Does post-mining rehabilitation on the Weipa bauxite plateau restore bird habitat values?

by

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Candidate's Declaration

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

Susan F. Gould Date:1 June 2009

Dedication

This work was inspired by love and gratitude for the birds of the Weipa bauxite plateau, who make the forests ring with song, but have no voice in shaping the plans of mankind for their forest homes. This thesis is dedicated to them.

Acknowledgements

It has been a privilege and a pleasure to be mentored by people who are experts in their respective fields, clear thinkers, and fine people to boot. I thank all of the members of my supervisory panel for their generosity in mentoring me through this process: Professor Brendan Mackey (chair) - Fenner School of Environment and Society at ANU, Professor Henry Nix - Fenner School of Environment and Society at ANU, Dr John Ludwig - CSIRO Sustainable Ecosystems Tropical Forest Research Centre at Atherton, David Tongway - Visiting Fellow Fenner School of Environment and Society at ANU, and Dr Penny Olsen - Visiting Fellow Botany and Zoology at ANU. Dr Ross Cunningham also provided early advice on research design before retiring from ANU.

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Abstract

Rehabilitation is seen as a key strategy for minimising biodiversity losses. Although most rehabilitation strategies aim to provide habitat for fauna, they usually focus entirely on establishing vegetation. Successful vegetation establishment, however, does not necessarily provide habitat to the same species that are threatened by habitat loss. Improved understanding of faunal response to rehabilitation is required if rehabilitation techniques are to be refined and deliver the hoped for biodiversity outcomes. This study aimed to assess to what extent post-mining rehabilitation on the Weipa bauxite plateau has restored the bird habitat values of the pre-mining native forest.

Bird assemblages, vegetation, and landscape functionality were compared between: (1) *Eucalyptus tetrodonta* open forest reference sites representative of the pre-mining native forest; (2) two reference land units of *Eucalyptus tetrodonta* tall woodland that have previously been nominated as ecologically appropriate analogues for the post-mining landscapes; and (3) a chronosequence of post-mining rehabilitation sites up to 23 years old.

Bird species richness and mean bird abundance increased with rehabilitation age. Bird species composition also changed and became more similar to native forest bird assemblages with increasing age. Significant differences remained, however, in mean bird abundance and composition of the bird assemblages between the oldest age class of mine rehabilitation and reference native forest land units. The mean bird species shortfall index in the oldest age class of mine rehabilitation was 63%, compared to a mean species shortfall index of 27% for pre-mining native forest sites. There were also significant differences in vegetation composition and structure between reference native forest land units and post-mining rehabilitation sites. Most importantly, the framework plant species that dominate the native vegetation community occurred at much lower densities in mine rehabilitation than in reference land units. Site detection rates of birds were strongly related to vegetation composition and structure.

It is concluded that mining and post-mining rehabilitation on the Weipa bauxite plateau has so far resulted in habitat conversion rather than habitat restoration. It therefore contributes to the causes of biodiversity decline. Post-mining rehabilitation created new habitat for 18 bird species not sourced from the pre-mining native forest. It also provided partial habitat for many of the generalist native forest bird species recorded, although their presence in the landscape remained dependent on access to native forest. However, rehabilitation did not provide the habitat resources that are required by habitat specialists and foraging specialists. This study found that the native forest bird species most sensitive to habitat loss, and most in need of habitat restoration, may be the last to return to rehabilitation if they return at all. The findings of this study have implications for rehabilitation practices, biodiversity conservation on the Weipa bauxite plateau, as well as broader implications for policies that rely on the assumption that rehabilitation can offset biodiversity losses.

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Chapter 1: Introduction

1.1 Research rationale and aim

Habitat loss, including habitat conversion, arising from human activities is recognised as one of the key drivers of biodiversity loss in Australia and globally (Primack, 2002; Sattler and Creighton, 2002; Millennium Ecosystem Assessment, 2005). If current rates of extinction continue it is estimated that as many as 30% of all species may be extinct by the mid 21st century. In Australia, although most states have introduced legislation aimed at reducing broadscale land clearing, the legacies of past vegetation clearing remain strong drivers of biodiversity decline (Cork *et al.*, 2006). Rehabilitation, including habitat restoration, is seen as one of a number of measures that can minimise human impacts on biodiversity (Australian and New Zealand Environment and Conservation Council, 1996) and is integral to recent Australian commonwealth and state government policy approaches to development approvals such as environmental offsets. Much depends, therefore, on our ability to successfully restore native habitats.

Many rehabilitation strategies that aim to provide habitat are passive with respect to fauna and focus almost entirely on establishing vegetation. The implicit assumption is based on what Palmer *et al.*, (1997: 295) refer to as the Field of Dreams hypothesis, that 'if you build it they will come'. Although vegetation based habitat resources are a pre-requisite for most vertebrate fauna, vegetation establishment is not a guarantee that the species most in need of habitat restoration will return. Large gaps remain in our understanding about which species return to rehabilitated areas and how long it takes them to return. Improved understanding of faunal response to rehabilitation across a range of ecosystem types and climate zones is required if rehabilitation techniques are to be improved and deliver the hoped for biodiversity outcomes. Assessing the effectiveness of rehabilitation techniques in restoring native habitats is a high research priority for managing biodiversity conservation (Australian and New Zealand Environment and Conservation Council and Biological Diversity Advisory Committee, 2001).

This study aimed to assess how effective post-mining rehabilitation on the Weipa bauxite plateau has been in restoring the bird habitat values of the pre-mining native vegetation. The Weipa bauxite plateau is an important location for studying the effectiveness of rehabilitation in restoring native habitat because of the potentially large scale of mining impacts and the extent of coverage of a unique ecosystem. The Weipa bauxite plateau is the world's largest proven bauxite reserve (Taylor *et al.*, 2008). The bauxite ore body extends along the west coast of Cape York Peninsula for 350 km north to south and 65 km east to west, covering an area of approximately 1.1 million ha. The distribution of a unique ecosystem of Darwin Stringybark (*Eucalyptus tetrodonta)* is highly correlated with the Weipa bauxite plateau. Current bauxite mining leases cover 585,526 ha, representing 47% of the *Eucalyptus tetrodonta* open forest on

Cape York Peninsula (Queensland Herbarium, 2005; Department of Mines and Energy, 2008). Minerals exploration leases cover an additional 314,231 ha of *E. tetrodonta* open forest. The total area of *E. tetrodonta* open forest on Cape York Peninsula that is covered by either mining lease or minerals exploration lease is 744,258 ha, 82% of the entire ecosystem. Less than 2% of the *E. tetrodonta* open forest is in the protected area estate (Accad *et al.*, 2008).

The Weipa bauxite plateau also offers several advantages for an investigation of this type. First, unlike much of temperate Australia, the opportunity still exists in the monsoonal eucalypt formations of northern Australia for comparing intact native ecosystems with rehabilitation. The vegetation of the northern tropical savannas, including the ecosystems of the Weipa bauxite plateau, is relatively intact compared to most other Australian habitats and tropical savannas globally (Woinarski, 1993; Woinarski *et al.*, 2007). The mining operation on the Weipa bauxite plateau is surrounded by continuous intact native vegetation. The opportunity still exists, therefore, to collect good quality baseline data on reference pre-mining ecosystems and to reduce the effects of potentially confounding variables.

Second, as a long term progressive surface mining operation, vegetation in the post-mining landscape on the Weipa bauxite plateau consists of a mosaic of rehabilitation ages. This provides the opportunity to study replicates of a chronosequence of rehabilitation sites rather than relying on a long term study. Comalco[1] has been mining bauxite at Weipa continuously since the late 1950's. Since at least 1981, the aim of post-mining rehabilitation at Weipa has been to establish 'self-sustaining maintenance-free vegetation comprising a variety of native plants which in turn support native fauna' (Reeders, 1985).

Various taxonomic groups have been used as indicators of mine rehabilitation success. This study does not assume that birds are indicators of biodiversity or habitat quality for other fauna. They are, however, an important component of the fauna in their own right. Birds are also sufficiently abundant to provide useful quantitative data and are easily monitored. Bird diversification over several trophic levels and their sensitivity to vegetation structure also make them potentially useful indicators of habitat change (Da Silva and Vickery, 2002; Pereira and Cooper, 2006; Olsen *et al.*, 2003).

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[1] Due to change in ownership the original Comalco is currently known as Rio Tinto Alcan.

1.2 Research questions

The overall research question addressed by this study was: To what extent has post-mining rehabilitation on the Weipa bauxite plateau restored the bird habitat values of the *Eucalyptus tetrodonta* open forest? This question was investigated via four key research questions:

1. How do bird assemblages in mine rehabilitation compare with native forest bird assemblages?

2. What is the temporal pattern of bird succession in mine rehabilitation on the Weipa bauxite plateau?

3. What environmental factors determine bird occupancy in mine rehabilitation and native forest on the Weipa bauxite plateau?

4. Is post-mining rehabilitation likely to restore habitat similar to that of pre-mining native forest on the Weipa bauxite plateau?

Bird species composition and abundance, vegetation composition and structure and landscape functionality were compared in: (1) native forest reference sites that are representative of the pre-mining *Eucalyptus tetrodonta* open forest land unit; (2) two native forest reference land units nominated as ecologically appropriate analogues for the post-mining landscape; and (3) a chronosequence of post-mining rehabilitation sites aged from one up to 23 years. This is the first study that has compared bird assemblages, vegetation and landscape function between reference sites and post-mining rehabilitation sites. It is also the first study on the Weipa bauxite plateau to directly compare bird assemblages and vegetation in pre-mining native forest and post-mining rehabilitation.

1.3 Thesis outline

To design a robust and suitably scaled study, it is necessary to understand the current state of knowledge of the topic as well as the ecology and environmental context of the study area. Chapter 2 provides the background information on the physical environment of the study area. It describes the geology, climate, hydrology, vegetation, fire regime, and birds of the Weipa bauxite plateau. Chapter 2 also outlines the bauxite mining and post-mining rehabilitation process.

The research literature relating to this topic covers a number of areas. Due to the broad scope of the literature, it is reviewed in two chapters. Chapter 3 provides a general framework for understanding bird-habitat relationships by reviewing the literature on patterns of bird community attributes, especially bird species richness. It is found that multiple factors operating over a range of spatial and temporal scales are involved in the explanation of patterns in bird distribution and abundance. The habitat templet[2] theory is found to be a useful conceptual framework for understanding bird-habitat relationships, particularly in successional environments.

In Chapter 4, the theoretical and empirical literature on bird response to successional environments is reviewed with reference to the habitat templet theory. Current theory on vegetation succession and its application to post-mining environments is outlined. A framework of the factors affecting the effectiveness of rehabilitation as habitat is described and previous research on bird assemblages in post-mining rehabilitation is reviewed focussing specifically on post bauxite mining in the Australian context. Methodological, definitional and interpretational problems in the literature are discussed and some of the limitations of previous research are identified.

The two literature review chapters and the background information on the physical environment inform the research design and sampling strategies used in this research, which are outlined in Chapter 5. The chapter begins by outlining some assumptions of the research design. Site selection and sampling procedures, and analytical techniques are described. Chapter 5 concludes by highlighting the original methodological approach to investigating faunal response to rehabilitation made by this research.

The results of this study are presented in two chapters. The findings in relation to birds are presented in Chapter 6 with reference to the first three key research questions. To begin with the survey completeness with respect to the bird assemblages of the *Eucalyptus tetrodonta* open forest sites is assessed. Summary values of the bird assemblages in post-mining rehabilitation sites are compared with reference values obtained from pre-mining native forest reference sites and post-mining landscape analogue land units. Bird assemblages are then compared at the level of individual species and foraging groups, and temporal patterns are explored. The results of analyses that explore relationships between the site detection rates of individual bird species and site attributes are presented.

In Chapter 7 the findings of this research in relation to the developing ecosystem in post-mining rehabilitation are presented with reference to the final key research question. To begin with, reference values for landscape function, vegetation composition and structure in the native forest reference land units are described. These reference values are compared with values for landscape function, vegetation composition and structure in post-mining rehabilitation. Results pertinent to interpreting the pathway of vegetation development in mine rehabilitation are presented. Each of the results chapters concludes with a brief summary of results.

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[2] templet is an alternative spelling for template.

The findings of this study are discussed in chapter 8. The results are interpreted with reference to the predictions of theoretical frameworks. In particular, the bird results are compared with the predicted patterns of faunal succession that arise from the habitat templet theory. The results of this study are also compared with research in other successional environments, particularly in post-mining rehabilitation. Chapter 8 concludes with a synthesis and discussion of the implications of the findings of this study for the native forest birds of the Weipa bauxite plateau.

In the concluding chapter, the main findings of this study are summarised with respect to the four key research questions posed. Conclusions are then presented in relation to the overall research question. The management and policy implications of the conclusions of this study are discussed. Finally, the limitations of this study and possible directions for future research are outlined.

Chapter 2: The study area

This chapter sets the environmental context and informs the research design of this study. It describes the location of the study area and the geology, climate, hydrology and vegetation of the Weipa bauxite plateau. The impacts of the fire regime on vegetation in the monsoonal eucalypt formations are discussed. Birds of the *Eucalyptus tetrodonta* open forest of Cape York Peninsula and their conservation status are described. The chapter concludes with a brief outline of the bauxite mining process at Weipa.

2.1 Location

The study area is the area of the Weipa bauxite plateau that is centred on the Weipa peninsula on Cape York Peninsula, Queensland (Figure 2.1). The Weipa bauxite plateau extends with minor erosional gaps for 350 km along the west coast of Cape York Peninsula, from Vrilya Point in the north to the Holroyd River in the south (McConachie *et al.*). It extends up to 65 km inland (Smart, 1977) and covers an area of approximately 11 000 km^2 (1.1 million ha).

Figure 2.1 Location of study area

Source: (Geoscience Australia, 2000)

2.2 Geology

The Weipa bauxite plateau is a dissected plateau remnant capped by aluminous (bauxitic) laterite. The western edge of the plateau forms striking red cliffs 5 to 15 m high along the coast of the Gulf of Carpentaria. The plateau rises to an elevation of 80 m along the inland eastern margin. The bauxite forms a layer approximately three metres thick, but thickening to 12 m locally (Taylor and Eggleton, 2004). It is underlain by a lateritic profile and is overlain by soils up to 1 m thick (Table 2.1and Figure 2.2). Elevation varies locally. Relief generally results from erosion, but there is also evidence of warping. Bauxite grades decrease with distance from the coast, correlated with increasing relief and decreasing rainfall (Smart, 1977).

Table 2.1 Description of the bauxite profile[1]

Thickness (m)	Description
0.5 - 1	**Topsoil**. Grey-brown sandy loam. Very thin A horizon. Soils are predominantly red or yellow earths with minor hydrosols in wet or seasonally wet locations. Podzols occur in association with sinkholes. Undulations in the topography lead to different soil moisture regimes and hydration states. The higher soils are red whereas those of the lower parts of the landscape are yellow.
0 - 5	**Regolithic sediment.** 'Redsoil' or re-worked bauxite. Occurs in localised elongate lenses < 5 m thick and up to 2 km long in topographic lows. Has a high content of fine material.
0 - 12	**Pisolitic bauxite**. This is the ore layer. It consists of approximately 70% pisoliths and 30% fines. It covers most of the western Weipa bauxite plateau. The upper part of the bauxite layer is primarily boehmite, a monohydrate of alumina. The lower part of the bauxite layer is primarily gibbsite, a trihydrate of alumina. There may be a sharp or gradual boundary to the underlying lateritic layer. The underlying laterite is the lower cut-off for mining.
0 - 5	**Lateritic duricrust.** An undulating irregular layer of ironstone (ferricrete) or ferruginous nodules in a matrix of pisoliths and sandy clay. Indurated mainly near watercourses.
5 - 15	**Mottled Zone.** An iron-oxide mottled layer consisting of 50 - 80% kaolinite and 20 - 30% hematite or goethite. Occurs everywhere. In places the top of this layer is sufficiently indurated with hematite or goethite to form a ferricrete. The uppermost mottled zone and ferricrete are the transition zone of Taylor and Eggleton (2008). The **transition zone** is about 1m thick and occurs at the top of the wet-season watertable. The chemical and mineralogical changes that produce bauxite pisoliths occur in the transition zone.
10 - 20	**Plasmic Zone.** Gradational from the mottled zone, up to 90% kaolinite. Occurs everywhere.
25 - 50	**Saprolite**

1. Redrawn after (Eggleton *et al.*, 2008; Smart, 1977)

Figure 2.2 Exposure of the bauxite profile

An exposure of the bauxite profile at Hey Point on the Embley River showing *Eucalyptus tetrodonta* open forest growing on a thin layer of topsoil overlying the richly coloured orange-red pisolithic bauxite layer and a pale mottled zone.

The Weipa bauxite consists of a loose accumulation of pisoliths, (i.e., spherical, pea-sized balls of ferricrete) (Smart, 1977), in little or no matrix. The loose pisolithic form of bauxite over such a large area is unique to the Weipa deposit (Taylor *et al.*, 2008; Chapman, 1963). Until recently, the Weipa bauxite was thought to have formed *in situ* by weathering (McConachie *et al.*, ; Chapman, 1963). Recent research has led to the view that pisoliths formed upslope within the substrate, or precursors of current substrates, and have been transported to their present position. Continual re-working of regolith containing pisoliths, earthy sediments and red soils has resulted in the accumulation of the bauxite layer. Most of the fines were winnowed as the regolith was re-worked leaving only the pisoliths and their cores (Taylor and Eggleton, 2004). *In situ* formation of bauxite in the transition zone of the regolith (see Table 2.1) is believed to be an ongoing contemporary process (Taylor and Eggleton, 2008).

The interactions of the seasonal watertable, termites, *Eucalyptus tetrodonta,* and fire with the regolith are thought to play a key role in the ongoing formation of the bauxite layer (Taylor and Eggleton, 2008). Analysis of termite mound composition has shown that the termites bring a considerable amount of kaolinite to the surface from as deep as 25 m in the regolith profile. Kaolinite weathers to form gibbsite, a species of alumina. It has been estimated that the whole bauxite layer and all clay in the soil of the bauxite profile may be a consequence of termite activity. The roots of *Eucalyptus tetrodonta* penetrate the bauxite layer at least to the top of the mottled zone (see Table 2.1) and commonly some distance further. The combined effects of treefall and fire are thought to be responsible for the mix of alumina species throughout the top of the soil profile (Eggleton and Taylor, 2008). Deep tree root channels provide tunnels for the vertical movement of material. Production of organic chemicals by tree roots is also thought to cause seasonal dissolution and precipitation of hematite and goethite (Taylor and Eggleton, 2008), which form ironstone.

2.3 Climate

The study area is in the megatherm seasonal bioclimate of Nix (1982), more generally referred to as the monsoonal or wet-dry tropics. The climate is characterised by a pattern of uniform high temperature and solar radiation. Rainfall is strongly seasonal (Suppiah, 1992) with a coefficient of variation for monthly rainfall of 100% (Nix, 1974; Nix and Kalma, 1972). Seasonality of the rainfall is due to seasonal movement of the equatorial trough[3] as it tracks the sun. From April to November, the region is under the influence of a continental belt of high pressure bringing dry south-easterly trade-winds overland. Southward shift of the high pressure belt from December to March causes southerly intrusion of the equatorial trough, which brings an inflow of moist, unstable air (Keast, 1981a; Bowman, 2002).

Although the seasonal pattern is predictable in a general sense, there is considerable inter-annual and intra-seasonal variability in the wet season. Compared to the wet-dry tropics globally, rainfall variability is high, but similar to temperate non-arid Australia (Ridpath, 1985). The timing of onset and retreat, and the duration and magnitude of the wet season are highly variable from year to year (Holland, 1986; Ridpath, 1985; Drosdowsky, 1996). Inter-annual variability in the timing of onset and intensity of the monsoon is associated with ENSO (Drosdowsky, 1996; Holland, 1986). During the negative phase of ENSO the amount of wet season rainfall tends to be reduced and there is reduced tropical cyclone activity (Bowman, 2002). Alternating wet (i.e., monsoonal), dry, and rainy periods (i.e., pre- and post monsoonal transitions) of varying duration occur within a wet season (Cook and Heerdegen, 2001). Periods of sustained rainfall are associated with macro-scale rain-producing systems such as tropical cyclones and monsoonal depressions. There are usually three to four such sequences lasting up to 10 days in a wet season. Rainy periods of occasional showers are associated with local

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[3] The equatorial trough is the belt of low pressure lying between the trade winds of the two hemispheres. It differs from the intertropical convergence zone (ITCZ) in the degree of convergence of trade winds although the equatorial trough and the ITCZ have the same general causes. The equatorial trough can be thought of as the meteorological equator separating the circulation systems of the two hemispheres (Colls and Whitaker, 1990).

convection and with the sea breeze convergence (McDonald and McAlpine, 1991). Tropical cyclones are recurrent although unpredictable events (Figure 2.3).

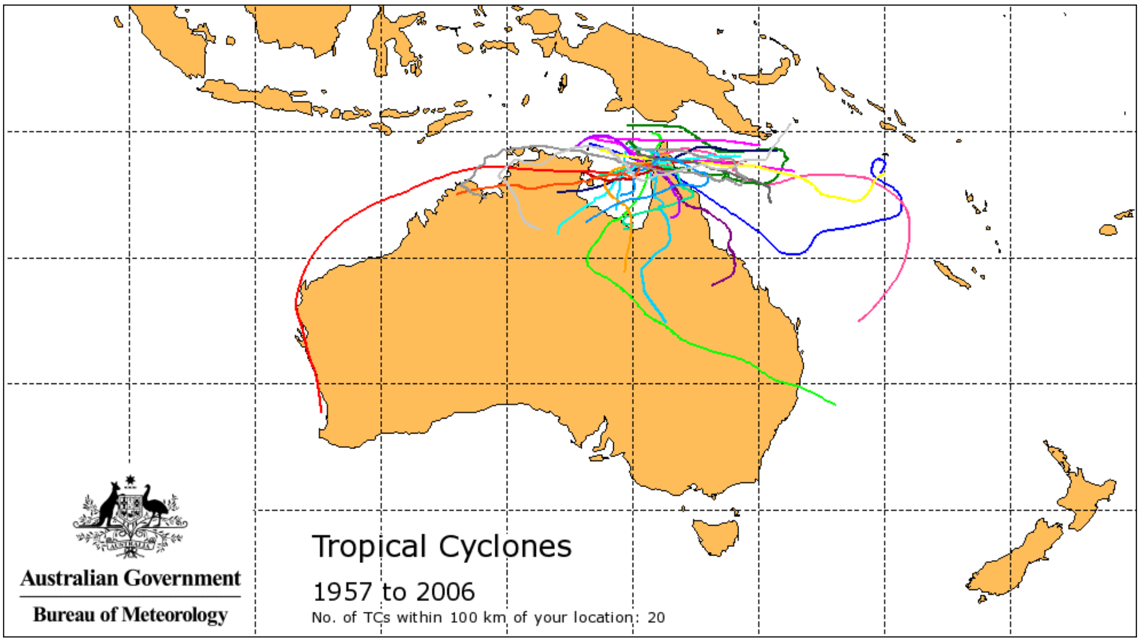

Figure 2.3 Cyclone tracks near Weipa over the last 50 years

Tropical cyclone tracks within 100 km of Weipa between 1957 and 2006 (Source: Bureau of Meteorology)

At Weipa, annual mean rainfall over 94 years of records is 1765 mm. It ranges from a minimum of 991 mm recorded in 1986 to a maximum of 2690 mm recorded in 1996 (Source: Bureau of Meteorology). Most of the rain falls between November and April (Figure 2.4). Solar radiation is high for most of the year but peaks in October and November prior to the onset of the wet season. The daily mean minimum temperature is 21.8° C and the daily mean maximum temperature is 32.3° C (Figure 2.5).

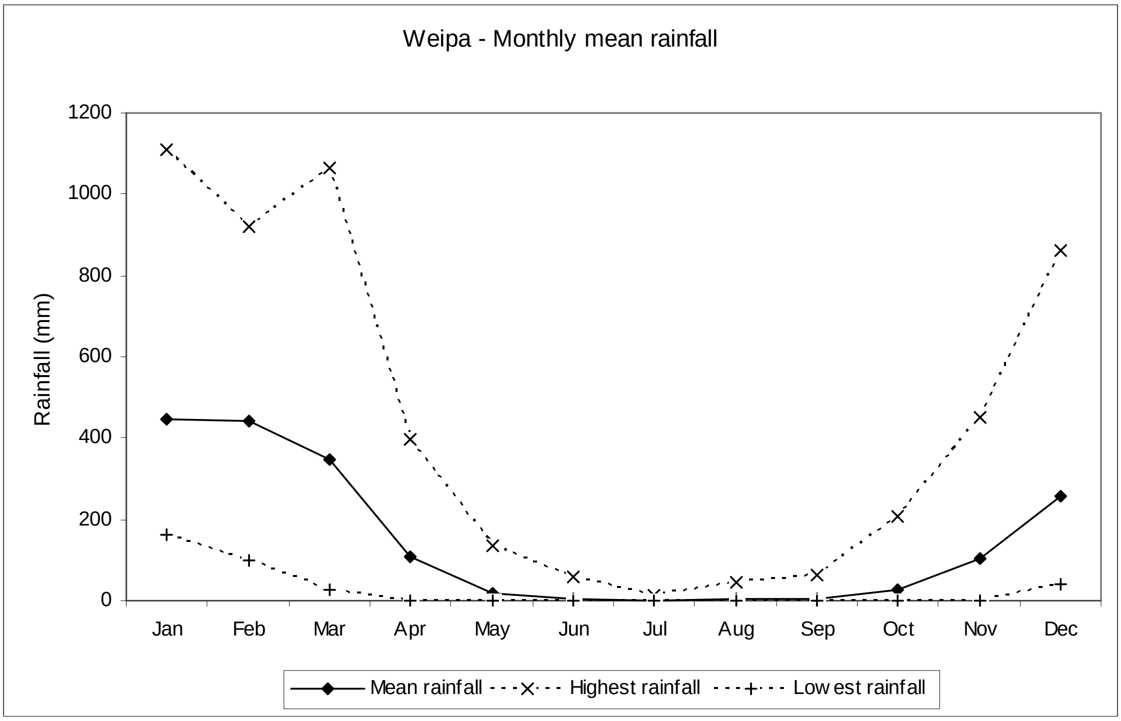

Figure 2.4 Monthly mean rainfall at Weipa

(Source: Bureau of Meteorology, Weipa Eastern Avenue data 1918 - 2008)

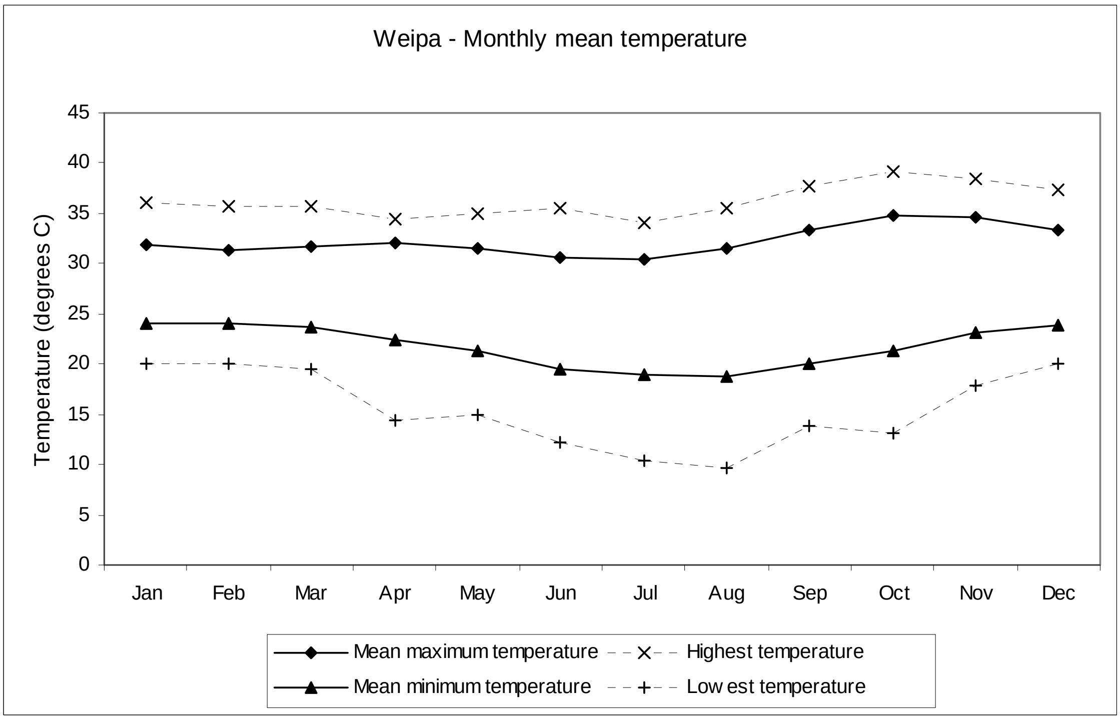

Figure 2.5 Monthly mean temperature at Weipa

(Source: Bureau of Meteorology, Weipa Eastern Avenue, 1959 - 1994)

2.4 Hydrology

The Weipa peninsula hydrology has been well studied, but not the areas of the Weipa bauxite plateau that occur north of the Mission River or south of the Embley River. On the Weipa peninsula, where most of the study sites are located, groundwater hydrology is strongly seasonal (Chapman, 1963). Rain drains vertically into an unconfined shallow aquifer that covers an area of approximately 260 sq km (1971). Early rains wet the porous upper strata. When these are wet, rain infiltrates rapidly through the upper strata including the bauxite layer, directly recharging the aquifer so that the watertable begins to rise. By the end of the wet season the watertable rises into the bauxite and ironstone layers. As it is dependent on rainfall for recharge, the aquifer is susceptible to drought conditions (Smart, 1977).

The watertable rises up to 8 m during the wet season, and falls again through the dry season with the shape of the hydrograph varying spatially across the Weipa peninsula (Watermark Numerical Computing). The height of the watertable above sea level generally increases with distance inland. Close to the coast, at the western end of the Weipa peninsula, the watertable is below sea level during the dry season minimum and up to 5 m above sea level during the wet season maximum. Inland the dry season minimum is up to 22 m above sea level with a wet season maximum height of 30 m.

During the wet season, rainfall greatly exceeds evaporation. Normally this would result in a well developed surface drainage network. Due to the high permeability of the soil, however, surface drainage features are poorly developed and there is very little surface run off in spite of high intensity rainfall events (Chapman, 1963). By the end of the wet season there is considerable flow through the permeable bauxitic, ironstone and clay layers. Flow through the aquifer is generally westward. The aquifer discharges into streams, which arise around the periphery of the bauxite plateau, and into the sea in the western part of the Weipa peninsula where the aquifer is below sea level (1971).

2.5 Vegetation

Land units based on terrain, soils, and plant composition and structure have been mapped for the Weipa area at a scale of 1:24 000 by Comalco (Gunness *et al.*, 1987). Four land units have been identified that occur on the bauxite plateau proper, three on the eroding slopes of the plateau, and ten on gentle slopes and drainage lines of the bauxite plateau. A further 23 land units are associated with drainage basins, sandplains and dunefields, marine terraces and saline flats in the Weipa area (Appendix 1). The most common land unit on the bauxite plateau proper and the main source of commercial bauxite, is land unit 2B, Darwin Stringybark (*Eucalyptus tetrodonta*) tall woodland / open forest on red earth soils. To improve readability, this is hereafter referred to as *Eucalyptus tetrodonta* open forest.

Eucalyptus tetrodonta is one of the most widely distributed species in northern Australia. It occurs on a wide range of soil types where annual mean rainfall exceeds 750 mm. Woodlands dominated by *E. tetrodonta* are the most widespread communities on Cape York Peninsula (Pedley and Isbell, 1971) and their distribution is used to define the southern boundary of the Cape York Peninsula bioregion (Frith, 1995). The *E. tetrodonta* open forest that occurs on the bauxite plateau is a unique regional ecosystem (Regional Ecosystem 3.5.2)[4] that is closely correlated with the bauxite plateau (Sattler and Williams, 1999) (Figure 2.6). It represents the maximum structural development of *E. tetrodonta* throughout its range in tropical Australia with trees often exceeding 30 m (Environmental Protection Agency, 2005). It is also floristically distinct from *Eucalyptus* formations in the Northern Territory and Kimberley region of Western Australia (Specht *et al.*, 1977).

Structurally, the *Eucalyptus tetrodonta* formations of the bauxite plateau vary from an open forest, to a tall woodland, and to a woodland (Environmental Protection Agency, 2005). Sub-canopy and shrub layers vary from sparse to very sparse. The ground layer is sparse to mid-dense. Variation in structure and composition is primarily related to soil moisture controlled by drainage conditions (Pedley and Isbell, 1971; Bowman and Minchin, 1987; Bowman *et al.*, 1988). Access to soil water modifies the severity and duration of the seasonal drought allowing the development of an open forest (Nix, 1982; Gillison, 1994; Specht *et al.*, 1977; Dunlop and Webb, 1991).

Floristically, *Eucalyptus tetrodonta* dominates forming a distinct but discontinuous canopy. Melville Island Bloodwood (*Corymbia nesophila*) is present as a sub-dominant and locally co-dominant canopy species (Environmental Protection Agency, 2005). Cooktown Ironwood (*Erythrophleum chlorostachys)* may be present as a tall sub-canopy layer. The shrub layer is dominated by *Acacia* and *Eucalyptus* species but includes *Grevillea glauca*, *Grevillea parallela, Alphitonia excelsa*, *Parinari nonda*, *Planchonia careya, Petalostigma pubescens, Brachychiton* spp*., Owenia vernicosa, Canarium australianum, Lophostemon suaveolens, Croton arnhemicus, Ficus opposita* and *Livistona muelleri* (Environmental Protection Agency, 2005). The ground layer is dominated by the perennial grasses, *Sorghum plumosum var. plumosum*, *Heteropogon triticeus*, *Alloteropsis semialata* and *Eulalia mackinlayi* (Environmental Protection Agency, 2005). Variation in the shrub, grass and herb layer is thought to be due to differences in soil texture (Specht *et al.*, 1977).

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[4] Regional Ecosystem numbering comes from the Queensland Environmental Protection Agency mapping of regional ecosystems throughout the state. The number 3 refers to the Cape York Peninsula bioregion; 5 refers to the land zone; and 2 is the final identifier that refers to the regional ecosystem (Sattler and Williams, 1999).

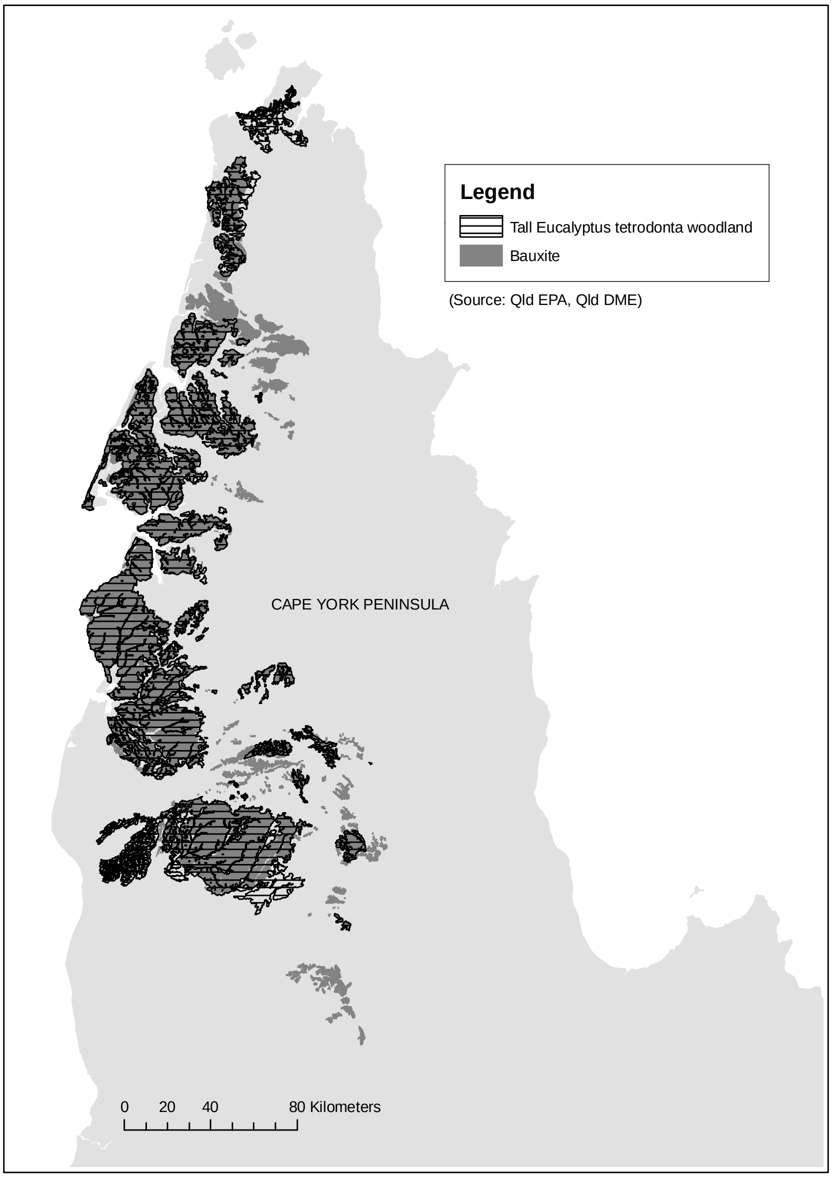

Figure 2.6 Distribution of *Eucalyptus tetrodonta* open forest (R.E. 3.5.2)

(Source: Qld EPA, Regional Ecosystems, 2003)

2.6 Fire in the tropical savannas

In combination with climate, fire has been a shaping force on the environment of northern Australia for millennia. Dry electrical storms are a natural source of ignition of fires in the late dry to early wet season. Prior to human occupation, fires ignited by lightning possibly occurred more frequently because of higher available fuel loads (Stocker, 1966). It is considered likely that lightning ignited fires were extensive although infrequent in the tropical savannas (Crowley, 1995).

The fire regime[5] in Australia's tropical savannas is very different from those in temperate Australia (Bowman, 1986). The leaves of tropical eucalypts are not highly flammable due to a lower volatile leaf oil content (Lacey *et al.*) and vegetation based fuel does not accumulate due to frequent dry season fire and high rates of litter breakdown during the wet season. Consequently, fires are generally restricted to the grassy ground layer and do not carry into the canopy layer although they may scorch it depending on the fire intensity (Lonsdale and Braithwaite, 1991).

In the *Eucalyptus tetrodonta* tall woodlands on Cape York Peninsula perennial grasses grow tall every wet season, and produce fuel loads that can carry grass fires at least every second year[6]. The fire season commences as grasses cure following the wet season. The area burnt each month increases as the dry season progresses, diminishes at the onset of the wet season and ceases between December and March when the country is usually too wet to burn (Felderhof and Gillieson, 2006). Under current management, fires (mostly grass fires) burn large areas of western Cape York Peninsula annually (Figure 2.7). A recent five year study found that on average 57% of the bioregion burns every year. Over five years, most of western Cape York Peninsula burnt more than once, much of the landscape burnt three, four or five times and only 5% of the study area remained unburnt (Felderhof and Gillieson, 2006).

Although much of the landscape burns annually in grass fires there is spatial variation in the effects of single fire events (Lacey *et al.*). Fire intensity varies with time of year and (Crowley, 1995; Lonsdale and Braithwaite, 1991) when combined with variation in vegetation, topography and climatic conditions at the time of fire results in patchy mosaics of burn intensity. Research from the Northern Territory suggests that the scale of fire mosaic patterns on the landscape has changed from pre-European fire patterns. No specific studies of altered fire regimes have been conducted for the Weipa bauxite plateau. Although the presumed extent of change has not been quantified, it is thought that relative to current burning practices, Aboriginal burning in the tropical savannas of northern Australia generally created a more finely scaled mosaic of burnt and unburnt areas. Aboriginal fires are also assumed to have been

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[5] The fire regime consists of fire intensity, fire frequency and season of burn (Gill, 1975).

[6] The Weipa bauxite plateau is in Gill's fuel type region 'C' where there is sufficient fuel for fires in most seasons (Gill, 1975).

relatively lower in intensity. Departure from Aboriginal fire regimes may have contributed to a loss of habitat diversity and the decline of several species of birds and mammals that were once common in parts of northern Australia (Bowman *et al.*, 2004; Russell-Smith *et al.*, 1997; Garnett and Crowley, 2000b).

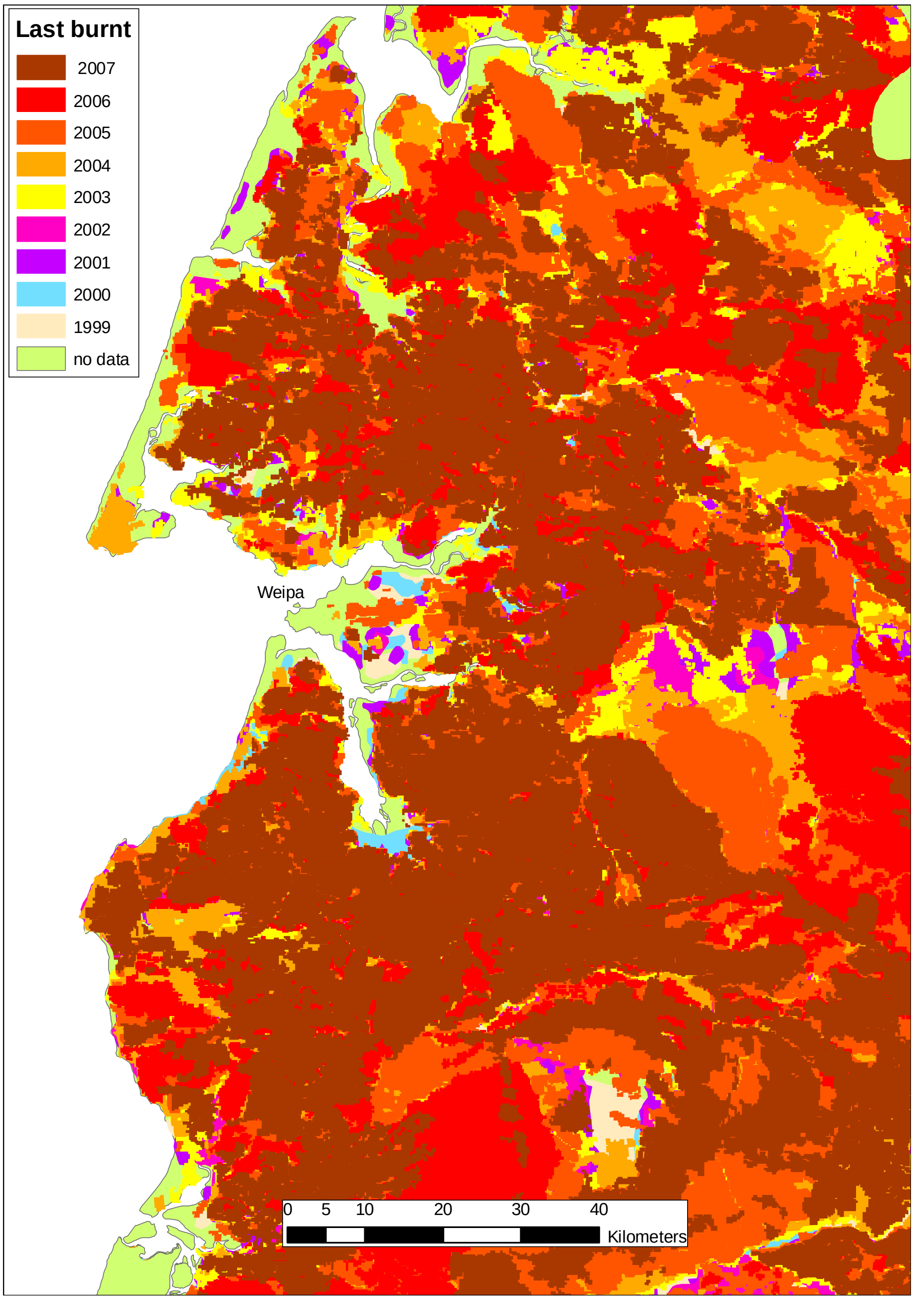

Figure 2.7 Fire scar map of the Weipa locality

(Source: Cape York Peninsula Sustainable Fire Project)

2.6.1 Effects of fire on vegetation in the tropical savannas

The tropical eucalypts have evolved different regeneration strategies from the temperate eucalypts, possibly in response to severe annual drought (Bowman, 1986). In the tropical savannas, *Eucalyptus* spp., and other woody perennial species, which are well established, have traits that enable them to survive frequent, low-intensity fire[7]. A Northern Territory study found that approximately 1% of canopy trees in the eucalypt dominated savannas died each year. Ten percent of these canopy deaths were due to lightning strike, 10% were due to windthrow and another 10% died standing as a result of termite attack. All of the remaining canopy deaths were due to the interaction between termite hollows and fire. After several fires, termite hollowed trees burn through on one side, where a fire scar has formed and the tree eventually falls (Braithwaite and Estbergs unpublished data cited in (Lonsdale and Braithwaite, 1991)).

The current composition and structure of the vegetation communities is stable under frequent, low-intensity burning. However, single fire events, especially hot, late dry season fires can destroy woody vegetation. A study of vegetation response to a late dry season fire in the Northern Territory found tree and shrub mortality of 14.3% within 6 months of a late dry season fire. Although no species were lost, species differed in their mortality rates leading to marked changes in relative species abundance. Both the smallest and the largest plants were most vulnerable to fire (Lonsdale and Braithwaite, 1991). Due to high fire frequency, however, fire intervals are not related to stand structure as they are in temperate eucalypt forests (Bowman, 1986). Short term structural and floristic changes in response to fire are often subtle (Lonsdale and Braithwaite, 1991).

Bowman and Minchin (1987) found complex interrelationships between vegetation, edaphic variation and fire regime in the tropical savannas. They proposed that the development of a dense understorey in *Eucalyptus tetrodonta* forest following artificial exclusion of fire could be explained by the impact of soil moisture supply on the growth rates of woody species. Under frequent fire, woody species rarely grow fast enough that they are able to escape damage to their above-ground biomass (Bowman and Minchin, 1987). Fensham (1990) compared sites with different fire histories and found that the basal area of the dominant species, *Eucalyptus tetrodonta* and *Erythrophleum chlorostachys,* were not significantly affected by fire history although all woody species were significantly more abundant as saplings in unburnt sites. He concluded that the interaction of edaphic conditions with fire frequency resulted in different outcomes for different species. Change in structure and overstorey composition in response to fire was more likely where site physical conditions imposed stress (Fensham, 1990).

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[7] Three broad categories of plant response to fire are recognised: (1) fire ephemerals, (2) obligate seeders, and (3) sprouters although a species may respond differently to two fires at the same site depending on the fire history and the condition of the individual at the time of burning. Sprouters possess traits that enable mature plants to recover vegetatively from 100% leaf scorch. In contrast, populations of non-sprouters (obligate seeders) recover from seed banks stored either in the soil or on the plant (Whelan, 1995).

Although the tropical eucalypt formations are dominated by sprouter species with traits that enable them to survive frequent, low-intensity fire, there is concern that lack of seedling recruitment due to frequent burning over long time frames may have a gradual but significant effect on the population structure of Australia's tropical savannas (Setterfield, 2002). Highly frequent fires may deplete nitrogen, cause a recruitment bottleneck in savanna trees and lead to declines in woody obligate-seeder shrubs (Williams *et al.*, 2003b). The relative abundance of species that reproduce vegetatively is expected to increase in response to frequent burning (Setterfield, 2002). Fire free periods of between three and five years may be required to ensure seedling recruitment (Williams *et al.*, 2003b) and allow for the growth of saplings to greater than two m (Williams *et al.*, 2003a). The longevity of fire tolerant canopy trees may conceal the consequences of seedling recruitment failure for some time.

Recently, the potentially wide scale impact of windthrow on stand structure in the monsoonal eucalypt woodlands has been realised. Where Tropical Cyclone Monica made landfall in April 2006, as many as 80% of trees were windthrown and trees were windthrown up to 130 km from landfall (Cook and Goyens, 2008). Previously, the bimodal size structure of woody vegetation in the tropical savannas has been attributed to lack of recruitment to the canopy as a result of grazing and frequent fire. Observations from TC Monica, however, indicate that wind is an important disturbance agent with the estimated one in 100 year event resulting in a 42% decrease in basal area (Cook and Goyens, 2008). Tropical cyclones have occurred frequently within the study area over the last 50 years (Figure 2.3). No events of similar intensity to TC Monica have been recorded to date on the Weipa bauxite plateau. However, stands of *Eucalyptus tetrodonta* with even stem sizes can be interpreted as even-aged stands that have developed following vegetation destruction due to cyclones (J. Clarkson, personal communication, 2008).

2.7 Birds of the *Eucalyptus tetrodonta* open forest

Cape York Peninsula has substantially higher bird species richness than the Top End of the Northern Territory and the Kimberley region of Western Australia. Almost half of all Australian bird species have been recorded on Cape York Peninsula (Frith, 1995). Of the 366 bird species recorded only two are endemic to Cape York Peninsula, white-streaked honeyeater *Trichodere cockerelli* [8], and golden-shouldered parrot *Psephotus chrysopterygius*. A further 18 species are restricted to Cape York Peninsula within Australia but are shared with Papua New Guinea (Frith, 1995). The high regional bird diversity of Cape York Peninsula compared to the Top End and the Kimberley regions is due to the high diversity of habitats and to the presence of

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[8] Bird nomenclature follows Christidis, L. and Boles, W. (2008*) Systematics and taxonomy of Australian birds,* CSIRO Publishing, Collingwood.

additional biogeographic elements, the Irian and Tumbunan elements[9], in the closed forests of Cape York Peninsula (Morton and Brennan, 1991).

The bird assemblages of the eucalypt formations on Cape York Peninsula are basically Torresian (Winter and Atherton, 1985). A gradient of descending numbers of predominantly Irian bird species and increasing numbers of Eyrean species runs from the northeast to the southwest of the region. The palm cockatoo *Probosciger aterrimus* is the sole representative of the Irian element within the *E. tetrodonta* tall woodland bird assemblages. Eyrean elements within the tall *E. tetrodonta* woodland bird assemblages include the emu *Dromaius novaehollandiae*, and the white-winged triller *Lalage sueurii*. Approximately 24% of the bird species that occur in the eucalypt forests and woodlands of northern Australia are confined to the Torresian biogeographic province.

The *Eucalyptus tetrodonta* open forest of the Weipa bauxite plateau is habitat for widespread and common bird species as well as species that are rare or have restricted distributions. Surveys of vertebrate fauna in a range of habitats in the Weipa region found that the *E. tetrodonta* open forest had the highest species richness for birds, reptiles, frogs and mammals and that a number of species occurred exclusively in the open forest (Cameron and Cogger, 1992; Winter and Atherton, 1985). A study of habitat preferences at Weipa, found that woodland habitats were as species rich as riparian habitats (Thomas, 2004). Woodland habitats had a unique composition of terrestrial vertebrate fauna and included fauna that is either exclusive to or occurs predominantly in woodland habitats.

Combining the records of a number of bird observers 148 bird species were recorded in eucalypt woodlands and open forests of Cape York Peninsula over the 20th century (Appendix 2). Of these, 22% (33/148) are of conservation concern and two more are protected by international treaties (Appendix 3). Many of the species of conservation concern are widely distributed and threatened primarily as a result of habitat loss in other parts of their ranges. A number are shared with temperate eucalyptus woodlands, which are the most threatened ecosystem in Australia and contain one of the highest concentrations of extinct and threatened birds of any region or habitat in Australia (Olsen *et al.*, 2006; Robinson and Traill, 1996).

Habitat loss and fragmentation due to land clearing is the major threatening process contributing to the decline of 11 bird species that have been recorded in the *Eucalyptus* woodlands and open forests of Cape York Peninsula including: square-tailed kite *Lophoictinia isura,* grey goshawk *Accipiter novaehollandiae,* red goshawk *Erythrotriorchis radiatus,* barking owl *Ninox connivens,* masked owl *Tyto novaehollandiae,* red-tailed black cockatoo

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[9] Five broad faunal geographic regions are recognised in Australia. The names of the faunal regions are also applied to the mobile fauna to reflect the evolutionary origins of the extant fauna. The avifaunal zones are broadly characterised as the southern eucalypt (Bassian), northern eucalypt (Torresian), arid (Eyrean), temperate montane rainforests of eastern Australia and highland New Guinea (Tumbunan), and lowland rainforests of northern Australia and New Guinea (Irian) (Schodde, 1982; Morton and Brennan, 1991; Keast, 1981b).

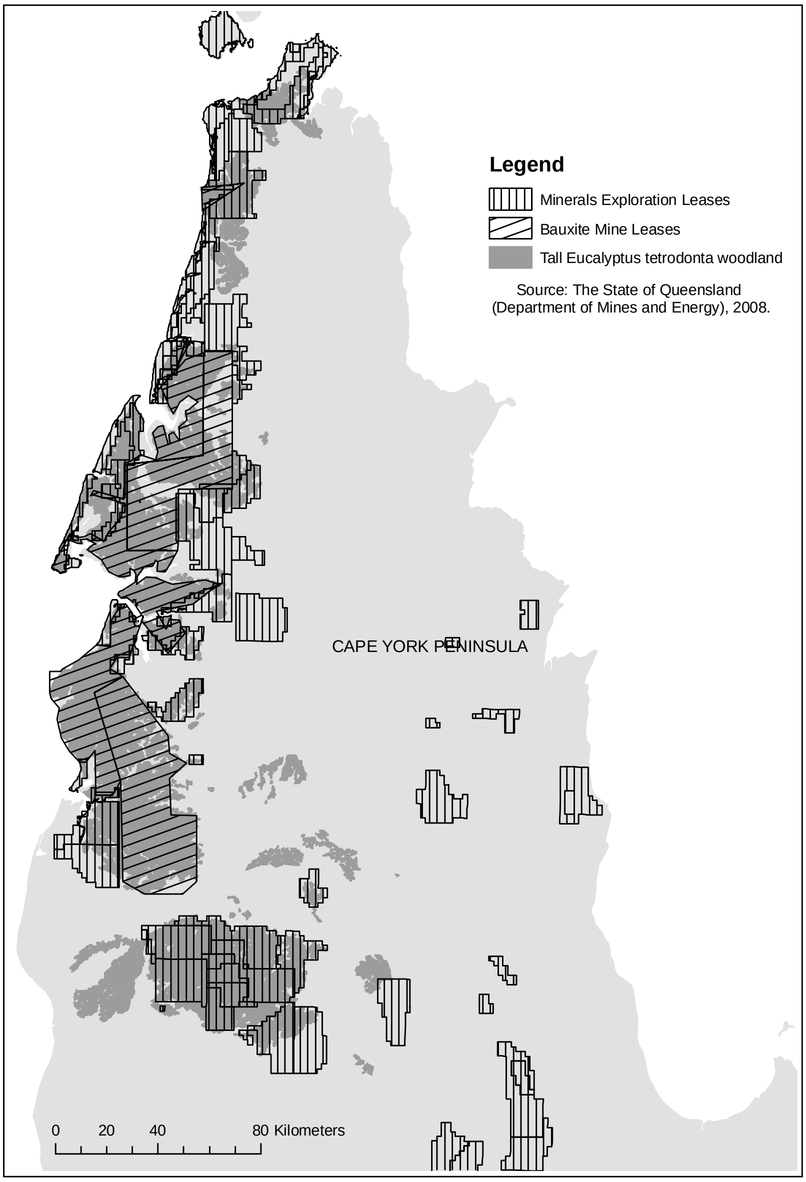
Calyptorhynchus banksii, brown treecreeper *Climacteris picumnus,* grey-crowned babbler *Pomatostomus temporalis,* bush stone-curlew *Burhinus grallarius,* and black-chinned honeyeater *Melithreptus gularis*. For these species, the loss of habitat has primarily occurred outside of Cape York Peninsula.

Another eight taxa are of conservation concern because of impacts on Cape York Peninsula: palm cockatoo, golden-shouldered parrot, the Cape York Peninsula sub-species of the brown treecreeper *Climacteris picumnus melanotus,* black-faced woodswallow *Artamus cinereus,* black-throated finch *Poephila cincta,* crimson finch *Neochmia phaeton,* star finch *Neochmia ruficauda,* and gouldian finch *Erythrura gouldiae*. The gouldian finch, crimson finch, black-throated finch, star finch and black-faced woodswallow were common in the Archer River area in the early 20th century (MacGillivray, 1918a; MacGillivray, 1918b). Since then, gouldian finches have disappeared from Cape York Peninsula and the other species have all declined in abundance or suffered range contractions (Garnett and Crowley, 1994). The Cape York Peninsula sub-species of the star finch *Neochmia ruficauda clarescens,* and the crimson finch *Neochmia phaeton evangelinae* are endangered (Garnett *et al.*, 2005; Todd *et al.*, 2003).

Altered fire regimes have been implicated in the contraction of the golden shouldered parrot, brown treecreeper, black-throated finch, star finch and gouldian finch. Widespread, late dry season fires are a likely cause of the contraction of the brown treecreeper, which appears to have limited ability to recolonise after fire (Garnett and Crowley, 1995). The impact of fire on tree hollows is of particular concern. Although fires contribute to hollow formation they can also destroy nests, young and nesting trees (Crowley, 1995). Destruction of hollow bearing trees by fire may also be a threat to palm cockatoos (Garnett and Crowley, 2000a; Murphy *et al.*, 2003; Murphy and Legge, 2007). More important in the long term, Murphy and Legge (2007) concluded, are the potential effects of increased frequent, widespread fires on stand structure and the maintenance of adequate numbers of large potentially hollow bearing trees.

2.8 Mining on the Weipa bauxite plateau

The Weipa bauxite is the world's largest proven bauxite resource. It covers an area of approximately 11 000 km^2 with an average thickness of 2.5 m (Taylor *et al.*, 2008). Current bauxite mine leases cover 47% of Regional Ecosystem 3.5.2, *Eucalyptus tetrodonta* open forest, on Cape York Peninsula. Together, bauxite mine leases and minerals exploration leases cover 82% of the total area of *E. tetrodonta* open forest on Cape York Peninsula (Figure 2.8).



Source: The State of Queensland (Department of Mines and Energy), 2008.

Figure 2.8 Distribution of bauxite mine leases and minerals exploration leases

Overlap of bauxite mine leases and minerals exploration leases with *Eucalyptus tetrodonta* open forest (R.E. 3.5.2).

As a progressive strip-mining operation the first step in the bauxite mining process is vegetation clearing (Figure 2.9). At current mining rates approximately 1,000 to 2,000 ha are cleared each year. The cleared vegetation is root-raked, windrowed and burnt, one or two years prior to mining. Sucker regrowth is then re-cleared and a layer of topsoil approximately 50 cm deep is stripped. Topsoil is either stockpiled or moved to an area where mining has been completed for use in rehabilitation. Topsoil stockpiles are sown with pasture grasses to control erosion (Comalco Mining and Refining, 2004).

Excavators or front end loaders then scrape up the loose pisolithic ore and dump it into haul trucks for removal. On average a layer of bauxite 2 to 3 m thick is removed down to the top of the ironstone layer. There is considerable variation in the depth of bauxite material that is removed, from as little as 50 cm up to 12 metres.

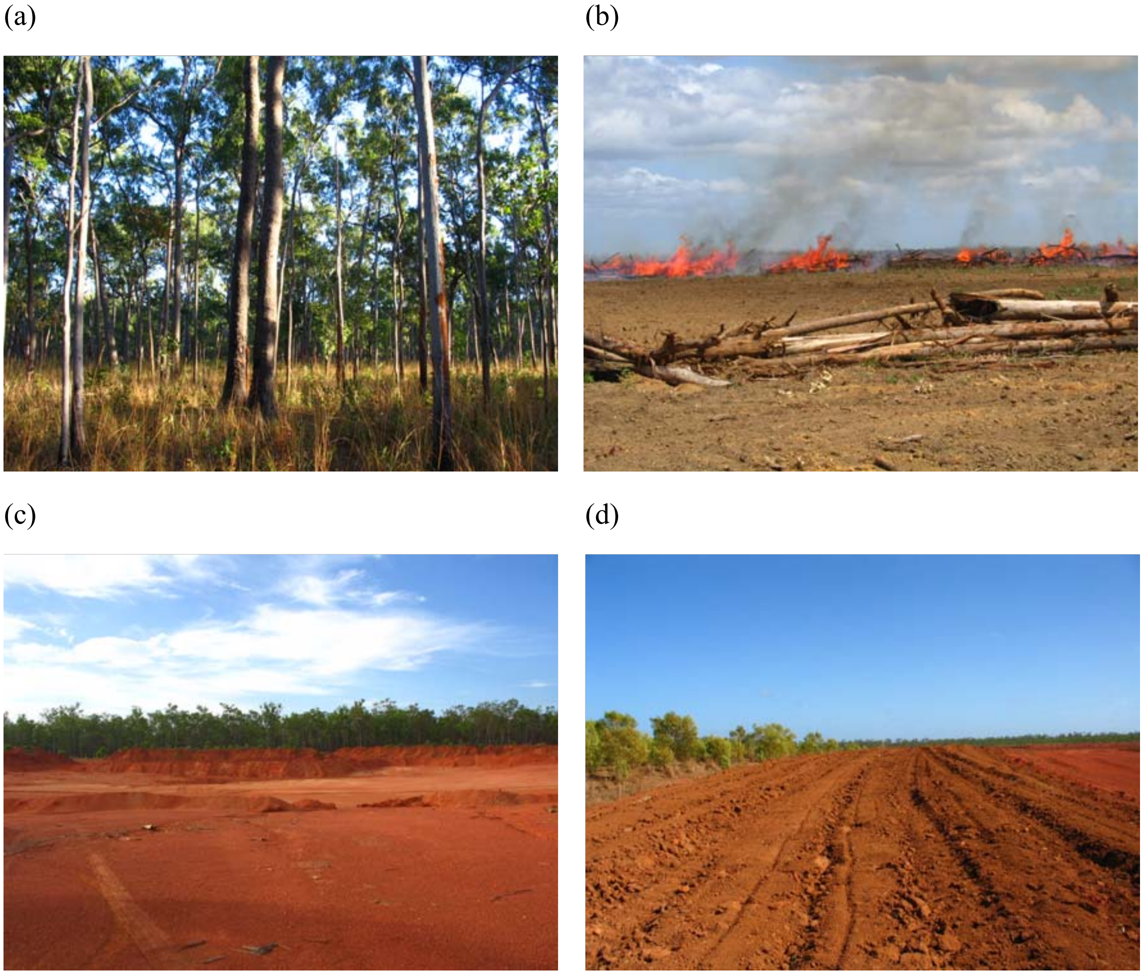


Figure 2.9 Mining sequence

(a) pre-mining *Eucalyptus tetrodonta* open forest (site A-1); (b) cleared vegetation is root-raked, windrowed and burnt; (c) following topsoil stripping the pisolithic bauxite is removed to the top of the underlying ironstone layer; (d) topsoil is replaced over the ironstone and is then deep-ripped, ploughed, fertilised and seeded.

Rio Tinto Alcan's current commitment is to reinstate 82% of the area mined to 'native ecosystems which are a mixture of local native tree and shrub species which create a habitat for fauna' (Comalco Mining and Refining, 2004). Prior to 2004, since at least 1981, the aim of their post-mining rehabilitation strategy was to establish 'self-sustaining maintenance-free vegetation comprising a variety of native plants which in turn support native fauna' (Reeders, 1985). For areas where the rehabilitation goal is to reinstate native ecosystems, either fresh or stockpiled topsoil is laid over the ironstone to a depth of approximately 30 cm. To control soil erosion and break up the compacted mine floor, sites are ripped along contours to a depth of 1.5 m (Comalco Mining and Refining, 2004). Following early wet season rains, sites are cultivated with a disc plough. Fertiliser and a seed mix are then applied with a spreader, giving seeds the opportunity to germinate and become established prior to the dry season. The seed mix is composed of a combination of species, some sourced locally. All of the species included in the seed mix are Australian native plant species but many are not part of the pre-mining vegetation community (Appendix 4). Fertiliser is applied at a rate of 100 kg / ha of superphosphate although application rates have varied through time (Appendix 7). In older mine rehabilitation sites, enrichment planting using tube-stock was also used. Enrichment planting was discontinued as a rehabilitation practice at Weipa in the early 1980's.

The mining process results in a landscape in which the post-mining land surface is on average 2 to 3 m lower than the original land surface and up to 12 m lower locally depending on variation in the thickness of the ore body. Consequently, the wet season watertable rises to the surface in some parts of the mine. There is also more topographic variation than in the pre-mining landscape, resulting from the undulating ironstone floor (Figure 2.10). The soil profile and drainage characteristics are also altered with potential effects on infiltration rates and water holding capacity (Reddell and Hopkins, 1994).

Observations of natural landscapes in the vicinity of the mine indicate that depth to the watertable and infiltration characteristics are fundamental in controlling the distribution of native plant communities. Consideration of these altered properties is therefore critical for understanding the ecology of developing vegetation in mine rehabilitation, defining appropriate rehabilitation targets and identifying appropriate reference ecosystems for comparison with mine rehabilitation. Reddell and Hopkins (1994) suggested that a range of alternative vegetation states will be required across the post-mining toposequence due to undulations in the mine floor and variable depth to watertables.

Figure 2.10 Undulations in the post-mining land surface

Rehabilitated area following wet season rains, showing undulations in the post-mining landscape and localised flooding.

2.9 Conclusion

This chapter has described the environmental context of this study including the dominant vegetation of the Weipa bauxite plateau. The level topography, highly seasonal climate, fire regime, termites and *Eucalyptus tetrodonta* interact to create an open forest that is remarkably homogeneous in structure and composition. There is a close association between the bauxite ore body and the *E. tetrodonta* open forest. Under the current disturbance regime, the vegetation on the Weipa bauxite plateau is characterised by a eucalypt dominated canopy, a well developed tall perennial tropical grass layer, and poorly developed shrub layers. The dominant plant species in the tropical *Eucalyptus* forests and woodlands are tolerant of the fire regime and provide stable habitat resources in spite of frequent widespread fire. No seral stages of different vegetation age are produced by single fire events although interactions between fire and soil moisture and/or fire history possibly account for variation that occurs in the vegetation composition and structure of the *E. tetrodonta* open forest. Severe wind associated with thunderstorms or cyclones may be more important than fire in determining stand structure. The dominant control on vegetation structure, however, is water availability (Williams *et al.*, 1996). The background information outlined in this chapter was drawn on in interpreting the vegetation and landscape in the study area and in designing a robust approach to answering the four broad research questions. The next chapter reviews the research literature on the environmental determinants of bird community attributes, focussing where information is available on the environmental determinants of bird assemblages in the monsoonal eucalypt formations of northern Australia.

Chapter 3: Bird-environment relationships

3.1 Introduction

This chapter aims to provide a framework for understanding bird-environment relationships that will inform the research design and interpretation of results. It does so by reviewing the literature on patterns of bird distributions, bird species richness and abundance. There is a great deal of debate in the literature on the causal mechanisms of species richness patterns. The aim here is not to attempt to resolve these debates but rather to provide a general framework for understanding patterns of bird distribution and abundance. Species richness is a coarse, aggregate measure made up of multiple species distributions that vary in their detail. However, it provides a useful starting point for understanding bird-environment relationships in general.

Essentially, there are two broad sets of processes by which faunal species richness can be increased: (i) by increasing the amount and/or variety of habitat resources; and (ii) by increasing the degree of partitioning of a given amount of habitat resources. Interacting biotic and abiotic factors are involved in both of these, some acting directly and others indirectly so that no general explanations emerge. Nevertheless, these categories provide a useful organising framework for this review of the theoretical and empirical literature in relation to bird species richness. Where available, empirical evidence for how they relate to bird assemblages in Australia's tropical *Eucalyptus* formations is also reviewed.

This review deliberately focuses on exogenous factors that are related to species richness. It is acknowledged that biotic interactions can be important in defining community structure, and can determine the difference between a species' theoretical, or fundamental niche, and its realised niche (Giller, 1984; Heglund, 2002). The focus of this review on environmental properties is justified, however, on theoretical and practical grounds. There are two justifications on theoretical grounds. First, regardless of species interactions, it is the environment that ultimately determines the amount and variety of habitat resources and therefore the number of potential niches available. Second, the niche-assembly perspective of communities, *sensu* Hubbell (2001), assumes that ecological communities are at or near equilibrium. There is abundant evidence, however, that many ecological communities are not in equilibrium and that non-equilibrium conditions reduce the effects of species interactions (Rohde, 2005). Focussing on understanding bird-environment relationships also appears to be the most practical step for informing the development of improved rehabilitation techniques.

3.2 The amount of habitat resources

There are two main ways that an increase in habitat resources can occur. First, the amount or volume of a given set of types of habitat resources can increase. Second, the variety, or number of types of habitat resources can increase.

Srivastava and Lawton (1998) outline four alternative explanations for how an increase in productivity of resources (i.e., amount of the resource base), could lead to an increase in faunal species richness. The first explanation (more individuals), proposes that productivity is limiting and more productive locations have more species because they contain more individuals. The number of species increases with productivity because populations with high abundance are less likely to go extinct. The second explanation (more specialisation), proposes that for a given variety of habitat resources there are more opportunities for specialists to occur in locations with high productivity because for each type of resource, the minimum amount required to support populations is provided. The third explanation (dynamic equilibrium), proposes that more productive locations have faster population recovery times from disturbance and consequently a higher equilibrium number of species. The fourth explanation (more trophic levels) proposes that more productive sites have smaller population sizes because of the addition of a higher trophic level that reduces competitive exclusion (Srivastava and Lawton, 1998). There is plenty of empirical evidence demonstrating correlation between productivity and species richness. However, multiple and interacting causal processes mean that quantifying the amount of variation that is specifically attributable to productivity remains complex.

3.2.1 Plant productivity, habitat resources and bird species richness

Gross primary productivity[10] is highly correlated with vegetation structure and with the availability of food resources for trophic groups along the food chain and is therefore a potentially important predictor of vegetation-based habitat resources (Berry *et al.*, 2007). Various surrogates of gross primary productivity have been found to be correlated with faunal species richness over a range of spatial scales. Species richness of a number of taxonomic groups is correlated with total net primary production (Wright, 1983), realised annual evapotranspiration (Currie and Paquin, 1987), annual potential evapotranspiration (Currie, 1991), and normalised difference vegetation index (NDVI) (Hurlbert and Haskell, 2003; Koh *et al.*, 2006).

The relationship between species richness and gross primary productivity however is not a simple one. First, under some conditions, increased productivity can lead to a reduction in diversity, known as the 'paradox of enrichment' (Huston, 1994). Second, the rate of

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[10] Gross primary productivity (GPP) is the total rate of photosynthesis per unit area per unit time of vegetation cover.

photosynthesis is controlled by the combined inputs of solar radiation, water, nutrients and carbon dioxide, which interact with landscape and the biota to produce complex spatial patterning. Third, the transformation of gross primary productivity into vegetation structure and food resources produces different outcomes in different ecosystems. Terrestrial ecosystems differ in the proportions of energy that are either lost in respiration or converted to biomass, in how carbohydrates are allocated to different plant parts, and in the time taken for conversion of new biomass into a range of food resources (Berry *et al.*, 2007). Finally, vegetation-based habitat resources are perceived differently by individual species.

A number of Australian researchers have found relationships between plant productivity and bird species richness. Nix (1982) showed that potential site productivity, estimated by climate and seasonality of plant response, determines the geographic distribution of taxonomic diversity at continental scales. A relationship between bird species richness and plant productivity in Australian *Eucalyptus* formations is supported by studies that have found correlations between bird species richness and foliage nutrient levels (Braithwaite *et al.*, 1989; Recher, 1985); tree basal area (Neave *et al.*, 1996); biomass (Milledge and Recher, 1985); arthropods and foliar nutrients (Recher *et al.*, 1996; Loyn, 1985b).

Bird movements confirm that plant productivity is a dynamic determinant of species richness (Hurlbert and Haskell, 2003). Seasonal changes in bird species richness and abundance in response to insect or carbohydrate availability have been demonstrated in a number of eucalypt habitats indicating that climate driven plant productivity is an important driver of bird distributions within eucalypt landscapes (Nix, 1974; Keast, 1967; Mac Nally and McGoldrick, 1997; Loyn, 1985b; Arnold, 2003; Farmer *et al.*, 2004; Woinarski *et al.*, 2000b; Franklin and Noske, 1999). Nix (1974) showed that seasonal movements of Australian birds and breeding activity are closely coupled to seasonal pulses in plant growth. He proposed a map of seasonal movements of Australian birds, in which dispersive movements are driven by plant productivity. Although the role of productivity has yet to be demonstrated for many individual species, analysis of intra-continental bird movements in Australia found evidence for large scale dispersive movements of bird species or populations that supports Nix's model of environmental controls (Griffioen and Clarke, 2002).

If high plant productivity *per se* was equally important to all species, then we might expect all species to occur in areas of high productivity. However, many species are specifically adapted to exploit environments of low productivity. Even so, productivity has been shown to be important across a range of gradients. Analysis of distributions of South African birds along productivity gradients (estimated from NDVI), shows that the probability of a species occurring in a given location is directly related to productivity (Storch *et al.*, 2007b). Rare species occurred more frequently in areas of high productivity.

How closely bird species richness or individual abundances are related to resource availability depends on the extent to which populations are resource limited. Depending on the environment and specific resource requirements, resources may often not be limiting (Wiens, 1989b). Absence of a direct relationship with plant productivity can be due to birds exploiting food resources that show a delayed response to plant productivity (Wiens, 1989b; Gilmore *et al.*, 2007). The abundance of an arid zone species, the Australian bustard (*Ardeotis australis*) for example, is positively correlated with above average rainfall in the previous year. Bustard abundance is negatively correlated with NDVI maxima, but positively associated with NDVI minima. This is interpreted as evidence that a minimum productivity threshold exists for this species. Apparently, favoured food resources, insects, small mammals and reptiles become more available to the bustards as grasses senesce (Ziembicki and Woinarski, 2007).

As Wiens (1989) noted, general environmental variation in factors such as rainfall offer no assurances that the important resources, as they are perceived by birds, vary in the same way (Wiens, 1989b). The time frame of species response to productivity varies from species to species depending on the pathway of transformation from rainfall to photosynthesis to food resource and the specific resource requirements of the species. Variation in bird response to productivity may also be due to differences in the way that carbohydrates are apportioned within a plant. Loyn (1985) suggested that forest types differ in the way that resources are apportioned to timber and edible plant resources.

The degree of correlation of species abundance or species richness with local scale productivity probably also depends on the spatial scale over which species forage. The occurrence of some highly nomadic species shows no direct relationship with plant productivity although they are related to food availability. In northern Australia, for example, movements of a number of Australian raptors occur in response to fire as they exploit resources made readily available in fire fronts and ephemeral post-fire habitats (Braithwaite and Estbergs, 1987).

Milledge and Recher (1985) proposed that bird species richness is a product of the combined effects of productivity (vegetation-based food resources) and vegetation complexity. Recher *et al.*, (1996) also proposed that vegetation complexity increases **with** productivity. There is empirical evidence that vegetation complexity increases as a function of productivity and that both act to increase species richness. Bailey *et al.*, (2004) found a positive correlation between productivity (as estimated by NDVI) and the number of functional guilds of birds. The guilds that are added as productivity increases are primarily tree foragers. This is interpreted as a positive relationship between productivity and vegetation complexity (Bailey *et al.*, 2004). Hurlbert (2004) found that increases in avian species richness along a productivity gradient (as estimated from NDVI) occur as a result of two separate factors. Firstly, more productive sites have more individuals. Secondly, in more productive sites, a given number of individuals support a greater number of species due to a change in relative species abundance with increasing habitat complexity (Hurlbert, 2004).

Ludwig *et al.*, (2004) argued that the functional integrity of landscapes, which is the intactness of soil and native vegetation patterns and processes that maintain those patterns, is related to biodiversity even at fine scales in Australia's rangelands. They found that birds are more efficient indicators of disturbances that reduced landscape functionality than invertebrates. In this case, it is the ability of the landscape to retain water and nutrient resources that determines the quantity and quality of vegetation patches, which in turn provides habitat resources for a variety of organisms (Ludwig *et al.*, 2004).

Species richness is related to the amount of evolutionary time[11] that has elapsed since the last major disturbance, the last glaciation for example. Clarke (2007) demonstrated that climate history accounts for some of the variation in patterns of regional species richness. This in turn affects species richness at sub-regional scales (Ricklefs, 1987). An analysis of the geographic distribution of terrestrial birds in Australia found that variation in species richness was partly attributable to historical climatic regimes with the isolation of refuges during cycles of aridity causing loss of species in some regions. The faunal zonation derived from this analysis, however, largely corresponds with continental scale vegetation distribution and **current** climatic conditions (Kikkawa and Pearse, 1969).

In summary, many empirical studies have demonstrated a correlation between plant productivity and bird species richness. The pattern of the relationship varies from one habitat type to another and from one species to another. Different specific causal mechanisms have been hypothesised to account for the observed patterns. The amount of variation in bird species richness that is attributable to productivity in different environments or for individual species remains to be quantified. The **amount** of available habitat resources is presumably also related to the presence and abundance of individuals and species competing for the same specific resources although few studies appear to have quantified this.

3.2.2 Area

The relationship of species richness with area provides insight into the mechanisms that may be involved in the explanation of species richness. Three alternative explanations have been proposed to explain the species-area relationship. In the random sampling hypothesis, sampling larger areas samples an increasing proportion of the species pool. The second explanation is the equilibrium concept in which larger areas permit higher total populations for species so that extinction probabilities will be lower than for small areas (MacArthur and Wilson, 1963; Mac Nally and Watson, 1997; Bellamy *et al.*, 1996; Storch *et al.*, 2007b; Huston,

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[11] The term evolutionary time is used here to refer to the long temporal scales that are involved in speciation. The time taken for species richness to recover following major extinction events, for example, is in the order of tens of millions of years (Wilson, 1992). It is noted, however, that evolution is an ongoing process of genetic change that occurs from generation to generation. As Mayr (2001:190) stated 'all macroeolutionary processes take place in populations and in the genotypes of individuals, and are thus simultaneously microevolutionary processes' (Mayr, 2001).

1994). A third explanation of species-area relationships is that area is simply a surrogate for environmental heterogeneity (Palmer, 2007).

Islands have been important in the thinking about species-area relationships. The concept of patches of remnant natural vegetation as habitat islands within human modified landscapes, combined with an interest in reserve design, has led to many landscape scale studies of birds. These have generally confirmed that bird species richness increases as area increases but have identified a number of other landscape properties that interact with area to determine different bird species distributions (Table 3.1).

Table 3.1 Landscape attributes related to bird distributions

Variable	**Source**
Area of habitat (patch size, remnant size)	(Howe, 1984; Arnold and Weeldenburg, 1990; Bellamy *et al.*, 1996; Stratford and Stouffer, 1999; Major *et al.*, 2001; MacDonald and Kirkpatrick, 2003; Peak and Thompson, 2006; Stouffer *et al.*, 2006; Hannah *et al.*, 2007)
Distance from potential source areas	(Stratford and Stouffer, 1999; Miller *et al.*, 2004; Stouffer *et al.*, 2006; Askins *et al.*, 1987; MacDonald and Kirkpatrick, 2003)
Landscape scale vegetation cover	(Drapeau *et al.*, 2000; Ford *et al.*, 2001; Arnold and Weeldenburg, 1990; Radford *et al.*, 2005; Askins *et al.*, 1987; Saunders, 1993)
Landscape context of habitat islands	(Bellamy *et al.*, 1996; Stouffer *et al.*, 2006; Villard *et al.*, 1999; Stratford and Stouffer, 1999; Recher and Serventy, 1991)
Habitat fragmentation	(Ford *et al.*, 2001; Howe, 1984)
Habitat diversity	(Bellamy *et al.*, 1996; Bohning-Gaese, 1997)
Habitat type / structure	(Kikkawa, 1968; Bohning-Gaese, 1997; Major *et al.*, 2001; Johnson and Mighell, 1999; Arnold and Weeldenburg, 1990; Miller *et al.*, 2004)
Habitat condition	(Ford *et al.*, 2001; Drapeau *et al.*, 2000; MacDonald and Kirkpatrick, 2003)
Time since clearing / isolation	(Bellamy *et al.*, 1996; Recher and Serventy, 1991)

3.3 The variety of habitat resources

A lognormal distribution of species abundance in multi-species samples is widespread although not universal in nature. According to niche theory, the lognormal distribution is caused by niche partitioning, or the way in which species divide resources. Niche partitioning theoretically increases species richness by enabling the segregation and coexistence of species that would otherwise be competing for the same resources (Giller, 1984). Alternatively, spatial heterogeneity caused by the lognormal distribution of environmental resources, rather than the ecological subdivision of resources by organisms, may be the underlying cause of the lognormal distribution of species abundances (Huston, 1994). There is a large body of empirical research on the correlation between species richness and spatial heterogeneity at various spatial scales.

3.3.1.1 A hierarchical model of spatial heterogeneity

Complex spatial patterning in the distribution of environmental inputs and species richness was noted by Nix (1982) who argued that climate interacting with terrain and soil to produce unique radiation, temperature, moisture and nutrient regimes are the primary environmental determinants of biogeographic pattern and evolutionary process. Building on this concept, Mackey and Lindenmayer (2001), outlined a hierarchical framework for modelling the spatial distribution of terrestrial vertebrate fauna. The organising principle for the framework is the scale-dependent delivery of four primary environmental resources that control biological productivity: heat, light, water, and mineral nutrients (Mackey and Lindenmayer, 2001; Nix, 1982). Through the interaction of the four primary environmental resources with spatial factors such as latitude and topography, complex patterns in their spatial distribution occur. With each change down in scale, patterns in the distribution of the primary environmental resources are increasingly fine scaled and drive further spatial heterogeneity through their effects on vegetation distribution and vegetation structure and through the effects of vegetation on sub-canopy processes.

Climatically and topographically controlled distribution of these four primary environmental resources determines spatial patterning at a range of spatial scales and across taxonomic groups. Direct correlations have been found between bird distributions and gradients of temperature, rainfall, soil nutrients and drainage (Neave *et al.*, 1996; Kikkawa, 1982; MacDonald and Kirkpatrick, 2003). More important for fauna, however, is how the primary environmental resources are transformed into vegetation based food and habitat resources. Spatial patterning in the delivery of the primary environmental resources determines spatial patterning in the distribution of major vegetation structural formations (Read, 1987), which are closely related to bird distributions (Keast, 1981c). It also determines spatial heterogeneity within eucalypt formations (Wardell-Johnson *et al.*, 1997; Keith, 1997).

3.3.2 Environmental heterogeneity and bird species richness

Environmental heterogeneity has been found to be correlated with species richness across a range of scales (Koh *et al.*, 2006; Paine, 1966; Tilman, 1994; Pianka, 1975; Currie and Paquin, 1987). A test of the effects of area and habitat diversity found that mammal species richness and diversity was more closely related to the diversity of available habitats than to area *per se* (Fox and Fox, 2000). A study of scale effects on avian species richness demonstrated that avian diversity is determined by spatial heterogeneity at the corresponding spatial scale. Small spatial scale emphasises the importance of habitat diversity whereas large scale emphasises variables related to energy, such as temperature (Bohning-Gaese, 1997).

Most environmental variables exhibit distance decay such that environmental heterogeneity increases inevitably as a function of spatial scale. Palmer (2007) has argued, that environmental heterogeneity should consequently largely determine the species-area relationship, depending on the strength and nature of species-environment relationships. Palmer outlines an environmental-texture model, in which the rate at which species accumulate as a function of scale is influenced by the rate at which new environments appear. This, in turn, is regulated by the geometry of multiple environmental gradients (Palmer, 2007). This conceptual model is similar to Mackey and Lindenmayer's (2001) hierarchical model in that climate, geology and geomorphology are important environmental determinants of species' distributions. Vegetation type is the most obvious biological expression of multiple environmental gradients, *sensu* Palmer (2007), or primary environmental resources *sensu* Mackey and Lindenmayer (2001).

3.3.3 Habitat diversity and bird species richness

Bird distributions are closely related to vegetation type (Keast, 1981c). Structurally similar vegetation types have similar assemblages of birds (Keast, 1981c; Kikkawa, 1968). Consequently, bird species richness at the level of beta diversity, *sensu* Cody (1975), is expected to increase as the diversity of vegetation types increases. The high species richness, for example, of the Torresian province in northern Australia compared to other provinces is attributed largely to greater habitat diversity (Kikkawa and Pearse, 1969). It has been shown empirically that for sites of equal area, the more species rich sites are those with the most habitat types (Rafe *et al.*, 1985; Boecklen, 1986). The relationship of avian species richness to habitat diversity is complex, however, as there is no simple relationship between bird species turnover and change in vegetation types.

To begin with, some vegetation types are more rich in bird species than others (Braithwaite *et al.*, 1989; Rafe *et al.*, 1985). Area for area, closed forests and riparian zones in tropical savannas, for example, have higher bird species richness than adjacent eucalypt woodlands (Woinarski *et al.*, 1988; Woinarski *et al.*, 2000a), and eucalypt forests have higher bird species richness than woodlands (Recher, 1985). This has been attributed to the specificity of resource requirements of closed forest bird species and the greater range of foraging opportunities available in closed forests (Woinarski *et al.*, 1988). Resources are possibly more closely packed in more productive environments reducing the area requirements of individual birds so that, for a given area, productive environments are more species rich.

The relationship of species richness to habitat diversity is complicated too by the fact that there is not a one to one mapping between bird species composition and vegetation type (Miller, 2000). Habitat-specificity (i.e., the degree to which a species relies on a particular vegetation type), differs from species to species and many species occur in a wide range of forest types (Milledge and Recher, 1985; Woinarski *et al.*, 1988; Recher, 2004; Kikkawa, 1982). Each species responds idiosyncratically to the availability of particular resources (Recher *et al.*, 1991), which is also affected by the type and number of species competing for the same resources. Species also differ in the spatio-temporal scales over which habitat mosaics are used (Law and Dickman, 1998). The degree of habitat-specificity has also been shown to vary within a species depending on population density (Wiens, 1989a). Regardless of differences in the degree of habitat-specificity, vegetation classifications based on floristic, structural and life form information, have been found to be useful in describing the distribution of bird species in the Top End of the Northern Territory (Woinarski *et al.*, 1988).

3.3.4 Bird-habitat relationships in *Eucalyptus* formations

Keast (1985b) noted the fundamental similarity in the bird assemblages of Australian eucalypt formations at a continental scale. He attributed this to the similarity in structure of the eucalypt formations throughout Australia and to climatic variability that requires that the fauna be somewhat generalised (Keast, 1985a). The same taxonomic groups are present throughout, many species occur in all regions, and they or congeneric counterparts fill the same niches. The bird communities are also similarly structured in how feeding zones are divided and in the proportion of avian biomass occupying each (Keast, 1985b).

Much of the vertebrate fauna found in eucalypt woodlands also occurs in formations with similar vegetation structure, but that are dominated by other plant genera such as *Acacia*, *Callitris*, *Casuarina* and *Melaleuca* (Woinarski *et al.*, 1997). However, distinctive attributes of eucalypts may play an important role in structuring avian assemblages. Eucalypt formations have more insectivores and nectarivores, but fewer frugivores compared to rainforests (Woinarski *et al.*, 1997). Eucalypt bird communities are distinguished by the abundance and diversity of species that feed on carbohydrates, and bark foraging specialists (Recher *et al.*, 1985). Eucalypt trunks, bark and branches are important for bark gleaning insectivores (Loyn, 1985b; Recher and Holmes, 1985; Noske, 1985). The greater availability of hollows may be an important factor in determining the relative richness of groups such as owls and parrots in eucalypt formations compared to other vegetation types (Woinarski *et al.*, 1997). Eucalypt seeds are the main food resource for some parrots (Recher, 1985).

Compared with other vegetation types, eucalypt formations offer a rich source and variety of carbohydrates. In addition to floral nectar and sap, exudates formed by leaf dwelling arthropods are a key resource for foliage gleaning and nectarivorous birds (Woinarski *et al.*, 1997). Manna, for example, is a sugary exudate that crystallizes around small injuries to eucalypt leaves, lerps are coverings exuded by psyllid nymphs, and honeydew is a viscous liquid exuded by some coccids and aphids. These food resources are far more abundant on eucalypt foliage than in associated non-eucalypts (Woinarski *et al.*, 1997). Recher *et al.*, (1996) found that the richest and most abundant eucalypt arthropod communities occurred in sites with the highest soil fertility and on the tree species with the highest levels of foliage nutrients. The rate at which some species of insectivorous birds selected between *Eucalyptus* species was related to the kinds of arthropods available and their abundance on the foliage (Recher *et al.*, 1996).

Eucalypts are fundamental to the ecology of foliage gleaning and nectarivorous birds. Seasonal variation in nectar availability accounts for species turnover in bird assemblages within the eucalypt formations, although most nectarivorous birds are flexible in their diets and foraging behaviours and may be insectivorous for part of the year. The relatively minor seasonal variation in the abundance of most invertebrate groups provides a temporal stability to bird assemblages which is in contrast to that of non-Australian forests and woodlands (Woinarski *et al.*, 1997).

