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Is restoring flora the same as restoring fauna? Lessons learned from koalas and mining rehabilitation

Romane H. Cristescu[1,2]*, Jonathan Rhodes[3], Céline Frére[4]† and Peter B. Banks[1,5]†

[1]*School of Biological, Earth and Environmental Sciences, University of New South Wales, Kensington, NSW, 2052, Australia;* [2]*Sustainable Minerals Institute, Centre for Mined Land Rehabilitation, The University of Queensland, Brisbane, QLD, 4072, Australia;* [3]*School of Geography, Planning and Environmental Management, The University of Queensland, Brisbane, QLD, 4072, Australia;* [4]*Centre for Ecology and Conservation, The University of Exeter, Cornwall campus, Penryn, TR,10 9EZ, UK; and* [5]*School of Biological Sciences, The University of Sydney, Sydney, NSW, 2006, Australia*

Summary

1. Rehabilitation of degraded and disturbed landscapes has become critical for counteracting habitat loss. The success of rehabilitation projects, to date, has focused on abiotic and flora-based criteria of success, leaving fauna unmonitored. This follows from the common paradigm that if flora recovers, fauna will recover too. However, we know very little about the extent to which this assumption is true. We addressed this issue by examining whether flora criteria used to assess mine rehabilitation reflected patterns in the recovery of an iconic species, the koala *Phascolarctos cinereus*, in eastern Australia.

2. We used rank tests to search for correlations between current mining flora criteria and fauna presence. We then developed *a priori* regression models to search for new abiotic and flora criteria that are biologically relevant to *Phascolarctos cinereus*. In a third step, we investigated correlations between rehabilitation success ranked on the best biologically relevant habitat variables and *Phascolarctos cinereus* recolonization.

3. We found that rehabilitation success based on current mining flora criteria (calculated at two different scales: rehabilitation blocks and monitoring plots) did not correlate with *Phascolarctos cinereus* presence.

4. In contrast to the current flora-based criteria, we found that variables that are biologically relevant to *Phascolarctos cinereus* had more influence on its presence. For instance, species richness in food trees favoured by *Phascolarctos cinereus* and tree canopy cover had a positive effect on its recolonization. However, correlations between biologically relevant habitat variables and fauna occurrence were still inconsistent.

5. *Synthesis and applications*. In our study, flora criteria for rehabilitation success did not correlate with fauna recolonization. We also found several additional difficulties in predicting fauna recolonization based on habitat variables, such as the choice of relevant scales and the geographic specificity of relevant variables. The choice between monitoring habitat proxies or fauna will ultimately be based on weighting costs and efficiency and will depend on the fauna species. However, we argue that in general, fauna species should be directly monitored to ensure the recolonization of i) species of interest (e.g. threatened and charismatic) and ii) fauna involved in long-term resilience of ecosystems.

Key-words: completion criteria, fauna monitoring, koala, mining, recolonization, rehabilitation

Introduction

Most ecosystems in the world today are affected to some degree by anthropogenic pressures (Vitousek *et al.* 1997). Restoration of disturbed landscapes (e.g. agricultural and

*Correspondence author. E-mail: romromfr@yahoo.fr
†These authors equally contributed to this work.

mining landscapes) has therefore become a necessary component of conservation (MacMahon & Holl 2001). In ecological restorations, success criteria focus primarily on abiotic and flora characteristics (Tongway & Hindley 2003; Ruiz-Jaen & Aide 2005). In particular, the quality of flora, in terms of structure, complexity and species diversity, is deemed a condition *sine qua non* for fauna recolonization. Monitoring flora characteristics is therefore sometimes assumed to be more relevant than monitoring fauna species (Lindenmayer, Margules & Botkin 2000). Some common flora characteristics (tree density, height and richness) are also comparatively easy to measure and show less seasonal variation than fauna (Ruiz-Jaen & Aide 2005). Once abiotic and flora criteria are deemed satisfactory, then it is usually assumed that fauna recovery will follow the same trends.

