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Does post-mining rehabilitation restore habitat equivalent to that removed by mining? A case study from the monsoonal tropics of northern Australia

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Abstract

Context. Rehabilitation is increasingly being promoted as a strategy for minimising and even reversing biodiversity loss. Many rehabilitation strategies that aim to provide habitat focus entirely on establishing vegetation. Successful vegetation establishment, however, does not necessarily provide habitat that is ecologically equivalent to that removed by vegetation clearing. Quantitative understanding of faunal responses to rehabilitation is required if rehabilitation techniques are to be refined and deliver desired biodiversity outcomes.

Aims. I aimed to assess the extent to which post-mining rehabilitation restores bird habitat equivalent to that removed in the mining process on the Weipa bauxite plateau.

Methods. The composition, abundance and richness of bird assemblages were compared between native forest sites and a 23-year chronosequence of post-mining rehabilitation sites. Native forest sites were made up of three Weipa bauxite plateau land units, including the land unit that represents pre-mining native forest, and two land units that are considered to be potential analogues for the post-mining landscape.

Key results. Bird abundance and bird species richness increased with rehabilitation age. Bird species richness in the two oldest age classes of mine rehabilitation was similar to values obtained from pre-mining native forest and post-mining landscape analogue sites. The composition of bird assemblages, however, was significantly different. Of all the bird species observed, 25% occurred exclusively in native forest sites, 19% occurred exclusively in mine-rehabilitation sites, and the remaining 56% were recorded in both native forest and mine-rehabilitation sites. Site bird-detection rates were significantly related to site vegetation structure, with inter-specific differences in bird response.

Conclusions. Post-mining rehabilitation at Weipa has partially made up for the loss of habitat caused by clearing for mining. Twenty-three years after rehabilitation commenced, however, a clear residual impact on biodiversity remains, with a third of native forest birds absent from mine rehabilitation, including several native forest specialists.

Implications. Rehabilitation can partially make up for biodiversity losses caused by the initial loss of habitat. There is no evidence, however, that rehabilitation can achieve 'no net loss'. Reliance on rehabilitation to achieve conservation outcomes does not address the fact that many fauna species require resources that are found only in mature forest.

Additional keywords: bauxite mining, bird assemblages, chronosequence, *Eucalyptus tetrodonta*, habitat restoration, northern Australia, post-mining rehabilitation.

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Introduction

Habitat loss, including habitat conversion, arising from human activities is recognised as one of the key drivers of biodiversity decline (Primack 2002; Sattler and Creighton 2002; Millennium Ecosystem Assessment 2005). In Australia, although most states have introduced legislation aimed at reducing broad-scale 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 a key strategy for minimising or even reversing human impacts on biodiversity (Australian and New Zealand Environment and Conservation Council 1996). It is also increasingly being referred to in biodiversity offset policies as a strategy for achieving 'no net loss' or even 'net gain' of biodiversity.

The terms rehabilitation, reconstruction and restoration all imply replacement of habitat, although none quantifies the biodiversity losses and gains. Most rehabilitation strategies, however, are passive with respect to fauna, i.e. they focus entirely on establishing vegetation. Vegetation establishment does provide habitat; however, the benefits of rehabilitation differ from one species to another. Successful vegetation establishment does not necessarily provide habitat that is ecologically equivalent to the habitat lost. In the case of birds, some groups of foraging specialists and habitat specialists are more sensitive to habitat loss than are others (Sekercioglu *et al*. 2004). Likewise, some bird species benefit from rehabilitation more than others and at different stages of the rehabilitation process. Bird assemblages related to different stages of vegetation succession have been found in (1) natural succession following logging or clearing (May 1982; Fisher 2001; Venier and Pearce 2005), (2) habitat reconstruction following agriculture (Martin *et al*. 2004) and (3) habitat restoration following mining (Armstrong and Nichols 2000; Nichols and Nichols 2003; Nichols and Grant 2007). Some species are exclusively associated with old-growth vegetation and do not even return to late successional forest (Loyn 1985; Sallabanks *et al*. 2006; Barlow *et al*. 2007). It may be that the species that are most sensitive to habitat loss, and therefore most in need of habitat restoration, are the last species to benefit from rehabilitation, if they return at all.

Landscape variables, including the size of rehabilitation patches (Fink *et al*. 2008) and the area of remnant vegetation in the landscape (Miller and Hobbs 2007; Lindenmayer *et al*. 2010), have all been shown to alter the habitat value of rehabilitation. Although broad patterns of response can be identified, bird species respond uniquely to vegetation and landscape attributes.