3.3.4.1 Bird-habitat relationships in tropical *Eucalyptus* formations

By definition, tropical eucalypt formations occupy a very different climatic regime compared with temperate eucalypt forests. Tropical eucalypt formations are floristically and structurally distinct in a number of respects that are significant for fauna (Woinarski *et al.*, 1997). The *Eucalyptus* species that evolved in tropical Australia are specifically adapted to the climatic conditions so that they rarely grow successfully in southern Australia (Brooker and Kleinig, 2004). The northern eucalypt formations are composed mainly of the sub-genera *Corymbia, Blakella* and *Eudesmia* whereas *Symphyomyrtus* and *Monocalyptus* dominate the southern eucalypt forests. The northern eucalypt formations are typically co-dominated by non-eucalypts from a wide taxonomic range. They also differ in that they characteristically have a dense, tall and diverse tropical grass understorey (Morton and Brennan, 1991). The trees generally do not have decorticating bark and branching angles are more vertical. They have very high populations of termites and the disturbance factors that affect plant demography and vegetation structure are different (Woinarski *et al.*, 1997). The tropics also differ from temperate parts of Australia in their climatic and isolation history.

Structural and floristic differences in the vegetation are reflected in the bird assemblages. Bird assemblages in the tropical eucalypt forests and woodlands differ in composition and structure from temperate eucalypt formations in having more granivores, fewer insectivores and fewer foliage gleaners (Woinarski *et al.*, 1997; Keast, 1985b; Woinarski and Tidemann, 1991; Morton and Brennan, 1991). Nevertheless, insectivores and nectarivores together constitute 60% of the bird species and close to 90% of the avian biomass. Granivores are the only other foraging group that is important in terms of biomass (Morton and Brennan, 1991).

Bird species richness in the monsoonal eucalypt formations has been found to change in association with gradients of canopy height, basal area and floristics (Woinarski *et al.*, 1999). Bird species richness declined from the eucalypt forests of high rainfall areas to the woodlands of lower rainfall areas. Changes in bird species composition were related to changes in vegetation structure associated with soil texture and rainfall. Subtle variation in vegetation structure within the eucalypt formation along the rainfall gradient, however, was not reflected in bird species richness.

Tree hollows are important for birds in northern Australia and are an important resource for a range of fauna including many that do not normally live in eucalypt formations. A number of duck species, for example, frequently use hollows for nesting or roosting (Taylor *et al.*, 2003). Taylor *et al.*, (2003) found that 18% of all bird species and 21% of forest birds in the Northern Territory used hollows. This compares to 15% of all land birds in Australia (Gibbons and Lindenmayer, 2002).

3.3.5 Vegetation structural complexity and bird species richness

The complexity of vegetation structure clearly differs between vegetation formations, but it also varies within vegetation formations in response to variation in topography, availability of the primary environmental resources and disturbance. Correlation between avian diversity and structural complexity of vegetation has been demonstrated empirically in many studies (Roth, 1976; Karr and Roth, 1971; MacArthur and MacArthur, 1961; Erdelen, 1984; MacArthur, 1964; Willson, 1974). It is not necessarily clear in these studies whether variation in structural complexity is due to different habitat types being compared or similar habitat types with structural variation. Wiens and Rotenberry (1981) found that bird habitat associations were most apparent when the habitat gradients surveyed were large, as in a continental survey including quite different habitat types. Differences in sampling scale, definitions and methods make it difficult to draw generalizations from these studies.

The main effect of an increase in the variety of habitat resources appears to be an increase in the number of foraging guilds rather than an increase in richness within foraging guilds. Addition of foraging guilds occurs as vegetation increases in structural complexity. According to niche theory, species richness is defined not only by the number of niches, but also by the niche width and niche overlap between species. Theoretically, species become specialised in their foraging behaviour to avoid inter-specific competition for limited resources (Giller, 1984). Although inter-specific competition potentially reduces the amount of a specific habitat resource leading to a need for increased specialisation, the physical structure of vegetation, limits the variety of foraging sites available.

A study that compared birds in a variety of vegetation types from grassland to shrubland to forest in Illinois, USA, demonstrated that bird species richness increased due to the addition of foraging guilds with new vegetation layers (Willson, 1974). Development of a ground cover layer added one guild with one or two species. Development of the shrub layer added one of four guilds with one to four species. When the tree layer began to develop up to seven more guilds with approximately 12 more species were added. With increasing development of the tree layer, one or two more guilds were added accounting for two or three more species. Development of the tree layer also changed the configuration of lower layers resulting in a re-sorting of guilds in all layers with some guilds expanding and others losing species (Willson, 1974).

3.3.5.1 Birds and structural complexity within *Eucalyptus* formations

Patterns of bird species richness and abundance in eucalypt forest and woodlands have been related to the structure and the productivity of the plant communities in which they reside (Recher, 1985). A review of fauna-habitat relationships found that avian diversity increased in eucalypt formations where a diversity of vegetation strata was available as foraging substrates (McElhinny *et al.*, 2006). Habitats with the highest avian diversity were those that supported a greater variety of foraging groups. Six habitat components were consistently identified as important bird resources:

1. foliage - a source of exudates and invertebrates,
2. flowers - a source of nectar and invertebrates,
3. bark - a source of exudates and invertebrates,
4. ground layer - a source of invertebrates and small vertebrates,
5. air spaces within and between canopy strata - a source of invertebrates, and
6. hollow-bearing trees - for nesting and shelter (McElhinny *et al.*, 2006).

Bird assemblages in temperate eucalypt forests are partitioned firstly by food type, then by substrate and then foraging behaviour (Table 3.2). With increased structural complexity of vegetation more types of food become available.

Table 3.2 Partitioning of bird assemblages in temperate eucalypt forests[1]

Food type, substrate and foraging behaviour			**Example**
Nectar			Lorikeets
Nectar + insects			Spinebill / Wattlebird / Friarbird / Honeyeater
Fruit			Mistletoebird
Seed	Trees and herbs		Parrots / Rosellas
	Herbs and grasses		Finches
Insects	Ground	Glean	Lyrebird / Chough / Fairywren
		Pounce	Robins
	Air	Hawk	Flycatchers / Fantail
		Aerial	Woodswallows
	Bark	Glean	Treecreepers / Sittellas
	Bark + ground		Shrike-thrush
	Loose bark		Shrike-tit
	Leaves + nectar		Honeyeaters
	Leaves	Glean	Pardalotes / Thornbills / Honeyeaters
	Leaves	Pounce	Whistlers / Cuckoo-shrike

[1] (Redrawn after Ford, 1989)

In some cases, indices of structural complexity have been correlated with bird species richness in eucalypt formations. More often, specific vegetation structural attributes are associated with a single species by providing a specific habitat resource. In some cases a combination of vegetation elements is required (Woinarski, 1985). The degree to which any single factor or combination of factors applies, depends on the specific requirements of the individual species. A range of specific structural attributes of vegetation have been found to be correlated with bird abundance, species richness, number of foraging guilds, or richness of foraging guilds in eucalypt formations (Table 3.3).

Table 3.3 Eucalypt structural attributes related to bird distributions

Vegetation attribute	**Source**
Canopy height (insectivores)	(Woinarski *et al.*, 1988; Recher *et al.*, 1991; Gilmore, 1985; Woinarski, 1985)
Foliage height diversity (species richness)	(Recher, 1969)
Number of vegetation layers	(Recher *et al.*, 1991)
Foliage cover / canopy volume	(Gilmore, 1985)
Structural complexity / diversity	(Recher, 1969; Loyn, 1980; Arnold, 2003; Erdelen, 1984; Morton and Brennan, 1991; Kikkawa, 1982)
Amount of open area (aerial insectivores)	(Shields *et al.*, 1985)
Tree basal area	(Braithwaite *et al.*, 1989; Recher *et al.*, 1991)
Tree density	(Ford and Bell, 1982)
Understorey density	(Woinarski *et al.*, 1988; Loyn, 1985b)
Vegetation biomass (insectivores)	(Gilmore, 1985)
A combination of vegetation elements	(Woinarski, 1985)
Floristic diversity	(Milledge and Recher, 1985; Recher *et al.*, 1991; Gilmore, 1985)
Presence of nectar rich trees (nectarivores)	(Woinarski *et al.*, 1988; Franklin and Noske, 1999; Woinarski *et al.*, 2000b; Mac Nally and McGoldrick, 1997; Milledge and Recher, 1985; Loyn, 1985b)
Bark attributes (bark foragers)	(Gilmore, 1985; Recher *et al.*, 1985; Recher and Davis, 1998; Loyn, 1985b)
Abundance and type of tree hollows	(Loyn, 1985b; Noske, 1985; Brigham and Fritz, 1997)
Dead wood	(Noske, 1985)
Presence of mistletoe (frugivores)	(Woinarski *et al.*, 1988; Watson, 2002; Milledge and Recher, 1985)
Leaf litter, logs and debris (ground foragers)	(Milledge and Recher, 1985; Gilmore, 1985; Recher and Davis, 1998; Laven and Mac Nally, 1998)

3.3.6 The effect of disturbance on environmental heterogeneity

Disturbance can increase species richness by increasing environmental heterogeneity over a range of spatial and temporal scales (Urban and Smith, 1989). Disturbance can create spatial heterogeneity at landscape scales. It can also affect the short term availability of food resources over small areas or cause long term changes in vegetation structure and composition, depending on the type, timing, frequency and intensity of disturbance.

The dynamic equilibrium hypothesis proposes that under equilibrium conditions species richness is reduced by competitive displacement whereas disturbance (abiotic or biotic), maintains diversity and enables local co-existence of species by reducing the rate of competitive displacement (Huston, 1979). According to this hypothesis, the outcome of disturbance in terms of species richness depends on the interaction between: (1) the frequency and/or magnitude of population reduction; and (2) the rate of competitive displacement and population growth rates. Alternatively, meta-population theory suggests that even if competitive exclusion does occur, disturbance can create spatial heterogeneity at landscape scales that allows species to co-exist at a regional scale in spite of local extinctions (Huston, 1994).

It has been shown, however, that there is no long term correlation between a species' abundance and its rate of increase (Levins, 1979). Connell (1978) argued that the degree of past disturbances is more important than differences in the rate of competitive displacement during recovery from disturbance. He proposed that disturbance determines the basic physical structure, or spatial heterogeneity, of communities and that diversity is higher when disturbances are intermediate on the scales of frequency and intensity (Connell, 1978). Disturbance appears to have both direct effects on fauna as well as indirect effects through its effects on habitats.

3.3.6.1 Effects of fire on environmental heterogeneity

Bird species turnover following fire has been documented in different vegetation formations over a range of time scales (Woinarski and Recher, 1997). Fire can directly impact bird populations by causing mortality at the time of fire and through longer term effects on post-fire survival, population size and recruitment (Loyn, 1997; Baker *et al.*, 1997). Changes in bird species composition following a single fire event depend on a range of factors including: species mobility; the effects of fire on key short term and long term resources; pre-fire population levels; post-fire climatic conditions; availability of refuge areas; and landscape connectivity (Woinarski and Recher, 1997). The effects of fire on bird populations are likely to be very important especially given the cumulative impacts of widespread habitat loss, fragmentation and degradation. Direct population effects, however, are not well known.

Fire can also have indirect effects on bird populations through its effects on vegetation composition and structure. Most of the documented impacts of fire on birds are indirect effects due to the effects of fire on vegetation structure and composition. Bird response to vegetation succession following fire differs between vegetation formations (Loyn, 1997). Over short time frames a number of species may be attracted to resources that become available in recently burnt areas (Woinarski and Recher, 1997; Loyn, 1997; Braithwaite and Estbergs, 1987; Woinarski, 1990). Fire can have short term effects on vegetation by removal of shrub layers, or hanging bark. It can also have long term effects on vegetation structure, composition and the availability of tree hollows (Kutt and Woinarski, 2007; Woinarski and Recher, 1997).

3.3.6.2 Effects of fire in monsoonal *Eucalyptus* formations

Intense seasonality of rainfall in monsoonal *Eucalyptus* formations annually produces a large body of grass followed by annual drought. High temperatures and the body of fuel result in frequent and widespread grass fires (see section 2.6). Various bird responses have been documented to the fire regime of the tropical *Eucalyptus* forests and woodlands.

Two experimental studies, one in Queensland and one in the Northern Territory have found that the season of burning has effects on the composition of bird assemblages in tropical eucalypt formations. In the Northern Territory, Woinarski (1990) found differences between foraging groups in their responses to fire regime. Granivores are most common in plots subjected to early annual burns. Omnivores are more common in biennial and early burnt plots. Insectivores and insectivore-nectarivores are most abundant in unburnt plots, whereas carnivores are significantly less common in unburnt plots. Birds are attracted to recently burnt areas because of increases either in food abundance, or food accessibility. Removal of tall, dense grasses, for example, increases access to fallen seeds for ground feeding granivores. Changes in bird composition following fire persist from less than 3 days to more than 4 months (Woinarski, 1990). The Queensland study found that the long term effects of the season of burn may differ from the short term responses (Valentine *et al.*, 2007).

A comparison of the effects of fire and grazing on vertebrates in tropical savannas found that terrestrial bird species are more prevalent in recently burnt areas and a smaller set of bird species is more common at unburnt sites (Kutt and Woinarski, 2007). Williams *et al.*, (2003) noted that late dry season fires can reduce flowering and fruiting leading to a reduction in faunal food resources (Williams *et al.*, 2003b). The rate of change in bird response to single fire events and the duration of the effects of fire on bird assemblages are very different from the long term-successional changes that occur in temperate *Eucalyptus* forests (Braithwaite and Estbergs, 1987). The frequency of fire in the tropical *Eucalyptus* open forests and woodlands is such that it almost continuously provides temporary habitat for mobile species. Braithwaite and Estbergs (1987) found rapid turnover in highly mobile birds with different species exploiting different phases of the ephemeral habitats created by fire. Different birds were present in: (1) the fire front lasting minutes; (2) the hot ash phase lasting days; (3) the cold black phase after smouldering has stopped and lasting up to three weeks; and (4) the vegetation re-sprouting phase. The accessibility and quality of food resources for a range of birds are enhanced in the few months following a fire.

Most of the studies on bird response to fire in the tropical savannas have found short term changes in bird abundance and species composition. Woinarski (1990) found little evidence of succession of bird species to long unburnt areas, although species that use shrubby understoreys occurred at higher densities in long unburnt areas. Another study found significant reduction in bird species diversity in tropical eucalypt forest due to the loss of a dense species-rich understorey (Woinarski *et al.*, 1988). A later study, in which fire had been excluded for 23 years, found significant differences in bird species composition between a burnt and a long unburnt site (Woinarski *et al.*, 2004). The response was attributed to significant changes in vegetation composition and structure that accompanied long term fire exclusion.

The previous lack of evidence for bird succession to 'long unburnt' areas appears to have been due to fire not having been excluded long enough for a strong gradient of vegetation composition and structure to have developed. It also appears to be due to the predominance of frequently burnt country in the landscape. Woinarski *et al.*, (2004) suggested that species that are disadvantaged by fire have disappeared, whereas more fire tolerant species persist in tiny fragments of long unburnt country that comprise less than 1% of the landscape (Woinarski *et al.*, 2004). There is also a gradient of bird tolerance to fire. Some species, such as the red-backed fairy wren *Malurus melanocephalus* are disadvantaged by fire but are able to persist in the landscape by exploiting unburnt patches of vegetation (Woinarski, 1990). Distance from source areas, and permeability of the landscape, may also have contributed to previous lack of evidence for bird succession to long unburnt areas.

A survey of birds in relation to vegetation patterns by Woinarski *et al.*, (1988) included *Eucalyptus* forest and woodlands sites that had been devastated by Tropical Cyclone Tracey 12 years prior to the study and undamaged *Eucalyptus* forest and woodland sites. They concluded that bird distribution is not related to cyclone damage or to repeated fire.

3.4 Partitioning of habitat resources

3.4.1 Stability of habitat resources

Various hypotheses have been proposed to account for relationships between species richness and environmental stability. The environmental stability hypothesis proposes that conditions of environmental stability favour the co-existence of fauna that are more specialised, or less plastic, in their behaviour and consequently use smaller niches (Klopfer and MacArthur, 1961). Stated another way, environmental instability sets a limit to the degree of specialisation (MacArthur and Levins, 1967), and therefore to the number of resident species that can form stable populations. If this is so, then the stability of habitat resources within a given habitat should be reflected in the ratio of sedentary specialised resident species to migratory or nomadic visitors more tolerant of variation. Empirical studies have demonstrated relationships between bird species richness and the stability of habitat resources over a range of spatial and temporal scales.

3.4.2 Stability of habitat resources and Australian bird assemblages

Keast (1967) found that the species richness of honeyeaters in Australia was related to climatic stability, particularly seasonality of rainfall. Climatically stable locations, with high rainfall and botanical diversity permitted the co-existence of the largest numbers of species, the highest proportion of residents and the highest proportion of breeders (Keast, 1967). That is, the movement behaviour of bird species varied with the stability of habitat resources.

Chan (2001) and Gilmore *et al*. (2007), found respectively that 44% and 51% of Australian land and freshwater birds exhibited some types of dispersive movement. Gilmore *et al*., (2007) proposed that dispersive movements can be interpreted in terms of spatio-temporal variation in the outcomes of individuals maximising Darwinian fitness by trading off energy intake against mortality risk. Bailey *et al*., (2004) demonstrated that the relationship of dispersive movement behaviours of species or populations to variation in productivity in the USA differed depending on the specific resource requirements, the stability of resource availability over the range of a species' distribution and on species' vagility. Migrant species account for correlation of species richness with productivity maxima (estimated by NDVI) as they are able to use their mobility to track temporal and spatial variation in productivity. In contrast, the species richness of resident birds is correlated with productivity minima (Bailey *et al*., 2004).

Temporal variation in the availability of specific resources across a species distribution may drive different dispersive movement behaviours of populations within a species (Gilmore *et al*., 2007). Wiens (1989) cited examples of populations fluctuating more at the limits of their distributions than at the centre, creating landscape scale 'source and sink patterns' determined by the stability of resources within source habitats. Keast (1967) also found that variation from the usual patterns of seasonal movements of honeyeaters occurs in the periphery of a species' distribution. If populations are the unit of evolution, then different movement behaviours of populations within a species associated with variation in environmental conditions certainly provide the material for speciation to occur. The minor seasonal variation in the abundance of most invertebrate groups in the *Eucalyptus* formations provides temporal stability of habitat resources which theoretically should lead to an increase in the richness of insectivores, particularly sedentary insectivores.

3.4.2.1 Stability of habitat resources in the tropical *Eucalyptus* formations

The tropical *Eucalyptus* forests and woodlands of Australia provide a rich nectar resource. In the Northern Territory, eucalypts dominate annual nectar production providing over 75% of annual production. *Eucalyptus tetrodonta* and *E. minata* account for almost a third of all nectar production (Woinarski *et al.*, 2000b). No comparable studies have been conducted in Queensland but given its dominance it is likely that *E. tetrodonta* is a very important source of nectar on Cape York Peninsula. The abundance of nectar in the northern eucalypt forest coincides with the troughs of many resources in winter in temperate and inland Australia. Consequently, there is a substantial continental scale migration to northern Australia by facultatively nectarivorous species such as the white-winged triller (Woinarski *et al.*, 2000b).

Although nectar is available year round, there is significant inter-annual and intra-seasonal variation in amount, habitat type and location as the flowering times of many eucalypt species are staggered through the dry season (Setterfield and Williams, 1996; Woinarski *et al.*, 2000b). Nectar is most abundant during the dry season when fruit and insects are scarce (Franklin, 1999). Birds respond to the spatially and temporally variable supply of resources and the provision of a widespread nectar resource in structurally simple flowers with two basic strategies, nomadism and dietary opportunism (Woinarski *et al.*, 2000b).

Nectarivores escape the resource limitation they would face if they were sedentary by tracking nectar resources over long distances. When the nectar resource is scarce, nectarivores become concentrated in spatially limited patches of rich nectar sources (Woinarski *et al.*, 2000b; Morton and Brennan, 1991). Some species escape resource limitation by resource switching. A high proportion of dry season nectarivores are insectivorous or frugivorous at various times (Franklin, 1997; Woinarski *et al.*, 2000b). The rich nectar resource also supplements the diets of many species that are not primarily nectarivorous. More than 20 bird species that are primarily insectivorous, carnivorous or granivorous are known to be opportunistic nectarivores, feeding from eucalypt flowers during the peak nectar production periods of the mid dry season (Franklin, 1999; Woinarski *et al.*, 2000b). Consequently, few of the northern Australian nectarivorous birds show the morphological specializations that characterize heathland nectarivores in temperate Australia (Woinarski *et al.*, 2000b).

3.4.2.2 Biotic interactions

Ultimately, the number of available niches that are able to support viable populations is defined by the spatial complexity and productivity of the environment. The environment, however, also consists of other species which are potentially using the same habitat resources. The niche diversification hypothesis proposes that species richness is defined not only by the number of niches, but also by the niche width and niche overlap between co-existing species (Connell, 1978). For a given range of habitat resources, more species can be packed in the more specialised they are. Partitioning of resources has been interpreted as avoidance of inter-specific competition that has resulted in niche differentiation and speciation. Differentiation results in each species becoming competitively superior at exploiting a particular subdivision of the habitat resources (Giller, 1984). Connell (1980) argued that the likelihood of co-evolution between competing species depends on how similar their resource requirements are and how often they meet. That likelihood is reduced when species have different tolerances to the physical environment, or in conditions of high environmental variability. He argued that there is little evidence that past competition has led to niche diversification (Connell, 1980).

Most evolutionary change is driven by behavioural shifts, and advantageous behavioural changes are likely to be reinforced by the selection of genetic determinants for such behaviour (Mayr, 2001). Opportunistic behaviour that results in exclusive use of a resource (niche diversification) is therefore likely to be selected although this is not necessarily the same as competition avoidance. Nevertheless, the evolutionary success of diversification in resource use leading to stable species co-existence is presumably determined by the stable supply of minimum amounts of the specific habitat resources on which species have become specialised to support viable populations.

Furthermore, the competitive equilibrium hypothesis assumes that communities are biologically saturated and in equilibrium. There is now abundant evidence that many ecological communities are not biologically saturated or in equilibrium (Rohde, 2005). There is some evidence for tropical plant communities that niche processes may alter community structure through regulating relative species abundance (Kelly *et al.*, 2008). However, niche diversification processes in assemblages of mobile species may not be the same as they are in sessile communities.

The predation hypothesis suggests that selective predation, on the most dominant or most abundant species can maintain high species richness over time. As with abiotic disturbances, the intensity of predation is important. The effects of predation on species richness have been demonstrated experimentally in benthic communities (Paine, 1966). Paine (1966) hypothesised that local species diversity is directly related to the efficiency with which predators prevent any one species from monopolising resources. In general, however, predation only appears to lead to high species richness under a limited set of conditions (Giller, 1984).

3.4.2.3 Biotic interactions in *Eucalyptus* formations

According to Ford (1989), the structure of bird assemblages in *Eucalyptus* woodlands and forests is determined primarily by the availability of suitable habitat resources. There is evidence that inter-specific competition for resources occurs in honeyeater assemblages but the evidence is less clear for insectivorous species (Ford, 1989). The available empirical evidence for inter-specific competition appears to relate mostly to interactions such as aggression rather than competition for the same resources. Most of the empirical evidence for biotic interactions among Australian birds comes from environments in which vegetation structure has been altered by human activities such as land clearing, forestry and agriculture.

Loyn (1985), for example, observed an influx of bird species following experimental removal of bell miners *Manorina melanophrys* in eucalypt forests in Victoria. The abundance of noisy miners *Manorina melanocephala*, a colonially nesting honeyeater, has been shown to affect bird species composition, richness and diversity by aggressive exclusion in remnants of eucalypt forest in Tasmanian (MacDonald and Kirkpatrick, 2003) and in remnants of tropical *Eucalyptus* woodland (Hannah *et al.*, 2007). Green and Catterall (1998) also found that noisy miners were possibly involved in the reduced abundance of individual forest birds from edge habitats. The composition of bird assemblages has also been found to be affected by biotic interactions involving two species of aggressive, colonial honeyeaters, the yellow-throated miner and white-plumed honeyeater in natural edge habitats, riverine eucalypt woodland habitat (Pavey and Nano, 2009). In this case the presence of the aggressive honeyeaters was related to the absence of most species of small birds (< 50 g) from riverine woodland.

Habitat degradation in particular has resulted in increasing abundance of aggressive species such as the noisy miner with adverse consequences for many birds (Olsen *et al.*, 2005). MacDonald *et al.*, (2003) found that the ability of noisy miners to exclude other bird species depends on the absence of understorey vegetation. Vegetation clumping in a rehabilitated minesite allows large honeyeaters to exclude small nectarivores (Comer and Wooller, 2002). Habitat loss also results in increasing inter-specific competition for scarce resources such as tree hollows (Saunders, 1993).

A number of authors have concluded that the bird assemblages of Australian *Eucalyptus* forests and woodlands are largely random and independent assemblages of species that can co-occur, with each species responding idiosyncratically to the availability of particular food resources, vegetation structure and nest sites over varying spatial and temporal scales (Keast, 1985b; Recher *et al.*, 1991; Mac Nally and McGoldrick, 1997; Woinarski and Tidemann, 1991). Mac Nally and McGoldrick (1997) suggested that the lack of evidence of structuring within bird assemblages of eucalypt forests in Victoria may be due to the absence of barriers to the movements of large numbers of birds across Australian landscapes.

Woinarski *et al.*, (1997) suggested a number of interacting factors that account for the homogeneity of bird assemblages of *Eucalyptus* formations. First, the dominance of *Eucalyptus* woodlands and forests in the Australian landscape provides access to extensive and relatively consistent supply of invertebrates. This consistency, combined with the ecological versatility and mobility of birds, has contributed to the homogeneity of bird assemblages in the *Eucalyptus* formations. They also suggested that the limited seasonal variation in invertebrate resources in *Eucalyptus* forests and woodlands underpins the life-history characteristics of Australian birds such as relatively low annual reproductive rates, their longevity, and the high incidence of co-operative breeding (Woinarski *et al.*, 1997).

In summary, the environment determines the amount and variety of habitat resources. The availability of specific resources, however, is theoretically also a function of the types and number of species competing for the same resources. There is little empirical evidence that competition for resources limits and structures bird assemblages in the *Eucalyptus* formations. The high mobility of birds, the variability of the environment and inter-specific variation in bird-habitat relationships may mean that resources are rarely so limiting that there is strong inter-specific competition for resources.

3.5 Habitat resources: a conceptual framework

This review so far has identified a number of variables that are involved in the explanation of bird species richness. Empirical evidence has been presented that area, productivity, habitat diversity, habitat complexity, habitat stability and disturbance are all correlated with bird species richness. Three broad problems confound any simple interpretation of species richness patterns. First, species richness only becomes meaningful when the spatio-temporal framework is clearly defined, habitat is defined and habitat-specificity is accounted for. Species richness has variance, and for any given habitat it needs to be defined in terms of the different habitat relationships of resident and non-resident birds. Second, no single variable can account for the multi-dimensionality of species' space. Multi-dimensionality is complicated by the fact that some of the variables that affect species indirectly through the amount and variety of habitat resources also act directly or indirectly on species. In addition to the indirect effects of temperature through plant productivity, for example, metabolic theory proposes that latitudinal and altitudinal patterns in species richness are due to temperature dependent effects on population density, population growth rates and speciation rates (Brown *et al.*, 2004). In addition to the indirect effects of rainfall through plant productivity, rainfall has been shown to directly influence the timing of reproduction in at least one bird species through its effects on plant chemistry which in turn alter bird reproductive hormones (Morrison *et al.*, 2006). Finally, understanding patterns of species richness has to account for the effects of spatial and temporal scale on pattern and process.

Southwood (1977, 1988) outlined a theoretical framework that is useful for making sense of the complexity of species-habitat relationships that underlie patterns of species richness. It describes the mechanism by which fauna-habitat relationships can evolve with reference to habitats that vary in their productivity, spatial complexity, resource stability and disturbance. In doing so it accounts for inter-specific variation in foraging and movement behaviours. Importantly from the point of view of this thesis, it places all of this within a temporal framework.

3.5.1 The habitat templet

Southwood (1988) proposed that certain combinations of adaptations for survival and reproduction are selected which improve individuals' fitness in ecological time. He described five categories of tactics on which natural selection can operate: (1) tolerance of harsh conditions; (2) defence against other organisms; (3) food harvesting and growth; (4) migration or change of habitat; and (5) reproductive effort. The optimum combination of tactics will be the one that confers the highest fitness with respect to the environment, or habitat(s), that a species occupies, including the presence of other species.

Habitat is characterised by two abiotic axes which represent spatial and temporal heterogeneity in habitat resources and a third axis of biotic interactions (Southwood, 1988). The disturbance axis represents the durational stability of a habitat and characterises it with respect to biomass, spatial complexity and trophic complexity (Southwood, 1977). The adversity axis represents the resource level and constancy of a habitat and characterises a habitat's productivity and stability of habitat resources. The axis of biotic interactions represents the impact of the biotic components of the environment and incorporates predation, parasitism and competition. When combined, these three axes enable predictions about relationships between characteristic life history strategies and habitat attributes.

Species are not confined to a single habitat, they also select habitat in ecological time (Rosenzweig, 1981). Trade-offs may involve fitness with respect to multiple habitats (Morrison *et al.*, 2006). Nevertheless, the habitat templet is a useful theoretical framework for understanding how strategies such as nomadism, opportunism and specialisation might be related and how they and species richness might vary in relation to environmental domains described by the axes of the habitat templet (Mackey *et al.*, 2008). It is a particularly useful theory for understanding species characteristics and distributions in relation to successional environments.

3.5.2 The effects of spatial and temporal scale on pattern and process

Many researchers have concluded that the significance with which any one environmental variable is correlated with bird species richness or abundance is scale (Cody, 1975; Wiens and Rotenberry, 1981; Addicott *et al.*, 1987; Currie and Paquin, 1987; Wiens, 1989a; Wiens, 1989c; Virkkala, 1991; Levins, 1992; Drapeau *et al.*, 2000; Gaston, 2000; Willis and Whittaker, 2002; Currie, 2007; Storch *et al.*, 2007a). Associations that appear to be significant at one spatial scale are non-significant at another. Some researchers are still trying to identify single factors that are important over different spatial scales. Many, however, have accepted that multiple factors operating simultaneously, some directly others indirectly, are involved in the explanation of species diversity (Rosenzweig, 1975; Colwell, 1992; Levins, 1992; Blackburn and Gaston, 1996; Recher, 1971; Willis and Whittaker, 2002; Cueto and Lopez de Casenave, 1999).

An alternative but not mutually exclusive approach, is that a few environmental determinants underlie species richness patterns but different processes operate on species richness at different spatio-temporal scales (Clarke, 2007). The concept that the environment consists of hierarchies of ecological processes influencing diversity at different spatial scales offers a useful conceptual framework for understanding the complexity of findings. A number of researchers have outlined hierarchically structured models[12] that incorporate both pattern and process (Holling, 1992; Allen and Hoekstra, 1992; Mackey and Lindenmayer, 2001; Clarke, 2007).

The hierarchical model of Mackey and Lindenmayer (2001) provides a spatially explicit framework for understanding scale dependent variation in the environmental factors that determine species' distributions. Their framework explicitly recognises the underlying environmental determinants of species' distributions and illustrates how patterning in spatial distribution changes with scale. It demonstrates how different processes are involved in the spatial delivery of the primary environmental resources at different scales (Table 3.4). The interaction of the four primary environmental resources with landscape involves large scale abiotic inputs and small scale biotic driven processes and drives increasingly fine-scaled spatial heterogeneity (Mackey and Lindenmayer, 2001).

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[12] Hierarchical systems can be nested or non-nested. In nested systems such as landscapes, levels of organisation are ordered by their spatial and temporal context and lower levels are spatially contained within higher levels. In non-nested systems, such as food chains, biological interactions decouple systems from direct physical determination by spatial patterning. Non-nested systems are therefore ordered by the differing scales of functioning and the flow of energy (Allen and Hoekstra, 1992; Wiens, 1989c).

Table 3.4 Hierarchical scale effects on species richness[1]

Spatial scale	Structuring variables and processes	Environmental heterogeneity	Outcome for species richness
Global scale	The shape of the planet, its tilt and its elliptic orbit around the sun drive latitudinal and seasonal variation in extra-terrestrial radiation and determine availability of solar energy. Planetary processes such as movement of tectonic plates, global cooling and evolutionary processes leading to speciation operate over millions of years.	Availability of solar energy is the primary constraint on global scale productivity which in turn defines distribution of biomes. Historically, major changes in distributions have occurred with cycles of aridity.	At the global scale, latitudinal productivity gradients are positively correlated with bird species richness. Evolutionary processes, specifically speciation, are reflected in the biogeographic distribution of species, and regional species richness at global and continental scales.
Meso scale	Interaction of broad scale topography with weather drives long term precipitation, thermal and radiation conditions. Lithology exerts a major control on soil profile and nutrient status of the regolith.	Thermal regimes and water availability are the main meso-scale determinants of photosynthesis and thus vegetation physiognomy and distribution of major vegetation structural formations.	Vegetation structural formation is the main determinant of faunal assemblages (gamma-diversity).
Topo-scale	Terrain modifies the delivery of precipitation, temperature, radiation, potential evaporation and nutrients. Disturbance, hydrological and erosive processes operate over hundreds to thousands of years.	Diversity of vegetation structural formations.	At the landscape scale species richness increases with the diversity of vegetation structural formations (beta-diversity).
Canopy / Sub-canopy-scale	Vegetation cover exerts a strong influence on the light, heat, water and mineral-nutrient conditions experienced by sub-canopy plants and animals. Physiological processes, biotic interactions and disturbance, operate over 1 to hundreds of years.	Complexity of vegetation structure.	Structural complexity of vegetation combined with productivity increase within-habitat bird species richness (alpha-diversity).

1. Redrawn after Holling (1992); Mackey and Lindenmayer (2001); Willis and Whittaker (2002); and Clarke (2007).

In their hierarchical model, the four primary environmental resources: heat, light, water, and mineral nutrients define the general environmental context within which individuals persist and constitute a basic set of constraints with which an individual must contend (Mackey and Lindenmayer, 2001). At what is commonly referred to as the landscape scale, climatically and topographically controlled variation in the spatial delivery of these environmental resources produces habitat diversity and heterogeneity within broadly similar vegetation structural formations. Mapping that combines topographic information with remotely sensed vegetation data has been proposed as a tool for rapidly assessing potential habitat for many species (Venier and Mackey, 1997). Combining this spatially explicit approach with the habitat templet theory enables testable predictions to be developed about genetic differences between populations and vertebrate life-history strategies, including their movement strategies across productivity regimes (Mackey *et al.*, 2008).

In addition to these effects, the hierarchically scaled patterns of availability of the primary environmental resources promote or constrain an animal's behaviour to produce a second hierarchy of distributional behaviour (Mackey and Lindenmayer, 2001). The hierarchy of distributional behaviour consists of the species *in toto*, populations, colonies or breeding pairs, and individuals. Mackey and Lindenmayer (2001) proposed that understanding the spatial distribution of an animal requires an understanding of how these two hierarchies intersect (Mackey and Lindenmayer, 2001).

The relative importance of ecological and evolutionary processes in determining biodiversity changes with increasing spatial scale, creating a spatially nested hierarchy of processes (Ricklefs, 1987; Clarke, 2007). At small spatial scales and the level of populations, biotic interactions, disturbance and micro-evolution are emphasised. At intermediate spatial scales spatial heterogeneity is emphasised. At large spatial scales, long term processes such as speciation, extinction and climate change are emphasised. At no spatial scale do either ecological or evolutionary processes have no effect but the balance changes with spatial scale. Clarke (2007) argued that to understand patterns on a global scale, the role of evolutionary processes operating over long time scales is important.

3.6 Conclusion

This review of the literature has identified a number of environmental variables that are correlated with bird species richness. This approach has been taken, not because species richness is all important but because it provides a useful entry point for the research literature on bird-environment relationships generally.

Bird species richness in the Australian *Eucalyptus* forests and woodlands is broadly related to plant productivity (amount of habitat resources), structural complexity of vegetation (variety of habitat resources) and the stability of habitat resources. The number and diversity of foraging substrates is of particular importance as the most species rich *Eucalyptus* formations are those with the highest number of foraging groups. Due to climatic variability, eucalypt bird assemblages are somewhat generalised, however, species richness generally increases as the number of habitat specialists increases. The number and diversity of specific habitat resources such as tree hollows, mistletoe and woody debris is important for habitat specialists and consequently species richness. The composition of bird assemblages in eucalypt formations is dynamic, with many species tracking the availability of seasonal resources through the landscape. Bird assemblages in eucalypt formations are also related to time since fire. So far, there is little evidence that biotic interactions are involved in explanations of bird species richness in the eucalypt formations except where the vegetation has been fragmented or degraded by human activities allowing opportunistic species to dominate.

The distribution of four primary environmental resources interacting with landscape and ecological processes over multiple spatial and temporal scales, underlie the spatial and temporal patterning of the amount, variety and stability of habitat resources. A model that characterises habitat in terms of two abiotic axes which represent spatial and temporal heterogeneity in habitat resources and a third axis of biotic interactions is found to be a useful theoretical framework for understanding bird-habitat relationships, particularly in a dynamic temporal situation. The habitat templet theory provides the framework for interpreting temporal patterns in bird assemblages in successional environments. Vegetation succession and the temporal changes that occur in faunal communities in successional environments, in particular in post-mining rehabilitation is the subject of the next chapter.

Chapter 4: Succession in post-mining rehabilitation

4.1 Introduction

The previous chapter concluded that the distribution of the four primary environmental resources underlie the spatial and temporal patterning of the amount, variety and stability of habitat resources. It was also concluded that the habitat templet theory which characterises habitat in terms of the spatial and temporal heterogeneity of habitat resources is a useful framework for understanding bird-habitat relationships, particularly within a dynamic temporal context.

In this chapter, the research literature on bird response to successional environments is reviewed with reference to the habitat templet theory. A number of factors are found to affect the habitat value of rehabilitation plantings. A framework of these factors is developed and previous research on birds in post-mining rehabilitation is reviewed with reference to this framework. This chapter concludes with a discussion of some of the methodological, definitional and interpretational problems that are found in the research literature and by identifying some limitations of previous research.

Given the importance of vegetation structure in defining bird assemblages and the interest of this study in understanding how vegetation in post-mining rehabilitation may develop over time, this chapter begins by briefly outlining vegetation succession and discussing it in the context of mine rehabilitation. This is provided as background and is not intended to be a comprehensive review of the research literature on vegetation dynamics, disturbance theory or ecosystem development.

4.2 Vegetation succession

The ecosystem concept provides the broad framework for restoration ecology and the theoretical basis for predicting change in human modified ecosystems (Ehrenfeld and Toth, 1997), although the picture is still far from complete. In turn, understanding succession has been central to the developing concept of the ecosystem. Early research on succession led to a view of ecosystems as analogous to organisms (Clements, 1916). This classical concept of vegetation succession suggested that a single, static climax state exists for late successional vegetation. This work was important in identifying patterns and processes of ecosystem change. Clements' organismic concept of the ecosystem, however, made a number of assumptions that subsequent ecological research and modelling have placed within a broader framework (Walker and Del Moral, 2003).

Current ecological thinking assumes that systems: (1) can be open; (2) are regulated by external processes arising outside their boundaries as well as internal processes; (3) exhibit multiple equilibria or end points; (4) have multiple and probabilistic successions; (5) are subject to natural disturbances; and (6) incorporate humans and their effects (Pickett and Parker, 1994; Paine, 2002). This thinking has resulted in the recognition that there may be considerable and persistent heterogeneity in late successional vegetation (Pickett and Cadenasso, 2005).

Succession is defined as the change that occurs in the species composition and/or structure of natural communities and their associated substrates over time (Walker *et al.*, 2007; Pickett and Cadenasso, 2005). It occurs with different rates and patterns in all natural communities in response to disturbance, which can vary in type, intensity and extent from the small gaps created by the death of a single individual to the bare substrates created by volcanic eruptions. It includes primary succession and secondary succession. Primary succession refers to the succcessions that occur on severely damaged or new substrates where there is no biological legacy from previous vegetation. Secondary succession refers to succession that occurs on intact substrates after some or all of the plants have been removed or killed but without eliminating the physical and biological changes to the substrate that resulted from the plant occupation (Huston, 1994; Walker *et al.*, 2007).

The drivers of ecological succession are grouped into two categories. In autogenic succession, change in the relative abundance of species is due to changes in the environment caused primarily by the organisms themselves. In allogenic succession, change in the relative abundance of species is due to changes in the environment driven primarily by external factors. The relative importance of allogenic and autogenic processes throughout the succession depends on the disturbance regime (Huston, 1994). The relative contributions of allogenic and autogenic processes are also likely to differ with the successional stage, the environmental context and the characteristics of the component species.

Early models described vegetation succession in terms of a sequence of discrete seral stages, or seres, in which groups of species successively appeared, altered the site conditions and then disappeared, to be replaced by another group of species. Egler (1954) termed this the 'relay-floristics' model. However, succession is currently understood to be a process of gradual alteration that occurs as a consequence of individual species characteristics interacting in a variable environment (Whittaker, 1953; Gleason, 1926; Huston and Smith, 1987). Pickett and Cadenasso (2005) outline a framework in which three sets of processes interact during succession: (1) differential site availability; (2) differential species availability; and (3) differential species performance. The specific mechanisms that operate within each set of processes are aspects of the abiotic environment, plants, animals, and microbes and their interactions. Modelling and empirical research have identified a multitude of physiological and life history characteristics of plants that can affect species availability and performance, including: mechanisms or strategies of propagule dispersal, establishment and persistence;

individual longevity and phenology (Egler, 1954; Gross, 1987; Connell and Slatyer, 1977; Noble and Slatyer, 1979); and specific mechanisms of persistence in response to fire (Naveh, 1975; Noble and Slatyer, 1980; Cattelino *et al.*, 1979).

The successional pathway is contingent on its history including: initial floristic composition (Egler, 1954), the accidents of arrival of species at the site (Watt, 1947), the outcomes of plant interactions (Connell and Slatyer, 1977), and the nature of the connections to the surrounding landscape (Pickett and Parker, 1994). These interact with the various strategies of member plant species so that succession can be understood as a sequence of conditional probabilities (Ulanowicz, 1996). Each disturbance event, the resultant site characteristics and the specific characteristics of the assembled plants on the site can result in unique successional pathways. Exactly which factors dominate in a succession depends on the history of the site and the specific individuals that reach the site (Pickett and Cadenasso, 2005).

Understanding and predicting the rate, direction and nature of successional change in response to disturbance events or regimes, depends on both deterministic and stochastic processes working at different spatial and temporal scales and different levels of ecological organisation. It also requires understanding the linkages between organisational levels and how disturbance propagates through different levels of ecological organisation (Pickett *et al.*, 1989; Pickett *et al.*, 1987) (Table 4.1). Three broad types of trajectory and endpoints are envisaged: (1) single equilibrium end-points; (2) persistent non-equilibrium end-points; and (3) multiple stable states. Empirical evidence is available to support each of these models (Suding and Gross, 2006).

Table 4.1 Expression of disturbance at various levels of ecological organisation[1]

Speed / Scale	Level	Attributes / mechanisms of change	
		Structure	Process
Fast / small ↕ Slow large	Individual	Physiology Behaviour Size	Mortality Growth Reproduction
	Population	Density Structure (age, size, genetic)	Evolution Extinction
	Community	Diversity Species composition Functional groups	Coexistence Competition Mutualism Predation
	Ecosystem	Nutrient pools Productivity	Resistance Resilience Nutrient flux / retention
	Landscape	Disturbance Propagule pressure	Connectedness Colonisation
	Region	Temperature Precipitation	Climate change Pollution inputs

1. Redrawn after (Pickett *et al.*, 1989; Suding and Gross, 2006)

Understanding succession is central to our current concepts of ecosystems and how they interconnect to form landscapes. Although ecosystems are conventionally described in terms of their characteristic species composition, structure, and landscape context, the current concept of ecosystems also emphasises their dynamic nature. Allen and Hoekstra (1992:100), for example, argued that ecosystems are process oriented rather than spatially ordered, and are more powerfully seen as 'sequences of events rather than as things in a place'. In their view, the characteristic identity of ecosystems is achieved through cycling of resources, without which they would be directly constrained by the environment. Contingency is central to this concept.

Although ecosystems themselves do not evolve in the conventional sense of the term, they are nevertheless the result of evolutionary processes. Natural ecosystems result from repeated cycles of natural selection operating on the available pool of species, or propagules. Community assemblages are therefore not merely the result of random processes but evolutionary processes. Natural ecosystems take the forms they do because, through natural selection, their component species in their relative abundances have proven to have the best set of adaptations to the local conditions including interactions with other species. Thus, although natural ecosystems are dynamic, their characteristic patterns tend to be stable with respect to the environmental conditions and disturbance regimes in which they persist. Allen and Hoekstra (1992:104) used the analogy of the woodsman's axe, 'it is still his ax(e) after he has replaced the handle four times and the head twice'.

This brief outline has highlighted the dynamic nature of ecosystems, especially with respect to their taxonomic composition. As Mackey *et al*., (2008) noted, ecosystems are complex phenomena with properties that confound their identification, classification and mapping. Nevertheless, without making any assumptions about the level of ecosystem functioning and recognising that taxonomic composition is contingent on history, ecosystems are stable enough that their composition and structure can be described. Ecosystems can be named and classified, and their distributions mapped with some degree of useful reliability.

The composition and structure, or 'woodsman's axe', of the tropical eucalypt savannas of northern Australia is highly stable with respect to the highly seasonal soil moisture regime, low nutrient availability and frequent low intensity fires (Braithwaite, 1996). Vegetation composition and structure on the Weipa bauxite plateau is characterised by a five 'framework' plant species that dominate the canopy (see 4.2.1.1 Definitions of 'framework' and 'desirable' species). These framework species provide environmental stability, core habitat resources, and contribute substantially to the overall function of the ecosystem (Reddell and Hopkins, 1994). The distribution of the *Eucalyptus tetrodonta* open forest (Figure 2.6) demonstrates how close the relationship of this ecosystem is with the unique topographic and substrate conditions of the Weipa bauxite plateau.

4.2.1.1 Definitions of 'framework' and 'desirable' species

The concept of framework species is based on the understanding that the dominant species of an ecosystem are stable with respect to the environmental conditions and disturbance regimes in which they persist. Framework species dominate the vegetation canopy and contribute substantially to community structure and function. Characteristics of framework species for the Weipa bauxite plateau environment are: (1) relatively long life spans; (2) high 'resistance' to fire; (3) vegetative regeneration strategies in response to stress and disturbance; and (4) high predictability of growth performance and developmental stages (Reddell and Hopkins, 1994).

In the tropical savannas, *Eucalyptus* spp. and well established woody perennial species survive fires by protection of aerial parts and vegetative recovery (Lacey *et al.*, 1982). Many species, including all of the *Eucalyptus* spp., produce lignotubers. Some species that lack lignotubers are still able to produce shoots from buried portions of the mainstem, despite repeated burning. Vegetative reproduction from root suckering is common in *Eucalyptus tetrodonta* and *Erythrophleum chlorostachys* and also occurs in a number of understorey trees and shrubs including some *Acacia* species. There is little seedling recruitment due to annual drought and frequent fire, and vegetative spread is common. Sprouting species have an advantage over non-sprouting species that regenerate from seed as their roots already exert physical control over a site and they have access to water from deeper in the soil profile than seedlings (Reddell and Hopkins, 1994). Mature framework species also have an important role in regulating ground water levels and contribute significantly to organic matter inputs (CSIRO, 1997). The stability imparted by framework species extends to stable provision of vegetation based habitat resources for fauna.

For the purposes of this study, framework species of the Weipa bauxite plateau are the five framework species identified as such by Reddell and Hopkins (1994): *Eucalyptus tetrodonta*, *Corymbia nesophila*, *Corymbia stockeri* subsp. *peninsularis*, *Erythrophleum chlorostachys* and *Corymbia polycarpa*. No changes have been made to their definition except to update species names to reflect currently accepted nomenclature.

Reddell and Hopkins (1994) also identified six sub-canopy and understorey species on the Weipa Plateau that were 'desirable for the long term development of community structure' because of their significant contribution to community structure and function. For the purposes of this analysis, 'desirable' species for the Weipa bauxite plateau species are those species identified by Reddell and Hopkins (1994): *Acacia rothii*, *Grevillea glauca*, *Grevillea parallela*, *Lophostemon suaveolens*, *Parinari nonda* and *Planchonia careya*. One sub-canopy species not listed by Reddell and Hopkins (1994), *Xylomelum scottianum,* was added to the definition of desirable species for this analysis on the basis of its contribution to vegetation in native forest reference sites.

4.2.2 Succession and mine rehabilitation

Native ecosystems are increasingly being used as target or reference ecosystems for post-mining rehabilitation in Australia and elsewhere (Bell, 2001; Herath *et al.*, 2008). Rehabilitation that matches local reference ecosystems with respect to plant species composition and structure, ecological function and faunal populations is considered likely to be sustainable (Chambers *et al.*, 1994; Nichols, 2006).

As strip mining, including bauxite mining, involves complete removal of vegetation and major disturbance of the substrate, rehabilitation of post-mining landscapes is analogous to primary rather than secondary succession. Post-mining rehabilitation strategies that aim to rehabilitate native ecosystems attempt to address both the autogenic and allogenic components of primary succession, including the deliberate manipulation of natural processes to achieve specific outcomes. It is increasingly being recognised, however, that ecosystem development in novel or degraded ecosystems is not uni-directional. Multiple directional pathways and alternative stable states are possible, including stable degraded states, and the presence of system thresholds may be linked to the functional composition of the system (Suding *et al.*, 2004; Hobbs and Norton, 1996; Whisenant, 2002). Consequently, there is increasing recognition among ecologists of the need to understand how successional processes operate in each system and to acquire detailed knowledge of the ecological characteristics of target ecosystems (Palmer *et al.*, 1997; Hobbs *et al.*, 2007).

Driven by the social and regulatory imperative to demonstrate sustainability, there is also increasing emphasis on functional succession (i.e., re-establishment of the capability of the land to capture and retain energy, water, nutrients) (Bell, 2001; Aronson and Le Floc'h, 1996; Ehrenfeld and Toth, 1997; Cooke and Johnson, 2002; Hobbs, 2002; Whisenant, 2002; Jasper *et al.*, 2006). Not only do the ecological processes themselves need to be restored, but monitoring techniques are required that enable assessment of change in those processes. Assessing change in functionality is required to inform the development of successful rehabilitation strategies and for regulatory purposes (Nichols, 2006). Demonstration of the long term sustainability of mine rehabilitation is of major concern to the state government as ongoing liabilities for rehabilitation are transferred to the state on lease surrender (Environmental Protection Agency, 2008).

Establishing vegetation that is similar in structure to a suitable local reference native ecosystem requires that rehabilitation strategies following bauxite mining address three broad issues. First, the physical substrate constraints imposed by the mining process and local environmental conditions (Bradshaw, 1983). Second, the maintenance of long term plant productivity through establishment of nutrient conserving and cycling processes (Ward, 2000). Third, the establishment of a vegetation community that is self-sustaining and resilient to the local environment conditions and disturbance regime.

4.2.2.1 Physical constraints

There are no substrate toxicity issues associated with bauxite mining. The major substrate constraints on the Weipa bauxite plateau include: soil compaction and impeded water infiltration due to the mining process; depth to the water table due to landscape lowering; high gravel content due to mixing of subsoil and topsoil; loss of soil structure due to soil handling; loss of organic matter; and soil infertility (Schwenke *et al.*, 2000b; Reddell and Hopkins, 1994; Schwenke *et al.*, 2000a). Soil ripping and cultivation are widespread practices used to ameliorate soil compaction (Bradshaw, 1983) and to improve water infiltration, including at Weipa. Soil structure, organic matter and fertility issues are addressed by the addition of fertilisers and vegetation establishment.

4.2.2.2 Plant nutrients

A study of nutrient distribution in the *Eucalyptus tetrodonta* open forest on the Weipa bauxite plateau has found that most of the nitrogen, phosphorus and other plant nutrients are stored and cycled in the vegetation and organic matter (Schwenke, 1999). The nutrients held in the vegetation are lost when the forest is cleared and burnt in preparation for mining. Topsoil stripping and replacement and in some cases stockpiling can also cause losses of soil nitrogen through volatilisation, leaching and denitrification (Todd *et al.*, 2000). Large declines in surface soil organic matter, organic carbon and total nitrogen during rehabilitation operations have been recorded at Weipa (Schwenke *et al.*, 2000b). Mixing of topsoil and subsoil may exacerbate nutrient deficiencies in highly weathered soil such as at Weipa where plant nutrition is highly dependent on soil organic matter content. It has become standard practice in Australia, including at Weipa, to address the nutrient losses associated with vegetation clearing through the application of fertiliser and the establishment of fast growing nitrogen fixing understorey species at high densities (Todd *et al.*, 2000).

4.2.2.3 Establishment of self-sustaining, resilient vegetation

Establishment of vegetation similar to native ecosystems in post-mining rehabilitation relies on a combination of seed banks in the topsoil that is re-laid following mining, direct seeding and planting of tubestock. At Alcoa's bauxite mine in Western Australia the topsoil seed bank accounts for 77% of the species re-established (Ward *et al.*, 1997). Research by Alcoa has found that handling and accurate replacement of topsoil is important for preserving the soil seed bank (Ward *et al.*, 1997; Ward *et al.*, 1996; Koch *et al.*, 1996; Grant *et al.*, 1996). Use of fresh topsoil also assists soil nutrition by preserving soil microflora and fauna (Reddell and Hopkins, 1994). Rehabilitation at Weipa also uses seed banks in re-applied topsoil. This is supplemented with sown seed made up of a mix of species. Application of mulch containing canopy material (Bellairs and Bell, 1993), and hand planting of seedlings of species that do not regenerate well from seed, is also used at some mine rehabilitation sites (Herath *et al.*, 2008) although not at Weipa.

As fire is a frequent disturbance in the monsoonal eucalypt woodlands (see section 2.6), the establishment of vegetation that can survive fire is considered to be critical to successful rehabilitation in northern Australia (CSIRO, 1997; Reddell and Hopkins, 1994). Previous researchers have recommended that the critical success criterion for mine rehabilitation at Weipa is that 'framework' species should be established at densities within ranges derived from the closest analogue ecosystems (Reddell and Hopkins, 1994) (see section 4.2.1.1).

4.3 Bird assembly in successional environments

Faunal succession is expected to occur in response to the physical changes in habitat structure that occur in succession. That there are relationships between vegetation successional maturity and community attributes such as biomass, spatial heterogeneity and species life history has been recognised for some time (Margalef, 1963; Odum, 1969). Southwood (1977) proposed a theoretical framework that relates the temporal changes that occur in faunal assemblages to the spatial and temporal changes that occur in habitat during succession. According to his habitat templet theory, habitat can be described in two abiotic dimensions: an axis of frequency of disturbance (durational stability), and an axis of adversity (environmental favourableness). Through habitat, combinations of individual life-history strategies are selected that improve individual fitness in relation to the two intersecting abiotic gradients. A third dimension, biotic interactions, incorporates the relative degree to which competition contributes to a decrease in individual fitness (Southwood, 1988).

As habitat is characterised by the habitat templet theory, succession represents the movement of a particular location along a gradient of durational stability while environmental favourableness remains unchanged (Southwood, 1977). The length of time that a particular habitat structure persists for, its durational stability, is determined by the rate of turnover in individual plant structures (Brown, 1991). In turn, the rate of turnover in individual plant structures is a result of turnover or change in dominance of plant species and associated changes in growth rates and longevity. The establishment of woody species is particularly important. As the durational stability of the habitat increases, biomass and spatial complexity also increase, resulting in an increase in the variety of niches (Brown, 1991; Southwood, 1977)

The axis of durational stability relates the number of fauna generations that occur in a particular location to the length of time that the particular habitat structure required for breeding exists and the frequency with which new similar habitats arise. As a result of trade-offs between adaptations for survival and reproduction, the habitat templet theory predicts that vegetation succession will reflect the ecological adaptations of the successive fauna species. Fauna associated with early successional habitats of short durational stability are characterised as small, mobile, opportunistic, and short-lived. As the permanency of habitat structures increases, these fauna will progressively be replaced by larger, more sedentary, specialised, and long-lived species (Southwood, 1977). Associated with the increased biomass and variety of niches, and the changes that are expected in faunal species' life-history attributes with increasing durational stability, a number of characteristics of faunal assemblages are expected to change. Niche-breadth should progressively narrow, trophic complexity should increase and the rate of faunal species turnover should gradually slow. As a consequence of these changes, an increase in faunal abundance and species richness is expected, and turnover in species composition leading to changes in the structure of faunal assemblages (May, 1982).

4.3.1 Bird species richness

Many empirical studies have confirmed that bird species richness and abundance increase as the structural complexity of vegetation increase (Nichols and Nichols, 2003; Armstrong and Nichols, 2000; Karr, 1968; Kavanagh *et al.*, 1985; Smith, 1985; Martin *et al.*, 2004; Passell, 2000; Laiolo *et al.*, 2004; Venier and Pearce, 2005; Loyn, 1985a; Willson, 1974; May, 1982). Some studies have found that species richness was highest in mature ecosystems compared to earlier successional stages (May, 1982; Martin *et al.*, 2004; Taylor *et al.*, 1997). In contrast, Fisher (2001) found a decrease in bird species richness and abundance with time since disturbance. He accounted for this by the low productivity of the site and suggested that, in this case, species richness was increased by disturbances that increased habitat heterogeneity in early regeneration. Willson (1974) found that the increase in bird species richness is primarily due to the addition of foraging guilds with new vegetation layers. May (1982), however, found that increases in species richness during secondary forest succession occur due to both addition and expansion of foraging guilds.

4.3.2 Turnover in bird species composition

Rather than a simple increase in the number of co-occurring species, species turnover is expected as habitat changes, with rapid turnover in early succession gradually slowing in late succession as the durational stability of habitat structures increases. Few studies have related rates of species turnover to successional stages. However, bird species turnover along successional gradients has been demonstrated in a range of environments with a number of studies identifying specific assemblages associated with different successional stages or age classes (Nichols and Nichols, 2003; Fisher, 2001; Sallabanks *et al.*, 2006; Smith, 1985; Loyn, 1985a; Taylor *et al.*, 1997; Martin *et al.*, 2004). Some species are found across all or many stages of a succession (Martin *et al.*, 2004; Fisher, 2001). For other species, the time at which they return is related to the presence of key resources such as food plants, tree hollows, mature eucalypts and leaf litter (Smith, 1985; Loyn, 1985a). Venier *et al.*, (2005) found in a secondary forest succession that half of the bird species were significant indicators of specific age classes and that there is an increase in specialisation through succession.

A number of empirical studies have found that some species are exclusively associated with old growth forest and do not return to even old successional forest (Taylor *et al.*, 1997; Sallabanks *et al.*, 2006; Barlow *et al.*, 2007). Loyn (1985a), for example, found that powerful owls *Ninox strenua*, sooty owls *Tyto tenebricosa*, and australian owlet-nightjars *Aegotheles cristatus* only occurred in Mountain Ash forest containing trees aged 200 years or more. A study of birds in habitat reconstruction at Gunnedah recorded no bark or log foraging species outside of remnant woodlands (Martin *et al.*, 2004). In contrast, Kavanagh *et al.*, (1985) found that no bird species were excluded from coupes that had been logged, although the diversity of birds in logged coupes was maintained by the proximity of unlogged forest. Martin *et al.*, (2004) analysed birds according to their conservation status and found that although habitat reconstruction sites were used by four declining species, a further four declining species were exclusively recorded in remnant woodland.

Turnover in species composition is an ongoing process that occurs over a range of spatial and temporal scales (Wiens, 1989b). It occurs in response to environmental variation within a site over a gradient of time frames, from seasonal variation in resource availability to long term vegetation change. Species turnover also occurs in response to factors external to a site (Willson, 1974; Wiens, 1989b). Within-site species turnover for example, may reflect demographic changes in meta-populations due to regional scale factors. Bird succession in long term studies has been shown to be correlated with landscape scale change in vegetation cover (Wiens, 1989b).

Vegetation complexity generally increases with age, but age *per se* is not necessarily a good predictor of development. Disturbance can disrupt community assembly (Mouquet *et al.*, 2003), particularly if it alters vegetation structure. Green and Catterall (1998) found little evidence that forest dependent birds were using regenerating areas. They suggested that this was due to ongoing disturbance in regenerating areas which virtually removed the understorey (Green and Catterall, 1998). A study of birds in secondary forest succession in Italy, found that management regime was a better predictor of bird species richness, abundance and diversity than age (Laiolo *et al.*, 2004). In this case it was due to removal of standing dead wood, the variable that was most strongly related to bird occurrence. Studies of birds in mine rehabilitation following bauxite mining have found that rehabilitation technique was more important for bird species richness than age of rehabilitation (Nichols and Watkins, 1984; Collins *et al.*, 1985; Armstrong and Nichols, 2000).

4.3.3 Changes in bird community structure

Empirical studies support the prediction that the niche-breadth of species narrows along successional gradients, with generalists predominating in early succession and specialists predominating in late succession as the durational stability of habitats increases. May (1982) demonstrated increases in both the number and the richness of bird specialist foraging guilds with secondary forest succession. In another study of secondary forest succession, Davis *et al.*, (2000) found that generalist foraging bird species, particularly foliage gleaning insectivores, are more common in early succession and foraging specialists are more common in later succession. Shifts in species and foraging guilds were related to fire frequency and the associated changes in vegetation structure (Davis *et al.*, 2000). A study of bird communities on surface mined lands found that as the height and diameter of trees increased, there is a gradual decrease in omnivorous and granivorous species and a corresponding increase in insectivorous species (Brenner and Kelly, 1981).

4.4 Habitat value of rehabilitation plantings

Although advances in understanding of vegetation succession have been incorporated into restoration ecology, most rehabilitation strategies are passive with respect to fauna. The role of vegetation change in faunal succession following disturbance has been demonstrated for various taxonomic groups including birds, lizards (Twigg and Fox, 1991; Taylor and Fox, 2001), ants, and mammals (Fox, 1990; Fox, 1996). As outlined in the previous chapter, however, other factors are also potentially involved in the explanation of temporal patterns in bird assemblages.

Ultimately, the species composition of a local community is determined by the addition of species through successful colonisation and establishment of breeding populations, and by the loss of species through local extinction (Wilson, 1992). The regional species pool limits the development of local communities to the number of potentially colonising species (Mouquet *et al.*, 2003; Wiens, 1989a) and species richness is a function of local and regional processes whose relative influences vary over time.

Isolation and area both have an important role in determining rates of species addition and loss. The larger the area, the more likely it is that dispersing individuals will encounter it. No area, however, simply receives a random influx of animals in proportion to its size and degree of isolation. Superimposed on these variables are the colonisation characteristics of the animals (Majer, 1989) and the permeability of the landscape (Hobbs, 2002). In the context of an island archipelago, Diamond (1975:371) documented that some bird species with strong flying ability had an 'insuperable psychological barrier to crossing water'. Presumably even terrestrial landscapes with intact vegetation cover can filter out species if they do not provide suitable protection or if key resources are not available.

Diamond (1975) derived rules of assembly for bird communities from observations of patterns of bird species distributions on islands. He showed that the incidence of bird species on any particular island depended on their dispersal abilities, their reproductive potential and their habitat-specificity (Diamond, 1975). A species' dispersal ability combined with distance from a source population is thought to be the primary determinant of whether a species will colonise a given area (Wiens, 1989a). For species with equal dispersal ability, population size in the source population affects the rate at which individuals will disperse. Therefore, rare species are less likely to contribute to the assembly of a local community than common species of equivalent dispersal ability (Wiens, 1989a).

Arriving, however, is not sufficient. Presence of one or more individuals does not indicate presence of a functionally significant demographic unit, or population (Smallwood, 2001). A species can only be considered a *bona fide* member of a local community if it is established as a population (Wiens, 1989b). Several factors influence the likelihood that populations will be established. Diamond (1975) showed that after arrival on an island, the size of the island affected bird incidence in ways that varied from species to species depending on: (1) whether the required habitat of a species occurs on small islands; (2) the minimum territory size for species in which each pair maintains an exclusive territory; (3) the minimum area required to supply resources that were patchy or seasonal; (4) population size in relation to short term and long term population fluctuations; and (5) the role of resource hotspots. Biotic interactions are also important and may be changed by the order in which species arrive if there are 'priority effects' as proposed by Diamond (1975). The presence of a predator may reduce the number of colonist individuals to such a low level that a persistent population does not become established.

These factors all operate to degrees that differ between species to determine which species will enter and remain in a community during its assembly. Wiens (1989a) noted that assembly is an ongoing process for two reasons. First, most communities are subjected to a continuing stream of potential colonists. Second, species may be removed from the community through local extinction. Changes in habitat within the area may render it unsuitable for some species whereas other species may suffer the effects of interactions with predators or competitors that develop after their initial establishment. Wiens (1989a) concluded that it is unlikely that any single factor plays an overwhelming role in determining bird community assembly in most situations.

Consequently, a number of factors need to be considered in addition to establishment of vegetation, if rehabilitation is to function effectively as habitat (Figure 4.1). First, rehabilitation projects should consider the landscape context of the rehabilitation including landscape permeability (Hobbs, 2002), area of rehabilitation (Fink *et al.*, 2008), distance to source populations (Scott *et al.*, 2001; Miller and Hobbs, 2007), species dispersal ability (Majer, 1989; Diamond, 1975; Wiens, 1989a), and population viability (George and Zack, 2001; Smallwood, 2001; Vesk and Mac Nally, 2006). Second, rehabilitation projects need to consider organism specific habitat requirements (Miller and Hobbs, 2007; Vesk and Mac Nally, 2006) including biotic interactions and availability of resources through time (Southwood, 1977).

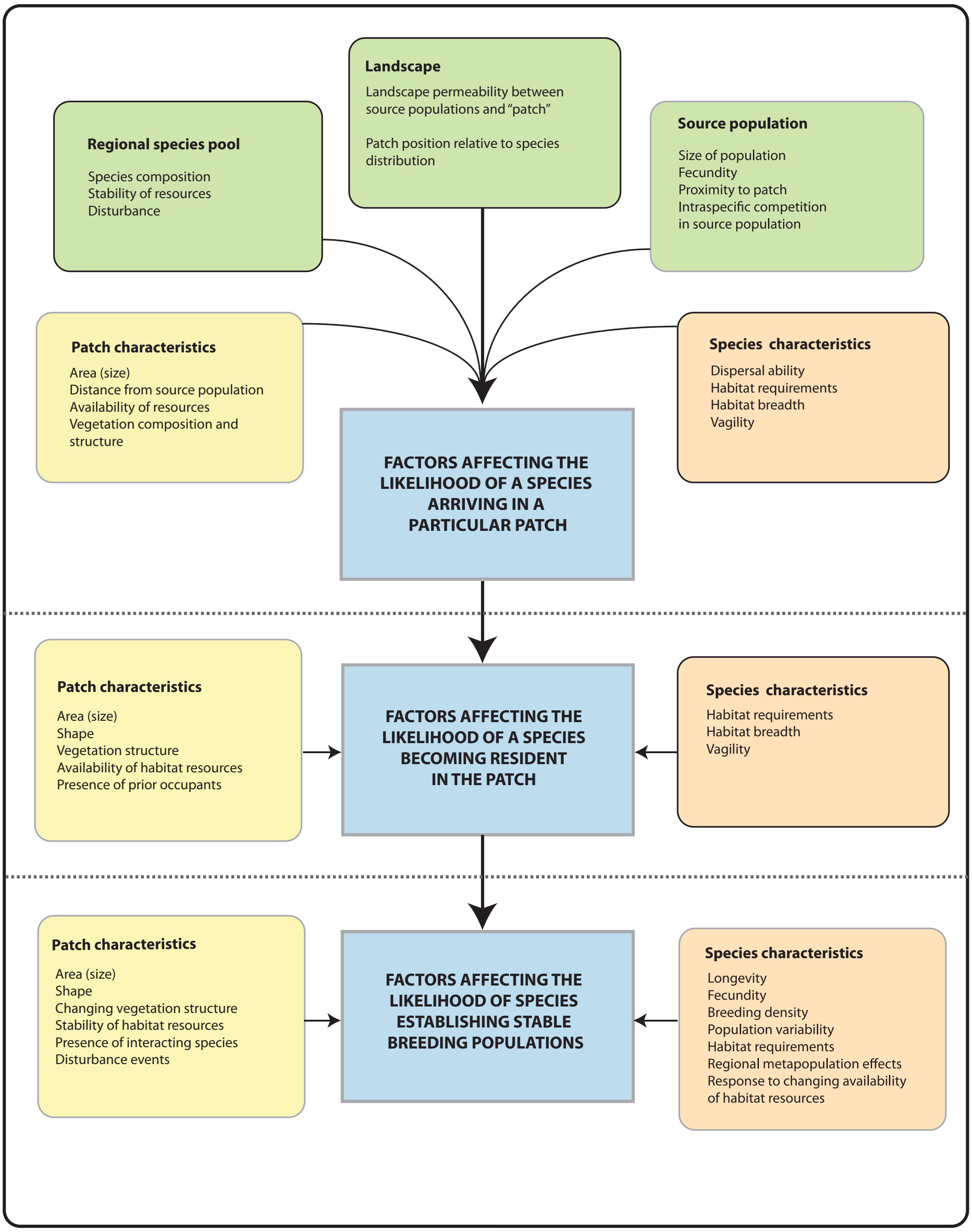

Figure 4.1 Factors influencing the effectiveness of rehabilitation as habitat

According to the habitat templet theory species characteristics and habitat characteristics are related.

4.5 Bird succession in post-mining rehabilitation

Bird succession was monitored by Alcoa in Western Australia in a long term study of rehabilitation following bauxite mining. Rehabilitation sites support similar bird species richness and total bird abundance to forest control sites although there are differences in bird species composition (Collins *et al.*, 1985). Compositional similarity increases over time (Armstrong and Nichols, 2000). Four to five year old rehabilitation sites have 65% (Nichols and Watkins, 1984) and 16 to 17 year old rehabilitation sites have 73.5% of the bird species recorded in unmined forest sites (Armstrong and Nichols, 2000). The absence of some of the missing bird species after 17 years is attributed to the absence of specific habitat resources that are only present in mature forest including tree hollows and fallen logs from mine rehabilitation. A number of bird species also occur exclusively in mine rehabilitation sites. Comparison of the species' densities suggests that there are significant differences in the occupancy rates of birds in the different treatments.

Variation in bird assemblages over time at Alcoa's Western Australia mine (Armstrong and Nichols, 2000) is attributed to successional development in the mine rehabilitation sites. The density of insectivorous species increases with time and the pattern of bird succession varies with rehabilitation strategy. It was concluded that vegetation development controls the composition of bird communities in rehabilitated mine sites (Armstrong and Nichols, 2000). Their study used low levels of replication, with only two healthy forest control sites, two dieback forest sites and three post-mining rehabilitation sites. Different rehabilitation techniques had been applied to each of the post-mining rehabilitation sites. The native forest 'control' sites had previously been logged so may already also be missing species dependent on habitat resources associated with mature vegetation. Low levels of repeat sampling also mean that presence/absence data was emphasised at the expense of site bird occupancy data.

Another long term fauna monitoring program using a different set of survey sites was commenced in 1991 at Alcoa's bauxite mine (Nichols and Nichols, 2003). In this study, birds, mammals, reptiles and ants were first surveyed in 1992 when the rehabilitation sites were two years old and subsequently in 1995, 1998 and 2001. After 8 years, bird species richness and diversity in rehabilitation sites is similar to unmined forest. By 11 years, bird species composition in rehabilitating mine sites is indistinguishable from stream zones in Jarrah forest that contained tall dense understorey (Nichols and Grant, 2007). Combining all studies, 95% of the 70 upland jarrah forest bird species had been recorded in mine rehabilitation. Species missing from mine rehabilitation included the trunk-gleaning insectivore, rufous treecreeper *Climacteris rufa*.

An assessment of the suitability of rehabilitated bauxite mine sites for bird breeding found that 21 species forage in the mine site while breeding (Curry and Nichols, 1986). Of these, 12 nest or rear fledglings in the rehabilitating sites while the remaining 9 species breed in the adjacent forest. The species that nest in nearby forest but not in rehabilitation all require mature, senescent or dead trees for nesting sites. It was concluded that the birds that nest in the mine sites represent virtually all of the locally occurring species that could be expected to find suitable nest sites in early stages of regrowth (Curry and Nichols, 1986). The significance of these results is not clear, however, as sampling effort was not standardised, there was little replication of sites with similar attributes and there is no mention of the proximity of the mine sites to native forest sites. Without the inclusion of landscape variables such as distance from remnant vegetation it is difficult to interpret how the results of these studies might be applied to other areas. A review of faunal response to rehabilitation plantings in agricultural areas found that few studies measured landscape level attributes (Munro *et al.*, 2007).

The effect of distance from remnant vegetation on bird use and occupancy of rehabilitating areas is a potentially important explanatory variable that, had it been incorporated into the research design would make their results more general. The configuration of remnant and mined areas at Alcoa's bauxite mining operation in Western Australia is very different from the Weipa bauxite mine. At the Alcoa mine, the average mine pit size is 20 ha and each pit is surrounded by native vegetation. In contrast, bauxite mining at Weipa is a progressive strip mining operation in which very little remnant vegetation is retained. Source populations for recolonising or visiting birds are therefore likely to be in much closer proximity to regeneration areas at the Alcoa mine than they are at Weipa.

4.5.1 Birds in post-mining rehabilitation at Weipa

Two unpublished studies have been conducted of vertebrate fauna in rehabilitating mine sites at Weipa. The first study, which was conducted in 1981 to 1982, aimed to establish an inventory of fauna and assess the extent to which fauna used mine rehabilitation and how this was influenced by the vegetation (Reeders and Morton, 1983). The sampling design used was appropriate for an inventory study but was not suitable for quantitative assessment of faunal use of mine rehabilitation. Rigorous assessment of the extent to which fauna used mine rehabilitation, requires comparable, quantitative data obtained from both mine rehabilitation and reference sites. However standardised sampling procedures were not used, survey sites were not of equal size or shape and sometimes included more than one habitat type. Only one 'control' site was surveyed, which was an island of native vegetation within the mine, and it was surveyed during just one of the three sampling periods. Thus, the baseline data for the reference faunal assemblages are incomplete. Their comparison with the local native forest species was based primarily on a species list compiled from reports of fauna surveys conducted by different researchers using different survey methods.

Surveys were conducted over three different sampling periods, December 1981, April 1982 and September 1982 to capture seasonal variation (Reeders and Morton, 1983). Seasonal variation was confounded by site type, however, because a different set of sites was surveyed during each sampling period. Finally, the post-mining rehabilitation survey sites incorporated a range of vegetation types including mixed species forest and woodland, pasture, pine plantation and native forest regrowth in an area that was cleared but not mined (Reeders and Morton, 1983). The mixed species forest and woodland makes up the majority of the mine rehabilitation by area and is the only rehabilitation type that specifically aims to provide habitat for fauna.

Examination of Reeder and Morton's appendices shows that although a number of bird species were recorded in the mine rehabilitation, most of them were probably using the regeneration opportunistically. Of the 103 species recorded, 12 species were simply flying overhead. Of the 91 species remaining after removing the overhead records, 19 were waterbirds that were recorded at an artificial lake or in flooded areas of the mine. Of the 72 bird species remaining after removing records of the waterbirds, 32 bird species were uncommon and 22 were infrequent. Fifteen bird species were common and nests of eight species were found. Only two species, rainbow lorikeet and golden-headed cisticola were abundant in mine rehabilitation sites (Reeders and Morton, 1983). It was concluded that 10 species were resident in mine rehabilitation. Of these, seven species were shared with the *Eucalyptus tetrodonta* open forest. Reeders and Morton (1983:74) claimed that their survey demonstrated that mine rehabilitation had 'created a habitat that provided food, shelter and protection for a majority of the local open forest species'. However, their claim is not supported by their survey design or by examination of their data.

A second study of vertebrate fauna in rehabilitated mine areas at Weipa was conducted from December 1995 to June 1997 (Winter and Alford, 1999). A stratified random design was used that covered a chronosequence of 60 mine rehabilitation sites but no native forest reference sites. In this study, a standardised survey procedure was used and all survey sites had been rehabilitated with the aim of establishing 'self-sustaining and maintenance free vegetation comprising a variety of native plants which in turn support native fauna'. Site variables measured were: vegetation structure, floristics, proximity to remnant *Eucalyptus tetrodonta* open forest, proximity to mesic habitats, season and fire. Four 15 minute surveys were conducted at each site over different seasons although site size was not standardised.

The composition and abundance of reference native forest fauna was estimated from a survey of terrestrial vertebrate fauna conducted by different researchers but using similar procedures (Winter and Alford, 1999). The data on reference faunal assemblages was collected in one sampling period, during the 1996 wet season by surveying 11 woodland sites twice for fifteen minutes. Total bird survey time in woodland sites was 5.5 hours compared to 60 hours of systematic bird survey effort in regenerated mine sites. Because of differences in sampling periods and search effort, only presence / absence data was used in the analysis.

Winter and Alford (1999) also compared the data from mine rehabilitation with a regional species list. By excluding species that were considered to be unavailable for colonisation, or inadequately sampled, they concluded that only three birds that are abundant in woodlands were absent from mine rehabilitation (Winter and Alford, 1999), emu *Dromaius novaehollandiae*, varied sittella *Daphoenositta chrysoptera*, and red-tailed black cockatoo *Calyptorhynchus banksii*. A further 17 woodland species were under-represented in mine rehabilitation. The authors concluded that 'a high proportion of the available regional fauna is successfully colonising regeneration' (Winter and Alford, 1999:68). It is not clear what criteria they applied to define successful colonisation as only one species was recorded breeding in regeneration.

These two studies of vertebrate fauna in post-mining rehabilitation at Weipa did not provide data that enables a quantitative understanding of the extent to which post-mining rehabilitation at Weipa provides habitat for native forest birds. The need remains to systematically and quantitatively compare bird assemblages in post-mining rehabilitation sites with native forest reference sites.

4.6 Limitations of previous research

4.6.1 Research design

Landscape scale studies are complex and implementing ideal research design can be impractical if not logistically impossible. Synchronous sampling over a large number of research sites while controlling for variability between observers, for example, is impossible. Lack of research funding and support also make it difficult to conduct long term or large scale research. Given these difficulties, it is inevitable that individual studies will be limited and open to critique. Nevertheless, an understanding of some of the issues with research design could be used to improve future studies. Some of the issues are:-

1. Spatial control: Assessment of change in response to landscape modification requires reference to appropriate intact landscapes. Many studies lack adequate spatial control although this is probably due in many cases to the absence of suitable reference landscapes.
2. Replication: When the effects of multiple factors are being analysed such as area, habitat diversity and isolation, adequate replication is required. Many studies are limited by inadequate spatial replication, although preliminary assessment of what level of replication is adequate can use up valuable research resources.
3. Design variables: As the two literature review chapters document, a large range of factors can affect the outcome of research. Lack of control or measurement of important variables can confound results or lead to things appearing to be significant that are actually not the most meaningful explanatory variables (Wiens, 1989a). It also reduces the ability to draw any clear conclusions. Landscape context, for example, is known to be an important contributing factor to bird occupancy and, therefore, needs to be

included as a design variable or at least measured so that bird occupancy in habitat remnants or rehabilitating areas can be analysed in a meaningful way and compared between studies.

4. Framing: To detect a response to a particular environmental gradient requires that sufficient variation along that gradient is incorporated into the research / sampling design. For example, to adequately test for the effects of isolation requires that the sites sampled cover a meaningful range along a gradient of isolation values.

4.6.2 Definitions and inconsistencies in terminology

4.6.2.1 The definition of species richness

Three scales of species richness have been identified that relate to different spatial scales and incorporate different levels of environmental heterogeneity (Bohning-Gaese, 1997; Cody, 1975). The number of species within a particular habitat type is referred to as alpha-diversity. Change in species richness relative to differences between habitat types is referred to as the beta-diversity or species turnover rate. Regional species diversity, referred to as epsilon or gamma diversity, is a measure of geographic species turnover and recognises that similar habitats within different regions support different species for historical reasons. However, many researchers do not specify the level of species richness that is being reported and match it with the scale of the environmental gradients measured. Furthermore, some researchers confuse the term species diversity with species richness (Hurlbert, 1971). Species richness is a simple count of the number of species, whereas species diversity incorporates a measure of the equality of species abundance (Krebs, 1997).

Erdelen (1984) found that foliage height diversity, for example, was correlated with bird species diversity only when different vegetation types were included in the analysis. It was not significant when forested areas alone were analysed. That is, foliage height diversity accounted for variation between different vegetation types (beta-diversity), but not within habitats of similar type (alpha-diversity). As species richness is also known to vary with time, area and environmental heterogeneity over a range of scales, separating out the effects of within habitat vegetation complexity on species richness requires that the level of species richness, the spatio-temporal scale and the level of environmental heterogeneity are all clearly defined.

Species richness increases as a function of time due to: (1) the effects of increased sampling effort; (2) species turnover in ecological time; and (3) speciation over long time scales (White, 2007). When comparing species richness then, the temporal extent of species richness values should be standardized. However, species richness in many of the large scale studies is simply defined as the long term cumulative list of all species ever recorded in a locality and may include vagrants that are not normally part of the community.

Implicit in much of the literature is that species richness is a static, spatially fixed attribute. However, species richness is an aggregate measure of multiple species' distributions. Each species' distribution is determined by the overlap between its unique requirements, and the environment. The distribution of each species is dynamic over a range of scales, particularly in the case of birds which are highly mobile. Although bird species differ in mobility, individuals and populations move around to meet their needs and numbers fluctuate. Finally, the landscapes that birds occupy are dynamic, made up of multiple, continuous environmental gradients intersecting to create an environment that is heterogeneous over a range of spatio-temporal scales. Bird species richness then should not be seen as a static, spatially fixed property of places. Rather, bird species richness is a dynamic phenomenon that varies spatially and temporally over a range of scales. When reporting species richness, therefore, there is a need to define the spatio-temporal frame that has been cast around it and to contextualize it in relation to prevailing environmental conditions and the disturbance regime.

4.7 Conclusions and relevance to this study

This and the previous literature review chapter have provided the context for understanding the multi-scaled dynamic nature of the exogenous factors that are involved in explanations of bird species distributions, particularly in successional environments. Habitat characteristics interact with species characteristics, and these occur within a landscape context that determines the rates of species addition and loss. The assembly of bird communities is an ongoing process. A number of hypotheses have been proposed to explain the patterns in bird species richness and bird community structure supported by a large body of empirical research. These two chapters have outlined some of these hypotheses and the empirical evidence for them, particularly in relation to birds in the tropical *Eucalyptus* woodlands. However, the dynamism and complexity of the patterns mean that the extent to which specific causal mechanisms operate in different ecosystems remains unquantified.

Bird distributions can be understood in terms of two intersecting hierarchies. In the first, large scale abiotic inputs interacting with landscape and small scale biotic driven processes drive increasingly fine-scaled spatial heterogeneity. In the second, the distribution of individual species is a scale-dependent hierarchical process. Interacting factors operate at multiple scales, and higher levels impose constraints upon lower levels. For the purposes of this research, some of the high level factors, such as climate and regional species pool will impose the same set of constraints on both post-mining rehabilitation sites and native forest reference sites. The exogenous factors that are most likely to contribute to differences in the bird assemblages therefore are differences in vegetation composition and structure. Landscape context variables, such as amount of natural vegetation and distance from natural vegetation are also expected to be related to differences in bird assemblages.

These two review chapters and the previous chapter describing the environmental context of the study have identified a number of issues that need to be accounted for in designing a comparative study of birds and their habitat. This study built on the previous work that has been conducted by incorporating knowledge of environmental factors that have been demonstrated to be related to bird species richness in the research design. In terms of research design, this study incorporated a number of elements that have not been used together in any previous assessments of bird response to succession in post-mining rehabilitation. These are outlined in the following chapter, Research design and methods.

Chapter 5: Research design and methods

5.1 Introduction

The aim of this study was to assess to what extent post-mining rehabilitation on the Weipa bauxite plateau has restored the bird habitat values of the *Eucalyptus tetrodonta* open forest. There are both spatial and temporal dimensions to approaching this question. The spatial dimension requires the acquisition of cross-sectional data on native forest reference sites and mine rehabilitation sites (i.e., 'synchronous' comparison between sites). The temporal dimension necessitates the acquisition of time-series (longitudinal) data that samples the changes in the vegetation community and bird assemblages in developing mine rehabilitation. Two broad approaches are available to the acquisition of time-series data for developing ecosystems. First, repeated measure studies in which the same sites are sampled repeatedly over time. The second approach substitutes space for time by 'synchronously' sampling sites of different age. In this study, space is substituted for time by sampling a chronosequence of mine rehabilitation sites. As an observational study, emphasis is placed on developing an appropriate sampling strategy. This chapter provides an overview of the study and outlines the research design and sampling procedures used. Potential problems with the research design and methods are identified, along with the strategies used for overcoming these. The chapter concludes by outlining the original methodological contribution made by this study.

5.2 Assumptions of the research design

Based on the literature review and the environmental context of the mining operation (i.e., the mine being surrounded by continuous intact native vegetation cover), I assumed that site vegetation structure would be the primary determinant of bird assemblages. The two literature review chapters outlined a number of additional environmental factors that affect bird species distributions and abundance over different spatial and temporal scales. I also assumed that native forest sites and mine rehabilitation sites would be equally affected by factors that operate over large spatial and temporal scales including: climate, regional biodiversity, historical influence, and disturbances that affect populations outside the study region. That is, the same pool of bird species was expected to be equally available to both native forest sites and mine rehabilitation sites. Any differences found in bird assemblages between sites should therefore be due primarily to variation in the vegetation structure between the study sites and to differences in the local landscape context of each site.

Landscape studies of birds have identified a range of variables other than vegetation structure that could also potentially affect bird distributions and abundance including patch configuration, edge effects, grazing, time since isolation, patch size and connectivity

(Woinarski, 1990; Cooper *et al.*, 2002; Major *et al.*, 2001; Recher and Serventy, 1991). These were assumed to have little applicability to this study as they relate to highly fragmented landscapes in which remnant vegetation occupies less than 20% of the landscape.

5.3 Research design

To compare bird assemblages and bird habitat, an adequate number of independent and representative sites must be sampled. The sampling strategy must capture inherent spatial variability and in the case of birds, temporal variability. As the amount of spatial variability was unknown, the minimum adequate number of research sites required to obtain accurate estimates of error was also unknown. In the absence of such prior knowledge, the number of research sites was maximised within practical constraints imposed by time, accessibility, resources and the sampling methods. An overview of the study is outlined in Table 5.1.

Table 5.1 Study overview

Stage 1	Develop research questions
Stage 2	Select and stratify design variables • Native forest sites - land unit / vegetation composition / vegetation structure • Mine rehabilitation sites - treatment / age / block size / distance from remnant vegetation
Stage 3	Select sites • 36 native forest sites - 28 pre-mining and 8 post-mining landscape analogue • 31 mine rehabilitation sites - 1 to 23 old
Stage 4	Collect vegetation data once at each site using a PCQ sampling method at 16 systematically located sampling points per site. All data was collected during May-June 2007.
	Collect landscape functionality data once at each site at 2 or 3 systematically located transects per stie using a standard procedure. All data was collected during June-July 2007.
	Collect bird data in 8 surveys per site over 16 months from September 2006 to December 2007 using a standardised 2 ha area search method. Each round of site visits was conducted 'synchronously' for all sites.

5.3.1 Grain and extent of the study

Landscape functionality, vegetation composition and structure and bird assemblages were sampled at 67 study sites across an area 45 km east to west, and 27 km north to south, an area of 1,215 km^2 (Figure 5.1). Thirty-six sites were *Eucalyptus tetrodonta* native forest reference sites, and 31 were mine rehabilitation sites. Study sites were 2 ha (100 m x 200 m). The size and shape of the research sites was dictated by the bird survey method. Vegetation and landscape functionality were sampled once at each site. To account for temporal variability, birds were sampled eight times at each site over sixteen months.

There are number of potential sources of error associated with this sampling design:

1. Underestimating the inherent variability within vegetation,
2. Non-representativeness of the study sites,
3. Differences in successional patterns between chronosequence sites,
4. Non-independence of sites and of bird observations,
5. Temporal effects on birds, vegetation and landscape function measurements, and
6. Systematic error introduced through sampling methods.

The following section outlines the strategies that were employed in the sampling design to reduce or avoid these potential sources of error.