Researchers have questioned this paradigm, and testing it is now a priority for restoration ecology (Clewell & Rieger 1997; Palmer, Ambrose & Poff 1997; Block *et al.* 2001; Koch 2007). Indeed, ensuring fauna recolonization is successful is crucial (see Craig *et al.* 2012), not only for the conservation of biodiversity in general, but for the long-term resilience of ecosystems (Fischer, Lindenmayer & Manning 2006), as fauna plays many crucial roles in ecosystems (Majer 1989; Nichols & Nichols 2003). A few studies to date have challenged the paradigm [e.g. invertebrates (Crisp, Dickinson & Gibbs 1998; Longcore 2003), amphibians (Mazerolle *et al.* 2006) and birds (Buffington *et al.* 2000)], but despite this, rehabilitation success of fauna, particularly mammals, is still ignored by both mining companies and regulators world-wide. This may be because no studies to date have investigated this paradigm in a context relevant to mining rehabilitation policies and regulations.

The flora-equals-fauna paradigm is directly relevant to mining rehabilitation, particularly as the number of mine closures around the world is increasing (World Bank & the International Finance Corporation 2002). Furthermore, the mining industry is growing at an unprecedented rate [for example, 2012 is a record year in the investments made to increase mining capacity in Australia (Australian Government 2012)]. Habitat clearance as a result of mining is a direct threat to many species [e.g. 225 amphibians, 216 reptiles, 322 birds and 266 mammals according to the IUCN Red List (IUCN 2012)], but despite this, the majority of mining countries still lack frameworks in regard to rehabilitation for mine closure (Clark & Cook-Clark 2005). Even in closely regulated countries, such as those in North America and Australia (World Bank & the International Finance Corporation 2002), there are no requirements to include fauna in rehabilitation monitoring (see United States and Canada coal mines, Smyth & Dearden 1998). For example, in the Mine Rehabilitation Code of Ontario, included as Schedule I of Regulation 240/00 that regulates mine closure, monitoring specifications are explicitly given for the stability of the site, surface and ground water, pollution and flora but not for fauna (Ontario Government 2011). As for Australia, governmental guidelines on mining rehabilitation state that in most cases, it is too difficult to directly measure fauna, and thus habitat variables, in particular flora, should be used as proxies (EPA 2006). This again underlines that flora criteria are assumed to serve the double goal of reflecting their own state as well as being a proxy for fauna recolonization success.

This study is the first to test the flora-equals-fauna paradigm in a context directly relevant to mining rehabilitation policies and regulations. To do so, we investigated whether rehabilitation success criteria, based on the current fulfilment of flora criteria implemented by the Sibelco mining company and endorsed by the Australian Government, correlated with recolonization patterns of the koala *Phascolarctos cinereus* Goldfuss, an iconic Australian species. Sibelco mining company is only the second mining company in Australia to have reached an agreement with all stakeholders, including the Government, regarding rehabilitation success criteria. Many more mining companies world-wide will soon follow this example; therefore, it is important that relevant criteria are being used. In this study, we first questioned whether flora criteria currently used by the mining company would accurately represent *P. cinereus* recolonization success, then whether other habitat variables chosen specifically for *P. cinereus* could give better results. For this second step, we searched for new abiotic and flora criteria relevant to *P. cinereus* and tested whether any of these variables correlated with *P. cinereus* recolonization patterns. *Phascolarctos cinereus* was chosen as a model fauna species for two reasons. First, the factors influencing *P. cinereus* distribution have been intensively studied, providing us with good working hypotheses for potential habitat variables influencing *P. cinereus* recolonization of rehabilitated landscapes. Second, *P. cinereus* is recognized as a charismatic species whose conservation is considered crucial by all stakeholders. Consequently, successfully rehabilitated flora is not sufficient to declare this mine ready for closure because recolonization by *P. cinereus* of postmining landscapes has to be ensured.

Materials and methods

STUDY SITE

North Stradbroke Island (NSI, 27º23′/27º45′S, 153º23′/153º33′E) is the largest of a group of sand islands in Moreton Bay, in the south-east region of Queensland, Australia. The island is roughly the shape of a triangle of 38 km by 11 km at its widest point (approximately 27 500 ha). NSI has a wet-dry subtropical climate (Specht 2009) and is formed predominantly of unconsolidated Cainozoic sediments (Laycock 1978). Open mining for mineral sand occurs on the island, with approximately 16% of its area mined so far. Mined areas are progressively rehabilitated according to premining landscape aspect, slope and elevation. Current methods of rehabilitation include spreading topsoil and seeding with seed either collected ahead of the mine path or within a

30 km radius of the rehabilitated site. Seed mixes vary considerably in accordance with premine vegetation surveys and generally have between 70 and 90 species. A hybrid sterile sorghum crop is sown for a fast-growing windbreak to protect the young native seedlings from wind exposure. Terolas (an anionic slow-set bitumen emulsion) is sprayed to stabilize the soil surface, prevent sand and soil movement and erosion (Bell, Carter & Hetherington 1986). One to two years after direct seeding, nursery stock is planted at an average rate of 1650–2000 seedlings per hectare.