Empirical information about faunal response to rehabilitation is required across a range of rehabilitation strategies, ecosystem types and climate zones to quantify biodiversity losses and gains. Understanding faunal response to rehabilitation will inform the development of improved rehabilitation techniques. The pattern of faunal response to rehabilitation in the monsoonal tropics of Australia is not well known, but see Brady and Noske (2009). Improved understanding of faunal response to rehabilitation is particularly needed in Australia's monsoonal tropics, given the escalating pressure to develop northern Australia, including large mining proposals. An evidence-based approach to assessing the biodiversity outcomes of rehabilitation is also a necessary step to quantifying the extent to which rehabilitation can be considered to make up for the impacts of development. I compared bird assemblages and vegetation in pre-mining native forest and post-mining rehabilitation on the Weipa bauxite plateau. Here, I report the key findings in relation to bird assemblages and discuss the implications for rehabilitation practice and policy.

Materials and methods

The study area

The Weipa bauxite plateau (centred on 12°40'S, 141°55'E) on the north-western coast of Cape York Peninsula, Queensland, Australia, is the world's largest proven bauxite reserve (Taylor *et al*. 2008), covering an area of ~1.1 million ha. The distribution of a unique ecosystem of tall woodland (i.e. foliage cover 10–30% and vegetation height >30 m), Regional Ecosystem 3.5.2 (R.E. 3.5.2) dominated by Darwin stringybark (*Eucalyptus tetrodonta*) is correlated with the bauxite plateau (Environmental Protection Agency 2005).

Rio Tinto Alcan (previously Comalco) has been mining bauxite on the Weipa bauxite plateau continuously since the early 1960s. Since 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). The current goal 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). The mining and rehabilitation process has resulted in a mosaic of rehabilitated areas of different age. At the time of commencement of the present study in 2006, ~10 500 ha had been cleared for mining, of which 7500 ha had been rehabilitated with the aim of establishing native vegetation.

Mining at Weipa is a strip-mining operation. One or two years before mining, native forest is cleared and the vegetation is root-raked, windrowed and burnt. Immediately before mining, regrowth is cleared and a layer of topsoil ~50 cm deep is removed (Comalco Mining and Refining 2004). The ore body, which lies immediately below the topsoil, is then removed down to an ironstone layer, lowering the original land surface on average by 2–3 m (Taylor *et al*. 2008). In areas where the goal is to reinstate native ecosystems, either freshly stripped topsoil or stockpiled topsoil is spread on the mine floor to a depth of ~30 cm. Sites are ripped to control soil erosion and break up the compacted mine floor and ironstone (Comalco Mining and Refining 2004). Following early wet-season rains, sites are cultivated with a disc plough, fertilised and direct-seeded with a mix of native plant species. Many of the species included in the seed mix do not belong to the plant communities of the Weipa bauxite plateau. Species used in the rehabilitation mix at Weipa have been selected for reliability of germination from seed and high growth rates, and included *Acacia*, *Grevillea* and *Eucalyptus* species from around Australia.

Study design

Bird assemblages and vegetation were surveyed at 67 sites, made up of 36 native forest sites and 31 mine rehabilitation sites. Native forest sites were stratified into three land units that are subdivisions of Regional Ecosystem 3.5.2 (Gunness *et al*. 1987). The three land units were (1) Land unit 2B, which is the main source of commercial bauxite and represents the pre-mining native forest (28 sites) and (2) two land units that are potential landscape analogues for the post-mining landscape, Land units 2C and 5K (4 sites of each land unit). Consideration of access to soil water is important for identifying appropriate reference ecosystems because it is a key determinant of vegetation composition and structure (Pedley and Isbell 1971; Specht *et al*. 1977; Bowman and Minchin 1987). Because of the changes in soil depth and hydrological characteristics that are caused by mining, Land units 2C (LU 2C) and 5K (LU 5K) have been nominated as more realistic reference ecosystems for post-mining rehabilitation at Weipa (Reddell and Hopkins 1994). Land unit 2C is tall shrubby woodland that occurs on the bauxite plateau in areas where drainage is impeded, and Land unit 5K is grassy woodland that occurs on lateritic slopes of the bauxite plateau (Fig. 1). A process of *a priori* stratification was used to represent variation in vegetation composition and structure within pre-mining native forest (LU2B).

