | Rufous Whistler | Striated Pardalote | White-browed Babbler | White-winged Chough | Varied Sittella | Yellow-rumped Thornbill |
|--|-------------------|--------------|-------------------|------------------|-----------------------|----------------------|-------------------|---------------------|-------|----------------|--------------------|--------------|---------------|-------------------|-------------------|----------------|-----------------|--------------------|----------------------|---------------------|-----------------|-------------------------|
| BRANCHES | | | | | | | | | | | | | | | | | | | | | | |
| Box Mistletoe (<i>Amyema miquelii</i>) | | | | | | | | | | | | | | | | | | | | | | • |
| Golden Wattle (<i>Acacia pycnantha</i>) | | | | | | | | | | | | | | | | | • | | | | | |
| Hakea (<i>Hakea carinata</i>) | | | | • | | | | | | | | | | | | | | | | | | |
| Kangaroo Thorn (<i>Acacia paradoxa</i>) | | | | | • | | | | | | | | | | | | | | • | | | |
| Long-leaved Box (<i>Eucalyptus goniocalyx</i>) | • | | | | | | | | | | • | | | | | | | | | | | |
| Olive (<i>Olea europaea</i>) | | | | | | | | | | | | | | | | | | | | | | • |
| Peppermint Box (<i>Eucalyptus odorata</i>) | • | | | | | | | | | | • | | | | | | | | | | • | |
| Slender Cypress-pine (<i>Callitris gracilis</i>) | | | | | • | | • | | | | • | • | | | | | • | • | • | | | |
| Slaty Sheoak (<i>Allocasuarina muelleriana</i>) | | | | | | | | • | | | | | | | | | | | | | | |
| South Australian Blue Gum (<i>Eucalyptus leucoxylon</i>) | • | • | | | | | | • | | | | | | | | | | | | | • | |
| HOLLOWS | | | | | | | | | | | | | | | | | | | | | | |
| Long-leaved Box (<i>Eucalyptus goniocalyx</i>) | | | | | | • | | | | | | | | | | | | | • | | | |
| Peppermint Box (<i>Eucalyptus odorata</i>) | | | | | • | | | | | | | | | | • | • | • | | | | | |
| Pink Gum (<i>Eucalyptus fasciculosa</i>) | | | | | | • | | | | | | | | | | | • | • | | | | |
| Slender Cypress-pine (<i>Callitris gracilis</i>) (in fissures) | | | | | • | | | | | | | | | | | | | | | | | |
| South Australian Blue Gum (<i>Eucalyptus leucoxylon</i>) | | | | | | • | • | • | | | • | | | | • | • | • | | | | | |
| VEGETATION General | | | | | | | | | | | | | | | | | | | | | | |
| Mount Lofty Grass-tree (<i>Xanthorrhoea quadrangulata</i> ssp. <i>quadrangulata</i>) (in dead foliage) | | | | | | | | • | | | | | | | | | | | | | | |

| Nesting substrates and associated plant species | Australian Magpie | Brown Falcon | Brown Treecreeper | Brush Bronzewing | Buff-rumped Thornbill | Collared Sparrowhawk | Common Bronzewing | Crescent Honeyeater | Galah | Grey Currawong | Grey Shrike-thrush | Hooded Robin | Musk Lorikeet | Rainbow Bee-eater | Red-rumped Parrot | Red Wattlebird | Rufous Whistler | Striated Pardalote | White-browed Babbler | White-winged Chough | Varied Sittella | Yellow-rumped Thornbill |
|---|-------------------|--------------|-------------------|------------------|-----------------------|----------------------|-------------------|---------------------|-------|----------------|--------------------|--------------|---------------|-------------------|-------------------|----------------|-----------------|--------------------|----------------------|---------------------|-----------------|-------------------------|
| VEGETATION General cont. | | | | | | | | | | | | | | | | | | | | | | |
| South Australian Blue Gum (<i>Eucalyptus leucoxylon</i>) (behind loose bark at base of the trunk) | | | | | • | | | | | | | | | | | | | | | | | |
| Spiky Guinea-flower (<i>Hibbertia exutiacies</i>) | | | | • | | | | | | | | | | | | | | | | | | |
| Yacca (<i>Xanthorrhoea semiplana</i>) | | | • | | | • | | | | | | | | | | | | | | | | |
| OTHER | | | | | | | | | | | | | | | • | | | | | | | |
| Excavating a hole in the sandy hillside by a creek | | | | | | | | | | | | | | | | | | | | | | |