We collected koala signs and conducted ground checking for koala faecal pellets (scats) in the undisturbed surroundings of rehabilitated areas before selecting our study sites. This ensured remnant populations of koalas existed and could potentially recolonize (e.g. one mine was not included in this study because surrounding populations could not be confirmed, see Fig. 1). The shape of the rehabilitated areas (ribbon-like, often less than 400 m wide) and koala mobility mean that all rehabilitated areas in this study were accessible to koalas.

STUDY PLOTS

As our goal was to compare flora mine criteria to *P. cinereus* presence, we studied plots used by the mine for assessing flora success criteria (Sibelco, unpublished data). These mine monitoring plots, each measuring 50 × 10 m, were compatible in size with plots used in previous *P. cinereus* scat surveys (Lunney *et al.* 2000). Plots were established along transects with a random start and then were evenly spaced across rehabilitated areas. We included all mining monitoring plots from 7 to 31 years postrehabilitation (N = 54). Seven years postmining was chosen as the youngest cut-off value for plots to be included in our analyses as koalas have been observed using them (Cristescu 2011). As practice for rehabilitation has evolved over time, we assigned plots to three groups on the basis of key methodological changes, ranging from times when rehabilitation methodology was mainly designed to stabilize landforms (N = 12), to when rehabilitation used refined seed mixes to allow a more diverse flora to develop (N = 15) to the current methods described above (N = 27, see 'Study Site'). We recorded plot coordinates by GPS (Garmin, eTrex®H, Olathe, KS, USA, accuracy ±7 m, AMG 84) and demarcated them with measuring tapes.

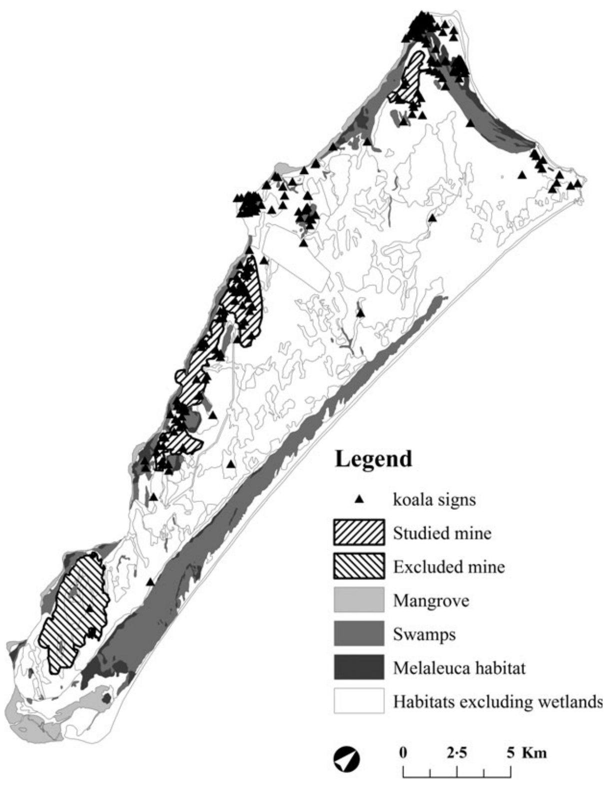

Fig. 1. Distribution of koala signs on North Stradbroke Island showing that the selection of mines included in this study was based on the presence of koala signs in the surroundings.

For each plot, we calculated environmental variables following the methodology of the mining monitoring or using the mining data base (Sibelco, unpublished data). All trees in the plot were counted. The percentage of canopy and ground cover (plants, bare or litter) was visually estimated every 2 m along the two transects forming the longer borders of the plots. These are part of the data routinely collected when assessing flora rehabilitation success (see Table S1 in Supporting Information). We then used *P. cinereus* literature (see specific papers in Table S2) to select *P. cinereus*-relevant variables: these were habitat characteristics relevant to *P. cinereus* and readily measurable (i.e. that could easily be measured by the mine).