Rehabilitation sites were selected only from areas that had been rehabilitated with the aim of establishing native vegetation. The 23-year chronosequence of rehabilitation sites

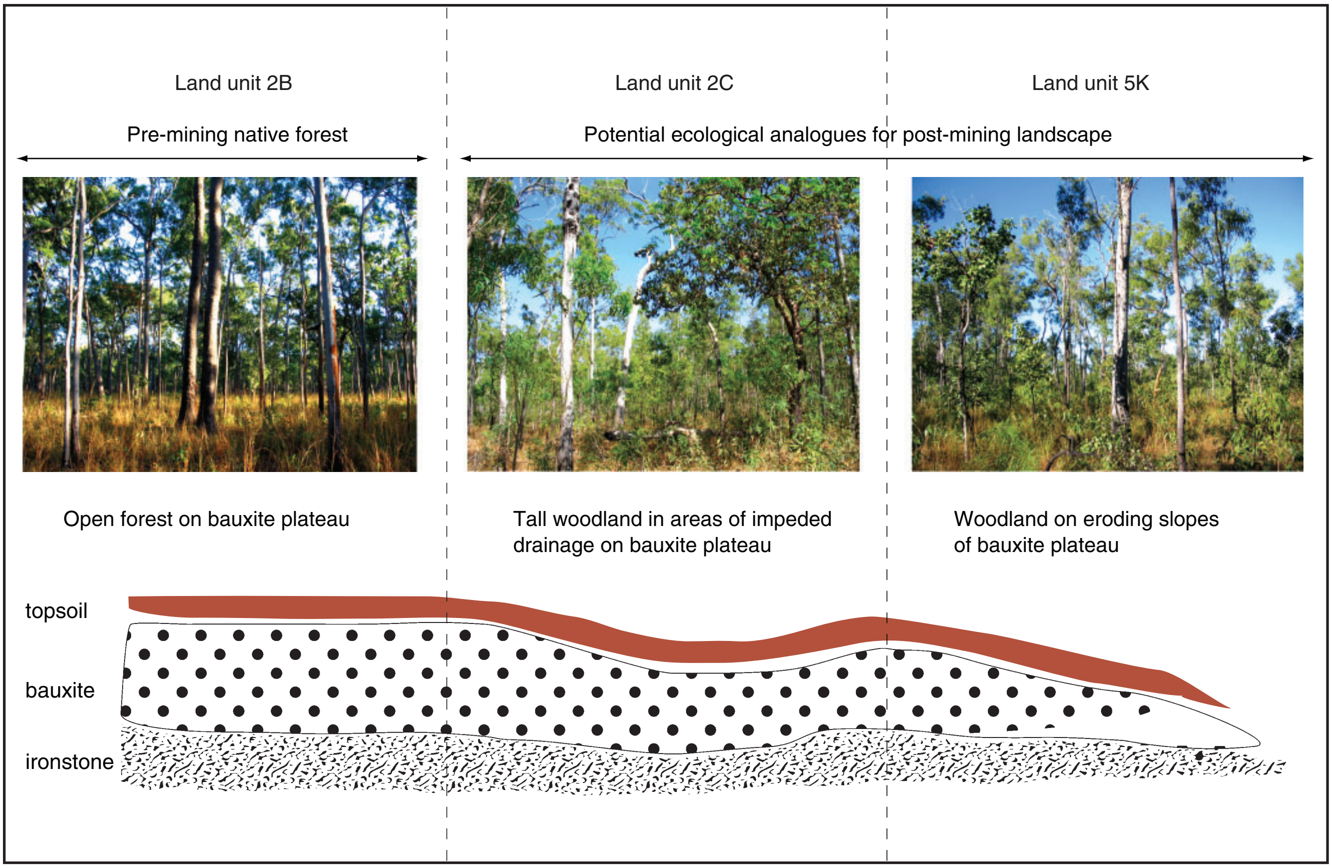

Fig. 1. Schematic of the Weipa bauxite plateau, showing the three reference native-forest land units.

corresponded to the period 1983–2006. This was the longest chronosequence available in which similar rehabilitation methods had been applied. Replicates of the following five age classes of rehabilitation were sampled across the chronosequence: 1–2 years old (six sites), 3–4 years old (six sites), 5–8 years old (six sites), 9–16 years old (eight sites) and 17–23 years old (five sites). All rehabilitation sites selected had been ripped, fertilised and sown with a mix of Australian native seeds. Enrichment planting with tubestock had been carried out in eight of the oldest rehabilitation sites.