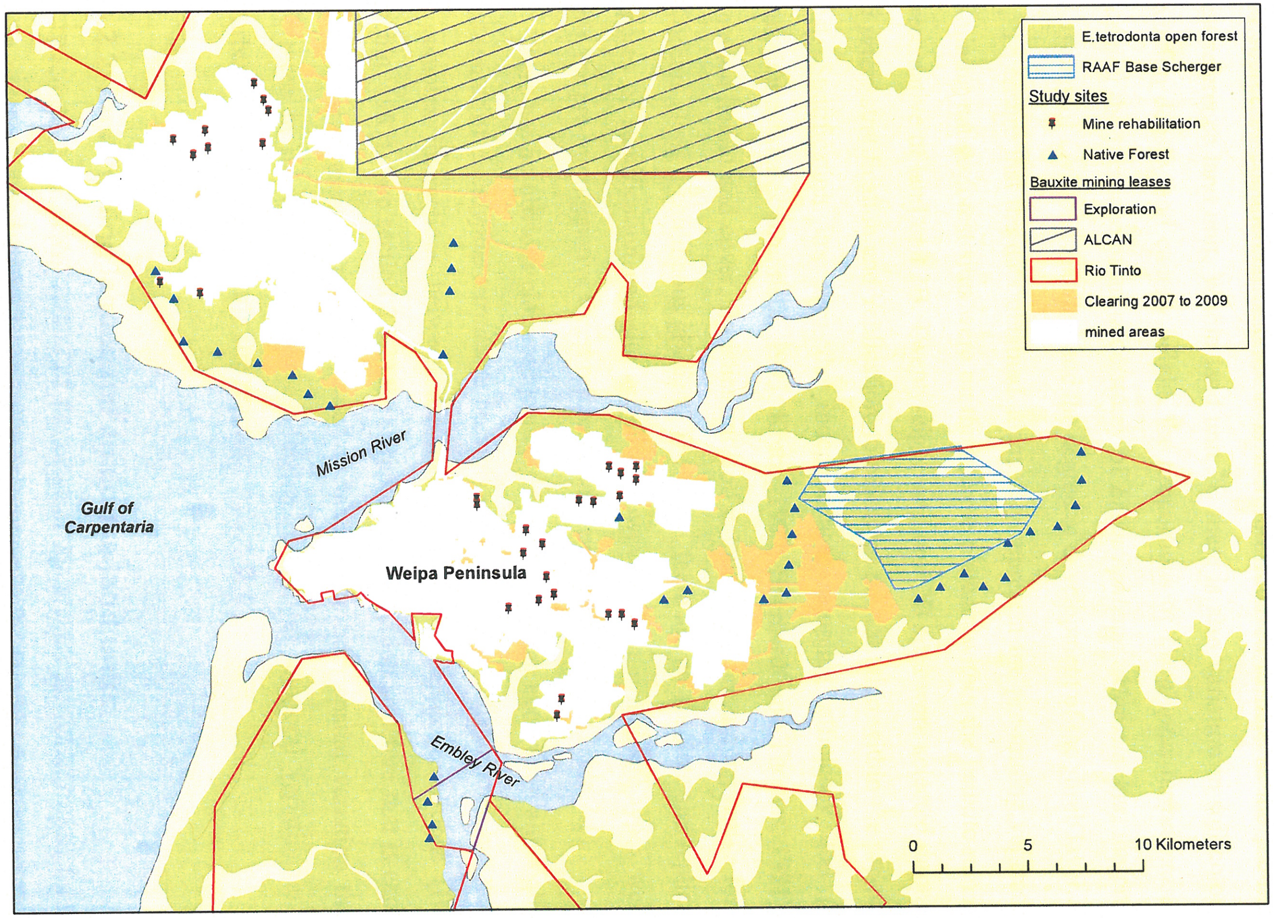

Figure 5.1 Location of study sites

5.3.2 Selection of native forest reference sites

5.3.2.1 Variability within the pre-mining native forest land unit

The dominant ecosystem that occurs on the Weipa bauxite plateau is Regional Ecosystem 3.5.2 (RE 3.5.2), tall woodland dominated by *Eucalyptus tetrodonta, Corymbia nesophila* ± *Erythrophleum chlorostachys* (Sattler and Williams, 1999). Although this ecosystem has been described as uniform over large areas, there is heterogeneity in the floristics and vegetation structure. A number of land units are recognised within RE 3.5.2 that differ in topography, soils, drainage, vegetation structure and floristics (Gunness *et al.*, 1987). Land unit 2B, *Eucalyptus tetrodonta* tall woodland / open forest on red earth soils is the most extensive of these and is the main source of commercial bauxite. It is therefore referred to here as the pre-mining native forest reference land unit. Within land unit 2B there is also variation in soil colour, drainage, vegetation density and canopy height, the proportions of *Eucalyptus tetrodonta* to *Corymbia nesophila,* the presence of an understorey of *Erythrophleum chlorostachys,* the presence of an understorey of *Livistona muelleri* and the degree of development of the shrub layer.

To ensure that the data obtained adequately represent the pre-mining vegetation state, all pre-mining native forest sites were on the bauxite plateau, within the bauxite mining lease and representative of land unit 2B. Twenty-eight land unit 2B sites were selected through a process of *a priori* stratification and 'replication' to represent the variation within the pre-mining native forest. Stratification improves the independence, sampling efficiency and precision of population estimates (Dutilleul, 1993; Krebs, 1997; Eberhardt and Thomas, 1991). Change in vegetation height and cover, floristics and soil colour in the pre-mining native forest is subtle and occurs along a continuous gradient. Therefore, there are no absolute site replicates. For the purposes of stratification and site selection however, six *a priori* site types were identified within the pre-mining native forest reference land unit 2B (LU 2B) (Table 2.1).

There are no obvious chronosequences in the pre-mining ecosystem although some even-sized stands of *Eucalyptus tetrodonta* possibly indicate previous disturbance by cyclones. Given the complex interactions between the fire regime and edaphic conditions (Bowman and Minchin, 1987), (see section 2.6.1) the variation in vegetation structure that is captured by the native forest reference sites is assumed to incorporate both spatial variation that is due to differences in substrate and water availability as well as temporal variation in response to disturbance history, particularly fire regime.

5.3.2.2 Post-mining landscape analogue land units

Reconstruction *sensu stricto* of the original *Eucalyptus tetrodonta* open forest, land unit 2B, is not considered to be technically or economically feasible because of the significant changes in soil depth and drainage characteristics caused by mining (Foster, 1985). In this case, use of landscape analogues for the post-mining landscape is recommended for identifying suitable references for species composition, structure and function (Walker and Reddell, 2007). An assessment by CSIRO recommended that land units 2C and 5K are the most ecologically appropriate landscape references for post-mining rehabilitation areas that are not seasonally inundated (Reddell and Hopkins, 1994). Other ecosystems are recommended for parts of the mine that are seasonally flooded but these represent a small proportion of the total post-mining landscape.

Both land units 2C and 5K are also part of RE 3.5.2 although they differ from the pre-mining land unit 2B in a number of respects. Land unit 5K (LU 5K) is a grassy woodland dominated by *Eucalyptus tetrodonta* and *Corymbia nesophila* that occurs on the lateritic eroding slopes of the bauxite plateau. Land unit 2C (LU 2C) is a tall woodland also dominated by *Eucalyptus tetrodonta* and *Corymbia nesophila* that occurs on the bauxite plateau in areas where drainage is impeded. It occurs in the headwaters of riparian zones where drainage lines are not incised. Land unit 2C is gradational to land unit 2B and has also been cleared for mining although it is less widespread than land unit 2B. Other studies have also recommended that vegetation composition and structure, and vertebrate faunal communities in woodland on ironstone, (i.e. land unit 5K), be studied to establish a benchmark for the anticipated mature vegetation community in post-mining rehabilitation (Reeders and Morton, 1983; Winter and Alford, 1999).

Consequently, in addition to the 28 pre-mining native forest LU 2B sites, native forest reference sites included four LU 2C sites and four LU 5K sites (Table 5.2). The intention of sampling land unit 2C and land unit 5K sites is to obtain ecologically appropriate alternative reference values for bird assemblages, vegetation and landscape function against which to compare the developing vegetation in post-mining rehabilitation sites. Although these three land units vary in structure, to improve readability they are referred to collectively in this thesis as native forest reference sites (or land units) except where there is a need to differentiate between them. Where there is a need to differentiate them from the pre-mining native forest LU 2B, these are referred to as the post-mining landscape analogue land units.

Table 5.2 Stratification of native forest sites

Landscape	**Soil type**	**Vegetation structure**	**Floristic variation**	**Site type**
Bauxite plateau	Red earth	Tall open forest to tall woodland	Tall, open, grassy	*E. tetrodonta* open forest to tall woodland with grassy understorey - LU 2B (type 1) 6 sites (Figure 5.2)
		Tall open forest to tall woodland	*Corymbia nesophila* co-dominant	*E. tetrodonta* open forest to tall woodland with grassy understorey, *E tetrodonta* dominant, *Corymbia nesophila* co-dominant - LU 2B (type 2) 2 sites (Figure 5.2)
		Tall open forest to tall woodland	*Erythrophleum chlorostachys* sub-canopy layer	*E. tetrodonta* open forest to tall woodland, sub-canopy layer of *Erythrophleum chlorostachys* - LU 2B (type 3) 5 sites (Figure 5.3)
	Red / yellow earth	Tall open forest to tall woodland	*Livistona muelleri*	*E. tetrodonta* open forest to tall woodland, sub-canopy layer *L. muelleri* -LU 2B (type 4) *Livistona* is also an understorey layer in LU 2C but in combination with different perennial grasses and soils. 1 site (Figure 5.3)
		Tall layered woodland	Well developed shrub layer	*E. tetrodonta* tall layered woodland - LU 2B (type 5) 10 sites (Figure 5.4)
		Tall, layered woodland	Thickening	*E. tetrodonta* tall woodland, thickening layer - LU 2B (type 6) 4 sites (Figure 5.4)
	Yellow / grey earth	Tall layered woodland	*L. suaveolens* *L. muelleri*	*E. tetrodonta* tall, layered woodland with *Livistona muelleri* and *Lophostemon suaveolens* - LU 2C 4 sites (Figure 5.5)
Eroding slopes of bauxite plateau	Yellow / red earths with outcropping ironstone or ironstone gravel	Tall woodland to open woodland	Woodland	*Corymbia nesophila / Eucalyptus tetrodonta* open woodland - LU 5K 4 sites (Figure 5.5)

Figure 5.2 Variation in pre-mining native forest reference sites - 1

(a) *E. tetrodonta* open forest to tall woodland with grassy understorey on red earth soils. Pre-mining native forest reference LU 2B (type 1); (b) *E. tetrodonta* open forest to tall woodland with grassy understorey on red earth soils, *Corymbia nesophila* co-dominant. Pre-mining native forest reference LU 2B (type 2).

(a)

(b)

Figure 5.3 Variation in pre-mining native forest reference sites - 2

(a) *E. tetrodonta* open forest to tall woodland on red earth soils, sub-canopy layer of *Erythrophleum chlorostachys*. Pre-mining native forest LU 2B (type 3); (b) *E. tetrodonta* open forest to tall woodland on red-yellow earth soils, sub-canopy layer of *Livistona muelleri.* Pre-mining native forest LU 2B (type 4). Gradational to LU 2C.

(a)

(b)

Figure 5.4 Variation in pre-mining native forest reference sites - 3

(a) *E. tetrodonta* tall layered woodland on red-yellow earth soils. Varying floristic composition and extent of development of the shrub layer. Pre-mining native forest LU 2B (type 5); (b) *E. tetrodonta* tall woodland on red-yellow earth soils, thickening tree layer. Pre-mining native forest LU 2B (type 6). Gradational to LU 2C.

(a)

(b)

Figure 5.5 Post-mining landscape analogue land units

(a) *E. tetrodonta* tall, layered woodland on yellow earth soils with *Livistona muelleri* and *Lophostemon suaveolens*. Post-mining landscape analogue LU 2C; (b) *C. nesophila* and *E. tetrodonta* tall open woodland on shallow yellowish-red soils with outcropping ironstone, or ironstone gravel. Post-mining landscape analogue LU 5K.

The land units have not been mapped for the entire study area and it is not known what proportion of the *Eucalyptus tetrodonta* tall woodland is made up of each *a priori* type. It was originally intended that an equal number of replicates of each site type would be sampled. Ultimately, however, sampling of site types was not equal. This was partly a consequence of recognising that absolute replicates did not exist and partly that it proved difficult to assign sites strictly to a particular type. The number of replicates identified for each site type reflects their extent in the study area in that more common site types were selected more frequently. Sites were also selected within the practical constraints of: (1) having access only to a limited area; (2) selecting sites across the accessible geographic range; (3) ensuring that sites were at least 1 km apart; (4) using existing access roads; and (5) not having more than one site replicate within a vegetation patch.

5.3.2.3 Independence of native forest reference sites

Statistical analysis requires the assumption that data observations are obtained from independent sample units. In the case of continuous vegetation cover where sampling units are not discrete, care must be taken to ensure that replicates of each site type are not pseudoreplicates, (i.e., drawn from the same vegetation 'patch' of a site type) *sensu* (Hurlbert, 1984). One way to achieve independence of observations is completely random selection of samples. However, completely random selection of sites would have been impractical and inefficient due to the difficulty in accessing randomly selected sites and the number of sites that would be needed to sample the inherent variability. Independence in site selection was achieved by the stratification process outlined above and by partially randomising site selection using ArcGIS.

After determining the *a priori* site types, a GPS was used to identify the approximate locations of different site types along the existing system of tracks. Randomisation was achieved by using ArcGIS to: (1) draw 2 ha shapes (100 x 200 m) aligned north-south along the long axis; (2) position sites near the marked locations of the different site types; (3) position each 2 ha site approximately 50 m away from the road or track; and (4) ensure sites were at least 1 km apart. This last requirement was to ensure that bird observations were independent (Eyre *et al.*, 2006). The exact location and boundaries of each site were then uploaded onto a mapping GPS so that boundaries of the study sites could be located.

The scale of variation in vegetation (i.e., site types) in combination with the requirement to keep sites 1 km apart, meant that no two reference sites were in the same vegetation 'patch'. Sites were selected across the range of the practically accessible area. As far as possible 'replicates' of each site type were selected from different geographical areas. It was not possible to intersperse LU 2C, with the other land units as all of the remaining areas of LU 2C that have not already been mined occurred in a slight depression in the centre of the Weipa peninsula.

5.3.3 Selection of mine rehabilitation sites

5.3.3.1 Representativeness of mine rehabilitation sites

Bauxite mining commenced at Weipa in the late 1950's and has been operating continuously since then. Various rehabilitation strategies have been used over this time. As this study aimed to evaluate the effectiveness of post-mining rehabilitation on the Weipa bauxite plateau in restoring bird habitat values of the *Eucalyptus tetrodonta* open forest, mine rehabilitation sites were only selected from sites that had been rehabilitated with the specific aim of restoring habitat. The current commitment is to reinstate 82% of the mined area to 'native ecosystems which are a mixture of local native tree and shrub species which create a habitat for fauna' (Comalco Mining and Refining, 2004:9). Prior to 2004, since at least 1981, the commitment for post-mining rehabilitation was to 'establish self-sustaining, maintenance-free vegetation comprising a variety of native plants which in turn support native fauna' (Reeders, 1985).

As of 2007, 12,653 ha had been mined of which 59% (7,456 ha) had been 'rehabilitated to native flora'. The total area of the mine rehabilitation blocks that were selected for this study was 584 ha. Assuming that the mine rehabilitation blocks selected were uniform throughout, study sites represent 7.8% of the total area that had been rehabilitated to native flora.

5.3.3.2 Variability within mine rehabilitation sites

This research substituted space for time by using sites of different age to construct a chronosequence of mine rehabilitation sites. Substituting space for time is an approach that has been widely used to study successional change following disturbance and has been verified by long-term studies of faunal succession in a post-mining environment (Fox, 1996). However, a large number of variables have the potential to affect rehabilitation outcomes (Appendix 5). It was therefore important to minimise the amount of variability between sites. This was achieved by accessing rehabilitation records and selecting sites in which soil treatment, the composition of the seed mix and fertiliser application rates were as constant as possible. All mine rehabilitation sites selected had been ripped, fertilized and sown with a mixture of seeds. The seed mix was composed of Australian native plants but included many species that are not part of the pre-mining native forest. Fresh rather than stockpiled topsoil had been used in most sites.

Changes have occurred in rates of fertiliser application, the composition of the seed mix, density of seed application (g/ha) and establishment method (seed or tubestock) through time (Appendix 4) and (Appendix 7). It was not possible, therefore, to hold treatments completely constant for all age classes of mine rehabilitation. Changes in treatments and the large number of uncontrolled and unknown site variables, mean that rehabilitating sites of different ages can only approximate a chronosequence (Pickett, 1988). It is possible, however, to obtain an estimate of approximately how landscape functionality, vegetation structure and bird assemblages change through time, by sampling multiple replicates of sites of the same age across a chronosequence. In this study, multiple replicates of five age classes of post-mining rehabilitation were sampled.

As most change in developing vegetation is expected to occur in the first few years, replicates were selected to approximate a growth curve of time. Thirty-one mine rehabilitation sites were selected including: six 1 to 2 year old sites, six 3 to 4 year old sites, six 5 to 8 year old sites, eight 9 to 16 year old sites, and five 17 to 23 year old sites (Appendix 7). Twenty-three years was the oldest available site, given the need to keep rehabilitation treatments as constant as possible.

The criteria used for selecting mine rehabilitation sites were:

1. Rehabilitation treatment,
2. Age (time since rehabilitation commenced),
3. Rehabilitation block size and shape, and
4. Distance from remnant native forest.

Rehabilitation block size had to be at least 6 ha and of an appropriate shape to allow placement of a 100 m x 200 m research site surrounded by a 50 m buffer and aligned north-south along the long axis within the block. Distance of mine rehabilitation sites from remnant native forest was varied. The landscape consists of remnant vegetation and a mosaic of post-mining rehabilitation of various ages. The mosaic of post-mining rehabilitation sites is separated from remnant vegetation by dirt perimeter tracks, which are used as fire breaks and access tracks. Distances of rehabilitation sites from remnant native forest varied from 15 to 1500 m.

As fire is thought to have an important influence on the pathway of vegetation development in rehabilitation (Murray *et al.*, 2000), burnt sites were included. The only fire history information that was available was fire scar data for the last 7 years from the Cape York Peninsula Sustainable Fire Project. The data are incomplete as cloud cover prevents clear satellite images for many months of the year and only fire scars greater than 20 ha are mapped. Fire scar boundaries are also inaccurate at large cartographic scales as the minimum pixel size is 6 ha. Nevertheless, fire history for burnt sites was reconstructed to the extent possible.

5.3.3.3 Independence of mine rehabilitation sites

Independence of mine rehabilitation sites was achieved by using ArcGIS to select appropriate rehabilitation blocks and to position research sites within them. Rio Tinto provided shapefiles that included the location, block size, age and rehabilitation treatment category of all mine rehabilitation areas. Site selection was refined by accessing rehabilitation records and selecting sites in which soil treatment, and seed and fertiliser application rates were as constant as possible. Planned re-treatment of weed infested mine rehabilitation sites during this study precluded site selection from large areas of the Andoom mine. The 2 ha study sites were then mapped using ArcGIS and uploaded onto a mapping GPS (Garmin 60CX). Other than confirming that all sites were accessible, mine rehabilitation sites were selected site unseen.

To maintain independence of site data, no two sites were within the same rehabilitation block. With the exception of two sites that were only 200 m apart, (but of different ages), all rehabilitation sites were at least 500 m apart. There was insufficient accessible area to require all sites to be 1 km apart. Rehabilitation sites of similar age were selected from different parts of the mine as far as practicable. A distance of 1 km has been recommended to ensure independence of bird observations (Eyre *et al.*, 2006). However, it was assumed that the interspersion of sites of different age combined with a distance of 500 m between sites was adequate to ensure independence of bird observations in mine rehabilitation sites.

5.4 Sampling procedures

5.4.1 Bird assemblages

To enable quantitative comparison of bird assemblages between sites a standardised bird survey procedure was required. To assess the extent to which mine rehabilitation provides habitat for native forest birds, the procedure also had to provide estimates of abundance and occupancy rates of individual species between sites. In this study site detection rates were used as surrogates for site occupancy rates.

5.4.1.1 Bird survey procedure

Approval to conduct bird surveys was granted by the ANU Animal Experimentation Ethics Committee, Protocol number S.RE.06.06. The bird survey method used was a modification of the 2 ha-20 minute area search used for the Atlas of Australian Birds (Barrett *et al.*, 2003). Study sites were 2 ha in size, 100 x 200 m and aligned north-south along the long axis. Each site was divided into two 1 ha plots. The boundaries of each 2 ha site (and 1 ha plot) were identified in the field using a mapping GPS (Garmin 60CX). No physical site markers were used.

During each survey, all birds seen or heard within a 1 ha plot were identified. The species were recorded and the number of individuals of each species counted for 10 minutes. This was repeated in the second hectare for 10 minutes avoiding counting any individuals already counted in the first hectare. Any uncommon birds detected in any part of the 2 ha site at any time during the survey were recorded rather than risk missing the record. Birds flying over or through the site were not counted unless they stopped within the site. Birds using the air space, such as raptors or aerial insectivores flying low over the canopy that appeared to be searching for prey within the site, however, were included. Foraging or breeding behaviour was noted although no attempt was made to quantify such behaviour.

This survey procedure is not strictly speaking a 2 ha-20 minute search because all birds were not surveyed across the total 2 ha area for the entire 20 minutes. However, the presence/absence data (species list for each survey) is assumed to be as close to complete as the vegetation and species detectability allowed. The abundance data obtained was conservative as the requirement to avoid double counting meant that multiple individuals were only recorded when more than one individual of the same species was detected simultaneously within the site. For example, two rufous whistlers *Pachycephala rufiventris* would only be recorded if both individuals were calling at the same time or close together in time from different parts of the site. Given this requirement, count data are expected to underestimate the true abundance, especially of more common species. Dividing survey effort in this way was intended to improve site coverage and the reliability of count data. Each site was surveyed eight times over 16 months from September 2006 to December 2007. Total bird survey effort over 16 months amounted to 2.6 hours for each site. Total bird survey effort across all sites, was 174.2 hours.

One of the post-mining landscape analogue LU 2C sites, A19, was partially cleared after 6 bird surveys and all other field work had been completed there. For the final two bird surveys, bird data was collected from a new site that overlapped the original site by 100 m and the survey data was included with the first 6 surveys for that site.

5.4.1.2 Sources of error in the bird survey procedure

With any sampling procedure some degree of error or bias in the data is introduced. Sampling procedures that standardise effort have been criticised on the grounds that standardising effort is done at the cost of sample completeness (Watson, 2004; Watson, 2003). There are problems too with assuming that a standard survey method will be equally efficient in different types of vegetation (Harden *et al.*, 1986). Plot size and survey effort can be changed to improve survey completeness, but whatever sampling procedure is adopted, standardising effort across different vegetation types is not able to eliminate all sources of error because the amount of bias caused by these factors is not consistent between different vegetation types (Harden *et al.*, 1986). There may also be a trade-off between survey completeness and accuracy of density measurements (Slater, 1994). A review of bird census procedures found that none of the available methods provide unbiased estimates of bird densities (Pyke and Recher, 1984).

Nevertheless, there are advantages to standardising effort that are more important than producing unbiased density estimates. The advantage of standardised bird survey procedures is that results can be compared across sites and across studies (Recher, 1988). Given that birds are widely used in ecological research, a standardised method can be justified in spite of the systematic errors caused by standardised procedures. The 2 ha-20 minute area search has been found to be an efficient bird survey method (Loyn, 1986), and is a widely used survey method across many vegetation types.

Other than standard procedures, many factors can cause systematic error in bird surveys (Kendeigh, 1944; Thompson, 2002) including: inter-annual differences in bird assemblages; season; weather; time of day; time since fire (Woinarski, 1990); inter-specific differences in bird behaviour, conspicuousness and detectability; and variability in observers. The following section outlines the survey protocols that were used to reduce the potential of these factors to cause error in the bird data.

5.4.1.3 Error due to temporal variability

Bird assemblages are dynamic, changing from year to year in response to climatic conditions (Maron *et al.*, 2005; Mac Nally, 1996), and from season to season (Farmer *et al.*, 2004; Slater, 1995; Arnold, 2003). The patterns of variation in distribution and abundance vary from species to species. Time of day has also been shown to affect the outcome of bird surveys in a number of studies (Slater, 1994; Craig and Roberts, 2001). Although individual species have different behaviour, in general the total number of species has been found to be negatively correlated with time since sunrise.

Repeat surveys can reduce the amount of error due to climate variability, seasonal turnover of species and disturbance events such as fire and provide a more accurate estimate of the amount of error (Field *et al.*, 2002). Unless long term surveys are undertaken with frequent repeat visits, even repeat bird surveys will be incomplete in comparison to the true but unknown long term bird assemblages. However, it has been found that the most important factor is the total amount of time spent at the site. In a comparison of survey techniques using 2 ha search areas, the techniques that gave the most complete species list had 120 minutes or more of total sampling effort. Species lists were usually complete after a total of 100 minutes at sites with a sparse understorey and 160 minutes at sites with a dense understorey (Slater, 1994).

Potential error in the bird data caused by temporal variability was reduced by:

1. Conducting eight repeat surveys at each site over 16 months,
2. Surveying all plots 'synchronously’ during each round of surveys,
3. Conducting all surveys between 6.30 am and 10.00 am, and
4. Surveying each site for a total of 160 minutes.

In this study, once a round of site visits had commenced, three to five bird surveys were conducted every day until all sites had been visited unless circumstances dictated otherwise. In November 2006, widespread fires and the operation of mine equipment on some parts of the mine lease prevented continuous surveying for safety reasons. A cyclone in February 2007 required a nine day break more than half way through the first round of wet season surveys. Strong wind sometimes delayed access to the sites accessed by boat. Strong trade winds during the dry season sometimes required that surveying be abandoned as early as 8.30 am. With 67 research sites it generally took 16 or 17 consecutive mornings to complete a round of surveys. Although this is not synchronous in the strict sense of the term, it was assumed that this would be sufficient to eliminate differences due to seasonal effects. The order in which sites were visited was changed with each round of surveys.

Sites were surveyed twice each season. Previous studies have found that the time span over which surveys were conducted within a season did not have a significant influence on estimates of species richness. Spacing of up to 93 days between surveys had little impact on the number of species recorded provided that repeat visits were conducted within a period during which the species assemblage was roughly constant (Field *et al.*, 2002). In this study, surveys 1 and 2 were conducted in September - November 2006, surveys 3 and 4 in late January to early April 2007, surveys 5 and 6 in June - August 2007, and surveys 7 and 8 in November - December 2007.

5.4.1.4 Error due to weather

Previous studies have found correlations between the number of bird species recorded and weather conditions. The number of species detected declines as wind increases and temperature falls (Slater, 1994; Rosenstock *et al.*, 2002). Given the number of sites and the requirement for synchronous surveys, it was impossible to survey all sites under exactly the same conditions. Error in bird data due to weather was reduced by not surveying when it was raining or very windy (defined as when large branches began swaying) and not surveying later than 10.00 am. Categorical site data was collected for each survey on wind and cloud cover.

5.4.1.5 Error due to differences in detectability

Not all survey methods are equally effective in identifying birds (Pagen *et al.*, 2002). Other than environmental variables, such as weather conditions, bird count data can be affected by differences in bird detectability (Rosenstock *et al.*, 2002). Using only point count observations can introduce error due to differences between species in detectability. For the types of habitats being sampled in this study, use of additional sampling methods such as mist-netting was not considered to add sufficient value to justify the effort involved. Therefore observational and audial surveying were the only methods used. Potential error due to differences between birds in detectability and difference between site vegetation density was reduced by actively searching each plot rather than doing point counts from the centre of each 1 ha plot (Loyn, 1986).

As there was only one observer collecting bird survey data, error due to differences between observers was not an issue (Kavanagh and Recher, 1983; Cunningham *et al.*, 1999). The observer already had many years of experience surveying birds in the research area and was able to identify most of the species by call from the commencement of the study. Some research has shown that the colour of observer clothing can also affect estimates of species richness (Riffell and Riffell, 2002). Unfortunately mine safety requirements required that high visibility clothing be worn while on any active parts of the mine lease.

5.4.2 Vegetation composition and structure

The purpose of vegetation sampling was twofold. It had to provide estimates of a number of vegetation attributes that have been found to be important determinants of bird distributions. Secondly, it had to provide estimates of vegetation characteristics that are important for analysing ecosystem development. Many of the structural attributes of vegetation that have been found to be positively associated with the presence, abundance or richness of one or more bird species in Australian eucalypt forest or woodlands are also important for characterising ecosystem function and ecosystem development (McElhinny *et al.*, 2005; Tongway and Hindley, 2004; McElhinny *et al.*, 2006).

The vegetation sampling procedure was intended to provide estimates of vegetation composition and structure across the entire two ha bird survey sites. The following variables were selected for measurement at each site: (1) density and cover of perennial grasses; (2) amount of cover and density of stems in four woody vegetation layers; (3) diameter of stems larger than 10 cm; and (4) height and depth of the canopy in each vegetation layer. Compositional data was derived from the sampling data.

5.4.2.1 Vegetation sampling procedure

Vegetation was sampled once at each site. All vegetation data was collected during May to June 2007. At this time the bush is drying out following the wet season, many grasses are in flower or seed and the widespread grass fires of the mid to late dry season have not yet started. Sampling all sites during the same period minimised any error in vegetation data due to temporal effects.

A point-centred quarter (PCQ) sampling method was used so that data could be collected across a large number of sites and across entire two ha sites rather than extrapolating data from a quadrat or belt transect (Krebs, 1997; Bonham, 1989). Plotless methods, such as PCQ, eliminate bias due to quadrat size, quadrat shape and edge effects. The PCQ method is also considered to be an efficient method of data collection that is statistically robust and a good estimator of density (Krebs, 1997; Reinke and Jones, 2006).

A trial of the sampling procedure sampled three vegetation size classes (perennial grass, woody stem < 1 cm dbh and woody stem > 1 cm dbh) at 25 points using a 10 m search radius. This resulted in low rates of canopy trees being sampled. The sampling procedure was therefore modified by reducing the number of sampling points, increasing the search radius and increasing the number of vegetation size classes sampled. Initially the number of vegetation layers was increased to four by stratifying the >1 cm dbh category into two layers: (1) >1<10 cm dbh; and (2) >10 cm dbh. This procedure still resulted in very few large canopy trees being sampled. As the canopy layer is potentially the most ecologically significant for birds and contains most of the vegetation biomass, an additional size class of >35 cm dbh was added. This size class was determined by measuring a number of tall *E. tetrodonta* trees that formed the tallest canopy layer at one of the native forest reference sites. Ideally, the number of sampling points required for sampling adequacy should be determined for each vegetation type. However, field tests of the PCQ method have shown that 15 sampling points provided means for density that were within 5% of values obtained from 30 sampling points (Bonham, 1989). The number of sampling points was therefore reduced to 16 and the search radius increased to 16.5 m.

Vegetation was sampled at 16 systematically located sampling points within each site. The coordinates for each sampling point were calculated using Microsoft Excel from the coordinates of the south-west corner of each site. Systematic sampling ensured that sampling points were independent and enabled the search radius at each sampling point to be maximised. Vegetation sampling points were loaded into a mapping GPS for identification in the field. The area around each sampling point was divided into four 90° quadrants. At each sampling point, a cross was pegged into the ground with one axis aligned north-south (Figure 5.6). Within each quadrant the closest individual plant within a 16.5 m radius was then identified for five *a priori* defined size classes. A limit was imposed on the search radius to ensure that double counting did not occur.

Stratification of target plants by size was designed to ensure that sufficient data was obtained for all potentially ecologically significant vegetation layers. Stem diameters (dbh) were measured at a height of 1.5 m. For each site there were potentially 64 distance measurements plus the associated plant measurements for each of following vegetation layers:

1. Perennial grass;
2. Low shrub - defined as single-stemmed woody plant < 1 cm diameter at 1.5 m, or multi-stemmed shrub < 2 m tall;
3. Tall shrub - defined as single-stemmed woody plant >1< 10 cm diameter at 1.5 m, or multi-stemmed shrub > 2 m tall;

4. Small tree - defined as woody plant, single stemmed to a height of 50 cm and at least one stem >10< 35 cm diameter at 1.5 m; and
5. Big tree - defined as woody stem > 35 cm diameter at 1.5 m.

For each target plant the following data were recorded: species identity; distance from the centre of the sampling point; heights to the base and top of the canopy; and two canopy dimensions, namely, length of the longest axis and width perpendicular to the longest axis. Stem diameter was only recorded for stems with a diameter at 1.5 m >10 cm. For multi-stemmed plants with at least one stem >10 cm, all stem diameters were recorded. Finally, the canopy density within the ellipse defined by the two canopy dimensions was estimated. For perennial grasses, diameter at the base, height, two canopy dimensions, and canopy density were recorded. If within a quadrant there was no plant of the target size class inside the search radius, a blank was recorded. A correction factor was applied to adjust for error introduced by vacant quadrants when calculating stem densities (Warde and Petranka, 1981; Mitchell, 2007).

Figure 5.6 Vegetation sampling

Identifying quadrants and measuring distance in a mine rehabilitation site

5.4.3 Landscape functionality

The purpose of sampling landscape functionality was to assess key aspects of functionality that are pre-requisites for ecosystem development and sustainability. Landscape Function Analysis (LFA) is a monitoring procedure used in some Australian mine rehabilitation sites that uses visual indicators of the status of biotic and abiotic processes. These indicators are used to monitor rehabilitation progress in terms of resilience, self-sustainability and stability (Tongway and Ludwig, 2006). Landscape function analysis does not assess the sustainability of the ecosystem *in toto*. It does, however, measure ecological attributes that are pre-requisites to succession and long term sustainability, namely water infiltration, nutrient cycling and soil stability.

Landscape functionality is defined as the manner in which scarce, vital resources are retained, used, and lost from landscapes. Landscape function analysis explicitly links spatial pattern with the abiotic and biotic processes that maintain that pattern (Tongway and Ludwig, 2006). A landscape is characterised as highly functional if it is organised so that vital resources tend to be retained and used within the hydrologic boundaries. To link pattern and process, the LFA procedure uses indicators of soil surface processes. These surface indicators are combined to develop site indices of soil stability, water infiltration, and nutrient cycling that are surrogates of a number of soil properties.

The use of these indices as surrogates of soil properties has been verified in a number of mine sites although the degree of verification differs depending on climatic and soil conditions (Tongway and Hindley, 2003). Laboratory analysis has verified that aggregate stability of the 0 to 1, and 1 to 3 cm layers of soil is significantly linearly related to the LFA soil stability index. Field measurement of saturated flow infiltration is significantly related to the LFA infiltration index. Measured soil respiration, which is a measure of microbial activity, and soil nutrient pool size are significantly linearly related to the LFA nutrient cycling index, although the slope of the relationship varies. Tongway and Hindley (2003) measured soil nutrient pool size in a laboratory including measures of organic nitrogen, organic carbon, plant available phosphorus, total sulphur and mineraliseable nitrogen. Plant available phosphorus was not related to the nutrient cycling index due to additions of superphosphate (Tongway and Hindley, 2003). The value of the LFA indices as surrogates of ecosystem properties, however, is not universally accepted. It has been argued that key ecosystem processes such as N mineralisation and nitrification should be used as indicators themselves due to poor correlation of mineralisation potential or microbial biomass with long term rates of mineralisation (Ward, 2000).

The LFA procedure was developed in arid and semi-arid landscapes where run-on and runoff processes interact with source-sink patches to concentrate scarce water and nutrients (Ludwig *et al.*, 1997). The procedure uses a trigger-transfer-reserve-pulse framework. The concept of landscape function underlying the LFA procedure incorporates the concept of alternate states and transitions *sensu* (Westoby *et al.*, 1989). Successional processes or transitions may occur between different states, however, the transitions are not unidirectional in space or time. Thus, the underlying conceptual framework of LFA is consistent with the concepts of ecosystems and succession outlined in chapter 4. The model of semi-arid landscape function whereby runoff and run-on processes interact with patches to concentrate water and nutrient resources has been confirmed experimentally (Tongway and Ludwig, 1996; Ludwig and Tongway, 1996). An experimental manipulation that created patches[13] (landscape sinks) was successful in promoting water infiltration and soil deposition. A wide range of soil physical, chemical and biological properties that promote plant growth were restored using this procedure.

The interpretational framework for the LFA procedure assumes that the model response of a site that is progressing towards sustainability is defined by a sigmoidal curve (Tongway and Hindley, 2003). The curve indicates a slow rise in young mine rehabilitation sites, followed by a more rapid rise in landscape function as the biota becomes established and begins to influence ecosystem processes. This rise flattens after a number of years, eventually reaching a plateau defined by the biogeochemical potential of the site. This plateau is the target value that is derived from assessment of landscape function in suitable reference local native ecosystems. Over time, LFA monitors the trajectory of a system with reference to these target values. This does not make any assumption about how functional or sustainable local native ecosystems are. Neither does it assume that the reference ecosystem is static. In fact ongoing sampling of the reference system and how it responds to disturbance is recommended. However, it does assume necessarily that, given the prevailing environmental conditions and disturbance regime, the selected reference ecosystem represents the best available example of a self sustaining system.

The LFA procedure has been verified at Alcan's Gove bauxite mine which has very similar soil and climatic conditions to Weipa (Table 5.3), although there may be differences in post-mining issues between Gove and Weipa due to landscape differences. Gove is much higher above sea level than Weipa. Monitoring of LFA indices at Alcan's Gove bauxite mine found that the values of all three indices increased with age of mine rehabilitation. The trajectories of the stability and infiltration indices in post-mining rehabilitation approximately corresponded to a sigmoidal response but the shape of the nutrient cycling index did not (Tongway and Hindley, 2003). Plant community development at Gove over a 26 year chronosequence was associated with soil development (Spain *et al.*, 2006).

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[13] A patch in LFA terminology is a physical or biological feature that accumulates resources in the landscape by restricting the downslope flow of water, soil, and organic materials. Patches are identified by the presence of accumulating resources such as leaf litter and alluvium. 'Inter-patches' are zones of resource loss, where water, soil, or organic materials are transported downslope by water or downwind by wind.

Table 5.3 A comparison of conditions at Gove and Weipa

	Gove	Weipa
Annual mean rainfall	1444 mm	1765 mm
Daily mean maxima - January - July	32°C 27.8°C	32°C 30.5°C
Daily mean minima - January - August	24.6°C 19.2°C	24°C 18.8°C
Native vegetation community	Tall open forest - dominated by *Eucalyptus tetrodonta* and *Eucalyptus miniata*	Tall open forest / tall woodland dominated by *Eucalyptus tetrodonta* and *Corymbia nesophila*

Ideally, time-series data would be used to monitor trends in landscape functionality in the mine rehabilitation. However, for practical reasons, the use of more readily available chronosequence data is widely used. The ecological appropriateness of using chronosequence data for LFA has been verified in a wide range of environments and has specifically been verified following bauxite mining at the Gove bauxite mine in the Northern Territory (Spain *et al.*, 2006; Tongway and Hindley, 2003).

Use of sites that represent an age-series in place of true time series data is only appropriate where similar rehabilitation techniques have been used over time. All of the mine rehabilitation sites used in this study had been ripped, fertilized and sown with a mixture of native seeds and most had been established with fresh rather than stockpiled topsoil although there has been some variation in rates of fertiliser application, rates of seed application and establishment methods over time. In the two oldest age classes of mine rehabilitation tubestock was planted in addition to direct seeding. In sites 16 years and older seed was applied aerially.

5.4.3.1 Landscape function analysis sampling procedure

Landscape function indices were sampled once at each site. All LFA data was collected from June to July 2007, prior to any fires to minimise any error due to temporal effects. Three x 30 m transects were assessed at all mine rehabilitation sites as there was considerable within-site variation in the distribution of surface materials. There was less within-site variation in the distribution of surface materials in native forest reference sites due to the dense cover of perennial grasses and slopes of less than one degree. Consequently only two 30 m transects were assessed at most of the native forest reference sites unless there was sufficient within site variability to warrant a third transect.

The origin of each transect was systematically located using the vegetation sampling points. Unless conditions dictated otherwise, transect origins were the vegetation sampling points A-6, B-4 and C-2. Transect origins were sometimes varied to ensure that a more accurate estimate of within-site variability was obtained or to ensure that the transect remained within the site if a transect ran downslope near the site perimeter. A tape was run out for 30 m due north from the transect origin and pegged at both ends except where there was a slope in which case the tape was run downslope, aligned with the maximum slope.

Landscape organisation was recorded by noting the location, length and width of patches and inter-patches (Figure 5.7) along the transect using the procedures outlined by Tongway and Hindley (2004). Soil surface assessment was conducted at up to five x 1 m query zones for each patch or inter-patch type along the transect. Within each query zone 11 soil surface assessment indicators were scored (Appendix 8). The indicators relate the status of a mix of biological and physical processes and are intended to record the increasing role of developing vegetation in providing ecosystem services during the progress of rehabilitation (Tongway and Ludwig, 2006).

Figure 5.7 Patches and inter-patches created by bank-trough structures

5.5 Data analysis

5.5.1 Analysis of bird data

All bird observations were entered into Access database which was used to filter data, generate summary data and do calculations. Data were exported to Microsoft Excel and put into the appropriate formats for use in JMP IN, GenStat 10 and PC-ORD 5. GenStat 10 was used for data transformations, calculations of summary statistics, statistical tests, analysis of variance, simple linear regressions and non-linear regressions (VSNi, 2007). JMP IN was also used for some statistical tests (SAS, 2001). Some data transformations were required so that data met assumptions of homogeneity of variance. Indicator species analysis and all multivariate analyses were conducted using PC-ORD 5 (McCune and Mefford, 2006). Estimated species richness of native forest sites and rarefaction curves were calculated using EstimateS (Colwell, 2008).

5.5.1.1 Analysis of variance and statistical tests

A combination of parametric and non-parametric procedures was used to analyse bird data. Bird data were analysed at the following levels: species, foraging group, site, visit number, land unit / age class, and site category (native forest / mine rehabilitation) (Appendix 9).

Repeated measures analysis fitting a power model for the correlation structure between successive site visits was conducted to analyse variance in bird abundance data using GenStat 10. Tests for differences in summary values (species richness, bird abundance and species shortfall index) at the level of age class / land unit values were conducted using Student's t-tests in JMP IN.

The mean abundance, mean detection rate and the percentage of sites in which each species was recorded were compared between native forest sites and mine rehabilitation sites by pooling site data. Non-parametric Mann-Whitney U-tests were used to compare the equality of values for individual species. For species whose mean abundance was significantly different between reference and mine rehabilitation sites profile graphs were produced in Excel to illustrate differences between reference sites and mine rehabilitation sites.

The bird assemblages of native forest reference sites and mine rehabilitation sites were also compared at the level of foraging groups by assigning each species to a foraging group. The mean abundance of each foraging group in native forest reference sites and mine rehabilitation sites was calculated for (1) each age class / land unit and (2) site category (i.e., reference / mine rehabilitation) from site data. The equality of abundance of foraging groups in native forest reference sites and mine rehabilitation sites was compared using non-parametric Mann-Whitney U-tests.

A non-parametric procedure, multi-response permutation procedure (MRPP) was used to test for differences in bird species composition with PC-ORD 5. The purpose of MRPP is to detect concentration within *a priori* defined groups (Zimmerman *et al.*, 1985). Species composition was compared using site presence / absence data for all 97 bird species recorded during surveys. The similarity matrix was calculated using the Bray-Curtis coefficient. Groups were the *a priori* defined design variables, site category and age class / land unit. The MRPP calculates an average within group delta for each group and compares it with an expected delta calculated to represent the mean delta for all possible partitions of the data (McCune and Grace, 2002). Tests for differences in species composition were conducted for native forest versus mine rehabilitation and pairwise tests were conducted for each age class and land unit.

5.5.1.2 Calculation of species shortfall index

The similarity of a sites' bird species composition to the pre-mining native forest reference condition was compared by calculating species shortfall index values for each site following the procedure outlined in Hannah *et al.*, (2007):

1. An ambient abundance value for each native forest bird species was derived by calculating its' mean abundance across all pre-mining native forest LU 2B sites,
2. Site values for the degree of deviation from the reference condition were calculated as

$$SSI = 100[a_x - \sum \min(a_{in}, a_{ix})]/a_x$$

where

a_{in} is the mean abundance of species *i* in pre-mining native forest LU 2B sites,

a_{ix} is the abundance of that species in site *x*, and

a_x is the sum of mean abundances of all species in pre-mining native forest LU 2B sites.

If a site contains the full complement of species recorded across all pre-mining native forest reference LU 2B sites, with each species present in at least equal abundance to its average abundance across all pre-mining native forest reference LU 2B sites, then there is no species shortfall and the site species shortfall index is 0%. If a site contains none of the species recorded in pre-mining native forest reference LU 2B sites, then the site species shortfall index is 100% (Hannah *et al.*, 2007).

5.5.1.3 Estimated species richness

The estimated species richness of native forest sites was calculated using the Chao2 species richness estimator in EstimateS (Colwell, 2008). Individual and sample based rarefaction curves were also calculated using EstimateS. Calculations used pooled bird abundance data from all 36 native forest reference sites over all eight site visits.

5.5.1.4 Indicator species values

Indicator species for native forest and mine rehabilitation sites were identified using indicator species analysis in PC-ORD 5. Values were calculated using the total abundance of each species in each site. This procedure calculates each species' (1) proportional abundance in each group relative to its abundance in all groups (exclusiveness); and (2) its proportional frequency in each group (faithfulness) (McCune and Grace, 2002). The values for exclusiveness and faithfulness are multiplied to obtain an overall indicator value. The statistical significance of the indicator values was tested using a Monte Carlo procedure with 4999 permutations.

5.5.1.5 Identification of bird-environment relationships

Canonical Correspondence Analysis was used in this study to explore species-environment relationships. Only data for species recorded in five or more sites were used for this analysis. The rationale for removing species recorded in less than five sites was that species that were recorded infrequently were considered unlikely to provide sufficient data to demonstrate meaningful habitat preferences.

Canonical Correspondence Analysis is a multivariate analysis technique developed to relate community composition to known variation in the environment (Ter Braak, 1986). The procedure constrains an ordination of one matrix (species abundance) by a multiple linear regression on variables in a second matrix (environmental variables) (McCune and Grace, 2002). It is described as a direct gradient analysis technique in which ordination axes are linear combinations of environmental variables (Ter Braak, 1987). It therefore enables species composition to be directly related to the measured environmental variables (Ter Braak, 1986). Canonical Correspondence Analysis has been reported to be robust to skewed species distributions, quantitative noise in species abundance data and highly intercorrelated environmental variables (Palmer, 1993).

Three separate CCA's were used to relate the detection rate of bird species observed in five or more sites and: (1) landscape variables; (2) 12 vegetation structure variables; and (3) vegetation composition variables. Row and column scores were standardized by centring and normalizing. Scaling of ordination scores was by column (species). Scores for graphing sites were linear combinations of the landscape, vegetation structure and vegetation composition variables. Monte Carlo randomization procedures using 998 runs were used to test for correlation between the bird species data and the environment data. Time of day was used as a random number seed. Canonical Correspondence Analysis was also used to explore the effect of variation in vegetation structure on bird detection rates in native forest references sites only. Canonical Correspondence Analysis (CCA) was also used to relate the abundance of individuals in each foraging group in each site, to site vegetation structure.

The effect of proximity to remnant vegetation on bird species richness in mine rehabilitation sites was tested using simple linear regression of distance from remnant vegetation against site species richness and species shortfall index. The effect of area of remnant native forest on mine rehabilitation sites was also tested using simple linear regression of area of remnant vegetation within a 500 m buffer of the site against site species richness and species shortfall index.

5.5.2 Analysis of landscape function and vegetation data

Landscape function index values for each transect were derived by entering soil surface indicator scores into a spreadsheet developed by CSIRO which has built in calculations for deriving transect index values for landscape organisation, soil stability, water infiltration and nutrient cycling (Tongway and Hindley, 2004). Vegetation data were entered into Excel spreadsheets and then imported into Access database for data validation and calculation of summary data. Summary data were exported from Access into Excel and put into the appropriate formats for analysis in GenStat 10 and PC-ORD 5. A combination of parametric and non-parametric procedures was used to analyse LFA and vegetation data. GenStat 10 was used for calculations of summary statistics, analysis of variance, simple linear regressions and non-linear regressions (VSNi, 2007). Multivariate analyses were conducted using PC-ORD 5 (McCune and Mefford, 2006). Site values were pooled to obtain summary values for land unit / age class and site category (Appendix 10).

A linear mixed model was used to analyse variance in landscape function index values in mine rehabilitation sites. Residual plots for regressions on untransformed data were checked for normality and no data transformations were required for the LFA data. The change in landscape function values with age of site was analysed for stability, infiltration and nutrient cycling indices using non-linear regression. Exponential asymptotic curves were fitted to the data using the standard curve fitting procedure in GenStat 10.

Vegetation composition was compared using the multi-response permutation procedure in PC-ORD 5. Two tests were conducted, one using canopy volume data of 93 woody species and the other using canopy volume data of 24 grass species. Similarity matrices were calculated using the Relative Sorensen distance measure to standardise data by site. Relativisation changes the analysis to proportions of canopy volume of species rather than absolute canopy volumes. The weighting of groups was n/sum(n). The MRPP procedure calculates an average within group delta for each group and compares it with an expected delta calculated to represent the mean delta for all possible partitions of the data (McCune and Grace, 2002). A dendrogram based on the canopy volumes of 93 woody species was calculated using hierarchical agglomerative cluster analysis, Relative Sorensen distance measure, and flexible beta ($\beta = -0.25$) linkage method.

The density of framework, desirable and other species was compared between native forest reference sites and mine rehabilitation sites using the non-parametric multi-response permutation procedure in PC-ORD 5. Two tests were conducted, one used site values for stem density of framework species and another used site values for the percentage of the total canopy volume made up of framework species. For both tests, similarity matrices were calculated using the Relative Sorensen distance measure. The weighting of groups was n/sum(n).

Linear mixed models were used to analyse variance in vegetation height, canopy volume and basal area. The change in vegetation structure with age of site was analysed using non-linear regression for mean vegetation height, mean canopy volume m^3/ha and mean basal area m^2/ha. Exponential asymptotic curves were fitted using the standard curve fitting procedure in GenStat 10. Residual plots on untransformed data were checked for normality. Vegetation height data were square root transformed, canopy volume data were log 10 transformed, and basal area data were transformed by adding 0.001 to all data and then log 10 transforming to meet assumptions of normality. The number of vegetation layers and foliage cover of each vegetation layer were compared for each age class and land unit using summary values and a histogram.

5.5.2.1 Age of native forest reference sites

All of the native forest reference sites had what was defined in this study as a big tree layer, (i.e., woody stems exceeding 35 cm dbh). All native forest reference sites had Cooktown Ironwoods (*Erythrophleum chlorostachys*) with stem diameters exceeding 35 cm dbh with the exception of seven sites (three LU 2C and three LU 5K sites). Many of these Cooktown Ironwoods had diameters well in excess of 35 cm dbh and occasionally exceeding 70 cm dbh. The seven sites that lacked large Cooktown Ironwoods all had eucalypts with stem diameters >35 cm. In all but one of these seven sites (A-22) there were *E. tetrodonta* or *Corymbia nesophila* trees with stem diameters in excess of 50 cm dbh.

Cooktown Ironwoods have slow growth rates and in the Northern Territory trees with stems of 35 cm dbh are estimated to be in excess of 300 years old (Cook *et al.*, 2005). It is unlikely that growth rates of *Erythrophleum chlorostachys* on the Weipa plateau vary significantly from growth rates in the Northern Territory as rainfall, particularly winter rainfall which is an important determinant of growth rates, is very similar to that in the Northern Territory (G. Cook, personal communication, 2008). Savanna eucalypts with stems of 50 cm in the Kakadu National Park are estimated to be approximately 215 (±140) years old (Werner, 2005). Natural forest is constantly in a state of renewal as mature individuals senesce and are replaced. However, given the presence of large trees in all reference sites, it was assumed throughout this analysis that vegetation in native forest reference sites is 200 years old.

5.6 Conclusion

This chapter has described the research design including the sampling methods and the measures that have been incorporated to reduce sampling error. This study incorporates a number of elements that have not been used together in any previous assessments of bird response to succession in mine rehabilitation.

1. A large number of site replicates is used (67) with more or less equal sampling of native forest reference sites and mine rehabilitation sites.
2. A random stratified design is used that includes landscape context as independent variables in addition to site variables.
3. All mine rehabilitation sites were rehabilitated using the same basic strategy, and are the most important rehabilitation type at Weipa by area.
4. Recognising the effects of area and time, and the dynamic nature of bird assemblages through time, this research uses 'synchronous' bird sampling across all sites using standardised search area and search effort.
5. Each site is visited eight times to enable individual bird species detection rates to be calculated for each site. This will give more rigour to conclusions about bird occupancy of mine rehabilitation sites.
6. Vegetation attributes are directly measured rather than scored.
7. No previous study of bird succession in mine rehabilitation has combined landscape function analysis, vegetation sampling and bird surveys.

The site selection strategy aimed to maximise variability in native forest reference sites to obtain a rigorous estimate of error. Conversely it aimed to minimise variability in the rehabilitation techniques applied to mine rehabilitation sites so that they represent a chronosequence as closely as possible. This study is the first at Weipa to directly compare mine rehabilitation sites with reference native forest sites. It is also the first study at Weipa to enable quantitative assessment of the effectiveness of mine rehabilitation in restoring habitat values. Results of the data analyses are presented in the two following chapters. Chapter six presents the results of the data analysis in relation to the three key research questions for birds. Chapter seven presents the findings of this research in relation to the developing ecosystem in mine rehabilitation.

Chapter 6: Bird results

6.1 Introduction

This study posed three key research questions in relation to birds. First, how do bird assemblages in mine rehabilitation compare with native forest bird assemblages? Second, what is the temporal pattern of bird succession in mine rehabilitation on the Weipa bauxite plateau? Third, what environmental factors determine bird occupancy in mine rehabilitation and native forest on the Weipa bauxite plateau? This chapter presents the results of the data analysis in relation to these three questions. To begin with, the completeness of the bird surveys in native forest reference sites is examined. Even though the bird data are assumed to be representative for comparative purposes, it is useful to understand how complete the reference data might be.

6.1.1 Bird survey completeness

Total survey effort was 179 hours of which 96 hours were in native forest and 83 hours were in mine rehabilitation. Ninety-seven bird species were recorded during surveys and an additional 14 species were recorded incidentally[14] (Table 6.1 and Appendix 2). Seventy-nine bird species were recorded during surveys in native forest reference sites. This compares with an estimate of species richness for native forest reference sites of 90 species (sd ±7).

Table 6.1 Breakdown of the number of bird species recorded

	Mine rehabilitation only	Native forest and mine rehabilitation	Native forest only	Total
Survey records	18	55	24	97
Incidental records[1]	2	1	11	14
Total	20	56	35	111

The individual-based rarefaction curve indicates that the species list for the native forest reference sites was close to complete as the rate of increase in species richness has almost reached an asymptote (Figure 6.1). The sample-based rarefaction curve has a different slope from that of the individual based rarefaction curve and has not yet reached an asymptote, although the rate of increase in bird species richness has slowed on both curves (Figure 6.1). These rarefaction curves indicate that although not all bird species that might occur in the *Eucalyptus tetrodonta* open forest were recorded in this study, most of the species that occur

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[14] Incidental records in the mine only include species that were observed in mine rehabilitation vegetation and not birds using unusual habitats such as surface water, bare ground or mine infrastructure.

consistently were recorded. It is assumed, therefore, that the native forest bird data provide a suitable reference dataset for comparison with post-mining rehabilitation.

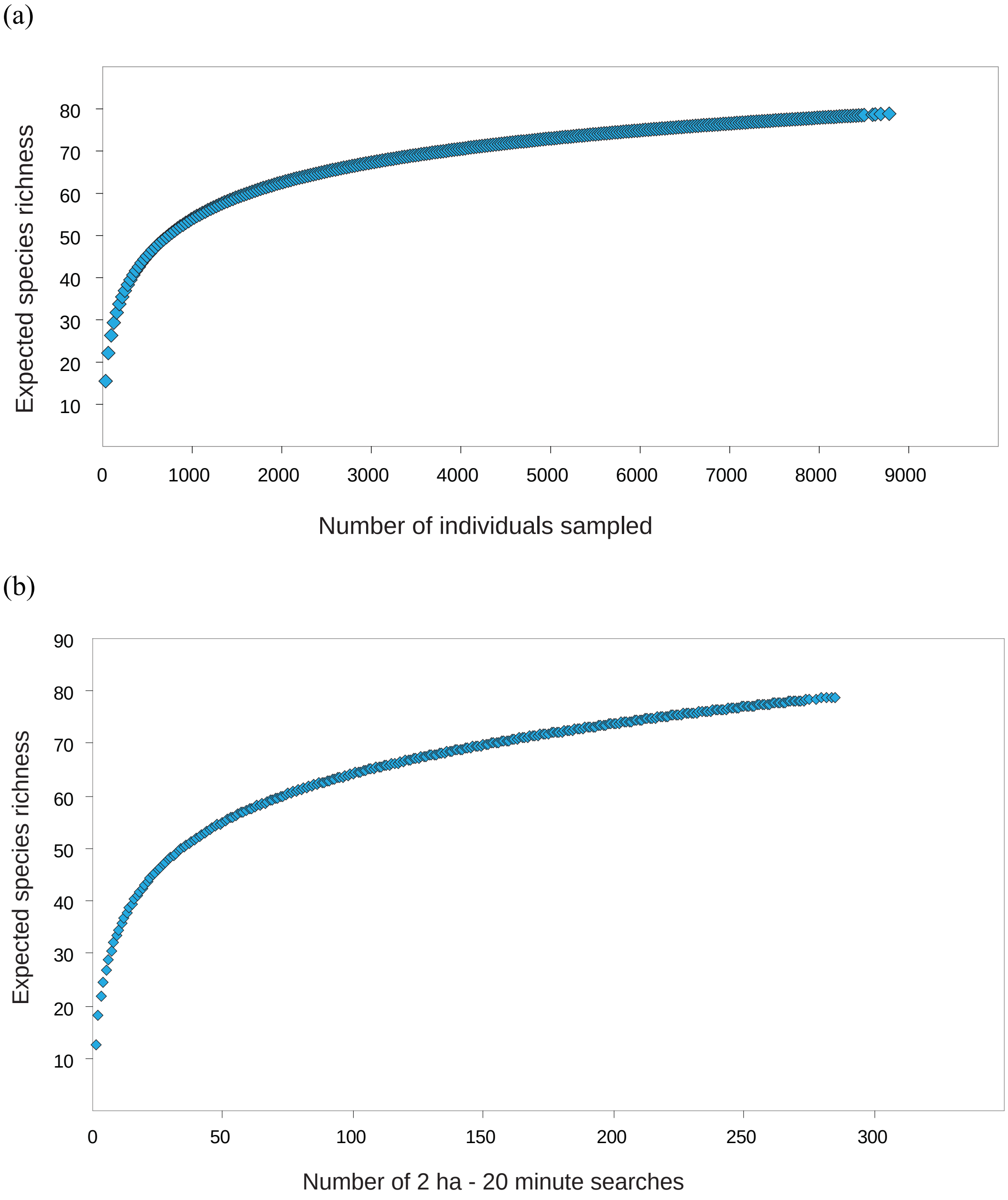

Figure 6.1 Individual- and sample-based rarefaction curves

Rarefaction curves were calculated using pooled data from all native forest sites and all site visits. (a) The individual-based rarefaction curve shows that the number of expected species for a given number of individuals (assuming spatial homogeneity) has almost reached an asymptote for native forest. (b) The sample-based rarefaction curve has not yet reached an asymptote indicating that the number of species would be expected to rise with more survey effort. A large number of individuals would have to be sampled, however, for more species to be added.

The 111 species recorded in total in this study can also be compared with a regional list of 148 *Eucalyptus* open forest / woodland bird species compiled for this study from historical records (Appendix 2). Forty-two species from the regional list were not recorded in this study. The absence of 37 of these species from this research can probably be attributed to:

1. The original records being unusual records for *Eucalyptus* open forest (seven species),
2. The species being at or beyond the natural limits of their distribution within the study area (10 species),
3. The species being sparsely distributed or seasonal visitors on Cape York Peninsula (eight species),
4. The regional population having declined or the range contracted since historical records began (seven species), and
5. The species being nocturnal, and no nocturnal surveys were conducted during this research (five species) (Appendix 11).

There is no apparent explanation for the failure to record the remaining five species that have previously been recorded in *Eucalyptus* open forest / woodland on Cape York Peninsula. The five unaccounted for species were: little eagle *Hieraaetus morphnoides*, white-throated gerygone *Gerygone albogularis*, double-barred finch *Taeniopygia bichenovii*, Australian swiftlet *Aerodramus terrareginae*, and bar-breasted honeyeater *Ramsayornis fasciatus*. The absence of these species may be due to inadequate spatial or temporal replication of surveys, inadequacies of the survey procedure, or non-detection for various reasons. The estimated species richness of 90 based on data collected in this study indicates that the value of 148 species for *E. tetrodonta* open forest may include many unusual or uncommon species.

Four species were recorded during this study that are not on the regional *Eucalyptus* open forest bird species list. Three of these, however, were only recorded in mine rehabilitation and are not usually associated with open forest: red-chested button-quail *Turnix maculosus*, lovely fairy-wren *Malurus annabilis*, and red-headed honeyeater *Myzomela erythrocephala*. The other species, azure kingfisher *Ceyx azureus*, was only recorded in *E. tetrodonta* open forest near the coast where two birds frequently passed through one of the study sites between the coast and a possible nesting location in the soil and root mass of a windthrown tree.

The 14 species that were only recorded incidentally were mostly raptors (five species) and aerial foragers (four species) (Appendix 2). Of the five other species that were only recorded incidentally, one, the red-tailed black cockatoo *Calyptorhynchus banksii*, is seasonally common in the *E. tetrodonta* open forest but is highly mobile and clumped in its distribution as large flocks gather to feed on *Eucalyptus* fruit. The remaining four species are low in abundance or non-sedentary: channel-billed cuckoo, great frigatebird, lesser frigatebird and radjah shelduck.

6.2 Bird assemblages of the pre-mining native forest

For the purposes of this thesis, the term 'native forest birds' includes all bird species that were recorded in a native forest reference site during this study. Indicator species analysis identified 26 bird species that characterised the bird assemblages of the native forest reference sites (Table 6.2). These included eight bird species that were sufficiently widespread that they were recorded as present in every native forest reference site in at least one out of eight site visits (Table 6.2). Insectivore-nectarivores were the most abundant birds in native forest reference sites followed by nectarivores and foliage-gleaning insectivores.

Of the native forest reference land units, post-mining landscape analogue LU 2C sites had the highest mean species richness and mean bird abundance (Table 6.3). Mean bird abundance was lower in post-mining landscape analogue land unit 5K than in the other two native forest land units but differences were not significant. There were no significant differences in bird species composition between the three native forest reference land units. The shortfall index values, however, indicate that there was variation in bird species composition and differences in species' abundances between sites and land units (Appendix 12).

Table 6.2 Indicator species in native forest and mine rehabilitation

P-values	Mine Rehabilitation (n=31)		Native Forest (n=36)	
	Species[3]	IV[1]	Species	IV
$P \leq 0.001$	Brown Honeyeater	97	Black-backed Butcherbird (U)[2]	99
	Olive-backed Sunbird	95	Lemon-bellied Flycatcher	94
	Bar-shouldered Dove	68	Little Friarbird (U)	92
	Golden-headed Cisticola	55	Noisy Friarbird (U)	88
	Graceful Honeyeater	54	Rainbow Lorikeet (U)	88
	Yellow-spotted Honeyeater	49	Rufous Whistler	85
	White-streaked Honeyeater	36	Leaden Flycatcher	84
			Mistletoebird (U)	84
			Grey Shrike-thrush	82
			Forest Kingfisher	80
			White-bellied Cuckoo-shrike (U)	78
			Laughing Kookaburra	75
			Cicadabird	72
			White-throated Honeyeater (U)	72
			Banded Honeyeater	69
			Yellow Honeyeater (U)	69
			Varied Sittella	58
			Sulphur-crested Cockatoo	51
			Brown Treecreeper	42
			Blue-winged Kookaburra	36
			Olive-backed Oriole	35
$0.01 \geq P > 0.001$	Peaceful Dove	59	Brush Cuckoo	35
	Red-browed Finch	26	Yellow-tinted Honeyeater	31
	Chestnut-breasted Mannikin	23		
	Australian Brush-turkey	23		
$0.05 \geq P > 0.01$	Dusky Honeyeater	28	Red-winged Parrot	53
	Australasian Pipit	16	Red-backed Fairy-wren	42
	Black Butcherbird	16	Pied Imperial-pigeon	17
	Brown Quail	16		
	Lovely Fairy-wren	13		

1. (IV) = indicator value. A perfect indicator of a particular group (100) is always present in that group and is exclusive to it. Species with lower indicator values reflect species that may still be exclusive to that group but are generally less abundant. Brown treecreepers for example, with an indicator value of 42, occurred exclusively in native forest reference sites but not in all native forest reference sites.

2. Native forest bird assemblages were characterised by the native forest indicator species. The symbol (U) indicates 'ubiquitous' species, (i.e. species that were recorded at least once in every native forest site).

3. The indicator species for mine rehabilitation sites were more likely to be recorded in mine rehabilitation than in native forest. Some of these species, however, characterised young age classes of mine rehabilitation whereas others characterised older age classes of mine rehabilitation.

Table 6.3 Community level bird statistics by age class and land unit

	Mine rehabilitation sites					Native forest reference sites		
						Pre-mining	Post-mining landscape analogues	
	1 - 2 y (n=6)	**3 - 4 y (n=6)**	**5 - 8 y (n=6)**	**9 - 16 y (n=8)**	**> 16 y (n=5)**	**LU 2B (n=28)**	**LU 2C (n=4)**	**LU 5K (n=4)**
Mean site species richness	9 (27%)	15 (22%)	20 (29%)	25 (18%)	28 (11%)	27 (15%)	29 (8%)	29 (16%)
Mean bird species richness per visit	3 (58%)	5 (34%)	6 (47%)	9 (37%)	10 (27%)	12 (24%)	14 (23%)	13 (26%)
Mean bird abundance per visit	5 (99%)	11 (65%)	14 (61%)	18 (44%)	20 (37%)	31 (30%)	34 (33%)	28 (32%)
Mean number of foraging groups	5 (33%)	8 (14.5%)	9 (12%)	9 (12%)	9 (8%)	10 (11%)	10 (12%)	11 (9%)
Mean shortfall index %	98 (1.3%)	90 (10.4%)	85 (12.7%)	79 (10.7%)	63 (19%)	27 (20.4%)	25 (14%)	34 (20%)

Age class and land unit means were calculated from site totals and site means. Values shown in brackets are coefficients of variation.

6.3 Comparison of bird assemblages

Site category, season, time since fire, and weather conditions all caused variation in bird abundance (Table 6.4). Mean bird abundance was highest in the early dry season of 2007 and lowest in the late dry season 2006 (Appendix 13). Mean bird abundance was highest in sites that had been burnt in the week of the survey. Generally, mean bird abundance was higher when it was cloudless and when there was no wind although the highest value for mean bird abundance was when there was no wind and some cloud cover. These effects may have been due to changes in either bird detectability or bird activity under different survey conditions. Site category, however, had by far the largest effect on bird abundance (Table 6.4).

Table 6.4 ANOVA results for bird abundance data

Site category	Season	Fire	Cloud*Wind
F (5, 289)	F (3, 335)	F (4, 478)	F (11, 498)
206.8 ***	33.35 ***	3.04 *	2.31 **

Bird data from all sites and all visits was used. A power model was fitted for the correlation structure between successive site visits. Data were pooled into six site categories: (1) 1 to 2 year old rehabilitation sites; (2) 3 to 4 year old rehabilitation sites; (3) 5 to 8 year old rehabilitation sites; (4) 9 to 16 year old rehabilitation sites; (5) > 16 year old rehabilitation sites; and (6) all native forest sites (Appendix 13). Bird abundance data were square root transformed for analysis to meet assumptions of normality. Significance of the F-statistic is indicated by asterisks: * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$; and *** = $P \leq 0.001$. ns = not significant.

6.3.1 Mean bird abundance

There was a clear trend of increasing mean bird abundance per site visit and decreasing coefficients of variation in values with rehabilitation age (Figure 6.2) and (Table 6.3). Mean bird abundance per site visit in the oldest age class of mine rehabilitation was considerably lower than the values for all three native forest reference land units. Pairwise comparison of means using the Student's t-test found that differences in mean bird abundance between the three native forest reference land units and the oldest age class of mine rehabilitation were all significant (*P* < 0.01).

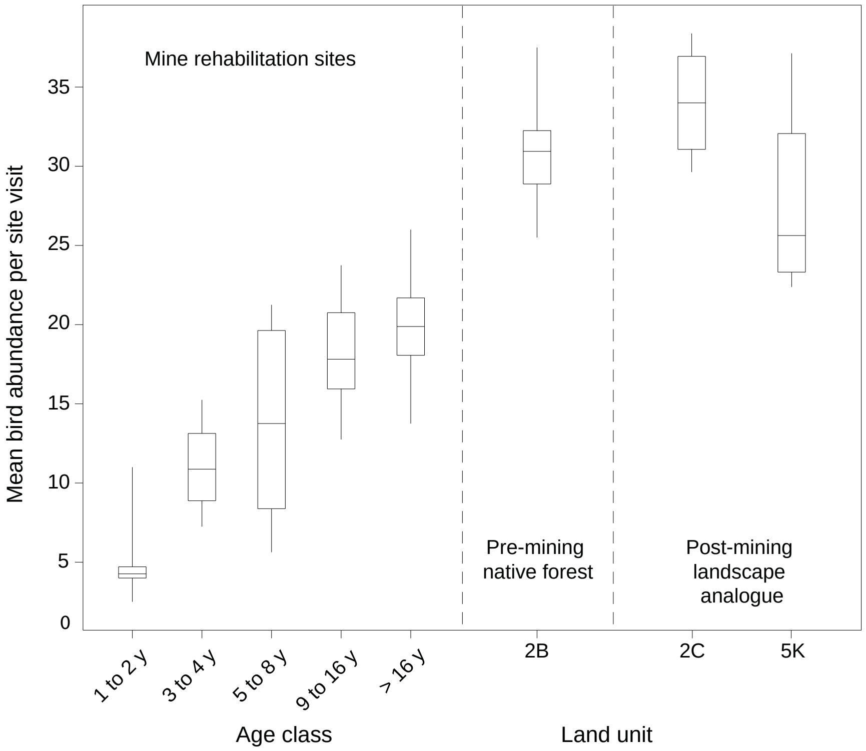

Figure 6.2 Increasing bird abundance with age of site

6.3.2 Species richness

Values for mean site species richness increased and coefficients of variation decreased in mine rehabilitation sites with increasing rehabilitation age (Figure 6.3) and (Table 6.3) Mean site species richness in the oldest age class of mine rehabilitation was comparable to values in all three native forest reference land units. Pairwise comparison of means using Student's t-tests found that differences in the mean site species richness values between the two oldest age classes of mine rehabilitation and the three native forest reference land units were not significant. All other pairwise comparisons of means were significantly different ($P < 0.01$).

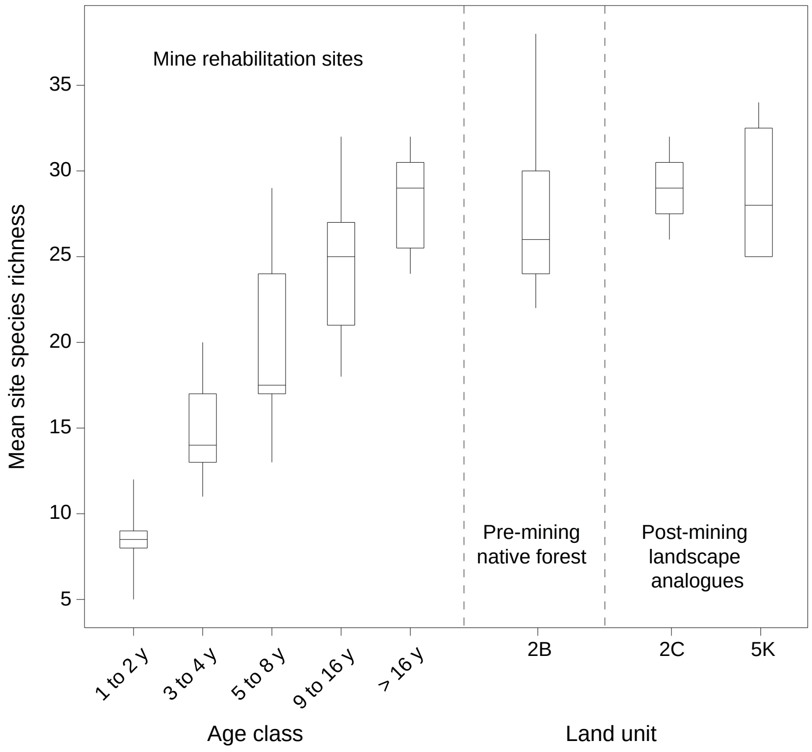

Figure 6.3 Increasing species richness with age of site

6.3.3 Composition of bird assemblages

There were significant differences in bird species composition between native forest reference sites and mine rehabilitation sites (Table 6.5). Bird species composition changed with rehabilitation age. Differences in bird species composition between the oldest age class of mine rehabilitation and pre-mining native forest reference LU 2B were also significant (Table 6.5). Pairwise comparison of bird species composition found no significant difference in bird species composition between the three native forest reference land units. Differences between the native forest reference land units and the oldest age class of mine rehabilitation were all significant (Appendix 14).

Table 6.5 MRPP results (bird species composition)

Comparison of species composition	T	A
Native forest cf. mine rehabilitation	-36 ***	0.21
Age classes / Land units	-17 ***	0.29
Premining native forest LU 2B cf. oldest age class of mine rehabilitation	-13 ***	0.12

Site presence/absence data for the 97 bird species recorded during surveys were used. The test statistic (T) describes the separation between *a priori* defined groups. The more negative is T, the stronger the separation. Significance of the T-statistic is indicated by asterisks: * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$; and *** = $P \leq 0.001$. ns = not significant. The chance corrected within group agreement (A) is a description of the effect size, independent of the sample size and statistical significance. Values for $A > 0.3$ are considered high (McCune and Grace, 2002).

6.3.4 Site species shortfall index

The site species shortfall index provides a measure for each site of the percentage difference in bird species composition and abundance, relative to the pre-mining native forest reference LU 2B condition. It therefore provides different information to species richness or mean abundance. Rather than simply counting the number of species, it is specific about which species are present and how abundant they are. Mine rehabilitation sites, for example, may have similar species richness values to native forest reference sites, but lack many of the species that are in native forest reference sites and provide habitat to species that are not part of the native forest reference bird assemblage.

None of the native forest reference sites had the full complement of native forest bird species. Species abundance also varied between native forest sites. No site, therefore, had a species shortfall index of 0% (Appendix 12). The lowest species shortfall index of any site was 16.8% for native forest site A-05, a pre-mining native forest reference LU 2B site.

Site species shortfall index values in mine rehabilitation sites decreased (i.e., became more similar to the pre-mining native forest reference sites) with increasing age (Figure 6.4). However, the mean species shortfall index in the oldest age class of mine rehabilitation was 63%. This value was much higher than the mean species shortfall index values in the three native forest reference land units (Table 6.3). Pairwise comparison of means found that differences in species shortfall index between the oldest age class of mine rehabilitation and all native forest reference land units were significant ($P < 0.01$). Differences between the pre-mining native forest reference LU 2B and the two post-mining analogue reference land units, 2C and 5K, were not significant.

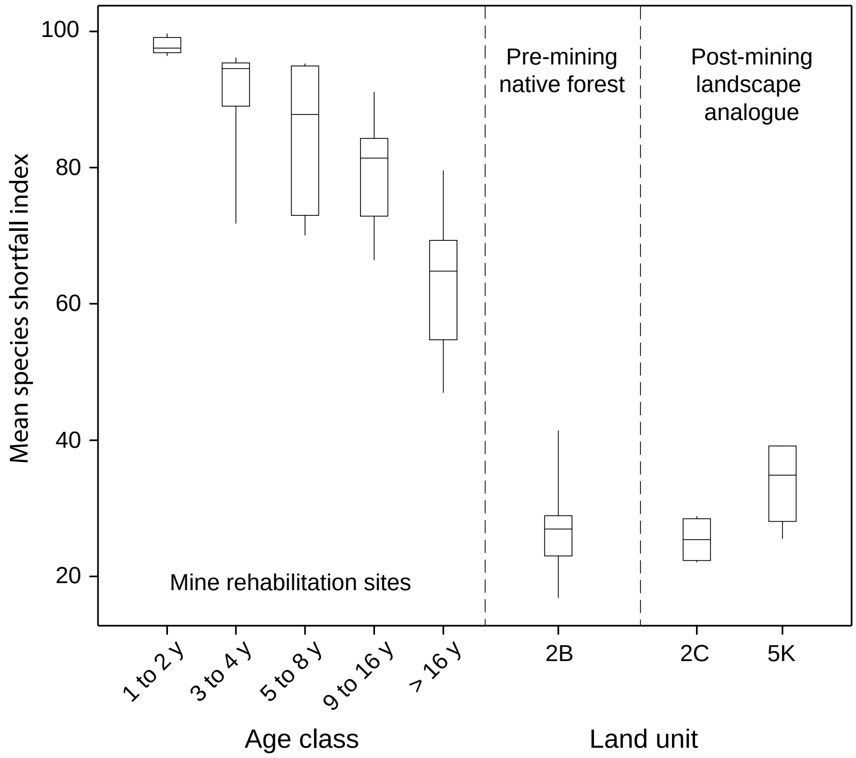

Figure 6.4 Decreasing species shortfall index with age of site

6.3.5 Comparison by individual species

Out of all 97 bird species recorded during surveys 25% (24/97) were recorded exclusively in native forest reference sites (Table 6.6). Nineteen percent (18/97) were recorded exclusively in mine rehabilitation sites (Table 6.7), and 56% (55/97) were recorded in both native forest and mine rehabilitation sites (Table 6.8). Excluding the bird species that were only recorded in mine rehabilitation sites, 79 native forest birds were recorded during surveys. Of these, 30% (24/79) were only recorded in native forest. Of the 14 species recorded incidentally, a further 11 species were only recorded in native forest, two were recorded exclusively in mine rehabilitation, and one was shared (Appendix 2).

Three aspects of species occurrence were compared between native forest and mine rehabilitation. First, the percentage of sites in which a species was recorded. The percentage of sites is assumed here to be a measure of a species' spatial saturation regardless of abundance or frequency. Second, the mean detection rate (i.e., number of times out of eight visits in which a species was recorded) was compared. Detection rate is assumed here to be a measure of a species' temporal saturation regardless of abundance and spatial distribution. Third, the mean abundance of each species was compared. Mean abundance is assumed here to be a measure of the number of individuals of a species, regardless of its' spatial or temporal distribution.

Profile graphs were used to illustrate the differences in these three aspects of bird species occurrence between native forest reference sites and mine rehabilitation sites for the 55 shared species (i.e., species that were recorded in both native forest and mine rehabilitation sites). As the results for mean abundance and mean detection rate were almost identical, graphs are only shown for (1) the percentage of sites in which shared species were recorded (Figure 6.5) and (2) for the mean detection rates of shared species (Figure 6.6). For each species the pattern of response was consistent across the three measures (Table 6.8). Species that occurred in a higher percentage of native forest sites also had higher detection rates in native forest sites and higher mean abundance in native forest sites.

Of the shared species, 40% (22/55) had significantly higher detection rates, were significantly more abundant and occurred in a significantly higher percentage of native forest reference sites than in mine rehabilitation sites (Table 6.8). Ten percent of the shared species (6/55) were significantly more abundant in mine rehabilitation sites than in native forest sites (Table 6.8). Of these six species, five also had significantly higher detection rates in mine rehabilitation sites (Figure 6.6). Differences in the mean abundance and mean detection rates of the remaining 50% (28/55) of shared bird species were not significant due to generally low numbers (Table 6.8). The exception to this was the striated pardalote *Pardalotus striatus*, which had a mean detection rate > 0.3 in both native forest reference sites and mine rehabilitation sites.

Table 6.6 Bird species recorded exclusively in native forest reference sites

Scientific name	Common name	Total number of sites in which recorded (n=67)	% of mine sites in which recorded (n=31)	% of reference sites in which recorded (n=36)	Mean detection rate in mine sites (n=31)	Mean detection rate in reference sites (n=36)	Mean abundance in mine sites (n=31)	Mean abundance in reference sites (n=36)
Dromaius novaehollandiae	Emu	2	0	6	0	0.007	0	0.014
Megapodius reinwardt	Orange-footed Scrubfowl	1	0	3	0	0.003	0	0.003
Ducula bicolor	Pied Imperial-pigeon	6	0	**17**	0	**0.024**	0	**0.038**
Podargus strigoides	Tawny Frogmouth	4	0	11	0	0.014	0	0.017
Hirundapus caudacutus	White-throated Needletail	3	0	8	0	0.010	0	0.014
Aviceda subcristata	Pacific Baza	2	0	6	0	0.007	0	0.010
Accipiter novaehollandiae	Grey Goshawk	1	0	3	0	0.003	0	0.003
Aquila audax	Wedge-tailed Eagle	1	0	3	0	0.007	0	0.014
Falco peregrinus	Peregrine Falcon	2	0	6	0	0.010	0	0.014
Probosciger aterrimus	Palm Cockatoo	1	0	3	0	0.003	0	0.003
Cuculus optatus	Oriental Cuckoo	1	0	3	0	0.003	0	0.003
Ninox novaeseelandiae	Southern Boobook	1	0	3	0	0.003	0	0.003
Eurystomus orientalis	Dollarbird	1	0	3	0	0.003	0	0.003
Climacteris picumnus	Brown Treecreeper	15	0	**42**	0	**0.177**	0	**0.306**
Pardalotus rubricatus	Red-browed Pardalote	1	0	3	0	0.007	0	0.021
Lichenostomus flavescens	Yellow-tinted Honeyeater	11	0	**31**	0	**0.090**	0	**0.358**
Entomyzon cyanotis	Blue-faced Honeyeater	3	0	8	0	0.010	0	0.024
Pomatostomus temporalis	Grey-crowned Babbler	5	0	14	0	0.024	0	0.059
Daphoenositta chrysoptera	Varied Sittella	21	0	**58**	0	**0.160**	0	**0.792**
Pachycephala simplex	Grey Whistler	1	0	3	0	0.007	0	0.007
Artamus minor	Little Woodswallow	1	0	3	0	0.003	0	0.007
Rhipidura albiscapa	Grey Fantail	2	0	6	0	0.007	0	0.007
Rhipidura rufiventris	Northern Fantail	1	0	3	0	0.003	0	0.010
Grallina cyanoleuca	Magpie-lark	1	0	3	0	0.003	0	0.003

Equality of values was tested using Mann –Whitney U-tests. Where there is a significant difference the higher of the two values is indicated in bold.

Table 6.7 Bird species recorded exclusively in mine rehabilitation sites

Scientific name	Common name	Total number of sites in which recorded (n=67)	% of mine sites in which recorded (n=31)	% of reference sites in which recorded (n=36)	Mean detection rate in mine sites (n=31)	Mean detection rate in reference sites (n=36)	Mean abundance in mine sites (n=31)	Mean abundance in reference sites (n=36)
Alectura lathami	Australian Brush-turkey	7	**23**	0	**0.044**	0	**0.065**	0
Coturnix ypsilonophora	Brown Quail	5	16	0	0.020	0	0.020	0
Caprimulgus marrurus	Large-tailed Nightjar	1	3	0	0.004	0	0.004	0
Falco longipennis	Australian Hobby	1	3	0	0.004	0	0.004	0
Turnix pyrrhothorax	Red-chested Button-quail	3	10	0	**0.020**	0	**0.032**	0
Cacomantis flabelliformis	Fan-tailed Cuckoo	1	3	0	0.004	0	0.004	0
Syma torotoro	Yellow-billed Kingfisher	1	3	0	0.004	0	0.004	0
Todiramphus sanctus	Sacred Kingfisher	2	6	0	**0.012**	0	**0.012**	0
Malurus annabilis	Lovely Fairy-wren	4	13	0	**0.024**	0	**0.060**	0
Gerygone palpebrosa	Fairy Gerygone	3	10	0	**0.020**	0	**0.028**	0
Myzomela erythrocephala	Red-headed Honeyeater	1	3	0	0.004	0	0.004	0
Lichmera indistincta	Brown Honeyeater	30	**97**	0	**0.589**	0	**2.012**	0
Trichodere cockerelli	White-streaked Honeyeater	11	**35**	0	**0.093**	0	**0.157**	0
Cracticus quoyi	Black Butcherbird	5	16	0	**0.036**	0	**0.040**	0
Cisticola exilis	Golden-headed Cisticola	17	**55**	0	**0.222**	0	**0.403**	0
Neochmia phaeton	Red-browed Finch	8	**26**	0	**0.044**	0	**0.129**	0
Lonchura castaneothorax	Chestnut-breasted Mannikin	7	**23**	0	**0.036**	0	**0.230**	0
Anthus novaeseelandiae	Australasian Pipit	5	16	0	**0.052**	0	**0.093**	0

Equality of values was tested using Mann –Whitney U-tests. Where there is a significant difference the higher of the two values is indicated in bold.

Table 6.8 Bird species recorded in both native forest reference sites and mine rehabilitation sites

Scientific name	Common name	Total number of sites in which recorded (n=67)	% of mine sites in which recorded (n=31)	% of reference sites in which recorded (n=36)	Mean detection rate in mine sites (n=31)	Mean detection rate in reference sites (n=36)	Mean abundance in mine sites (n=31)	Mean abundance in reference sites (n=36)
Geopelia striata	Peaceful Dove	45	**81**	56	**0.278**	0.163	**0.569**	0.208
Geopelia humeralis	Bar-shouldered Dove	51	87	67	**0.379**	0.174	**0.927**	0.264
Podargus papuensis	Papuan Frogmouth	2	3	3	0.004	0.003	0.004	0.007
Haliastur sphenurus	Whistling Kite	14	13	28	0.016	0.035	0.016	0.042
Haliastur indus	Brahminy Kite	5	10	6	0.012	0.007	0.012	0.007
Milvus migrans	Black Kite	2	3	3	0.004	0.003	0.004	0.003
Accipiter fasciatus	Brown Goshawk	7	6	14	0.008	0.021	0.008	0.021
Accipiter cirrhocephalus	Collared Sparrowhawk	14	16	25	0.020	0.042	0.020	0.042
Falco berigora	Brown Falcon	4	6	6	0.008	0.007	0.012	0.007
Burhinus grallarius	Bush Stone-curlew	4	6	6	0.008	0.007	0.012	0.010
Cacatua sanguinea	Sulphur-crested Cockatoo	29	19	**64**	0.052	**0.181**	0.085	**0.330**
Trichoglossus haematodus	Rainbow Lorikeet	54	58	**100**	0.145	**0.771**	0.601	**4.212**
Aprosmictus erythropterus	Red-winged Parrot	46	55	**81**	0.117	**0.233**	0.262	**0.514**
Platycercus adscitus	Pale-headed Rosella	34	42	58	0.093	0.122	0.169	0.250
Centropus phasianus	Pheasant Coucal	14	16	25	0.036	0.042	0.040	0.049
Eudynamys orientalis	Common Koel	11	10	22	0.012	0.038	0.012	0.052
Chrysococcyz minutillus	Little Bronze-cuckoo	2	3	3	0.004	0.003	0.004	0.014
Cacomantis variolosus	Brush Cuckoo	14	3	**36**	0.004	**0.080**	0.004	**0.094**
Dacelo novaeguinae	Laughing Kookaburra	32	10	**81**	0.028	**0.316**	0.048	**0.611**
Dacelo leachii	Blue-winged Kookaburra	16	6	**39**	0.008	**0.076**	0.012	**0.128**
Todiramphus macleayii	Forest Kingfisher	35	16	**83**	0.024	**0.382**	0.024	**0.517**
Merops ornatus	Rainbow Bee-eater	22	26	39	0.048	0.090	0.073	0.201
Ptilonorhynchus nuchalis	Great Bowerbird	16	**35**	14	0.069	0.059	0.073	0.059
Malurus melanocephalus	Red-backed Fairy-wren	32	32	**61**	0.089	**0.188**	0.226	**0.344**
Smicromis brevirostris	Weebill	13	29	11	0.065	0.045	0.077	0.101
Pardalotus striatus	Striated Pardalote	57	**97**	75	0.339	0.309	0.480	0.458
Meliphaga notata	Yellow-spotted Honeyeater	18	**52**	6	**0.254**	0.021	**0.601**	0.031
Meliphaga gracilis	Graceful Honeyeater	24	**65**	11	**0.218**	0.045	**0.464**	0.087
Lichenostomus flavus	Yellow Honeyeater	55	**61**	100	**0.298**	0.611	**0.435**	0.972

Table 6.8 Bird species recorded in both native forest reference sites and mine rehabilitation sites[1] - continued

Scientific name	Common name	Total number of sites in which recorded (n=67)	% of mine sites in which recorded (n=31)	% of reference sites in which recorded (n=36)	Mean detection rate in mine sites (n=31)	Mean detection rate in reference sites (n=36)	Mean abundance in mine sites (n=31)	Mean abundance in reference sites (n=36)
Ramsayornis modestus	Brown-backed Honeyeater	12	16	19	0.036	0.031	0.065	0.045
Myzomela obscura	Dusky Honeyeater	16	32	17	0.125	0.024	0.202	0.028
Cissomela pectoralis	Banded Honeyeater	37	32	**75**	0.052	**0.222**	0.109	**1.354**
Melithreptus albogularis	White-throated Honeyeater	54	58	**100**	0.395	**1.000**	1.375	**5.566**
Philemon buceroides	Helmeted Friarbird	2	3	3	0.004	0.003	0.004	0.003
Philemon buceroides	Noisy Friarbird	47	35	**100**	0.073	**0.545**	0.137	**1.003**
Philemon citreogularis	Little Friarbird	53	55	**100**	0.177	**0.920**	0.278	**3.066**
Coracina novaehollandiae	Black-faced Cuckoo-shrike	24	26	44	0.036	0.080	0.125	0.122
Coracina papuensis	White-bellied Cuckoo-shrike	55	61	**100**	0.226	**0.712**	0.355	**1.257**
Coracina tenuirostris	Cicadabird	29	6	**75**	0.008	**0.160**	0.008	**0.198**
Lalage sueurii	White-winged Triller	9	6	19	0.012	0.028	0.016	0.038
Lalage leucomela	Varied Triller	6	**16**	3	0.028	0.003	**0.028**	0.003
Pachycephala rufiventris	Rufous Whistler	34	10	**86**	0.020	**0.691**	0.020	**1.163**
Collurincla harmonica	Grey Shrike-thrush	31	3	**83**	0.008	**0.406**	0.008	**0.500**
Sphechotheres vieilloti	Figbird	2	3	3	0.004	0.017	0.004	0.028
Oriolus flavocinctus	Yellow Oriole	5	3	11	0.004	0.028	0.004	0.038
Oriolus sagittatus	Olive-backed Oriole	16	6	**39**	0.008	**0.063**	0.008	**0.069**
Artamus leucorynchus	White-breasted Woodswallow	7	6	14	0.008	0.017	0.012	0.111
Cracticus mentalis	Black-backed Butcherbird	38	6	**100**	0.008	**0.615**	0.012	**1.003**
Cracticus nigrogularis	Pied Butcherbird	6	3	14	0.004	0.031	0.004	0.045
Dicrurus bracteatus	Spangled Drongo	26	39	39	0.077	0.076	0.097	0.080
Corvus orru	Torresian Crow	22	26	39	0.048	0.056	0.149	0.090
Myiagra rubecula	Leaden Flycatcher	51	52	**97**	0.145	**0.646**	0.173	**1.063**
Microeca flavigaster	Lemon-bellied Flycatcher	42	23	**97**	0.052	**0.757**	0.052	**1.670**
Dicaeum hirundinaceum	Mistletoebird	56	65	**100**	0.173	**0.674**	0.194	**0.983**
Nectarinia jugularis	Olive-backed Sunbird	34	**97**	11	**0.730**	0.017	**1.544**	0.024

Equality of values was tested using Mann –Whitney U-tests. Where there is a significant difference the higher of the two values is indicated in bold.