We searched each plot and recorded two *P. cinereus* presence variables: the number of *P. cinereus* scats and the number of scat locations (i.e. the number of trees under which scats were found). We searched the ground of the entire plot (50 × 10 m) for koala scats, as scat deposition is not limited to the base of the trees (Ellis *et al.* 1998). This took up to 4 h and was conducted by a single researcher (RC) to standardize observer bias (Neff 1968). Four hours is a longer search time than many studies, but a long search time was necessary for a high scat recovery (Cristescu *et al.* 2012).

Phascolarctos cinereus-relevant variables at a fine scale were as follows, for *P. cinereus* main food trees (*Eucalyptus* and *Corymbia*, Martin & Handasyde 1999): species richness, density, mean circumference at breast height and mean tree height. Landscape variables were slope, aspect and elevation, as well as distance from the plot to the nearest wetland (which represent primary *P. cinereus* habitat and could constitute a source for recolonization, Table S2, Fig. 1) and distance to undisturbed habitat (a measure of ease of recolonization, Tables S2 and 1). We extracted landscape variables from a 2008 airborne laser scan of the island (Sibelco, unpublished data) in Terramodel 10·61.

DATA ANALYSIS

Comparing mining flora criteria to Phascolarctos cinereus *occurrence*

Criteria used by the mine to assess rehabilitation success for flora include species presence, density, ground cover and presence of weeds (Table S1), with each criterion having a specific threshold. For example, tree density in rehabilitated areas must not be significantly less than 75% of tree density in mining reference sites (these reference sites come either from the premining surveys or from the mine surroundings). Species density, richness and other criteria are measured by monitoring plots described above.

Table 1. Description of the explanatory variables contained in the different models

Explanatory variables	Description
Method	Factor with three levels based on rehabilitation method (old method designed to stabilize landforms; improved method using refined seed mixes; current method as defined in text)
Plants	Percentage of ground cover of plants in the plot (visually assessed)
Bare	Square root of percentage of bare ground in the plot
Elevation	Elevation in metres above sea level of the plot
Density	Log of mean density of *Eucalyptus* and *Corymbia* species in the plot (trees/ha)
Richness	Number of *Eucalyptus* and *Corymbia* species in the plot
Circumference (CBH)	Square root of mean circumference in cm of *Eucalyptus* and *Corymbia* species in the plot
Canopy	Percentage of canopy cover in the plot
Slope	Slope of the plot in percentage
Distance to undisturbed	Euclidean distance in metres from the plot to the edge of closest undisturbed areas
Distance to wetlands	Square root Euclidean distance in metres from the plot to the edge of swamps, *Melaleuca* forest or Mangroves

Plots are combined to calculate average and confidence interval per block of rehabilitation. Blocks are defined as a section of rehabilitated area of the same age and location (N = 10, mean size = 35·3 ha [9·8–67·0 ha]). The mine determines rehabilitation success, that is, the number of criteria met, based on these averages per block. When all criteria are met for a block of rehabilitation, the mine can proceed towards relinquishing this block as part of the mine closure. The number of criteria that the mine assesses per rehabilitated block varies from 23 to 63, depending largely on different blocks having different numbers of species in the reference sites and the age of the block (older blocks have less criteria). For example, some blocks will need to re-establish ten native tree species compared to the vegetation type that was disturbed, some only five native species.

We classified rehabilitation success per rehabilitation block in two ways: i) on the basis of flora criteria as defined by the mining company (Table S1) from the highest percentage of success criteria met to the lowest and ii) on the basis of the number of *P. cinereus* scats or scat locations from highest number of evidence to the lowest. Our data were non-normal, overdispersed and zero-inflated, so we compared the correlation between the flora and fauna rankings with Kendall's τ and Spearman's ρ tests in PAWS Statistics 18·0 (IBM 2009). Both correlation methods were concordant so we only present the Kendall's results. In the eventuality of blocks being heterogeneous in term of flora and fauna, averaging (per block) the results of individual monitoring plots could disrupt a correlation between flora and fauna. Thus, we also calculated the correlation between the rankings of flora and fauna at the plot level.

Searching for most relevant habitat criteria for Phascolarctos cinereus

We constructed models to compare and rank *P. cinereus*-relevant habitat variables based on the number of scat locations in each plot. We avoided stepwise selection techniques (Mac Nally 2000) and data dredging (Burnham & Anderson 2002) using *a priori* models to draw inferences (Johnson & Omland 2004). This method decreases the possible selection of noise variables (Flack & Chang 1987).