Independence in site selection was achieved through stratification and partial randomisation using GIS. All study sites were 2 ha (100 m × 200 m) and aligned north to south along the long axis. Native forest sites were at least 1 km apart. Rehabilitation sites were at least 500 m apart, with the exception of two sites that were only 200 m apart but of different ages. Rehabilitation sites of similar age were selected from different parts of the mine wherever possible.

Bird surveys

A standardised bird-survey procedure was used to obtain estimates of species abundance and species detection rates that could be compared among sites. The survey procedure was a modification of the 2-ha 20-min area-search method (Barrett *et al.* 2003). Each site was divided into two 1-ha plots. Plot boundaries were identified in the field using a GPS (Garmin 60CX). During each survey, all birds seen or heard within a 1-ha plot were identified and recorded, and the number of individuals of each species was counted for 10 min. This procedure was repeated in the second hectare, avoiding recounting any individuals already counted in the first hectare. The requirement to avoid double counting individuals meant that multiple individuals of the same species were recorded only when they were detected more or less simultaneously within the site. Species count data are therefore conservative. Birds flying over or through the site were not counted. 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, were included. Potential error in bird data was reduced by adopting the following protocols: (1) each site was surveyed eight times over 16 months (September 2006 to December 2007), with a total survey time of 160 min at each site; (2) all surveys were conducted between 0630 hours and 1000 hours; (3) the order in which sites were visited changed with each round of surveys; (4) no surveys were conducted when it was raining or very windy (when large branches began swaying); (5) sites were searched actively rather than doing point counts; and (6) one observer (SG) conducted all bird surveys. Each round of surveys took 16 or 17 consecutive days to complete by surveying 3–5 sites a morning, weather permitting. Estimates of error owing to time since fire and weather conditions were obtained by collecting categorical data during each survey on wind, cloud cover and recent fire history. These variables were included in the analysis of bird-abundance data.

Vegetation sampling

Vegetation was sampled once at each site during May to June 2007. A point-centred quarter-sampling method was used (Bonham 1989) as data representative of the entire 2-ha study site were required. Vegetation data were collected at 16 systematically located sampling points within each site, each with a search radius of 16.5 m. Data were collected for up to five *a priori* defined vegetation layers within each quadrant, including the following: (1) perennial grass, defined as deep-rooted grass plant that resists detachment; (2) low shrub, defined as single-stemmed woody plant with stem diameter <1 cm or multi-stemmed shrub <2 m tall; (3) tall shrub, defined as single-stemmed woody plant with stem diameter >1 cm and <10 cm, or multi-stemmed shrub >2 m tall; (4) small tree, defined as woody plant, single-stemmed to a height of 50 cm and with at least one stem with diameter >10 cm and <35 cm; and (5) big tree, defined as woody stem with stem diameter >35 cm. Vertical stratification was necessary to ensure that sufficient data were obtained for all potentially ecologically significant vegetation layers. The following data were recorded for all vegetation layers: species identity; distance from the centre of sampling point; and canopy length, width and density. For perennial grasses, diameter at base and canopy height were also recorded. For all woody vegetation, heights to the base and the top of the canopy were recorded. Stem diameters were also recorded for woody plants with stem diameters >10 cm at a height of 1.5 m. Basal areas were calculated for multi-stemmed plants by calculating the basal area of individual stems and adding them together. Canopy density was visually estimated within an ellipse defined by the canopy length and width, using a standard visual reference for Australian vegetation (Mc Donald *et al*. 1998). If within a quadrant there was no plant of the target vegetation layer inside the search radius, a blank was recorded. A correction factor was applied when calculating stem densities to adjust for error introduced by vacant quadrants (Warde and Petranka 1981; Mitchell 2007). Distances, heights and canopy dimensions were measured using a LaserAce hypsometer (Measurement Devices Ltd., Aberdeen, Scotland), except where it was more efficient to use hand-held measuring tapes.

Landscape variables

Five landscape variables were calculated for each study site using GIS (ESRI ArcGIS 9). The variables were (1) area of remnant vegetation within a 500-m buffer of the site, (2) distance to remnant vegetation, (3) distance to the coast, (4) distance to the edge of the bauxite plateau and (5) distance to mesic vegetation. Drainage lines were used as a surrogate for mesic vegetation.