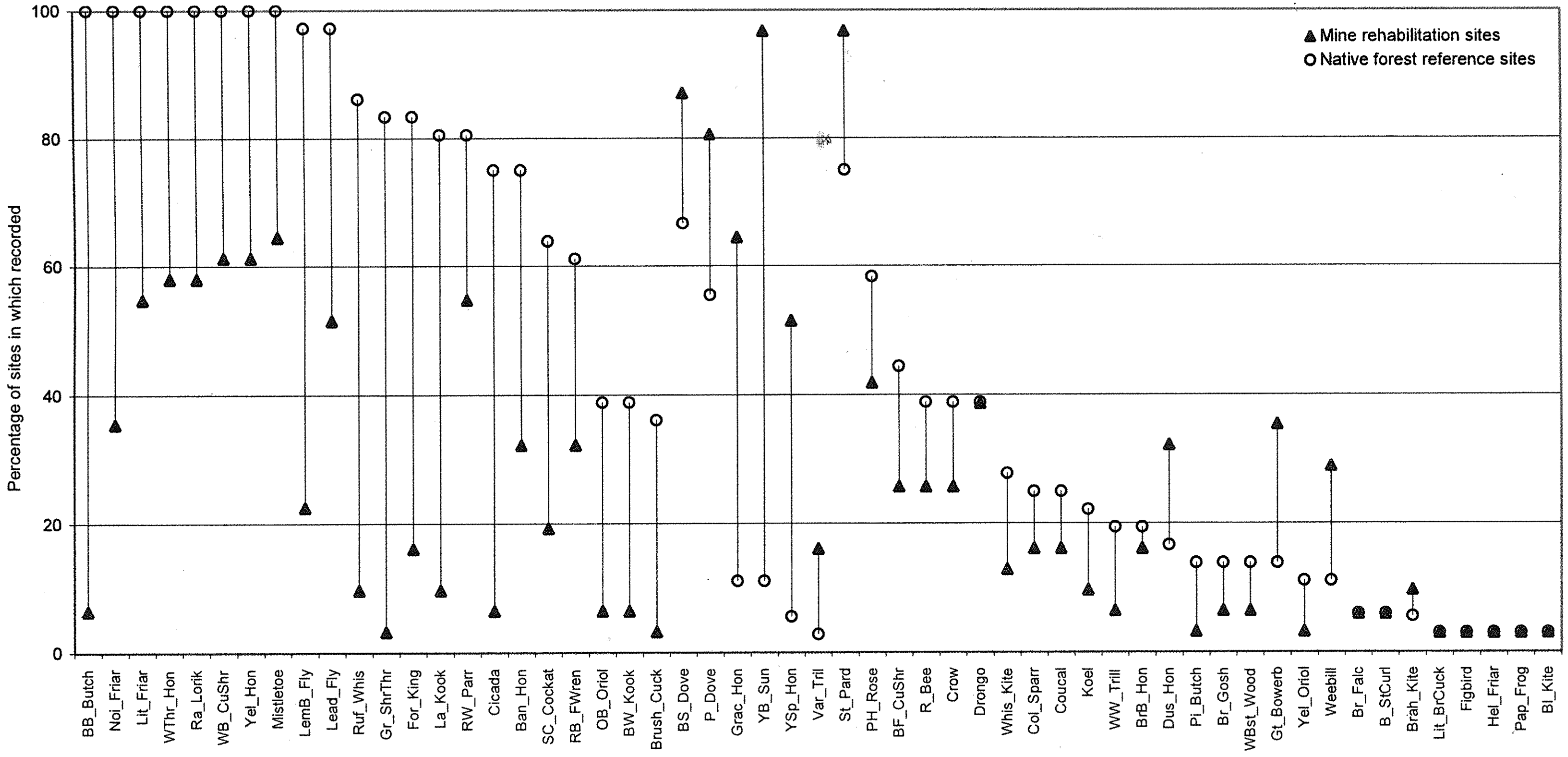

Figure 6.5 Profile graph of percentage of sites in which shared species were recorded

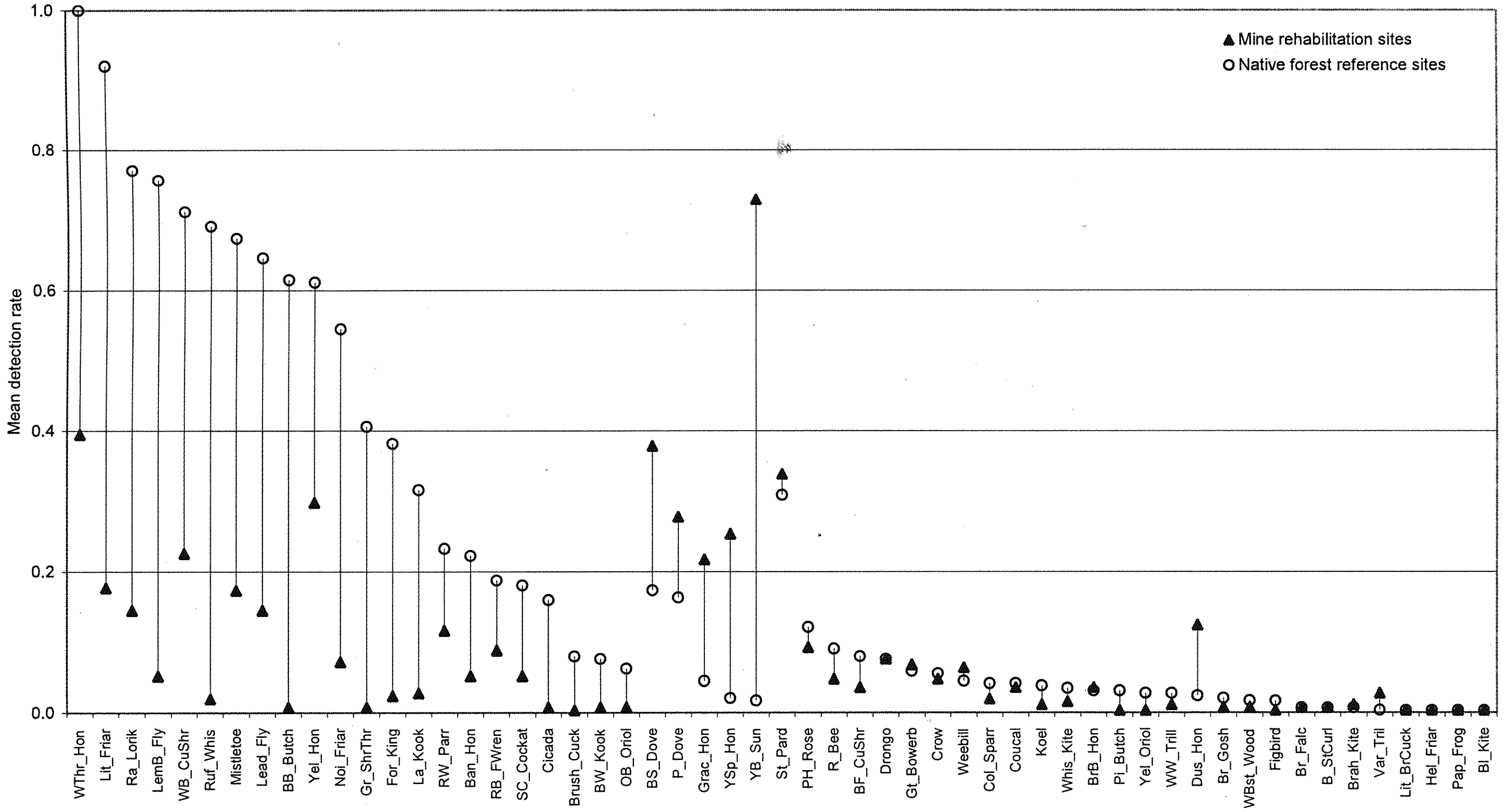

Figure 6.6 Profile graph of mean detection rates of shared species

6.3.6 Comparison by foraging group

Bird assemblages were dominated by insectivore-nectarivores across all native forest land units and age classes of mine rehabilitation. Differences between native forest reference sites and mine rehabilitation sites in the relative abundance of foraging groups and mean abundance of individuals within foraging groups reflected differences in vegetation structure. Foraging groups that had significantly higher mean abundance in mine rehabilitation sites were species that use open ground and grass or low shrubs (Table 6.9). The foraging group that each species was assigned to is shown in Appendix 15.

Table 6.9 Mean abundances of foraging groups

Foraging group	Mean abundance in native forest reference sites (n=36)	Mean abundance in mine rehabilitation sites (n=31)
Carnivore	1.5 (± 0.3)	0.9 (± 0.2)
Frugivore	**10.3 (± 1.1)**	2.4 (± 0.5)
Granivore	3.8 (± 0.6)	**15.3 (± 2.8)**
Ground forager	0.03 (± 0.03)	**0.5 (± 0.2)**
Insectivore - aerial	1.1 (± 0.6)	0.1 (± 0.1)
Insectivore - foliage-gleaner	**34.6 (± 2.2)**	11.1 (± 1.5)
Insectivore - grass and ground	0.5 (± 0.2)	**4.9 (± 1.0)**
Insectivore - hawker	**24.3 (± 1.9)**	3.1 (± 0.7)
Insectivore - trunk-gleaner	**9.3 (± 1.4)**	0
Insectivore-nectarivore	**92.3 (± 3.3)**	45.6 (± 5.5)
Nectarivore	**41.9 (± 2.9)**	18.3 (± 2.4)
Omnivore	0.7 (± 0.2)	1.2 (± 0.6)
Pouncing predator	**18.4 (± 1.3)**	0.9 (± 0.4)
Seed Eater (trees)	**8.8 (± 1.0)**	4.1 (± 0.9)

Values shown in brackets are 1 SE. The mean number of individuals of each foraging group was calculated from pooled site data. Equality of values was tested using Mann-Whitney U-tests. Where a statistically significant difference was found the higher of the two values is shown in bold.

Foraging groups that had significantly higher mean abundance in native forest reference sites included groups that use different components of mature vegetation including fruit (mainly mistletoe), the vegetation canopy, nectar, perches and canopy gaps between trees, tree trunks and branches, and tree seed. Trunk-gleaning insectivores, species that probe the bark of mature trees for insects, were entirely absent from mine rehabilitation sites.

6.4 Temporal patterns in bird succession

6.4.1 Temporal patterns in individual bird species

All age classes of mine rehabilitation included at least some bird species that were not recorded in any of the native forest reference sites (Table 6.10). Apart from the brown honeyeater *Lichmera indistincta*, which was present in all age classes of mine rehabilitation, there was turnover in these species with increasing age class. Only the striated pardalote had a mean detection rate >0.08 across all age classes of mine rehabilitation and native forest reference sites. When the detection rate of individual bird species was analysed by age class three patterns emerged:

(i) turnover of species in young rehabilitation sites, most of which are associated with open ground, grassy habitats or heathy habitats;

(ii) accumulation of species that are generally associated with mesic habitats with increasing rehabilitation age; and

(iii) accumulation of native forest species with increasing rehabilitation age (Table 6.10).

Table 6.10 Species detection rates by age class

Temporal pattern	Common name	Mine rehabilitation sites					Native Forest
		1 - 2 y	3 - 4 y	5 - 8 y	9 - 16 y	> 16 y	
Turnover of species in young rehabilitation sites	Australasian Pipit	0.21					
	Red-browed Finch	0.08			0.09		
	Australian Brush turkey		0.1				
	Pheasant Coucal*			0.08			
	Torresian Crow*			0.15			
	White-streaked Honeyeater			0.23	0.16		
	Golden-headed Cisticola	0.42	0.23	0.19	0.23		
Accumulation of species with increasing rehabilitation age	Olive-backed Sunbird	0.23	0.85	0.77	0.94	0.8	
	Brown Honeyeater	0.33	0.71	0.77	0.66	0.43	
	Yellow-spotted Honeyeater		0.08	0.08	0.45	0.65	
	Graceful Honeyeater*			0.21	0.33	0.48	
	Weebill*				0.08	0.2	
	Dusky Honeyeater				0.19	0.45	
	Black Butcherbird				0.09	0.08	
	Brown-backed Honeyeater					0.1	
	Varied Triller*					0.08	
All age classes	Striated Pardalote	0.27	0.29	0.21	0.47	0.43	0.31
Accumulation of predominantly native forest species with increasing rehabilitation age	Mistletoebird		0.21	0.15	0.2	0.28	0.67
	Peaceful Dove		0.29	0.21	0.45	0.35	0.16
	Bar-shouldered Dove		0.42	0.44	0.59	0.3	0.17
	Yellow Honeyeater		0.1	0.25	0.39	0.75	0.61
	White-bellied Cuckoo-shrike		0.15	0.29	0.22	0.53	0.71
	Little Friarbird		0.08	0.13	0.23	0.45	0.92
	Black-faced Cuckoo-shrike		0.08				0.08
	Pale-headed Rosella			0.08	0.13	0.18	0.12
	Red-winged Parrot			0.15	0.20	0.20	0.23
	Rainbow Lorikeet			0.08	0.20	0.40	0.77
	White-throated Honeyeatear			0.33	0.63	0.98	1.00
	Leaden Flycatcher			0.17	0.22	0.33	0.65
	Noisy Friarbird			0.15		0.10	0.55
	Banded Honeyeater			0.10			0.22
	Lemon-bellied Flycatcher				0.14	0.10	0.76
	Drongo				0.13	0.23	0.08
	Red-backed Fairy-wren				0.28		0.19
	Rainbow Bee-eater					0.18	0.09
	Laughing Kookaburra					0.08	0.32
	Sulphur-crested Cockatoo					0.25	0.18
	Forest Kingfisher					0.08	0.38
Species exclusive to native forest or with mean detection rates < 0.08 in mine rehabilitation	Brown Treecreeper						0.18
	Varied Sittella						0.16
	Yellow-tinted Honeyeater						0.09
	Brush Cuckoo						0.08
	Grey Shrike-thrush						0.41
	Blue-winged Kookaburra						0.08
	Black-backed Butcherbird						0.61
	Cicadabird						0.16
	Rufous Whistler						0.69

Only values ≥ 0.08 are shown. Asterisks indicate species that are generally associated with native forest and were recorded in native forest but have been placed in these categories as they had mean detection rates < 0.08 in native forest.

6.4.2 Temporal patterns in bird community structure

The youngest age class of mine rehabilitation had the lowest mean number of foraging groups per site. There was a step up in the mean number of foraging groups per site in 3 to 4 year old mine rehabilitation sites with the addition of four foraging groups (Table 6.11). Subsequent to this the number of foraging groups stayed more or less the same with some turnover. Native forest reference sites had the highest number of foraging groups.

Foraging group richness, (i.e., the number of species within each foraging group), also generally increased with rehabilitation age although the pattern was not simply one of accumulation (Table 6.11). The species richness of seven foraging groups was highest in native forest sites. The species richness of insectivore-nectarivores, and grass and ground foraging insectivores was highest in 5 to 8 year old mine rehabilitation sites. Granivore species richness was highest in 9 to 16 year old mine rehabilitation sites and pouncing predator species richness was highest in the oldest age class of mine rehabilitation (Table 6.11).

Granivore abundance was also highest in 9 to 16 year old mine rehabilitation sites (Figure 6.7). The abundance of grass and ground foraging insectivores was highest in the youngest age class of mine rehabilitation and gradually declined with increasing rehabilitation age. The abundance of all other foraging groups generally increased with increasing rehabilitation age (Figure 6.7). The relative abundance of all foraging groups also changed and in the native forest reference sites was more evenly distributed between foraging groups, probably reflecting the maturity and layering of the vegetation structure.

Table 6.11 Number and richness of foraging groups by age class[1]

Foraging group	**Mine rehabilitation sites**					**Native forest**
	1 - 2 y	**3 - 4 y**	**5 - 8 y**	**9 - 16 y**	**> 16 y**	
Carnivore	2	5	4	2	5	**13**
Frugivore	1	2	4	3	5	**8**
Granivore	5	4	4	**6**	3	2
Ground forager	0	1	1	1	0	1
Insectivore - aerial	0	1	1	0	1	**3**
Insectivore - foliage-gleaner	3	5	8	10	10	**15**
Insectivore – grass & ground forager	2	3	**5**	3	1	3
Insectivore - hawker	0	2	3	4	4	**7**
Insectivore - trunk-gleaner	0	0	0	0	0	**3**
Insectivore-nectarivore	5	7	**11**	10	10	10
Nectarivore	1	3	3	3	4	4
Omnivore	1	1	1	0	0	1
Pouncing predator	0	1	1	5	**6**	5
Seed eater (trees)	1	2	3	3	3	**4**
Total number of foraging groups	**9**	**13**	**13**	**11**	**11**	**14**

1. Values shown in bold indicate the highest values but do not indicate statistical significance.

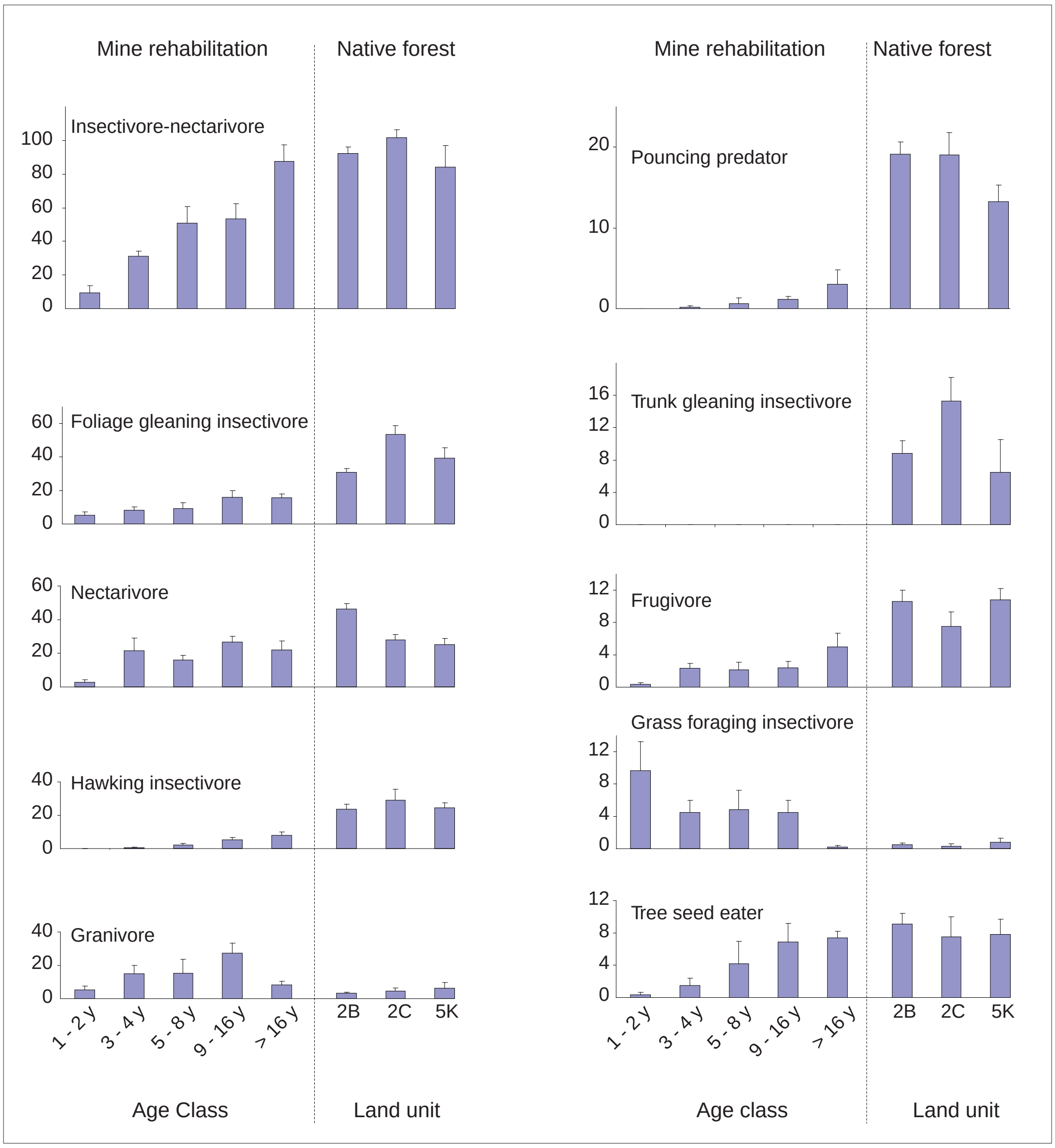

Figure 6.7 Mean foraging group abundance by age class and land unit

The vertical scale of the five bar graphs on the right hand side has been increased to emphasise patterns. Otherwise, all bar graphs on the left hand side are of the same vertical scale and all bar graphs on the right hand side are of the same vertical scale. Four foraging groups that had low overall abundance have not been included in this diagram: carnivores, ground foragers, aerial insectivores and omnivores.

6.4.3 Bird species associated with mature framework trees

Nineteen of the bird species that occurred in five or more sites were consistently associated with mature framework trees (i.e., area of remnant vegetation, vegetation height, stem density of big trees and canopy volume of *Eucalyptus* spp. in CCA ordinations). Five of these species were absent from all age classes of mine rehabilitation including all three foraging specialist trunk-gleaning insectivores (Table 6.12). The yellow-tinted honeyeater *Lichenostomus flavescens* is an insectivore that also forages on trunks and branches although it does not specialise on this as a substrate. The pied imperial-pigeon *Ducula bicolor* is a migratory frugivore that nests in tall *Eucalyptus* spp. on the Weipa bauxite plateau.

The remaining 14 species are not foraging specialists. For example, little friarbird *Philemon citreogularis*, banded honeyeater *Cissomela pectoralis* and noisy friarbird *Philemon corniculatus* are insectivore-nectarivores that obtain most of their food from flowers and invertebrates in the foliage, mainly in the upper canopy but also understorey shrubs. All of these species are arboreal with respect to their requirements for nesting materials, nesting sites, perching sites, roosting sites or foraging substrates (Higgins *et al.*, 2001; Higgins and Peter, 2002; Higgins *et al.*, 2006). Detection rates for these 14 species generally increased with rehabilitation age but in the oldest age class of mine rehabilitation remained very low in comparison to native forest reference sites (Table 6.12).

Table 6.12 Mean detection rates of species associated with big framework trees

Bird species	**Mine rehabilitation sites**					**Native Forest**
	1 - 2 y	**3 - 4 y**	**5 - 8 y**	**9 - 16 y**	**> 16 y**	
Little Friarbird	0.02	0.08	0.13	0.23	0.45	0.92
Banded Honeyeater	0.02	0.04	0.10	0.06	0.03	0.22
Leaden Flycatcher		0.02	0.17	0.22	0.33	0.65
Rainbow Lorikeet		0.06	0.08	0.2	0.4	0.77
Noisy Friarbird		0.06	0.15	0.06	0.1	0.55
Olive-backed Oriole			0.02		0.03	0.06
Laughing Kookaburra			0.04	0.03	0.08	0.32
Rufous Whistler			0.06	0.03		0.69
Black-backed Butcherbird				0.02	0.03	0.61
Blue-winged Kookaburra				0.02	0.03	0.08
Cicadabird				0.02	0.03	0.16
Forest Kingfisher				0.05	0.08	0.38
Lemon-bellied Flycatcher				0.14	0.10	0.76
Grey Shrike-thrush					0.05	0.41
Brown Treecreeper						0.18
Yellow-tinted Honeyeater						0.09
Varied Sittella						0.16
Grey-crowned Babbler						0.02
Pied Imperial-pigeon						0.02

6.5 Bird response to vegetation and landscape

6.5.1 Bird response to vegetation structure

The site detection rates of birds (62 bird species in main matrix) were significantly related to site vegetation structure variables (12 vegetation structure variables in secondary matrix) (Appendix 16). Total variance in the bird data was 1.836. Cumulative variance explained was 32.1%, of which 23.1% was in the first axis and 6.4% was in the second axis. Only the first two axes were interpreted. The eigenvalue for the first axis was significantly higher than expected by chance (P < 0.001). The first axis was strongly correlated with the vegetation structure variables (P < 0.001) (Appendix 16). The variables most strongly related to axis 1 were mean height of the tallest vegetation layer and stem density, basal area and canopy volume of big trees (Table 6.13). The vegetation structure variables most strongly related to axis 2 were stem density of small trees, canopy volume of small trees and basal area of small trees.

Table 6.13 Correlation coefficients[1] between vegetation structure variables and ordination axes

Variable	Axis 1	Axis 2
Mean height of the tallest vegetation layer	**0.98**	-0.00
Stem density of big trees	**0.83**	-0.24
Stem density of small trees	0.20	**0.80**
Stem density of tall shrubs	-0.27	0.44
Stem density of low shrubs	0.48	0.24
Basal area of big trees	**0.80**	-0.22
Basal area of small trees	0.38	**0.74**
Canopy volume of big trees	**0.75**	-0.21
Canopy volume of small trees	0.30	**0.76**
Canopy volume of tall shrubs	-0.63	**0.56**
Canopy volume of low shrubs	0.38	-0.08
Canopy volume of perennial grasses	0.71	-0.27

1. The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case the site detection rate of birds, per unit change in the corresponding environmental variable (Ter Braak, 1986).

The first axis of the ordination almost completely separated native forest reference sites and mine rehabilitation sites (Figure 6.8). All native forest sites with the exception of site A-22, a low shrubby post-mining landscape analogue LU 5K site, were positioned to the right on axis 1 and all mine rehabilitation sites were positioned to the left. Of the 62 bird species in the ordination, 34 were positively related to the first axis. Six species groupings could be identified in the ordination on the basis of their scores (+ / -) in relation to the ordination axes (Appendix 16).

Groups I, II and III were species whose detection rates were highest in native forest reference sites (Figure 6.9). The site detection rates of Group I species were positively related to vegetation height (> 0.35) and the stem density and basal area of big trees, but negatively related to the stem density, basal area and canopy volume of small trees (> 0.35). Group I birds were: brown treecreeper, yellow-tinted honeyeater, grey shrike-thrush *Colluricincla harmonica*, rufous whistler *Pachycephala rufiventris*, and banded honeyeater.

The site detection rates of Group II species were also positively related to vegetation height and to stem density and basal area of big trees but were not negatively related to the stem density of small trees. Group II birds were: varied sittella *Daphoenositta chrysoptera*, cicadabird *Coracina tenuirostris*, grey-crowned babbler *Pomatostomus termporalis*, black-backed butcherbird *Cracticus mentalis*, forest kingfisher *Todiramphus sanctus*, lemon-bellied flycatcher *Microeca flavigaster*, laughing kookaburra *Dacelo novaeguinae*, brush cuckoo *Cacomantis variolosus*, blue-winged kookaburra *Dacelo leachii*, noisy friarbird, rainbow lorikeet *Trichoglossus haematodus*, little friarbird, leaden flycatcher *Myiagra rubecula*, olive-backed oriole *Oriolus sagittatus*, and mistletoebird *Dicaeum hirundinaceum*.

The site detection rates of Group III species were positively related to vegetation height (> 0.35) and to the stem density, basal area and canopy volume of small trees (>0.35). Group III species were: yellow oriole *Oriolus flavocinctus*, pied imperial-pigeon, koel *Eudynamys orientalis*, and sulphur-crested cockatoo *Cacatua galerita*.

Groups IV and V were species whose highest detection rates were in mine rehabilitation sites where vegetation height was much lower. The site detection rates of Group IV species were positively related to the stem density, basal area and canopy volume of small trees and tall shrubs (> 0.35). These species occurred most frequently in the two oldest age classes of mine rehabilitation sites and only occasionally in native forest sites. Within native forest sites they occurred more frequently in sites with a higher density of small trees and tall shrubs. Group IV species were: white-streaked honeyeater, black butcherbird *Cracticus quoyi*, yellow-spotted honeyeater *Meliphaga notata*, varied triller *Lalage leucomela*, graceful honeyeater *Meliphaga gracilis*, dusky honeyeater *Myzomela obscura*, brahminy kite *Haliastur indus*, great bowerbird *Ptilonorhynchus nuchalis*, brown-backed honeyeater *Ramsayornis fasciatus*, and spangled drongo *Dicrurus bracteatus*. With the exception of brahminy kite, all of these species generally prefer dense or mesic vegetation (Higgins *et al.*, 2006; Higgins *et al.*, 2001; Higgins and Davies, 1996).

The site detection rates of Group V species were negatively related to big trees (> 0.35), small trees and tall shrubs (> 0.35). These species had their highest detection rates in young mine rehabilitation sites with low shrub layers and no tree layers. Group V species were: Australasian pipit *Anthus novaeseelandiae*, golden-headed cisticola *Cisticola exilis*, chestnut-breasted mannikin *Lonchura castaneothorax*, red-browed finch *Neochmia temporalis*, brown quail *Coturnix ypsilonophora*, and brown honeyeater. The remaining species were shared between native forest and mine rehabilitation sites and did not show strong responses to the ordination axes.

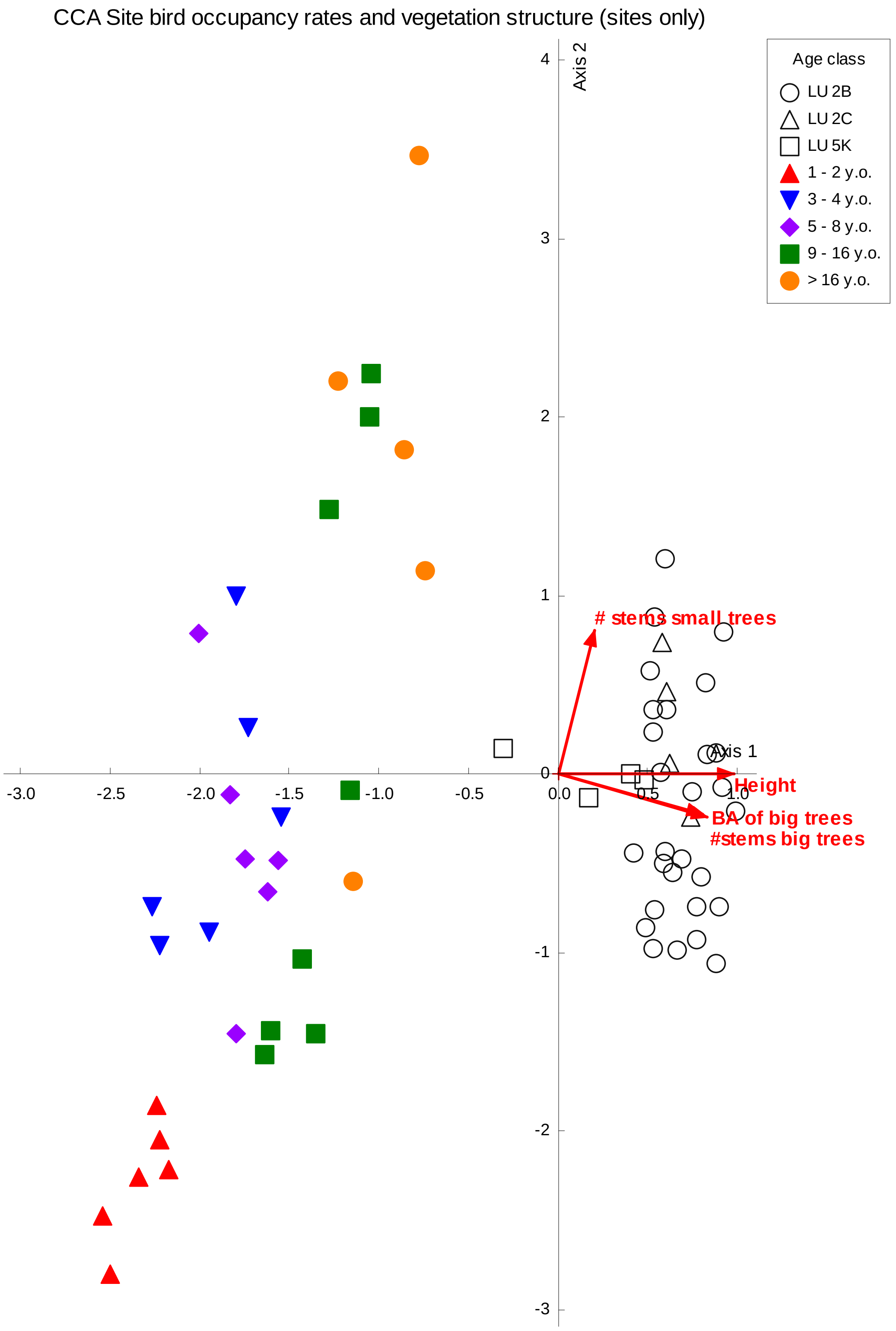

Figure 6.8 Ordination graph of birds and vegetation structure (sites)

Ordination of sites in environmental space as defined by CCA. Axes are centred and scaled with unit variance. Scores for graphing sites are linear combinations of vegetation structure variables. To reduce clutter only variables with bi-plot scores > 0.65 are shown. The first (horizontal) axis can be interpreted as a gradient of increasing (left to right) contribution from big trees. The second (vertical) axis can be interpreted as a gradient of increasing (bottom to top) contribution from small trees. Each site point falls approximately on the centroid of the species points that occur at that site, so that it can be inferred which species were most likely to have occurred at a site.

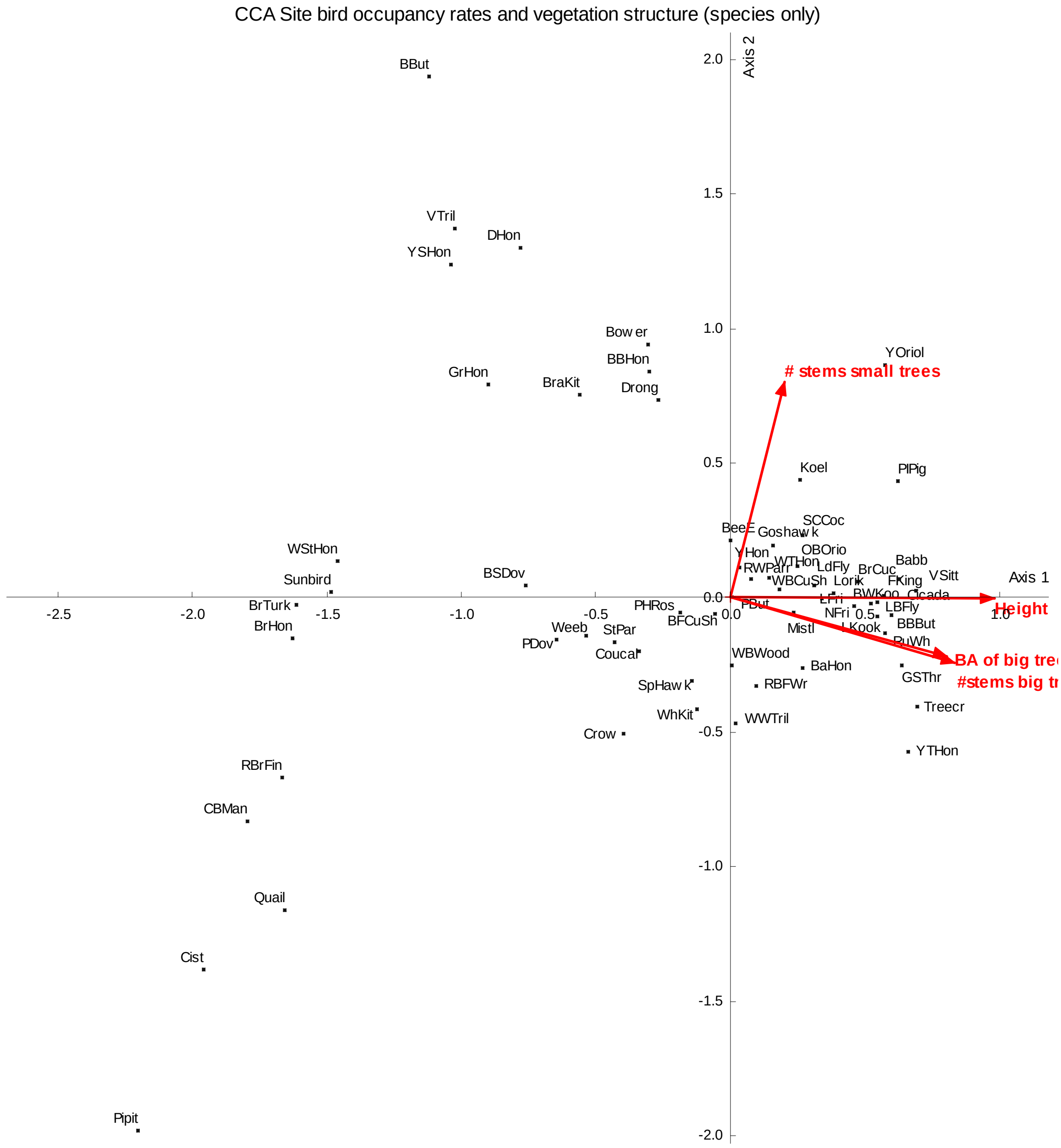

Figure 6.9 Ordination graph of birds and vegetation structure (species)

Ordination of species in environmental space as defined by CCA. Axes are centred and scaled with unit variance. Scores for graphing sites are linear combinations of vegetation structure variables. To reduce clutter only variables with bi-plot scores > 0.65 are shown. Species scores are weighted mean site scores. This enables direct spatial interpretation of the relationship between species and environmental variables. Imagining the environmental line extended throughout the ordination space and perpendiculars drawn from each species to the environmental line: (a) the approximate ranking of species response curves to that variable; and (b) whether a species has a higher than average or lower than average optimum on that variable can be seen.

6.5.2 Bird response to vegetation composition

The site detection rates of birds (62 bird species in main matrix) were significantly related to site vegetation composition variables (10 vegetation composition variables in secondary matrix). This analysis produced very similar results to the ordination based on vegetation structure. This is because vegetation composition and vegetation structure were strongly related in this dataset (Appendix 17). Mine rehabilitation sites were dominated by *Acacia* spp., *Alstonia* spp. and 'other plant genera' and have very different vegetation structure from native forest sites. Apart from being much older and taller, native forest sites are dominated by *Eucalyptus* spp. and their vegetation structure is dominated by big trees. In this case, native forest reference sites are synonymous with eucalypts, which in turn are synonymous with big trees.

Total variance in the bird data was 1.836. Cumulative variance explained was 26.7% of which 19.2% was in the first axis and 5.4% was in the second axis (Appendix 18). Only the first two axes were interpreted. The eigenvalue for the first axis was significantly higher than expected by chance ($P < 0.001$). The first axis was strongly related to the vegetation composition variables ($P < 0.001$). The variables most strongly positively related to axis 1 were canopy volumes of *Eucalyptus* spp. and *Corymbia* spp. (Table 6.14). Canopy volume of *Acacia* spp. was strongly negatively related to axis 1. The variable most strongly related to axis 2 was canopy volume of *Alstonia* spp. Large positive values for *Alstonia* spp. characterise mature mine rehabilitation sites.

Table 6.14 Correlation coefficients[1] between vegetation composition variables and ordination axes

Variable	Axis 1	Axis 2
Canopy volume of *Acacia* spp	**-0.68**	0.37
Canopy volume of *Alstonia* spp	-0.35	**0.68**
Canopy volume of *Corymbia* spp	**0.75**	0.01
Canopy volume of *Eucalyptus* spp	**0.80**	0.27
Canopy volume of *Erythrophleum chlorostachys*	**0.55**	0.12
Canopy volume of *Livistona muelleri*	0.10	-0.01
Canopy volume of *Melaleuca* spp	-0.26	0.56
Canopy volume of *Parinari nonda*	0.06	0.55
Canopy volume of *Xylomelum scottianum*	0.53	-0.04
Canopy volume of all other plant genera	**-0.68**	**0.62**

1. The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case the site detection rate of birds, per unit change in the corresponding environmental variable (Ter Braak, 1986).

The first axis of the ordination completely separated native forest sites from mine rehabilitation sites. All of the native forest sites were positioned to the right on axis 1, and all mine rehabilitation sites were positioned to the left (Figure 6.10). Thirty-three of the 62 bird species in the ordination were positively related to the first axis.

Four species groupings could be identified in the ordination (Figure 6.11). The site detection rates of Group I species were positively related (> 0.3) to high proportions of *Eucalyptus* spp. and *Erythrophleum chlorostachys* in the plant canopy and low proportions of *Acacia* spp. and 'other plant genera'. Group I species included: yellow-tinted honeyeater, varied sittella, rufous whistler, black-backed butcherbird, lemon-bellied flycatcher, brown treecreeper, grey shrike-thrush, forest kingfisher, cicadabird, laughing kookaburra, blue-winged kookaburra, noisy friarbird, grey-crowned babbler, banded honeyeater, rainbow lorikeet, leaden flycatcher, pied imperial-pigeon, and little friarbird.

The site detection rates of Group II species were positively related to the proportion of *Alstonia* spp, *Acacia* spp and 'other' spp in the plant canopy. This group occurred most frequently in the two oldest age classes of mine rehabilitation. Group II species were: black butcherbird, varied triller, yellow-spotted honeyeater, dusky honeyeater, graceful honeyeater, brahminy kite, great bowerbird, brown-backed honeyeater, and spangled drongo.

The site detection rates of Group III species were negatively related to the proportion of *Eucalyptus* spp., *Erythrophleum chlorostachys* and *Alstonia* spp. in the plant canopy. This group occurred most frequently in young mine rehabilitation sites. Group III species were: Australian brush-turkey *Alectura lathami*, white-streaked honeyeater, chestnut-breasted mannikin, brown honeyeater, olive-backed sunbird *Nectarinia jugularis*, red-browed finch *Neochmia temporalis*, golden-headed cisticola, brown quail, Australasian pipit, bar-shouldered dove *Geopelia humeralis,* peaceful dove *Geopelia striata*, weebill *Smicrornis brevirostris*, and pheasant coucal *Centropus phasianus*.

The remaining species, Group IV, did not show a strong response to the ordination axes and were shared between native forest reference sites and mine rehabilitation sites.

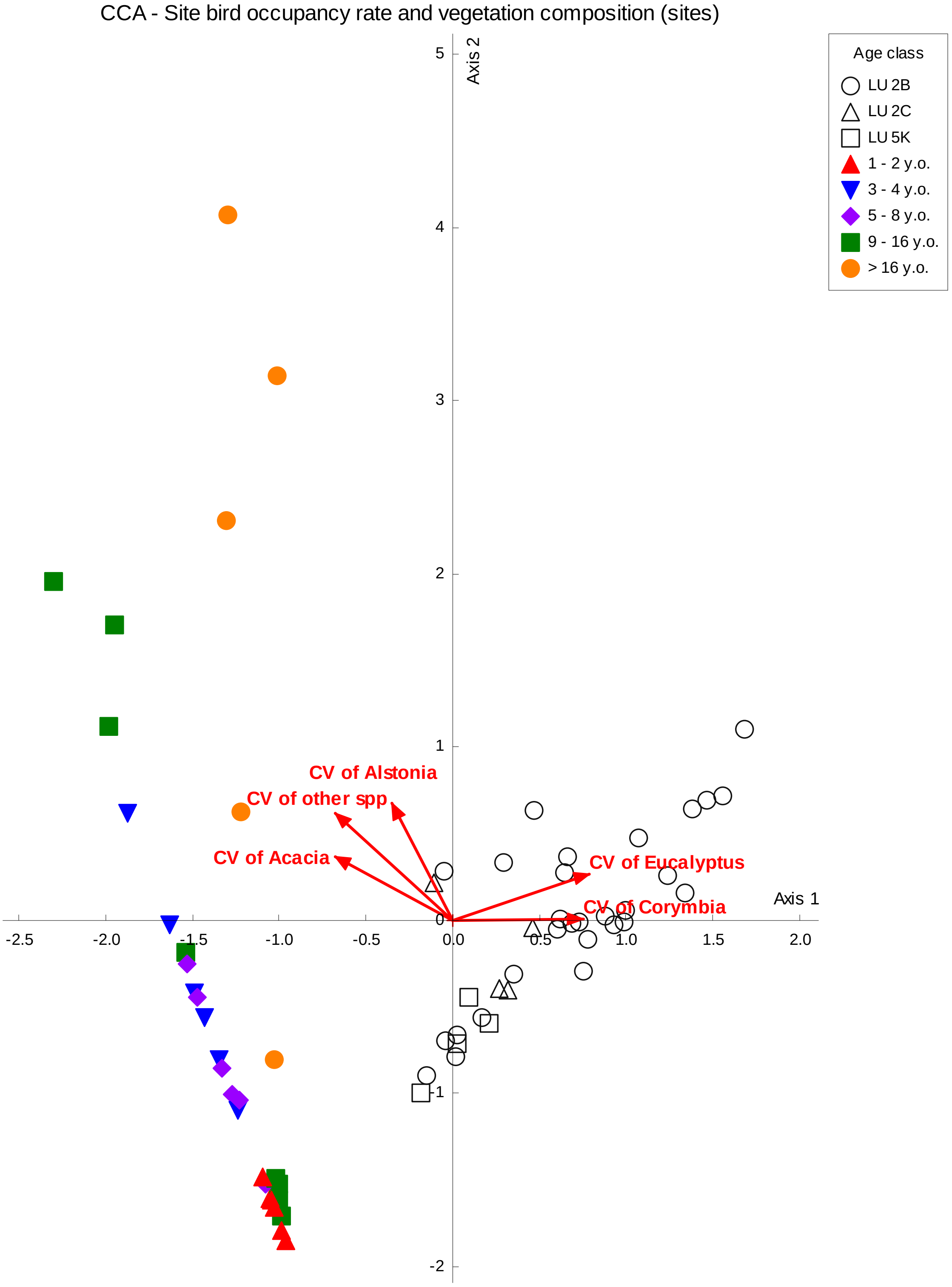

Figure 6.10 Ordination graph of birds and vegetation composition (sites)

Ordination of sites in environmental space as defined by CCA. Axes are centred and scaled with unit variance. Scores for graphing sites are linear combinations of vegetation composition variables. To reduce clutter only variables with bi-plot scores > 0.4 are shown. The first (horizontal) axis can be interpreted as a gradient of increasing (left to right) contribution from *Eucalyptus* species. The second (vertical) axis can be interpreted as a gradient of increasing (bottom to top) contribution from *Alstonia* and other species. Each site point is placed approximately on the centroid of the species points that occur at that site so that it may be inferred which species were most likely to have occurred at a site.

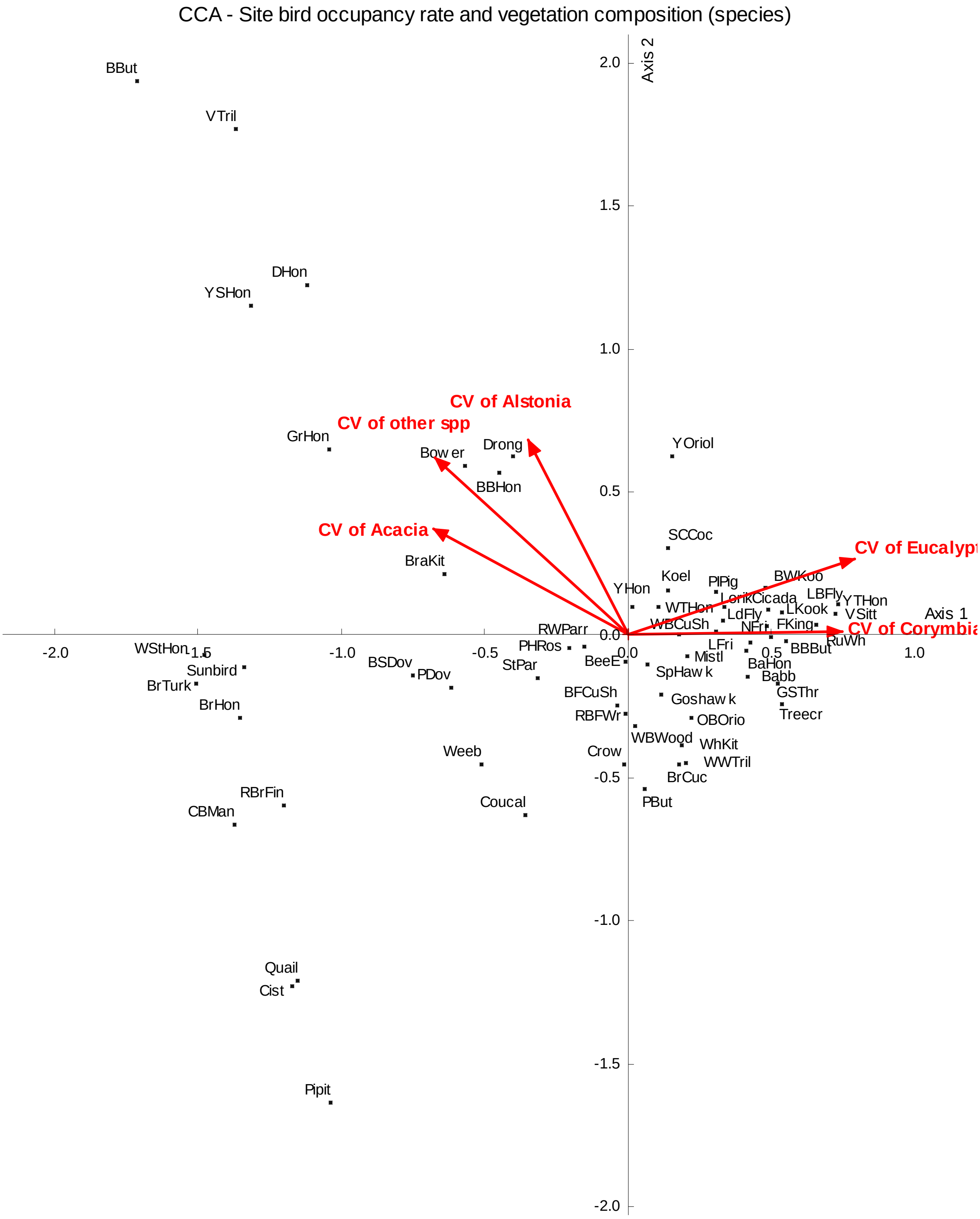

Figure 6.11 Ordination graph of birds and vegetation composition (species)

Ordination of species in environmental space as defined by CCA. Axes are centred and scaled with unit variance. Site scores are linear combinations of vegetation composition variables. To reduce clutter only variables with bi-plot scores > 0.4 are shown. Species scores are weighted mean site scores. This enables direct spatial interpretation of the relationship between species and environmental variables. Imagining the environmental line extended throughout the ordination space and perpendiculars drawn from each species to the environmental line: (a) the approximate ranking of species response curves to that variable; and (b) whether a species has a higher than average or lower than average optimum on that variable can be seen.

6.5.3 Foraging group response to vegetation structure

Canonical correspondence analysis confirmed the importance of vegetation structure to the abundance of individuals in foraging guilds. The abundance of birds within the 14 foraging groups (main matrix) was significantly related to vegetation structure variables (secondary matrix) (Appendix 19). The total variance in the foraging group data was 0.6786. Cumulative variance explained was 30.9% of which 27% was in the first axis and 3.9% in the second axis (Appendix 19). The first axis of the ordination almost completely separated native forest reference sites and mine rehabilitation sites (Figure 6.12). All native forest sites with the exception of two post-mining landscape analogue LU 5K sites were positioned to the right on axis 1. All mine rehabilitation sites except for one in the oldest age class were positioned to the left. The vegetation structure variables most strongly related to axis 1 were mean height of the tallest vegetation layer and stem density of big trees (Table 6.15).

Table 6.15 Correlation coefficients[1] between vegetation structure variables and ordination axes

Variable	Axis 1	Axis 2
Mean height of the tallest vegetation layer	**0.92**	0.28
Stem density of big trees	**0.70**	0.12
Stem density of small trees	0.43	-0.53
Stem density of tall shrubs	-0.12	-0.21
Stem density of low shrubs	0.45	0.08
Basal area of big trees	0.68	0.12
Basal area of small trees	0.58	-0.51
Canopy volume of big trees	0.64	0.14
Canopy volume of small trees	0.53	**-0.60**
Canopy volume of tall shrubs	-0.39	**-0.61**
Canopy volume of low shrubs	0.28	0.38
Canopy volume of perennial grasses	0.56	0.18

1. The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case abundance of foraging groups, per unit change in the corresponding environmental variable (Ter Braak, 1986).

Trunk-gleaning insectivores, pouncing predators, hawking insectivores, aerial insectivores, frugivores, foliage-gleaning insectivores and nectarivores were positioned to the right on axis 1 (Figure 6.13). Ground foragers, granivores, grass-foraging insectivores, omnivores and carnivores were positioned to the left on axis one. Insectivore-nectarivores and tree-seed eaters were virtually centred on axis 1.

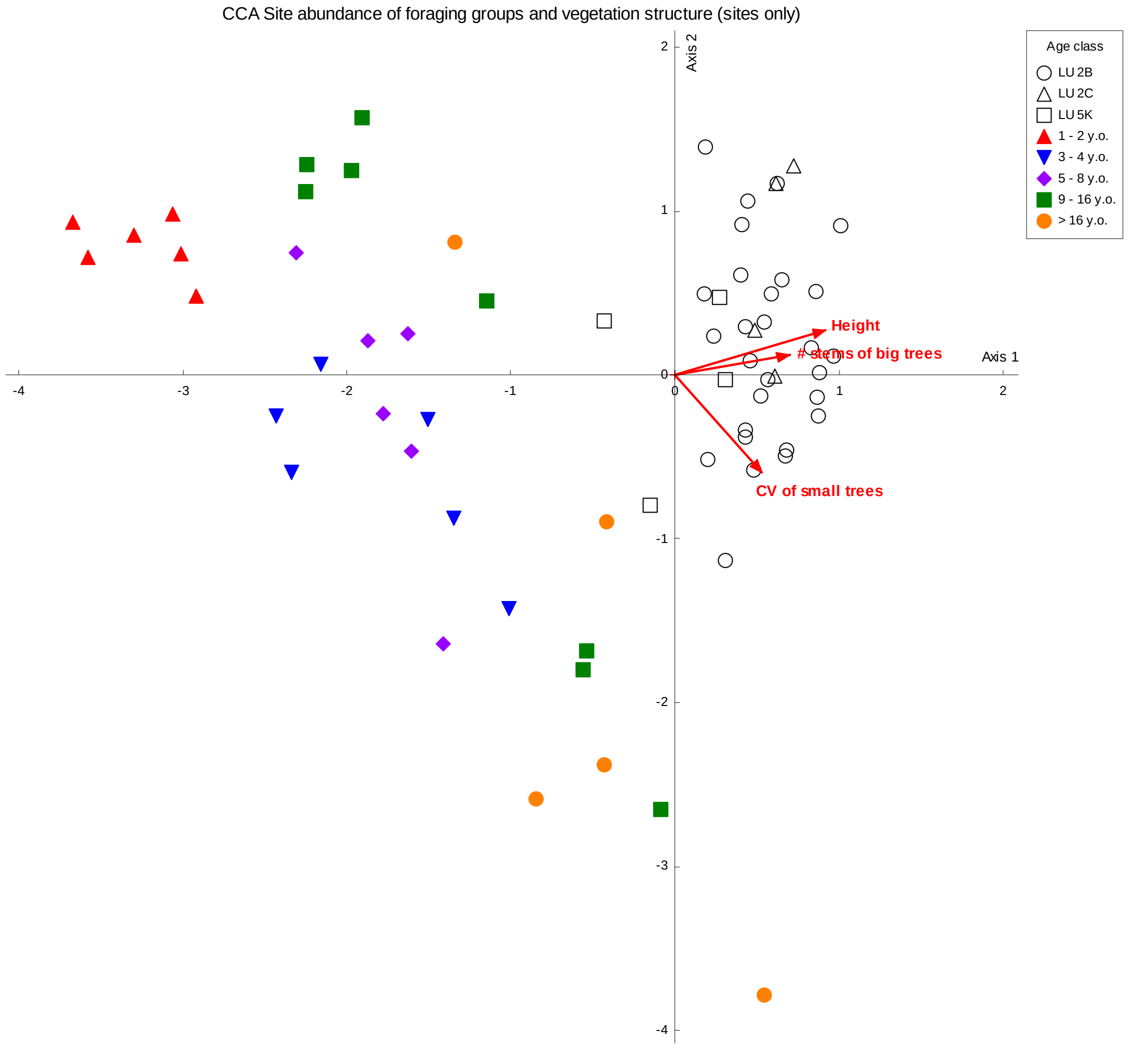

Figure 6.12 Ordination graph of foraging groups and vegetation structure (sites)

Ordination of sites in environmental space as defined by CCA. Axes are centred and scaled with unit variance. Site scores are linear combinations of vegetation structure variables. To reduce clutter only variables with bi-plot cut off scores > 0.54 are shown. The first (horizontal) axis can be interpreted as a gradient of increasing (left to right) contribution from big trees. The second (vertical) axis can be interpreted as a gradient of decreasing (bottom to top) contribution from small trees. Each site point is placed approximately on the centroid of the foraging group points that occur at that site so that it may be inferred which foraging groups were most likely to have occurred at a site.

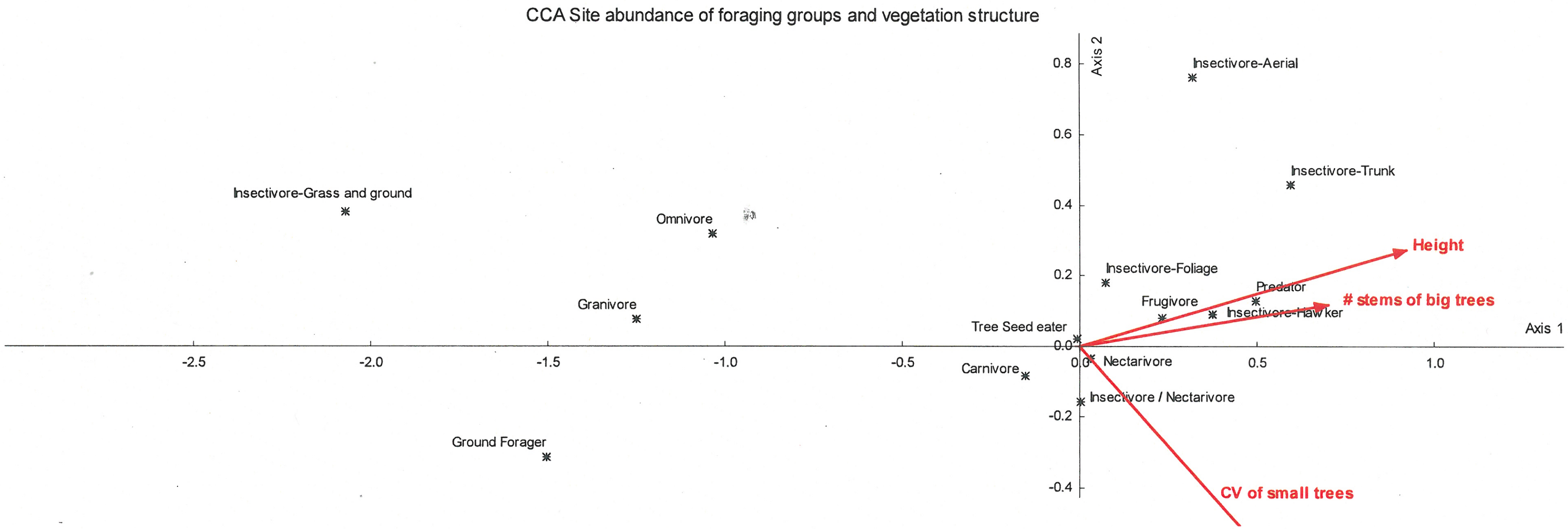

Figure 6.13 Ordination graph of foraging groups and vegetation structure (foraging groups)

Ordination of foraging groups in environmental space as defined by CCA. Axis scores are centred and scaled to unit variance. Scores for graphing sites are linear combinations of vegetation structure variables. To reduce clutter only variables with bi-plot scores > 0.54 are shown. Foraging groups are weighted mean site scores. This enables direct spatial interpretation of the relationship between foraging groups and environmental variables. Imagining the environmental line extended throughout the ordination space and perpendiculars drawn from each foraging group to the environmental line: (a) the approximate ranking of foraging group responses to that variable; and (b) whether a foraging group has a higher than average or lower than average optimum on that variable can be seen. Star symbols represent optima in foraging group responses to the environmental gradients.

6.5.4 Bird response to landscape variables

The site detection rates of birds (62 bird species in the main matrix) were significantly related to site landscape context variables (seven landscape variables in the secondary matrix) (Appendix 20). Total variance in the bird data was 1.836. Cumulative variance explained was 29.2%, of which 22.7% was in the first axis and 3.7% was in the second axis. Only the first two axes were interpreted. The eigenvalue for the first axis was significantly higher than expected by chance ($P < 0.001$). Separate analysis using data from mine rehabilitation sites only was non-significant. The first axis was strongly related to the landscape variables ($P < 0.001$). The landscape context variables most strongly related to axis 1 were amount of remnant vegetation within a 500 m buffer of the site and distance from remnant vegetation (Table 6.16).

Table 6.16 Correlation coefficients[1] between landscape variables and ordination axes

Variable	Axis 1	Axis 2
Area of remnant vegetation in a 500 m buffer	**0.99**	-0.02
Distance from remnant vegetation	**-0.79**	-0.55
Distance from the coast	0.30	-0.21
Distance from the edge of the bauxite plateau	-0.22	-0.15
Distance from mesic vegetation	-0.003	**-0.71**

1. The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case abundance of foraging groups, per unit change in the corresponding environmental variable (Ter Braak, 1986).

The first axis of the ordination can be interpreted as a gradient of remnant vegetation. It completely differentiated native forest reference sites and mine rehabilitation sites (Figure 6.14). All native forest sites were positioned to the right on axis 1 and all mine rehabilitation sites were positioned to the left. The site detection rates of half of the bird species in the ordination were positively related to the first axis and conversely half of the species were negatively related to the first axis (Figure 6.15).

Four species groupings can be identified. Group I species were strongly positively related (> 0.35) to the area of remnant vegetation and had their highest detection rates in native forest reference sites. Group I species were: yellow-tinted honeyeater, varied sittella, rufous whistler, black-backed butcherbird, grey shrike-thrush, brown treecreeper, cicadabird, lemon-bellied flycatcher, laughing kookaburra, forest kingfisher, grey-crowned babbler, blue-winged kookaburra, noisy friarbird, banded honeyeater, rainbow lorikeet, pied imperial-pigeon, olive-backed oriole, pied butcherbird *Cracticus nigrogularis*, little friarbird, and brush cuckoo.

Group II species were strongly negatively related to area of remnant vegetation (> 0.35) and negatively related to distance from mesic vegetation. They were not sourced from native forest but their site detection rates declined as distance from mesic native vegetation increased (Figure 6.15). Although species in this group showed a preference for mine rehabilitation sites they presumably obtained some benefit from or were possibly sourced from mesic native vegetation in the landscape. Group II birds were: black butcherbird, yellow-spotted honeyeater, red-browed finch, varied triller, dusky honeyeater, graceful honeyeater, and great bowerbird (Figure 6.15). With the exception of red-browed finch, all of these species are usually found in mesic vegetation (Higgins *et al.*, 2006; Higgins *et al.*, 2001; Higgins and Davies, 1996).

The response of Group II birds probably reflects the composition of the remnant native vegetation within mine rehabilitation site buffers which was not necessarily entirely *Eucalyptus tetrodonta* open forest. Within the mine, the main areas of remnant vegetation retained are buffers surrounding drainage lines. The remnant vegetation within 500 m buffers of mine rehabilitation sites, therefore, sometimes included mesic vegetation. The site buffers of seven mine rehabilitation sites included other land units: LU 3C (notophyll vine forest), LU 4B (mesophyll palm forest and swamp forest), LU 5J (Bloodwood / *Banksia* spp. woodland), and LU 8B (Closed scrub within spring fed swamp) (Appendix 1).

Group III birds had their highest site detection rates in the younger age classes of mine rehabilitation. Their site detection rates increased as the distance from native vegetation and as distance from mesic vegetation increased. These species were not sourced from native forest or any of the mesic vegetation types in the surrounding landscape. Group III birds were: brown quail, chestnut-breasted mannikin, white-streaked honeyeater, golden-headed cisticola, brahminy kite, Australasian pipit, brown honeyeater, Australian brush-turkey, and olive-backed sunbird (Figure 6.15). Four of these are usually found in grassland and the remainder except for brahminy kite are found in a range of wooded habitats (Higgins *et al.*, 2006; Marchant and Higgins, 1993; Higgins *et al.*, 2001)

Group IV birds were located in the centre of the ordination and were shared between native forest sites and mine rehabilitation sites. The detection rates of these species declined as distance from remnant vegetation increased. Group IV species were: yellow honeyeater, rainbow bee-eater *Merops ornatus*, red-winged parrot *Aprosmictus erythropterus*, and pale-headed rosella *Platycercus adscitus*.

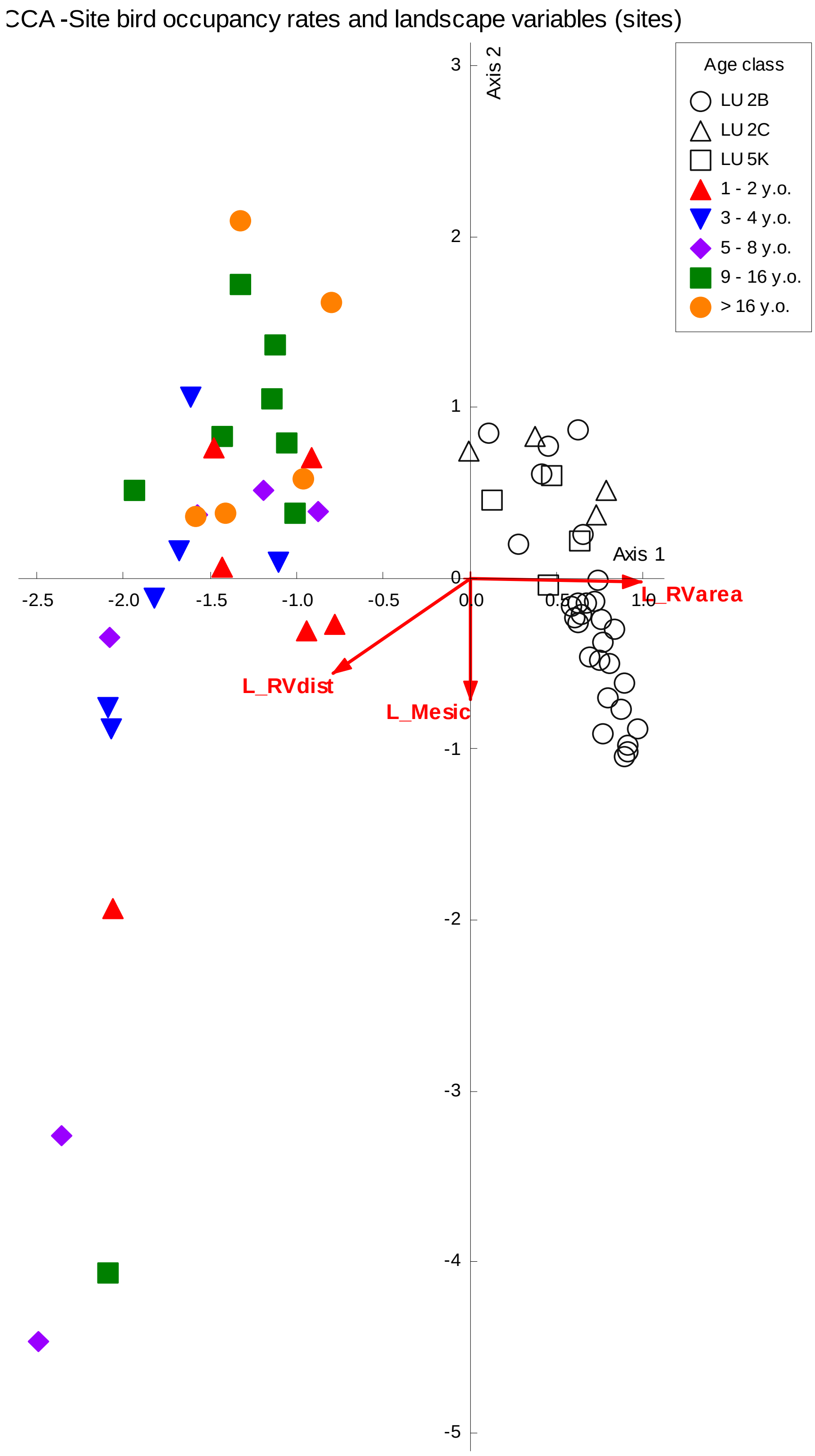

Figure 6.14 Ordination graph of birds and landscape context (sites)

Ordination of sites in environmental space as defined by CCA. Axis scores are centred and scaled to unit variance. Scores for graphing sites are linear combinations of landscape variables. To reduce clutter only variables with bi-plot scores > 0.4 are shown. The first (horizontal) axis can be interpreted as a gradient of increasing (lef to right) area of remnant vegetation. The second (vertical) axis can be interpreted as a gradient of increasing (bottom to top) distance from mesic vegetation. Each site point is placed approximately on the centroid of the species points that occur at that site so that it may be inferred which species were most likely to have occurred at a site.

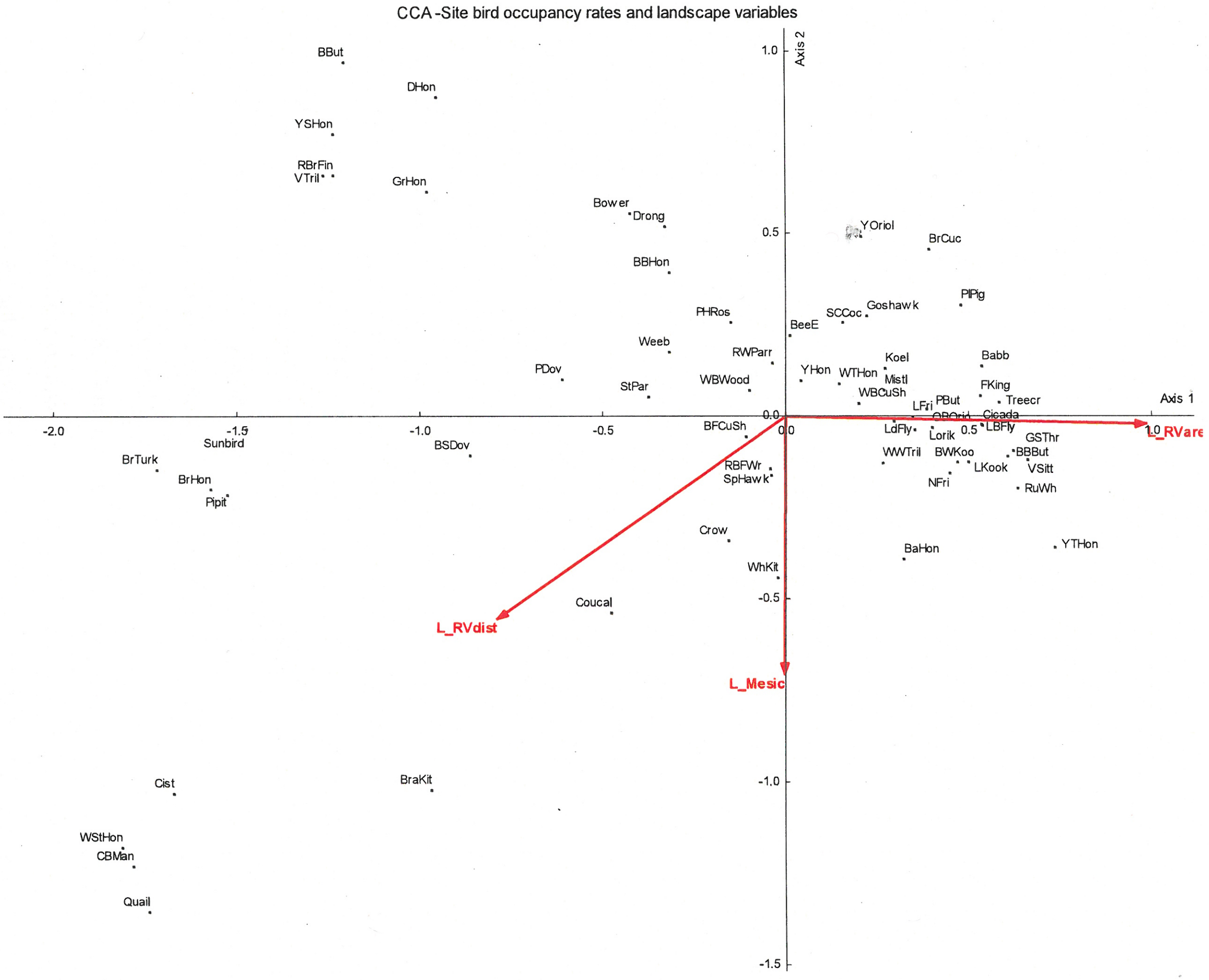

Figure 6.15 Ordination graph of birds and landscape context (species)
Ordination of species in environmental space as defined by CCA. Axis scores were centred and scaled to unit variance. Scores for graphing sites are linear combinations of landscape variables. To reduce clutter only variables with bi-plot scores > 0.4 are shown. Species scores are weighted mean site scores. This enables direct spatial interpretation of the relationship between species and environmental variables. Imagining the environmental line extended throughout the ordination space and perpendiculars drawn from each species to the environmental line: (a) the approximate ranking of species response curves to that variable; and (b) whether a species has a higher than average or lower than average optimum on that variable can be seen.

6.5.5 Effect of proximity to remnant native vegetation

There was no relationship between bird species richness in mine rehabilitation sites and distance from remnant vegetation. Nor was there a relationship between bird species richness in mine rehabilitation sites and area of remnant vegetation within a 500 m site buffer (Table 6.17). There was still no relationship when the data were analysed by age class. Site species shortfall index values which provide a measure of the species richness of native forest birds were also not related to distance of mine rehabilitation sites from remnant native vegetation or amount of remnant vegetation within a 500 m site buffer.

Table 6.17 Linear regressions between bird summary values and landscape variables[1]

Response variables	**Distance (m) of site from remnant vegetation**			
	y-intercept (± s.e.)	slope (± s.e)	adjusted R^2	T (29 d.f.)
Site species richness √[2]	4.4 (± 0.26)	-0.0003 (± 0.0005)	0	-0.66 ns
Species shortfall index % √	8.8 (± 0.22)	0.0007 (± 0.0004)	0.062	1.72 ns
	Area (ha) remnant vegetation within a 500 m buffer of the site			
	y-intercept (± s.e.)	slope (± s.e)	adjusted R^2	T (29 d.f.)
Site species richness √	4.15 (± 0.27)	0.006 (± 0.010)	0	0.6 ns
Site species shortfall index % √	9.3 (± 0.23)	-0.010 (± 0.008)	0.014	-1.19 ns

1. Only data from mine rehabilitation sites is included in the analysis.
2. Bird data were square root transformed. Significance of the T-statistic is indicated by asterisks * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$; and *** = $P \leq 0.001$. ns = not significant.

6.5.6 Native forest bird response to variation in vegetation structure

Separate ordinations were used to explore the relationship between site detection rates of birds and vegetation structure in the 31 mine rehabilitation sites only and in the 36 native forest reference sites only. The results of the CCA for mine rehabilitation sites were significant and explained a large part of the variation in site detection rates of birds however they are not reported as they simply confirmed the importance of mean vegetation height and density of small trees in mine rehabilitation (Appendix 21).

Canonical Correspondence Analysis was also used to explore the relationship between vegetation structure in the 36 native forest reference sites and site detection rates of birds (only species observed in five or more native forest reference sites). The site detection rates of native forest birds (46 species in main matrix) were significantly related to vegetation structure (12 site vegetation structure variables in secondary matrix) in native forest reference sites (Appendix 22). The total variance in the bird data was 0.6416. Cumulative variance explained was 23.8% of which 10% was in the first axis and 8% was in the second axis. Only the first two axes were interpreted. The eigenvalue for the first axis was significantly higher than expected by chance ($P = 0.004$). The first axis of the ordination was significantly related to the vegetation structure variables ($P = 0.034$). With only native forest sites in the ordination, vegetation height did not strongly influence the structure of the ordination. The first axis was positively related to the canopy volume of big trees and canopy volume of grasses and negatively related to the number of stems of small trees, tall shrubs and low shrubs. The first (horizontal) axis of the ordination can therefore be interpreted as a gradient of increasing (left to right) site openness and grassiness. The second (vertical) axis can be interpreted as a gradient of increasing (bottom to top) contribution from small trees.

Examination of ordination graphs of individual species shows a number of patterns. The site detection rates of all but one species, varied across the native forest sites. The exception was the white-throated honeyeater *Melithreptus albogularis* which occurred at a high detection rate across all sites. There was insufficient data to detect any clear patterns of response in 23 out of the 46 species that occurred in five or more native forest sites. There was variation in the site detection rates of the remaining 23 species, (i.e., patchiness, at the scale of 2 ha). However the pattern of response for 9 of these species did not appear to be related to the ordination axes. Presumably factors other than variation in the measured attributes of vegetation structure caused variation in the site detection rates. Site detection rates of the remaining 14 species varied in response to the relative contributions of big trees, small trees, canopy volume of grasses and number of stems of low shrubs and tall shrubs.

Site detection rates for the banded honeyeater, noisy friarbird, rufous whistler and yellow-tinted honeyeater were positively related (> 0.35) to both ordination axes, indicating a preference for sites with high canopy volume of big trees, and small trees, high canopy volume of perennial grasses and low contribution of low shrubs and tall shrubs. All four of these species feed high in the canopy of trees (Higgins *et al.*, 2001; Higgins and Peter, 2002). Both the banded honeyeater and noisy friarbird are non-sedentary insectivore-nectarivores and the rufous whistler is a sedentary species. The noisy friarbird and rufous whistler also roost near the top of the canopy. An example of the detection rate of one of these species, the banded honeyeater, is shown in Figure 6.16.

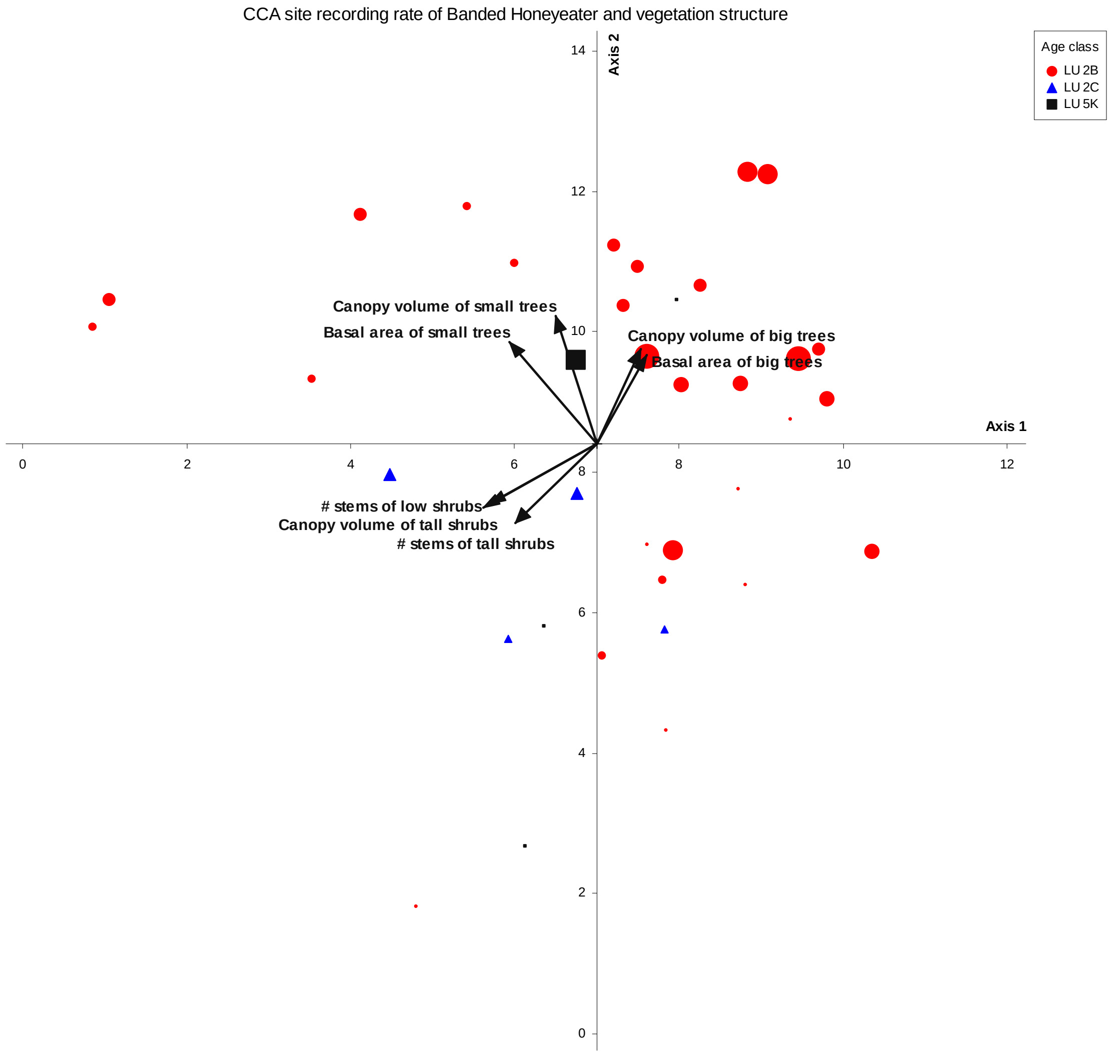

Figure 6.16 Detection rate of banded honeyeater in native forest sites

The site detection rate of each species was one of 8 values between 0 and 1. The larger the value of the site detection rate, the larger the size of the site symbol in the ordination. Values of zero are indicated by the very small symbols.

Site detection rates for brown treecreeper, grey shrike-thrush, red-backed fairy-wren, pale-headed rosella, and striated pardalote were positively related (> 0.35) to axis one and negatively related to axis 2 indicating a preference for sites with a high canopy volume of big trees and high canopy volume of perennial grasses but with lower contribution from the small tree layer. Four of these species are sedentary insectivores. The brown treecreeper is a trunk-gleaning insectivore that forages among crevices on trunks and large limbs especially of rough-barked eucalypts (Higgins *et al.*, 2001). Studies in temperate woodlands have recorded them foraging on the ground, however, in this study they were only observed on tree trunks at heights > 2 m. The striated pardalote forages high in the outer foliage of trees and the grey shrike-thrush forages in foliage and branches of trees at all levels (Higgins and Peter, 2002). Similarly, the red-backed fairy-wren forages in foliage of grasses and shrubs at all levels up to 30 m (Higgins *et al.*, 2001). An example of the detection rate of one of these species, the grey shrike-thrush, is shown in Figure 6.17.

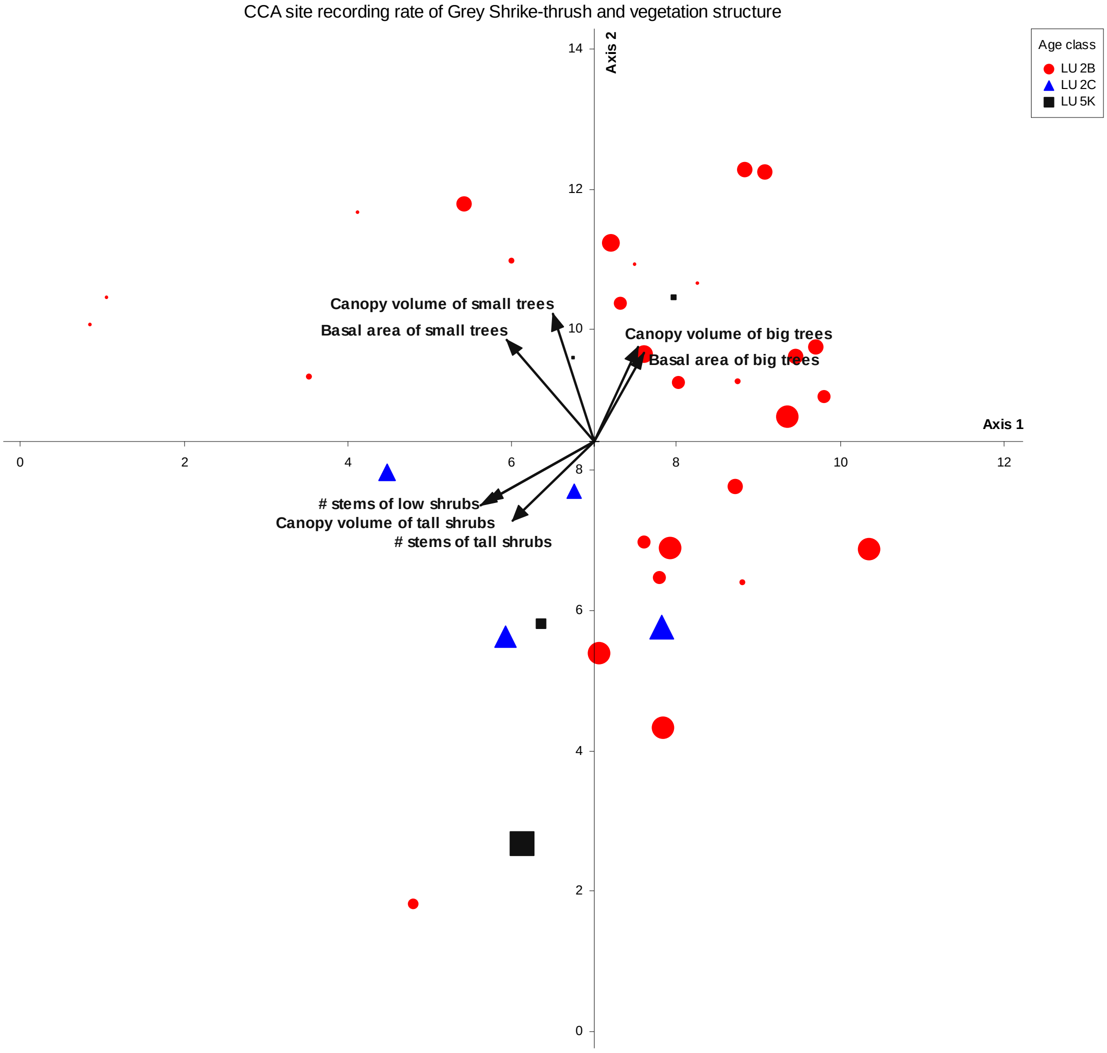

Figure 6.17 Detection rate of grey shrike-thrush in native forest sites

The site detection rate of each species was one of 8 values between 0 and 1. The larger the value the larger the size of the site symbol in the ordination. Values of zero are indicated by the very small symbols.