Our response variable was the number of scat locations instead of the number of scats to avoid unnecessary over-dispersion due to the lack of independence between *P. cinereus* scats (Zuur *et al.* 2009). We fitted our response variable using a generalized linear model with a Poisson distribution, the standard distribution for counts (McCullagh & Nelder 1989). To control for a greater frequency of zeros than expected under a standard Poisson distribution (Zuur *et al.* 2009), we used a zero-inflated mixture modelling approach (Lambert 1992). This produces zeros from two different processes: a binomial process (corresponding to false zeros) and a count process (true zeros and positive values, Zuur *et al.* 2009). Our models consequently consisted of two parts, a binomial part (called zero models, because they correspond to false zeros) and a count part:

Scat locations ~ binomial part (variables influencing the occurrence of false zeros, i.e. variables linked with scat detectability and decay rate) + count part (variables influencing *P. cinereus* occurrence, i.e. habitat variables).

Finding the best model was thus a two-step process: (i) searching for the best binomial part by comparing all different binomial parts, while the count part is fixed to the full model (including all variables) and (ii) the best binomial part is retained, while count parts including different variables are tested. Binomial and count parts were defined by a set of *a priori* models (all models are given in Table S3).

The binomial part of the zero models included variables influencing the probability of not finding scats when in fact koalas were using a plot (i.e. false zeros). False zeros can result from variation in scat detectability in relation to the complexity of the ground layer (Cristescu *et al.* 2012). Thus, we included the percentage of plant ground cover and bare ground in the zero models (we excluded litter because it correlated with the other two). The method used for rehabilitation, which influences the litter characteristics, was another variable in the binomial part. We included plot elevation as a surrogate for temperature and humidity, which influence the rate of scat decay (Rhodes *et al.* 2011; Cristescu 2012), and hence could also lead to false zeros (see Appendix S4 for other possible biases). We compared all models with one single variable and combinations of two variables, giving a total of 10 zero models. We used the best model for the binomial part for each of the count models described below.

To populate the count part of the models, we used different combinations of koala-relevant variables (Fig. 2, Table S2). Models 1–10 incorporated every combination of two and three fine-scale variables related to food and shelter trees for koalas (density, richness, circumference and canopy). Models 11–14 were composed of every combination of two to three landscape variables (elevation, slope and distance to wetlands). Model 15 emphasized a fragmentation approach, where distance to undisturbed populations and population sources (i.e. wetlands) would be influential. Models 16–33 investigated multilevel models. Food

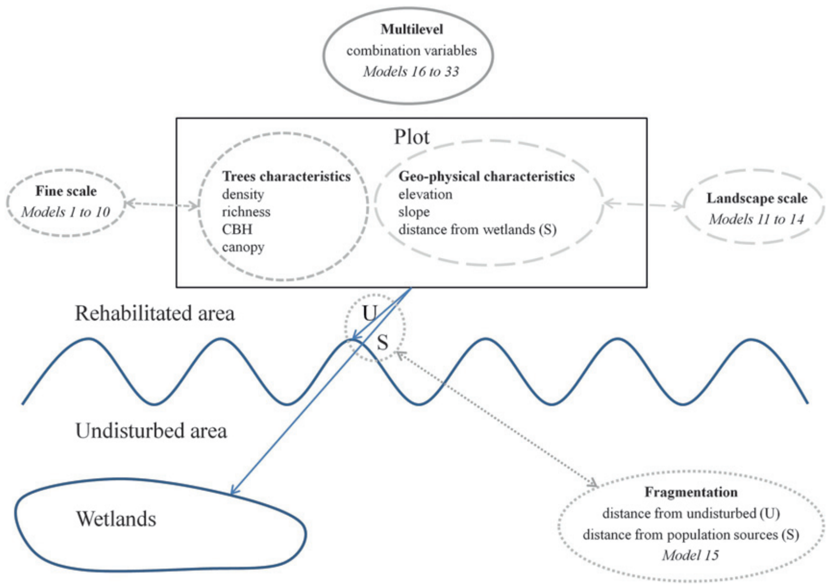

Fig. 2. Schematization of model construction based on *Phascolarctos cinereus*-relevant variables.

and shelter were emphasized by including in each model a combination of two fine-scale variables and then adding in turn variables from landscape and fragmentation models that have been most recurrently found to influence koala distribution in the literature (elevation, distance to source or undisturbed area). Model 34 consisted only of these three variables. Although we recognize the influence of foliar content in koala distribution (Moore & Foley 2005), preliminary unpublished results found no correlation and are not presented here.