Data analysis

Bird data from all site visits were pooled to obtain site values. Species-detection rates were calculated for each site as the number of times a species was recorded per number of site visits. Site data were pooled to obtain summary bird statistics for the categories of age class of mine rehabilitation, native forest land unit and site category (mine rehabilitation cf. native forest). One of the post-mining landscape analogue LU 2C sites (A-19) was partially cleared after six bird surveys and all other fieldwork had been completed there. For the final two bird surveys, bird data were collected from a site that overlapped the original site by 100 m and the survey data were included with the first six surveys for that site. The analysis for A-19 was based on the original site-vegetation data and landscape variables.

Data transformations, calculations of summary statistics, statistical tests and analysis of variance were conducted using GENSTAT 10 (VSNi 2007). Repeated-measures analysis was used to analyse variance in bird-abundance data. A power model was fitted to account for the correlation structure between successive site visits. Multivariate analyses were conducted using PC-ORD 5 (McCune and Mefford 2006). Exploratory data analysis was also conducted with Primer-E v6 with Permanova+ (Anderson *et al*. 2008).

The degree of similarity in bird species composition to a reference condition was calculated for each site using the species shortfall index procedure described in Hannah *et al.* (2007). Hence,

(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; and
(2) site values for the degree of deviation from the reference condition were calculated as

$$\frac{100[a_x - \Sigma\min(a_{in}, a_{ix})]}{a_x}$$

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.

The shortfall index provides a measure of overall community dissimilarity, which includes species composition and mean species abundance, relative to a reference community. In this case, reference community values were obtained from Land unit 2B sites, i.e. pre-mining native forest. If a site contained the full complement of species recorded across all LU 2B sites, with each species present in at least equal abundance to its average abundance across all LU 2B sites, then there was no species shortfall and the site shortfall index value was 0%. If a site contained none of the species recorded in LU 2B sites, then the site shortfall index value was 100% (Hannah *et al*. 2007). Student's *t*-tests were used to test for differences in species shortfall index values between the age classes of mine rehabilitation and native-forest land units.

Exploratory data analysis of three groups of explanatory variables (5 × landscape, 12 × vegetation structure and 10 × vegetation composition) using the DISTLM procedure showed that vegetation structure accounted for most of the variation in the bird data. Canonical correspondence analysis was used to relate site species-detection rates with site vegetation-structure variables. Only 62 bird species that were recorded in five or more sites were included in the analysis. Row and column scores were standardised by centering and normalising. Scaling of ordination scores was by column (species). Scores for graphing sites were linear combinations of the vegetation-structure variables. Monte Carlo randomisation procedures using 998 runs were used to test for correlation between the bird data and the environment data. Time of day was used as a random number seed.

Results

In total, 97 bird species were recorded. Mean bird abundance and species richness increased consistently with rehabilitation age class (Table 1). In the two oldest age classes of rehabilitation, mean site species richness attained values similar to those in native forest, whereas bird abundance remained lower than in native-forest land units. Site category, season, time since fire, and weather conditions all contributed to variation in bird abundance (Table 2), with site category having the largest effect.

The composition of bird assemblages became more similar to native-forest bird assemblages with increasing rehabilitation age (Table 1); however, large and significant ($P < 0.001$) differences remained between the bird community in the oldest age class of mine rehabilitation and native-forest bird assemblages. The mean site species shortfall index in the oldest age class of mine rehabilitation was 63% compared with 27% at pre-mining native forest sites. In total, 19% (18/97) of species occurred only in mine rehabilitation, 25% (24/97) occurred only in native forest, and 56% (55/97) occurred in both native forest and mine rehabilitation. Excluding the 18 bird species that were recorded only in mine rehabilitation, there were 79 species of native forest bird of which 30% (24/79) were only recorded in native forest. Native-forest bird species that were absent from mine rehabilitation included brown treecreeper, grey goshawk, grey-crowned babbler, palm cockatoo, oriental cuckoo, southern boobook, dollarbird, yellow-tinted honeyeater and varied sittella.

Bird detection rates were significantly related to site vegetation structure. Ordination of species detection rates and vegetation structure completely separated native-forest sites from mine-rehabilitation sites along the first axis (Fig. 2). Total variance in the bird data was 1.836. Cumulative variance explained by the ordination 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 ($P < 0.001$) higher than expected by chance. The first ordination axis was strongly correlated with the vegetation-structure variables ($P < 0.001$). Four broad response groupings can be identified on the basis of the direction (+/–) and strength of their scores in relation to the ordination axes (Table 3). The habitat preference of Group I birds can be characterised as grassy, open forest. The habitat preference of Group II birds can be characterised as an open forest to tall woodland, with a well developed understorey. The habitat preference of Group III birds can be characterised as a dense shrubland to woodland, typical of the older age classes of mine rehabilitation. Finally, the habitat preference of Group IV birds can be characterised as grassland to low shrubland; vegetation that is typical of the younger age classes of mine rehabilitation.