Site detection rates of the cicadabird and drongo were negatively related to axis 1, indicating a preference for sites with less contribution from big trees. The site detection rates of the bar-shouldered dove, red-winged parrot and brush cuckoo were negatively related to both ordination axes indicating a preference for sites with less contribution from big and small trees and a higher contribution from the shrub layers. An example of the detection rate of one of these species, the bar-shouldered dove, is shown in Figure 6.18.

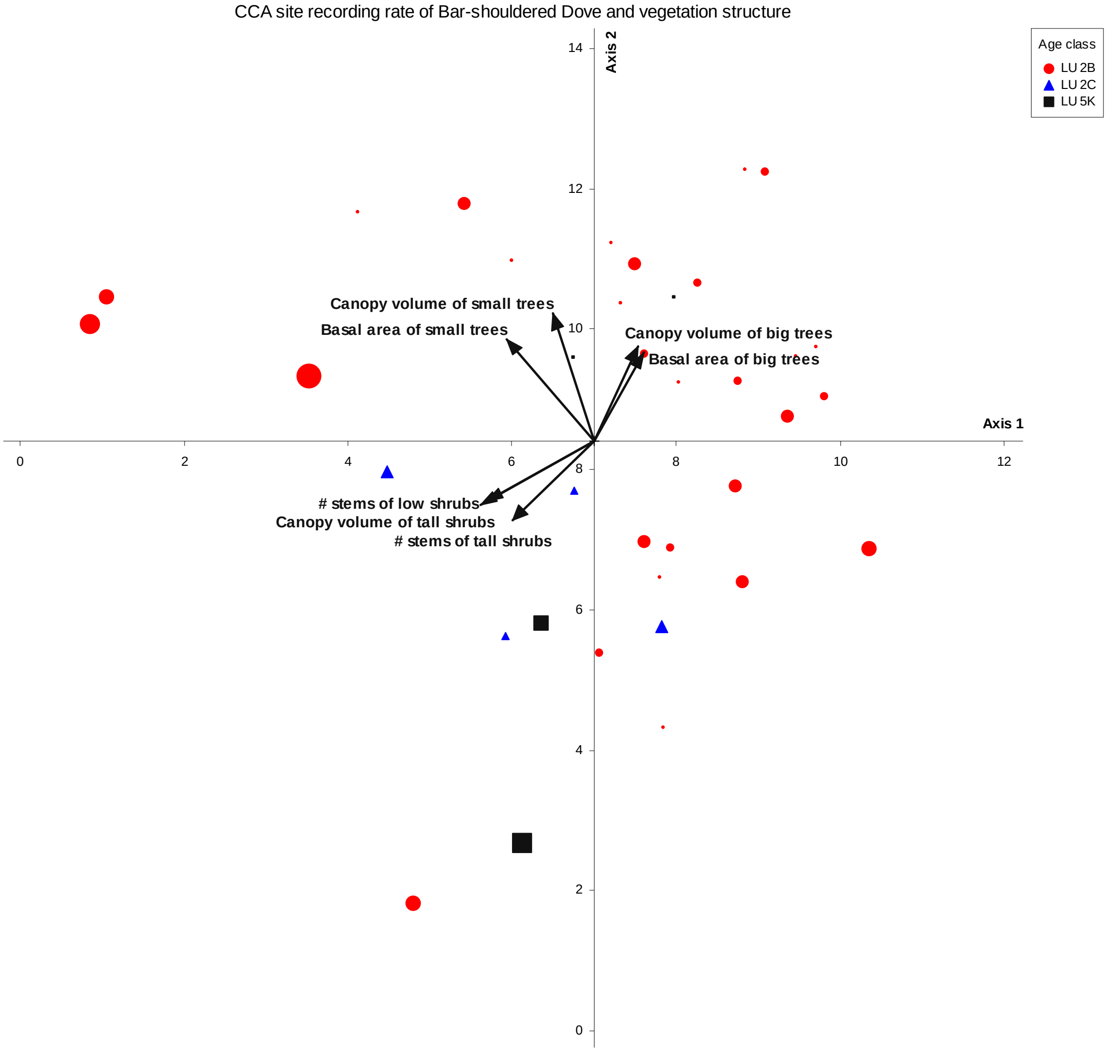

Figure 6.18 Detection rate of bar-shouldered dove in native forest sites

The site detection rate of each species was one of 8 values between 0 and 1. The larger the value the larger the size of the site symbol in the ordination. Values of zero are indicated by the very small symbols.

6.5.7 Highly correlated variables

When all sites were included in the analyses, simple linear regression of bird summary values against vegetation height and percent foliage cover of framework species yielded similar results to the regression against area of remnant vegetation, although height had the highest coefficient of determination (Table 6.18). This is probably due partly to the area of remnant vegetation within a 500 m buffer around a site being highly correlated with site vegetation structure in native forest reference sites. More of a gradient in values of area of remnant vegetation within a 500 m buffer of native forest reference sites would be needed to separate the effects of within site vegetation variables from site landscape context variables.

Table 6.18 Comparison of regressions of highly correlated variables

Response variables	**Area (ha) of remnant native forest in a 500 m buffer of the site √**			
	y-intercept (± s.e.)	slope (± s.e)	adjusted R^2	T (65 d.f.)
Site species richness √	3.9 (± 0.19)	0.12 (± 0.02)	0.27	5.1 ***
Site species shortfall index % √	10.5 (± 0.28)	-0.48 (± 0.03)	0.75	-14.1 ***
	Mean height of tallest vegetation layer (m) √			
	y-intercept (± s.e.)	slope (± s.e)	adjusted R^2	T (65 d.f.)
Site bird species richness √	2.8 (± 0.2)	0.52 (± 0.05)	0.58	9.6 ***
Site species shortfall index % √	12.9 (± 0.27)	-1.58 (± 0.06	0.89	-22.2***
	% of foliage cover made up of framework species √			
	y-intercept (± s.e.)	slope (± s.e)	adjusted R^2	T (65 d.f.)
Site species richness √	4.12 (±0.13)	0.17 (±0.02)	0.36	6.2***
Site species shortfall index % √	9.37 (± 0.14)	-0.66 (±0.03)	0.88	-21.8 ***

Outputs from simple linear regression of bird summary statistics and mean height of the tallest vegetation layer, area of remnant vegetation and percent foliage cover composed of framework species. All data were square root transformed to meet assumptions of normality. Significance of the T-statistic is indicated by asterisks * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$; and *** = $P \leq 0.001$. ns = not significant.

The similarity in results is probably also partly due to the relationships between vegetation structure and vegetation composition. Native forest sites had the tallest and most well developed vegetation structure and were also dominated by framework species, mostly *Eucalyptus* spp., whereas mine rehabilitation sites had low vegetation and were dominated by *Acacia* spp. and other non-framework species. This was not an avoidable confounding of variables caused by the research design but simply an outcome of the large differences in vegetation composition between native forest reference sites and mine rehabilitation sites and associated morphological differences. The high degree of relatedness of potentially explanatory variables makes it difficult to isolate the independent effects of any one of these variables.

6.6 Summary of bird results

Site category, season, time since fire, and weather conditions all had significant effects on bird abundance, but site category (including age class) had the largest effect. Mean bird abundance and bird species richness increased with age of rehabilitation site. There were no significant differences in mean species richness values between the two oldest age classes of mine rehabilitation and the three native forest reference land units. Mean bird abundance in the oldest age class of mine rehabilitation, however, was significantly lower than in all three native forest reference land units. The composition of the bird assemblage changed with increasing rehabilitation age. The bird species composition in the oldest age class of mine rehabilitation sites, however, remained significantly different from all three native forest reference land units.

Of the species recorded during surveys, 56% (55/97) were recorded in both native forest and mine rehabilitation sites. Of these shared bird species 40% (22/55) were significantly more abundant in native forest sites. Ten percent (6/55) of shared species were significantly more abundant in mine rehabilitation sites and there were no significant differences between the remaining 50% of shared species. Striated pardalote was the only species that was reasonably abundant in both native forest reference sites and mine rehabilitation sites. Of the native forest birds recorded during surveys 30% (24/79) were recorded exclusively in native forest.

The main temporal patterns were: (1) turnover of species generally associated with open, grassy or heathy habitats; (2) accumulation of species generally associated with mesic habitats; and (3) accumulation of native forest bird species. The composition of the bird assemblage became increasingly similar to the pre-mining native forest reference bird assemblage with increasing age. However, the mean species shortfall index in the oldest age class of mine rehabilitation was 63% compared to 27% for the pre-mining native forest reference land unit 2B indicating that there were still large differences in bird species composition.

Turnover of bird species was reflected in the temporal pattern of foraging group richness and abundance. The number of foraging groups and the species richness within each foraging group generally increased with age class of mine rehabilitation. The species richness of four foraging groups peaked in different age classes of mine rehabilitation. The number of foraging groups and the species richness of seven foraging groups was highest in native forest reference sites. The abundance of two foraging groups peaked in different age classes of mine rehabilitation. The pattern in all other foraging groups was generally one of increasing abundance with age class.

This analysis confirmed that site vegetation structure was an important determinant of bird assemblages. Bird species responded individually to vegetation structure. Although vegetation height was overwhelmingly the most important measure of vegetation structure, it cannot necessarily be assumed that it is height that birds are responding to as vegetation height incorporates a great deal of information about vegetation structure. Vegetation composition and landscape context were also important determinants of bird assemblages although these two variables were strongly related to vegetation structure. Birds had idiosyncratic species responses to area of remnant vegetation within a 500 m site buffer. The site detection rates of some species were also related to distance from mesic vegetation. There were consistent bird response groupings in relation to vegetation structure, vegetation composition and landscape context. These groupings are consistent with the significant differences in bird species composition between native forest and mine rehabilitation sites.

Nineteen bird species that were recorded in five or more sites were consistently associated with mature framework trees. Of these, five species including one entire specialist foraging group, trunk-gleaning insectivores, were absent from all age classes of mine rehabilitation. The remaining 14 species that were consistently associated with mature framework trees occurred to various degrees in mine rehabilitation sites, although their detection rates even in the oldest age class of mine rehabilitation sites were very low relative to native forest.

The relationship between birds and vegetation structure was most evident at the level of foraging groups. Foraging groups that use different components of mature vegetation were significantly more abundant in native forest reference sites, whereas foraging groups that use open ground, grass or low shrubs were significantly more abundant in mine rehabilitation sites. There were no significant differences between foraging groups that hunt or search for food from the air.

Variation in the bird assemblages between native forest reference sites was significantly related to variation in vegetation structure. The pattern of variation in site detection rates of 14 native forest bird species was related to ordination axes. Bird species responded differently to between site variation in the relative contributions of big trees, small trees, tall shrubs and perennial grasses. However, for many of the native forest birds, site detection rates appeared to be influenced by factors other than variation in the measured attributes of vegetation structure at the scale of 2 ha.

Chapter 7: Vegetation and landscape functionality results

7.1 Introduction

This study posed one key question in relation to the developing ecosystem in post-mining rehabilitation. Is post-mining rehabilitation likely to restore habitat similar to that of pre-mining native forest on the Weipa bauxite plateau? Some of the complexities of vegetation succession were identified in chapter 4. Making predictions about long term trajectories of vegetation succession is uncertain, particularly in novel environments. Nevertheless, enough is understood of the processes involved in vegetation succession that broad predictions are possible. Demonstration that restoration of the pre-mining native forest, or an ecologically appropriate analogue for the post-mining landscape was possible would require that even young mine rehabilitation sites satisfy a number of conditions. This chapter presents the results of the data analysis in relation to the fourth key research question. To begin with the vegetation composition and structure and landscape function in the pre-mining native forest reference LU 2B and the two post-mining landscape analogue land units, 2C and 5K are described to establish the reference conditions against which the post-mining rehabilitation is compared. The chapter concludes with a summary of the findings.

7.2 Reference vegetation and landscape functionality values

7.2.1 Pre-mining native forest - Land unit 2B

Twenty-eight research sites covered much of the variation within this land unit. A number of the sites included in the analysis for LU 2B were gradational to LU 2C and could not easily be categorised as one or the other. Gradational sites, for example, had some *Lophostemon suaveolens* and *Livistona muelleri*, which are species that become increasingly common in LU 2C.

Vegetation in LU 2B was typically grassy, layered tall woodland to tall open forest (Figure 7.1). The big tree layer was more developed in the pre-mining native forest reference LU 2B than in the two post-mining landscape analogue land units (Table 7.1). Consequently, values for mean vegetation height, mean total basal area, mean total % foliage cover and mean total canopy volume were all highest in LU 2B (Table 7.2). The canopy was dominated by *Eucalyptus tetrodonta* and locally *Corymbia nesophila* was co-dominant (Appendix 23). In some locations *Erythrophleum chlorostachys* formed a distinctive sub-canopy to canopy layer. There was considerable variation in floristics and structure of the shrub layers, which is probably associated with drainage conditions and disturbance history. The perennial grass layer in LU 2B was dense and was dominated by *Heteropogon triticeus* and *Sarga plumosum var plumosum* (Appendix 24).

Landscape function analysis transects in LU 2B usually consisted of a single continuous patch comprised of perennial grasses, sometimes with suckering low woody vegetation. The high density of perennial grasses resulted in consistently high values for patch area index and landscape organisation index (Table 7.3) which indicates high levels of overland resource control. Bare soil (inter-patches) occurred in 10 out of the 28 LU 2B sites. These were generally small areas associated with tree bases, termite mounds, and fire scars and showed no sign of downslope extension. There was little evidence of soil sediment.

7.2.2 Post-mining landscape analogue - Land unit 2C

Land unit 2C occurs in areas that are less well drained than the surrounding LU 2B. The level of the wet season watertable can rise locally above the ground surface. Yellow to grey soils indicate that these areas are periodically waterlogged (D. Tongway personal communication, 2006).

Vegetation in LU 2C was typically shrubby, layered tall woodland (Figure 7.2). Species composition of canopy species was similar to LU 2B, although the big tree layer was not as well developed (Table 7.1) and the contribution of framework species to the canopy was lower (Table 7.4). The canopy layer was dominated by *Eucalyptus tetrodonta* and *Corymbia nesophila*. In contrast to LU 2B, *Erythrophleum chlorostachys* was only a minor component of the canopy and sub-canopy layers (Appendix 23). The small tree and the tall shrub layers were more developed (Table 7.1) and incorporated species that are generally absent from LU 2B. Substantial contributions of *Livistona muelleri* and *Lophostemon suaveolens* to the vegetation are indicative of LU 2C. The perennial grass layer in LU 2C was less dense than in LU 2B and the species composition differed. The perennial grasses that dominated LU 2C sites were *Eriachne pallescens* and *Alloteropsis semialata* although *Heteropogon triticeus and Sarga plumosum var plumosum* were still substantial components of the perennial grass layer (Appendix 24).

Values for patch area index, and landscape organisation index were lower in LU 2C sites than in the other native forest reference land units (Table 7.3). All LU 2C sites had areas of bare soil. These inter-patches were larger on average than in LU 2B sites. These areas of bare soil were mostly associated with tree bases, termite mounds and fire scars. Some areas of bare soil occurred simply due to low grass cover and low levels of leaf litter. There was some evidence of sand sediment. Two of the LU 2C sites with low levels of tree leaf litter and vegetation cover had been burnt in a hot, late dry season fire in the previous December (2006), seven months prior to the LFA sampling.

7.2.3 Post-mining landscape analogue - Land unit 5K

Land unit 5K occurs on gentle eroding slopes of the bauxite plateau. Outcropping ironstone or dense ironstone gravel is characteristic. Soils are shallow, gravelly reddish-brown loams overlain with a thick layer of ironstone gravel.

Vegetation in LU 5K was typically layered, tall open woodland (Figure 38b). Both of the tree layers were less developed than in LU 2B (Table 28). The canopy layer was dominated by *Corymbia nesophila* with *Eucalyptus tetrodonta* and *Erythrophleum chlorostachys* as co-dominants (Appendix 23). The shrub layers were similar in volume to land unit 2C but due to the openness of the tree layers contributed a higher proportion to the total canopy volume. The perennial grass layer in LU 5K was open to dense. Species that contributed most to the canopy volume of perennial grasses were *Heteropogon triticeus* and *Sarga plumosum var plumosum* (Appendix 24).

Two of the LU 5K sites occurred on gentle slopes (approximately 3°), and had a thick layer of ironstone gravel indicating that runoff and sheet erosion has been operating over a long period. Therefore, the gravel layer was not the result of disturbance. The combined ironstone lag and perennial grass were interpreted as 'patch' in LFA terminology in spite of low levels of tree leaf litter because they appeared to be stable slopes with no evidence of active erosion or deposition. The higher levels of water runoff implied by the ironstone gravel surface probably result in lower values for soil moisture in LU 5K sites than in the other native forest reference land units (D.Tongway, personal communication, 2007). The other two LU 5K sites were level and had a combination of outcropping ironstone boulders and perennial grasses. Values for patch area index and landscape organisation index were lower than for LU 2B but higher than LU 2C due to the higher density of perennial grasses (Table 7.3).

Figure 7.1 Pre-mining native forest land unit 2B

(a)

(b)

Figure 7.2 Post-mining landscape analogue land units

(a) Land unit 2C (b) Land unit 5K

7.2.4 Summary of native forest reference values

The native forest reference sites indicate the vegetation and soil potential for the local environment. Ideally, sites would be monitored through time to assess changes in landscape function values in response to disturbance and determine a range for the reference landscape function values. Landscape function analysis was only conducted once at each site and within a short time period. Nevertheless, the values obtained are considered to be representative and to provide a reasonable estimate of error as 23 out of the 36 native forest reference sites had burnt at different times in the dry season prior to this assessment. Of these 23 sites, two sites had experienced late dry season fires that damaged the canopy. Under current management, native forest sites on Cape York Peninsula are likely to burn on an annual or biennial basis (Felderhof and Gillieson, 2006).

The bauxite plateau is a very level landscape. The two land units that occur on the bauxite plateau generally had slopes of less than one degree. Land unit 5K sites had gentle slopes up to 3 degrees. Topographic variation across all three native forest reference land units was finely scaled (tens of centimetres), and was mostly due to the presence of tree bases (Figure 7.3). The presence of unconsolidated materials at the surface was entirely due to biotic processes. Termite and ant activity, animal diggings and occasional tree overturn were the main processes leading to presentation of unconsolidated materials in native forest reference sites (Figure 7.3). The other process leading to soil surface variation was fire history. Fire scars and altered grass species composition appear to persist for years in places where logs had fallen and burnt on the ground (Figure 7.3). Given the level topography and well-developed perennial vegetation, there is low potential for downslope or downwind movement of unconsolidated materials. At the same time there is high potential for resource retention by biogenic processes. Resource control in the native forest reference sites is almost entirely due to the density of perennial vegetation. The interaction of the monsoonal climate and fire with biotic processes over long periods of time has probably contributed to the homogeneity of topography and vegetation on the bauxite plateau.

Figure 7.3 Causes of spatial variation in native forest reference sites

(a) Bioturbation by termites and ants; and (b) tree overturn are the main processes causing presentation of unconsolidated materials at the surface. (c) Intense heat from burning of fallen logs leaves persistent signatures and alters grass species composition. (d) Tree pedestals cause minor topographic variation.

Values for density of big trees, mean height of the tallest vegetation layer, mean total percent foliage cover, mean total canopy volume and mean total basal area all varied between the native forest reference land units (Table 7.2). The relative contribution of each woody vegetation layer to the total canopy volume also varied as the density of big trees decreased and the density of small trees increased. From land unit 2B to 2C and 5K, the contribution of big trees to total canopy volume correspondingly decreased from 69% to 51% to 42%. At the same time the contribution of small trees to the total canopy volume increased from 26% to 39% to 45%. The contribution of the combined shrub layers to total canopy volume also increased from 5% to 10% to 13% respectively. Differences in vegetation structure between the three native forest reference land units were significant ($P < 0.01$). Perennial grass cover was highest in LU 2B and lowest in LU 2C.

Framework species contributed more than 83% of the total canopy volume of woody species in all native forest reference land units with a maximum in LU 2B and a minimum in LU 2C (Table 7.4). Together, framework and 'desirable' species contributed more than 98% of the total canopy volume of woody species in all native forest reference land units. A comparison of vegetation composition between the three native forest reference land units based on relativised canopy volume data for woody species found no significant differences. Hierarchical cluster analysis grouped many of the *a priori* site types together, although no tight or exclusive groupings were found. Land units 2C and 5K were all in one half of the dendrogram interspersed with some of the types of LU 2B. There was some clustering of the various types of LU 2B in the other half of the dendrogram but with some interspersion of types.

The differences in vegetation structure between land units were reflected in values for the landscape function indices (Table 7.3). The main cause of variation in values for landscape organisation was variation in the density of perennial grasses, which in turn appeared to be related to variation in soil moisture due to drainage or runoff, the abundance of rocks on the surface, and fire history. Although native forest reference sites are burnt frequently they retained relatively high stability values due to the ability of the dominant plant species to survive fire. Across the native forest reference land units, mean values for stability, infiltration and nutrient cycling were highest in LU 2B and lowest in LU 5K. The most important factor contributing to these values was dense, perennial, fire tolerant vegetation cover.

There was very little evidence of erosion or movement of surface materials in any of the native forest reference land units. Mean inter-patch length was lowest in LU 2B and highest in LU 2C (Table 7.3). Values for mean basal area of perennial grasses in land units 2C and 5K were lower than LU 2B (Table 7.1). However, the density of perennial grasses in all reference sites was such that any unconsolidated materials were captured over very short distances indicating good overland resource control. Where deposited materials occurred, they usually consisted of a sparse layer of sand < 5% or 5 to 20% cover, evenly distributed over the surface. A few shallow depressions in land unit 2C where the grass cover was sparse due either to poor drainage or fire had 20 to 50% deposited materials, consisting of sand. There was no evidence of soil sediment in any LU 5K sites. The density of perennial grasses in combination with woody debris contributed to high surface roughness scores which raised values for the infiltration index in native forest reference sites.

Most reference sites had high scores for surface resistance to erosion (Class 5), which is an indicator that increases stability index values but reduces infiltration index values. The high scores for surface resistance to erosion were due to high levels of perennial grass cover. All soils were very stable in the slake tests, and no aggregate breakdown or clay dispersion was observed. Slake tests contributed to high surface stability index values and raised the values for the infiltration index. All soil surface textures were sandy loams, with moderate infiltration rates.

There were very few surface crusts in the native forest reference sites, which also contributed to high stability index values. Where surface crusts occurred, they were associated with localised reduction in perennial vegetation canopy cover, perennial grass cover and leaf litter in patches where fires may have been more intense. Physical crusts were formed by raindrop action on bare soil and biologically derived crusts occurred in association with termite mounds.

Variation in perennial vegetation cover appeared to be the main factor causing variation in index values for stability, infiltration and nutrient cycling in reference sites. Differences in scores for rainsplash protection, perennial vegetation cover and leaf litter cover between sites were a response to variation in the density of perennial grasses, in the amount of woody vegetation foliage cover and to fire history. Fire removed much of the litter layer which reduced nutrient cycling index values in sites that had been recently burnt. Frequent fire probably accounts for the generally low values for litter cover and decomposition. The low values for leaf litter were the main cause of reduced infiltration and nutrient cycling values.

Table 7.1 Individual vegetation layer statistics by age class and land unit

	Mine rehabilitation sites					Native forest sites		
						Pre-mining	Post-mining landscape analgoues	
Indicator	1 - 2 y (n=6)	3 - 4 y (n=6)	5 - 8 y (n=6)	9 - 16 y (n=8)	> 16 y (n=5)	LU 2B (n=28)	LU 2C (n=4)	LU 5K (n=4)
Layer 1 - Perennial Grasses								
Mean basal area m^2/ha (se)[1]	31 (± 13)	45 (± 16)	29 (± 10)	5 (± 2)	6 (± 2)	589 (± 49)	267 (± 70)	480 (± 113)
Mean % foliage cover (se)	4 (± 1.2)	3 (± 0.7)	4 (± 1.2)	1 (± 0.8)	1 (± 0.4)	56 (± 5.4)	29 (± 7.5)	43 (± 11.3)
Mean canopy volume m^3/ha (se)	400 (± 150)	500 (± 150)	700 (± 200)	200 (± 100)	150 (± 50)	6,900 (± 550)	4,250 (± 1600)	5,750 (± 1550)
Layer 2 - Woody stems < 1 cm dbh (Low shrub layer)								
Mean height (m) (se)	0.9 (± 0.1)	0.9 (± 0.1)	0.7 (± 0.1)	0.8 (± 0.1)	0.5 (± 0.1)	0.5 (± 0.1)	0.5 (± 0.01)	0.5 (± 0.1)
Mean % foliage cover (se)	4 (± 1.0)	2 (± 0.3)	2 (± 0.7)	3 (± 0.4)	3 (± 1.2)	10 (± 1.0)	8 (± 0.9)	9 (± 0.9)
Mean canopy volume m^3/ha (se)	1,000 (± 250)	650 (± 150)	600 (± 250)	700 (± 150)	650 (± 250)	1,400 (± 150)	1,550 (± 450)	1,250 (± 150)
Mean number of stems/ha (se)	1,570 (± 350)	1,410 (± 290)	2,920 (± 840)	7,030 (± 1960)	6,240 (± 2150)	11,820 (±1250)	12,630 (± 1710)	13,200 (± 2770)
Layer 3 - Woody stems > 1 < 10 cm dbh (Tall shrub layer)								
Mean height (m) (se)	2 (± 1.0)	4 (± 0.1)	4 (± 0.1)	5 (± 0.5)	5 (± 0.2)	5 (± 0.2)	5 (± 0.7)	5 (± 0.5)
Mean % foliage cover (se)	4 (± 1.9)	24 (± 5.0)	15 (± 2.7)	10 (± 2.7)	15 (± 3.5)	3 (± 0.4)	6 (± 1.5)	7 (± 1.5)
Mean canopy volume m^3/ha (se)[2]	1,250 (± 600)	10,150 (± 1750)	7,700 (± 1550)	5,450 (± 1650)	8,050 (± 2050)	1,400 (± 150)	2,500 (± 500)	2,500 (± 500)
Mean number of stems/ha (se)[3]	120 (± 50)	770 (± 180)	500 (±100)	560 (± 160)	1,000 (± 220)	320 (± 40)	730 (± 210)	610 (± 150)
Layer 4 - Woody stems > 10 < 35 cm dbh (Small tree layer)								
Mean height (m) (se)	0	4 (± 1.5)	7 (± 0.3)	9 (± 0.5)	10 (± 0.7)	15 (± 0.3)	14 (± 0.4)	12 (± 1.1)
Mean % foliage cover (se)	0	3 (± 1.6)	4 (± 0.9)	12 (± 4.9)	33 (± 8.2)	13 (± 0.7)	15 (± 0.9)	15 (± 3.5)
Mean canopy volume m^3/ha (se)	0	2,100 (± 1150)	2,700 (± 650)	9,050 (± 3850)	28,500 (± 7600)	14,200 (± 850)	15,300 (± 1600)	13,450 (± 3250)
Mean number of stems/ha (se)	0	20 (± 10)	40 (± 10)	120 (± 40)	320 (± 60)	140 (± 10)	160 (± 20)	180 (± 40)
Mean basal area m^2/ha (se)	0	0.2 (± 0.1)	0.6 (± 0.1)	2.5 (± 1.0)	8.4 (± 2.2)	4.5 (± 0.3)	5.0 (± 0.5)	4.7 (± 0.7)
Layer 5 - Woody stems > 35 cm dbh (Big tree layer)								
Mean height (m) (se)	0	0	0	0	0	24 (± 0.2)	23 (± 0.5)	19 (± 1.5)
Mean % foliage cover (se)	0	0	0	0	0	26 (± 2.0)	14 (± 2.1)	9 (± 3.9)
Mean canopy volume m^3/ha (se)	0	0	0	0	0	38,450 (± 3050)	20,150 (± 3750)	12,350 (± 5200)
Mean number of stems/ha (se)	0	0	0	0	0	50 (± 5)	30 (± 5)	30 (± 10)
Mean basal area m^2/ha (se)	0	0	0	0	0	9.6 (± 0.6)	5.0 (± 0.9)	3.7 (± 1.3)

1. Values for mean basal area of perennial grasses, mean % foliage cover and mean height have been rounded to the nearest unit. Standard errors have been rounded to one decimal place due to the small sample size.
2. Values for means and standard errors of canopy volume have been rounded to the nearest 50 due to the large values and large range of values.
3. Values for mean total number of stems have been rounded to the nearest 10.

Table 7.2 Total values for woody vegetation by age class and land unit

	Mine rehabilitation sites					Native forest sites		
						Pre-mining	Post-mining landscape analogues	
Indicator	1 - 2 y (n=6)	3 - 4 y (n=6)	5 - 8 y (n=6)	9 - 16 y (n=8)	> 16 y (n=5)	LU 2B (n=28)	LU 2C (n=4)	LU 5K (n=4)
Mean height of tallest layer (m) [2]	2 (± 1.0)[1]	4 (± 1.5)	7 (± 0.3)	9 (± 0.5)	10 (± 0.7)	24 (± 0.2)	23 (± 0.5)	19 (± 1.5)
Mean total % foliage cover	8 (± 2.9)	29 (± 6.9)	21 (± 4.3)	25 (± 8.0)	52 (± 12.9)	52 (± 4.1)	43 (± 5.4)	40 (± 9.8)
Mean total canopy volume m^3/ha [3]	2,250 (± 850)	12,900 (± 3050)	11,000 (± 2450)	15,200 (± 5650)	37,200 (± 9900)	55,450 (± 4200)	39,500 (± 6300)	29,550 (± 9100)
Mean total number of stems/ha [4]	1,690 (± 400)	2,200 (± 480)	3,460 (± 950)	7,710 (± 2160)	7,560 (± 2430)	12,330 (± 1585)	13,550 (± 1945)	14,020 (± 2970)
Mean total basal area m^2/ha [5]	0	0.2 (± 0.1)	0.6 (± 0.1)	2.5 (± 1.0)	8.4 (± 2.2)	14.1 (± 0.9)	10.0 (± 1.4)	8.4 (± 2.0)

1. Values shown in brackets are 1 SE
2. Values for mean height and mean total % foliage cover have been rounded to the nearest unit. Standard errors have been rounded to one decimal place due to the small sample size.
3. Values for means and standard errors of canopy volume have been rounded to the nearest 50 due to the large values and large range of values.
4. Values for mean total number of stems have been rounded to the nearest 10.
5. Similar stand values for basal area in m^2/ha do not mean that biomass values are the same. First, species differ in the allometric relationships between basal area and biomass. Second, most biomass is in big trees. The high value for basal area m^2/ha in the oldest age class of mine rehabilitation is achieved through high stem densities of small trees which do not have the same biomass for basal area as big trees.

Table 7.3 Landscape function indices by age class and land unit

Landscape function indicator	Mine rehabilitation sites					Native forest sites		
						Pre-mining	Post-mining landscape analogues	
	1 - 2 y (n=6)	3 - 4 y (n=6)	5 - 8 y (n=6)	9 - 16 y (n=8)	> 16 y (n=5)	LU 2B (n=28)	LU 2C (n=4)	LU 5K (n=4)
Mean Patch Area Index Value	0.18 (± 0.07)	0.53 (± 0.11)	0.69 (± 0.15)	0.87(± 0.06)	0.93 (± 0.05)	0.96 (± 0.01)	0.77 (± 0.11)	0.93 (± 0.05)
Mean Landscape Organisation Index Value	0.18 (± 0.07)	0.37 (± 0.08)	0.58 (± 0.13)	0.84 (± 0.04)	0.93 (± 0.04	0.98 (± 0.01)	0.81 (± 0.07)	0.93 (±0.07)
Mean Inter-patch Length (m)	4.4 (± 1.00)	6.7 (± 2.44)	4.6 (± 1.58)	2.5 (± 0.92)	1.6 (± 1.36)	0.4 (± 0.14)	2.5 (± 0.83)	0.7 (± 0.7)
Mean Stability Index Value	51 (± 2.13)	57 (± 1.15)	58 (± 2.13)	68 (± 2.16)	64 (± 1.01)	74 (± 1.19)	72 (± 3.50)	71 (± 2.94)
Mean Infiltration Index Value	36 (± 1.55)	39 (± 2.31)	43 (± 1.15)	47 (± 2.73)	52 (± 2.00)	48 (± 0.96)	47 (± 3.00)	44 (± 1.67)
Mean Nutrient Cycling Value	21 (± 1.88)	26 (± 2.29)	29 (± 1.71)	37 (± 3.26)	42 (± 2.98)	35 (± 1.32)	35 (± 3.69)	33 (± 1.49)

Values shown in brackets are 1 SE. Values for mean stability index, infiltration index and nutrient cycling index have been rounded to the nearest unit.

Table 7.4 Summary statistics for framework and desirable species

Indicator	Mine rehabilitation sites					Native forest sites		
						Pre-mining	Post-mining landscape analogues	
	1 - 2 y (n=6)	3 - 4 y (n=6)	5 - 8 y (n=6)	9 - 16 y (n=8)	> 16 y (n=5)	LU 2B (n=28)	LU 2C (n=4)	LU 5K (n=4)
Stems per hectare of framework species	45 (± 30)	67 (± 43)	163 (± 86)	356 (± 228)	240 (± 95)	4955 (± 500)	2103 (± 330)	7517 (± 2891)
FC[1] of framework species	0.03 (± 0.02)	0.1 (± 0.08)	0.3 (± 0.2)	0.7 (± 0.4)	5.2 (± 2.3)	42.4 (± 2.3)	26.7 (± 2.2)	31.8 (± 6.3)
FC of desirable species	0.5 (± 0.3)	1.7 (± 1.1)	4.1 (± 2.0)	1.6 (± 0.5)	5.7 (± 2.3)	6.0 (± 0.6)	12.5 (± 1.3)	6.9 (± 3.7)
FC of other species	7.7 (± 2.1)	26.9 (± 5.2)	16.5 (± 3.5)	22.8 (± 7.2)	41.2 (± 7.0)	3.6 (± 0.8)	4.1 (± 1.3)	1.6 (± 0.7)
% of total CV[2] consisting of framework species	< 1 (± 0.2)	< 1 (± 0.1)	< 1 (± 0.6)	6 (± 3.1)	10 (± 3.8)	93 (± 1.0)	83 (± 2.2)	88 (± 7.3)
% of total CV consisting of desirable species	8 (± 4.7)	8 (± 6.3)	21 (± 9.8)	9 (± 2.5)	10 (± 4.0)	5 (± 0.7)	15 (± 2.3)	11 (± 7.4)
% of total CV consisting of other species	92	92	78	85	80	2	2	1

1. FC = percent foliage cover. FC values are absolute values and add up to the total foliage cover values (not 100).
2. CV = canopy volume. CV values are the percentage of the total canopy volume. CV values add up to 100. CV values have been rounded to the nearest unit.

7.3 Vegetation and landscape functionality in mine rehabilitation sites

Mine rehabilitation sites were stratified by age. Field work, however, took place over nearly two years. Consequently, sites that were originally selected as 1 to 2 years old, were actually 2 to 3 years old at the time of vegetation sampling and so on. In presenting the results, the original age categories are retained to avoid confusion. Fertiliser type, rate of fertiliser application, method of seed application, and fire history of each site is tabled in Appendix 7. Composition of seed mix for the mine rehabilitation sites is tabled in Appendix 4.

7.3.1 Mine rehabilitation - 1 to 2 years old

There were two woody vegetation layers in this age class, low shrubs and tall shrubs (Figure 7.4). Values for mean vegetation height and mean total percent foliage cover were lowest in this age class (Table 7.2). Low shrubs contributed 44% and tall shrubs 56% of the total canopy volume (Table 7.1). Species that contributed most to the canopy volume were *Acacia torulosa, Dodonea* spp. and *Trema aspera* (Appendix 23). The perennial grass layer was sparse. Species that contributed most to canopy volume of perennial grasses were *Sarga plumosum var plumosum, Heteropogon contortus, Heteropogon triticeus* and *Eriachne* spp. (Appendix 24). These species established from the soil seedbank applied as fresh topsoil.

Low values for landscape organisation 1 to 2 year old sites (Table 7.3) were due to a combination of large areas of bare soil, low levels of leaf litter, low levels of plant cover and inefficient bank-trough structures (Figure 7.4). A number of transects consisted entirely of inter-patch. Sheet erosion was usually evident, deposited materials were often present, and vegetation cover was low. In spite of this, the values for the stability index were moderate. Surface stability in these sites was due to the presence of surface crusts and cryptogams and because assimilation of even small amounts of organic matter into the soil were sufficient to stabilise soil aggregates. Mean infiltration index values were low in spite of the sandy loam texture due to low values for perennial vegetation cover, leaf litter cover and surface roughness. Mean nutrient cycling index values were also low, primarily due to low perennial vegetation cover and leaf litter cover but also due to low surface roughness and less stable soils.

7.3.2 Mine rehabilitation - 3 to 4 years old

There were three woody vegetation layers in the 3 to 4 year old age class due to the development of a small tree layer (Table 7.1) and (Figure 7.5). Mean vegetation height was double and mean total canopy volume was more than triple the values in the 1 to 2 year old age class (Table 7.2). Low shrubs contributed 5%, tall shrubs 79% and small trees 16% of the total canopy volume. Species that contributed most to the canopy volume were *Dodonea* spp., *Acacia torulosa, Grevillea pteridifolia,* and *Trema aspera* (Appendix 23). The perennial grass layer was sparse (Table 7.1). Relative to 1 to 2 year old sites, foliage cover of perennial grasses was lower and basal area of perennial grasses was higher. This is possibly due to consumption of the grass canopy by termites which was observed in this age class. Species that contributed most to the perennial grass canopy volume were *Hetropogon triticeus, Sarga plumosum var plumosum* and *Andropogon gayanus* (Appendix 24).

Mean landscape organisation values in the 3 to 4 year age class were double the mean values of 1 to 2 year old sites, but were still low relative to native forest reference sites (Table 7.3). Higher landscape organisation values were mainly due to more extensive cover of leaf litter associated with the growth of shrubs. Higher mean values for perennial vegetation cover, combined with increasing values for leaf litter cover, were associated with a step up in values for stability and nutrient cycling and a smaller increase for infiltration.

7.3.3 Mine rehabilitation - 5 to 8 years old

There were three woody vegetation layers in the 5 to 8 year old age class (Table 7.1) and (Figure 7.5). Mean vegetation height was almost double that in the 3 to 4 year old age class (Table 7.2). Mean total basal area was more than double that in 3 to 4 year old sites. The increased basal area was accompanied by increased canopy volume of small trees. However, the mean percent foliage cover for all woody layers combined was lower than in the younger age class. Low shrubs contributed 5%, tall shrubs 70% and small trees 25% to the total canopy volume. Species that contributed most to the canopy volume were *Acacia rothii*, *Acacia auriculiformis, Acacia torulosa, Grevillea pteridifolia,* and *Dodonea* spp. (Appendix 23).

The perennial grass layer was sparse. The mean basal area of perennial grasses was lower than in 3 to 4 year old sites although mean percent foliage cover was slightly higher. Species that contributed most to the perennial grass canopy volume were *Andropogon gayanus, Eriachne* spp., *Heteropogon triticeus* and *Sarga plumosum var plumosum.* The higher value for percent foliage cover and lower basal area was possibly due to the influence of *Andropogon gayanus* which had a large canopy volume for its basal area relative to other grass species.

Mean landscape organisation values were higher than in 3 to 4 year old sites although the standard error was high due to the presence of large inter-patches in some sites (Table 7.3). Although the mean total percent foliage cover of woody vegetation was lower than in 3 to 4 year old sites, index values for landscape organisation, stability, infiltration and nutrient cycling were slightly higher due to accumulating leaf litter and increasing evidence of decomposition.

7.3.4 Mine rehabilitation - 9 to 16 years old

There were three woody vegetation layers in the 9 to 16 year old age class (Table 7.1) and (Figure 7.6). Mean vegetation height was slightly higher relative to the 5 to 8 year old age class, and mean total basal area had expanded fourfold (Table 7.2). Mean total percent foliage cover was higher than in the 5 to 8 year old age class and was accompanied by a shift in dominance from tall shrubs to small trees. Low shrubs contributed 5%, tall shrubs 36% and small trees 59% of the total canopy volume. Species that contributed most to the canopy volume were *Acacia mangium, Dodonaea* spp.*, Alstonia scholaris,* and *Grevillea pteridifolia* (Appendix 23). The perennial grass layer was very sparse in this age class. Species that contributed most to the perennial grass canopy volume were *Brachiaria decumbens* and *Andropogon gayanus* (Appendix 24).

Higher mean values for landscape organisation in this age class were due largely to the developing perennial vegetation, and accumulating leaf litter. The mean stability index value was highest in this age class of mine rehabilitation and did not increase past this point.

7.3.5 Mine rehabilitation - > 16 years old

There were still only three woody vegetation layers in the > 16 year old age class. Relative to the 9 to 16 year old age class, mean vegetation height had increased slightly, mean total percent foliage cover was double and mean total basal area of woody stems had more than tripled (Table 7.2). Low shrubs contributed 2%, tall shrubs 22% and small trees 76% to the total canopy volume. Species that contributed most to the canopy volume were *Alstonia scholaris, Eucalyptus brassiana, Alphitonia excelsa, Acacia mangium, Grevillea pteridifolia* and *Parinari nonda (*Appendix 23). The perennial grass layer was most poorly developed in this age class. Species that contributed most to the perennial grass canopy volume were *Eriachne* spp. and *Sarga plumosum var plumosum* (Appendix 24).

Mean values for landscape function indices were higher in this age class, with the exception of the stability index, and the standard error of the mean values decreased. This was due to the increasing contribution of the maturing vegetation to leaf litter cover and perennial woody vegetation cover across all sites in this age class (Figure 7.6).

(a)

(b)

Figure 7.4 Young mine rehabilitation sites

(a) 1 to 2 year old site B-11 (Gar-6) at the end of its 2nd year; and (b) poorly aligned bank-trough structures in another 1 to 2 year old site B-20 (Herring-18) at the end of its 2nd year. The intention of bank-trough structures is to reduce water run-off, facilitate plant establishment and promote soil stabilisation. As such they should be aligned along slope contours. However, due to the undulations in the post-mining land surface, many of the bank-trough structures are aligned downslope, promoting downslope water flow and creating instability.

(a)

(b)

Figure 7.5 Mine rehabilitation sites - 1

(a) 3 to 4 year old site B-03 (Kauri-5) at the end of its 4th year; and (b) 5 to 8 year old site B-15 (Mackerel-6) at the end of its 7th year.

Figure 7.6 Mine rehabilitation sites - 2

(a) 9 to 16 year old site B-7 (Grevillea-1) at the end of its 12th year; and (b) 17 to 23 year old site B-24 (Bass-2) at the end of its 22nd year.

7.4 Comparison of landscape functionality

Overall, spatial heterogeneity in the mine rehabilitation sites was less organised than in the native forest reference sites. In contrast to the native forest reference sites, very little of the spatial heterogeneity was caused by fire or biotic processes. There was no evidence of animal diggings and relatively few signs of termite activity. There was much more topographic variation. Engineered bank and trough structures created topographic variation over scales of a few metres squared although they varied in width, spacing and persistence. There was also larger scale (10 to 100 m^2) topographic variation due to undulations in the post-mining landscape corresponding with the top of the ironstone layer (Figure 2.10).

Values for all landscape function indices increased with rehabilitation age, although there was large variation in values in the two oldest age classes of mine rehabilitation. Age class and soil type (fresh or stockpiled) accounted for a significant amount of variation in all landscape function indices (Table 7.5). Fire history (burnt or unburnt) also accounted for a significant amount of variation in the infiltration index and nutrient cycling index values but not stability. It should be noted that a site was classified as burnt if it had been burnt at any time since rehabilitation. The category 'burnt', therefore, does not differentiate between time since fire or age of rehabilitation at the time of burning.

Table 7.5 ANOVA results for LFA indices in mine rehabilitation

	Explanatory variables		
Response variables	**Age Class**	**Fire history (Burnt/unburnt)**	**Soil type (Fresh/Stockpiled)**
	F (4, 83)	F (1, 83)	F (1, 83)
Stability Index Value	19.2 ***	1.3 ns	8.3 **
Infiltration Index Value	18.9 ***	12.4 ***	4.82 *
Nutrient Cycling Index Value	26.7 ***	19.7 ***	4.96 *

Data used for the variance components analysis were transect values not site values. Transect values rather than site values were used for this analysis because in a number of cases some transects within a site had been burnt while other transects had not. Use of mean site values would not have accurately reflected the effects of fire. Significance of the F-statistic is indicated by asterisks * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$; and *** = $P \leq 0.001$. ns = not significant.

7.4.1 Stability index values

Stability index values increased with rehabilitation age (Table 7.3). Non-linear regression found that age accounted for 56.6% of variance ($P < 0.001$) in stability index values in mine rehabilitation sites (Figure 7.7). Values around the fitted curve show a great deal of variation in 9 to 16 year old sites. Soil type accounted for a small but significant component of the variation in stability index values (Table 7.5). Stability index values were lower in sites where stockpiled soil had been used than in sites where fresh soil had been used.

Although soil type accounted for a significant component of variation, it was unrealistic to fit separate regression models for stability index against age of site as all of the rehabilitation sites in which stockpiled soil had been used were aged between 9 and 16 years. The regression model predicted an upper asymptote value of 67 for stability index. This predicted value is slightly higher than the observed mean stability index values in the oldest age class of mine rehabilitation but lower than the mean observed values for all native forest reference land units (Table 7.3).

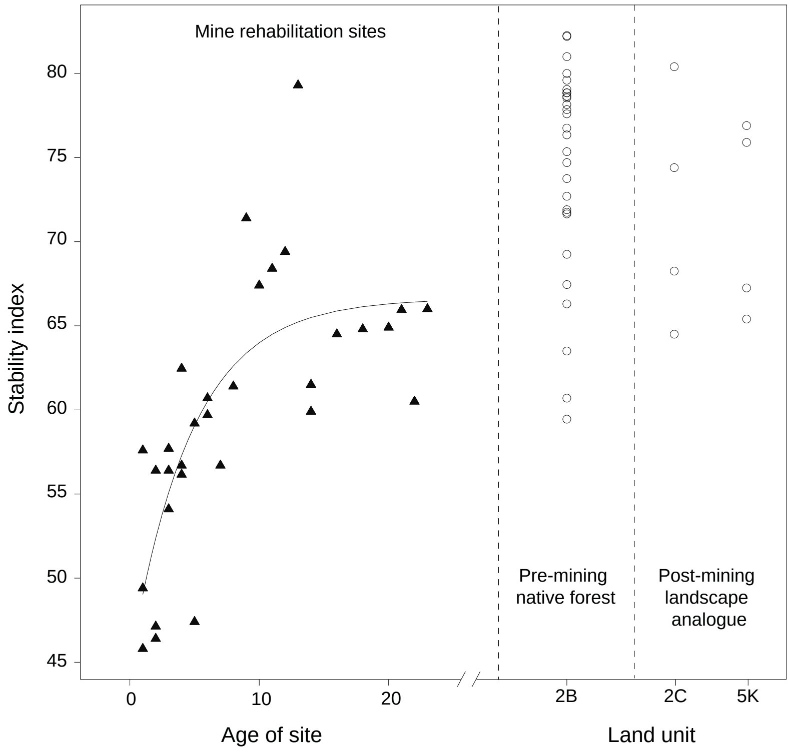

Figure 7.7 Regression of stability index values against age of site

Non-linear regression of mean site stability index values against age of site using an exponential asymptotic curve. The regression model predicted an upper asymptote of 67 for stability in mine rehabilitation sites.

Relatively lower stability index values in even the oldest age class of mine rehabilitation were due to consistently lower scores for rainsplash protection and surface resistance to disturbance than in native forest reference sites. Rainsplash protection assesses the degree to which projected perennial vegetation cover up to a height of 0.5 metres intercepts raindrops (Appendix 8). Low rainsplash protection values are consistent with the low percent foliage cover for low shrubs and the low values for basal area of perennial grasses and mean percent foliage cover for perennial grasses in the older mine rehabilitation sites (Table 7.1).

Surface resistance to disturbance assesses the ease with which the soil can be mechanically disturbed. Native forest reference sites consistently scored the highest possible value for surface resistance to disturbance due to the dense perennial grass cover and the presence of macro-organic matter incorporated into the surface soil layers. Stability index values increased with age in mine rehabilitation sites due to increasing leaf litter cover and decomposition values which were higher than in native forest reference sites. However, the leaf litter effect was not sufficient to compensate for the low percent foliage cover of low shrubs and low density of perennial grasses.

7.4.2 Infiltration index values

Infiltration index values with rehabilitation age. The observed mean infiltration index value in the oldest age class of mine rehabilitation exceeded the observed mean values in the native forest reference land units (Table 7.3). Non-linear regression found that age accounted for 58.6% of variance ($P < 0.001$) in infiltration index values in mine rehabilitation sites (Figure 7.8). Values showed a great deal of variation in sites aged 9 to 16 years. Fire history accounted for a large and significant component of the variation in infiltration index values ($P < 0.001$) and soil type accounted for a small but significant component of the variation ($P < 0.05$) (Table 7.5).

Infiltration index values continued to increase with age after stability had reached an asymptote. The regression model predicted that infiltration values reach an upper asymptote value of 51 at approximately 20 years. Separate regressions for burnt and unburnt sites predicted different asymptotic values for unburnt sites (60) and burnt sites (47) (Figure 7.8). The predicted asymptote even for burnt mine rehabilitation sites is within the range of observed values and slightly higher than the observed means for all native forest reference land units (Table 7.3).

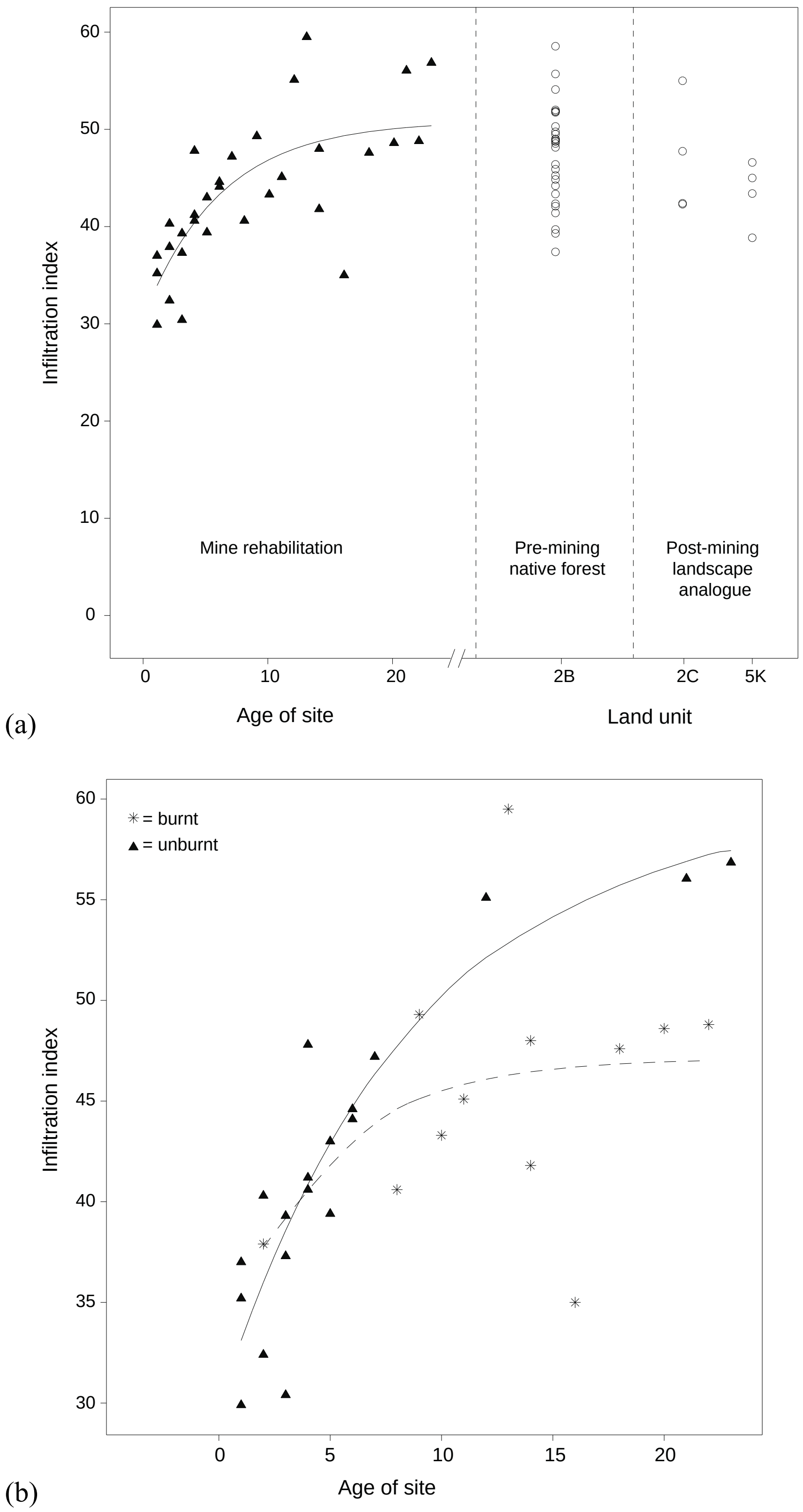

Figure 7.8 Regression of infiltration index values against age of site

Non-linear regression of mean site infiltration index values against age of site using an exponential asymptotic curve. (a) When all mine rehabilitation sites were combined the regression model predicted an upper asymptote of 51 for stability in mine rehabilitation sites. (b) Separate regression models for burnt and unburnt sites in mine rehabilitation predicted an upper asymptote for infiltration of 60 for unburnt sites and 47 for burnt sites.

Soil texture is an important determinant of the infiltration index value. There were no differences in soil texture between mine rehabilitation sites and native forest reference sites, all were sandy loams. Differences in infiltration index values therefore reflect differences in litter cover, perennial vegetation cover, surface resistance to disturbance and surface roughness. High scores for litter cover and decomposition in mine rehabilitation raised the overall infiltration index value relative to native forest reference sites (Table 7.3). Leaf litter cover scores were consistently higher in the older age classes of mine rehabilitation sites than in native forest reference sites even though three out of the five >16 year old sites had been burnt at some time. This probably reflects time since fire as two of the burnt sites had been burnt two years prior to assessment (Appendix 7).

Given the similarity in soil texture across all sites and higher leaf litter cover and decomposition in mine rehabilitation, relatively higher infiltration index values might be expected. However, relatively lower scores for surface resistance to disturbance and soil surface roughness in mine rehabilitation sites consistently reduced the infiltration index value. Infiltration index values in mine rehabilitation were also reduced due to relatively lower scores for perennial vegetation cover in sites which had been burnt. This is consistent with the finding that fire history accounted for a significant proportion of the variation in values for canopy volume in mine rehabilitation. Consistently high scores for perennial vegetation cover raised infiltration index values in native forest reference sites. Although frequent fire results in low scores for leaf litter in the native forest reference sites, it does not affect canopy volume.

7.4.3 Nutrient cycling index values

Nutrient cycling values with rehabilitation age (Table 7.3). Observed mean nutrient cycling index values in the two oldest age classes of mine rehabilitation were higher than in all native forest reference land units. Non-linear regression found that age accounted for 63.8% of variance ($P < 0.001$) in nutrient cycling values in mine rehabilitation sites (Figure 7.9). Values show a great deal of variation in sites aged 9 to 16 years. Fire history accounted for almost as much variation in nutrient cycling values as age ($P < 0.001$), and soil type accounted for a smaller but significant component of the variation ($P < 0.05$) (Table 7.5).

Nutrient cycling index values appeared to reach an upper asymptote after approximately 20 years. The regression model predicted that nutrient cycling values reach an upper asymptote value of 57 for unburnt sites and 37 for burnt sites (Figure 7.9). The predicted asymptote even for burnt mine rehabilitation sites exceeded the observed mean values for nutrient cycling in all of the native forest reference land units.

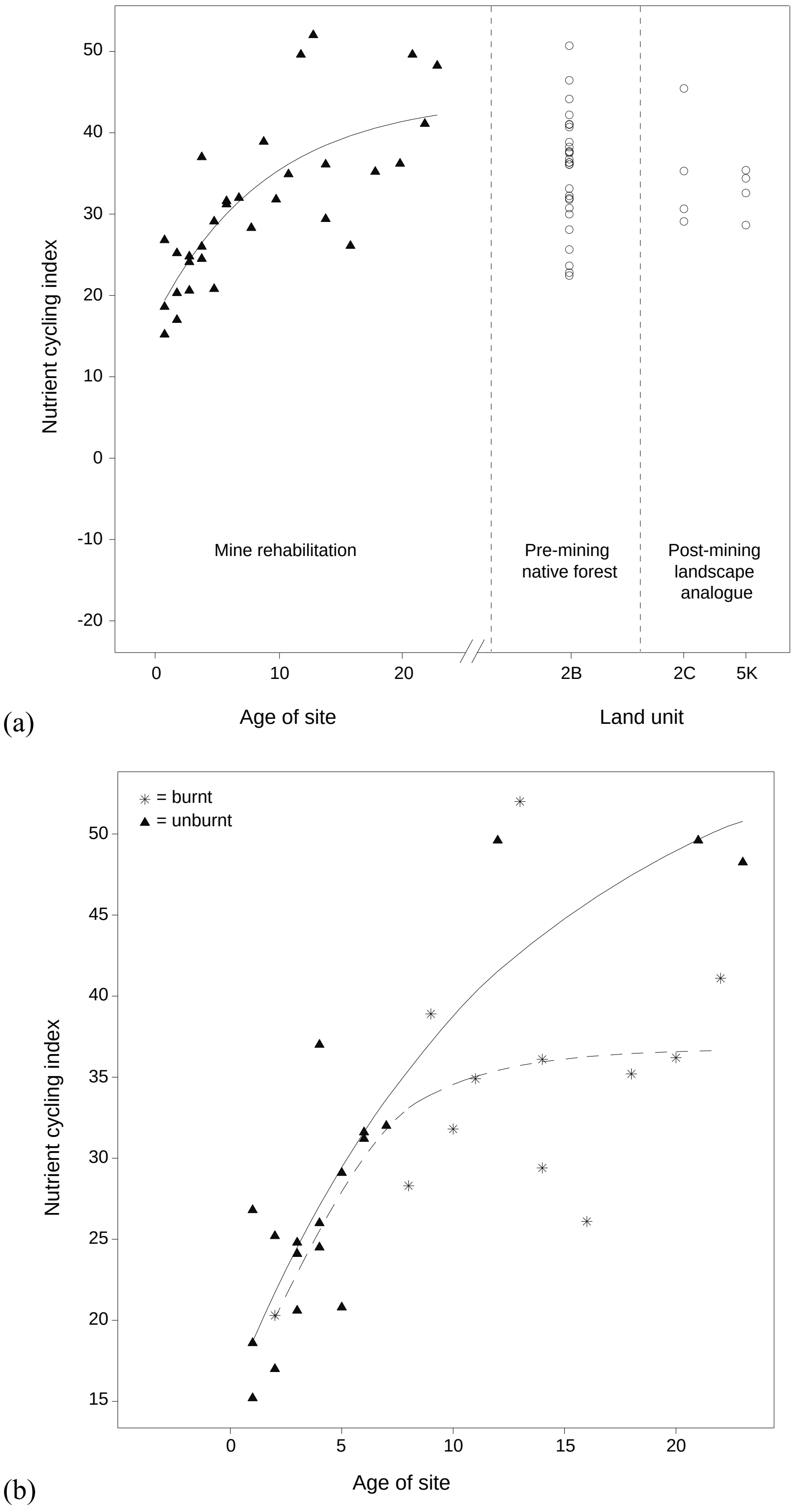

Figure 7.9 Regression of nutrient cycling index values against age of site

Non-linear regression of mean site nutrient cycling index values against age of site using an exponential asymptotic curve. (a) When all rehabilitation sites were combined the regression predicted an upper asymptote of 44 for nutrient cycling in mine rehabilitation sites. (b) Separate regressions for burnt and unburnt sites predicted an upper asymptote for nutrient cycling of 57 for unburnt sites and 37 for burnt sites.

The high nutrient cycling index values were mainly due to the extensive and deep leaf litter cover that was able to accumulate in mine rehabilitation sites that are largely protected from fire. In contrast, most of the native forest reference sites are burnt annually or biennially and leaf litter is not able to accumulate.

Fire had different effects on the landscape function indicators in native forest reference sites compared with mine rehabilitation sites. Frequent fire in native forest reference sites resulted in consistently lower values for leaf litter cover and decomposition. Because the vegetation is fire tolerant, however, there is still high density of perennial grasses and foliage cover of perennial woody vegetation. The relatively small contribution of leaf litter to stability, infiltration and nutrient cycling values in the reference sites was therefore compensated for by the perennial vegetation cover. Dense perennial grass cover in native forest sites resulted in consistently higher scores for surface resistance to disturbance. Higher shrub cover in native forest sites resulted in consistently higher scores for rainsplash protection. Scores for perennial vegetation cover in native forest sites were not reduced by burning of native forest sites in the same way as they were in mine rehabilitation sites due to the canopy height and its fire tolerant qualities. Native forest reference sites also generally had higher scores for soil surface roughness. The pathways by which the stability, infiltration and nutrient cycling values are achieved are therefore different in native forest than in mine rehabilitation.

7.5 Comparison of vegetation composition

7.5.1 Plant species composition

Vegetation composition in mine rehabilitation sites was very different from the native forest reference sites and included many species that are not part of the pre-mining vegetation community. Of 24 perennial grass species recorded in total, 13 species (54%) occurred only in mine rehabilitation and 3 species (12%) occurred only in native forest (Appendix 24). Of 91 woody species recorded, 41 species (45%) occurred only in mine rehabilitation sites, and 14 species (15%) occurred only in native forest reference sites (Appendix 23). There were large and significant differences between native forest reference land units and mine rehabilitation sites in the relative contributions of perennial grass species and woody plant species to the total biomass of the respective vegetation communities (Table 7.6).

Table 7.6 MRPP results (plant species composition)

Species composition measure	T	A
Canopy volume by grass species	-14 ***	0.1
Canopy volume by woody species	-41 ***	0.29

Relativised canopy volume data of 24 grass species was used in one test and relativised canopy volume data of 93 woody species was used in the other. The test statistic (T) describes the separation between *a priori* defined groups. The more negative is T, the stronger the separation. Significance of the T-statistic is indicated by asterisks: * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$; and *** = $P \leq 0.001$. The chance corrected within group agreement (A) is a description of the effect size, independent of the sample size. Values > 0.3 are considered to be high (McCune and Grace, 2002).

Hierarchical cluster analysis based on the relativised canopy volume data of woody plant species separated native forest and mine rehabilitation sites into two distinct groups (Figure 7.10). As the dendrogram was based on relativised data, it reflects relative species contributions to the total canopy volume.

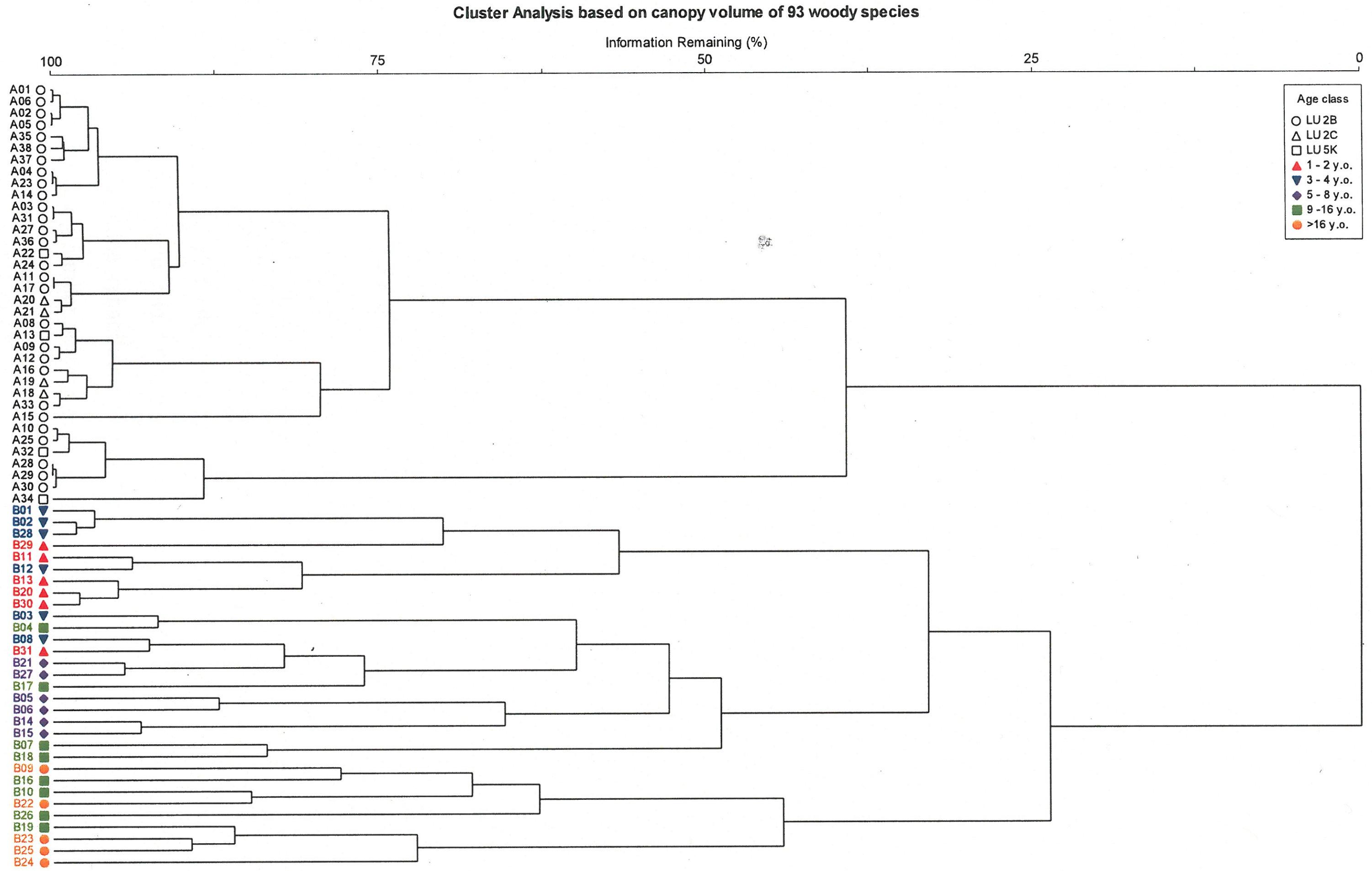

Figure 7.10 Dendrogram of sites based on relativised canopy volume of woody species (Percent chaining = 2.09%)

7.5.2 Density of framework species

The stem density of framework species in mine rehabilitation sites of all age classes was much lower than in all native forest reference land units (Table 7.4). Framework and desirable species are defined in section 4.2.1.1. Differences between native forest reference sites and mine rehabilitation sites in the stem density of framework species and in the percentage of the canopy volume that is made up of framework species were large and significant (Table 7.7).

Table 7.7 MRPP results (density of framework species)

Species composition measure	T^1	A^2
Stems / ha of framework species	-36 ***	0.39
% canopy volume made up of framework species	-45 ***	0.76

1. The test statistic (T) describes the separation between *a priori* defined groups. The more negative is T, the stronger the separation. Significance of the T-statistic is indicated by asterisks: * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$; and *** = $P \leq 0.001$.
2. The chance corrected within group agreement (A) is a description of the effect size, independent of sample size. Values > 0.3 are considered to be high (McCune and Grace, 2002).

Framework species made up at least 83% of the canopy volume in all native forest reference land units. Together, framework species and desirable species contributed more than 98% of the total canopy volume of woody species in all native forest reference land units. In contrast, framework species accounted for < 1% of the total canopy volume in the three youngest age classes of mine rehabilitation (Table 7.4). The contribution of framework species was higher in the two oldest age classes of mine rehabilitation although framework species never contributed more than 10% of the total canopy volume in mine rehabilitation sites. The higher values in the two oldest age classes of mine rehabilitation possibly reflect differences in initial rehabilitation strategies rather than successional or developmental change. In the three youngest age classes of mine rehabilitation only direct seeding was used. Aerial seeding was used in all of the mine rehabilitation sites 16 years and older. Supplementary hand planting of tube stock including framework and desirable species was also used in three out of the eight 9 to 16 year old mine rehabilitation sites and in all of the > 16 year old sites.

Sub-canopy and understorey species of the Weipa plateau that are considered desirable to the long term development of community structure made up more of the canopy volume in mine rehabilitation sites than did framework species. However, framework species and desirable species combined never made up more than 22% of the total canopy volume in mine rehabilitation sites (Table 7.4). Desirable species contributed a higher percentage of the total canopy volume in mine rehabilitation sites than they did in the pre-mining native forest reference LU 2B and a similar percentage to the post-mining landscape analogue land units, 2C and 5K. Most of the canopy volume in mine rehabilitation sites was made up of other species (Figure 7.11).

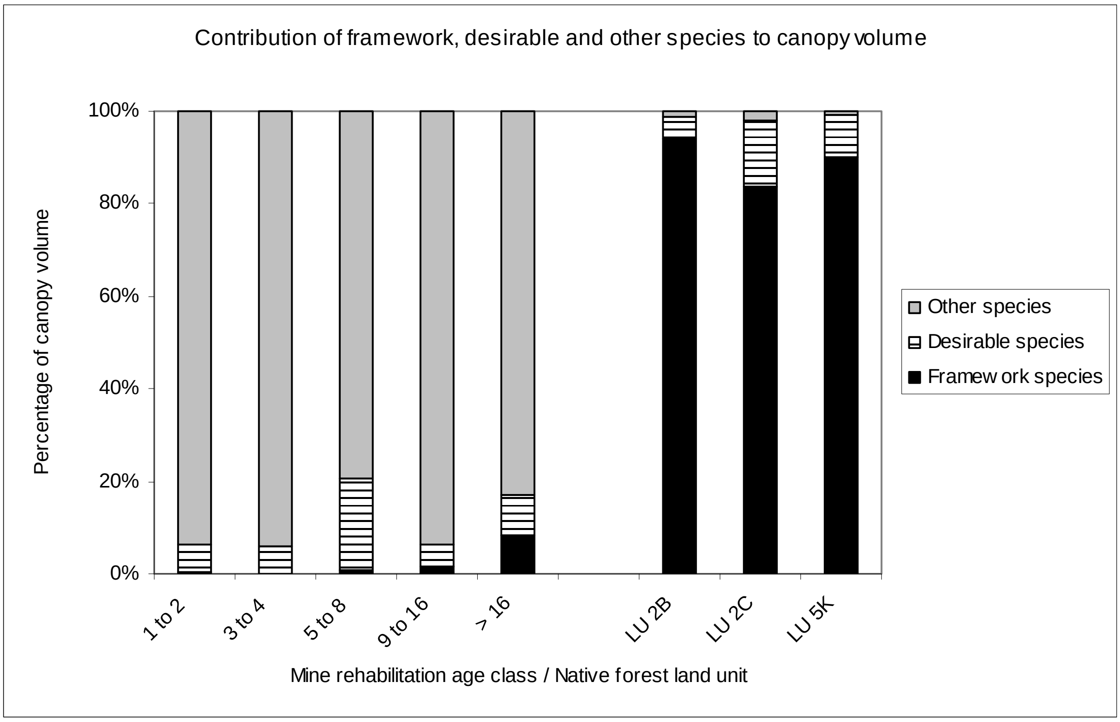

Figure 7.11 Canopy volume of framework, desirable and other species

7.5.3 Density of destabilising species

It was initially intended to assess the density of species with significant potential to disrupt ecosystem development. Reddell and Hopkins (1994) identified aggressive weedy species, especially exotic grasses that produce high biomass, as species with significant potential to disrupt the pathway of vegetation development. Due to planned re-treatment of gamba grass *Andropogon gayanus*[15] infested areas, however, site selection was precluded from large areas of mine rehabilitation. The mine rehabilitation sites that were selected are therefore not considered to be representative with respect to this question. However, *Andropogon gayanus* was sampled in five of the mine rehabilitation sites and *Leucaena leucocephala*[16] was sampled in an additional five sites, which when combined represent 32% of the mine rehabilitation sites.

Reddell and Hopkins (1994) also identified high densities and stand dominance of *Acacia* spp. as having the potential to disrupt ecosystem development, although the level of foliage

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[15] Gamba grass is an invasive, tall, tussock-forming, perennial grass native to tropical Africa. Although it was deliberately introduced to Australia as a pasture grass, it is now considered a serious environmental weed and is a declared plant in Queensland. It produces heavy fuel loads that can cause intense fires that are damaging to native plant communities (Csurhes, 1998).

[16] *Leucaeana leucocephala* is a small tree, native to South America that was introduced to Australia to provide cattle fodder. It has naturalised throughout disturbed areas in coastal Queensland. It forms dense thickets that suppress the regeneration of native plants (Csurhes, 1998).

cover or canopy volume that would be classified as stand dominance was not defined. Comparison of the relative contributions of the main plant genera to the canopy volume shows that generally non-sprouting species (obligate seeders) *Acacia* spp., *Dodonea* spp., *Grevillea* spp., and *Alstonia* spp. dominate the canopy volume in all but the oldest age class of mine rehabilitation (Figure 7.12).

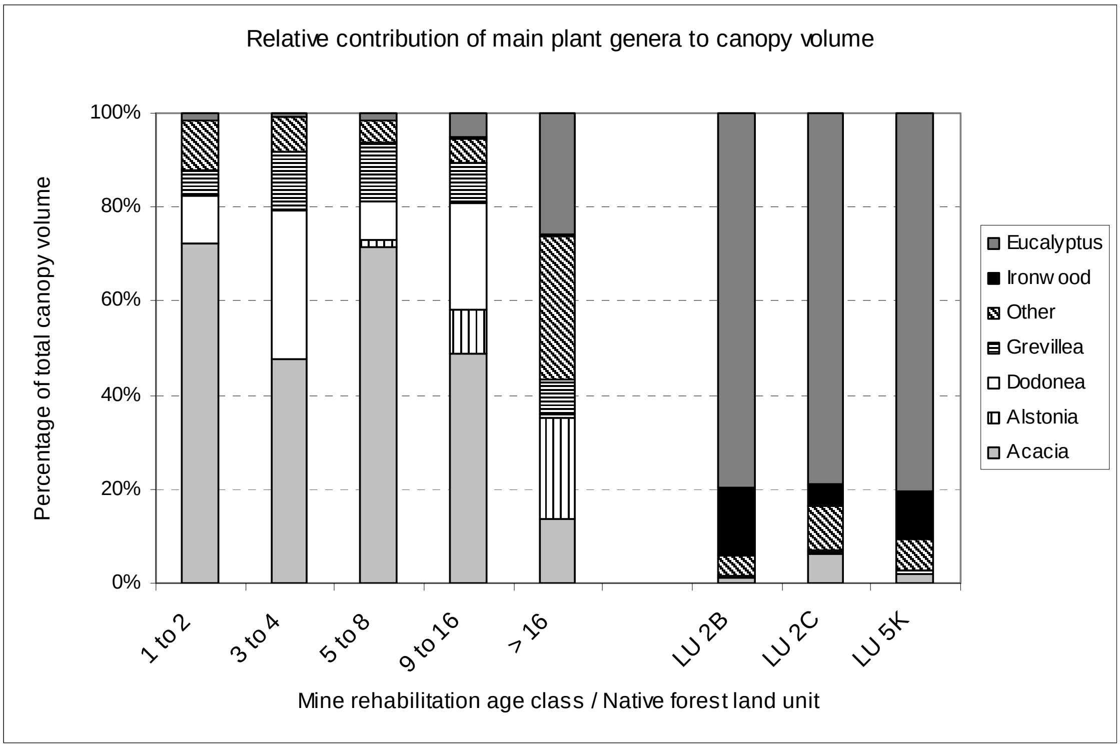

Figure 7.12 Contributions of main plant genera to canopy volume

7.6 Comparison of vegetation structure

Values for vegetation height, canopy volume and basal area all increased with rehabilitation age although there was large variation in values for canopy volume and basal area in the two oldest age classes of mine rehabilitation. Age class accounted for most of the variation in values (Table 7.8). Soil type was not related to variation in any of the vegetation attributes. Fire history, however, accounted for a significant component of the variation in basal area and canopy volume values.

Table 7.8 ANOVA results for summary vegetation values

	Explanatory variables		
Response variables	**Age Class**	**Fire history (Burnt / Unburnt))**	**Soil type (Fresh / Stockpiled)**
	F (4, 24)	F (1, 24)	F (1, 24)
Mean vegetation height √	21.3 ***	0.3 ns	0.4 ns
Total canopy volume √	10.2 ***	4.9 *	0.1 ns
Total basal area √	16.9 ***	4.6 *	0.1 ns

Data used were site values from mine rehabilitation sites only. Vegetation data were square root transformed for analysis to meet assumptions of normality. Significance of the F-statistic is indicated by asterisks * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$;*** = $P \leq 0.001$, ns = not significant.

7.6.1 Vegetation height

The mean height of the tallest vegetation layer increased with rehabilitation age (Figure 7.13). Most of the height increase occurred in the first five years after which the rate of increase slowed. Height reached an asymptote at approximately 15 years. Non-linear regression analysis found that age accounted for 88.4% of variance ($P < 0.001$) in vegetation height values in mine rehabilitation sites. There was little variation in site values. The regression model predicted an upper asymptote for vegetation height in mine rehabilitation sites of 10.6 m, well below the mean observed height of the tallest stratum in all native forest reference land units (Table 7.2). The observed value for mean vegetation height in the oldest age class of mine rehabilitation had already reached the predicted upper asymptote value. Fire history had no effect on vegetation height.

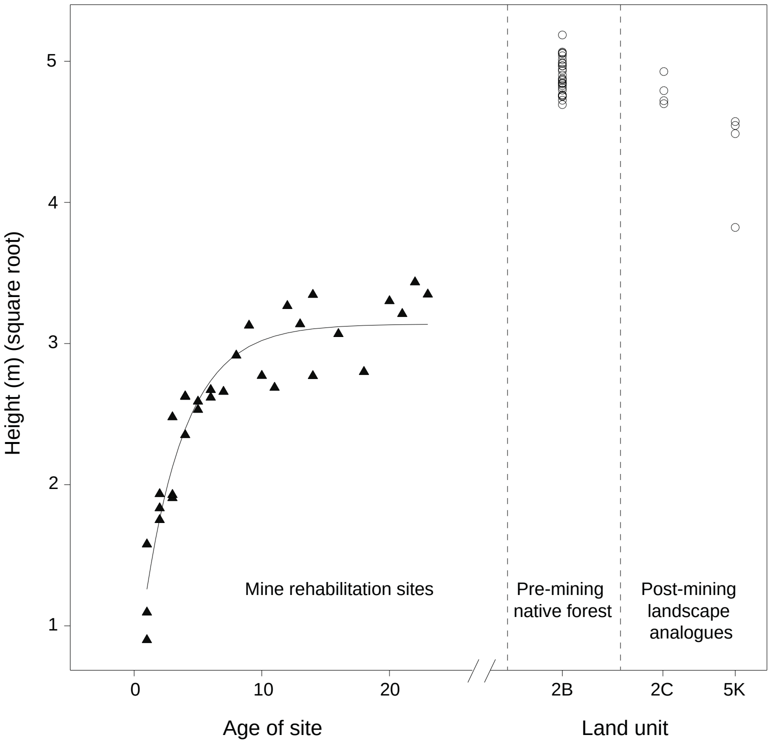

Figure 7.13 Regression of vegetation height against age of site

Non-linear regression of mean site values for vegetation height (m) (data were square root transformed) against age of site using an exponential asymptotic curve. The regression predicted an asymptote for vegetation height in mine rehabilitation sites of 10.6 m.

7.6.2 Canopy volume

Canopy volume values showed a similar pattern of response to height. Non-linear regression found that most of the change in canopy volume occurred in the first five years, after which the rate of growth slowed and reached an asymptote at approximately 15 years (Figure 7.14). Age accounted for 59.9% of the variance ($P < 0.001$) in total canopy volume values in mine rehabilitation sites. The large amount of variation in site canopy volume values around the fitted curve reflects the effect of fire history. Although fire history (burnt versus unburnt) had significant effects on canopy volume, fitting separate regression curves for burnt and unburnt mine rehabilitation sites was not able to predict a meaningful asymptote for burnt sites (Figure 7.14). This may be due to the fact that the fire history category did not account for differences between sites in the length of time that had elapsed between burning and vegetation sampling. The fire history category also did not account for differences in the age of the site at the time of burning. The age of rehabilitation at the time of burning could be expected to result in very different post-fire vegetation responses. Age of rehabilitation may also interact with fire to alter the intensity of the fire event.

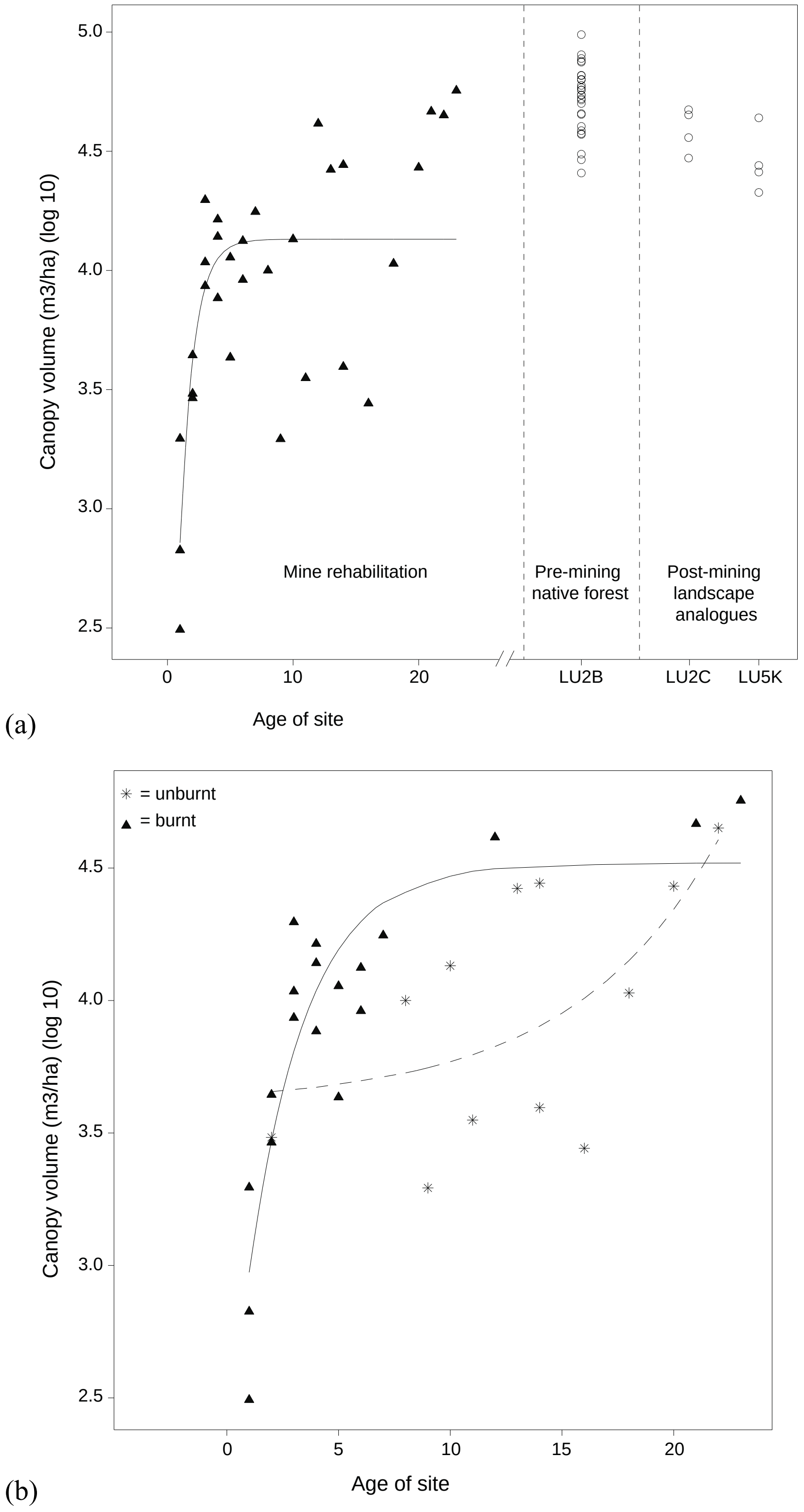

Figure 7.14 Regression of canopy volume against age of site

Non-linear regression of mean site canopy volume (m^3/ha) against site age using an exponential asymptotic curve (data were $\log_{10}$ transformed). (a) When all mine rehabilitation sites were combined the regression predicted an upper asymptote for canopy volume in mine rehabilitation sites of 13,520 m^3/ha. (b) Separate regression of data for burnt and unburnt mine rehabilitation sites predicted an upper asymptote of 33,036 m^3/ha for unburnt sites and was not able to predict a meaningful asymptote for burnt sites.

The regression model predicted an upper asymptote value for canopy volume in unburnt mine rehabilitation sites of 33,036 m^3/ha. This is slightly lower than the observed mean value for the oldest age class of mine rehabilitation and indicates that canopy volume is close to its maximum development in mine rehabilitation. The predicted asymptote and the observed mean value for canopy volume in the oldest age class of mine rehabilitation is comparable to the observed mean total canopy volume values for the two post-mining landscape analogue land units, 5K and 2C, but much lower than the pre-mining native forest reference LU 2B (Table 7.2).

7.6.3 Basal area

The pattern of response for basal area was similar but slightly delayed compared with both height and canopy volume. This may reflect the sequence with which different parts of the perennial woody vegetation develop. Non-linear regression found that age accounted for 83.4% of variance in basal area in mine rehabilitation sites ($P < 0.001$). Most of the change occurred in the first 8 years after which the rate of expansion slowed and reached an asymptote at approximately 20 years (Figure 7.15). There was less variation in values around the fitted curve than there was for canopy volume although fire history still accounted for a significant component of variation (Table 7.8).

Separate regression models predicted very different upper asymptote values for basal area in burnt and unburnt sites. The predicted upper asymptote for unburnt mine rehabilitation sites was 15.8 m^2/ha which exceeds the observed mean basal area of all native forest reference land units (Table 7.2). The predicted upper asymptote for burnt sites, however, is 5.7 m^2/ha, well below the mean observed basal area of all native forest reference land units. The observed mean total basal area in the oldest age class of mine rehabilitation is equal to that in the post-mining landscape analogue land unit 5K[17] but lower than the two other native forest reference land units.

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[17] Similar stand values for basal area in m^2/ha do not mean that biomass values are the same. First, there are interspecific differences in the allometric relationships between basal area and biomass. Second, most biomass is in big trees. The high value for basal area m^2/ha in the oldest age class of mine rehabilitation is achieved through high stem densities of small trees which do not have the same biomass for basal area as big trees. For example, a single *E. obliqua* tree with a stem diameter of 202 cm (dbh) has a basal area of 32,127 cm^2 and stem biomass of 20,800 kg. A stand of 409 trees of *E. obliqua* with stem diameters of 20 cm (dbh) has the same basal area (32,127 cm^2) but a much smaller stem biomass of 7,550 kg (H. Keith, 2009, personal communication).