Models were constructed, compared and validated in R 2·12·0 (R Development Core Team 2010). We graphically analysed explanatory variables, square- or log-transformed skewed variables and then standardized them to allow comparison of model parameter estimates (Quinn & Keough 2006). Prior to the inclusion of any variables in the models, we tested collinearity using variance inflation factors (VIF). Any VIF superior to three was examined and eliminated if it was theoretically sound, that is, measuring conceptually similar things (O'Brien 2007). On the basis of their VIF, tree height, age of rehabilitation and method were excluded from the count part. We included a maximum of three different variables in each count part (and two variables in binomial parts) to minimize the risk of spurious effects due to our small sample size (Burnham & Anderson 2002; Babyak 2004).

Zero-inflated Poisson models were created for all models in the R-package 'pscl' (Jackman *et al.* 2010). Spatial correlations in the data and the residuals were analysed with R-package 'ncf' (Bjornstad 2009). We estimated the global goodness-of-fit between the global model and the null model with a likelihood ratio test (Mundry 2011), which is preferred to a Wald test in case of small sample size (Pawitan 2001). To rank the models, we used Akaike Information Criterion corrected for small sample size, AICc (Burnham & Anderson 2002).

Multimodel inference methods were used to determine the relative importance of explanatory variables based on our set of models (Anderson *et al.* 2001). We calculated, based on AICc, Akaike differences (Δ) between each model and the most parsimonious model; Akaike weights, a measure of the probability that any given model is the best model; and the evidence ratios, which indicate how much more likely the best model is compared to each of the other models. To account for model uncertainty, we calculated the model average parameter estimates and the unconditional standard error of each estimate (Burnham & Anderson 2002).

Comparing P. cinereus-*relevant variables to* Phascolarctos cinereus *occurrence*

We selected the *P. cinereus*-relevant habitat variables with the highest relative importance (by summing the Akaike weights of the models where the variable appeared, Burnham & Anderson 2002). We then reiterated our Step 1 by calculating the correlation between habitat and *P. cinereus* rankings of rehabilitation with Kendall's τ tests.

Results

COMPARING MINING FLORA CRITERIA TO *PHASCOLARCTOS CINEREUS* OCCURRENCE

Details of plots are given in Table 2. *Phascolarctos cinereus* were located in five blocks and 24 plots. Ranking of rehabilitation blocks or monitoring plots based on flora criteria used by the mine did not correlate with ranking based on *P. cinereus* scats or scat locations (Table 3a).

SEARCHING FOR RELEVANT HABITAT CRITERIA FOR *PHASCOLARCTOS CINEREUS*

The goodness-of-fit of the global model was significantly better than a model with just the intercept (likelihood ratio test, $\chi^2 = 55{\cdot}35$, $P < 0{\cdot}001$); thus, our models fitted the data significantly better than the null model. Residuals plotted against each explanatory variable (included or not included in the models, Table 2) showed no patterns, indicating that our selection of variables was valid and there was no violation of independence. Spline correlograms provided no evidence of spatial autocorrelation (Bjornstad 2009).

Table 2. Characteristics of the plots searched for *Phascolarctos cinereus* scats

	Mean	SEM	Minimum	Maximum
Year of rehabilitation	1993	1	1978	2002
Number of scat locations in plot	1·3	0·3	0	8
Number of scats in plot	32·2	11·9	0	444
Time of search (minutes)	132·8	6·4	60	240
Density *Eucalyptus and Corymbia spp.* per ha	1501·1	201·0	200	9540
Richness in *Eucalyptus and Corymbia spp.*	4·4	0·2	1	8
Mean CBH *Eucalyptus and Corymbia spp.* (cm)	23·6	2·1	6·2	76·1
Mean height *Eucalyptus and Corymbia spp.* (m)	5·4	0·4	2·4	15·0
Canopy cover (%)	48·3	3·3	10	92
Ground cover: plants (%)	21·8	4·0	0	100
Ground cover: bare (%)	17·5	2·4	0	72
Elevation (m)	71·8	3·6	21·3	145·6
Aspect (degree)	187·8	13·6	4·0	329·5
Slope (%)	20·4	1·7	2·5	47·6
Distance to wetlands (m)	515·3	58·9	70	1920
Distance to undisturbed (m)	163·4	14·7	20	435

SEM, standard error of the mean.