Discussion

Faunal succession is expected to occur in response to changes in habitat that occur during vegetation succession. Relationships between the structural complexity of vegetation and the richness and abundance of bird assemblages, are well established generally (Karr 1968; Willson 1974; Kavanagh *et al*. 1985; Laiolo *et al*. 2004; Venier and Pearce 2005). The observed rates of change in these indices at Weipa are interpreted, therefore, as responses to physical changes in habitat structure as rehabilitation ages.

Twenty-three years after the rehabilitation commenced, the composition of bird assemblages in post-mining rehabilitation at Weipa remained significantly different from that of native-forest bird assemblages. Similar results have been documented at several post-mining rehabilitation sites (Parrotta *et al*. 1997; Brady 2005). In particular, foraging specialists and species that require specific vegetation-based habitat resources, such as tree hollows, are often absent from rehabilitation (Loyn 1985; Armstrong and Nichols 2000; Martin *et al*. 2004). Armstrong and Nichols (2000) concluded that vegetation development controls the composition of the bird communities at rehabilitation sites. Changing physical structure, however, does not account for the role of plant-species composition in providing specific vegetation-based habitat resources.

An obvious interpretation of the differences in the composition of bird assemblages is simply that more time is needed for bird assemblages in rehabilitation to converge on reference bird assemblages. Because vegetation composition and structure determine the availability of habitat resources that are central to understanding bird distributions, such an

Table 1. Bird statistics by age class and land unit

Species richness is the mean value calculated from site totals over eight site visits. Abundance is the mean value calculated from the number of individual birds detected per site visit. Shortfall index is the mean site shortfall index. See text for definition of the land units. Percentages are coefficients of variation

Site category	Land unit/age class	Species richness	Abundance	Shortfall index
Rehabilitation	1–2 years ($n = 6$)	9 (27%)	5 (99%)	98 (1.3%)
	3–4 years ($n = 6$)	15 (22%)	11 (65%)	90 (10.4%)
	5–8 years ($n = 6$)	20 (29%)	14 (61%)	85 (12.7%)
	9–16 years ($n = 8$)	25 (18%)	18 (44%)	79 (10.7%)
	17–23 years ($n = 5$)	28 (11%)	20 (37%)	63 (19%)
Native forest	LU 2B ($n = 28$)	27 (15%)	31 (30%)	27 (20.4%)
	LU 2C ($n = 4$)	29 (8%)	34 (33%)	25 (14%)
	LU 5K ($n = 4$)	29 (16%)	28 (32%)	34 (20%)

Table 2. ANOVA results for bird-abundance data

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

Site category	Season	Fire	Cloud × wind
$F_{(5, 289)}$ 206.8***	$F_{(3, 335)}$ 33.35***	$F_{(4, 478)}$ 3.04*	$F_{(11, 498)}$ 2.31**