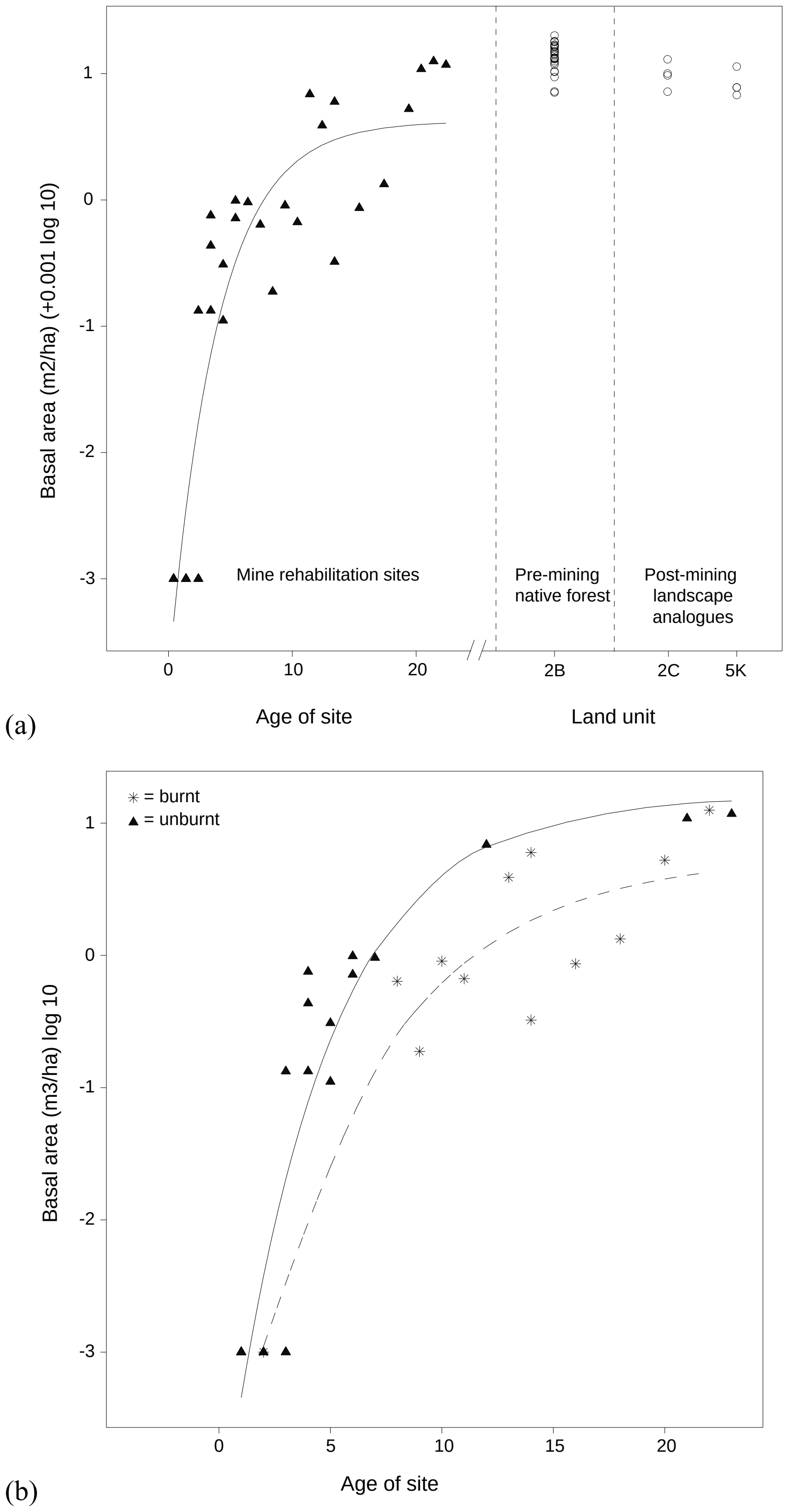

Figure 7.15 Regression of basal area against age of site

Non-linear regression of mean site basal area (m^2/ha) against site age using an exponential asymptotic curve (0.001 was added to all values then data were $\log_{10}$ transformed). (a) When all mine rehabilitation sites were combined, the regression predicted an upper asymptote for basal area in mine rehabilitation sites of 4.2 m^2/ha. (b) Separate regressions for burnt and unburnt mine rehabilitation sites predicted an upper asymptote of 15.8 m^2/ha for basal area for unburnt sites and 5.7 m^2/ha for burnt sites.

7.6.4 Vegetation layering

In the oldest age class of mine rehabilitation, the observed mean total basal area was equal to the mean for LU 5K sites. The observed mean total canopy volume in the oldest age class of mine rehabilitation was comparable to the two post-mining landscape analogue land units, 2C and 5K (Table 7.2). Although these total values approached values of the native forest reference land units, the structure of the vegetation layers was very different (Figure 7.16). Vegetation in all native forest reference land units was taller, had greater canopy depth and was more open. Although the cross-sectional area of canopy (mean percent foliage cover) in the oldest age class of mine rehabilitation was the same as in pre-mining native forest reference LU 2B, the canopy volume in LU 2B was much higher because of the greater depth of the vegetation canopy (Table 7.2).

No big tree layer had developed in the oldest age class of mine rehabilitation sites (Figure 7.16). In contrast to the native forest reference land units, most of canopy volume (77%), and all of the basal area in mine rehabilitation sites was in the small tree layer. The small tree layer in mine rehabilitation sites had twice as many stems per hectare, almost twice the basal area and double the percent foliage cover compared to the small tree layer in any of the native forest reference land units (Table 7.1). Apart from higher stem densities of small trees, differences in structure were also attributable to differences in plant species composition. Species with large leaves or dense canopies were major components of the small tree layer in mine rehabilitation sites including *Alstonia scholaris*, *Acacia mangium*, *Alphitonia excelsa* and *Parinari nonda* (Appendix 23).

The basal area and canopy volume of perennial grasses declined with rehabilitation age. Perennial grass cover was much lower in all mine rehabilitation sites than in the reference native forest land units (Figure 7.16). The highest value for basal area of perennial grasses in mine rehabilitation sites, which was only 8% of the mean value for the pre-mining native forest reference LU 2B, was in the three to four year old age class.

If a layer is defined as being present when it has more than 5% foliage cover (my definition), then native forest reference land units were all at least 4-layered, whereas mine rehabilitation sites were no more than 2-layered. Using this definition, LU 2B lacks a tall shrub layer as it has only 3.3% foliage cover. Likewise, perennial grasses and low shrubs do not exceed a mean of 5% foliage cover in any of the age classes of mine rehabilitation although some individual sites may have exceeded this value.

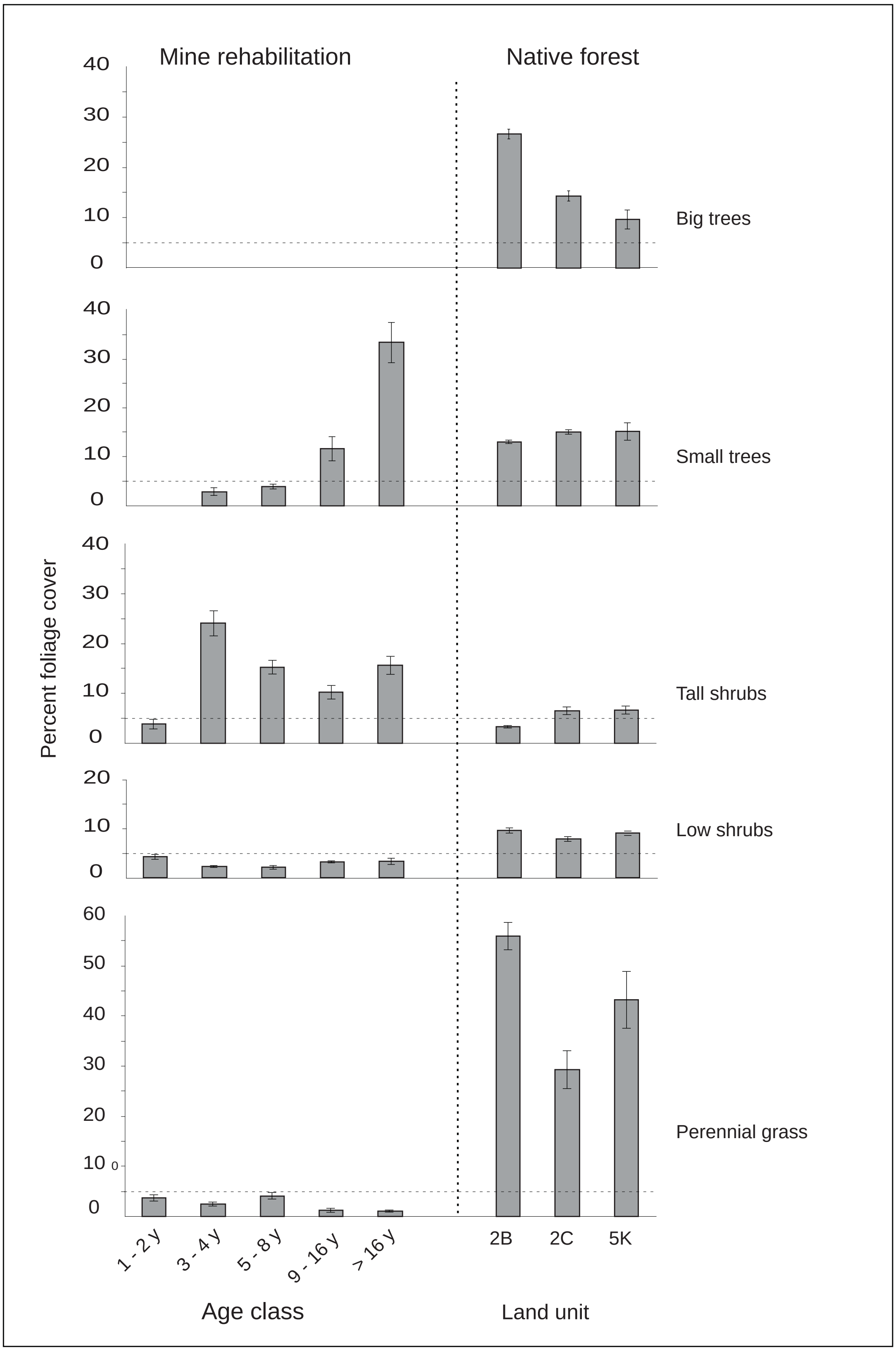

Figure 7.16 Percent foliage cover of vegetation layers by land unit

Mean percent foliage cover of each vegetation layer by age class of mine rehabilitation and native forest reference land unit. A dotted line is drawn at 5% foliage cover which is arbitrarily used here to define whether a layer is present or not. Error bars show standard error of the mean.

7.6.5 Confounding of variables

There was confounding of variables in mine rehabilitation sites aged 9 to 16 years. All six of the sites that had been regenerated using stockpiled material were in the same age class (9 to 16 years old). This was an unavoidable consequence of there having been a number of years during which no fresh topsoil was used in any of the mine rehabilitation. This may have contributed to the finding that soil type was not a significant factor in determining landscape function index values.

Fire history was not confounded with age class. Fourteen of the mine rehabilitation sites had been burnt covering all age classes except 3 to 4 year old sites. All but one of the sites in the 9 to 16 year age class had been burnt and some sites had been burnt more than once. The combined effects of fire history and stockpiled topsoil on this age class of mine rehabilitation probably account for the larger variability in landscape function indices, canopy volume and basal area in 9 to 16 year old sites compared to the other age classes.

7.7 Summary of vegetation and landscape function results

All landscape function values increased with rehabilitation age and acquired values within the range of reference values for all native forest reference land units. The observed mean values in the oldest age class of mine rehabilitation sites were slightly higher than the observed mean values for infiltration and nutrient cycling and slightly lower than the observed mean value for stability across all three native forest reference land units.

Vegetation composition in rehabilitating mine sites was significantly different from all native forest reference land units. The vegetation in mine rehabilitation included many species that are not part of the pre-mining vegetation community and included native species that are not part of the regional flora, and exotic species. Only 34% of perennial grass species and 40% of woody species recorded were found in both native forest and mine rehabilitation although at very different densities. The vegetation canopy in mine rehabilitation was dominated by species that are not part of the vegetation in native forest reference land units. Framework species occurred at significantly lower stem densities in mine rehabilitation sites than in all native forest reference land units. The relative contributions of framework and desirable species were also substantially lower in all age classes of mine rehabilitation than in all native forest reference land units.

It was not possible to accurately quantify the contribution of destabilising species as the mine rehabilitation sites selected were not representative with respect to this question. However, the vegetation canopy volume in mine rehabilitation sites was dominated by species that may contribute to environmental instability. Foliage cover in all but the oldest age class of mine rehabilitation sites was dominated by non-sprouters (obligate seeder species) including *Acacia* spp., *Dodonea* spp., *Grevillea* spp. and *Alstonia* spp. Aggressive weedy species were sampled in 32% of the mine rehabilitation sites.

Perennial woody vegetation in rehabilitating mine sites was in a phase of rapid growth in the first 5 to 8 years followed by a slower phase of consolidation and expansion. Vegetation height and canopy volume reached asymptotes after approximately 15 years. Height reached an asymptote at a value much lower than any of the native forest reference land units. Mean total canopy volume reached an asymptote between the values for the two post-mining landscape analogue land units, 2C and 5K. Although the observed values for mean total basal area[18] and mean total canopy volume in the oldest age class of mine rehabilitation were similar to the post-mining landscape analogue LU 5K, there were significant differences in how the vegetation was distributed into vegetation layers. Most notably, the complete absence of a big tree layer and virtual absence of perennial grass and low shrub layers from all age classes of mine rehabilitation. Conversely, the tall shrub layer and small tree layer had much higher levels of foliage cover in mine rehabilitation than in native forest reference land units.

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[18] Species differ in the allometric relationships between basal area and biomass. The amount of biomass is related to the stem diameter but most biomass is stored in big trees. Because of differences in species composition between mine rehabilitation and native forest sites and differences in the size class distribution of stems (native forest sites had a big tree layer while mine rehabilitation sites did not), similar values for mean basal m^2/ha area do not translate into similar values for total biomass. See also footnote 15 on page 185.

Chapter 8: Discussion of results

8.1 Introduction

As characterised by the habitat templet theory, succession represents the movement of a particular location along a gradient of durational stability (Southwood, 1977). Durational stability is a measure of the rate at which a habitat changes in relation to the needs of an organism (Southwood, 1988). As the durational stability of a habitat increases, biomass and spatial complexity also increase resulting in an increase in the variety of niches (Brown, 1991; Southwood, 1977). These changes could be expected to result in a turnover in the type of habitat resources, and increases in the variety, amount and stability of habitat resources. Using the habitat templet as a theoretical framework, a number of related changes are expected to occur in habitat resources and bird assemblages along successional gradients. The first section of this chapter discusses the results of this study in relation to the expectations that arise from the habitat templet theory and compares the results of this study with those of other studies.

A number of factors influence the effectiveness of rehabilitation as habitat. Site vegetation characteristics and landscape context have been identified as two key factors that can influence the effectiveness of mine rehabilitation as habitat for birds. The second section of this chapter discusses how the bird assemblages in native forest and mine rehabilitation were related to vegetation and landscape variables and compares the findings of this study with the results of other studies. The question of whether post-mining rehabilitation on the Weipa bauxite plateau is likely to restore habitat similar to that of pre-mining native forest is then addressed with reference to the predictions of successional theory and to the findings of other studies on rehabilitation. This chapter concludes with a discussion of the consequences of bird-habitat relationships in the context of mining and post-mining rehabilitation on the Weipa bauxite plateau and rehabilitation generally.

Although succession is not limited to changing species composition, the term succession is often used to refer to a change in plant species composition. The term succession is used here to refer to changing vegetation structure, with or without change in plant taxonomic composition.

8.2 Temporal patterns in bird assemblages

A number of characteristics of faunal assemblages are expected to change with increasing durational stability. These changes occur because of the relationships between faunal species' life history characteristics with plant biomass and variety of niches. As the permanency of habitat structures increases, niche breadth is expected to narrow. The small, mobile, opportunistic, short-lived species associated with early successional habitats of short duration are expected to be progressively replaced by larger, more sedentary, specialised, and long-lived species (Southwood, 1977). As a consequence of the expected changes in the life history characteristics of component fauna species, an increase in abundance and species richness is also expected. Turnover in species composition is also expected leading to changes in the structure of faunal assemblages (May, 1982). The rate of faunal species turnover is expected to be related to the rate of vegetation change.

8.2.1 Bird species richness and abundance

Bird species richness in post-mining rehabilitation on the Weipa bauxite plateau increased with rehabilitation age. The increase in species richness was associated with changes in bird community structure that were most evident at the level of foraging group. As May (1982) found, bird species richness increased with rehabilitation age as a result of two processes: (1) addition of new foraging groups; and (2) expansion of existing foraging groups. The results of this study also support May's finding that the increase in the number of foraging groups is greatest early in succession (May, 1982).

The increase in bird species richness in mine rehabilitation at Weipa was initially due to the addition of foraging groups associated with an increase in vegetation height, canopy volume and the addition of a woody vegetation layer. From the youngest mine rehabilitation age class (1 to 2 years old) to the next age class (3 to 4 years old), mean vegetation height doubled from 2 to 4 m, canopy volume tripled and the number of woody vegetation layers increased from two to three with the addition of a small tree layer. Associated with these vegetation changes, four foraging groups were added and seven foraging groups expanded. The four foraging groups that were added were ground foragers, aerial insectivores, hawking insectivores, and pouncing predators. Subsequent to this, no more foraging groups were added and further increases in species richness occurred due to expansion and species turnover within the existing foraging groups. The rate of change in bird assemblages in mine rehabilitation closely paralleled changes in vegetation structure, with vegetation height the strongest predictor of bird species richness. These findings confirm the importance of the amount and variety of habitat resources to species richness patterns.

The pattern of change in mine rehabilitation at Weipa, however, was not one of simple accumulation of species. As vegetation changed, there was turnover in bird species composition and foraging group structure changed. The species richness of four foraging groups was highest in mine rehabilitation sites. The species richness of insectivore-nectarivores and grass and ground feeding insectivores was highest in 5 to 8 year old mine rehabilitation sites. Granivore species richness was highest in 9 to 16 year old mine rehabilitation sites. Pouncing predator species richness was highest in the oldest age class of mine rehabilitation, which was 17 to 23 years old.

Vegetation in native forest sites had greater biomass and more woody vegetation layers than mine rehabilitation. The greater variety and amount of habitat resources associated with this vegetation allowed for more foraging groups and a higher species richness of seven foraging groups. The species richness of carnivores, frugivores, tree-seed eaters, hawking insectivores, trunk gleaning insectivores, foliage-gleaning insectivores and aerial insectivores was highest in native forest sites.

Bird abundance in mine rehabilitation at Weipa increased consistently with increasing rehabilitation age. However, mean bird abundance in the oldest age class of mine rehabilitation remained significantly lower than in all of the native forest reference land units. The finding of high bird species richness and low mean bird abundance may be due to the opportunistic use of resources in mine rehabilitation by native forest birds. Although species richness generally increases with the number of individuals sampled and a log-normal distribution of species abundance is expected (Gotelli and Colwell, 2001), such a relationship is not universal. Modelling that accounts for differences in immigration rates has demonstrated different species abundance distributions (Bell, 2000; Hubbell, 2001). Opportunistic use of resources in mine rehabilitation by native forest birds, leading to high immigration (and emigration) rates, might be expected because of seasonally high densities of flowering shrubs in mine rehabilitation. Thus, although numerical species richness values in the two oldest age classes of mine rehabilitation and native forest land units are similar, they appear to differ in the ratios of resident to opportunistic species.

The opportunistic visitation hypothesis is supported by an examination of the site detection rates of individual bird species. Forty percent of the species that occurred in both native forest and mine rehabilitation sites had higher mean detection rates in native forest sites. The site detection rates of all of these species also declined as distance from remnant vegetation increased. Only 10% of shared species had higher mean detection rates in mine rehabilitation sites. Differences between the remaining 50% (28/55) of shared species were not significant due to generally low abundances.

Furthermore, with the exception of the 10% of shared species that had higher mean detection rates in mine rehabilitation sites, all of the shared bird species are known to be dependent on mature *Eucalyptus* trees for many of their habitat resources (Higgins *et al.*, 2006; Higgins *et al.*, 2001; Higgins, 1999; Higgins and Peter, 2002). It is concluded, therefore, that most of the shared bird species depend on native forest for the majority of their habitat resources[19] and use resources in mine rehabilitation sites opportunistically. In spite of high numerical species richness values, therefore, the bird assemblages in mine rehabilitation appear to be made up of a high proportion of opportunistic visitors.

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[19] The term habitat resources, includes all of a species resource requirements including food resources, protection (shelter) and breeding resources.

8.2.2 Rate of change

Although the pattern of increasing bird species richness and bird abundance observed in this study was consistent with other studies, the rate of increase, was slower than rates reported from post bauxite mining sites elsewhere. In Western Australia, bird species richness in mine rehabilitation sites is similar to unmined Jarrah forest after 8 years (Nichols and Nichols, 2003; Nichols and Grant, 2007). Some post-mining rehabilitation sites as young as 5 years support similar bird species richness and abundance as unmined Jarrah forest (Nichols and Watkins, 1984). The rate of increase in species richness and abundance in mine rehabilitation at Gove is slower and only reaches similar values to native forest control sites after approximately 19 years (Brady, 2005). As the rate of change in bird species richness is closely related to vegetation development, particularly vegetation height, differences in the rate and magnitude of change in bird assemblages between this study and other post-mining rehabilitation sites probably reflect differences in rehabilitation strategies and vegetation development between the mining operations.

8.2.3 Succession and 'specialisation'

An increase in bird specialisation along successional gradients has been reported in a few studies of birds in secondary forest succession (May, 1982; Venier and Pearce, 2005). Conversely, reduced woody vegetation cover in response to fire in a North American oak savanna resulted in a decrease in bird specialisation due to the decline in insectivores of the upper canopy (Davis *et al.*, 2000). The usefulness of the term specialisation, however, is questionable. Specialisation is a relative concept and an increase in the number of specialists does not mean that specialisation *per se* increases (Helle and Monkkonen, 1990). There are two alternative, complementary explanations for increasing specialisation. In the first, specialisation results from niche diversification (i.e., an increase in the degree of partitioning of a given amount of habitat resources). In the second, specialisation is related to the amount and variety of habitat resources.

If it can be assumed that the stem density of big trees and vegetation height are surrogates for the amount and variety of habitat resources respectively, then bird species richness and the number of bird specialists in mine rehabilitation and native forest at Weipa was related to the amount and variety of habitat resources. The species richness of seven foraging groups was highest in native forest reference sites. The abundance of most foraging groups increased with rehabilitation age and was also highest in native forest sites. Of these, the abundances of hawking insectivores, trunk-gleaning insectivores, frugivores and pouncing predators were all strongly related to vegetation height and the stem density of big trees. The amount and variety of specific habitat resources used by these foraging groups, including perches, canopy gaps, trunks and large branches, tree hollows, arboreal termitaria, and mistletoe might be expected to increase with increasing vegetation maturity and density of big trees. The form of the relationships between specific structural attributes such as canopy gaps and the number of big trees no doubt differs from one vegetation type to another. However, the number of big trees has been found to be an important structural attribute of vegetation for characterising wildlife habitat (McElhinny *et al.*, 2005). It has been found to be significantly correlated with occupancy by hollow dependent fauna. The density of big trees is specifically associated with the presence or abundance of a number of vertebrate faunal groups including birds in Australian *Eucalyptus* formations (McElhinny *et al.*, 2006).

Tree-seed eaters also used habitat resources associated with mature vegetation. The richness and abundance of tree-seed eaters also increased with rehabilitation age and were significantly higher in native forest. However, their site detection rates were not strongly related to vegetation height or the stem density of big trees. This finding is probably due to the presence of seed and fruit bearing trees in the older age classes of mine rehabilitation sites. Pale-headed rosellas were observed feeding on flowers and fruit of *Grevillea pteridifolia* and sulphur-crested cockatoos were observed feeding on fruit of *Alphitonia excelsa* in mine rehabilitation.

The absence of some specialist foraging groups from mine rehabilitation is interpreted as being due to the vegetation lacking the specific type of habitat resources that they require. Hawking insectivores and pouncing predators were absent from the youngest age class of mine rehabilitation, presumably due to a lack of suitable perches and foraging space. The entire specialist foraging group of trunk-gleaning insectivores, including brown treecreeper, grey-crowned babbler and varied sittella were only found in native forest sites. In addition to being foraging specialists, these three are sedentary species and co-operative breeders. A number of Australian studies of birds along successional gradients have also found that specialist trunk-gleaning insectivores only occurred in mature or remnant *Eucalyptus* formations (Fisher, 2001; Martin *et al.*, 2004). Loyn (1985) found that bark-gleaning species were still scarce or absent after 10 years in a secondary forest succession following logging. The trunk-gleaning rufous treecreeper *Climacteris rufa* and varied sittella were two of 12 species that had still not been recorded in Alcoa's mine rehabilitation in Western Australia 17 years after rehabilitation (Armstrong and Nichols, 2000).

Not all specialists, however, were associated with increasing vegetation maturity. Some foraging groups specialise in the ephemeral habitats that are associated with early successional environments. In this study, the species richness and the abundance of grass and ground foraging insectivores, granivores, and other ground foragers were highest in young mine rehabilitation sites. Their abundances were negatively related to surrogates for amount and variety of habitat resources (i.e., stem density of big trees and vegetation height). Appropriate vegetation structural surrogates for amount and variety of habitat resources apparently differ with the specific type of habitat resources and from one foraging group to another.

With some qualification, therefore, the results of this study indicate that specialisation does increase through succession, depending on how specialist is defined. However there was not a simple increase in the number of specialists. Based on the results of this study, it would be more accurate to say that the number and richness of co-existing eucalypt foraging groups increased through succession associated with increasing amount and variety of eucalypt based habitat resources.

Knowing what resources or habitats a bird specialises in is important for understanding their long term conservation needs. However, some of the resources or habitats that birds specialise in are ephemeral or are only important at particular life stages. This study found that even for bird species that are resource specialists with respect to one or more particular resources, not all of their resource requirements were habitat-specific. For example, two species of insectivore-nectarivore that are generalists in their foraging behaviour, little friarbird and banded honeyeater, were strongly related to mature framework trees but occurred in mine rehabilitation sites as young as 2 years old. Little friarbirds need stringy bark for roosting and breeding. They weave strips of bark to create suspended nests often high in the outer branches of trees. They also roost in trees. However, although they mostly forage on flowers and foliage in the tree canopy they also forage on shrubs (Higgins *et al.*, 2001), hence, their presence at low detection rates in young mine rehabilitation sites. Banded honeyeaters, too, forage mainly in the tree canopy but also in shrubs. They also construct nests made of strips of bark in the outer foliage of trees (Higgins *et al.*, 2001). This emphasises the need to differentiate between habitat specialists and resource specialists, and to specify in what respects a species is a generalist and in what respects it is a specialist.

8.2.4 Bird community structure

In addition to the temporal axis of durational stability, the habitat templet theory characterises habitats in terms of their spatial favourability. Depending on the particular resource, a habitat consists of favourable patches and unfavourable inter-patches (Southwood, 1977). The favourableness of a habitat (i.e., the density and spatial arrangement of particular resources), determines the carrying capacity and individual abundance. The productivity and spatial arrangement of resources then, should be related to the abundance of individuals that use those particular resources and therefore to community structure.

The findings of this study suggest that the way in which habitat is related to foraging group structure is complex and involves a set of conditional relationships. To begin with, the taxonomic composition of the vegetation determines the presence of specific **types** of habitat resources and their phenology. The structural complexity of vegetation, which is in turn determined by plant taxonomic composition, disturbance history and maturity, determine the **variety** of habitat resources. Plant productivity is also related to plant taxonomic composition and vegetation structure. Together these three factors: plant species composition, structural complexity of vegetation and plant productivity determine the specific types, the variety of different types and the overall **amount** of habitat resources. These three factors are related to bird foraging group structure through their combined effects on the distribution and abundance of individual species. Bailey *et al*., (2004) found that the number of bird foraging groups was related to both plant productivity and vegetation complexity. Hurlbert (2004) found that species richness increased with productivity as a result of two related processes, higher numbers of individuals and changes in species abundance distributions that were associated with increasing habitat complexity.

Finally, through the mechanism of natural selection, individual bird species become uniquely matched to their habitats through trade-offs that optimise fitness in relation to different combinations of durational stability, productivity, disturbance and environmental conditions. As long as a minimum amount of a particular food resource is consistently available, mobility can be 'traded off' against an increase in foraging efficiency at exploiting that resource. The more specialised a species becomes in its foraging requirements, the more tightly matched it is to a particular habitat type and presumably the more sedentary it becomes. Various combinations of organism size, prey size, metabolic requirements, flight efficiency, habitat productivity and spatial and temporal distribution of resources could result in a range of outcomes that optimise fitness in relation to specific resources. Species that are less specialised in their foraging requirements retain mobility and access to a range of habitat types and are therefore less habitat-specific. The relationship between habitat type and bird foraging group structure then, appears to be highly conditional.

In this study, foraging group abundance was significantly related to vegetation structure. Grass and ground foraging insectivores were most abundant in 1 to 2 year old mine rehabilitation sites and then declined with age. Ground foragers were absent from the youngest age class, most abundant in 3 to 4 year old mine rehabilitation sites and then declined in abundance with rehabilitation age. Granivores were present in all age classes, increased consistently to a peak in 9 to 16 year old mine rehabilitation sites and then declined to their lowest abundance in native forest sites. With the exception of omnivores, for which no clear pattern emerged, and trunk-gleaning insectivores, which only occurred in native forest, the abundance of all other foraging groups increased with rehabilitation age. The abundance of ten out of 14 foraging groups was highest in native forest sites, which had the greatest biomass and the most well developed vegetation layering.

Native forest bird assemblages were dominated by three groups that forage on foliage and flowers, foliage-gleaning insectivores, insectivore-nectarivores, and nectarivores. The species richness of foliage gleaning insectivores was also highest in native forest. The high abundance and species richness of foliage-gleaning insectivores in native forest sites may simply reflect the greater volume of foliage (i.e., amount of substrate for habitat resources) available there. Alternatively, the increased richness of foliage-gleaning insectivores in native forest sites may reflect the degree of specialisation that this group has reached in *Eucalyptus* formations where there is relative stability in the season-to-season and year-to-year availability of foliage and its invertebrates (i.e., there is high **stability** of foliage based habitat resources).

Trunk-gleaning insectivores were absent from all age classes of mine rehabilitation. Compared to the oldest age class of mine rehabilitation, pouncing predators were six times more abundant in native forest and hawking insectivores were three times more abundant. Foliage-gleaning insectivores, nectarivores and frugivores were all twice as abundant in native forest than in the oldest age class of mine rehabilitation. The presence of these foraging groups is related firstly to the availability of specific **types** of habitat resources that are associated with particular vegetation composition. That their abundance increases with vegetation biomass indicates that the **amount** of specific habitat resources is also important.

The finding of this and other studies that certain species or foraging groups only occur in mature vegetation has further implications. For more specialised species, in addition to the availability of specific types of resources, the density at which those resources occur and their spatial configuration is important. Presumably the fitness of these specialists is optimised by particular spacing of habitat resources. In this study, site detection rates of four sedentary insectivores were related to the basal area of big trees and canopy volume of perennial grass but negatively related to small trees, indicating that within *Eucalyptus tetrodonta* open forest, particular spatial arrangements of habitat resources were important.

If the amount of specific habitat resources is important, however, there is an apparent paradox in the comparatively low density of perennial grasses found in mine rehabilitation sites and the high abundance of granivores and grass and ground foraging insectivores in these sites. The habitat templ*et al*so predicts that bird distribution and abundance is related to resource minima. Brady (2005) quantified bird abundance in relation to the abundance of nectar, insects and grass seeds in post bauxite mining rehabilitation at Gove. He found that bird distributions were not simply related to maxima of resource availability. Habitat structure and the minimum resource availability were both involved. Ziembicki *et al*., (2007) found that Australian bustard *Ardeotis australis* abundance was positively associated with productivity minima. An assessment of dispersive bird movements in relation to variation in productivity, found that the species richness of **resident** birds was related to productivity minima (Bailey *et al*., 2004). At Gove, Brady (2005) concluded that habitat structure was more important in determining the composition of the bird community than was the availability of specific food resources. It appears, therefore, that a hierarchy of factors is involved when birds select habitat. Depending on the degree of habitat-specificity, habitat type (defined by plant species composition, relative species abundance, biomass, vertical structural complexity and horizontal spatial order), appears to be the most important factor.

A hierarchy of factors in habitat selection provides a plausible explanation for the high abundance of grass and ground foraging insectivores in mine rehabilitation sites at Weipa. As long as a minimum amount of the specific type of food resources is available, then attributes of habitat structure determine habitat selection. The low, open grassy habitat structure of young mine rehabilitation sites with grass may be associated with lower predation risk from pouncing predators than grassy forested sites. Alternatively, protection of mine rehabilitation sites at Weipa from fire may make the grass cover in mine rehabilitation sites preferable to that in native forest. Golden-headed cisticola, which was the most abundant of the grass foraging insectivores in this study is disadvantaged by fire (Woinarski and Recher, 1997).

It may be that different hierarchies apply to different bird species depending on how sedentary and specialised they are. For species that are highly specialised, habitat selection may involve a spatially nested hierarchy of factors with habitat type (incorporating the type of habitat resources and amount of habitat resources that are consistently available), the most important criterion. In contrast, for generalist species, hierarchies are likely to be non-spatially nested and productivity maxima may be the most important criterion determining habitat selection.

8.3 Bird assemblages, vegetation and landscape

8.3.1 Composition of bird assemblages

It has been well established that the composition of bird assemblages changes with vegetation succession. A number of studies have identified specific bird assemblages, or indicator species associated with different vegetation ages or vegetation states in different types of successional environments. Age-specific bird assemblages have been found in: (1) natural secondary forest succession following logging or clearing (Laiolo *et al.*, 2004; Smith, 1985; Taylor *et al.*, 1997; Sallabanks *et al.*, 2006; Loyn, 1985a; Venier and Pearce, 2005; Fisher, 2001; May, 1982); (2) habitat reconstruction following agriculture (Martin *et al.*, 2004); (3) plantation forestry (Woinarski, 1979); and (4) primary succession in habitat reconstruction following mining (Nichols and Nichols, 2003; Karr, 1968; Nichols and Grant, 2007; Nichols and Watkins, 1984; Armstrong and Nichols, 2000). The vegetation stage at which bird species enter the succession is usually related to the development of key habitat resources such as food plants, tree hollows, mature eucalypts, and leaf litter (Smith, 1985; Loyn, 1985a). That vegetation structure has a role in determining the composition of bird assemblages is evident from these studies.

Three patterns emerged from changing composition of the bird assemblage with increasing rehabilitation age on the Weipa bauxite plateau. First, there was turnover of species generally associated with open, grassy or heathy habitats in the four youngest age classes of mine rehabilitation. Second, bird species generally associated with mesic habitats accumulated with increasing rehabilitation age. Third, native forest bird species accumulated with increasing rehabilitation age. Consequently, the bird assemblage in mine rehabilitation became increasing similar to the native forest bird assemblages with increasing rehabilitation age. The composition of the bird assemblage in the oldest age class of mine rehabilitation at Weipa, however, remained significantly different from the pre-mining native forest reference LU 2B, and both of the post-mining landscape analogue land units, 2C and 5K.

A number of studies of birds in post-mining rehabilitation have also found that, although the composition of bird assemblages changes through succession, significant differences remain between old rehabilitation sites and reference sites. Winter and Alford (1999) found that the composition of bird assemblages in mine rehabilitation at Weipa changes over time but bird assemblages in the older age classes of mine rehabilitation remain significantly different from Darwin Stringybark woodland (Winter and Alford, 1999). In a similar environment at nearby Gove, (across the Gulf of Carpentaria), bird species composition changes with rehabilitation age but in the oldest age class of mine rehabilitation is still significantly different from that found in native forest control sites (Brady, 2005). In a completely different environment in the far away Amazon, the abundance and diversity of birds and mammals in rehabilitation following bauxite mining remains low after 10 years and is characteristic of fairly young secondary forests (Parrotta *et al.*, 1997).

The results of the Weipa, Gove and Amazon post bauxite mining studies contrast with findings from Alcoa's bauxite mine in Western Australia, where Nichols *et al.*, (2003) concluded that bird communities in post-mining rehabilitation are very similar to those in unmined forest sites after only eight years (Nichols and Nichols, 2003). Eleven years after mine rehabilitation in Western Australia, bird species composition is indistinguishable from that of unmined forest in stream zones, which contain tall dense understorey (Nichols and Grant, 2007). In this case the reference Jarrah forests are not un-impacted as they have been previously logged and are locally affected by dieback. Furthermore, bird species composition does not indicate that species have colonised mine rehabilitation (Smallwood, 2001). Without demonstration that breeding populations have established, all that can be said of the species whose presence has been recorded in mine rehabilitation is that they are utilising some aspect of the habitat resources provided in mine rehabilitation. If newly created habitat in mine rehabilitation does not replace the ecological values of the native forest, mine rehabilitation may even act as a sink for dispersing individuals (Keagy *et al.*, 2005).

This study did not include natural mesic vegetation land units on the Weipa bauxite plateau, such as riparian zones and associated ecotones. It is therefore not possible to quantify the degree of similarity between bird assemblages in mine rehabilitation sites and mesic habitats of the Weipa bauxite plateau. However, it is possible that bird assemblages in older mine rehabilitation sites at Weipa are more similar to those of nearby mesic habitats than they are to those of *Eucalyptus tetrodonta* open forest. Bird assemblages in the older age classes of mine rehabilitation at Weipa were characterised by a mix of native forest birds and species that are known to prefer dense or mesic vegetation, including olive-backed sunbird, brown honeyeater, yellow-spotted honeyeater, graceful honeyeater, and dusky honeyeater (Higgins *et al.*, 2001; Higgins *et al.*, 2006). Other mesic species that were strongly related to older mine rehabilitation sites were white-streaked honeyeater, black butcherbird, varied triller, great bowerbird, brown-backed honeyeater and spangled drongo. Thomas (2004) compared habitat preferences of vertebrate fauna in the Weipa area and found distinct differences in bird assemblages between woodlands, ecotones and riparian zones. Fifty percent of the species that were recorded exclusively in mine rehabilitation in this study were found by Thomas (2004) to occur only in ecotone or riparian vegetation in the Weipa area.

8.3.2 Habitat-specific bird assemblages

In this study, vegetation height was statistically, the single most important vegetation structure variable related to overall bird species richness. The strong height gradient from the youngest mine rehabilitation sites to native forest sites accounted for the strength of the relationship between site species richness values and mean vegetation height. Vegetation height, however, is evidently not the only determinant of species richness. In the oldest age class of mine rehabilitation bird species richness reached similar values to those of native forest sites even though mean vegetation height was only half that in native forest reference land units.

A number of factors contribute to this finding. First, vegetation height integrates other information about vegetation structure. Height can be thought of as a surrogate for habitat structural complexity (i.e., variety of habitat resources). Mean vegetation height was strongly correlated with a number of vegetation structure variables that other studies have been found to be also related to bird species richness, including tree density (Ford and Bell, 1982), canopy volume (Gilmore, 1985), and tree basal area (Braithwaite *et al.*, 1989; Recher *et al.*, 1991). Height may also be a surrogate for productivity and temporal stability of habitat resources associated with insects, providing birds some indication of the likely suitability of future as well as current site conditions.

An increase in height however, does not simply add structure. It is associated with a change in how vegetation is structured. Venier and Pearce (2005), for example, found that although the total amount of vertical vegetation structure increased with age there was a concurrent decrease in cover in the lower strata. This change was reflected in the composition of bird assemblages (Venier and Pearce, 2005). The form of the relationship between vegetation height and structure also changes with vegetation composition. Mean vegetation height in native forest sites at Weipa was related to the canopy volume of big trees, mostly *Eucalyptus* species.

In contrast, mean vegetation height in mine rehabilitation sites was related to the canopy volume of small trees, mostly *Alstonia spectabilis* and *Acacia* species.

Second, bird species responded idiosyncratically to vegetation structure. In mine rehabilitation sites, the site detection rates of a number of bird species increased in response to the canopy volume of small trees. In native forest sites, the site detection rates of a different set of bird species increased in response to the basal area of big trees. Other studies of birds in vegetation succession have also found that bird species respond idiosyncratically to different vegetation attributes (Venier and Pearce, 2005; Woinarski, 1979; Taylor *et al.*, 1997).

Third, in spite of similar bird species richness values, the composition of the bird assemblage in the oldest age class of mine rehabilitation at Weipa was significantly different from native forest bird assemblages. The different habitat-specific bird assemblages of the oldest age classes of mine rehabilitation and native forest are concealed by the similar numerical species richness values. Differences in the composition of the bird assemblages reflect the idiosyncratic nature of bird-habitat relationships. They also highlight the interactive nature of bird-habitat relationships, that is, birds actively select habitat.

The relationship between vegetation height, vegetation structure, vegetation composition and bird assemblages is habitat-specific and differs from one vegetation type to another. This interpretation is supported by the finding of Woinarski *et al*., (1988) that there was good agreement between the distribution of bird species and vegetation classification in the Top End of the Northern Territory providing that the classification is based on a broad range of parameters including both floristic and structural data.

Within native forest reference sites variation in the site detection rates of bird species was related to differences between sites in the relative contributions of the different vegetation layers. Winter and Atherton (1985) also identified patchiness in the distribution of birds and mammals within *Eucalyptus tetrodonta* open forest of the Weipa bauxite plateau. They recommended that a detailed comparative survey be conducted of the fauna and flora of the open forest to determine the scale of the patchiness and key habitat variables.

This study confirmed that there is patchiness in bird distributions within the *Eucalyptus tetrodonta* open forest. The data identified consistent patterns of bird response to variation in vegetation structure for 14 bird species. There was variation in the responses of a further nine bird species although the pattern of response was not related to the ordination axes. The lack of relationship between site detection rates and ordination axes for some species is probably due to a mismatch between the extent and grain of the sampling scale used in this study and the scale at which those species respond to the environment. Species distributions may be aggregated at spatial grains not sampled by this study. Even for species that are aggregated at a scale that could be detected using 2 ha study sites, the explanatory variables that potentially cause the aggregation may not be detected by a vegetation sampling procedure that uses means of vegetation data collected from systematic sampling points across the whole site. Summary vegetation data collected at the two hectare scale is only a first approximation of the habitat resources that actually define individual species' distribution and abundance. Determining the key resources and species-specific grain and extent first requires detailed observations of individual species foraging and movement behaviour. Meaningful single species modelling will then be supported by collection of species-specific data at an appropriate spatial grain and extent.

8.3.3 The landscape context of rehabilitation

Rehabilitation projects are embedded within landscapes. The effectiveness of passive rehabilitation projects in creating habitat that will be used by birds is expected to be determined both by attributes of the developing vegetation in the rehabilitation, and the landscape context. The permeability of the landscape (Hobbs, 2002) and the distance to potential source populations (Scott *et al.*, 2001; Miller and Hobbs, 2007) is expected to differ between bird species depending on: (1) their dispersal abilities; (2) their specific habitat requirements; and (3) how their populations have been affected by the initial loss of habitat. The effect of distance from source populations on bird populations in habitat remnants has been demonstrated in agricultural landscapes. Few studies, however, have quantified the effects of distance from source populations on birds in rehabilitation sites. Kavanagh *et al.*, (1985) found that the diversity of birds in clearfelled plots that were naturally regenerating was maintained by the proximity of unlogged forest.

This study found that bird species richness in mine rehabilitation sites was not increased by proximity to remnant vegetation or area of remnant vegetation within a 500 m buffer of a site, whereas site detection rates of individual bird species were. Three factors might account for this result. First, within-site vegetation attributes are more important in defining bird assemblages than landscape context. Even sites adjacent to remnant vegetation did not draw species from remnant vegetation if they did not provide suitable habitat. Second, the remnant vegetation within a 500 m site buffer may not be the source habitat for many of the bird species that occurred in mine rehabilitation sites. Alternative source populations may have come from other vegetation, including remnant mesic vegetation or other rehabilitation sites. The site detection rates of seven species were related to distance from mesic vegetation. Rather than habitat islands, vegetation in mine rehabilitation sites is part of a mosaic of vegetation of different rehabilitation ages. It is possible, therefore, that this mosaic of different aged vegetation in mine rehabilitation becomes a source of birds within the post-mining landscape. Third, the distance and area gradients sampled were not appropriately scaled.

Presumably species richness values in mine rehabilitation sites were not affected by proximity to areas of remnant vegetation because at the same time that the site detection rates of native forest birds decreased, the site detection rates of species sourced from other habitats increased. It appears, therefore, that adjacent remnant vegetation only becomes a source of faunal species for mine rehabilitation when vegetation structure in mine rehabilitation sites becomes sufficiently similar to the remnant vegetation or when it provides suitable habitat resources. The effectiveness of remnant vegetation as a source of dispersing fauna then, depends not just on how far away it is, but on how closely the different vegetation states in mine rehabilitation match individual species' habitat resource requirements. If this is so, then similarity of habitats and habitat resources in unlogged forest and naturally regenerating forest plots may account for the finding of Kavanagh *et al.*, (1985) that proximity to unlogged forests maintained bird diversity in plots of regenerating forest.

If this interpretation is correct, the importance of different habitat types in the landscape as sources of birds might be expected to change as different vegetation states develop within rehabilitation. That is, the relative importance of within-site habitat factors and landscape context factors is likely to vary throughout the succession depending on the extent to which vegetation in rehabilitation matches potential source habitats in the surrounding landscape. Increased habitat diversity in the surrounding landscape might also be expected to increase the diversity of source populations that could use different vegetation states in mine rehabilitation. Species richness in all age classes of mine rehabilitation may therefore be related to diversity of habitats in the landscape. This interpretation also has implications for the equilibrium theory of species number. In addition to the effects of distance from source habitat and island (or patch) area, the likelihood of immigrant species surviving on an island will presumably be a function of the similarity between source and destination habitats.

Further insight into the role of remnant landscape vegetation would require clear definition of habitat-specific bird assemblages and vegetation development states. It would also require comparing different vegetation states in mine rehabilitation and their associated bird assemblages with similar habitat types in the surrounding landscape and distances to them. That, however, was not the question of interest for this study. The results of this study show that if within-site habitat is not suitable, then the proximity of potential source populations is irrelevant. This is most important for species with specialised habitat requirements such as those that are dependent on mature trees.

8.4 The trajectory of post-mining rehabilitation

The long term outcomes of vegetation succession in response to small scale, low intensity natural disturbances acting on natural ecosystems are to a large extent predictable because of the biological legacies that exist at the outset of succession, including floristic composition. Plant species assemblages in natural ecosystems are not random. Through natural selection, a set of species that differ in their tolerances and performances, co-exist while competing for the same environmental resources. Although the relative contribution of individual species to total community biomass can change in response to disturbance, the floristic composition of natural ecosystems is generally in equilibrium with the site conditions (including interactions with other species) and the disturbance regime. The outcomes of differential plant species performance on vegetation structure will differ with the spatial scale and intensity of disturbance. Over long time frames, however, as long as the site conditions and disturbance regime remain stable, the outcomes of differential species performance following disturbance of natural ecosystems will result in characterisitic floristic composition and structure.

The pathways of vegetation change are far less predictable following intense disturbance such as that associated with bauxite mining. The bauxite mining process removes the accumulated biological legacies of the native forest including nutrient pools, biomass, soil physical structure, and organic matter. It also permanently alters site physical conditions including soil depth, soil texture, and hydrology. Post-mining rehabilitation at Weipa also uses a 'novel' combination of plant species (i.e., plant species that are derived from multiple areas and habitats whose ecological roles have not been selected within a single community). The outcomes of the interactions of these species, and the way that resources will be partitioned among them are unknown. Therefore, even though individual plant functional traits may be known, the patterns of vegetation change at the community level are unpredictable in 'novel' ecosystems.

Predictions about the long term pathway of vegetation development in such synthetic ecosystems are therefore highly uncertain. Nevertheless, enough is understood of successional processes that it is possible to predict with some confidence whether certain specific outcomes are likely. Demonstration that restoration of the pre-mining native forest was a possible outcome would require that even young mine rehabilitation sites satisfy a number of conditions. The following section discusses the results of this study in relation to the developing ecosystem in mine rehabilitation and compares them with the results of other studies.

8.4.1 Development of landscape functionality

Most of the accumulated nutrients on the Weipa bauxite plateau are held in the standing biomass and organic matter (Schwenke, 1999). These resources are lost when vegetation is cleared and burnt, and soil is disturbed. For rehabilitation to be successful in the long term it must address soil nutrient deficiencies, and ensure that the developing vegetation has the ability to retain and accumulate nutrients. This study used indicators of landscape function to assess this ability.

The interpretational framework for LFA predicts that the change in landscape function index values will approximate a sigmoidal response. The response in landscape function index values at Weipa was better represented by exponential asymptotic curves. This is probably due to the rapid biological response shown by vegetation in this climate (D. Tongway, personal communication, 2009). For the first one to two years there was little vegetation cover, < 12%, including perennial grasses and woody vegetation. During this time, soils are highly vulnerable to erosion associated with intense rainfall. In the absence of vegetation cover, control of resource loss is dependent on physical structures. Bank and trough structures in mine rehabilitation were not well defined and were often not aligned along contours. Consequently, there was evidence of a lot of sediment movement during this phase. However, due to the low topographic relief and the permeable soil texture there was little evidence of serious soil erosion in the form of rills and gullies. The incorporation of even small amounts of organic matter from annual grasses resulted in a rapid increase in soil aggregate stability, which raised the stability index.

As vegetation developed and exerted biological control over resources there was a concurrent increase in landscape function indices. All landscape function index values showed a rapid increase in the first 5 years, coinciding with the growth in vegetation height and canopy volume. After 5 years, the rate of change slowed until asymptote values were reached. The stability index reached an asymptote at approximately 12 years. Vegetation height and canopy volume reached asymptotes at approximately 15 years. Infiltration and nutrient cycling indices continued to increase after the stability index had reached an asymptote and appeared to reach asymptotes at approximately 20 years. Infiltration and nutrient cycling indices appeared to reach asymptotes at approximately the same time that basal area reached an asymptote. To a large extent, the changes in the landscape function indices were related to changes in the vegetation.

The extensive leaf litter cover that developed in mine rehabilitation sites was critical to the landscape function index values. Leaf litter in sites that had never been burnt in the two oldest age classes of mine rehabilitation was generally extensive, of local origin, sometimes exceeded 2 cm in depth, and showed signs of decomposition. These signs provided evidence of good resource retention on these sites. Leaf litter raised stability index values and infiltration index values in mine rehabilitation sites although relative to native forest sites these sites generally scored low values for rainsplash protection and surface resistance to disturbance.

Although the landscape organisation, infiltration and nutrient cycling indices in the oldest age class of mine rehabilitation matched or exceeded the mean values for all native forest reference land units, the pathways by which these functions were acquired were different from all native forest reference land units. The most important factor contributing to landscape function values in native forest reference sites was the density of perennial, fire tolerant vegetation. Native forest sites are dominated by perennial grasses and woody species that recover rapidly from fire by re-sprouting. Although the above-ground perennial grass cover is removed by fire, the plant species composition and structure of the vegetation is little affected by the low intensity grass fires that are frequent in the tropical eucalypt savannas. Consequently, vegetation control of soil and water resources is retained following fire in native forest sites.

Variation in landscape function values in native forest sites was largely a response to variation in the density of perennial grasses, in the amount of woody vegetation foliage cover and the patchy effects of fire.

The substantial contribution of leaf litter to landscape function index values in mine rehabilitation sites highlights the vulnerability of these landscape functions to fire. The introduction to mine rehabilitation of a novel assemblage of plant species with a different vertical vegetation structure may even result in a different type of fire regime in mine rehabilitation from the frequent low intensity grass fires that occur in the native forest. A number of authors have commented on the increased vulnerability to fire associated with the deep accumulated leaf litter in mine rehabilitation at Weipa (Reddell and Hopkins, 1994; Murray and Mulligan, 2003; Roberts, 1994). Vulnerability of landscape function to fire was also observed in post-mining rehabilitation sites at Gove (D. Tongway, personal communication, 2009).

Fire history was significantly related to reduced values for the infiltration and nutrient cycling indices. Leaf litter in sites that had been burnt in the two oldest age classes of mine rehabilitation was more patchy and showed little evidence of decomposition. Nevertheless, the predicted asymptote values for infiltration and nutrient cycling in burnt mine rehabilitation sites were comparable to the mean observed values for all native forest reference land units. The stability of the mechanisms via which those values have been attained, however, has not been rigorously tested. Mine rehabilitation sites were largely protected from fire and only three of the mine rehabilitation sites had been burnt more than once. In contrast, the native forest reference sites are burnt on a regular basis. These results are, therefore, not conclusive with respect to the effects of fire on landscape function values. They indicate that the mine rehabilitation sites have the potential to be self-sustaining as long as there are no other major physical constraints on long term development and as long as the vegetation is stable.

There may be physical constraints on long term vegetation development in mine rehabilitation at Weipa that are not detected by LFA. Soil organic matter quality and quantity in mine rehabilitation at Weipa was assessed in a chronosequence of up to 22 years (Schwenke *et al.*, 2000a). The study found positive signs of re-establishment of organic matter and nutrient cycling although equilibrium conditions were estimated to be more than a decade away. In spite of this, Schwenke *et al.*, (2000a) concluded that the potential for improving the physical properties of the soil in mine rehabilitation is severely limited. Compared with control sites, almost all soils in mine rehabilitation are shallower, have more compacted profiles, and contain a higher proportion of gravel. Soil compaction influences movement and retention of water and organic matter, and indirectly the chemical composition and decomposition dynamics of developing soils (Walker and Del Moral, 2003). The relationship between compaction and soil properties is complex and depends on the degree of initial organic matter removal, soil particle size and drainage and precipitation patterns. Soil compaction can be a persistent problem.

8.4.2 Vegetation development

As previously outlined there are many uncertainties in predicting the long term outcomes of vegetation change in post-mining landscapes. A number of recent studies, however, have found that initial rehabilitation treatments have lasting effects on plant species composition and vegetation structure. Research on vegetation succession following mining at Alcoa's bauxite mine in Western Australia found that the pathway of vegetation change in rehabilitating mine sites is best described by the initial floristic composition model. That is, plant species composition at the commencement of rehabilitation determines subsequent shifts in dominance with different species successively becoming dominant as they mature (Norman *et al.*, 2006; Grant and Loneragan, 2001). A study of a long term trial plot at Weipa also found that vegetation development is largely determined by the initial composition and density of applied seed (Murray and Mulligan, 2003). A comparison of plant species composition following sand-mining in New South Wales found significant differences between mined and unmined sites. Differences increase over time due to an increase in the abundance of exotic species and increasing differences in the abundance of native species (Buckney and Morrison, 1992). A study of woodland restoration on abandoned pasture in New South Wales found no evidence of directional change in vegetation dynamics (Wilkins *et al.*, 2003). The results of these studies highlight the importance of establishing suitable plant species early in rehabilitation rather than assuming that natural successional processes will take control, or that vegetation is climatically and edaphically pre-determined. Site history and the set of individuals that 'reach' a site dominate subsequent succession (Pickett and Cadenasso, 2005).

8.4.2.1 Plant species composition

This study found large and significant differences between native forest reference land units and mine rehabilitation sites in plant species composition. Most significantly for long term vegetation development, framework species (i.e., those species that dominate the composition, structure and function of the native forest reference land units) occurred at significantly lower densities in mine rehabilitation than in all native forest reference land units. Framework species made up more than 83% of the canopy volume of woody species in all native forest reference land units. In contrast, the contribution of framework species to the canopy volume of mine rehabilitation sites never exceeded 10%. The relative contribution of framework species to the canopy volume was highest in the oldest age class of mine rehabilitation. Rather than a gradient of values, however, there was step up in values between the three youngest age classes, in which framework species made up < 1% of the canopy volume, and the two oldest age classes of mine rehabilitation. This suggests that differences between the age classes of mine rehabilitation may be due to rehabilitation strategies rather than developmental change. Supplementary hand planting of tube stock including framework and desirable species was used in three out of the eight 9 to 16 year old mine rehabilitation sites and in all of the > 16 year old sites. Aerial seeding was also used in all mine rehabilitation sites 16 years and older. Only direct seeding was used in the three youngest age classes of mine rehabilitation in which the contribution of framework species was so low.

There were substantial differences in the fire response categories of the dominant plant species between native forest and mine rehabilitation on the Weipa bauxite plateau. Framework species for the native forest reference land units are all sprouters. Domination of the native forest by framework species, combined with their ability to survive fire means that plant species composition and vegetation structure in the native forest are stable following fire. Framework species also provide stability of habitat resources and of landscape functions. In contrast, the vegetation canopy in mine rehabilitation was dominated by species that are not part of the vegetation community in native forest reference land units. These included *Acacia* spp., *Dodonea* spp., *Grevillea* spp., and in the older age classes, *Alstonia* spp.. These species are predominantly non-sprouter species. Poor recruitment of sprouting species has been found to be a common issue in post-mining rehabilitation (Herath *et al.*, 2008; Bellairs and Bell, 1993; Ward *et al.*, 1997; Ward *et al.*, 1996).

A similar contrast in fire response categories between mine rehabilitation sites and native forest control sites was reported at Alcoa's bauxite mine in Western Australia. Non-sprouter species dominate vegetation cover in mine rehabilitation whereas sprouter species dominate density and cover in the native forest (Norman *et al.*, 2006). Differences in the fire responses of dominant plant species have important implications for how vegetation in mine rehabilitation will respond to fire. They also have implications for long term vegetation development as seeder species (i.e., non-sprouters) invest less resources in modifying soil properties and improving their edaphic environment (D. Tongway, personal communication, 2009).

Without management intervention, changes in the plant species composition in mine rehabilitation will be subject to the outcomes of differential species performance in the altered site conditions and is likely to be very slow. Roberts (1994) analysed the size class distribution of *Acacia* spp. and *Eucalyptus* spp. in mine rehabilitation at Weipa. He concluded that *Acacia* is not likely to decline in dominance due to high recruitment rates of *Acacia* seedlings. The sustainability of *Eucalyptus* populations in mine rehabilitation, however, is not assured due to the decline in stem density of eucalypts over time and very low recruitment rates of *Eucalyptus* seedlings (Roberts, 1994).

Walker and del Moral (2003) stated that colonisation from undisturbed natural areas appears to be the key factor promoting succession. At Weipa, recruitment of local native species into post-mining areas from the mine perimeter is likely to be a slow process for a number of reasons. First, recruitment from mine boundaries will be slowed by the large areas of mine without any remnant vegetation retained. Second, natural recruitment by *Eucalyptus tetrodonta* is thought to have a limited reliance on recruitment from seedlings due to the development of vegetative reproduction as a protective mechanism for frequent fire (Bell and Williams, 1997). Third, naturally slow recruitment rates may be further slowed by the dense *Acacia* cover and leaf litter cover in rehabilitating mine sites.

8.4.2.2 Vegetation structure

Initial rehabilitation treatment has also been found to have lasting effects on vegetation structure. Vegetation structure in post-mining rehabilitation at Alcoa's bauxite mine in Western Australia differed significantly depending on the initial seed mix and rehabilitation treatment (Norman *et al.*, 2006). Fox *et al.*, (1996) conducted a long term study of the effects of different types of initial disturbance on vegetation structure. They compared the effects of sand-mining and subsequent rehabilitation to native species, with the effects of fire on un-mined *Eucalyptus* forest. Significant differences in vegetation structure remain 17 years after post-mining rehabilitation (Fox *et al.*, 1996). At a phosphate mine in Idaho, the effects of different soil types and rehabilitation treatments are more evident after 14 years than 4 years after rehabilitation (Chambers *et al.*, 1994).

The substantial differences in plant species composition in mine rehabilitation at Weipa, were reflected in the significant and substantial structural differences in vegetation structure. Mean vegetation height in the oldest age class of mine rehabilitation, which was 17 to 23 years old, was approximately half that of any of the native forest reference land units. Mean values for total basal area and total canopy volume in the oldest age class of mine rehabilitation were similar to the post-mining landscape analogue LU 5K. It should be noted, however, that due to differences in plant species composition (there are interspecific differences in the allometric relationships between basal area and above-ground biomass) and size class distribution of woody stems between native forest and mine rehabilitation, similar basal area values do not translate into similar total above-ground biomass.

There were significant differences in size class distribution of woody stems, and consequently vegetation layering, between the oldest age class of mine rehabilitation and native forest land units. Tall shrubs and small trees contributed a much higher proportion of the foliage cover in the oldest age classes of mine rehabilitation than in all native forest reference land units. In contrast, there was no big tree layer[20] and the perennial grass and low shrub layers were virtually absent from mine rehabilitation. Vegetation in the oldest age class of mine rehabilitation can be characterised as a 2-layered mixed species low open forest *sensu* (Read, 1987). In contrast, vegetation in the pre-mining native forest, land unit 2B, can be characterised as a 4-layered *Eucalyptus* open forest. Vegetation in the two post-mining landscape analogue

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[20] See footnote 15 on p.185

land units, 2C and 5K, can be characterised as a 5-layered *Eucalyptus* woodland (Read, 1987). The big tree layer accounts for most of the biomass.

Differences in structure are expected between sites of such different age. However, the trends in vegetation structure in mine rehabilitation suggest that further change in values is likely to be very slow. There was a phase of rapid growth of woody vegetation in mine rehabilitation for the first 5 to 8 years followed by a slower phase of consolidation and expansion. Mean vegetation height and mean canopy volume in mine rehabilitation sites reached asymptotes after approximately 15 years while basal area reached an asymptote after approximately 20 years. The mean vegetation height in mine rehabilitation reached an asymptote at a value approximately half the mean vegetation height for any of the native forest reference land units. It is not clear whether the asymptotes for mean vegetation height, mean total canopy volume and mean total basal area in mine rehabilitation at Weipa are simply due to the dominant plant species having reached their maximum development potential or whether limiting abiotic factors such as soil physical properties, soil compaction or depth to the watertable are involved.

The pattern of vegetation development in bauxite mine rehabilitation at Weipa differs from that reported for bauxite mine rehabilitation at Gove (Spain *et al.*, 2006). Over a 26 year chronosequence in mine rehabilitation at Gove, there was a transient phase of *Acacia* dominance between 3 and 10 years. This was followed by a phase of increasing dominance by *Eucalyptus* and decline of *Acacia* spp., during which the three tiered structure of the native forest was attained. Over the 26 year chronosequence, vegetation height and basal area in mine rehabilitation at Gove was reported to attain values similar to, or greater than, those of the native forest reference sites, although unmined native forests on deeper soils were much taller (Spain *et al.*, 2006).

A number of factors may contribute to the differences in vegetation structure between mine rehabilitation sites at Gove and Weipa. First, the native forest reference sites at Weipa and Gove differ. Mean vegetation height of the tallest vegetation layer in native forest reference land units at Weipa found in this study was highest at 24 m in LU 2B (maximum height 35 m) and lowest at 19 m in LU 5K (maximum height 27 m). In contrast, mean vegetation height of native forest reference sites at Gove ranged from a low of 7.5 m up to a maximum 18 m. The unmined native forest at Gove is also described as three-layered in contrast to the four or five-layered native forest at Weipa. Presumably, the native forest reference sites at Gove are on shallower soils than the native forest reference sites at Weipa. Dry season burning of native forest at Gove is also frequent.

Second, the post-mining landscape at Gove does not impose the same physical constraints as at Weipa. Soils in post-mining rehabilitation at Gove are well above the watertable compared to Weipa (D. Tongway, personal communication, 2009). Third, differences in vegetation structure between Gove and Weipa may be due to differences in the relative abundances of

Acacia species and *Eucalyptus* species. Spain *et al.*, (2006) found different height trajectories for *Acacia* spp. and *Eucalyptus* spp.. The trajectory for the mean height of *Acacia* spp. at Gove was similar to the trajectory for mean vegetation height at Weipa. It also reached an asymptote well below the values for control sites while the mean height of *Eucalyptus* spp. continued to increase. Fourth, differences in vegetation sampling strategies may have resulted in the Gove vegetation data being biased towards smaller stems than the Weipa vegetation data.

It is likely that vegetation structure in mine rehabilitation at Weipa will continue to slowly change. In the long term, however, the degree of similarity in vegetation structure between native forest and mine rehabilitation sites will be limited by the differences in plant species composition. This study found that vegetation composition and structure on the Weipa bauxite plateau were strongly related. Restoration of the vertical vegetation structure and horizontal spatial order of the native forest reference land units would appear, therefore, to be dependent on restoration of the plant species composition of the native forest.

8.4.2.3 Destabilising species

Acacia species are often thought of as having a facilitative role in plant succession because of their nitrogen fixing capabilities and contribution to soil development. Domination of vegetation by shrubby nitrogen fixing species, however, can inhibit the establishment of shade intolerant species so that the positive effects of shade and organic matter are offset by competition for light and nutrients (Walker and Del Moral, 2003). Reddell and Hopkins (1994) suggested that the low fire tolerance, relatively short life spans and high soil seed banks generally associated with *Acacia* species may contribute to ecosystem instability in the post-mining environment at Weipa. Furthermore, dense *Acacia* canopies are associated with production of high leaf litter loads which increase site flammability and suppression of framework species by shading (Ward *et al.*, 1990; Grant and Loneragan, 1999). Transient *Acacia* dominance followed by natural self-thinning was reported for Alcoa's bauxite mine in Western Australia (Ward *et al.*, 1990) and for Alcan's bauxite mine at Gove (Spain *et al.*, 2006). However, Reddell and Hopkins (1994) argued that there is no evidence in the monsoonal tropics that *Acacia* dominated communities are a seral stage in a vegetation succession leading to a climax *Eucalyptus* dominated woodland.

At Weipa, vegetation cover in all except the oldest age class of mine rehabilitation was dominated by non-sprouter species, *Acacia*, *Dodonea*, *Grevillea*, and *Alstonia* species. Murray and Mulligan (2003) found that the development of vegetation structure in mine rehabilitation at Weipa is driven by long-lived *Acacia*, *Grevillea* and *Dodonaea* species. These species contribute to very high levels of leaf litter cover, a substantially reduced grass layer and ongoing suppression of *Eucalyptus* spp. expression at the canopy level. In a long term trial plot, *Acacia* spp. contribute 30 to 50%, *Grevillea* spp. up to 20% and *Eucalyptus* spp. less than 5% cover

(Murray and Mulligan, 2003). Roberts (1994) suggested that the thick mat of *Acacia* leaf litter that dominates mine rehabilitation sites at Weipa possibly has adverse effects on seedling establishment and contributes to ongoing suppression of understorey development. Koch and Davies (1993) cited in (Grant *et al.*, 1997) also found that a dense *Acacia* understorey in mine rehabilitation at Alcoa's bauxite mine in Western Australia may have suppressed the growth of smaller species due to competition for light, water and nutrients.

Exotic species can also seriously disrupt or alter the trajectory of succession, leading to stable degraded states (Prach *et al.*, 2007; Suding *et al.*, 2004). The presence of even low densities of exotic species such as *Leucaena leucocephala* and *Andropogon gayanus* are considered to pose a threat to long term vegetation development in mine rehabilitation at Weipa (Reddell and Hopkins, 1994; Murray and Mulligan, 2003). Fire and exotic species can interact to further facilitate the presence of exotic species. Gamba grass *Andropogon gayanus*, for example, is an exotic grass from Africa that produces heavy fuel loads that promote extremely hot fires that are damaging to native vegetation (Csurhes and Edwards, 1998).

The mine rehabilitation sites used in this study were not representative with respect to the question of exotic species. Planned bulldozing and re-treatment of weed infested mine rehabilitation sites during this study precluded site selection from large areas of the mine. However, *Andropogon gayanus* was recorded in five of the mine rehabilitation sites and *Leucaena leucocephala* was recorded in an additional five sites. Leaving questions of *Acacia* dominance aside, presence of exotic species, therefore, potentially posed a threat to long term vegetation development in 32% of the mine rehabilitation sites studied. Given the constraints on site selection this can be considered a minimum value. Roberts (1994) also found that weed infestation in mine rehabilitation sites at Weipa possibly poses a threat to vegetation succession.

8.4.2.4 The effects of fire on the developing vegetation

The question of how the vegetation in mine rehabilitation will respond to fire is central to understanding future pathways of vegetation change. It has been argued that for sustainability to be demonstrated, the vegetation in rehabilitating sites must have the capacity to resist the effects of likely perturbations (Spain *et al.*, 2006; Nichols, 2006). As fire is a frequent disturbance in the monsoonal eucalypt woodlands (see section 2.6), the establishment of fire tolerant vegetation is considered to be critical to successful rehabilitation in northern Australia (CSIRO, 1997; Reddell and Hopkins, 1994).

Mine rehabilitation sites at Weipa are generally protected from fire by conducting early dry season burns around the mine using mine perimeter roads acting as fire breaks. Protection from fire is necessary for young vegetation during establishment and early development. A few studies have found that the length of time that young mine rehabilitation sites should be protected from fire depends on sprouting species being well enough established to survive fire

(Grant and Loneragan, 2001; Ross *et al.*, 2004). However, given the environmental context of mine rehabilitation on the Weipa bauxite plateau, if vegetation in mine rehabilitation is to become maintenance free, the vegetation will need to acquire the ability to survive the fire regime.

No records of the timing of the accidental fires that have occurred in mine rehabilitation have been maintained. This study was therefore not able to rigorously assess the effects of fire on vegetation in mine rehabilitation. Nevertheless, analysis of the vegetation data indicates that fire is likely to threaten the stability of vegetation cover in mine rehabilitation at Weipa. Fire history caused significant reductions in values for canopy volume and basal area in mine rehabilitation sites. Murray and Mulligan (2003) assessed the effects of fire on vegetation development in mine rehabilitation sites at Weipa. The effects of fire differ with the age class of rehabilitation at the time of burning, and plant species composition. They found that vegetation structure declines in sites that are less than 6 years old when first burnt. Other sites experience a large increase in the density of *Grevillea* spp. suspected to result from the high seed load occurring in the soil seed bank. Sites older than 20 years remain stable in terms of density, richness and diversity following fire although they do not develop in a similar way to the native vegetation due to differences in floristic composition.

Controlled trials have been conducted on the effects of fire on plant species composition in mine rehabilitation at Alcoa's bauxite mine in Western Australia. Burning 11 to 13 year old mine rehabilitation sites results in increased *Acacia* density and weed density compared to native forest sites (Grant and Loneragan, 1999). Ross *et al*., (2004) found that the effects of fire on mined areas of different rehabilitation age following sand-mining in New South Wales differ significantly with rehabilitation age. Burning rehabilitation sites that are more than 10 years old increases native species richness and seedling densities. By 20 to 26 years after mining, burning brings species richness within the range of unmined sites of similar disturbance interval. Below this threshold, however, mined areas lack the capacity to recover. The effects of fire on mine rehabilitation are not related to time *per se,* however, but to the individual reproductive characteristics of the component plant species. However, the effects of fire and mining are synergistic and the outcomes for native species richness are not predictable from the effects of the two separate disturbances acting separately (Ross *et al*., 2004). Grant and Loneragan (1999) concluded that, on its own, fire would not be sufficient to increase the similarity in plant composition and structure between mine rehabilitation and native forest without other manipulations. The effects of repeated fire on vegetation and soil seed banks do not appear to have been tested.

8.4.2.5 LFA as a tool for monitoring rehabilitation

The goal of landscape function analysis in the context of mine rehabilitation is to provide a predictive measure of ecosystem or landscape trajectory (Tongway and Ludwig, 2006). In focussing on the functional role of vegetation at a fine spatial scale, it separates ecological processes from plant species composition. However, the identity of constituent plant species is the basic determinant of ecosystem functioning (Leps, 2006). Over long time frames vegetation has an important role in the development of soil physical and biological structure (Walker and Del Moral, 2003). Floristic composition determines how stable vegetation cover will be for a given disturbance regime. Stable vegetation communities provide a constant supply of organic matter to the soil which is important to the development of equilibrium soil conditions (Schwenke *et al.*, 2000a). An understanding of floristic composition would therefore appear to be fundamental to predicting the ecosystem trajectory.

In a hierarchy of factors, climate and parent material are meso-scale determinants of vegetation structural formations. Their effects, however, are mediated by topographic and edaphic conditions. At the topographic scale, site conditions, differential species performance and disturbance interact over time to determine the site specific vegetation outcomes (Pickett and Cadenasso, 2005). In turn, vegetation composition and structure are the primary determinants of small scale surface processes. Given the novel plant species assemblages used in post-mining rehabilitation and the altered site hydrological conditions, there is no assurance that the vegetation in mine rehabilitation will ultimately resemble the pre-mining native forest.

Tongway and Ludwig (2006) claimed that inappropriate species selection would be revealed by a failure of the nutrient cycling index to increase. However, as it was applied in this study, LFA was insensitive to floristic composition in mine rehabilitation sites, which was dominated by non-sprouter species. The LFA index values were also unrelated to the rate of establishment of framework species. Given the importance of floristic composition in determining the long term pathway of vegetation change it should be monitored alongside landscape functionality from the outset of rehabilitation. This is particularly the case where rehabilitation sites are protected from fire and where establishment of sprouter species is important.

8.4.3 Is mine rehabilitation likely to restore habitat similar to that of pre-mining native forest?

The term habitat is used in various ways. It is used in the sense of habitat type to describe an area supporting a particular type of vegetation, or it can be used to refer to the specific combination of resources and environmental conditions that promote the occupancy of a particular species (Morrison *et al.*, 2006). Vegetation type is not a synonym for habitat. Other resources such as access to water are important habitat attributes. Nevertheless, vegetation composition and structure determine the type, number and amount and stability of habitat resources that are central to understanding bird distributions and abundance. In this study, the term habitat is used to refer to the vegetation, including plant species composition, relative plant species abundance, biomass, vertical structural complexity and horizontal spatial order. Asking the question, 'is mine rehabilitation likely to restore habitat similar to that of pre-mining native forest?', is therefore equivalent to asking whether mine rehabilitation will restore vegetation similar to the *Eucalyptus tetrodonta* open forest of the Weipa bauxite plateau.

This study found no significant differences in the composition of the bird assemblages and mean bird abundance between the three native forest reference land units. The two post-mining landscape analogue land units, 2C and 5K are therefore suitable reference ecosystems for restoration of bird habitat.

The differences in vegetation composition and structure found by this study between all native forest reference land units and the mine rehabilitation sites were such that it is concluded that mine rehabilitation results in habitat conversion rather than habitat restoration. Rehabilitation is unlikely to restore vegetation similar in composition or structure to the pre-mining *Eucalyptus tetrodonta* open forest in the foreseeable future. Given the permanent changes to soil and hydrology caused by mining, restoration of the pre-mining native forest *sensu stricto* is probably not feasible. Furthermore, mine rehabilitation is not likely to restore vegetation similar in composition or structure to either of the two post-mining landscape analogue land units in the foreseeable future.

A number of lines of evidence lead to these conclusions. First, restoration of the vertical structure and horizontal spatial order of the pre-mining *Eucalyptus tetrodonta* open forest or post-mining landscape analogue land units would appear to be dependent on achieving similar plant species composition. However, there are large differences in the vegetation composition between mine rehabilitation and all native forest reference land units. Plant species composition in mine rehabilitation at Weipa is expected to increase in similarity to the native forest vegetation community at very slow rates, if at all. This inference is drawn because of: (1) the importance of initial floristic composition to vegetation development found at other mine rehabilitation sites; (2) the slow expected rate of natural recruitment from the mine boundary; and (3) the presence of species in mine rehabilitation that are likely to destabilise vegetation development. The results of previous studies indicate that fire is unlikely to result in significant changes in the plant species composition of the vegetation in mine rehabilitation. Second, the observed trends in development of vegetation structure in mine rehabilitation indicate that further change in values is likely to be very slow and dependent on a change in the plant species composition.

The vegetation established in mine rehabilitation at Weipa did provide habitat resources for birds. It provided habitat resources for 18 bird species that are not part of the pre-mining native forest bird assemblage. There was evidence that one of these species, white-streaked honeyeater, was breeding in mine rehabilitation. Mine rehabilitation also provided habitat resources that were used to varying degrees by 70% (55/79) of the *Eucalyptus tetrodonta* open forest birds including many birds that are dependent on mature vegetation for at least some habitat resources. There was evidence that two of these native forest bird species, striated pardalote, and olive-backed sunbird were breeding in mine rehabilitation. However, for the majority of the native forest birds, mine rehabilitation only provided partial restoration of the habitat values of the pre-mining native forest. The presence of these species in the post-mining landscape remains dependent on their having access to native forest. Mine rehabilitation did not provide habitat at all for 30% (24/79) of the *Eucalyptus tetrodonta* open forest birds including the native forest habitat specialists.

Including the species that were recorded incidentally and excluding the species that were only recorded in mine rehabilitation, there were 91 native forest birds. Of these 62% (56/91) occurred in mine rehabilitation and 38% (35/91) only occurred in native forest. These findings are similar to the findings of a study of birds in tropical primary, secondary and plantation forests, which found that more than 40% of all bird species occur exclusively in primary forest (Barlow *et al.*, 2007).

Based on the low stem densities of framework species, the vegetation in mine rehabilitation is likely to continue to only provide partial restoration of habitat for the native forest birds into the foreseeable future. Even over long time frames mine rehabilitation is unlikely to provide habitat in the total sense, (i.e., including foraging, roosting and breeding resources) for the majority of native forest birds that are specialised on habitat resources associated with mature *Eucalyptus tetrodonta.* This is particularly the case for sedentary species and habitat specialists, species that in addition to requiring access to specific types of habitat resources, require that those resources be available at minimum densities and in particular spatial configurations.

8.5 Conservation implications of habitat conversion for the native forest birds

This study found that the composition of bird assemblages is related to specific habitat types. It also found that a number of bird species are specifically associated with mature *Eucalyptus tetrodonta* habitat. Clearing *Eucalyptus tetrodonta* open forest, followed by bauxite mining and rehabilitation on the Weipa bauxite plateau has both immediate and long term impacts on the spatial and temporal configuration of habitats and associated bird populations. The temporal configuration, or age structure, of vegetation is altered with immediate and long term consequences for populations that depend on the habitat resources associated with mature eucalypts. These are resources that only develop over time. Long term change in the spatial configuration of habitats also occurs as the *Eucalyptus tetrodonta* open forest is replaced by vegetation that is dominated by non-eucalypts.

Altering vegetation age structure results in a long term reduction in the availability of key age-dependent habitat resources such as tree hollows. This can cause population sizes to be chronically limited with consequences for the maintenance of viable local bird populations (Vesk and Mac Nally, 2006). For viable local populations to be maintained, sufficient suitable habitat resources are required to: (1) sustain existing individuals; (2) allow adults to successfully breed; and (3) have populations that are large enough to cope with demographic fluctuations and exogenous impacts (Vesk and Mac Nally, 2006). Even assuming that vegetation in mine rehabilitation was made up of appropriate plant species, the long time lags in the provision of habitat resources associated with mature vegetation (Vesk and Mac Nally, 2006; Vesk *et al.*, 2008), mean that mine rehabilitation would not be expected to provide crucial limiting nesting resources such as tree hollows for many decades. Data from old rehabilitation plantings indicate that in excess of 100 years may be required before many important habitat resources become available (Mac Nally, 2008).

Mature, stable *Eucalyptus* formations provide a specific set of habitat resources that is associated with habitat-specific bird assemblages. Although the degree of habitat-specificity varies, most of the native forest birds on the Weipa bauxite plateau depend at some time on habitat resources that are specifically associated with mature native forest. Even less habitat-specific native forest birds that were recorded in mine rehabilitation depend on mature vegetation some of the time. For example, eight of the native forest birds that occurred in mine rehabilitation at Weipa still need access to hollows for breeding (Table 8.1). A few studies have found that rehabilitation does not provide habitat for breeding birds that are dependent on habitat resources associated with mature vegetation including tree hollows, fissures, bark crevices, high forks or canopy foliage (Curry and Nichols, 1986; Mac Nally *et al.*, 2008). Underlining the importance of mature native forest for bird populations, 9 out of 11 species that are of conservation concern found in this study occurred only in native forest (Table 8.1). One species of conservation concern, the bush stone-curlew, occurred in both native forest and mine rehabilitation, and one occurred only in mine rehabilitation, the Australian bustard.

The more sedentary and the more specialised a species is in its foraging, roosting or nesting requirements, the more strongly it is related to mature, stable vegetation. These specialist species are most vulnerable to loss or conversion of habitat as they have traded off flexibility against specialisation. Increased specialisation is highly correlated with increased likelihood of extinction. A conservative estimate is that 41% of the bird species that are limited to one habitat type are extinction prone (Sekercioglu *et al.*, 2004). The findings of this and other studies confirm that specialist species, such as trunk-gleaning insectivores, that are dependent on mature vegetation do not return even decades after rehabilitation.

Table 8.1 Native forest birds absent from or significantly less abundant in mine rehabilitation

	Native forest bird species absent from mine rehabilitation [1]	Native forest bird species significantly less abundant in mine rehabilitation
	emu, great frigatebird, lesser frigatebird, pied imperial pigeon, tawny frogmouth, osprey, pacific baza, white-bellied sea eagle, wedge-tailed eagle, peregrine falcon, azure kingfisher, red-browed pardalote, yellow-tinted honeyeater, blue-faced honeyeater, varied sittella, grey fantail, northern fantail	brush cuckoo, red-backed fairy-wren, yellow honeyeater, banded honeyeater, white-throated honeyeater, noisy friarbird, little friarbird, white-bellied cuckoo-shrike, cicadabird, rufous whistler, olive-backed oriole, black-backed butcherbird, leaden flycatcher, lemon-bellied flycatcher, mistletoebird
Species of conservation concern that are threatened by land clearing [5]	square-tailed kite, grey goshawk, grey-crowned babbler, brown treecreeper (H) [4] red-tailed black-cockatoo (H)	bush stone-curlew [2]
Additional species of conservation concern	radjah shelduck (H), palm cockatoo (H), channel-billed cuckoo, oriental cuckoo	
Hollow users	southern boobook, dollarbird, little woodswallow	sulphur-crested cockatoo, rainbow lorikeet, red-winged parrot, laughing kookaburra, blue-winged kookaburra, forest kingfisher, grey shrike-thrush, pale-headed rosella [3]

1. Six species that were only recorded in native forest but are probably not dependent on native forest habitat resources were removed from this table including orange-footed scrubfowl, fork-tailed swift, white-throated needletail, grey whistler, magpie-lark and tree martin.
2. Bush stone-curlew was recorded in both native forest and mine rehabilitation in low numbers.
3. Pale-headed Rosella occurred in mine rehabilitation and was more abundant in native forest sites but not significantly so.
4. (H) indicates hollow users.
5. Information on threats from (Marchant and Higgins, 1993; Higgins, 1999; Higgins and Peter, 2002; Higgins *et al.*, 2001).

Habitat loss or habitat conversion due to bauxite mining will lead to localised population reductions of native forest birds on the Weipa bauxite plateau. A review of the effects of habitat loss on temperate eucalypt woodland birds concluded that for many species the population declines that occur following vegetation clearing are far greater than simply the proportion of habitat lost (Johnson *et al.*, 2007). The effects of habitat loss on population viability will differ for each species (Morrison *et al.*, 2006). Even subtle differences between congeneric species can result in significantly different outcomes (McCarthy *et al.*, 2000). Most of the native forest bird species recorded in this study are widespread and are currently not considered to be of conservation concern. Depending on effects elsewhere, population reductions on the Weipa bauxite plateau are therefore not critical to the global conservation status of many of the native forest bird species. However, nine taxa recorded exclusively in native forest in this study are of conservation concern (Table 8.1). The distribution of two of these is restricted to Cape York Peninsula, the Cape York Peninsula sub-species of the brown treecreeper, and the palm cockatoo. Both of these taxa use *Eucalyptus tetrodonta* open forest on the Weipa bauxite plateau and therefore suffer habitat loss due to mining.

The *Eucalyptus tetrodonta* open forest of the bauxite plateau is important habitat for the Cape York Peninsula sub-species of the brown treecreeper (Garnett and Crowley, 1995). The status of this taxon is Near Threatened because the population has declined in density over more than half of its range (Garnett and Crowley, 2000a). Early in the 20th century, the brown treecreeper was recorded as widespread and abundant in Cape York Peninsula. It is now a rare bird on Cape York Peninsula. The northern population of the Cape York Peninsula form of the brown treecreeper appears to be confined to the Weipa area (Garnett and Crowley, 1995). In this study brown treecreepers were recorded exclusively in native forest, and showed a preference for open grassy sites where the woody vegetation was dominated by large trees (> 35 cm dbh). Any process that causes further population reductions of the Cape York Peninsula sub-species of the brown treecreeper is of concern.

In Australia, palm cockatoos are only found on northern Cape York Peninsula where in the late 19th century and early 20th century they were recorded as common (Barnard, 1911; MacGillivray, 1914). Their current conservation status is now Rare (Qld) and Near Threatened (Garnett and Crowley, 2000a) due to the small size of the adult population. Various human activities have been implicated in population declines and range contraction of the palm cockatoo on Cape York Peninsula. Murphy *et al.*, (2003) suggest that the most serious threat to palm cockatoos on Cape York Peninsula may come from altered fire regimes. Their slow life history makes palm cockatoos sensitive to environmental perturbations. Recent modelling of palm cockatoo population viability on eastern Cape York Peninsula suggests that their low reproductive rate does not balance mortality (Heinsohn *et al.*, in press). The Iron Range palm cockatoo population is therefore possibly in decline.

Palm cockatoos are obligate hollow-nesters that nest in near vertical, hollow limbs or hollow broken off trunks usually in *Eucalyptus tetrodonta* or *Corymbia* species (Wood, 1988; Murphy and Legge, 2007). Throughout their range they inhabit rainforests and adjacent woodlands and reach their highest breeding densities in woodlands. Nearly all palm cockatoo nests are in eucalypt woodland in the vicinity of rainforest (including riparian forest). On average nests are 320 m, but up to 1.04 km, from rainforest (including riparian forest) (Murphy *et al.*, 2003). Suitable nest holes are a limiting resource for palm cockatoos (Heinsohn *et al.*, 2003). Although many *Eucalyptus tetrodonta* trees have hollows, evidence of interference competition between palm cockatoos and year long nest defence by male palm cockatoos suggests that there is strong competition for nest sites (Murphy *et al.*, 2003). Palm cockatoos were only observed in low numbers in this study, but were observed exclusively in native forest. Native forest clearing on the bauxite plateau particularly in the ecotones between woodland and vine forests, and woodland and riparian zones is therefore likely to reduce the availability of palm cockatoo nest trees which may result in increased intra-specific competition for nest hollows.

Furthermore, palm cockatoos have been observed to compete with sulphur-crested cockatoos for hollows in eucalypt woodland on eastern Cape York Peninsula (Heinsohn *et al.*, 2003). Loss of sulphur-crested cockatoo nest trees on the bauxite plateau due to native vegetation clearing may therefore contribute to increased inter-specific competition for the remaining nest trees. The pattern of mining means that the remnant vegetation after clearing for mining is almost exclusively within the ecotone where palm cockatoo breeding density would be expected to be highest. Increased intra- and inter-specific competition for nest hollows is likely to result in even lower reproductive success. Native vegetation clearing is also likely to cause local reduction in food availability. Palm cockatoos feed on kernels of large fruit in open forest (Wood, 1988). Forshaw (1964) reported that nearly all of the food plants known to be important to palm cockatoos are found in open forest. Palm cockatoo food trees, include *Canarium australianum*, *Parinari nonda*, *Grevillea glauca*, *Pandanus* spp., and *Persoonia falcata* (Forshaw and Muller, 1978; Forshaw, 1964). Any process that potentially reduces the breeding success of a slow breeding species with a small adult population and a restricted distribution is of concern.

Habitat loss on the Weipa bauxite plateau is also likely to impact on local populations of at least five other species[21] that are rare or threatened (Appendix 3). These five species were recorded exclusively in native forest in this study: grey goshawk, square-tailed kite, grey-crowned babbler, red-tailed black cockatoo and radjah shelduck. None of these is restricted to Cape York Peninsula but habitat loss anywhere in a species' distribution becomes part of the cumulative impacts on it and contributes to its overall decline.

Mature vegetation on the Weipa peninsula is also important for two migratory marine species. The roost sites of great frigatebird *Fregata minor* and lesser frigatebird *Fregata ariel* are confined to patches of remnant *Eucalyptus tetrodonta* vegetation on the western Weipa peninsula. They are the only known mainland roost sites of these species globally. It has been estimated that these roost sites are likely to support a substantial proportion of the national and possibly international populations of both species (Mustoe, 2008). Mature vegetation especially eucalypts with horizontal or bare branches are preferred roosting trees. Presumably, particular attributes of roosting trees are required for successful take-off and landings of these large seabirds.

This study has found that not only the age structure of vegetation is altered. Mining and subsequent rehabilitation on the Weipa bauxite plateau has resulted in habitat conversion. The vegetation in mine rehabilitation is a novel assemblage composed of different plant species that form different habitat structures and support a significantly different assemblage of bird species. Restoration of vegetation similar in composition and structure to *Eucalyptus tetrodonta*

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[21] Not all species that have previously been recorded in the *Eucalyptus tetrodonta* open forest and are of conservation concern were recorded in this study.

woodlands in post-mining rehabilitation is not anticipated in the foreseeable future. The post-mining rehabilitation techniques used at Weipa therefore result in at least a long term and possibly a permanent reconfiguration of habitat types on the Weipa bauxite plateau. Mining and post-mining rehabilitation on the Weipa bauxite plateau therefore contribute to the causes of biodiversity decline.

It has been argued that increased habitat diversity associated with mining and subsequent rehabilitation is associated with increased species richness compared to natural vegetation (Brenner and Kelly, 1981; Steele and Grant, 1982; Reeders and Morton, 1983). This has the implication that higher species richness is an improvement. By converting habitat, mining and subsequent rehabilitation on the Weipa bauxite plateau does increase habitat diversity and consequently beta diversity *sensu* Cody (1975), within the area of operations. However, the change in habitat configuration needs to be seen from a broader perspective. When the age structure and spatial configuration of habitats is changed, the landscape scale abundance of species also changes. Species associated with the newly created habitat benefit at the expense of species associated with mature habitats. A local increase in species richness achieved by spatial reconfiguration of habitat, therefore, is not equivalent to an increase in species richness at the landscape or global scale. As this discussion has highlighted it is likely to be associated with a reduction in the conservation status of specialist species, and species dependent on habitat resources associated with mature vegetation.