Table 3. Correlation rankings between koala occurrence and a) mine flora criteria b) most relevant koala habitat variables, calculated by block of rehabilitation and by monitoring plots

a)

		Blocks	Plots
Scat numbers	Kendall's τ-b	0·277	0·098
	P value	*0·291*	*0·421*
Scat locations	Kendall's τ-b	0·277	0·061
	P value	*0·291*	*0·623*

b)

		Blocks		Plots	
		Richness*	Canopy†	Richness*	Canopy†
Scat numbers	Kendall's τ-b	0·629	0·239	0·303	0·215
	P value	0·051	0·506	0·020	0·078
Scat locations	Kendall's τ-b	0·584	0·291	0·330	0·231
	P value	0·077	0·415	0·012	0·060

*Mean number of *Eucalyptus* and *Corymbia* species in the area.
†Mean percentage of canopy cover in the area.

When zero models were compared, we found that the most parsimonious model based on AICc included plants and bare ground (ΔAICc > 2). As a result, we assumed a binomial part model including plants and bare ground for the 34 models with different count parts.

The ranking of the 34 count models did not strongly support any single model (Table 4). We found that the best model had an AICc weight of 0·38 (M22), and two other models were also well supported (ΔAICc < 2). *Eucalyptus* and *Corymbia* species richness and canopy cover were found in all three models.

Across all models, *Eucalyptus* and *Corymbia* species richness had a positive coefficient (β = 0·70 SE = 0·16) and the highest relative importance (0·98). Similarly to richness, canopy cover had a positive coefficient (β = 0·67 SE = 0·21) and a relative importance of 0·97. In contrast, elevation had a negative coefficient (β = −0·31 SE = 0·16) and a relative importance of 0·39. For other estimates, we found that the sign of the coefficient was unstable across models (undisturbed, wetland, CBH) and/or the importance was close to zero (density, undisturbed, CBH and slope). As a result, the influence of these variables could not be reliably ascertained.

Table 4. Ranks of the three best supported models 'Scat locations ~ count part + binomial part' comparing the count parts based on AICc (the count part focuses on variables influencing koala occurrence)

Model	Variables in count part*	*K*†	AiCc‡	Δ AICc§	AICc weight	Evidence ratio
M22	Richness + canopy + elevation	7	131·5	0·0	0·38	1·0
M23	Richness + canopy + wetland	7	132·3	0·8	0·25	1·5
M3	Richness + canopy	6	133·0	1·5	0·18	2·1

*Richness, mean number of *Eucalyptus* and *Corymbia* species in the plot; canopy, percentage of canopy cover in the plot; elevation, elevation in metres above sea level of the plot; wetland, square root Euclidean distance in metres from the plot to the edge of swamps; *Melaleuca* forest or Mangroves (see Table 1).
†Number of parameters.
‡Akaike Information Criterion corrected for small sample size.
§AICc differences.

COMPARING *P. CINEREUS*-RELEVANT VARIABLES TO *PHASCOLARCTOS CINEREUS* OCCURRENCE

The two variables with highest relative importance (*P. cinereus* tree richness and canopy cover) averaged across each block of rehabilitation did not correlate with averaged number of *P. cinereus* scats or scat locations across the same blocks (Table 3b). At the plot level, rankings of the best plots based on tree richness correlated to scat numbers and scat locations, but rankings on based canopy cover did not (Table 3b).

Discussion

This study demonstrates that the assumption currently used by mining companies and endorsed by governmental legislators, i.e. that flora can be used as a surrogate for fauna, is not always a robust assumption. Indeed, rankings based on the mine's flora criteria did not correlate with *P. cinereus* occurrence (Table 3a) and more often than not, mine rehabilitated areas with high flora quality (>70% of flora criteria met) had no *P. cinereus* signs (Figure S5). Thus, in our study, there was no threshold in flora criteria success that would ensure fauna occurrence. Even habitat variables chosen specifically to be relevant to *P. cinereus* did not associate consistently with *P. cinereus* occurrence. These results signal to the mining industry and its supporting policies that any flora-only-based assessment of rehabilitation success may not be accurately assessing fauna recolonization. Stakeholders play a powerful role in mine closure legislation, as ultimately the success of mine closure encompasses social and political considerations (Hobbs 2007). Our results suggest that the industry and its legislators need to consider the limits of the fauna/flora paradigm and develop frameworks where fauna is assessed in its own right, and other stakeholders should use their role to ensure this is a priority.