Appendix C: Food resources used by birds during the surveys

| Foraging resources | | Brown Thornbill | Brown Treecreeper | Buff-rumped Thornbill | Crimson Rosella | Diamond Firetail | Dusky Woodswallow | Golden Whistler | Grey Currawong | Grey Shrike-thrush | Hooded Robin | Jacky Winter | Mistletoebird | Musk Lorikeet | Rainbow Bee-eater | Red-browed Finch | Red-capped Robin | Red-rumped Parrot | Rufous Whistler | Silveryeye | Spotted Pardalote | Striated Pardalote | Striated Thornbill | Varied Sittella | Weebill | Welcome Swallow | White-browed Babbler | White-browed Scrubwren | White-winged Chough | Willie Wagtail | Yellow-faced Honeyeater | Yellow-rumped Thornbill | Yellow Thornbill | Zebra Finch |
|--|--|-----------------|-------------------|-----------------------|-----------------|------------------|-------------------|-----------------|----------------|--------------------|--------------|--------------|---------------|---------------|-------------------|------------------|------------------|-------------------|-----------------|------------|-------------------|--------------------|--------------------|-----------------|---------|-----------------|----------------------|------------------------|---------------------|----------------|-------------------------|-------------------------|------------------|-------------|
| SEEDS | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Bearded Oat (<i>Avena barbata</i>) | | | | • | • | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | |
| Drooping Sheoak (<i>Allocasuarina verticillata</i>) | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | | |
| Olive (<i>Olea europaea</i>) (stones and fruit) | | | | | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Perennial Veldtgrass (<i>Ehrharta calycina</i>) | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Pussy Tail (<i>Pentaschistis pallid</i>) | | | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | |
| Silvery Hair-grass (<i>Aira caryophyllea</i>) | | | | | | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Slaty Sheoak (<i>Allocasuarina muelleriana</i>) | | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | • | | | | |

| Foraging resources | | Brown Thornbill | Brown Treecreeper | Buff-rumped Thornbill | Crimson Rosella | Diamond Firetail | Dusky Woodswallow | Golden Whistler | Grey Currawong | Grey Shrike-thrush | Hooded Robin | Jacky Winter | Mistletoebird | Musk Lorikeet | Rainbow Bee-eater | Red-browed Finch | Red-capped Robin | Red-rumped Parrot | Rufous Whistler | Silveryeye | Spotted Pardalote | Striated Pardalote | Striated Thornbill | Varied Sittella | Weebill | Welcome Swallow | White-browed Babbler | White-browed Scrubwren | White-winged Chough | Willie Wagtail | Yellow-faced Honeyeater | Yellow-rumped Thornbill | Yellow Thornbill | Zebra Finch |
|--|--|-----------------|-------------------|-----------------------|-----------------|------------------|-------------------|-----------------|----------------|--------------------|--------------|--------------|---------------|---------------|-------------------|------------------|------------------|-------------------|-----------------|------------|-------------------|--------------------|--------------------|-----------------|---------|-----------------|----------------------|------------------------|---------------------|----------------|-------------------------|-------------------------|------------------|-------------|
| SEEDS continued | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Slender Cypress-pine (<i>Callitris gracilis</i>) | | | | • | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | | |
| Small Purslane (<i>Calandrinia eremaea</i>) | | | | | • | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | |
| Smooth Cat's-ear (<i>Hypochoeris glabra</i>) | | | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | |
| Sticky Hop-bush (<i>Dodonaea viscosa</i>) | | | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | | |
| VEGETATION | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Box Mistletoe (<i>Amyema miquelii</i>) (fruit) | | | | | | | | • | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Field Mushrooms (<i>Agaricus campestris</i>) | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | | | | |
| Flame Heath (<i>Astroloma conostephoides</i>) (flowers) | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | | | | |
| Guildford Grass (<i>Romulea rosea</i>) (bulbs) | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | | | | |
| Lesser Guildford Grass (<i>Romulea minutiflora</i>) (bulbs) | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | | | | |