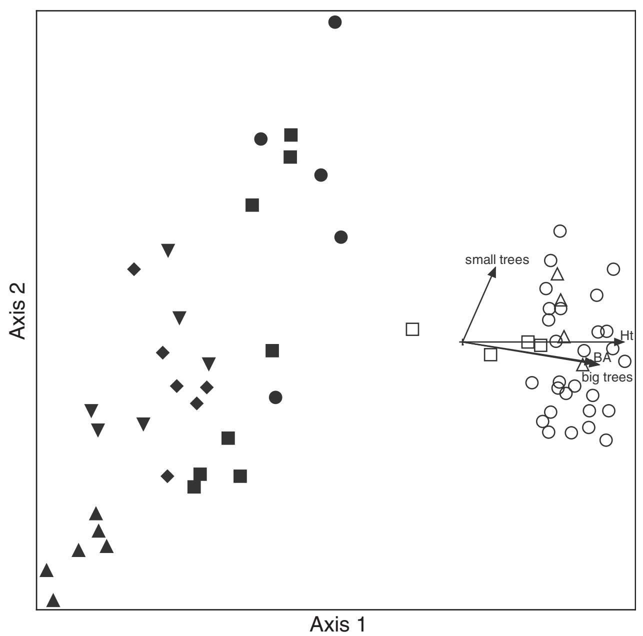

Fig. 2. Ordination graph of birds and vegetation structure (sites only). Bird detection rates birds (62 species in main matrix) were strongly related to vegetation structure (12 variables in the second matrix). The first ordination axis can be interpreted as height gradient. The second ordination axis can be interpreted as a density gradient. The variables most strongly related to Axis 2 were stem density of small trees, canopy volume of small trees and basal area of small trees. Native-forest sites are represented by open symbols: ○ = LU 2B; △ = LU 2C; □ = LU 5K. Mine rehabilitation age classes are represented by solid symbols: ▲ = 1 to 2 years, ▼ = 3 to 4 years, ◆ = 5 to 8 years, ■ = 9 to 16 years and ● = 17 to 23 years. Joint plot labels are Ht = mean height; big trees = number of big tree stems, BA = basal area of big trees; and small trees = number of small tree stems. To minimise clutter in the diagram, only joint plots with correlation coefficients >0.8 are shown. These are 'intraset correlations' (Ter Braak 1986) that relate to the rate of change in the community composition; in this case, the site detection rates of birds, per unit change in the corresponding environmental variable.

interpretation of the bird data might be supported by evidence of vegetation succession in rehabilitation. Apart from the structural changes associated with plant growth, however, there is no evidence that the vegetation community in post-mining rehabilitation at Weipa is converging on native-forest ecosystems (Gould 2010).

There are large and significant differences between the taxonomic composition of the plant community in rehabilitation and that in native forest at Weipa (Gould 2010). Analysis of taxonomic composition on the basis of canopy volume of plant species found a mean 90% dissimilarity between all pairs of sites (each pair consisted of one native-forest site and one rehabilitation site). There were initial rapid increases in vegetation height; however, the rate of increase slowed after 8 years and the observed mean vegetation height in the oldest age class of rehabilitation was half that of reference values. There is increasing body of empirical evidence that vegetation dynamics in novel ecosystems, such as in post-mining rehabilitation, are not unidirectional; and that a single climax state cannot be assumed for late successional vegetation and associated fauna (Foster *et al.* 2003; Hobbs *et al.* 2006; Chazdon 2008; Cramer *et al.* 2008). Successional pathways in novel ecosystems have been shown to be highly contingent (Pickett and Parker 1994; Paine 2002; Walker and Del Moral 2003), with multiple possible stable states (Pickett and Cadenasso 2005). Site history and rehabilitation treatments have lasting effects on vegetation composition and structure in post-mining rehabilitation (Fox *et al.* 1996; Grant and Loneragan 2001; Norman *et al.* 2006) and differences in vegetation composition and structure between reference ecosystems and rehabilitation may even increase over time (Buckney and Morrison 1992; Chambers *et al.* 1994).

A plausible interpretation of the response trajectory for the bird species shortfall index is that it may not continue in the direction of reference bird assemblages. The initial decline in bird species shortfall index values is interpreted as a response to the initial rapid changes in habitat structure that occurred with early vegetation growth in post-mining rehabilitation. There is no available evidence to support the interpretation that bird assemblages in post-mining rehabilitation at Weipa will inevitably converge on reference bird assemblages.

Lindenmayer *et al.* (2010) emphasised the role of landscape-context variables and size of rehabilitation areas in faunal use of rehabilitation in agricultural landscapes. At Weipa, within-site vegetation was the most important variable determining bird detection rates. The difference between studies may be due to the largely intact regional vegetation cover surrounding the Weipa mine compared with agricultural landscapes. Nevertheless, distance to source populations was irrelevant for a third of the Darwin stringybark-forest birds if the site did not provide suitable habitat.

Implications for rehabilitation

Rehabilitation at Weipa has been successful to the extent that vegetation has been established and it is providing habitat for fauna. It has, therefore, partially made up for the local biodiversity losses caused by the initial vegetation clearing. Nevertheless, 23 years after rehabilitation commenced there is a clear residual impact of mining on the bird assemblages. Improving biodiversity outcomes will depend on the development of specific strategies for the species that are in need of habitat restoration. For example, the bird species that were entirely absent from rehabilitation included all of the local trunk-gleaning insectivores. Incorporating understanding of the requirements

Table 3. Bird response to vegetation structure

The vegetation-structure variables most strongly related to Axis 1 were mean height of the tallest vegetation stratum, stem density and basal area of big trees. The vegetation-structure variable most strongly related to Axis 2 was stem density of small trees. Only species with at least one score >0.5 are listed