The lack of correlation between habitat proxies and fauna could also be exacerbated for fauna limited by other factors than those routinely included in mining criteria (e.g. tree hollows). Moreover, finding relevant proxies for fauna less well studied than *P. cinereus*, and for which relevant habitat characteristics are unknown, would be even more difficult. Our results suggest that some fauna must be constrained by more than commonly assessed habitat proxies. We cannot underestimate the influence on species recolonization of factors such as interactions with other species, social structure and behaviour or dispersal abilities (Majer 1989; Soulé *et al.* 2005; Ellis, Melzer & Bercovitch 2009; Fletcher & Sieving 2010). Finally, the landscape context will influence the recolonization of rehabilitated areas by fauna. Indeed, fauna recolonization relies not only on the presence of fauna in undisturbed surrounding areas, but also on the ability of these populations to produce dispersing individuals and the connectivity between these populations and rehabilitated areas. As a result, any rehabilitation project has to be integrative and preserve species in the areas not directly impacted by mining (see example of predator control in Nichols & Nichols 2003). While developing fauna criteria for recolonization success, managers need to be aware of the influences of the landscape context on faunal recolonization.

Similar to Weaver's (1995) study, we found that the scale of observation influenced potential correlations (as habitat variables were found to be more correlated to *P. cinereus*'s occurrence at the plot level rather than at the scale used by the mining company). This indicates that the scale at which habitat criteria are assessed might not appropriately reflects the scale at which variables influence fauna recolonization. In addition, there might be more than one relevant scale for fauna depending on the habitat variables chosen (Cale & Hobbs 1994; Lindenmayer 2000; Cunningham *et al.* 2007). Even if habitat criteria could be put in place to assess fauna recolonization at appropriate scales, other issues remain. For instance, locally developed habitat variables correlated with fauna recolonization may not be applicable to other areas. Indeed, it has been shown that the relationship between fauna and habitat variables can vary substantially from region to region (Whittingham *et al.* 2007; McAlpine *et al.* 2008), and in fact, habitat thresholds can also vary across different regions (Rhodes *et al.* 2008).

Given all these difficulties, it seems that using habitat proxies will often not accurately reflect fauna recolonization. Ultimately, there are two main considerations for choosing between using habitat proxies and fauna criteria.

First, one needs to weigh the costs of assessing the relevance of proxies against using a direct fauna criterion (i.e. the expense of ensuring the available proxy is relevant against the expense of directly monitoring the fauna). Second, one needs to assess the consequences of using a criterion uncoupled from what it is meant to represent (i.e. how risky is it to use a habitat proxy that does not represent the fauna of interest?). This could potentially result in a management decision not based on real trends (Goldstein 1999). The final choice will depend on the fauna species of concern. For species of particular interest (e.g. vulnerable, charismatic), such as *P. cinereus*, we strongly recommend that fauna monitoring be developed, as the cost of taking an inappropriate decision will override any other cost. Moreover, fauna monitoring data are easily understood and thus have been identified as the most effective in terms of communication with stakeholders (McIntyre, McIvor & Heard 2002). This is an essential part of mine closure and restoration ecology in general as, ultimately, restoration success is measured against societal expectations (Hobbs 2004).

The obvious danger with assuming that restoring flora equals restoring fauna is to declare a site restored when only one component of its biodiversity has actually returned, while the fate of fauna remains unknown. This represents a serious threat for the conservation of biodiversity in general and for the long-term resilience of ecosystems (Fischer, Lindenmayer & Manning 2006). Consequently, even in cases where fauna species are not the direct target of rehabilitation (e.g. where there is no charismatic or threatened species to re-establish), fauna should still become an integral part of assessing rehabilitation success. Good candidates for fauna criteria include fauna involved in ecosystem processes and functions (e.g. pollinators, detritivores, Andres & Mateos 2006), as well as keystone species (Simberloff 1998) or species particularly sensitive to threats (Lambeck 1997). If the lack of congruence between flora and fauna success that we found in this study is common in restoration, developing cost effective, relevant and feasible fauna criteria is crucial. This may well be the next challenge in achieving true ecosystem restoration.

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Table S1. Flora completion criteria for mining rehabilitation.

Table S2. Variables influencing the presence of *P. cinereus*.

Table S3. Descriptions and ranks of all models.

Appendix S4. Possible biases associated with our field method.

Figure S5. Plots of fauna versus current flora criteria.