| Foraging resources | | Brown Thornbill | Brown Treecreeper | Buff-rumped Thornbill | Crimson Rosella | Diamond Firetail | Dusky Woodswallow | Golden Whistler | Grey Currawong | Grey Shrike-thrush | Hooded Robin | Jacky Winter | Mistletoebird | Musk Lorikeet | Rainbow Bee-eater | Red-browed Finch | Red-capped Robin | Red-rumped Parrot | Rufous Whistler | Silveryeye | Spotted Pardalote | Striated Pardalote | Striated Thornbill | Varied Sittella | Weebill | Welcome Swallow | White-browed Babbler | White-browed Scrubwren | White-winged Chough | Willie Wagtail | Yellow-faced Honeyeater | Yellow-rumped Thornbill | Yellow Thornbill | Zebra Finch |
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| VEGETATION continued | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Long-leaved Box (<i>Eucalyptus goniocalyx</i>) (honey dew) | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Orchids (<i>Microtis</i> sp. and/or <i>Prasophyllum</i> sp.) (bulbs) | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Oyster Bay pine (<i>Callitris rhomboidea</i>) (male cones on branchlet tips) | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | | |
| Smooth Cat's-ear (<i>Hypochoeris glabra</i>) (flower) | | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | |
| Sods of mosses, herbs & grass shoots | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Sourso (Oxalis pes-caprae) (bulbs & occasionally leaves) | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Thread Iris (<i>Moraea setifolia</i>) (bulbs) | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| INVERTEBRATES | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>Camponotus</i> spp | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>Camponotus</i> undescribed | | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | | | |

| Foraging resources | | Brown Thornbill | Brown Treecreeper | Buff-rumped Thornbill | Crimson Rosella | Diamond Firetail | Dusky Woodswallow | Golden Whistler | Grey Currawong | Grey Shrike-thrush | Hooded Robin | Jacky Winter | Mistletoebird | Musk Lorikeet | Rainbow Bee-eater | Red-browed Finch | Red-capped Robin | Red-rumped Parrot | Rufous Whistler | Silveryeye | Spotted Pardalote | Striated Pardalote | Striated Thornbill | Varied Sittella | Weebill | Welcome Swallow | White-browed Babbler | White-browed Scrubwren | White-winged Chough | Willie Wagtail | Yellow-faced Honeyeater | Yellow-rumped Thornbill | Yellow Thornbill | Zebra Finch |
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| INVERTEBRATES continued | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>C. claripes</i> | | | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>C. ephippium</i> | | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>C. minimus</i> | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>C. piliventris</i> | | • | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | |
| Ant - <i>C. zeuxis</i> | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>Dolichoderus</i> spp | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>Iridomyrmex</i> spp | | • | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | |
| Ant - <i>I. purpureus</i> | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>Notoncus</i> sp. | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>Pheidole</i> spp | | • | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - <i>Prolasius</i> sp. | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Ant - Sub-Family Myrmicinae | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Beetle | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Beetle Larvae (birds unearthed) | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Caterpillar | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Cockroach | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Cocoon (unidentified invertebrate) | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Dragonfly | | | | | | | | | | | | | | | | | | | | | | | | | • | • | | | | | | | | |

| Foraging resources | | Brown Throombill | Brown Treecreeper | Buff-rumped Thornbill | Crimson Rosella | Diamond Firetail | Dusky Woodswallow | Golden Whistler | Grey Currawong | Grey Shrike-thrush | Hooded Robin | Jacky Winter | Mistletoebird | Musk Lorikeet | Rainbow Bee-eater | Red-browed Finch | Red-capped Robin | Red-rumped Parrot | Rufous Whistler | Silveryeye | Spotted Pardalote | Striated Pardalote | Striated Thornbill | Varied Sittella | Weebill | Welcome Swallow | White-browed Babbler | White-browed Scrubwren | White-winged Chough | Willie Wagtail | Yellow-faced Honeyeater | Yellow-rumped Thornbill | Yellow Throombill | Zebra Finch |
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| INVERTEBRATES continued | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| European Honey Bee (<i>Apis mellifera</i>) | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | | | |
| Fly Larvae | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Grasshopper / Locust | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | | |
| Grub | | • | • | | • | • | | • | • | • | • | | | | • | | • | • | • | • | | | | | | • | • | | | | | | | |
| Lacewing | | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | |
| Lerp | | • | • | | | | | | | | | | | | • | | | • | • | • | • | | | | | • | | | | | | | | |
| Moth | | • | | | | | | | | | | | | | | | • | | | | | | | | • | | | | | | | | | |
| Portuguese Millipede (<i>Ommatoiulus oreletii</i>) | | | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | | |
| Spider | | | • | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | |
| Termites (<i>Nasutermes</i> spp) | | | | | | | | | | | | | | | | | | | | | | | | | • | • | | | | | | | | |
| OTHER | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| Honey dew excreted from wattle tick scale (<i>Cryptes baccatus</i>) | | | | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | |
| Invertebrates associated with a kangaroo carcass | | | | | | | | | | | | | | | | | | | | | | | | | • | | | | | | | | | |
| Lizard (Scincidae) | | | | | | | | | | | | | | | | | • | | | | | | | | | | | | | | | | | |