Group	Species	Axis 1	Axis 2
I	Brown treecreeper (*Climacteris picumnus*)	0.6956	–0.4092
	Yellow-tinted honeyeater (*Lichenostomus flavescens*)	0.6607	–0.5749
	Grey shrike-thrush (*Colluricincla harmonica*)	0.6378	–0.2560
	Rufous whistler (*Pachycephala rufiventris*)	0.5766	–0.1336
	Cicadabird (*Coracina tenuirostris*)	0.5481	–0.0185
	Black-backed butcherbird (*Cracticus mentalis*)	0.5989	–0.0665
	Lemon-bellied flycatcher (*Microeca flavigaster*)	0.5233	–0.0235
	Laughing kookaburra (*Dacelo novaeguinae*)	0.5462	–0.0705
II	Varied sittella (*Daphoenositta chrysoptera*)	0.689	0.0260
	Grey-crowned babbler (*Pomatostomus temporalis*)	0.6290	0.0660
	Forest kingfisher (*Todiramphus macleayii*)	0.5712	0.0055
	Brush cuckoo (*Cacomantis variolosus*)	0.4720	0.0577
	Yellow oriole (*Oriolus flavocinctus*)	0.5731	0.8631
	Pied imperial-pigeon (*Ducula bicolour*)	0.6246	0.4326
	Koel (*Eudynamys orientalis*)	0.2566	0.4372
III	Olive-backed sunbird (*Nectarinia jugularis*)	–1.4840	0.0215
	White-streaked honeyeater (*Trichodere cockerelli*)	–1.4629	0.1360
	Black butcherbird (*Cracticus quoyi*)	–1.1230	1.9342
	Yellow-spotted honeyeater (*Meliphaga notata*)	~–1.0386	1.2343
	Varied triller (*Lalage leucomela*)	–1.0267	1.3691
	Graceful honeyeater(*Meliphaga gracilis*)	–0.9026	0.7902
	Dusky honeyeater (*Myzomela obscura*)	–0.7831	1.2996
	Bar-shouldered dove (*Geopelia humeralis*)	–0.7606	0.0442
	Brahminy kite (*Haliastur indus*)	–0.5584	0.7541
	Great bowerbird (*Ptilonorhynchus nuchalis*)	–0.3045	0.9399
	Brown-backed honeyeater (*Ramsayornis modestus*)	–0.3022	0.8373
	Spangled drongo (*Dicrurus bracteatus*)	–0.2703	0.7307
IV	Australasian pipit (*Anthus novaeseelandiae*)	–2.2057	–1.9830
	Golden-headed cisticola (*Cisticola exilis*)	–1.9613	–1.3822
	Chestnut-breasted mannikin (*Lonchura castaneothorax*)	–1.7980	–0.8324
	Ed-browed finch (*Neochmia temporalis*)	–1.6675	–0.6695
	Brown quail (*Coturnix ypsilophora*)	–1.6584	–1.6525
	Brown honeyeater (*Lichmera indistincta*)	–1.6292	–0.1538
	Australian brush turkey (*Alectura lathami*)	–1.6156	–0.0264
	Peaceful dove (*Geopelia striata*)	–0.6446	–0.1563

of trunk-gleaning insectivores with respect to stem densities of specific tree species into rehabilitation goals will improve the long-term outcomes for this foraging group. Development of specific strategies based on the habitat requirements of local biodiversity will require (1) setting clearly defined biodiversity goals, (2) understanding the multiple factors that determine the effectiveness of rehabilitation as habitat, especially as they relate to target species, (3) commitment to providing sufficient resources to support ongoing monitoring and adaptive management of rehabilitation and (4) developing strategies to address the long time delays in provision of key habitat resources.

Implications for conservation policy

The proposition that residual adverse impacts on biodiversity can be offset implies that biodiversity is exchangeable, one species for another. Species identity is fundamental to the concept of biodiversity. Some species have irreplaceable roles in ecosystems and population declines may disrupt ecological processes (Sekercioglu *et al*. 2004). There is a need to clearly articulate the concepts and measures used in biodiversity offsets, particularly with respect to the issues of currency and no net loss (Ten Kate *et al*. 2004;). In the absence of full understanding about species roles in ecosystem processes, careful quantification of species losses and gains is required when assessing the extent to which rehabilitation makes up for the impacts of development. It is important that policy makers and regulators understand and explicitly acknowledge the limitations of rehabilitation as a tool for minimising and reversing biodiversity losses. As Gibbons and Lindenmayer (2007) point out, there is only a limited set of circumstances under which no net loss of biodiversity can be assumed.

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