Emphasis on numerical values of species richness fails to account for changes in species composition, including a reduction in the abundance of the characteristic local fauna. Species richness is an aggregate measure of multiple species distributions that conveys nothing of the ecological or evolutionary history of each species. Comparisons between intact and disturbed environments that focus on species richness without reference to differences in species composition, fail to recognise the importance of characteristic bird assemblages. Characteristic species assemblages, including species identity, relative species abundance, foraging guild composition, and habitat specificity, which have co-evolved within a particular environmental context provide a more ecologically appropriate reference against which to measure rehabilitation success.

The findings of this study are consistent with the predictions of the habitat templet theory. This interpretation is based on the use of vegetation surrogates rather than direct measurement of specific habitat resources, however, the composition of bird assemblages and community structure were related to vegetation structure, which is assumed to determine both the amount and variety of habitat resources. The presence of trunk-foraging specialists was also related to particular spatial configuration of habitat, which is assumed to represent habitat favourability.

Most studies of bird-habitat relationships have focused on the role of vegetation structure. The habitat templet theory also focuses on the role of durational stability and productivity on vegetation structure. Although this study found that vegetation structure was important, plant species composition was central to the question of differences in habitat structure and bird assemblages. Plant species composition determines what specific types of habitat resources are available as well as vegetation structure. The relationships between vegetation height, vegetation structure, plant species composition and the composition of bird assemblages therefore differed from one vegetation type to another. The role of habitat in shaping foraging group structure also appeared to be conditional on the interrelated factors of plant species composition, structural complexity and productivity of the vegetation. Plant species composition is also central to the question of the trajectory of landscape function.

Chapter 9: Conclusions

9.1 Introduction

To assess how effective post-mining rehabilitation on the Weipa bauxite plateau has been in restoring habitat values this study posed the broad question: To what extent has post-mining rehabilitation on the Weipa bauxite restored the bird habitat values of the *Eucalyptus tetrodonta* open forest? In this concluding chapter, the findings of this study are summarised in relation to the four key research questions posed. Overall conclusions are then drawn with respect to the broad research question, drawing on both the results and the discussion. The management and policy implications of these conclusions are then discussed. Finally, some limitations of this research and future research needs are outlined. To begin with, the question of the suitability of birds as a taxonomic group for monitoring the effectiveness of post-mining rehabilitation on the Weipa bauxite plateau is addressed.

9.1.1 Birds as indicators of rehabilitation success

This study compared site detection rates of one taxonomic group of vertebrate fauna, namely, birds. This was not intended to imply that birds are indicators of how other vertebrate groups might respond to rehabilitation. Birds are an important component of biodiversity in their own right. They also provide useful quantitative data by virtue of their abundance and detectability. The suitability of birds as indicators of habitat quality in general has been questioned given their high mobility and the ability of many bird species to use a variety of habitats opportunistically (Hilty and Merenlender, 2000). In this study, birds were sufficiently differentiated in their foraging behaviour and habitat preferences to be useful indicators of habitat change in post-mining rehabilitation at the scale of the Weipa mining operation. Bird assemblages (including species composition, species abundance and foraging group structure) in mine rehabilitation reflected the changing vegetation structure. Birds therefore proved to be a useful taxonomic group for assessing the effectiveness of mine rehabilitation in restoring habitat on the Weipa bauxite plateau. It appears that the suitability of different taxonomic groups for assessing the effectiveness of rehabilitation is at least partly determined by the scale of rehabilitation in relation to faunal movements.

9.2 Findings in relation to key research questions

9.2.1 How do the bird assemblages of post-mining rehabilitation compare with native forest bird assemblages?

This study found significant differences in the composition and abundance of bird assemblages between mine rehabilitation sites and all native forest land units. Site category (including age class), season, time since fire, and weather conditions all had significant effects on bird abundance. Site category, however, had by far the largest effect. Differences in mean site species richness between the two oldest age classes of mine rehabilitation and the native forest reference land units were not significant. There were significant differences, however, in bird species composition between the oldest age class of mine rehabilitation and all native forest reference land units. The mean species shortfall index, which is a measure of the similarity in species composition and mean species abundance to a reference condition, in the oldest age class of mine rehabilitation was 63% compared with a mean species shortfall index of 27% for the pre-mining native forest land unit 2B.

In total, 111 bird species were recorded, 97 in surveys, and 14 incidentally. Of the 97 bird species for which abundance data was obtained, 25% (24/97) were recorded exclusively in native forest sites, 19% (18/97) were recorded exclusively in mine rehabilitation, and 56% (55/97) were shared between native forest and mine rehabilitation. Excluding the 18 species that occurred exclusively in mine rehabilitation, 79 native forest bird species were recorded in surveys of which 30% (24/79) were recorded exclusively in native forest.

Forty percent (22/55) of the shared species had significantly higher mean detection rates and mean abundance in native forest sites. Ten per cent (6/55) of the shared bird species had significantly higher mean detection rates and mean abundance in mine rehabilitation sites. Differences in the site detection rates and mean abundances of the remaining 50% (28/55) of shared species were not significant, largely due to low overall abundances. The striated pardalote was the only species that was reasonably abundant in both native forest and mine rehabilitation sites. There was evidence that two of the shared species, striated pardalote and olive-backed sunbird, were breeding in mine rehabilitation.

Insectivore-nectarivores dominated bird assemblages in native forest and all age classes of mine rehabilitation except the youngest age class, 1 to 2 years old, in which grass and ground foraging insectivores dominated. Native forest sites had the highest number of foraging groups, the highest species richness of seven foraging groups, and the highest mean abundance of eight out of 14 foraging groups. Frugivores, foliage-gleaning insectivores, hawking insectivores, insectivore-nectarivores, nectarivores, pouncing predators and tree seed eaters were all significantly more abundant in native forest. One entire foraging group, trunk-gleaning insectivores was completely absent from mine rehabilitation. Conversely, granivores, ground foragers, and grass and ground foraging insectivores were significantly more abundant in mine rehabilitation. There were no significant differences between native forest and mine rehabilitation in the abundance of omnivores, aerial insectivores and carnivores.

9.2.2 What is the temporal pattern of bird succession in post-mining rehabilitation on the Weipa bauxite plateau?

A number of attributes of the bird assemblages in mine rehabilitation changed with rehabilitation age and were closely related to changes in vegetation structure, particularly vegetation height, which was the strongest predictor of bird species richness. In addition to increasing mean bird abundance and mean site species richness, the composition of the bird assemblages changed with increasing rehabilitation age. Three trends characterised the changing bird species composition along the successional gradient: (1) turnover of bird species generally associated with open, grassy or heathy habitats in the four youngest age classes of mine rehabilitation; (2) accumulation of bird species generally associated with mesic habitats with increasing rehabilitation age; and (3) accumulation of native forest bird species with increasing rehabilitation age. Although the composition of the bird assemblage became increasingly similar to native forest bird assemblages with increasing age, it remained significantly different even in the oldest age class of mine rehabilitation.

Changing bird species composition was most clearly reflected in the temporal patterns of community structure. The number of foraging groups, the species richness within each foraging group, and mean foraging group abundance all increased with rehabilitation age, and generally had their highest values in native forest. The species richness of four foraging groups, however, peaked in different age classes of mine rehabilitation. The mean abundance of two foraging groups also peaked in different age classes of mine rehabilitation.

9.2.3 What environmental factors determine bird occupancy in pre-mining native forest and post-mining rehabilitation on the Weipa bauxite plateau?

Site vegetation structure was an important determinant of bird species composition. Vegetation height was strongly correlated with bird species richness and with the site detection rates of many individual bird species. Although height was the strongest vegetation structure correlate, birds were not necessarily responding to height *per se* because height incorporates a great deal of unspecified information about vegetation structure.

Bird species responded idiosyncratically to site vegetation structure. Nevertheless, there were consistent response groupings of birds indicating habitat-specific bird assemblages. Ordination based on site detection rates of birds and site vegetation structure completely separated native forest reference land units and mine rehabilitation sites. In native forest sites, variation in the site detection rates of birds was related to variation between sites in the relative contributions of different vegetation layers.

The pattern of relationships between birds and vegetation structure was most evident at the level of foraging groups. Foraging groups that use different components of mature vegetation, such as canopy gaps, tree trunks, tree seed, fruit and perches for pouncing predators, were significantly more abundant in native forest reference sites. Foraging groups that use open ground, grass or low shrubs were significantly more abundant in mine rehabilitation sites. There were no significant differences between mine rehabilitation and native forest in the abundance of foraging groups that hunt or search for food from the air.

Distance to remnant vegetation, and the area of remnant vegetation within a 500 m site buffer were not significantly related to summary measures such as species richness or species shortfall. Bird species responded idiosyncratically to the distance from remnant vegetation and area of remnant vegetation within a 500 m site buffer. The site detection rates of some bird species were also related to distance from mesic vegetation.

9.2.4 Is the post-mining rehabilitation process likely to restore habitat similar to that of pre-mining native forest on the Weipa bauxite plateau?

All landscape function values increased with rehabilitation age and acquired values within the range of reference values for all native forest reference land units. Vegetation composition in rehabilitating mine sites was very different from all native forest reference land units. The vegetation in mine rehabilitation sites included many species that are not part of the pre-mining vegetation community, including native species that are not part of the regional flora, and exotic species. Only 34% of perennial grass species and 40% of woody plant species were recorded in both mine rehabilitation and native forest reference sites. There were large and significant differences between native forest reference land units and mine rehabilitation sites in the relative contributions of perennial grass species and woody plant species to the total canopy volume.

Most importantly, the framework plant species that dominate vegetation in the native forest, and which are all sprouters, occurred at substantially lower densities in mine rehabilitation sites than in all native forest reference land units. There were also large and significant differences between all age classes of mine rehabilitation and all native forest reference land units in the relative contribution of framework plant species to the total canopy volume. The canopy volume in mine rehabilitation sites was dominated by non-sprouter species including *Acacia*, *Dodonea* and *Grevillea* species and *Alstonia* in the older age classes. Given the high fire frequency on the Weipa bauxite plateau, these non-sprouting species may contribute to environmental instability. Exotic species with the potential to seriously destabilise vegetation succession were recorded in 32% of the mine rehabilitation sites studied.

Associated with the differences in vegetation composition, there were substantial differences in vegetation structure between the oldest age class of mine rehabilitation and native forest. Although structural differences are expected in vegetation of such different ages, the trends in vegetation structure indicate that ongoing change is likely to be very slow. Perennial woody vegetation in rehabilitating mine sites grew rapidly in the first five to eight years followed by a period of slow growth. Vegetation height and canopy volume reached asymptotes after approximately 15 years. Mean vegetation height reached an asymptote at approximately half the value of the native forest reference land units. Canopy volume reached an asymptote between the values for the two post-mining landscape analogue land units, 5K and 2C. Basal area reached an asymptote after approximately 20 years.

The observed values for mean total basal area and mean total canopy volume in the oldest age class of mine rehabilitation were similar to the post-mining landscape analogue land unit 5K. It should be noted, however, that due to differences in plant species composition and in the size class distribution of woody stems, similar values for basal area m^2/ha do not translate into similar values for total biomass (see section 8.4.2.2). There were large differences between mine rehabilitation and native forest reference land units in how the total vegetation biomass was distributed into vegetation layers. Most notably, there was a complete absence of a big tree layer in the oldest age class of mine rehabilitation, which included sites up to 23 years old. The basal area and canopy volume in mine rehabilitation was associated with a high stem density of tall shrubs and small trees. There was a virtual absence of perennial grass and low shrub layers in all age classes of mine rehabilitation.

9.3 Overall conclusions

The post-mining rehabilitation strategies used to date at Weipa have established vegetation cover. In the absence of fire, landscape functionality should not pose an impediment to long term vegetation development. However, the large and significant differences between mine rehabilitation and native forest sites in the composition and structure of bird assemblages (including species composition, species abundance and foraging group structure), lead to the conclusion that the rehabilitation strategies used to date have so far only partially restored the bird habitat values of the pre-mining *Eucalyptus tetrodonta* open forest. Nor have they restored the bird habitat values of either of the two land units nominated as ecologically appropriate analogues for the post-mining landscape. Rather, it would be more accurate to describe the post-mining rehabilitation as having so far resulted in habitat conversion that provides some habitat resources but not all of the habitat requirements of the original native forest bird community.

No support was found for the proposition that restoration of habitat similar to the pre-mining native forest can be expected at some time in the future. Given the lack of available long term monitoring data (the oldest mine rehabilitation site studied was 23 years old), the possibility cannot be excluded that vegetation in mine rehabilitation could at some time shift to a successional pathway that results in vegetation more like the native forest reference land units. However, given the large differences between native forest and mine rehabilitation sites in plant species composition and densities of framework species; the observed trends in vegetation structure; and the presence of destabilising species in mine rehabilitation, it is reasonable to conclude that the rehabilitation strategies used to date are unlikely to restore habitat similar in composition and structure to any of the native forest reference land units in the short to medium term. The distribution of the pre-mining native forest is strongly correlated with the bauxite plateau, indicating that the development of an open forest structure is dependent on the hydrological conditions of the bauxite profile. The altered hydrological conditions in the post-mining landscape provide further reason to doubt that vegetation in the post-mining landscape will ever resemble the pre-mining native forest. Mining and subsequent rehabilitation has therefore resulted in long term if not permanent conversion of the pre-mining habitat.

It is concluded that vegetation based habitat resources (defined by vegetation composition and structure) are key determinants of the composition and structure of bird assemblages at Weipa. Specifically, mature *Eucalyptus* formations provide a set of habitat resources that is associated with habitat-specific native forest bird assemblages. Although this study found that many of the native forest bird species used habitat resources across a variety of vegetation states, many others that specialise in habitat structure or habitat resources associated with mature *Eucalyptus* vegetation did not. Rehabilitation did not replace the habitat structure or habitat resources associated with mature *Eucalyptus* vegetation. Changes in the composition and structure of the vegetation, therefore, result in changes in the composition and structure of bird assemblages.

The vegetation that has been established in post-mining rehabilitation at Weipa provided habitat resources for 18 bird species that were not sourced from the pre-mining native forest. It also provided at least some habitat resources for many of the mobile, generalist native forest birds. Comparison of mean detection rates and effects of distance from remnant vegetation on individual species, however, indicates that most birds are using resources in mine rehabilitation on an opportunistic basis and cannot be said to have colonised the new habitat. The dependence of these bird species on habitat resources provided by mature vegetation means that the presence of even many generalist species in mine rehabilitation is conditional on the presence of remnant native forest in the landscape.

Importantly, post-mining rehabilitation did not provide habitat for at all for sedentary species and foraging specialists. Many of the native forest species that were absent from mine rehabilitation are species of conservation concern due to habitat loss in other parts of their ranges. Thus it appears that the native forest bird species that are most sensitive to habitat loss and for which there is most need to restore habitat, may be the last to return to it, if they return at all. The post-mining rehabilitation strategies used to date at Weipa have so far not been effective in restoring habitat resources for the native forest bird species that are most sensitive to habitat loss.

9.4 Implications of findings

This thesis began by highlighting the need for improved understanding of faunal responses to rehabilitation across a range of ecosystem types and climatic zones. Site-specific evidence has been presented on the extent to which native forest birds utilise post-mining rehabilitation on the Weipa bauxite plateau. Evidence has been presented that post-mining rehabilitation at Weipa has so far only partially restored bird habitat values. Full restoration of the bird habitat values of the *Eucalyptus tetrodonta* open forest is not anticipated in the short to medium term. Although rehabilitation did provide habitat resources for many birds, it did not provide the specific habitat resources or habitat structure required by many of the native forest birds. Even if species richness is restored, providing habitat through rehabilitation does not address the causes of biodiversity decline if it involves a change in the composition of faunal assemblages. Population declines and range contractions are part of the process of biodiversity decline. If rehabilitation is to be an effective strategy for minimising the biodiversity losses caused by habitat loss, it must be demonstrated that it can support populations of the same species that are threatened by the initial loss of habitat. The long term if not permanent conversion of *Eucalyptus tetrodonta* open forest habitat by mining and subsequent rehabilitation has implications for rehabilitation practices generally, implications for biodiversity management on the Weipa bauxite plateau, as well as broader policy implications.

9.4.1 Implications for rehabilitation practices

Broadscale rehabilitation techniques that rely on vegetation succession to deliver long term rehabilitation and biodiversity goals are unlikely to be sufficient to restore habitat. Minimising biodiversity losses requires that each rehabilitation project adopt some broad principles and develop a range of site-specific rehabilitation techniques. With respect to the broad principles, there is a need for rehabilitation projects to set clearly defined, financially and technically achievable biodiversity goals. These goals should include target plant species composition and structure, and target fauna for which rehabilitation is intended. Rehabilitation success can only be measured against clearly stated, specific goals. Second, it needs to be recognised that rehabilitation is a long term process with an uncertain trajectory. This requires that there be a commitment to providing sufficient resources to support ongoing adaptive management of rehabilitation. This will require ongoing monitoring of rehabilitation to assess progress against stated rehabilitation goals, and to inform the refinement of rehabilitation techniques.

With respect to the development of site-specific rehabilitation techniques, ongoing monitoring is need to inform improved models of site-specific vegetation succession. More accurate models of vegetation succession are needed to support delivery of the specific biodiversity goals. The findings of this study also highlight the need to monitor the fauna for which rehabilitation is intended to provide habitat. Monitoring combined with improved models of vegetation succession in post-mining landscapes and backed up by long term resources will enable timely intervention to ensure that departure from desired rehabilitation pathways is addressed. Finally, minimising biodiversity losses by providing habitat for target species requires an understanding of the specific habitat requirements of target species including foraging, roosting, breeding and shelter resources. Providing these specific habitat resources is likely to require a range of tailored site-specific management strategies in addition to broad scale rehabilitation of vegetation.

9.4.2 Biodiversity management implications for the Weipa bauxite plateau

The conclusions of this study also have implications for managing biodiversity on the Weipa bauxite plateau. The findings of this study that (1) mine rehabilitation does not provide habitat for a third of the native forest birds; and (2) many of the native forest bird species were using habitat resources in mine rehabilitation opportunistically and depend on the presence of native forest in the landscape; lead to the conclusion that there is a need to retain reserves and corridors of *Eucalyptus tetrodonta* open forest around and within the mine.

At the level of species conservation, replacement of the *Eucalyptus tetrodonta* open forest with mine rehabilitation can be expected to result in local declines in the abundances of many of the native forest dependent birds. Nine of these are species of conservation concern. Although mining has so far impacted on less than 2% of the *Eucalyptus tetrodonta* open forest, less than 2% of this ecosystem is reserved in the protected area estate (Accad *et al.*, 2008). Preventing biodiversity decline requires maintenance of viable species populations throughout their ranges. Careful analysis of the potential long term and accumulated impacts of mining on local species populations is required to ensure that viable local populations are maintained.

The long term conservation needs of two bird taxa in particular, which are confined to Cape York Peninsula, should be determined as a priority. These are the palm cockatoo and the Cape York Peninsula sub-species of the brown treecreeper. Both are sedentary hollow users and closely associated with mature *Eucalyptus tetrodonta.* The palm cockatoo is rare, has a small adult population and low reproductive success. Modelling of the population viability of palm cockatoos on Cape York Peninsula suggests that the population may be in decline (Heinsohn *et* *al.*, in press). The northern population of the Cape York Peninsula form of the brown treecreeper is near threatened and appears to be confined to the Weipa area (Garnett and Crowley, 1995).

One strategy for minimising biodiversity losses would be a network of protected areas based on a regional assessment of long term biodiversity conservation requirements. This study found that patchiness in the site detection rates of a number of native forest bird species was related to between-site variation in vegetation structure. Differences in site detection rates probably also reflect the different area requirements or territory sizes of different bird species. This highlights the need for any systematic conservation planning process to incorporate the variation in vegetation composition and structure that occurs within the *Eucalyptus tetrodonta* open forest and to be aware of the overlap between mining impacts and species distributions.

9.4.3 Broad policy implications

The findings of this study also have broader implications for policies that aim to use rehabilitation as strategy for minimising biodiversity losses. Environmental offsetting policies permit clearing of mature habitat based on the assumption that habitat loss can be offset by habitat restoration *in situ* or elsewhere. There is an increasing number of such policies including: environmental offsets under the Australian government EPBC Act, the Queensland government's Environmental Offsets Policy, the New South Wales government's Biodiversity Banking and Offsets Scheme, the Victorian government's BushBroker scheme, and the Western Australian government's Environmental Offsets Policy. The implications apply particularly to regions in which large amounts of native forests and woodlands have already been cleared and the amount of remnant vegetation that remains may already be less than what is required for the long term conservation of all bird species.

The assumption that rehabilitation can offset the loss of mature habitat effectively assumes that space can be substituted for time. This study has found that rehabilitation (space) is not able to substitute those habitat resources that are related to mature vegetation (time). Issues of plant species composition aside, environmental offset policies that treat mature and young habitats as ecologically equivalent, do not account for the fundamental role of vegetation structure in determining suitable fauna habitat. Nor do they account for the loss of habitat resources specifically associated with mature vegetation, such as tree hollows and trunk foraging surfaces. This study found that the native forest bird species most sensitive to habitat loss and for which there is most need to restore habitat, may be the last to return to it, if they return at all. Great care should be taken, therefore, in using environmental offsets in areas where vegetation clearing will reduce the habitat of species of conservation concern. Even in landscapes that have not been mined, rehabilitation will not restore the age-specific habitat resource requirements of many forest and woodland dependent species except over long time frames.

As a signatory to the Convention on Biological Diversity, Australia has a stated commitment to the conservation of biological diversity including preventing the causes of significant reduction or loss of biological diversity (Commonwealth of Australia, 1995). Habitat loss, including habitat conversion, is recognised as the most serious threat to biodiversity (Primack, 2002). The willingness of regulators to continue to permit land clearing in the face of this fact appears to be at least partly based on the assumption that biodiversity losses due to land clearing can be offset by rehabilitation. The findings of this study point to limitations in this assumption. Policy makers, regulators, industry and community need to understand the limitations of rehabilitation. This is not an argument against rehabilitation *per se*. Rehabilitation is especially important in landscapes where there is little extant mature vegetation and where there are real opportunities for improving biodiversity conservation outcomes through rehabilitation. However, the results of this study highlight the need for an empirical approach to assessing the effectiveness of rehabilitation and overall impacts on biodiversity.

9.5 Limitations of this study and directions for future research

As a cross-sectional study of bird habitat relationships in sites representing a chronosequence, this study was necessarily limited in duration to ensure that vegetation and landscape function data for sites of particular age remained related to site bird data. Inter-annual variation in bird assemblages may be expected in response to climatic variation and off-site impacts on nomadic and migratory species. Longer term variation may also occur in response to climate change. Therefore, although bird data was collected over 16 months and across a range of seasonal conditions, it only represents a snap-shot in time of bird assemblages. This approach does not invalidate the differences found in bird-habitat relationships between native forest and mine rehabilitation. However, the bird data obtained over the duration of this study may not fully represent the bird assemblages of the *Eucalyptus tetrodonta* open forest and post-mining rehabilitation on the Weipa bauxite plateau.

The conclusions that could be drawn about individual species-environment relationships from this study were also limited by the spatial grain of the sampling procedures. Use of a standardised scale of observation and systematic vegetation sampling procedures meant that this study was only able to identify species-environment relationships at a broad habitat scale. However, this study identified patchiness in the site detection rates of a number of bird species, some of which are of conservation concern. Identification of the specific environmental variables that are related to the occupancy rates of these species and the spatial scale at which those species occupy the landscape is important for management. Detailed studies in which the spatial grain and extent are matched to the movement and foraging behaviours of individual species are justified for at least two species of conservation concern that occur on the Weipa bauxite plateau. Research to identify the distribution, population size, movement behaviour, and habitat resource requirements of the palm cockatoo, and the Cape York Peninsula form of the brown treecreeper is required to inform conservation management of these species and to improve rehabilitation strategies. Ideally this would also quantify the population sizes and factors affecting the reproductive success of both taxa.

There was a large amount of variation in the landscape function and vegetation data particularly for the two oldest age classes of mine rehabilitation. Inclusion of more replicates of these age classes may have strengthened the results[22]. Likewise, inclusion of more replicates of the two post-mining landscape analogue land units and less of the pre-mining native forest reference land unit may have reduced the standard error in the vegetation data for the two post-mining landscape analogue land units.

The conclusions that could be drawn about the importance of landscape context were also limited by the research design. The nature of the mining process on the Weipa bauxite plateau results in no isolated patches of remnant vegetation that are (1) on economic ore body and (2) surrounded by mine rehabilitation. Annual mine expansion and uncertainty in the mine plan also make it risky to select sites close to the mine boundary if sites are required for more than one year. Nevertheless, the research design would have been improved by selection of native forest sites that represented more of a gradient in values in the amount of remnant vegetation surrounding sites.

Inter-specific competition has been found to be an important factor affecting bird use of remnant vegetation. This is particularly the case when vegetation structure has been altered or key vegetation based habitat resources such as nesting hollows become limiting. This study did not attempt to quantify inter-specific interactions between birds. Future research into the effects of inter-specific competition on the distribution and abundance of birds may be warranted. In particular this could focus on the effects of intra- and inter-specific competition for tree hollows on hollow-dependent species such as the palm cockatoo. Research into the effects of vegetation loss on the ability of aggressive bird species to competitively exclude other species or exploit food resources to a level that adversely impacts other species may also be warranted.

This study quantified and compared individual bird species' detection rates in post-mining rehabilitation and native forest but did not quantify or compare breeding rates. Establishment of breeding populations would provide conclusive evidence that vegetation in mine rehabilitation is providing habitat for resident species. If further research on bird utilisation of mine

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[22] Having a large number of native forest reference sites enabled collection of previously unavailable baseline bird and vegetation data. This was of practical as well as scientific value. Baseline data on the stem densities of different species in the three native forest reference land units has already been provided to Rio Tinto’s rehabilitation specialist to assist with the development of post-mining rehabilitation targets.

rehabilitation is intended, quantitative data on bird breeding rates in both native forest sites and mine rehabilitation may be warranted.

The conclusions of this study are based on a 23 year chronosequence of mine rehabilitation sites. The extent to which conclusions can be drawn about the long term trends in vegetation composition and structure are therefore limited. Long term research on the pathway of vegetation development is required. Ideally this would take the form of time-series monitoring of controlled and replicated trials that compare different rehabilitation techniques. This information is needed to refine rehabilitation strategies and improve understanding of the factors involved in causing variation in vegetation development. More information about how the developing vegetation community in mine rehabilitation will respond to disturbance, particularly fire, is also required to inform ongoing management of rehabilitation and refinement of rehabilitation strategies. In particular, more information is urgently needed about the requirements for successful establishment of the local framework species.

Used appropriately, fire is a useful land management tool. If the vegetation in mine rehabilitation is ever to provide stable habitat and be integrated into the landscape, it will need to be able to survive the fire regime. For these two reasons it is particularly important to understand how vegetation in mine rehabilitation will respond to fire. The results of this study indicate that fire history accounted for a large component of the variation in vegetation and landscape function data. These results were evident even with a low level of replication of burnt mine rehabilitation sites. Accurate modelling of the effects of fire will require a controlled and replicated trial with a large number of sites in which the effects of fire on different age classes of mine rehabilitation is tested. Ideally, vegetation composition and structure and landscape function would be assessed prior to burning, the timing and intensity of the fire would be recorded, and the post-fire responses (including vegetation, landscape functionality and fauna) would be monitored over a number of years.

Coda

This is the first study that has directly compared birds, vegetation and landscape function in reference native forest ecosystems and a chronosequence of post-mining rehabilitation sites. The mosaic of age classes in the post-mining landscape and the scale of the operation at Weipa provide interesting opportunities for researching temporal patterns in ecosystem development. The scale of post-mining rehabilitation also offers the opportunity to conduct controlled replicated long term experiments that compare different rehabilitation treatments and management interventions. There is a pressing need to improve our understanding of vegetation succession in post-mining landscapes and how vegetation in post-mining rehabilitation may respond to a range of management interventions. Further research that makes use of these opportunities in a robust way is to be encouraged.

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Appendices

Appendix 1 Land units of the Weipa region[1]

(Redrawn after Gunness et al., 1987) Shaded rows indicate the land units included in this study

Terrain	Internal drainage	Parent material	Soils	Vegetation structure	Key plant species	Land unit
Plateau	excessive	laterite, bauxite	red earths	Tall woodland	*E. tetrodonta, C. nesophila*	2b
				Notophyll vine forest		3c
		laterite, sandstone	earthy sands	Tall woodland	*E. leptophleba*	2a
	imperfect	laterite, bauxite	yellow earths	Tall woodland	*E. tetrodonta, C. nesophila*	2c
Eroding slopes	medium	laterite	red and yellow earths, ironstone outcrops	Grassy woodland	*E. nesophila, E. tetrodonta*	5k
	medium to slow	laterite, bauxite	red and yellow earths	Grassy woodland	*E. cullenii, E. tetrodonta, E. chlorostachys*	5b
	imperfect	colluvium	yellow and gleyed podzolics	Grassy woodland	*E. confertiflora, E. tetrodonta*	5f
Gentle lower slopes and drainage lines	excessive	laterite, sandstone	earthy sands	Tall woodland	*E. leptophleba*	2a
		laterite, bauxite	red earths	Notophyll vine forest		3c
	imperfect	colluvium	bleached yellow podzolics	Woodland to tall woodland	*Pandanus sp., E. polycarpa*	5e
			loams to sands	Woodland	*Banksia □entate, Grevillea glauca, Parinari nonda*	5j
			yellow or gleyed podzolics	Grassy woodland	*Melaleuca viridiflora, Imperata cylindrica*	7d
	very poor – water table at or near surface	variable	variable	Gallery forest	*Melaleuca spp., Lophostemon suaveolens,* scrub species	4a
		alluvium	gleyed soils	Mesophyll palm forest	*Xanthostemon crenulatus*	4b
				Sclerophyll fern forest	*Melaleuca spp., S. palustris*	4c
		sandstone	podzol-like soils	Closed scrub	*Leptocarpus* sp., insectivorous plants	8b
		sandy colluvium	podzol-like soils	Sedgeland		12a

Appendix 1 Land units of the Weipa region continued

Terrain	Internal drainage	Parent material	Soils	Vegetation structure	Key plant species	Land unit
Basins	Poor (swamp outer margin)	variable	gleyed or yellow podzolics	Low woodland and tall shrubland	*Melaleuca viridiflora, Lophostemon suaveolens*	7b
	Very poor (swamp – persistent level)	variable	podzolics (bleached horizons)	Closed or open forest	*Melaleuca* spp.	3b
	Very poor (zone below average water table)	variable	podzolics or grey clays	Sedgeland	*Eleocharis* sp.	12b
Sandplains and dunefields	well-drained	coastal sediments – partly lateritic	massive siliceous sands	Woodland	*E. polycarpa, E, chlorostachys (E. leptophleba absent)*	5a
	excessive (beach)	sand	siliceous sand	Grassland, herbfield		12c
				Woodland	*Casuarina equisetifolia*	5c
	excessive (rear)	sand	siliceous sand plus shellgrit	Sclerophyll vine woodland	*Acacia crassicarpa, Canarium australianum*	7a
				Semi-evergreen notophyll vine forest		3a
		calcareous sand	calcareous sand	Semi-evergreen vine thicket	*Pongamia pinnata, V. negundo, D. umbellatum*	6a
	excessive (swale)	sand	siliceous sand	Semi-evergreen notophyll vine forest		3a
				Paperbark woodland	*Melaleuca dealbata*	5d
				Paperbark woodland	*Melaleuca viridiflora*	7c
Marine terraces	poor	alluvium	bleached duplex soils	Grassy woodland	*E. papuana*	5i
		coastal sediments	clay and duplex soils	Grassland	*V. □longate, Pandanus* sp.	12d
		coastal sediments	solonchaks	Sedgeland, grassland	*Fimbristylis sp., S. virginicus*	12f
Saline flats	very poor – supratidal (seldom flooded by tide)	coastal sediments	solonchaks	Sedgeland, grassland	*Fimbristylis sp., S. virginicus*	12f
		estuarine sediments	saline podzolics	Closed scrub	*M. acacioides*	8c
			solonchaks (saltpan)	Sparse herbfield	*Arthrocnemum* spp.	12e
Mangroves (mud flats)	poor – brackish	estuarine sediments	firm muds	Closed forest – freshwater zone	*M. cajuputi, A. speciosum*	3e
	poor – supratidal, flooded by highest tides			Shrubland	*A. annulata, E. agallocha, A. eucalyptifolia*	9a
	very poor – Intertidal (frequently flooded to 0.5 m)			Closed scrub, low closed forest	*Ceriops* spp.	6c
				Closed scrub, low closed forest	*Bruguiera* spp.	6b
	very poor – flooded by every tide (up to 1 m)		Muds, loose muds	Closed forest	*Rhizophora* spp.	3d

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula

The following records have been compiled from the reports of various ornithologists. It is not always clear that records were from Eucalyptus tetrodonta woodlands, authors variously refer to open forest, open pockets or make no record of habitat. However any species that would seem to be unlikely are only listed where the author has specifically stated that it was recorded in open forest.

1. Barnard spent four months spent on northern Cape York Peninsula in 1910-11 (Barnard, 1911).
2. MacGillivray (1914) reported on the observations of McLennan who worked for MacGillivray as a collector for 6 months in the Gulf of Carpentaria and northern Cape York Peninsula during 1911 (MacGillivray, 1914). MacGillivray reported the observations of McLennan just to the south of Albatross Bay on a walk inland a couple of miles through messmate forest country on 24th May 1914 and on trips to the Archer River in 1914 and 1915 (MacGillivray, 1917; MacGillivray, 1918a; MacGillivray, 1918b).
3. Thomson conducted three expeditions to Cape York Peninsula in 1928, 1929 and 1932 (Thomson, 1935).
4. Beruldsen spent ten days spent at Weipa. Note that little time was spent in the open forest (Beruldsen, 1979).
5. Kikkawa visited Weipa in February 1975 (Kikkawa, 1975).
6. Reeders and Morton conducted a survey of birds in mine regeneration including a habitat island of *Eucalyptus tetrodonta* open forest. The records reported here are for the habitat island only (Reeders and Morton, 1983)
7. Biological Environmental Research Services undertook a pre-feasiblity environmental assessment for Shell in 1982 (BERS, 1982)
8. Keast conducted a comparison of birds in northern and southern eucalypt woodlands including surveys in Eucalytpus tetrodonta woodland on Cape York Peninsula (Keast, 1985b).
9. Dames and Moore conducted an EIS of the Alcan bauxite lease including fauna surveys (Dames and Moore, 1996).

10 Winter and Atherton undertook an inventory survey of vertebrate fauna of the Weipa region during 2 dry season visits, one wet season visit and one post-wet season visit in 1980 and 1981 (Winter and Atherton, 1985);

11. Lethbridge and Macmillan conducted surveys of terrestrial vertebrate fauna in vine forests and surrounding woodland on the Rio Tinto mine lease at Andoom (Lethbridge and Macmillan, 1996)
12. Thomas conducted a survey of habitat preferences of vertebrate fauna at Weipa (Thomas, 2004).
13. Gould PhD research (2006-2007). Survey - indicates that the species was recorded during one of the 2 hectare 20 minute bird surveys; Incidental – indicates that the species was only recorded incidentally. *F* – indicates that the species was recorded in native forest; M – indicates that the species was recorded in mine rehabilitation; Shaded cells in column 13 indicate species that were not observed during this research; Shaded rows indicate species recorded in this research that have not previously been recorded for *Eucalyptus tetrodonta* open forest.

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Casuariidae													
Emu *Dromaius novaehollandiae*	Plentiful	Numerous	Numerous		Noted		Wet			Uncommon	Noted		Survey – F
Southern Cassowary *Casuarius casuarius*							Dry						▨

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Megapodiidae													
Australian Brush-turkey *Alectura lathami*							Wet						Survey – M
Orange-footed Scrubfowl *Megapodius reinwardt*													Survey – F
Phasianidae													
Brown Quail *Coturnix ypsilophora*				Noted ?					Dry season				Survey – M
King Quail *Excalfactoria chinensis*			Coastal										
Anatidae													
Radjah Shelduck *Tadorna radjah*										Noted		Noted	Incidental – F
Columbidae													
Common Bronzewing *Phaps chalcoptera*		Noted	Noted										
Crested Pigeon *Ocyphaps lophotes*					Noted								
Squatter Pigeon *Geophaps scripta*			Frequently seen										
Diamond Dove *Geopelia cuneata*		Noted	Well distributed										
Peaceful Dove *Geopelia striata*	Plentiful	Common	Numerous	Noted			Wet	Noted	Wet / Dry	Abundant		Noted	Survey – F / M
Bar-shouldered Dove *Geopelia humeralis*	Plentiful, breeding	Common	Numerous		Common	Noted	Wet	Noted	Wet / Dry	Abundant	Common	Noted	Survey – F / M
Pied Imperial-Pigeon *Ducula bicolor*					Noted					Common seasonally		Noted	Survey – F

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Podargidae													
Tawny Frogmouth *Podargus strigoides*		Common	Numerous				Wet / Dry		Wet	Abundant	Noted	Noted	Survey – F
Papuan Frogmouth *Podargus papuensis*	Fairly common	Common	Numerous		Noted					Common		Noted	Survey – F / M
Marbled Frogmouth *Podargus ocellatus*												Noted (?)	
Eurostopodidae													
Large-tailed Nightjar *Caprimulgus macrurus*		Common	Numerous						Wet	Common			Survey – M
Aegothelidae													
Australian Owlet-nightjar *Aegotheles cristatus*		Noted			Noted					Uncommon			
Apodidae													
Australian Swiftlet *Aerodramus terrareginae*								Noted				Noted	
White-throated Needletail *Hirundapus caudacutus*	Large numbers		Frequent on wing										Survey – F
Fork-tailed Swift *Apus pacificus*		Noted, Migrating					Wet						Incidental – F
Fregatidae													
Great / Lesser Frigatebird *Fregata sp*										Roosting		Roosting	Incidental – F
Ardeidae													
Nankeen Night-Heron *Nycticorax caledonicus*							Wet						
Threskiornithidae													
Australian White Ibis *Threskiornis molucca*												Noted	

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Accipitridae													
Eastern Osprey *Pandion cristatus*		Noted	Coastal				Wet			Nesting			Incidental – F
Black-shouldered Kite *Elanus axillaris*		Noted	Rarely seen										
Square-tailed Kite *Lophoictinia isura*		A few	Noted									Noted	Incidental – F
Black-breasted Buzzard *Hamirostra melanosternon*	Noted	Noted							Wet season				Incidental – F / M
Pacific Baza *Aviceda subcristata*									Dry season				Survey – F
White-bellied Sea Eagle *Haliaeetus leucogaster*		Noted	Near rivers				Wet		Wet	Noted			Incidental – F
Whistling Kite *Haliastur sphenurus*	Fairly plentiful	Plentiful	Numerous		Noted		Wet		Wet	Common			Survey – F / M
Brahminy Kite *Haliastur indus*							Wet			Noted		Noted	Survey – F / M
Black Kite *Milvus migrans*		Noted	Numerous						Wet / Dry	Common			Survey – F / M
Brown Goshawk *Accipiter fasciatus*	Fairly common	Common	Numerous				Wet		Wet / Dry				Survey – F / M
Collared Sparrowhawk *Accipiter cirrocephalus*	Fairly common	Noted	Noted						Wet				Survey – F / M
Grey Goshawk *Accipiter novaehollandiae*	Plentiful, breeding	Noted	Noted		Noted		Dry		Dry		Noted		Survey – F
Red Goshawk *Erythrotriorchis radiatus*									Noted				
Wedge-tailed Eagle *Aquila audax*		Noted	Widely distributed		Noted					Uncommon	Noted		Survey – F
Little Eagle *Hieraaetus morphhnoides*										Scarce, breeding			

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Falconidae													
Nankeen Kestrel *Falco cenchroides*		Occasional	Abundant		Noted					Common			Incidental – M
Brown Falcon *Falco berigora*		Common	Numerous		Noted		Wet / Dry		Dry	Common	Noted	Noted	Survey – F / M
Australian Hobby *Falco longipennis*		Noted	Plentiful										Survey – M
Peregrine Falcon *Falco peregrinus*							Wet						Survey – F
Otididae													
Australian Bustard *Ardeotis australis*		Plentiful	Nowhere common		Noted				Dry	Uncommon			Incidental – M
Burhinidae													
Bush Stone-curlew *Burhinus grallarius*		Plentiful	Numerous						Dry	Scarce	Noted		Survey – F / M
Charadriidae													
Masked Lapwing *Vanellus miles*										Noted			
Turnicidae													
Red-backed Button-quail *Turnix maculosus*		Occasional	Common							Uncommon			
Red-chested Button-quail *Turnix pyrrhothorax*													Survey – M
Cacatuidae													
Palm Cockatoo *Probosciger aterrimus*	Noted, Breeding	Common, breeding	Noted		Fairly common		Wet			Common	Noted	Noted	Survey – F
Red-tailed Black-Cockatoo *Calyptorhynchus banksii*			Numerous		Noted					Common		Noted	Incidental – F
Galah *Eolophus roseicapillus*			Noted										

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Little Corella *Cacatua sanguinea*		Noted											
Sulphur-crested Cockatoo *Cacatua galerita*	Plentiful, breeding	Common, breeding	Very common		Noted		Wet		Wet / Dry	Common	Noted	Noted	Survey – F / M
Psittacidae													
Rainbow Lorikeet *Trichoglossus haematodus*	Great numbers	Very numerous	Very numerous		Noted	Noted	Wet / Dry	Noted	Wet / Dry	Abundant	Common	Noted	Survey – F / M
Varied Lorikeet *Psitteuteles versicolor*		Large flocks	Numerous							Scarce			
Red-winged Parrot *Aprosmictus erythropterus*	Fairly plentiful	Common	Numerous		Common		Wet / Dry		Wet	Abundant	Common	Noted	Survey – F / M
Pale-headed Rosella *Platycerus adscitus*	Occasiona l	Noted	Numerous		Noted		Wet		Wet / Dry	Uncommon	Noted	Noted	Survey – F / M
Golden-shouldered Parrot *Psephotus chrysopterygius*			Open forest Watson R										
Cuculidae													
Pheasant Coucal *Centropus phasianinus*	Plentiful	Noted	Frequent		Noted		Wet / Dry	Noted	Wet	Common	Noted	Noted	Survey – F / M
Eastern Koel *Eudynamys orientalis*					Noted					Common, seasonally		Noted	Survey – F / M
Channel-billed Cuckoo *Scythrops novaehollandiae*		Occasional	Frequently heard		Noted								Incidental – F
Black-eared Cuckoo *Chalcites osculans*										Scarce			
Little Bronze-Cuckoo *Chalcites minutillus*	Fairly plentiful	Common					Wet			Scarce			Survey – F / M
Pallid Cuckoo *Cacomantis pallidus*		Common	Frequently noted										

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Fan-tailed Cuckoo *Cacomantis flabelliformis*		Plentiful								Uncommon			Survey – M
Brush Cuckoo *Cacomantis variolosus*		Common								Uncommon , seasonal	Noted		Survey – F / M
Oriental Cuckoo *Cuculus optatus*	Fairly plentiful	Occasional											Survey – F
Strigidae													
Rufous Owl *Ninox rufa*										Scarce	Noted		
Barking Owl *Ninox connivens*		Noted					Wet			Noted			
Southern Boobook *Ninox novaeseelandiae*	Numerous	Noted	Very common		Noted		Wet / Dry		Wet / Dry	Abundant		Noted	Survey – F
Tytonidae													
Masked Owl *Tyto novaehollandiae*										Scarce			
Eastern Barn Owl *Tyto javanica*		Noted	Noted										
Halcyonidae													
Laughing Kookaburra *Dacelo novaeguinae*		Common	Numerous		Noted		Wet / Dry		Wet / Dry	Common	Noted	Noted	Survey – F / M
Blue-winged Kookaburra *Dacelo leachii*	Fairly plentiful	Common	Found throughout				Wet / Dry	Noted	Wet / Dry	Abundant	Noted	Noted	Survey – F / M
Yellow-billed Kingfisher *Syma torotoro*		Noted, breeding									Noted		Survey – M
Forest Kingfisher *Todiramphus macleayii*	Noted, breeding	Numerous, breeding	Numerous		Noted		Wet / Dry	Noted	Dry	Common	Noted	Noted	Survey – F / M
Sacred Kingfisher *Todiramphus sanctus*			Noted				Dry			Uncommon			Survey – M
Azure Kingfisher *Ceyx azureus*													Incidental – F

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Meropidae													
Rainbow Bee-eater *Merops ornatus*	Noted, migrating	Common	Numerous		Noted			Noted	Wet / Dry	Abundant	Common	Noted	Survey – F / M
Coraciidae													
Dollarbird *Eurystomus orientalis*	Noted, migrating	Plentiful	Noted, seasonal		Noted		Wet	Noted		Uncommon, seasonal			Survey – F
Climacteridae													
Brown Treecreeper *Climacteris picumnus*		Scattered pairs	Numerous	Noted	Noted		Wet		Dry	Uncommon	Noted	Noted	Survey – F
Ptilonorhynchidae													
Great Bowerbird *Ptilonorhynchus nuchalis*	Noted	Common	Abundant		Noted		Wet				Noted		Survey – F / M
Maluridae													
Red-backed Fairy-wren *Malurus melanocephalus*	Common, breeding	Common	Numerous				Wet	Noted	Wet / Dry	Common	Common	Noted	Survey – F / M
Lovely Fairy Wren *Malurus annabilis*													Survey – M
Acanthizidae													
Weebill *Smicrornis brevirostris*		Occasional	Frequent	Noted					Wet			Noted	Survey – F / M
Fairy Gerygone *Gerygone palpebrosa*										Noted			Survey – M
White-throated Gerygone *Gerygone albogularis*		Noted	Noted					Noted		Scarce		Noted	
Pardalotidae													
Red-browed Pardalote *Pardalotus rubricatus*		Noted											Survey – F
Striated Pardalote *Pardalotus striatus*		Numerous	Numerous				Wet	Noted	Wet / Dry	Common		Noted	Survey – F / M

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Meliphagidae													
Yellow-spotted Honeyeater *Meliphaga notata*		Common						Noted		Noted	Noted		Survey – F / M
Graceful Honeyeater *Meliphaga gracilis*	Noted	Common					Wet	Noted		Noted	Noted		Survey – F / M
Yellow Honeyeater *Lichenostomus flavus*		Common	Numerous	Noted			Wet	Noted	Dry	Abundant	Noted	Noted	Survey – F / M
Yellow-tinted Honeyeater *Lichenostomus flavescens*				Noted						Scarce, seasonal			Survey – F
Brown-backed Honeyeater *Ramsayornis modestus*		Common							Dry	Uncommon		Noted	Survey – F / M
Bar-breasted Honeyeater *Ramsayornis fasciatus*		Noted								Uncommon			
Dusky Honeyeater *Myzomela obscura*		Common	Noted		Noted			Noted		Common			Survey – F / M
Red-headed Honeyeater *Myzomela erythrocephala*													Survey – M
Banded Honeyeater *Cissomela pectoralis*		Numerous	Numerous							Common		Noted	Survey – F / M
Brown Honeyeater *Lichmera indistincta*		Noted							Wet	Uncommon			Survey – M
White-streaked Honeyeater *Trichodere cockerelli*		Uncommon											Survey – M
Black-chinned Honeyeater *Melithreptus gularis*		Occasional											
White-throated Honeyeater *Melithreptus albogularis*	Very numerous	Common	Numerous		Noted	Noted	Wet	Noted	Dry	Abundant	Common	Noted	Survey – F / M
Blue-faced Honeyeater *Entomyzon cyanotis*	Plentiful	Numerous	Numerous				Wet		Wet	Abundant		Noted	Survey – F
Helmeted Friarbird *Philemon buceroides*	Plentiful	Numerous, Breeding	Numerous		Noted			Noted					Survey – F / M

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Silver-crowned Friarbird *Philemon argenticeps*	Plentiful	Numerous, Breeding		Noted					Wet / Dry	Common			
Noisy Friarbird *Philemon corniculatus*		Common					Wet			Scarce	Noted	Noted	Survey – F / M
Little Friarbird *Philemon citreogularis*	Numerous	Common	Most numerous						Wet / Dry	Common	Noted	Noted	Survey – F / M
Tawny-breasted Honeyeater *Xanthotis flaviventer*		Common											
Pomatostomidae													
Grey-crowned Babbler *Pomatostomus temporalis*	Small flocks	Small flocks	Numerous	Noted			Wet	Noted	Wet	Uncommon	Noted	Noted	Survey – F
Neosittidae													
Varied Sittella *Daphoenositta chrysoptera*	Noted	Frequent	Numerous				Wet		Wet / Dry	Uncommon	Noted	Noted	Survey – F
Campephagidae													
Black-faced Cuckoo-shrike *Coracina novaehollandiae*		Common seasonally	Abundant		Noted		Dry			Common	Noted	Noted	Survey – F / M
White-bellied Cuckoo-shrike *Coracina papuensis*	Fairly plentiful	Common	Very numerous		Noted		Wet	Noted	Wet / Dry	Common	Common	Noted	Survey – F / M
Cicadabird *Coracina tenuirostris*	Occasional	Common	Noted				Wet			Uncommon	Noted	Noted	Survey – F / M
White-winged Triller *Lalage sueurii*	Occasional	Common, seasonal							Dry			Noted	Survey – F / M
Varied Triller *Lalage leucomela*	Plentiful	Common	Numerous						Dry	Uncommon		Noted	Survey – F / M
Pachycephalidae													
Grey Whistler *Pachycephala simplex*												Noted	Survey – F
Rufous Whistler *Pachycephala rufiventris*	Noted	Common	Abundant		Noted		Wet	Noted	Dry	Common	Noted	Noted	Survey – F / M
Grey Shrike-thrush *Colluricincla harmonica*	Common	Common	Common				Wet	Noted		Uncommon	Common	Noted	Survey – F / M

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Oriolidae													
Australasian Figbird *Sphecotheres vieilloti*	Plentiful	Common			Noted								Survey – F / M
Yellow Oriole *Oriolus flavocinctus*	Plentiful	Common			Noted			Noted		Noted	Noted		Survey – F / M
Olive-backed Oriole *Oriolus sagittatus*		Common	Numerous						Dry	Uncommon		Noted	Survey – F / M
Artamidae													
White-breasted Woodswallow *Artamus leucorynchus*		Common, seasonal	Numerous, coastal		Winter visitor				Wet / Dry	Uncommon			Survey – F / M
Black-faced Woodswallow *Artamus cinereus*		Common											
Dusky Woodswallow *Artamus cyanopterus*										Scarce			
Little Woodswallow *Artamus minor*		Common							Wet				Survey – F
Black Butcherbird *Cracticus quoyi*	Breeding							Noted		Noted			Survey – M
Black-backed Butcherbird *Cracticus mentalis*	Noted, Breeding	Numerous	Abundant		Noted		Wet		Wet / Dry	Common	Noted	Noted	Survey – F / M
Pied Butcherbird *Cracticus nigrogularis*		Plentiful	Common						Wet	Common		Noted	Survey – F / M
Pied Currawong *Strepera graculina*			Central range							Scarce			
Dicruridae													
Spangled Drongo *Dicrurus bracteatus*	Common, migratory	Breeding	Noted		Noted		Wet / Dry	Noted		Common			Survey – F / M

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Rhipiduridae													
Grey Fantail *Rhipidura albiscapa*		Noted							Dry	Uncommon, seasonal		Noted	Survey – F
Northern Fantail *Rhipidura rufiventris*		Plentiful	Noted							Uncommon			Survey – F
Willie Wagtail *Rhipidura leucophrys*		Noted	Noted							Noted			
Corvidae													
Torresian Crow *Corvus orru*		Numerous	Abundant		Noted	Noted	Wet		Wet / Dry	Abundant		Noted	Survey – F / M
Monarchidae													
Leaden Flycatcher *Myiagra rubecula*	Fairly plentiful	Common, Breeding	Numerous		Noted		Wet	Noted	Dry	Common	Noted	Noted	Survey – F / M
Magpie-lark *Grallina cyanoleuca*		Common seasonally	Numerous		Winter visitor		Dry		Wet			Noted	Survey – F
Petroicidae													
Jacky Winter *Microeca fascinans*			Noted						Wet				
Lemon-bellied Flycatcher *Microeca flavigaster*	Fairly plentiful	Common	Common				Wet	Noted	Dry	Common	Common	Noted	Survey – F / M
Cisticolidae													
Golden-headed Cisticola *Cisticola exilis*	Plentiful	Common	Numerous										Survey – M
Hirundinidae													
Fairy Martin *Petrochelidon ariel*		Noted								Common			
Tree Martin *Petrochelidon nigricans*		Noted								Common			Incidental – F

Appendix 2 Birds of the open eucalypt forests and woodlands of Cape York Peninsula – continued

Species	1	2	3	4	5	6	7	8	9	10	11	12	13
Nectaridiidae													
Mistletoebird *Dicaeum hirundinaceum*	Plentiful	Common	Numerous		Noted		Wet	Noted	Wet / Dry	Common	Common	Noted	Survey – F / M
Olive-backed Sunbird *Nectarinia jugularis*		Common	Numerous, coastal							Common near coast			Survey – F / M
Estrildidae													
Double-barred Finch *Taeniopygia bichenovii*		Common	Common							Uncommon			
Black-throated Finch *Poephila cincta*		Common	Well distributed						Dry	Uncommon			
Crimson Finch *Neochmia phaeton*		Common	Not uncommon										
Star Finch *Neochmia ruficauda*												Noted	
Red-browed Finch *Neochmia temporalis*		Noted	Not common		Noted					Vine forest fringes			Survey – M
Gouldian Finch *Erythrura gouldiae*		Common											
Chestnut-breasted Mannikin *Lonchura castaneothorax*		Common			Noted								Survey – M
Motacillidae													
Australasian Pipit *Anthus novaeseelandiae*		Noted	Not uncommon										Survey – M

Appendix 3 Conservation significance of the birds of CYP eucalypt forest

Nomenclature follows (Christidis and Boles, 2008). Where different names are used in legislation or by Qld EPA this is noted in the comments on distribution, residency and conservation status. Hollow users are indicated. Sources (Higgins and Davies, 1996; Marchant and Higgins, 1990; Garnett and Crowley, 2000a; Clayton *et al.*, 2006; Gibbons and Lindenmayer, 2002; Griffioen and Clarke, 2002; Marchant and Higgins, 1993; Higgins, 1999; Higgins *et al.*, 2001; Higgins and Peter, 2002; Higgins *et al.*, 2006).

Species	**Australian distribution and conservation status**
Emu *Dromaius novaehollandiae*	Monotypic. Endemic to Australia. Resident where conditions permit, otherwise dispersive. Widespread in NSW and Qld but generally absent from rainforest and from east coast NSW.
Southern Cassowary *Casuarius casuarius*	Polytypic. Extralimital. Sedentary but nomadic within large home range. CYP south to Townsville. Normally in rainforest but recorded in open woodland. Two populations recognised. Northern population - Vuln (Qld) & End (Com). Southern population - End (Qld & Com).
Australian Brush-turkey *Alectura lathami*	Polytypic. Endemic to Australia. Sedentary. CYP, north-east and eastern Australia south to Hawkesbury R.
Orange-footed Scrubfowl *Megapodius reinwardt*	Polytypic. Extralimital. Sedentary. Kimberley, NT, CYP south to Mitchell R. in west and Yeppoon in east.
Brown Quail *Coturnix ypsilonophora*	Polytypic. Extralimital. Resident. North, east, southeast and southwest of mainland Australia. Vuln (SA)
King Quail *Excalfactoria chinensis*	Polytypic. Extralimital. Probably resident with some local movement. Fragmented distribution in north and east Australia. In Qld = *Coturnix chinensis*. Thr (Vic), End (SA)
Radjah Shelduck *Tadorna radjah*	Polytypic. Extralimital. Northern Australia. Local movements between breeding territories and coast. Hollow user. Rare (Qld)
Common Bronzewing *Phaps chalcoptera*	Monotypic. Endemic to Australia. Widespread. Absent western, northern and central Cape York Peninsula. Usually sedentary or resident but at least part of population moves regularly
Crested Pigeon *Ocyphaps lophotes*	Polytypic. Endemic to Australia. Throughout much of Australia, sparsely distributed on CYP. Sedentary, resident or locally dispersive.
Squatter Pigeon *Geophaps scripta*	Polytypic. Endemic to Australia. Locally dispersive. Widespread in north-east and eastern Queensland. In Qld two subspecies are recognised. Northern subspecies = *G.s.peninsulae,* Southern subspecies = *G.s.scripta. G.s.scripta* Vuln (Qld, NSW & Com), Near Thr (APAB)
Diamond Dove *Geopelia cuneata*	Monotypic. Endemic to Australia. Widespread throughout Australia and in all regions of Queensland except Cape York Peninsula and east of Great Dividing Range. Dispersive. Hollow user. Thr (Vic)
Peaceful Dove *Geopelia striata*	Polytypic. Extralimital. Widespread in northern and eastern mainland Australia. Sedentary
Bar-shouldered Dove *Geopelia humeralis*	Polytypic. Extralimital. Northern and eastern Australia. Sedentary.
Pied Imperial-Pigeon *Ducula bicolor*	Polytypic. Extralimital. Northern Australia, widespread in CYP and Torres Strait. Intercontinental migrant, NT and Qld populations migratory. Breed colonially on Qld islands and solitarily at Weipa.
Tawny Frogmouth *Podargus strigoides*	Polytypic. Endemic to Australia. Widespread. Sedentary. Was Freckled Frogmouth.
Papuan Frogmouth *Podargus papuensis*	Monotypic. Extralimital. Restricted to northeast Qld in Australia. Sedentary.
Marbled Frogmouth *Podargus ocellatus*	Polytypic. Extralimital. In Australia, restricted to north-east Cape York Peninsula, south-east Queensland and north-east NSW. Sedentary. Vuln (Qld & NSW).
Large-tailed Nightjar *Caprimulgus macrurus*	Polytypic. Extralimital. Northern and eastern Australia. Sedentary.
Australian Owlet-nightjar *Aegotheles cristatus*	Polytypic. Extralimital. Widespread in Australia. Sedentary. Hollow user.

Appendix 3 Conservation significance of the birds of CYP eucalypt forest - continued

Species	**Australian distribution and conservation status**
Australian Swiftlet *Aerodramus terrareginae*	Polytypic. North-east Queensland. Sedentary. In Qld = White-rumped Swiftlet = *Collocalia spodiopygius.* Rare (Qld)
White-throated Needletail *Hirundapus caudacutus*	Polytypic. Extralimital. Widespread in eastern and south-eastern Australia. Breeds in northern hemisphere, migrates to Australasia in non-breeding season.
Fork-tailed Swift *Apus pacificus*	Polytypic. Extralimital. Widespread non-breeding visitor to Australia. Breeds in northern hemisphere, migrates through SE Asia to Australia.
Great Frigatebird *Fregata minor*	Monotypic. Tropical Indian and Pacific Oceans. Listed as migratory species and as marine species under the EPBC Act. Also listed in CAMBA and JAMBA. Roost in large numbers in *Eucalyptus tetrodonta* open forest at Weipa.
Lesser Frigatebird *Fregata ariel*	Polytypic. Tropical Indian and Pacific Oceans. Listed as migratory species and as marine species under the EPBC Act. Also listed in CAMBA and JAMBA. Roost in large numbers in *Eucalyptus tetrodonta* open forest at Weipa.
Nankeen Night-Heron *Nycticorax caledonicus*	Polytypic. Extralimital. Widespread in northern, eastern and south-western Australia. Nomadic.
Australian White Ibis *Threskiornis molucca*	Monotypic. Extralimital. Throughout northern, eastern and south-western Australia. Adults sedentary.
Osprey *Pandion cristatus*	Polytypic. Extralimital. Sedentary. Coastal northern and eastern Australia from Broome to south coast NSW. In Qld = *Pandion haliaetus.* Rare (SA) Vuln (NSW)
Black-shouldered Kite *Elanus axillaris*	Monotypic. Endemic to Australia. Dispersive in many parts of range. Occurs throughout Australia, irregular visitor to Torres Strait and CYP.
Square-tailed Kite *Lophoictinia isura*	Monotypic. Endemic to Australia. Widespread but sparse throughout mainland. Migratory through much of range. Breeds throughout range. Rare (Qld), Vuln (NSW, SA), Thr (Vic), Near threat (NT).
Black-breasted Buzzard *Hamirostra melanosternon*	Monotypic. Endemic to Australia. Continuous across much of northern and central Australia. Partially migratory in northern Australia. Seasonal movements related to rainfall. Depart north during rainfall. Vuln (NSW), Rare (SA).
Pacific Baza *Aviceda subcristata*	Polytypic. Extralimital. Northern and eastern Australia. Breeding and movements poorly understood.
White-bellied Sea Eagle *Haliaeetus leucogaster*	Monotypic. Extralimital. Most of coastal Australia, occasionally inland along rivers. Resident, established pairs sedentary. Thr (Vic), Vuln (Tas & SA).
Whistling Kite *Haliastur sphenurus*	Monotypic. Extralimital. Common and widespread throughout mainland Australia. Partly migratory, dispersive, mostly resident in northern and western Australia.
Brahminy Kite *Haliastur indus*	Polytypic. Extralimital. Continuous along northwest, northern and east coasts. Mostly resident.
Black Kite *Milvus migrans*	Polytypic. Extralimital. Throughout mainland Australia, common in Queensland. Migrate south during northern wet season, nomadic, dispersing in response to food availability. Dry season visitor to CYP.
Brown Goshawk *Accipiter fasciatus*	Polytypic. Extralimital. Widespread throughout Australia. Partly migratory over much of range with movements related to food availability. Adult pairs sedentary.
Collared Sparrowhawk *Accipiter cirrhocephalus*	Polytypic. Extralimital. Widespread but uncommon throughout mainland Australia.
Grey / White Goshawk *Accipiter novaehollandiae*	Polytypic. Extralimital. Northern Australia, patchily distributed in east and southeast mainland. Breeds CYP. Resident, established pairs sedentary. Rare (Qld & SA), Thr (ACT), End (Tas)
Red Goshawk *Erythrotriorchis radiatus*	Monotypic. Endemic to Australia. Continuously but sparsely distributed in north-west, northern and eastern Australia. Sedentary. Wooded forested lands of tropical Australia. Prefer forest with mosaic of vegetation types, large bird populations and permanent water. End (Qld & NSW), Vul (NT & Com), Rare (WA), Vuln (APAB), High (Q-BoT).
Wedge-tailed Eagle *Aquila audax*	Polytypic. Extralimital. Widespread throughout Australia. Established pairs resident.
Little Eagle *Hieraaetus morphnoides*	Polytypic. Extralimital. Widespread in wooded and forested land in mainland Australia, less common near coast. Some populations resident, others migratory or dispersive.
Australian Kestrel *Falco cenchroides*	Polytypic. Extralimital. Widespread and common through mainland Australia. Partly migratory, move south during northern wet season. Hollow user.

Appendix 3 Conservation significance of the birds of CYP eucalypt forest - continued

Species	**Australian distribution and conservation status**
Brown Falcon *Falco berigora*	Polytypic. Extralimital. Widespread in Australia. Established pairs resident through much of range, juveniles dispersive. Migratory, move south during northern wet season. Irruptions may be regulated by dry periods.
Australian Hobby *Falco longipennis*	Polytypic. Extralimital. Throughout mainland Australia. Partly migratory, resident in many regions of Australia.
Peregrine Falcon *Falco peregrinus*	Polytypic. Extralimital. Sporadic records for CYP. Sedentary within breeding range. Hollow user. Rare (S.A.)
Australian Bustard *Ardeotis australis*	Monotypic. Mainly inland northern Australia. Widespread in all regions of Queensland. Breeding widespread in northern Australia. Dispersive. Irregular widespread movements over long distances in response to food availability. Becoming scarce, significant range contraction. End (NSW) Thr (Vic), Vul (SA), Near Thr (NT & APAB)
Bush Stone-curlew *Burhinus grallarius*	Monotypic. Widespread in north and north-eastern Australia. Absent or scattered inland. Sedentary. Territorial when breeding, some local movement when not breeding. Range contracting. End (NSW), Vuln (SA), Near Thr (NT & APAB)
Masked Lapwing *Vanellus miles*	Polytypic. Extralimital. Widespread in eastern Australia. Resident. In Qld two subspecies are recognised. Northern subspecies = *V.m. miles* and Southern subspecies = *V.m.novaehollandiae*
Red-backed Button-quail *Turnix maculosus*	Polytypic. Extralimital. Disjunct populations in northern and eastern Australia. From southwest Gulf of Carpentaria north to Torres Strait and south along east coast to southeast Qld. Some populations sedentary, others dispersive.
Red-chested Button-quail *Turnix pyrrhothorax*	Monotypic. Endemic to northern, eastern and inland Australia. Generally uncommon. Rare (SA)
Palm Cockatoo *Probosciger aterrimus*	Polytypic. Extralimital. Australian distribution confined to Cape York Peninsula. Probably sedentary. Hollow user. Rare (Qld), Near Thr (APAB)
Red-tailed Black Cockatoo *Calyptorhynchus banksii*	Polytypic. Endemic to Australia. Dispersive. Mainly northern, north-eastern and south-west Australia. Five subspecies in total with disjunct distributions. Hollow user. In Qld two subsp are recognised *C.b. banksii* and *C.b.samueli*. *C.banksii* is Vuln (NSW) and Thr (Vic), *C.b.samueli*. is Rare (SA) and Near Thr (NT), *C.b.graptogyne* from southern Australia is End (Com), Vul (NSW) & Thr (Vic)
Galah *Eolophus roseicapillus*	Polytypic. Endemic to Australia. Resident. Widespread throughout Australia. Hollow user. In Qld = *Cacatua roseicapilla*
Little Corella *Cacatua sanguinea*	Polytypic. Extralimital. Widespread in arid and semi-arid Australia, also coastal and sub-coastal south-western Cape York Peninsula. Hollow user.
Sulphur-crested Cockatoo *Cacatua galerita*	Polytypic. Extralimital. Widespread in northern, eastern and south-east Australia. Varied movements considered resident. Hollow user.
Rainbow Lorikeet *Trichoglossus haematodus*	Polytypic. Extralimital. Nomadic. In Qld two subsp are recognised Rainbow Lorikeet = *T.h. moluccanus* and Red-collared Lorikeet = *T.h. rubritorquis*. Subsp *T.h.moluccans* widespread from Cape York Peninsula to Tasmania and west to Kangaroo Island and Eyre Peninsula. Hollow user.
Varied Lorikeet *Psitteuteles versicolor*	Monotypic. Endemic to northern Australia from Kimberley through the north to central Qld and CYP to Archer River. Nomadic. Hollow user.
Red-winged Parrot *Aprosmictus erythropterus*	Polytypic. Extralimital. Resident or locally nomadic. Northern and eastern Australia from Kimberley to central NSW. Hollow user. Rare (SA).
Pale-headed Rosella *Platycercus adscitus*	Polytypic. Endemic to north-east and eastern Australia. In Qld two subsp are recognised. Northern form = *P.a.adscitus* and Southern form = *P.a.palliceps.* Northern form from CYP south to about Cooktown. Hollow user.
Golden-shouldered Parrot *Psephotus chrysopterygius*	Monotypic. Endemic to CYP. Sedentary. End (Qld &Com). Also listed as Critical - Qld BoT
Pheasant Coucal *Centropus phasianus*	Polytypic. Extralimital. Resident. Widespread in northern and eastern Australia from Pilbara to south-east NSW.
Koel *Eudynamys orientalis*	Polytypic. Extralimital. Intercontinental migrant, breeds in Australia and migrates to New Guinea. Widespread in northern and eastern Australia from Kimberley to south-east NSW. In Queensland = *Eudynamyx scolopacea*.
Channel-billed Cuckoo *Scythrops novaehollandiae*	Polytypic. Extralimital. Intercontinental migrant, breeds in Australia and migrates to New Guinea. Widespread in northern and eastern Australia. Near Thr (NT)

Appendix 3 Conservation significance of the birds of CYP eucalypt forest - continued

Species	Australian distribution and conservation status
Black-eared Cuckoo *Chalcites osculans*	Monotypic. Extralimital. Widespread in mainland Australia. Movements not well understood. In Qld = *Chrysococcyx osculans*.
Little Bronze-cuckoo *Chrysococcyx minutillus*	Polytypic. Extralimital. Partly migratory. In Australia widespread from Kimberley to north-eastern NSW. In Qld two separate species are recognised *Chrysococcyx minutillus* and Gould's Bronze-cuckoo *Chrysococcyx russatus.*
Pallid Cuckoo *Cacomantis pallidus*	Monotypic. Extralimital. Migratory or partly migratory. Widespread in most regions of Australia. In Qld = *Cuculus pallidus*.
Fan-tailed Cuckoo *Cacomantis flabelliformis*	Polytypic. Extralimital. Migratory. Widespread in CYP and eastern Qld.
Brush Cuckoo *Cacomantis variolosus*	Polytypic. Extralimital. Intercontinental migrant. In Australia occurs from south-west Kimberley around north and east to Victoria.
Oriental Cuckoo *Cuculus optatus*	Polytypic. Extralimital. Partly migratory and partly resident or sedentary. Breeds northern hemisphere, winters in southeast Asia, New Guinea and coastal northern Australia. In Australia mostly occurs around north and east coast from Kimberley to near Sydney. In Qld = *Cuculus saturatus*. Near Thr (NT)
Rufous Owl *Ninox rufa*	Polytypic. Extralimital. Sedentary. Disjunct populations from Kimberley in west to Rockhampton in east. Hollow user. In Qld two subsp are recognised. Cape York Peninsula subsp = *N.r.meesi* and Southern subsp = *N.r.queenslandica*. *N.r.meesi* is Rare (Qld), *N.r.queenslandica* Vuln (Qld), Near Thr (APAB)
Barking Owl *Ninox connivens*	Polytypic. Extralimital. Sedentary. In Australia, widespread in south-west, north and east. Hollow user. Vuln (NSW), Thr (Vic), Rare (SA).
Southern Boobook *Ninox novaeseelandiae*	Polytypic. Extralimital. Considered resident in some areas, partly migratory. In Australia widespread throughout mainland. Hollow user.
Masked Owl *Tyto novaehollandiae*	Polytypic. Extralimital. Sedentary. Sparse records in the north, most records from coastal south-eastern Australia. Hollow user. In Qld two subsp are recognised. Northern subsp = *T.n.kimberli* and Southern subsp = *T.n.novaehollandiae*. *T.n.kimberli* occurs in north-east Qld, northern NT and north-east W.A. Vuln (Qld, Com, NSW & NT), End (Tas & SA), Thr (Vic), Near Thr (APAB)
Eastern Barn Owl *Tyto javanica*	Polytypic. Extralimital. Widespread on Australian mainland. Frequent hollow user. In Qld = *Tyto alba*.
Laughing Kookaburra *Dacelo novaeguinae*	Polytypic. Endemic to eastern mainland Australia. Sedentary. Widespread from western CYP to Eyre Peninsula S.A. Hollow user.
Blue-winged Kookaburra *Dacelo leachii*	Polytypic. Extralimital. Resident. Widespread in northern Australia. Hollow user.
Yellow-billed Kingfisher *Syma torotoro*	Polytypic. Extralimital. Sedentary. In Australia, restricted to CYP. Hollow user.
Forest Kingfisher *Todiramphus macleayii*	Polytypic. Extralimital. Sedentary, resident, partly migratory. Widespread in Top End and eastern Qld. Hollow user.
Sacred Kingfisher *Todiramphus sanctus*	Polytypic. Extralimital. Migratory, move south during northern wet season. Breeds in Australia in austral winter. In Australia, widespread in east, west and north. Hollow user.
Rainbow Bee-eater *Merops ornatus*	Monotypic. Extralimital. Resident in north, migratory in south. Widespread on mainland Australia.
Dollarbird *Eurystomus orientalis*	Polytypic. Extralimital. Widespread in northern and eastern Australia. Intercontinental migrant, breeds in Australia in Austral summer and spends non-breeding season in New Guinea. Hollow user.
Brown Treecreeper *Climacteris picumnus*	Polytypic. Endemic to eastern Australia. Sedentary. Hollow user. In Qld three subspecies are recognised Cape York Peninsula subspecies = *C.p.melanotus,* inland eastern subspecies = *C.p.picumnus* and south-eastern subspecies = *C.p.victoriae*. Vuln (NSW & ACT), Near Thr (APAB).
Great Bowerbird *Ptilonorhynchus nuchalis*	Polytypic. Endemic to tropical northern Australia where widespread. Resident or sedentary. In Qld = *Chlamydera nuchalis*
Red-backed Fairy-wren *Malurus melanocephalus*	Polytypic. Endemic to northern and eastern Australia. Resident or sedentary.
Lovely Fairy-wren *Malurus annabilis*	Monotypic. Endemic to north-east Queensland. Resident or sedentary.

Appendix 3 Conservation significance of the birds of CYP eucalypt forest - continued

Species	**Australian distribution and conservation status**
Weebill *Smicrornis brevirostris*	Polytypic. Endemic to mainland Australia. Resident or sedentary with some local movements. Widespread throughout Australian mainland.
Fairy Gerygone *Gerygone palpebrosa*	Polytypic. Extralimital. Possibly resident or sedentary. In Australia confined to northern and eastern Queensland.
White-throated Gerygone *Gerygone albogularis*	Polytypic. Extralimital. Migratory, over-winter in northern tropics and moving south during northern wet season. In Australia, Kimberley, Top end, northern and eastern Qld, eastern NSW and Victoria In Qld = *Gerygone olivacea.* Rare (SA).
Red-browed Pardalote *Pardalotus rubricatus*	Polytypic. Endemic to Australia. Sedentary. Widespread but scattered distributions in Queensland, NT and W.A north of Carnarvon, also north-western NSW.
Striated Pardalote *Pardalotus striatus*	Polytypic. Endemic to Australia. Resident, migratory or dispersive, varying with population. Widespread throughout Australian mainland and Tasmania. Hollow user.
Yellow-spotted Honeyeater *Meliphaga notata*	Polytypic. Endemic to north-east Qld. Resident.
Graceful Honeyeater *Meliphaga gracilis*	Polytypic. Extralimital. Resident or sedentary. In Australia confined to north-east Queensland.
Yellow Honeyeater *Lichenostomus flavus*	Polytypic. Endemic to north-east Qld. Resident.
Yellow-tinted Honeyeater *Lichenostomus flavescens*	Polytypic. Extralimital. Widespread in coastal and sub-coastal northern Australia. Sedentary, local movements.
Brown-backed Honeyeater *Ramsayornis modestus*	Monotypic. Extralimital. Movements not well known. Confined in Australia to north-east Qld.
Bar-breasted Honeyeater *Ramsayornis fasciatus*	Monotypic. Endemic to northern Australia. Resident or nomadic. Widespread in northern Qld, NT and Kimberley.
Dusky Honeyeater *Myzomela obscura*	Polytypic. Extralimital. Resident with some nomadic movements. In Australia occurs in Top End, coastal and subcoastal Qld south to Moreton Bay, north to CYP and southern Torres Strait.
Red-headed Honeyeater *Myzomela erythrocephala*	Polytypic. Extralimital. Resident, nomadic or migratory. Coastal areas of Kimberley, coastal NT and coastal CYP.
Banded Honeyeater *Cissomela pectoralis*	Monotypic. Endemic to northern Australia. Nomadic, seasonal movements. Kimberley, Top End and northern Queensland. In Qld = *Certhionyx pectoralis.*
Brown Honeyeater *Lichmera indistincta*	Polytypic. Extralimital. Widespread in Australia. Resident, nomadic.
White-streaked Honeyeater *Trichodere cockerelli*	Monotypic. Endemic to Cape York Peninsula. Mostly confined to northern CYP. Locally nomadic in response to flowering.
Black-chinned Honeyeater *Melithreptus gularis*	Polytypic. Endemic to mainland Australia. Widespread but scattered in northern and eastern Australia. Migratory. In Qld two subspecies are recognised Golden-backed Honeyeater = *M.g.laetior* and Black-chinned Honeyeater = *M.g.gularis.* Both subsp are Rare (Qld), Vuln (NSW & SA).
White-throated Honeyeater *Melithreptus albogularis*	Polytypic. Extralimital. Widespread in northern and north-eastern Australia. Resident.
Blue-faced Honeyeater *Entomyzon cyanotis*	Polytypic. Widespread in northern and eastern Australia. Sedentary, resident or locally nomadic. Rare (SA).
Helmeted Friarbird *Philemon buceroides*	Polytypic. Extralimital. In Australia occurs in Top End and north-east Queensland.
Silver-crowned Friarbird *Philemon argenticeps*	Polytypic. Endemic to northern Australia. Widespread in Kimberly, Top End, Gulf country and northern and eastern CYP.
Noisy Friarbird *Philemon corniculatus*	Polytypic. Extralimital. Widespread in eastern and south-eastern Australia. Movements complex, migratory.
Little Friarbird *Philemon citreogularis*	Polytypic. Extralimital. Resident, nomadic and partly migratory. Widespread in northern and eastern Australia. Rare (SA).
Tawny-breasted Honeyeater *Xanthotis flaviventer*	Polytypic. Extralimital. Resident. In Australia confined to CYP.

Appendix 3 Conservation significance of the birds of CYP eucalypt forest - continued

Species	**Australian distribution and conservation status**
Grey-crowned Babbler *Pomatostomus temporalis*	Polytypic. Extralimital. Sedentary. In Qld two subsp are recognised North-western subsp = *P.t.rubeculus* and Eastern subsp = *P.t.temporalis.* It occurs in eastern Australia from CYP to north-east NSW then south and west through central NSW and Vic to south east SA. Vuln (NSW), Thr (Vic), Rare (SA), Near Thr (APAB).
Varied Sittella *Daphoenositta chrysoptera*	Polytypic. Endemic to Australia. Sedentary, resident. Widespread in Australian mainland.
Black-faced Cuckoo-shrike *Coracina novaehollandiae*	Polytypic. Extralimital. Partly migratory and partly resident or sedentary. Widespread in Australia.
White-bellied Cuckoo-shrike *Coracina papuensis*	Polytypic. Extralimital. Movements not well known. Widespread in northern and eastern Australia. Rare (SA).
Cicadabird *Coracina tenuirostris*	Polytypic. Extralimital. Partly resident and partly migratory. Intercontinental migrant. Widespread in northern and eastern Australia.
White-winged Triller *Lalage sueurii*	Polytypic. Extralimital. Widespread in Australia. Partial migrant with episodic or irregular appearance at many sites.
Varied Triller *Lalage leucomela*	Polytypic. Extralimital. Disjunct distribution including Kimberley, Top End northern and eastern Australia.
Grey Whistler *Pachycephala simplex*	Polytypic. Extralimital. Resident or sedentary. Australian distribution confined to coastal Top End, north-eastern CYP and north eastern Qld from Cooktown to Paluma. In Qld = *Pachycephala simplex peninsulae*
Rufous Whistler *Pachycephala rufiventris*	Polytypic. Extralimital. Widespread on Australian mainland. Resident and partly migratory.
Grey Shrike-thrush *Colluricincla harmonica*	Polytypic. Extralimital. Widespread throughout Australia. Sedentary. Hollow user.
Australasian Figbird *Sphechotheres vieilloti*	Polytypic. Extralimital. Mainly resident or sedentary. Scattered in north, widespread in eastern Australia. In Qld = *Sphecotheres viridis*
Yellow Oriole *Oriolus flavocinctus*	Polytypic. Extralimital. Resident. Widespread in northern Australia with three disjunct populations in Kimberly, Top End and northern Qld.
Olive-backed Oriole *Oriolus sagittatus*	Polytypic. Extralimital. Partly migratory and partly resident. Widespread in northern and eastern Australia.
White-breasted Woodswallow *Artamus leucorynchus*	Polytypic. Extralimital. Movements complex, varying geographically, over-winter in tropical north and move south during northern wet season. Widespread in northern and eastern Australia, scattered in central Australia. Hollow user. Rare (SA).
Black-faced Woodswallow *Artamus cinereus*	Polytypic. Extralimital. Mainly resident or sedentary. Widespread in Australia. Hollow user. In Qld three subsp are recognised*. A.c. albiventris, A.c. melanops* and the CYP subspecies *A.c. normani. A.c.normani* is listed as High on the Qld BoT
Dusky Woodswallow *Artamus cyanopterus*	Polytypic. Endemic to Australia. Partly migratory and partly resident. Widespread in eastern, southern and south-western Australia. Hollow user.
Little Woodswallow *Artamus minor*	Polytypic. Endemic to Australia. Partly migratory and partly resident, move south during northern wet season. Widespread on Australian mainland north of 30 S. Hollow user.
Black Butcherbird *Cracticus quoyi*	Polytypic. Extralimital. Resident possibly sedentary. Northern and north-eastern Australia. In Qld two subsp are recognised *C.q. jardini* and *C.q. rufescens. C.q.jardini* occurs on coastal CYP
Black-backed Butcherbird *Cracticus mentalis*	Polytypic. Extralimital. In Australia confined to Cape York Peninsula. Considered sedentary.
Pied Butcherbird *Cracticus nigrogularis*	Polytypic. Endemic to mainland Australia where widespread. Resident.
Pied Currawong *Strepera graculina*	Polytypic. Endemic to Australia. Widespread on eastern mainland. Partly resident. In Qld three subspecies are recognised. Eastern = *S.g. graculina. Wet Tropics = S.f.robinsoni* and CYP = *S.g. magnirostris.*
Spangled Drongo *Dicrurus bracteatus*	Polytypic. Extralimital. Partly resident or sedentary and partly migratory. Widespread in northern and eastern Australia. In Qld two subsp are recognised CYP = *D.b.atrabectus* and Eastern Australia = *D.b.bracteatus.*

Appendix 3 Conservation significance of the birds of CYP eucalypt forest - continued

Species	**Australian distribution and conservation status**
Grey Fantail *Rhipidura albiscapa*	Polytypic. Extralimital. Movements complex, migratory. Widespread in Australia. In Qld = *Rhipidura fuliginosa.*
Northern Fantail *Rhipidura rufiventris*	Polytypic. Extralimital. Movements poorly known. Widespread in northern Australia.
Willie Wagtail *Rhipidura leucophrys*	Polytypic. Extralimital. Largely sedentary. Widespread throughout mainland Australia. In Qld two subsp are recognised Northern = *R.l. picata* and Southern = *R.l. leucophrys*
Torresian Crow *Corvus orru*	Polytypic. Extralimital. Geographical differences in movements. Widespread in most areas of mainland except extreme south-east and south-west.
Leaden Flycatcher *Myiagra rubecula*	Polytypic. Extralimital. Partly resident, partly migratory, resident in northern Australia. Widespread in northern and eastern Australia.
Magpie-lark *Grallina cyanoleuca*	Polytypic. Extralimital. Partly sedentary and resident. Widespread on mainland Australia
Jacky Winter *Microeca fascinans*	Polytypic. Widespread in Australia. Resident, sedentary, seasonal visitor and nomadic.
Lemon-bellied Flycatcher *Microeca flavigaster*	Polytypic. Extralimital. Confined to northern and north-eastern Australia. Resident or sedentary.
Golden-headed Cisticola *Cisticola exilis*	Polytypic. Extralimital. Probably sedentary, some local movements. Widespread in northern, eastern and south-eastern Australia. Rare (SA).
Fairy Martin *Petrochelidon ariel*	Monotypic. Widespread on mainland Australia. Migratory, part migratory and resident. Hollow user. In Qld = *Hirundo ariel*
Tree Martin *Petrochelidon nigricans*	Polytypic. Extralimital. Widespread in Australia. Largely migratory in eastern Australia, southern populations moving north to winter in northern Australia. Hollow user. In Qld = *Hirundo nigricans*
Mistletoebird *Dicaeum hirundinaceum*	Polytypic. Extralimital. Widespread in Australia although sparsely scattered in some areas. Largely resident with some local dispersion.
Olive-backed Sunbird *Nectarinia jugularis*	Polytypic. Extralimital. In Australia mostly confined to northern and eastern Queensland. Resident or sedentary.
Double-barred Finch *Taeniopygia bichenovii*	Polytypic. Endemic to northern and eastern Australia where widespread. Partly migratory, partly resident.
Black-throated Finch *Poephila cincta*	Polytypic. Endemic to Australia. Now confined to northern Queensland. Significant range contraction. Probably sedentary or resident. Hollow user. In Qld two subsp are recognised Black-rumped subsp = *P.c.atropygialis and* White-rumped subsp = *P.c.cincta*. *P.c.cincta* is Vuln (Qld), End (Com & NT)
Crimson Finch *Neochmia phaeton*	Polytypic. Extralimital. Widespread in northern Australia. On CYP occur between Archer River and Kowanyama. Hollow user. In Qld two subsp are recognised. White-bellied subsp = *N.p.evangelinae* and *N.p.phaeton*. Both subsp are Vuln (Qld). *N.p.evangelinae* End (Com) and is listed High Qld BoT
Star Finch *Neochmia ruficauda*	Polytypic. Endemic to Australia. Sedentary or resident with some local dispersal. Scattered distribution from Pilbara to Top End to northern Queensland. Significant range contraction has occurred. In Qld two subspecies are recognised. Northern subsp = *N.r.clarescens* and Southern subsp = *N.r.ruficauda*. *N.r.clarescens* End (Com), Near Thr (NT) and High (Qld BoT), *N.r.ruficauda* is End (Qld) & Crit End (Com).
Red-browed Finch *Neochmia temporalis*	Polytypic. Endemic to eastern mainland Australia where widespread from CYP to Kangaroo Island. Mainly sedentary or resident.
Gouldian Finch *Erythrura gouldiae*	Monotypic. Endemic to northern Australia. Apparently resident and largely sedentary. Hollow user. End (Qld, NT & Com), Rare (WA).
Chestnut-breasted Mannikin *Lonchura castaneothorax*	Polytypic. Widespread but scattered in northern and eastern Australia. Resident.
Australasian Pipit *Anthus novaeseelandiae*	Polytypic. Extralimital. Widespread in Australia and New Zealand. Resident or sedentary.

Appendix 4 Mine rehabilitation seed mix used at Weipa

	Species name	Age of rehabilitation site in 2006																			
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	16	18	20	21	22	23
Species sown in mine rehabilitation that are potentially found in the pre-mining native forest plant community	*Corymbia polycarpa*				■	■		■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Acacia aulacocarpa																■			■	
	Acacia crassicarpa			■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Acacia leptocarpa				■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	
	Acacia platycarpa	■	■	■	■	■		■	■	■	■	■	■	■	■	■	■				■
	Acacia rothii		■	■			■	■	■	■	■	■	■	■	■	■	■	■			
	Adenanthera abrosperma																	■			
	Alphitonia excelsa																	■	■	■	■
	Alstonia actinophylla						■	■	■	■			■	■	■	■	■	■	■		
	Atalaya variifolia																	■	■		
	Corymbia confertiflora			■	■		■	■	■	■											
	Corymbia stockerii	■	■			■		■	■		■	■	■	■	■	■	■	■			
	Deplanchea tetraphylla																	■	■		
	Erythrophleum chlorostachys	■	■	■	■	■	■	■	■	■	■	■	■	■				■		■	■
	Eucalyptus cullenii	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	
	Eucalyptus leptophleba	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Corymbia nesophila	■	■	■	■	■	■	■	■			■	■	■	■	■	■	■			■
	Eucalyptus tetrodonta	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Grevillea glauca			■		■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Grevillea parallela																		■		
	Lophostemon suaveolens										■		■	■	■	■	■	■	■	■	
	Pandanus sp.																■		■		■
	Parinari nonda	■													■	■	■	■	■	■	■

Appendix 4 Mine rehabilitation seed mix used at Weipa – continued

	Species name	Age of rehabilitation site in 2006																			
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	16	18	20	21	22	23
Species sown in mine rehabilitation that are not part of the pre-mining native forest plant community	*Acacia auriculiformis*	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	
	Acacia brassii			▨												▨	▨			▨	
	Acacia dimidiata			▨	▨	▨	▨	▨	▨		▨	▨	▨	▨	▨	▨	▨		▨		▨
	Acacia dunii											▨									
	Acacia fleckeri																▨				
	Acacia holosericea											▨									
	Acacia legnota	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨		▨	▨	▨
	Acacia mangium	▨	▨			▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
	Acacia polystachya											▨									
	Acacia simsii				▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
	Acacia torulosa	▨	▨	▨	▨	▨	▨	▨	▨		▨	▨	▨	▨	▨	▨	▨	▨	▨		▨
	Allocasuarina littoralis																▨	▨		▨	
	Alstonia spectabilis	▨	▨	▨	▨		▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
	Banksia dentata															▨	▨	▨			
	Callistemon ciminalis																		▨		
	Callitris intratropica																	▨		▨	▨
	Casuarina equisetifolia																▨	▨	▨	▨	▨
	Cochlospermum gillivraei	▨	▨	▨	▨		▨	▨	▨	▨	▨	▨					▨	▨	▨	▨	▨
	Cook stylo													▨							
	Denhamia obscura																▨				
	Dodonaea platyptera					▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨			
	Dodonaea triquetra		▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨		
	Erythrina vespertilio																▨	▨	▨	▨	
	Eucalyptus alba	▨	▨		▨	▨	▨	▨	▨	▨		▨	▨	▨	▨	▨	▨	▨	▨		
	Eucalyptus brassiana	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨		▨	▨	▨	▨	▨	▨	▨
	Eucalyptus camaldulensis											▨	▨	▨	▨	▨	▨	▨	▨	▨	▨

Appendix 4 Mine rehabilitation seed mix used at Weipa – continued

	Species name	Age of rehabilitation site in 2006																			
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	16	18	20	21	22	23
Species sown in mine rehabilitation that are not part of the pre-mining native forest plant community	*Eucalyptus clarksoniana*	■	■	■	■	■		■	■		■			■							
	Eucalyptus miniata	■	■	■	■	■					■	■	■	■	■	■	■				
	Eucalyptus papuana						■	■	■				■	■				■	■	■	
	Eucalyptus phoenicia			■																	
	Eucalyptus ptychocarpa											■									
	Eucalyptus stageriana						■	■	■			■									
	Ficus platypoda																■		■		
	Grevillea banksii															■	■		■	■	■
	Grevillea heliosperma	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Grevillea pteridifolia	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Hakea persiehana																■	■	■		
	Leptospermum brachyandrum			■																	
	Leptospermum longifolium															■		■		■	■
	Manilkara kauki											■						■	■		
	Melaleuca arcana																	■		■	
	Melaleuca brassii																			■	
	Melaleuca dealbata																	■	■	■	■
	Melaleuca leucadendra	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Melaleuca saligna													■	■	■	■	■		■	■
	Melaleuca sp.					■	■	■	■	■	■	■	■	■	■	■					
	Melaleuca stenostachya																	■			■
	Melaleuca symphyocarpa	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Melaleuca viridiflora	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
	Sterculia quadrifida							■	■	■	■	■						■	■		■

Appendix 4 Mine rehabilitation seed mix used at Weipa – continued

	Species name	Age of rehabilitation site in 2006																			
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	16	18	20	21	22	23
Species sown in mine rehabilitation that are not from the pre-mining plant community	*Thespesia populnea*																		▨		
	Tristania suaveolens																				▨
	Wild Bush Yam													▨							
	Wongai Plum													▨							
	Xanthostemon crenulatus																	▨			▨
	Xanthostemon tetrapetala															▨					
Exotic species	Verano stylo													▨	▨						
	Seca stylo																				
	Khaya senegalensis												▨								

Appendix 5 Variables that may affect post-mining rehabilitation plots

Initial conditions / treatments	**Enabling / constraining variables**	**Priority effects**
minefloor: • graded / ungraded	landform	weed invasion
topsoil replacement: • dual stripping – separate A and B horizons / mixed A and B horizons / subsoil included • fresh soil / stockpiled soil	depth to water table and infiltration characteristics	*Acacia* senescence
topsoil condition while handling: • wet / moist / dry	ratio of remnant vegetation to rehabilitation	follow up treatments
depth of topsoil replaced on mine floor	habitat features	fire
time topsoil stockpiled for	distance to remnant vegetation	
topsoil: • ploughed / unploughed	timing and amount of rainfall	
depth and spacing of ripping		
alignment of ripping		
prior rainfall		
method of establishment: • aerial seeding / handseeding / handplanting / marshall spreader		
fertilizer type and application rate		
seed mix: • species mix • density of application • seed treatments • seed viability		

Appendix 6 Native forest reference sites

Site #	UTM of SW corner (WGS 84)	Land unit	*a priori* site type
A01	54L 624884 8605884	2B (type 3)	Erythrophleum chlorostachys sub-canopy
A02	54L 624906 8604668	2B (type 5)	Tall, layered
A03	54L 624643 8603563	2B (type 1)	Tall, open, grassy
A04	54L 623841 8602657	2B (type 1)	Tall, open, grassy
A05	54L 622660 8602429	2B (type 4)	Livistona muelleri sub-canopy
A06	54L 621678 8601951	2B (type 5)	Tall, layered
A08	54L 621552 8600441	2B (type 3)	Erythrophleum chlorostachys sub-canopy
A09	54L 620598 8600040	2B (type 3)	Erythrophleum chlorostachys sub-canopy
A10	54L 619757 8600618	2B (type 6)	Thickening
A11	54L 618718 8600055	2B (type 6)	Thickening
A12	54L 617763 8599536	2B (type 1)	Tall, open, grassy
A13	54L 612071 8604735	5K	
A14	54L 612397 8603528	2B (type 1)	Tall, open, grassy
A15	54L 612280 8602399	2B (type 3)	Erythrophleum chlorostachys sub-canopy
A16	54L 612128 8601050	2B (type 6)	Thickening
A17	54L 612003 8599831	2B (type 6)	Thickening
A18	54L 611027 8599552	2C	
A19	54L 607693 8599974	2C	
A20	54L 606669 8599581	2C	
A21	54L 604749 8603199	2C	
A22	54L 597638 8615208	5K	
A23	54L 597540 8614102	2B (type 3)	Erythrophleum chlorostachys sub-canopy
A24	54L 597453 8613102	2B (type 1)	Tall, open, grassy
A25	54L 597140 8610323	2B (type 2)	Corymiba nesophila co-dominant
A27	54L 592197 8608137	2B (type 5)	Tall, layered
A28	54L 591254 8608659	2B (type 2)	Corymiba nesophila co-dominant
A29	54L 590573 8609473	2B (type 1)	Tall, open, grassy
A30	54L 589074 8610026	2B (type 5)	Tall, layered
A31	54L 587318 8610533	2B (type 5)	Tall, layered
A32	54L 585857 8610987	5K	
A33	54L 585460 8612854	2B (type 5)	Tall, layered
A34	54L 584685 8614076	5K	
A35	54L 596395 8589249	2B (type 5)	Tall, layered
A36	54L 596508 8589841	2B (type 5)	Tall, layered
A37	54L 596316 8590849	2B (type 5)	Tall, layered
A38	54L 596601 8591927	2B (type 5)	Tall, layered

Appendix 7 Mine rehabilitation sites

Age class	Age in 2006	Year started	Site #	Block name	UTM of SW corner (WGS 84) 54L	Soil	Fertiliser	Seed	Block size (ha)	Distance from native forest (m)	Fire history
1 – 2	1	2005/2006	B13	Gar-7	605496 8604751	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	34.66	270-470	Never burnt
1 – 2	1	2005/2006	B29	Snapper-18	604271 8598876	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	9.22	185-390	Never burnt
1 – 2	1	2005/2006	B31	Whiting-9	601985 8594492	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	27.24	115-330	Never burnt
1 – 2	2	2004/2005	B11	Gar-6	605508 8605325	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	15.12	155-290	Never burnt
1 – 2	2	2004/2005	B20	Herring-28	598555 8603997	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	9.75	30-160	Burnt 2006
1 – 2	2	2004/2005	B30	Whiting-6	602204 8595201	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	23.1	695-820	Never burnt
3 – 4	3	2003/2004	B01	Kauri-7	589471 8621470	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	18.01	105-310	Never burnt
3 – 4	3	2003/2004	B02	Kauri-8	589065 8622196	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	7.61	155-350	Never burnt
3 – 4	3	2003/2004	B28	Snapper-11	604838 8598885	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	5.39	405-620	Never burnt
3 – 4	4	2002/2003	B03	Kauri-5	589683 8620999	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	8.76	250-415	Never burnt
3 – 4	4	2002/2003	B08	Grevillea-19	585531 8619763	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	28.86	670-865	Never burnt
3 – 4	4	2002/2003	B12	Gar-3	604835 8605038	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	9.34	505-695	Never burnt
5 – 8	5	2001/2002	B14	Gar-1	604780 8604052	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	59.03	195-410	Never burnt
5 – 8	5	2001/2002	B27	Snapper-10	605401 8598462	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	49.21	135-260	Never burnt

Appendix 7 Mine rehabilitation sites – continued

Age class	Age in 2006	Year started	Site #	Block name	UTM of SW corner (WGS 84) 54L	Soil	Fertiliser	Seed	Block size (ha)	Distance from native forest (m)	Fire history
5 – 8	6	2000/2001	B05	Jacaranda-24	587044 8619390	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	13.14	1160-1380	Never burnt
5 – 8	6	2000/2001	B06	Grevillea-14	586389 8619087	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	12.43	1360-1540	Never burnt
5 – 8	7	1999/2000	B15	Mackerel – 6	604322 8605338	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	9.06	430-640	Never burnt
5 – 8	8	1998/1999	B21	Herring-18	598566 8603727	Fresh	100 kg/ha superphosphate ground application	Marshall spreader	5.82	15-215	Burnt in 2006
9 – 16	9	1997/1998	B26	Nanagai-26	599916 8599177	Stockpile	100 kg/ha superphosphate ground application	Marshall spreader	25.48	1055-1210	Burnt 2001
9 – 16	10	1996/1997	B04	Nunda-15	589404 8619566	Stockpile	100 kg/ha superphosphate ground application	Marshall spreader	9.19	200-340	Never burnt
9 – 16	11	1995/1996	B17	Groper – 5	603644 8603813	Fresh	100 kg/ha superphosphate ground application + CK5 250 kg/ha applied aerially	Marshall spreader + tubestock	13.53	35-235	Burnt in 1999 and 2006
9 – 16	12	1994/1995	B07	Grevillea-1	586919 8620157	Stockpile	100 kg/ha superphosphate ground application + 250 kg/ha superphosphate applied aerially	Marshall spreader	33.9	430-630	Never burnt
9 – 16	13	1993/1994	B18	Pike-29	603006 8603875	Fresh	100 kg/ha superphosphate ground application + 180 kg/ha triphosphate applied aerially	Marshall spreader + tubestock	10.91	30-240	Partially burnt in 2006
9 – 16	14	1992/1993	B10	Enna 15	586608 8613024	Stockpile	100 kg/ha superphosphate ground application + SM 450 kg/ha applied aerially	Marshall spreader	19	70-270	Burnt in 1999, 2000, 2001, 2003 and 2006
9 – 16	14	1992/1993	B19	Pike-25	600692 8602608	Stockpile	100 kg/ha superphosphate ground application + SM 450 kg/ha applied aerially	Marshall spreader	19.45	35-235	Partially burnt in 2006
9 – 16	16	1990/1991	B16	Trevalley-22	601404 8601986	Stockpile	SM 500 kg/ha applied aerially	Aerial application + tubestock	7.49	50-160	Burnt in 2005

Appendix 7 Mine rehabilitation sites – continued

Age class	Age in 2006	Year started	Site #	Block name	UTM of SW corner (WGS 84) 54L	Soil	Fertiliser	Seed	Block size (ha)	Distance from native forest (m)	Fire history
> 16	18	1988/1989	B09	Enna 8A	584882 8613531	Fresh	SM 300 kg/ha applied aerially	Aerial application + tubestock	15	50-250	Burnt in 2001, 2003, 2005 and 2006
> 16	20	1986/1987	B22	Trevalley 11	600563 8601582	Fresh	SM 300 kg/ha applied aerially	Aerial application + tubestock	8.03	15-170	Burnt in 2005
> 16	21	1985/1986	B24	Bass 2	601895 8599772	Fresh	SM 300 kg/ha applied aerially	Aerial application + tubestock	19.6	165-235	Never burnt
> 16	22	1984/1985	B23	Trout - 18	601568 8600531	Fresh	SM 300 kg/ha applied aerially	Aerial application + tubestock	26.07	105-255	Burnt in 2005
> 16	23	1983/1984	B25	Trout 10	601233 8599519	Fresh	SM 300 kg/ha applied aerially	Aerial application + tubestock	17.36	240-340	Never burnt

Appendix 8 Soil surface assessment indicators[1]

Indicator	Description
Rainsplash protection	Assesses the degree to which physical surface cover (rocks > 2 cm plus woody material) and projected plant cover of perennial vegetation to a height of 0.5 m (5 categories).
Perennial vegetation cover	Assesses the contribution of below-ground biomass of perennial vegetation by summing butt lengths of perennial grasses and estimating the cover and density of shrubs and trees overhanging the query zone (4 categories).
Litter cover	Assesses the amount, origin and degree of decomposition of plant litter including annual grasses, ephemeral herbage and detached leaves, stems etc (10 categories).
Cryptogam cover	Assesses the cover of cryptogams visible on the soil surface (5 categories).
Crust brokenness	Assesses the extent to which the surface crust is broken leaving loosely attached soil material available for erosion (5 categories).
Soil erosion type and severity	Assesses the type of erosion (five types) and severity of recent / current soil erosion (4 categories).
Deposited materials	Assesses the nature and amount of alluvium transported to and deposited on the query zone (4 categories).
Soil surface roughness	Assesses the surface roughness for its capacity to capture and retain mobile resources such as water, propagules, topsoil and organic matter (5 categories).
Surface resistance to disturbance	Assesses the ease with which the soil can be mechanically disturbed to yield material suitable for erosion by wind or water (5 categories).
Slake test	Assesses the stability of natural soil fragments to rapid wetting (5 categories).
Soil texture	Classifies the texture of the surface soil and relates it to permeability (4 categories)

Redrawn from Tongway and Hindley (2004)

Appendix 9 Bird response variables

Level	Response variable	Calculation
Species	Presence / absence	Species presence / absence values were calculated for each site from pooled data over 8 site visits. 1 = recorded at least once out of 8 site visits, 0 = never recorded.
	Total abundance	Species total abundance values were calculated for each site as the total number of individuals over 8 site visits.
	Mean abundance	Species mean abundance values were calculated for each site as the total number of individuals recorded over 8 site visits / 8.
	Detection rate	Species detection rates were calculated for each site as the number of site visits in which it was recorded in the site / 8. 1 = present during every site visit.
Foraging group	Mean abundance	Each species was assigned to one of 14 foraging groups. For each foraging group, mean abundance was calculated for each site as the total number of individuals within a foraging group over eight site visits / 8.
Site	Species richness	Count of all bird species observed during eight site visits based on presence / absence data.
	Mean bird species richness per visit	Count of all bird species observed / 8.
	Mean bird abundance per visit	Count of all individuals of all species observed during eight site visits / 8.
	Foraging group richness	Count of the number of foraging groups recorded in the site over eight site visits based on presence / absence data.
	Species shortfall index	section 5.5.1.1

Appendix 10 Landscape context and vegetation variables

Explanatory Variable	**How calculated / estimated**
Age	The number of wet seasons since rehabilitation commenced. Site age was the age in September 2006 when field work commenced. All native forest sites are assumed to be 200 years old.
Area of remnant vegetation in a 500 m buffer around a site	Area of remnant native vegetation in a 500 m buffer around the site was calculated using ArcGIS and clearing data supplied by Rio Tinto. Remnant vegetation included a number of vegetation types other than *Eucalyptus tetrodonta* open forest.
Distance from remnant vegetation	Shortest distance from the centre of the site to remnant vegetation or patch of remnant vegetation shown on ArcGIS using clearing data supplied by Rio Tinto
Distance from coast	Shortest distance from the site to the coast, calculated using ArcGIS
Distance from the edge of the bauxite plateau	Shortest distance from the site to the edge of the bauxite plateau was calculated using 1: 100 000 topographic data in ArcGIS
Distance from mesic vegetation	Shortest distance from the site to the nearest stream. Measured using 1: 100 000 drainage data in ArcGIS
Vegetation height	Mean height of the tallest vegetation layer in metres
Stem density of big trees	Mean number of stems exceeding 35 cm dbh per ha
Stem density of small trees	Mean number of stems > 10 < 35 cm dbh per ha
Stem density of tall shrubs	Mean number of stems > 1 < 10 cm dbh per ha
Stem density of low shrubs	Mean number of stems < 1 cm dbh per ha
Basal area of big trees	Mean basal area of stems exceeding 35 cm dbh in m^2 / ha
Basal area of small trees	Mean basal area of stems > 10 < 35 cm dbh in m^2 / ha
Canopy volume of big trees	Mean canopy volume of stems exceeding 35 cm dbh in m^3 / ha
Canopy volume of small trees	Mean canopy volume of stems > 10 < 35 cm dbh in m^3 / ha
Canopy volume of tall shrubs	Mean canopy volume of stems > 1 < 10 cm dbh in m^3 / ha
Canopy volume of low shrubs	Mean canopy volume of stems < 1 cm dbh in m^3 / ha
Canopy volume of perennial grasses	Mean canopy volume of perennial grasses in m^3 / ha
Canopy volume of *Acacia spp*	Mean canopy volume of Acacias in m^3 / ha
Canopy volume of *Alstonia spp*	Mean canopy volume of *Alstonia spp* in m^3 / ha
Canopy volume of *Corymbia spp*	Mean canopy volume of *Corymbia spp* in m^3 / ha
Canopy volume of *Eucalyptus* spp	Mean canopy volume of *Eucalyptus spp* in m^3 / ha
Canopy volume of *Erythrophleum chlorostachys*	Mean canopy volume of *Erythrophleum chlorostachys* in m^3 / ha
Canopy volume of *Livistona muelleri*	Mean canopy volume of *Livistona muelleri* in m^3 / ha
Canopy volume of *Melaleuca spp*	Mean canopy volume of *Melaleuca spp* in m^3 / ha
Canopy volume of *Parinari nonda*	Mean canopy volume of *Parinari nonda* in m^3 / ha
Canopy volume of *Xylomelum scottianum*	Mean canopy volume of *Xylomelum scottianum* in m^3 / ha
Canopy volume of other species	Mean canopy volume of all other species in m^3 / ha

Appendix 11 Unrecorded species

# of species	Possible reason for failure to record	Unrecorded species
7	Original record is an unusual record for *Eucalyptus tetrodonta* open forest	Southern Cassowary
		Tawny-breasted Honeyeater
		Masked Lapwing
		Nankeen Night Heron
		Marbled Frogmouth
		Australian White Ibis
		Red-backed Button-quail
10	The regional list of *Eucalyptus tetrodonta* species covers a much larger area than the area covered in this research. Species is at or beyond the limits of its regular distribution in the research area	Common Bronzewing
		Crested Pigeon
		Squatter Pigeon
		Diamond Dove
		Galah
		Little Corella
		Black-chinned Honeyeater
		Dusky Woodswallow
		Pied Currawong
		Black-eared Cuckoo
7	Species has a declining population or has suffered a range contraction since records began	Red Goshawk
		Golden-shouldered Parrot
		Black-faced Woodswallow
		Black-throated Finch
		Crimson Finch
		Star Finch
		Gouldian Finch
8	Species is sparsely distributed or a seasonal visitor on north-western Cape York Peninsula	King Quail
		Black-shouldered Kite
		Varied Lorikeet
		Pallid Cuckoo
		Silver-crowned Friarbird
		Willie Wagtail
		Jacky Winter
		Fairy Martin
5	Species is a nightbird – no nocturnal surveys were conducted	Australian Owlet-nightjar
		Rufous Owl
		Barking Owl
		Masked Owl
		Eastern Barn Owl
5	No explanation is available	Little Eagle
		White-throated Gerygone
		Double-barred Finch
		Australian Swiftlet
		Bar-breasted Honeyeater

Appendix 12 Bird summary values by site

Site ID	Site category	Age class	Species richness (All visits)	Mean species richness per visit	Mean abundance per visit (all species)	Native forest species shortfall %
A01	Native Forest	OG2B	23	10.38	28.88	27.04
A02	Native Forest	OG2B	24	10.25	32.88	22.11
A03	Native Forest	OG2B	25	11.13	31.38	23.67
A04	Native Forest	OG2B	23	10.38	31.13	25.13
A05	Native Forest	OG2B	30	13.75	35.75	16.84
A06	Native Forest	OG2B	22	11.25	29	20.21
A08	Native Forest	OG2B	24	10.88	30.63	19.43
A09	Native Forest	OG2B	27	13.0	35.63	22.31
A10	Native Forest	OG2B	23	10.5	29.13	25.50
A11	Native Forest	OG2B	28	11.25	28.5	28.92
A12	Native Forest	OG2B	30	14.25	31.5	27.27
A13	Native Forest	OG5K	25	12.0	27	30.59
A14	Native Forest	OG2B	26	12.5	28.88	26.83
A15	Native Forest	OG2B	28	12.38	34.38	27.69
A16	Native Forest	OG2B	25	12.5	30.75	21.75
A17	Native Forest	OG2B	24	11.75	32	28.13
A18	Native Forest	OG2C	29	14.25	35.5	22.57
A19	Native Forest	OG2C	32	14.63	38.38	22.07
A20	Native Forest	OG2C	26	12.63	32.5	28.18
A21	Native Forest	OG2C	29	13.5	29.63	28.75
A22	Native Forest	OG5K	25	10.25	24.25	39.15
A23	Native Forest	OG2B	24	11.0	29	41.35
A24	Native Forest	OG2B	26	12.25	27.25	27.35
A25	Native Forest	OG2B	34	16.5	37.5	26.63
A27	Native Forest	OG2B	31	12.75	28.75	28.85
A28	Native Forest	OG2B	30	12.13	29	25.70
A29	Native Forest	OG2B	30	13.75	32.25	20.19
A30	Native Forest	OG2B	25	9.75	25.5	36.54
A31	Native Forest	OG2B	24	9.38	27.25	34.69
A32	Native Forest	OG5K	34	15.88	37.13	25.53
A33	Native Forest	OG2B	29	14.13	31.63	30.81
A34	Native Forest	OG5K	31	11.75	22.38	39.14
A35	Native Forest	OG2B	24	12.38	28.63	32.47
A36	Native Forest	OG2B	35	15.25	32.25	27.26
A37	Native Forest	OG2B	38	15.5	31.5	34.49
A38	Native Forest	OG2B	28	13.88	35.63	24.68
B01	Mine Rehab	3 to 4	13	4.5	10.5	95.17
B02	Mine Rehab	3 to 4	11	3.75	7.25	95.39
B03	Mine Rehab	3 to 4	17	5.75	15.25	71.81
B04	Mine Rehab	9 to 16	26	8.13	17	78.85
B05	Mine Rehab	5 to 8	17	5.5	21.25	95.28
B06	Mine Rehab	5 to 8	17	6.0	11.88	90.65
B07	Mine Rehab	9 to 16	18	6.0	14.88	91.11
B08	Mine Rehab	3 to 4	14	4.25	13.13	96.19
B09	Mine Rehab	>16	32	10.88	20.25	57.32
B10	Mine Rehab	9 to 16	26	8.0	17.13	77.87
B11	Mine Rehab	1 to 2	8	2.6	4.4	97.53

Appendix 12 Bird summary values by site – continued

Site ID	Site category	Age class	Species richness (All visits)	Mean species richness per visit	Mean abundance per visit (all species)	Native forest species shortfall %
B12	Mine Rehab	3 to 4	20	4.38	11.25	89.03
B13	Mine Rehab	1 to 2	5	1.67	2.5	99.69
B14	Mine Rehab	5 to 8	13	3.13	5.63	94.95
B15	Mine Rehab	5 to 8	24	6.88	15.63	72.98
B16	Mine Rehab	9 to 16	28	9.38	22.38	84.05
B17	Mine Rehab	9 to 16	20	7.13	12.75	84.53
B18	Mine Rehab	9 to 16	32	12.63	23.75	66.50
B19	Mine Rehab	9 to 16	24	9.13	18.5	67.90
B20	Mine Rehab	1 to 2	9	2.38	4.13	96.48
B21	Mine Rehab	5 to 8	29	9.75	19.63	70.10
B22	Mine Rehab	>16	29	10.63	19.88	65.94
B23	Mine Rehab	>16	26	10.13	19.5	64.80
B24	Mine Rehab	>16	24	8.38	13.75	79.58
B25	Mine Rehab	>16	30	11.75	26	47.01
B26	Mine Rehab	9 to 16	22	8.38	19.13	83.92
B27	Mine Rehab	5 to 8	18	5.25	8.38	84.99
B28	Mine Rehab	3 to 4	14	4.25	8.88	93.94
B29	Mine Rehab	1 to 2	8	2.5	4	99.15
B30	Mine Rehab	1 to 2	12	3.63	11	96.89
B31	Mine Rehab	1 to 2	9	2.14	4.71	97.61

Appendix 13 Bird Abundance ANOVA Results

Site category, season, time since fire, and weather conditions all caused variation in bird abundance. Mean bird abundance was highest in the early dry season of 2006 and lowest in the late dry season 2007. Mean bird abundance was highest in sites that had been burnt in the week prior to the survey. Generally, bird abundance was higher when it was cloudless and when there was no wind although the highest value for mean bird abundance was when there was no wind and some cloud cover. These effects may have been due to changes in bird detectability or bird activity under different conditions. Site category had the greatest effect on bird abundance.

ANOVA results for bird abundance data

Site category	Season	Fire	Cloud*Wind
F (5, 289)	F (3, 335)	F (4, 478)	F (11, 498)
206.8 ***	33.35 ***	3.04 *	2.31 **

Values shown are fixed effects. Bird abundance was square root transformed for analysis to meet assumptions of normality. Values shown are F-statistics (variance ratios) which indicate the size of the effect. Numbers in brackets are numerator degrees of freedom (n.d.f) and denominator degrees of freedom (d.d.f). The shape of the F-distribution depends on the degrees of freedom. The lower the degrees of freedom, the larger the value of F needed to be significant. Significance of the F-statistic is indicated by asterisks * = $0.05 \geq P > 0.01$; ** = $0.01 \geq P > 0.001$;*** = $P \leq 0.001$, ns = not significant.

Predicted means – Site category (average s.e. of differences = 0.16)

1 to 2 y.o.	3 to 4 y.o.	5 to 8 y.o.	9 to 16 y.o.	> 16 y.o.	Native Forest
1.87	3.28	3.69	4.27	4.45	5.57

Predicted means - Season (average s.e. of differences = 0.12)

Late dry season 2006	Early dry season 2007	Early wet season 2007	Storms 2007
3.36	4.55	3.58	3.94

Predicted means - Fire (average s.e. of differences = 0.3)

F1	F2	F3	F4	F5
5.15	3.67	3.47	3.46	3.54

Fire codes are for time since fire. F1 = < 7 days, F2 = > 7 day < 1 month, F3 = > 1 < 3 months, F4 = > 3 < 6 months, F5 = > 6 months

Predicted means - Cloud*Wind interaction (average s.e of differences = 0.3)

	W0	**W1**	**W2**	**W3**
C0	4.21	4.06	4.28	4.01
C1	4.18	4.04	3.84	3.72
C2	4.17	4.22	3.80	2.97
C3	4.4	3.75	4.19	3.68
C4	4.3	*	2.33	3.14

Wind codes are indicators of wind strength. W0 = Still, W1 = Slight movement - no rustling of leaves, W2 = Rustling of leaves in canopy, W3 = Swaying of small branches, W4 = Swaying of large branches. Cloud codes are for the percentage of sky covered by cloud. C0 = cloudless, C1 = 5 - 25% cloud cover, C2 =26 - 75% cloud cover, C3 = 76 - 95% cloud cover, C4 = completely overcast.

Appendix 14 Pairwise differences in bird species composition

	1-2 y	**3-4 y**	**5-8 y**	**9-16 y**	**> 16 y**	**LU2B**	**LU2C**	**LU5K**
1-2 y.o		T= -3 P<0.005	T= -4 P<0.002	T= -7 P<0.001	T= -6 P<0.001	T= -21 P<0.001	T= -6 P<0.002	T= -5 P<0.002
3-4 y.o.			**T<1** **n.s.**	T= -5 P<0.001	T= -4 P<0.002	T= -19 P<0.001	T= -5 P<0.002	T= -5 P<0.002
5-8 y.o.				**T<1** **n.s.**	T= -7 P<0.01	T= -15 P<0.001	T= -5 P<0.002	T= -5 P<0.005
9-16 y.o.					T= -3 P<0.01	T= -19 P<0.001	T= -6 P<0.001	T= -6 P<0.001
17-24 y.o						T= -13 P<0.001	T= -5 P<0.005	T= -4 P<0.005
LU2B							**T<1** **n.s.**	**T<1** **n.s.**
LU2C								**T<1** **n.s.**

The test statistic (T) is the difference between the observed and expected deltas divided by the square root of the variance in delta. It describes the separation between groups, the more negative is T, the stronger the separation. The chance corrected within group agreement (A) is a description of the effect size, independent of the sample size and statistical significance. Values for A > 0.3 are considered high. The probability value expresses the likelihood of obtaining a delta as extreme or more extreme than the observed delta given the distribution of possible deltas (McCune and Grace, 2002).

Appendix 15 Foraging groups

Foraging Group	Code	Common name
Carnivore	C	Whistling Kite Brahminy Kite Black Kite Nankeen Kestrel Southern Boobook Australian Hobby Black Butcherbird Grey Goshwak Brown Goshawk Collared Sparrowhawk Pacific Baza Papuan Frogmouth Lesser Frigatebird Brown Falcon Peregrine Falcon Wedge-tailed Eagle Tawny Frogmouth
Frugivore	F	Emu Common Koel Mistletoebird Figbird Olive-backed Oriole Yellow Oriole Pied Imperial-Pigeon Great Bowerbird
Granivore	G	Red-chested Button-quail Bar-shouldered Dove Peaceful Dove Red-browed Finch Brown Quail Chestnut-breasted Mannikin
Ground forager	GF	Australian Brush-turkey Orange-footed Scrubfowl
Insectivore - aerial	I-A	Tree Martin White-throated Needletail Large-tailed Nightjar White-breasted Woodswallow Little Woodswallow
Insectivore - foliage gleaner	I-FG	White-winged Triller Little Bronze-Cuckoo Cicadabird Grey Shrike-thrush Red-backed Fairy-wren Varied Triller Oriental Cuckoo Weebill Rufous Whistler Fairy Gerygone Black-faced Cuckoo-Shrike Fan-tailed Cuckoo Channel-billed Cuckoo Brush Cuckoo Grey Whistler

Appendix 15 Foraging groups - continued

Foraging Group	Code	Common name
		Red-browed Pardalote White-bellied Cuckoo-Shrike Striated Pardalote
Insectivore - grass and ground forager	I-G	Richards Pipit Magpie-lark Lovely Fairy-wren Bush Stone-curlew Golden-headed Cisticola Pheasant Coucal
Insectivore - hawker	I-H	Northern Fantail Grey Fantail Leaden Flycatcher Lemon-bellied Flycatcher Spangled Drongo Rainbow Bee-eater Dollarbird
Insectivore - trunk gleaner	I-T	Brown Treecreeper Varied Sittella Grey-crowned Babbler
Insectivore / nectarivore	I/N	Brown Honeyeater Yellow-spotted Honeyeater Little Friarbird Blue-faced Honeyeater Brown-backed Honeyeater Dusky Honeyeater Graceful Honeyeater Red-headed Honeyeater White-streaked Honeyeater White-throated Honeyeater Yellow Honeyeater Yellow-tinted Honeyeater Banded Honeyeater
Nectarivore	N	Rainbow Lorikeet Yellow-bellied Sunbird Helmeted Friarbird Noisy Friarbird
Omnivore	O	Torresian Crow
Pouncing predator	P	Forest Kingfisher Sacred Kingfisher Pied Butcherbird Black-backed Butcherbird Yellow-billed Kingfisher Blue-winged Kookaburra Laughing Kookaburra
Seed eater (trees)	S	Pale-headed Rosella Red-tailed Black-Cockatoo Red-winged Parrot Sulphur-crested Cockatoo Palm Cockatoo

Appendix 16 CCA Site bird detection rates and vegetation structure

Data matrices

Main matrix: 67 sites (rows) / 62 species (columns)
Second matrix: 67 sites (rows) / 14 VegStruc (columns)

Options selected

Axis scores centered and standardised to unit variance
Axes scaled to optimise representation of columns: species
Scores for species are weighted mean scores for sites
Scores for graphing sites are linear combinations of VegStruc
Monte Carlo test: null hypothesis is no relationship between matrices
Random number seed: 2045

Axis summary statistics

Number of canonical axes: 3
Total variance ('inertia') in the species data: 1.8360

	Axis 1	**Axis 2**	**Axis 3**
Eigenvalue	0.424	0.118	0.047
Variance in species data			
% of variance explained	23.1	6.4	2.6
Cumulative % explained	23.1	29.5	32.1
Pearson Correlation, Spp-Envt*	0.942	0.775	0.717
Kendall (Rank) Corr., Spp-Envt	0.713	0.613	0.636

* Correlation between sample scores for an axis derived from the species data and the sample scores that are linear combinations of the environmental variables. Set to 0.000 if axis is not canonical.

Multiple regression results

Regression of sites in species space on VegStruc

	Canonical Coefficients						
	Standardised			**Original Units**			
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**	**S.Dev**
1 V_HtMean	0.917	0.536	1.181	0.126	0.074	0.163	0.726E+01
2 V_St_L5	0.777	-0.689	1.025	0.030	-0.026	0.039	0.262E+02
3 V_St_L4	0.337	0.337	0.417	0.004	0.004	0.004	0.958E+02
4 V_St_L3	0.164	-0.077	0.547	0.000	0.000	0.002	0.356E+03
5 V_St_L2	-0.088	0.436	0.210	0.000	0.000	0.000	0.664E+04
6 V_BA_L5	-0.238	0.383	-0.655	-0.049	0.080	-0.136	0.481E+01
7 V_BA_L4	-0.516	-0.361	-1.747	-0.195	-0.136	-0.660	0.265E+01
8 V_CV_L5	-0.359	0.064	-0.780	0.000	0.000	0.000	0.206E+05
9 V_CV_L4	0.248	0.551	0.427	0.000	0.000	0.000	0.876E+04
10 V_CV_L3	-0.142	0.677	0.406	0.000	0.000	0.000	0.357E+04
11 V_CV_L2	0.009	-0.188	0.000	0.000	0.000	0.000	0.781E+03
12 V_CV_L1	-0.057	-0.188	-0.214	0.000	0.000	0.000	0.378E+04

Final scores and raw data totals (weights) for 62 species

	Axis 1	**Axis 2**	**Axis 3**	**Totals**
1 BrTurk	-1.615667	-0.026475	0.510450	1.3750
2 BaHon	0.270262	-0.263541	-0.302153	9.6250
3 BBBut	0.598961	-0.066514	-0.008531	22.3750
4 BFCuSh	-0.059237	-0.063999	0.197639	4.0000
5 BBut	-1.123067	1.934252	-0.546944	1.1250
6 Goshawk	0.156701	0.193378	0.220897	1.0000
7 BrHon	-1.629274	-0.153829	0.197725	18.2500

Final scores and raw data totals (weights) for 62 species - continued

	Axis 1	**Axis 2**	**Axis 3**	**Totals**
8 Quail	-1.658430	-1.165258	-0.086845	0.6250
9 Treecr	0.695600	-0.409200	0.413736	6.3750
10 BraKit	-0.558472	0.754192	0.346726	0.6250
11 BBHon	-0.302267	0.837385	0.220905	2.2500
12 BrCuc	0.472033	0.057794	0.899418	3.0000
13 BSDov	-0.760676	0.044201	0.318839	18.0000
14 BWKoo	0.458595	-0.032570	-0.423107	3.0000
15 CBMan	-1.798097	-0.832417	-0.234520	1.1250
16 Cicada	0.548121	-0.018579	-0.145663	6.0000
17 SpHawk	-0.145481	-0.311319	-0.288261	2.1250
18 Coucal	-0.339102	-0.202666	0.355701	2.6250
19 Crow	-0.396527	-0.506038	-0.251387	3.5000
20 Drong	-0.270382	0.730731	-0.148937	5.1250
21 DHon	-0.783103	1.299601	-0.425133	4.7500
22 FKing	0.571269	0.005548	0.034624	14.5000
23 Babb	0.629769	0.066086	0.324387	0.8750
24 Cist	-1.961394	-1.382220	-0.586817	6.8750
25 GSThr	0.637844	-0.256013	0.194083	14.8750
26 GrHon	-0.902684	0.790221	-0.186528	8.3750
27 Bower	-0.304586	0.939958	0.082213	4.2500
28 Koel	0.256699	0.437232	0.030413	1.7500
29 LKook	0.546291	-0.070595	-0.024305	12.2500
30 LdFly	0.311763	0.041213	-0.015596	27.7500
31 LBFly	0.523375	-0.023576	-0.094054	28.8750
32 LFri	0.342113	-0.006722	0.036221	38.6250
33 Mistl	0.234041	-0.058741	0.088734	29.6250
34 NFri	0.408045	-0.115236	-0.042278	21.8750
35 OBOrio	0.248713	0.117346	0.150188	2.5000
36 PDov	-0.644619	-0.156355	0.206083	14.5000
37 PIPig	0.624669	0.432624	0.291564	0.8750
38 PHRos	-0.186002	-0.059803	0.067246	7.2500
39 PBut	0.006861	-0.000035	0.035864	1.2500
40 BeeE	0.001603	0.210416	0.272734	4.7500
41 Pipit	-2.205732	-1.983069	-1.414228	1.6250
42 Lorik	0.382999	0.011997	-0.034811	32.2500
43 RBFWr	0.097054	-0.329276	0.325175	9.5000
44 RBrFin	-1.667568	-0.669506	-0.717562	1.3750
45 RuWh	0.576687	-0.133631	-0.170878	25.5000
46 RWParr	0.075143	0.067826	0.189626	12.0000
47 SCCoc	0.267007	0.228607	-0.054271	8.1250
48 StPar	-0.432265	-0.166410	-0.034389	21.6250
49 VSitt	0.689623	0.026064	0.069835	5.7500
50 VTril	-1.026745	1.369102	-0.681609	1.0000
51 WBCuSh	0.182224	0.027060	-0.000362	32.6250
52 WBWood	0.004842	-0.254918	0.396162	0.8750
53 Weeb	-0.538340	-0.145731	0.295145	3.6250
54 WhKit	-0.126485	-0.416943	-0.119931	1.7500
55 WStHon	-1.462988	0.136034	0.484797	2.8750
56 WTHon	0.146067	0.073293	-0.068068	48.2500
57 WWTril	0.019450	-0.469812	-0.045136	1.3750
58 Sunbird	-1.484094	0.021500	0.191489	23.2500
59 YHon	0.034126	0.111183	-0.074183	31.2500
60 YOriol	0.573118	0.863190	-0.023031	1.1250
61 YSHon	-1.038689	1.234368	-0.544970	8.6250
62 YTHon	0.660704	-0.574979	-0.910952	3.2500

Correlations and biplot scores for 12 Vegetation Structure variables

	Correlations*			Biplot Scores		
Variable	Axis 1	Axis 2	Axis 3	Axis 1	Axis 2	Axis 3
1 V_HtMean	0.981	-0.004	-0.038	0.981	-0.004	-0.038
2 V_St_L5	0.831	-0.243	-0.215	0.831	-0.243	-0.215
3 V_St_L4	0.199	0.801	-0.301	0.199	0.801	-0.301
4 V_St_L3	-0.265	0.440	0.542	-0.265	0.440	0.542
5 V_St_L2	0.483	0.237	0.140	0.483	0.237	0.140
6 V_BA_L5	0.803	-0.222	-0.256	0.803	-0.222	-0.256
7 V_BA_L4	0.377	0.738	-0.435	0.377	0.738	-0.435
8 V_CV_L5	0.748	-0.212	-0.274	0.748	-0.212	-0.274
9 V_CV_L4	0.295	0.757	-0.474	0.295	0.757	-0.474
10 V_CV_L3	-0.634	0.559	0.296	-0.634	0.559	0.296
11 V_CV_L2	0.377	-0.078	0.169	0.377	-0.078	0.169
12 V_CV_L1	0.707	-0.272	-0.029	0.707	-0.272	-0.029

* Correlations are 'intraset correlations' of ter Braak (1986). The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case the site detection rate of birds, per unit change in the corresponding environmental variables (Ter Braak, 1986).

Monte Carlo test results – eigenvalues

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	Eigenvalue	Mean	Min	Max	p
1	0.424	0.090	0.041	0.190	0.0010
2	0.118	0.042	0.025	0.072	
3	0.047	0.030	0.021	0.044	

p = proportion of randomised runs with eigenvalue greater than or equal to the observed eigenvalue; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Monte Carlo test results – species-environment correlations

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	Spp-Envt Corr.	Mean	Min	Max	p
1	0.942	0.499	0.367	0.713	0.0010
2	0.775	0.589	0.343	0.821	
3	0.717	0.646	0.372	0.831	

p = proportion of randomised runs with species-environment correlation greater than or equal to the observed species-environment correlation; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Appendix 17 Correlations between vegetation structure and composition

StL5	**0.87**																				
StL4	0.38	0.11																			
StL3	-0.17	-0.37	0.26																		
StL2	**0.66**	**0.52**	0.35	-0.07																	
BAL5	**0.86**	**0.99**	0.12	-0.36	**0.51**																
BAL4	**0.54**	0.31	**0.93**	0.16	0.40	0.31															
CVL5	**0.82**	**0.97**	0.09	-0.37	0.48	**0.98**	0.28														
CVL4	0.46	0.28	**0.89**	0.14	0.30	0.28	**0.96**	0.27													
CVL3	-0.5	**-0.56**	0.11	**0.70**	-0.37	**-0.54**	-0.03	**-0.52**	0.04												
CVL2	0.44	0.34	0.07	-0.01	**0.56**	0.37	0.10	0.37	0.01	-0.29											
CVL1	**0.78**	**0.79**	0.11	-0.24	**0.53**	**0.79**	0.23	**0.72**	0.15	-0.49	0.44										
CVAc	-0.40	-0.48	0.2	0.33	-0.27	-0.47	0.08	-0.45	0.17	**0.73**	-0.28	-0.43									
CVAl	-0.14	-0.25	**0.63**	0.32	0.01	-0.24	**0.59**	-0.23	**0.62**	0.36	-0.22	-0.25	0.24								
CVCo	0.71	**0.79**	0.08	-0.30	0.42	**0.75**	0.2	**0.76**	0.18	-0.48	0.39	**0.69**	-0.42	-0.21							
CVEr	**0.56**	**0.77**	0.01	-0.33	0.25	**0.78**	0.21	**0.76**	0.22	-0.38	0.09	**0.62**	-0.35	-0.15	0.48						
CVEu	**0.81**	**0.84**	0.28	-0.23	**0.50**	**0.86**	0.46	**0.89**	0.46	-0.44	0.38	**0.57**	-0.41	-0.09	**0.51**	**0.576**					
CVLi	0.15	0.04	0.05	0.09	0.06	0.03	0.07	0.08	0.05	-0.06	-0.00	-0.02	-0.07	-0.05	0.08	0.061	0.08				
CVMe	-0.11	-0.20	**0.55**	0.28	-0.02	-0.19	**0.57**	-0.19	**0.57**	0.29	-0.19	-0.21	0.26	**0.82**	-0.17	-0.127	-0.08	-0.04			
CVNo	0.18	0.09	0.48	0.07	0.13	0.11	**0.51**	0.09	**0.52**	-0.02	0.01	0.09	-0.04	0.31	0.04	0.135	0.13	-0.05	0.19		
CVOt	-0.37	-0.47	0.39	**0.55**	-0.21	-0.46	0.28	-0.43	0.35	**0.72**	-0.22	-0.46	**0.62**	0.46	-0.43	-0.322	-0.35	-0.03	0.40	0.41	
CVXy	**0.51**	0.36	0.35	-0.13	0.27	0.35	0.27	0.30	0.17	-0.31	0.20	0.42	-0.26	-0.15	0.44	0.056	0.29	0.07	-0.11	0.03	-0.28
	Ht	StL5	StL4	StL3	StL2	BAL5	BAL4	CVL5	CVL4	CVL3	CVL2	CVL1	CVAc	CVAl	CVCo	CV_Er	CVEu	CVLi	CVMe	CVNo	CVOt

Correlations higher than 0.5 are shown in bold

Appendix 17 Correlations between vegetation structure and composition - continued

Vegetation structure variables
Ht = Mean height of the tallest vegetation layer
StL5 = Stem density of big trees (layer 5)
StL4 = Stem density of small trees (layer 4)
StL3 = Stem density of tall shrubs (layer 3)
StL2 = Stem density of low shrubs (layer 2)
BAL5 = Basal area of big trees
BAL4 = Basal area of small trees
CVL5 = Canopy volume of big trees
CVL4 = Canopy volume of small trees
CVL3 = Canopy volume of tall shrubs
CVL2 = Canopy volume of low shrubs
CVL1 = Canopy volume of perennial grasses

Vegetation composition variables
CVAc = Canopy volume of *Acacia* spp
CVAl = Canopy volume of *Alstonia* spp
CVCo = Canopy volume of *Corymbia* spp
CVEr = Canopy volume of *Erythrophleum chlorostachys*
CVEu = Canopy volume of *Eucalyptus* spp
CVLi = Canopy volume of *Livistona*
CVMe = Canopy volume of *Melaleuca* spp
CVNo = Canopy volume of *Parinari nonda*
CVOt = Canopy volume of other species
CVXy = Canopy volume of *Xylomelum scottianum*

Appendix 18 CCA Site bird detection rates and vegetation composition

Data matrices

Main matrix: 67 sites (rows) / 62 species (columns)
Second matrix: 67 sites (rows) / 12 VegComp (columns)

Options selected

Axis scores centered and standardised to unit variance
Axes scaled to optimise representation of columns: species
Scores for species are weighted mean scores for sites
Scores for graphing sites are linear combinations of VegComp
Monte Carlo test: null hypothesis is no relationship between matrices
Random number seed: 4278

Axis summary statistics

Number of canonical axes: 3
Total variance ('inertia') in the species data: 1.8360

	Axis 1	**Axis 2**	**Axis 3**
Eigenvalue	0.352	0.099	0.040
Variance in species data			
% of variance explained	19.2	5.4	2.2
Cumulative % explained	19.2	24.5	26.7
Pearson Correlation, Spp-Envt*	0.881	0.717	0.830
Kendall (Rank) Corr., Spp-Envt	0.632	0.664	0.558

* Correlation between sample scores for an axis derived from the species data and the sample scores that are linear combinations of the environmental variables. Set to 0.000 if axis is not canonical.

Multiple regression results:

Regression of sites in species space on VegComp

	Canonical Coefficients						
	Standardised			**Original Units**			
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**	**S.Dev**
1 V_CV_Ac	-0.105	0.285	-0.310	0.000	0.000	0.000	0.412E+04
2 V_CV_Als	-0.155	0.558	-0.082	0.000	0.000	0.000	0.306E+04
3 V_CV_Co	0.306	0.267	0.391	0.000	0.000	0.000	0.856E+04
4 V_CV_Eu	0.444	0.462	0.076	0.000	0.000	0.000	0.165E+05
5 V_CV_Er	0.097	0.168	-0.981	0.000	0.000	0.000	0.641E+04
6 V_CV_Liv	0.047	0.005	0.445	0.000	0.000	0.003	0.133E+03
7 V_CV_Mel	0.145	-0.093	0.456	0.000	0.000	0.000	0.914E+03
8 V_CV_Non	0.119	0.156	0.188	0.000	0.000	0.000	0.105E+04
9 V_CV_Xyl	0.248	0.138	-0.173	0.000	0.000	0.000	0.819E+03
10 V_CV_Oth	-0.203	0.553	-0.149	0.000	0.000	0.000	0.344E+04

Final scores and raw data totals (weights) for 62 species

	Axis 1	**Axis 2**	**Axis 3**	**Totals**
1 BrTurk	-1.508007	-0.173891	-0.331832	1.3750
2 BaHon	0.409914	-0.058771	-0.450814	9.6250
3 BBBut	0.549268	-0.023373	-0.006874	22.3750
4 BFCuSh	-0.036153	-0.247144	-0.069440	4.0000
5 BBut	-1.717276	1.934471	-0.167675	1.1250
6 Goshawk	0.113770	-0.212582	0.184223	1.0000
7 BrHon	-1.358075	-0.294038	-0.172551	18.2500
8 Quail	-1.155396	-1.212817	0.125348	0.6250
9 Treecr	0.534528	-0.246164	0.288292	6.3750

Final scores and raw data totals (weights) for 62 species - continued

	Axis 1	Axis 2	Axis 3	Totals
10 BraKit	-0.641593	0.211539	0.145990	0.6250
11 BBHon	-0.451869	0.567034	0.087185	2.2500
12 BrCuc	0.176119	-0.453214	0.511215	3.0000
13 BSDov	-0.751818	-0.143988	-0.077087	18.0000
14 BWKoo	0.478416	0.161995	-0.251007	3.0000
15 CBMan	-1.375221	-0.663935	-0.239912	1.1250
16 Cicada	0.490241	0.086098	-0.359820	6.0000
17 SpHawk	0.065462	-0.104451	-0.043620	2.1250
18 Coucal	-0.359042	-0.633777	-0.029711	2.6250
19 Crow	-0.013908	-0.457399	-0.111599	3.5000
20 Drong	-0.400705	0.623655	0.018267	5.1250
21 DHon	-1.119276	1.222563	0.367362	4.7500
22 FKing	0.497068	-0.010430	-0.076653	14.5000
23 Babb	0.415279	-0.147131	0.668711	0.8750
24 Cist	-1.174420	-1.230143	0.045954	6.8750
25 GSThr	0.522570	-0.170699	0.109779	14.8750
26 GrHon	-1.044997	0.648235	-0.091607	8.3750
27 Bower	-0.571200	0.591226	0.451550	4.2500
28 Koel	0.139456	0.150977	-0.296488	1.7500
29 LKook	0.482581	0.029651	0.042393	12.2500
30 LdFly	0.329241	0.049958	0.059277	27.7500
31 LBFly	0.535815	0.079023	-0.025215	28.8750
32 LFri	0.305974	0.010707	-0.008897	38.6250
33 Mistl	0.207893	-0.075625	-0.055519	29.6250
34 NFri	0.426508	-0.027232	-0.136868	21.8750
35 OBOrio	0.220456	-0.291949	0.102815	2.5000
36 PDov	-0.616550	-0.185237	0.021528	14.5000
37 PIPig	0.307536	0.146767	0.012148	0.8750
38 PHRos	-0.203855	-0.049627	0.390262	7.2500
39 PBut	0.058877	-0.543447	0.598298	1.2500
40 BeeE	-0.010136	-0.094204	0.142397	4.7500
41 Pipit	-1.038635	-1.637951	0.191921	1.6250
42 Lorik	0.335541	0.094810	-0.042231	32.2500
43 RBFWr	-0.010961	-0.276225	0.006922	9.5000
44 RBrFin	-1.200614	-0.600745	0.168156	1.3750
45 RuWh	0.658401	0.031465	-0.006462	25.5000
46 RWParr	-0.155157	-0.041819	-0.201464	12.0000
47 SCCoc	0.141301	0.303235	0.324165	8.1250
48 StPar	-0.318274	-0.153068	0.091071	21.6250
49 VSitt	0.725047	0.074115	0.300047	5.7500
50 VTril	-1.371429	1.765841	-0.174120	1.0000
51 WBCuSh	0.176301	0.000694	0.069870	32.6250
52 WBWood	0.026196	-0.322541	0.168633	0.8750
53 Weeb	-0.511683	-0.453733	0.978580	3.6250
54 WhKit	0.186620	-0.386267	0.023649	1.7500
55 WStHon	-1.481662	-0.074081	-0.351332	2.8750
56 WTHon	0.103979	0.095103	0.019171	48.2500
57 WWTril	0.200396	-0.451547	-0.039665	1.3750
58 Sunbird	-1.339082	-0.113594	-0.096981	23.2500
59 YHon	0.013562	0.094807	0.101778	31.2500
60 YOriol	0.154704	0.620681	0.158950	1.1250
61 YSHon	-1.318052	1.150938	0.011965	8.6250
62 YTHon	0.732964	0.104400	-1.634012	3.2500

Correlations and biplot scores for 10 Vegetation Composition variables

	Correlations*			Biplot Scores		
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**
1 V_CV_Ac	-0.682	0.368	-0.129	-0.682	0.368	-0.129
2 V_CV_Als	-0.351	0.680	0.252	-0.351	0.680	0.252
3 V_CV_Co	0.749	0.010	0.146	0.749	0.010	0.146
4 V_CV_Eu	0.790	0.266	-0.106	0.790	0.266	-0.106
5 V_CV_Er	0.550	0.124	-0.637	0.550	0.124	-0.637
6 V_CV_Liv	0.103	-0.011	0.408	0.103	-0.011	0.408
7 V_CV_Mel	-0.264	0.559	0.319	-0.264	0.559	0.319
8 V_CV_Non	0.060	0.545	0.028	0.060	0.545	0.028
9 V_CV_Xyl	0.520	-0.037	0.116	0.520	-0.037	0.116
10 V_CV_Oth	-0.676	0.617	0.038	-0.676	0.617	0.038

* Correlations are 'intraset correlations' of ter Braak (1986). The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case the site detection rate of birds, per unit change in the corresponding environmental variables (Ter Braak, 1986).

Monte Carlo test results – eigenvalues

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	**Eigenvalue**	**Mean**	**Min**	**Max**	**p**
1	0.352	0.082	0.037	0.161	0.0010
2	0.099	0.040	0.022	0.079	
3	0.040	0.029	0.018	0.049	

p = proportion of randomised runs with eigenvalue greater than or equal to the observed eigenvalue; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Monte Carol test results – species-environment correlations

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	**Spp-Envt Corr.**	**Mean**	**Min**	**Max**	**p**
1	0.881	0.488	0.340	0.839	0.0010
2	0.717	0.585	0.337	0.858	
3	0.830	0.626	0.362	0.871	

p = proportion of randomised runs with species-environment correlation greater than or equal to the observed species-environment correlation; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Appendix 19 CCA of foraging group abundance and vegetation structure

Data matrices

Main matrix:	67 sites (rows) / 14 guilds (columns)
Second matrix:	67 sites (rows) / 14 VegStruc (columns)

Options selected

Axis scores centered and standardised to unit variance
Axes scaled to optimise representation of columns: guilds
Scores for guilds are weighted mean scores for sites)
Scores for graphing sites are linear combinations of VegStruc
Monte Carlo test: null hypothesis is no relationship between matrices
Random number seed: 5634

Axis summary statistics

Number of canonical axes: 3
Total variance ('inertia') in the species data: 0.6786

	Axis 1	**Axis 2**	**Axis 3**
Eigenvalue	0.183	0.026	0.014
Variance in species data			
% of variance explained	27.0	3.9	2.1
Cumulative % explained	27.0	30.9	32.9
Pearson Correlation, Spp-Envt*	0.882	0.680	0.614
Kendall (Rank) Corr., Spp-Envt	0.698	0.537	0.444

* Correlation between sample scores for an axis derived from the species data and the sample scores that are linear combinations of the environmental variables. Set to 0.000 if axis is not canonical.

Multiple regression results:

Regression of sites in guilds space on VegStruc

	Canonical Coefficients						
	Standardised			**Original Units**			
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**	**S.Dev**
1 V_HtMean	1.196	0.843	0.493	0.168	0.118	0.069	0.712E+01
2 V_St_L5	0.352	-1.467	1.240	0.013	-0.056	0.047	0.262E+02
3 V_St_L4	0.463	-0.168	-0.249	0.005	-0.002	-0.003	0.920E+02
4 V_St_L3	0.111	0.303	-0.324	0.000	0.001	-0.001	0.347E+03
5 V_St_L2	-0.019	-0.258	-0.706	0.000	0.000	0.000	0.654E+04
6 V_BA_L5	0.014	0.315	0.017	0.003	0.065	0.004	0.482E+01
7 V_BA_L4	-0.747	-0.062	-1.706	-0.294	-0.025	-0.672	0.254E+01
8 V_CV_L5	-0.264	0.527	-1.031	0.000	0.000	0.000	0.209E+05
9 V_CV_L4	0.447	-0.491	1.666	0.000	0.000	0.000	0.836E+04
10 V_CV_L3	0.221	-0.809	0.405	0.000	0.000	0.000	0.351E+04
11 V_CV_L2	-0.064	0.122	0.335	0.000	0.000	0.000	0.778E+03
12 V_CV_L1	-0.116	-0.230	-0.205	0.000	0.000	0.000	0.372E+04

Final scores and raw data totals (weights) for 14 guilds

	Axis 1	Axis 2	Axis 3	Totals
1 Carnivor	-0.151656	-0.083922	-0.251921	85.0000
2 Frugivor	0.232104	0.080963	-0.261748	442.0000
3 Granivor	-1.249848	0.076913	-0.078457	609.0000
4 Ground F	-1.501917	-0.311137	-0.112925	17.0000
5 I-Aerial	0.318985	0.765241	0.430217	42.0000
6 I-Foliag	0.074263	0.179591	-0.028191	1588.0000
7 I-Grass	-2.068072	0.380316	0.025675	169.0000
8 I-Hawker	0.374498	0.090452	-0.067176	972.0000
9 I-Trunk	0.592956	0.459935	0.014526	333.0000
10 Ins/Nect	0.003265	-0.155913	-0.021329	4735.0000
11 Nectariv	0.030208	-0.035097	0.208807	2077.0000
12 Omnivore	-1.036154	0.319092	0.390536	63.0000
13 Predator	0.497245	0.129071	0.003116	693.0000
14 TreeSeed	-0.008586	0.019627	-0.202340	444.0000

Correlations and biplot scores for 12 VegStruc

	Correlations*			Biplot Scores		
Variable	Axis 1	Axis 2	Axis 3	Axis 1	Axis 2	Axis 3
1 V_HtMean	0.919	0.275	0.006	0.919	0.275	0.006
2 V_St_L5	0.701	0.118	0.258	0.701	0.118	0.258
3 V_St_L4	0.429	-0.527	-0.455	0.429	-0.527	-0.455
4 V_St_L3	-0.116	-0.209	-0.363	-0.116	-0.209	-0.363
5 V_St_L2	0.450	0.078	-0.578	0.450	0.078	-0.578
6 V_BA_L5	0.683	0.124	0.225	0.683	0.124	0.225
7 V_BA_L4	0.579	-0.509	-0.316	0.579	-0.509	-0.316
8 V_CV_L5	0.643	0.141	0.290	0.643	0.141	0.290
9 V_CV_L4	0.530	-0.596	-0.077	0.530	-0.596	-0.077
10 V_CV_L3	-0.388	-0.611	-0.003	-0.388	-0.611	-0.003
11 V_CV_L2	0.278	0.376	-0.228	0.278	0.376	-0.228
12 V_CV_L1	0.557	0.180	-0.005	0.557	0.180	-0.005

* Correlations are 'intraset correlations' of ter Braak (1986)

Monte Carlo test results – eigenvalues

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	Eigenvalue	Mean	Min	Max	p
1	0.183	0.038	0.015	0.078	0.0010
2	0.026	0.019	0.009	0.039	
3	0.014	0.013	0.007	0.022	

p = proportion of randomised runs with eigenvalue greater than or equal to the observed eigenvalue; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Monte Carlo test results – species-environment correlations

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	Spp-Envt Corr.	Mean	Min	Max	p
1	0.882	0.466	0.294	0.692	0.0010
2	0.680	0.513	0.250	0.746	
3	0.614	0.464	0.243	0.717	

Appendix 20 CCA site bird detection rates and landscape variables

Data matrices

Main matrix: 67 sites (rows) / 62 species (columns)
Second matrix: 67 sites (rows) / 7 landscap (columns)

Options selected

Axis scores centered and standardised to unit variance.
Axes scaled to optimise representation of columns: species (i.e.
Scores for species are weighted mean scores for sites).
Scores for graphing sites are linear combinations of landscap.
Monte Carlo test: null hypothesis is no relationship between matrices. Random number seed: 4614

Axis summary statistics

Number of canonical axes: 3
Total variance ('inertia') in the species data: 1.8360

	Axis 1	**Axis 2**	**Axis 3**
Eigenvalue	0.416	0.067	0.052
Variance in species data			
% of variance explained	22.7	3.7	2.8
Cumulative % explained	22.7	26.3	29.2
Pearson Correlation, Spp-Envt*	0.940	0.647	0.822
Kendall (Rank) Corr., Spp-Envt	0.674	0.436	0.619

* Correlation between sample scores for an axis derived from the species data and the sample scores that are linear combinations of the environmental variables. Set to 0.000 if axis is not canonical.

Multiple regression results:

Regression of sites in species space on landscap

	Canonical Coefficients						
	Standardised			**Original Units**			
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**	**S.Dev**
1 L_RVarea	0.875	-0.785	-0.791	0.021	-0.019	-0.019	0.408E+02
2 L_RVdist	-0.186	-1.119	-0.599	-0.001	-0.004	-0.002	0.253E+03
3 L_Coast	0.024	-0.179	0.896	0.002	-0.013	0.066	0.135E+02
4 L_BP	0.077	0.256	-0.178	0.000	0.000	0.000	0.659E+03
5 L_Mesic	0.095	-0.514	0.262	0.000	-0.001	0.001	0.390E+03

Final scores and raw data totals (weights) for 62 species

	Axis 1	**Axis 2**	**Axis 3**	**Totals**
1 BrTurk	-1.716948	-0.145295	-0.400279	1.3750
2 BaHon	0.321901	-0.392118	0.411768	9.6250
3 BBBut	0.608541	-0.110911	-0.055636	22.3750
4 BFCuSh	-0.108958	-0.058368	-0.254586	4.0000
5 BBut	-1.203635	0.968671	0.670508	1.1250
6 Goshawk	0.220098	0.273322	-0.344323	1.0000
7 BrHon	-1.569688	-0.200385	0.154981	18.2500
8 Quail	-1.738009	-1.352336	-0.502464	0.6250
9 Treecr	0.585308	0.035445	-0.413218	6.3750
10 BraKit	-0.967208	-1.020819	-0.670079	0.6250
11 BBHon	-0.318753	0.392993	-0.147779	2.2500
12 BrCuc	0.391766	0.456237	-0.752714	3.0000
13 BSDov	-0.861428	-0.107252	-0.301761	18.0000
14 BWKoo	0.469079	-0.125352	0.089469	3.0000
15 CBMan	-1.777299	-1.226640	-0.656967	1.1250
16 Cicada	0.540305	-0.028384	-0.120580	6.0000

Final scores and raw data totals (weights) for 62 species continued

	Axis 1	Axis 2	Axis 3	Totals
17 SpHawk	-0.038623	-0.160620	0.135778	2.1250
18 Coucal	-0.477282	-0.538633	-0.883322	2.6250
19 Crow	-0.156439	-0.340863	0.578648	3.5000
20 Drong	-0.329646	0.519234	-0.142797	5.1250
21 DHon	-0.950806	0.873679	0.163824	4.7500
22 FKing	0.531985	0.052636	-0.298999	14.5000
23 Babb	0.535157	0.133580	-0.111763	0.8750
24 Cist	-1.667273	-1.030217	-0.039063	6.8750
25 GSThr	0.623404	-0.097072	0.031248	14.8750
26 GrHon	-0.979661	0.613318	0.055746	8.3750
27 Bower	-0.424414	0.555067	-0.441652	4.2500
28 Koel	0.273690	0.127966	-0.612860	1.7500
29 LKook	0.499963	-0.126700	0.026216	12.2500
30 LdFly	0.295704	-0.014308	0.013442	27.7500
31 LBFly	0.535405	-0.024906	0.056776	28.8750
32 LFri	0.347047	-0.003885	-0.017686	38.6250
33 Mistl	0.270644	0.067805	-0.133468	29.6250
34 NFri	0.448696	-0.155692	0.240528	21.8750
35 OBOrio	0.402013	-0.034102	-0.368513	2.5000
36 PDov	-0.610216	0.099617	-0.031540	14.5000
37 PIPig	0.478471	0.301072	-1.163751	0.8750
38 PHRos	-0.150716	0.254289	0.113287	7.2500
39 PBut	0.386266	0.017348	-0.306657	1.2500
40 BeeE	0.011446	0.218783	-0.485261	4.7500
41 Pipit	-1.522519	-0.214879	0.879480	1.6250
42 Lorik	0.354565	-0.040223	0.026632	32.2500
43 RBFWr	-0.042821	-0.142799	-0.183241	9.5000
44 RBrFin	-1.233840	0.658260	0.709676	1.3750
45 RuWh	0.633750	-0.196809	0.238632	25.5000
46 RWParr	-0.035131	0.144349	-0.300846	12.0000
47 SCCoc	0.155014	0.255655	-0.332981	8.1250
48 StPar	-0.373816	0.050017	0.308818	21.6250
49 VSitt	0.660956	-0.121085	0.144706	5.7500
50 VTril	-1.261043	0.658297	-0.013952	1.0000
51 WBCuSh	0.199202	0.033016	-0.017225	32.6250
52 WBWood	-0.097626	0.069245	0.064746	0.8750
53 Weeb	-0.316426	0.172847	0.092125	3.6250
54 WhKit	-0.019540	-0.442472	0.208015	1.7500
55 WStHon	-1.807099	-1.176167	-0.863250	2.8750
56 WTHon	0.146179	0.086731	0.083858	48.2500
57 WWTril	0.267672	-0.128195	0.585005	1.3750
58 Sunbird	-1.473389	-0.034798	0.069500	23.2500
59 YHon	0.041586	0.096056	0.072660	31.2500
60 YOriol	0.205131	0.491377	-1.199371	1.1250
61 YSHon	-1.233998	0.772233	0.197703	8.6250
62 YTHon	0.737567	-0.357836	0.180630	3.2500

Correlations and biplot scores for 5 landscape context variables

	Correlations*			Biplot Scores		
Variable	Axis 1	Axis 2	Axis 3	Axis 1	Axis 2	Axis 3
1 L_RVarea	0.987	-0.022	-0.098	0.987	-0.022	-0.098
2 L_RVdist	-0.790	-0.553	-0.171	-0.790	-0.553	-0.171
3 L_Coast	0.300	-0.211	0.854	0.300	-0.211	0.854
4 L_BP	-0.221	-0.147	0.099	-0.221	-0.147	0.099
5 L_Mesic	-0.003	-0.708	0.275	-0.003	-0.708	0.275

* Correlations are 'intraset correlations' of ter Braak (1986). The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case the site detection rate of birds, per unit change in the corresponding environmental variables (Ter Braak, 1986).

Monte Carlo test results – eigenvalues

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	Eigenvalue	Mean	Min	Max	p
1	0.416	0.049	0.020	0.126	0.0010
2	0.067	0.025	0.015	0.053	
3	0.052	0.018	0.011	0.031	

p = proportion of randomised runs with eigenvalue greater than or equal to the observed eigenvalue; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations). p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Monte Carlo test results – species-environment correlations

		Randomised data			
	Real data	Monte Carlo test, 998 runs			
Axis	Spp-Envt Corr.	Mean	Min	Max	p
1	0.940	0.437	0.303	0.734	0.0010
2	0.647	0.536	0.276	0.798	
3	0.822	0.543	0.273	0.766	

p = proportion of randomised runs with species-environment correlation greater than or equal to the observed species-environment correlation; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations). p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Appendix 21 CCA Site bird detection rates and vegetation structure (mine rehabilitation sites only)

Data matrices

Main matrix: 31 sites (rows) / 57 species (columns)
Second matrix: 31 sites (rows) / 11 VegStruc (columns)

Options selected

Axis scores centered and standardised to unit variance
Axes scaled to optimise representation of columns: species
Scores for species are weighted mean scores for sites
Scores for graphing sites are linear combinations of VegStruc
Monte Carlo test: null hypothesis is no relationship between matrices
Random number seed: 4340

Axis summary statistics

Number of canonical axes: 3
Total variance ('inertia') in the species data: 1.5900

	Axis 1	**Axis 2**	**Axis 3**
Eigenvalue	0.282	0.119	0.079
Variance in species data			
% of variance explained	17.7	7.5	5.0
Cumulative % explained	17.7	25.2	30.2
Pearson Correlation, Spp-Envt*	0.922	0.880	0.909
Kendall (Rank) Corr., Spp-Envt	0.785	0.712	0.673

* Correlation between sample scores for an axis derived from the species data and the sample scores that are linear combinations of the environmental variables. Set to 0.000 if axis is not canonical.

Multiple regression results:

Regression of sites in species space on VegStruc

	Canonical Coefficients						
	Standardized			**Original Units**			
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**	**S.Dev**
1 V_HtMean	0.838	-1.114	-0.051	0.336	-0.446	-0.020	0.250E+01
2 V_St_L4	0.272	2.637	1.164	0.002	0.018	0.008	0.146E+03
3 V_St_L3	0.054	-0.302	0.548	0.000	-0.001	0.001	0.451E+03
4 V_St_L2	0.030	-0.105	-0.484	0.000	0.000	0.000	0.452E+04
5 V_BA_L4	0.750	1.548	3.392	0.181	0.374	0.820	0.414E+01
6 V_CV_L4	-1.155	-2.850	-4.801	0.000	0.000	0.000	0.142E+05
7 V_CV_L3	0.256	-0.290	-0.545	0.000	0.000	0.000	0.444E+04
8 V_CV_L2	0.123	-0.394	0.170	0.000	-0.001	0.000	0.442E+03
9 V_CV_L1	-0.286	-0.296	-0.504	-0.001	-0.001	-0.001	0.401E+03

Final scores and raw data totals (weights) for 57 species

	Axis 1	**Axis 2**	**Axis 3**	**Totals**
1 Aus_BrTu	-0.334225	-0.423530	-0.355763	1.3750
2 Ban_Hon	-0.489537	-0.257809	-0.079065	1.6250
3 BB_Butch	0.719776	-0.274000	0.416114	0.2500
4 BF_CuShr	-0.693169	-0.265318	-0.353814	1.1250
5 Bl_Butch	0.958562	0.746976	-0.544388	1.1250
6 Br_Gosh	0.445944	-0.617003	-0.139146	0.2500

Final scores and raw data totals (weights) for 57 species

	Axis 1	Axis 2	Axis 3	Totals
7 Br_Hon	-0.423075	-0.148509	-0.096118	18.2500
8 Br_Quail	-0.421020	-0.494388	0.412802	0.6250
9 Brah_Kit	-0.250911	0.019955	-0.737790	0.3750
10 BrB_Hon	0.852165	0.037332	0.086261	1.1250
11 Brush_Cu	0.616351	-0.106669	2.757219	0.1250
12 BS_Dove	-0.184917	-0.349382	-0.110872	11.7500
13 BW_Kook	-0.150762	1.302664	0.113481	0.2500
14 ChBr_Man	-0.898565	-0.535063	-0.282367	1.1250
15 Cicada	0.887387	0.216257	-0.309968	0.2500
16 Col_Spar	-1.204573	1.022089	-0.023886	0.6250
17 Coucal	-0.763950	-0.281084	-0.640002	1.1250
18 Crow	-0.868206	-0.190373	0.009856	1.5000
19 Drongo	0.707411	0.436867	-0.040925	2.3750
20 Dus_Hon	0.815698	0.638964	-0.239195	3.8750
21 For_King	0.543651	-0.301323	0.724540	0.7500
22 GH_Cisti	-1.418105	0.423939	0.072094	6.8750
23 Gr_ShrTh	0.616351	-0.106669	2.757219	0.2500
24 Grac_Hon	0.340022	0.055075	-0.314804	6.7500
25 Gt_Bower	0.465210	0.175001	-0.069023	2.1250
26 Koel	0.797041	0.108615	0.712428	0.3750
27 La_Kook	0.422871	-0.151845	0.216848	0.8750
28 Lead_Fly	0.390425	0.083815	0.011249	4.5000
29 LemB_Fly	0.836884	0.335233	-0.000477	1.6250
30 Lit_Fria	0.388441	-0.069068	0.352235	5.5000
31 Mistleto	0.018395	-0.155132	0.246021	5.3750
32 Noi_Fria	0.173031	-0.514440	-0.094809	2.2500
33 OB_Oriol	0.270097	-0.079873	0.907978	0.2500
34 P_Dove	-0.113686	-0.143937	0.092158	8.6250
35 PH_Rose	0.278431	-0.198566	0.069111	2.8750
36 Pi_Butch	-0.735302	0.021872	-0.794468	0.1250
37 R_Bee	0.435653	-0.209518	0.888997	1.5000
38 R_Pipit	-2.071168	1.705803	0.415246	1.6250
39 Ra_Lorik	0.540501	0.132084	-0.014116	4.5000
40 RB_FWren	0.285475	-0.578408	0.414475	2.7500
41 RBr_Fin	-0.502753	0.495645	0.335334	1.3750
42 Ruf_Whis	0.329376	-0.431044	-0.397307	0.6250
43 RW_Parr	0.306740	-0.363722	-0.130453	3.6250
44 SC_Cocka	0.702215	0.950073	-0.161541	1.6250
45 St_Pard	-0.297828	0.103481	0.113526	10.5000
46 Var_Tril	0.360806	0.370333	-0.804844	0.8750
47 WB_CuShr	0.193030	0.004494	0.134308	7.0000
48 WBst_Woo	-0.059476	-0.042398	0.981375	0.2500
49 Weebill	0.314157	-0.098899	0.877942	2.0000
50 Whis_Kit	-1.836874	1.034910	-0.147377	0.5000
51 WStr_Hon	-0.140789	-0.736797	-0.413055	2.8750
52 WThr_Hon	0.439409	0.166027	0.061351	12.2500
53 WW_Trill	-0.074441	-1.191580	-0.141287	0.3750
54 YB_Sun	-0.208150	-0.166519	-0.065276	22.6250
55 Yel_Hon	0.335879	0.199281	0.194736	9.2500
56 Yel_Orio	1.361457	0.559302	0.002538	0.1250
57 YSp_Hon	0.565542	0.373730	-0.454667	7.8750

Correlations and biplot scores for 9 Vegetation Structure variables

	Correlations*			Biplot Scores		
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**
1 V_HtMean	0.915	0.052	-0.238	0.915	0.052	-0.238
2 V_St_L4	0.722	0.488	-0.308	0.722	0.488	-0.308
3 V_St_L3	0.408	-0.252	0.252	0.408	-0.252	0.252
4 V_St_L2	0.483	0.107	-0.112	0.483	0.107	-0.112
5 V_BA_L4	0.650	0.509	-0.309	0.650	0.509	-0.309
6 V_CV_L4	0.637	0.488	-0.377	0.637	0.488	-0.377
7 V_CV_L3	0.374	-0.198	-0.289	0.374	-0.198	-0.289
8 V_CV_L2	-0.094	-0.354	0.420	-0.094	-0.354	0.420
9 V_CV_L1	-0.573	-0.387	-0.319	-0.573	-0.387	-0.319

* Correlations are 'intraset correlations' of ter Braak (1986). The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case the site detection rate of birds, per unit change in the corresponding environmental variables (Ter Braak, 1986).

Monte Carlo test results – eigenvalues

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	**Eigenvalue**	**Mean**	**Min**	**Max**	**p**
1	0.282	0.121	0.069	0.205	0.0010
2	0.119	0.077	0.055	0.106	
3	0.079	0.062	0.044	0.084	

p = proportion of randomised runs with eigenvalue greater than or equal to the observed eigenvalue; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Monte Carlo test results – species-environment correlations

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	**Spp-Envt Corr.**	**Mean**	**Min**	**Max**	**p**
1	0.922	0.738	0.593	0.904	0.0010
2	0.880	0.830	0.591	0.952	
3	0.909	0.824	0.563	0.959	

p = proportion of randomised runs with species-environment correlation greater than or equal to the observed species-environment correlation; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Appendix 22 CCA Site bird detection rates and vegetation structure (native forest sites only)

Data matrices

Main matrix: 36 sites (rows) / 46 species (columns)
Second matrix: 36 F_sites (rows) / 13 V_struct (columns)

Options selected

Axis scores centered and standardised to unit variance
Axes scaled to optimise representation of rows: sites
Scores for sites are weighted mean scores for species
Scores for graphing sites are linear combinations of V_struct
Monte Carlo test: null hypothesis is no relationship between matrices
Random number seed: 1981

Axis summary statistics

Number of canonical axes: 3
Total variance ('inertia') in the species data: 0.6416

	Axis 1	**Axis 2**	**Axis 3**
Eigenvalue	0.064	0.051	0.038
Variance in species data			
% of variance explained	10.0	8.0	5.9
Cumulative % explained	10.0	18.0	23.8
Pearson Correlation, Spp-Envt*	0.899	0.891	0.868
Kendall (Rank) Corr., Spp-Envt	0.597	0.695	0.638

* Correlation between sample scores for an axis derived from the species data and the sample scores that are linear combinations of the environmental variables. Set to 0.000 if axis is not canonical.

Multiple regression results:

Regression of sites in species space on V_struct

	Canonical Coefficients						
	Standardised			**Original Units**			
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**	**S.Dev**
1 V_HtMean	-0.048	-0.075	-0.148	-0.024	-0.038	-0.075	0.199E+01
2 V_St_L5	0.161	-0.268	-0.291	0.009	-0.015	-0.017	0.176E+02
3 V_St_L4	0.142	-0.135	0.031	0.002	-0.002	0.000	0.623E+02
4 V_St_L3	0.223	-0.111	0.064	0.001	0.000	0.000	0.269E+03
5 V_St_L2	-0.108	-0.061	0.060	0.000	0.000	0.000	0.624E+04
6 V_BA_L5	-0.016	0.100	0.262	-0.005	0.029	0.076	0.346E+01
7 V_BA_L4	-0.220	0.348	-0.118	-0.155	0.245	-0.083	0.142E+01
8 V_CV_L5	-0.104	0.273	0.016	0.000	0.000	0.000	0.167E+05
9 V_CV_L4	0.097	-0.096	0.100	0.000	0.000	0.000	0.443E+04
10 V_CV_L3	-0.298	0.010	-0.101	0.000	0.000	0.000	0.918E+03
11 V_CV_L2	-0.013	0.013	0.028	0.000	0.000	0.000	0.795E+03
12 V_CV_L1	0.066	0.005	-0.085	0.000	0.000	0.000	0.300E+04

Final scores and raw data totals (weights) for 46 species

	Axis 1	Axis 2	Axis 3	Totals
1 Ban_Hon	0.974766	1.904032	-1.255676	8.0000
2 BB_Butch	-0.331266	0.252515	0.105282	22.1250
3 BF_CuShr	-1.987333	0.036692	0.364410	2.8750
4 Br_Gosh	-1.178268	-1.379422	0.173305	0.7500
5 Br_Tree	1.576364	-2.812878	-2.313247	6.3750
6 BrB_Hon	-2.033759	-0.363961	1.362509	1.1250
7 Brush_Cu	-0.532969	-5.024201	2.137817	2.8750
8 BS_Dove	-1.657461	-1.297125	-0.508039	6.2500
9 BW_Kook	-0.049965	1.763478	0.751445	2.7500
10 Cicada	-0.795997	0.967602	-0.699548	5.7500
11 Col_Spar	0.502586	-0.134753	1.134303	1.5000
12 Coucal	0.311649	-3.796044	2.969920	1.5000
13 Crow	1.443102	1.760116	-0.065270	2.0000
14 Drongo	-3.427129	0.609139	-0.083639	2.7500
15 Dus_Hon	-3.957081	-3.875572	-1.071192	0.8750
16 For_King	-0.458553	0.063743	-0.534637	13.7500
17 GC_Babb	0.367978	-0.892709	3.689720	0.8750
18 Gr_ShrTh	1.151426	-1.153703	-0.584697	14.6250
19 Gt_Bower	-8.068213	1.600035	-3.363365	2.1250
20 Koel	-2.872302	1.623614	0.915480	1.3750
21 La_Kook	-0.047778	0.327078	-1.297252	11.3750
22 Lead_Fly	-0.030076	0.202568	0.351267	23.2500
23 LemB_Fly	0.037042	0.563819	0.381847	27.2500
24 Lit_Fria	0.106541	0.065650	0.226655	33.1250
25 Mistleto	-0.275564	-0.108800	-0.126470	24.2500
26 Noi_Fria	0.715497	0.575131	0.261571	19.6250
27 OB_Oriol	-0.654167	0.276690	4.019806	2.2500
28 P_Dove	0.337152	-3.011460	0.207329	5.8750
29 PdIm_Pig	-6.022048	0.694438	-1.583494	0.8750
30 PH_Rose	1.340687	-0.503891	0.616413	4.3750
31 Pi_Butch	0.823051	1.420672	11.878898	1.1250
32 R_Bee	-3.174791	-0.108529	1.636831	3.2500
33 Ra_Lorik	0.267744	0.057261	-0.262657	27.7500
34 RB_FWren	0.756206	-1.656534	-1.632733	6.7500
35 Ruf_Whis	0.580297	1.001976	0.406429	24.8750
36 RW_Parr	-0.624538	-0.853393	-1.252867	8.3750
37 SC_Cocka	0.228607	-1.213268	0.147392	6.5000
38 St_Pard	0.611771	-0.944119	0.301031	11.1250
39 V_Sitt	-0.034303	-0.020846	0.655186	5.7500
40 WB_CuShr	-0.322061	0.052427	0.020306	25.6250
41 WBst_Woo	0.293753	-0.438465	0.619664	0.6250
42 Whis_Kit	-0.550828	-0.276209	0.669234	1.2500
43 WThr_Hon	0.130841	0.150611	0.095339	36.0000
44 WW_Trill	0.913683	0.223591	-1.188513	1.0000
45 Yel_Hon	-0.262899	0.043619	0.204573	22.0000
46 YTin_Hon	3.294322	3.577280	-2.792854	3.2500

Correlations and biplot scores for 12 Vegetation structure variables

	Correlations*			Biplot Scores		
Variable	**Axis 1**	**Axis 2**	**Axis 3**	**Axis 1**	**Axis 2**	**Axis 3**
1 V_HtMean	-0.272	0.210	-0.668	-0.069	0.048	-0.130
2 V_St_L5	0.336	0.396	-0.442	0.085	0.090	-0.086
3 V_St_L4	-0.336	0.211	0.240	-0.085	0.048	0.047
4 V_St_L3	-0.449	-0.428	0.212	-0.113	-0.097	0.041
5 V_St_L2	-0.593	-0.326	0.122	-0.150	-0.074	0.024
6 V_BA_L5	0.272	0.481	-0.387	0.069	0.109	-0.075
7 V_BA_L4	-0.483	0.553	0.117	-0.122	0.125	0.023
8 V_CV_L5	0.241	0.513	-0.301	0.061	0.116	-0.058
9 V_CV_L4	-0.230	0.692	0.272	-0.058	0.157	0.053
10 V_CV_L3	-0.628	-0.349	0.119	-0.159	-0.079	0.023
11 V_CV_L2	-0.359	-0.064	0.193	-0.091	-0.014	0.038
12 V_CV_L1	0.355	-0.183	-0.446	0.090	-0.041	-0.087

* Correlations are 'intraset' correlations of ter Braak (1986). The correlation coefficients (intraset correlations) relate to the rate of change in community composition, in this case the site detection rate of native forest birds, per unit change in the corresponding environmental variables (Ter Braak, 1986).

Monte Carlo test results – eigenvalues

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	**Eigenvalue**	**Mean**	**Min**	**Max**	**p**
1	0.064	0.047	0.030	0.066	0.0040
2	0.051	0.036	0.025	0.054	
3	0.038	0.029	0.019	0.041	

p = proportion of randomised runs with eigenvalue greater than or equal to the observed eigenvalue; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Monte Carlo test results – species-environment correlations

	Real data	Randomised data Monte Carlo test, 998 runs			
Axis	**Spp-Envt Corr.**	**Mean**	**Min**	**Max**	**p**
1	0.899	0.835	0.724	0.930	0.0340
2	0.891	0.806	0.673	0.945	
3	0.868	0.793	0.650	0.949	

p = proportion of randomised runs with species-environment correlation greater than or equal to the observed species-environment correlation; i.e., p = (1 + no. permutations >= observed)/(1 + no. permutations) p is not reported for axes 2 and 3 because using a simple randomisation test for these axes may bias the p values.

Appendix 23 Mean canopy volume[1] of woody species m^3/ha by land unit

	Mine rehabilitation					Native Forest		
						Pre-mining	Post-mining landscape analogue	
Plant name	1 -2	3 - 4	5- 8	9 - 16	>16	LU2B	LU2C	LU5K
Acacia auriculiformis	3	216	1501	841				
Acacia aulacocarpa		327						
Acacia brassi		38	297	26	70			
Acacia crassicarpa	9	150	403	649	7	4		
Acacia dimidiata	2	33	39	97				
Acacia holosericea			0	0				
Acacia legnota	0		89		0			
Acacia leptocarpa	0	309	1046	433	775			
Acacia mangium	16	48	90	3906	3077			
Acacia platycarpa		230	44		67			
Acacia rothii	130	722	2096	527	741	748	2513	518
Acacia simsii	148	714	96	7	154			
Acacia spp.	40	519	889	784	76			21
Acacia torulosa	1254	2813	1248	158	77			
Adenanthera abrosperma	0				22	1		1
Alphitonia excelsa	9	5	15	51	4029	48	137	21
Alstonia actinophylla				84	419	14	129	
Alstonia spp.		3	144	1306	7646			
Anthobolus filicifolius			0			0		
Atalaya variifolia								31
Brachychiton sp.	0			0		31		2
Breynia sp.		0	0		1	5	7	0
Callitris spp.					396			
Canarium australianum						49		
Capparis sp.						4		
Casuarina sp.					1265			
Clerodendrum sp.	1	13	0	3	18	3		5
Cochlospermum sp.			2	26	91			
Corymbia nesophila	6		34	37	64	10721	8919	9806
Corymbia polycarpa		2	1	24	1368			
Corymbia spp.	0					46		145
Corymbia stockeri	0	0	39	3	6	2102	1285	2068
Croton arnhemicus	0	0	0		6	4	0	4
Dodonaea spp.	226	4068	915	3416	260			
Dolicandrone heterophylla			3			3	3	9
Erythrophleum chlorostachys	0	0		9	154	8068	1886	3003
Eucalyptus alba		7	10	50	192			
Eucalyptus brassiana				224	5595			
Eucalyptus camaldulensis				7	89			
Eucalyptus confertiflora			0					
Eucalyptus cullenii			5	7	94			
Eucalyptus leptophleba			2	19	172	142		
Eucalyptus miniata	17	30	7	40	4			
Eucalyptus papuana					132			
Eucalyptus ptychocarpa				1				
Eucalyptus tetrodonta	1	20	22	174	1534	31171	20915	11763
Ficus sp.	5	20	96	144	237	93		8
Flueggia virosa	0	1	62	5	104	70		
Gmelina sp.						6		

Mean canopy volume[1] of woody species m^3/ha by land unit - continued

						Native Forest		
	Mine rehabilitation					Pre-mining	Post-mining landscape analogue	
Plant name	**1 -2**	**3 - 4**	**5- 8**	**9 - 16**	**>16**	**LU2B**	**LU2C**	**LU5K**
Grevillea banksii					0			
Grevillea glauca			47	50	115	85	111	
Grevillea heliosperma	13	16	256	216	561			
Grevillea parallela						71		277
Grevillea pteridifolia	106	1596	1070	1052	2095			
Grewia retusa		0	0	1	1	3	7	2
Hibbertia sp.					0	2		5
Hibiscus sp.						2		
Indigofera pratensis						3		
Khaya senegalensis		0						
Leptospermum longifolium				8	81			
Leucaena leucocephala	1			134	0			
Livistona muelleri				10	0	9	240	
Lophostemon suaveolens			37	35	152		1479	64
Maytenus cunninghamii						6	38	
Melaleuca dealbata				1				
Melaleuca leucadendron				6	1265			
Melaleuca sp.	0		4	6	462			
Melaleuca stenostachya					50	0	76	
Melaleuca symphiocarpa		1		65	529			
Melaleuca viridiflora			1	8	89			
Morinda reticulata		0	0		0	5	1	10
Owenia vernicosa						24		
Pandanus spiralis				31	52	14		18
Parinari nonda				129	2089	663	395	380
Persoonia falcata						1		
Planchonia careya	0	0	5	3	94	243	295	638
Pogonobolus reticulatus					5	24	63	4
Pouteria pohlmaniana						30	5	
Sida retusa				4				
Siphonodon sp.			0		58	21	1	
Sterculia quadrifida				0	65			
Stylosanthes spp.	1	8	5	49	0			
Syzygium suborbiculare						73	6	
Timonius timon			15		25	8		
Trema aspera	217	911	254	60	36			
Unknown 'cheese'						8	0	0
Unknown eucalypt	11	25	58	209	319	0		
Unknown 'w'	1	0	5	21		6	0	1
Wrightia saligna						1		
Xanthostemon sp.				1				
Xylomelum scottianum	0	0			63	800	948	776

1. Values have been rounded to units.

Appendix 24 Mean canopy volume[1] of grasses (m^3/ha) by land unit

	Mine rehabilitation					Native Forest		
						Pre-mining	Post-mining landscape analogue	
Plant name	**1 - 2**	**3 - 4**	**5 - 8**	**9 - 16**	**> 16**	**LU2B**	**LU2C**	**LU5K**
Alloteropsis semialata					1	2662	34057	1273
Andropogon gayanus		825	14286	1181	44			
Botriochloa spp.	1	85						
Brachiaria decumbens		9	27	1615	3			
Brachiaria humidicola				51	48			
Brachiaria pubigera				1				
Cymbopogon spp				6	6	82		
Ectrosia spp.	121	8	415	20	1			
Eragrostis spp.		1	1	13	7			
Eriachne spp.	1361	136	5469	107	1493	5283	46002	4517
Eulalia spp.						316		
Haemotodus spp.						1		1
Heteropogon contortus	3761	8	3	62				
Heteropogon triticeus	1408	13709	2353	2	352	71411	14361	89460
Lomandra spp.			1	1	4	352	166	421
Mnesithea rottboellioides	49	12			1	1978	41	3813
Panicum maximum				79				
Panicum spp.		1						
Paspalum spp.				330				
Pseudopogonatherum contortum			1					
Sarga plumosum var plumosum	4213	2102	1098	260	1342	57267	397	16938
Setaria spp.				1				
Unknown 'spindly'						96		
Unknown spp.		71		1	1	944	164	643

1. Values have been rounded to units