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Reevaluating suitable habitat for reintroductions: lessons learnt from the eastern barred bandicoot recovery program

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Abstract

Reintroduction and translocation programs are widely used conservation tools but their success rates are low. Poor success rates for reintroduction programs are commonly attributed to insufficient knowledge of species' habitat requirements, especially if they are critically endangered. Yet conservation managers are frequently required to make decisions about suitable reintroduction sites when information is incomplete or uncertain. A widely used strategy to assist the selection of reintroduction sites – habitat suitability models – may rely on assumptions and simplifications to fill gaps in existing data. It is essential that these models are then evaluated and refined as new evidence becomes available. In this study, we examine the effectiveness of a reintroduction program based on habitat suitability modelling: that for the critically endangered eastern barred bandicoot *Perameles gunnii* in Australia. After collecting a variety of data from the reintroduction sites, we found that habitat preferences for this species could be accurately predicted using a simple logistic regression model within two predictor variables rather than the five previously used to select reintroduction sites. This made it possible to better focus limited resources on the most suitable reintroduction sites. We believe that building such a process of review into a reintroduction program can contribute to improving its success, while ensuring that scarce conservation resources are used more effectively.

Introduction

Translocation and reintroduction of threatened species is a widespread conservation tool. Within the US, for example, an estimated 70% of all recovery plans for threatened or endangered species have recommended this approach (Tear *et al*., 1993). However, success rates are typically <50% (Griffith *et al*., 1989; Kleiman, 1989; Fischer & Lindenmayer, 2000) and can be as low as 11% (Beck *et al*., 1994) and, though the number of reintroduction programs has increased over the past two decades (Griffith *et al*., 1989; Wolf, Garland & Griffith, 1998), their success rate has not (Fischer & Lindenmayer, 2000). While it is generally accepted that a successful reintroduction or translocation program requires 'good quality' habitat (Kleiman, 1989), a frequently cited reason for program failure is insufficient knowledge of what determines habitat quality (Griffith *et al*., 1989; Wolf *et al*., 1998). This has been a particular problem in the management of critically endangered species, which have often been little studied under natural conditions and can be virtually extinct in the wild or have remnant wild populations in marginal, highly modified or fragmented habitats.

To assist the reintroduction process, several studies have explored the use of analytical methods of identifying suitable habitat before release (e.g. Merrill *et al*., 1999; Macdonald *et al*., 2000; Carroll *et al*., 2003). Multivariate statistical models (Capen, 1981), pattern recognition (Williams, Russell & Seitz, 1977) and using the numerical Habitat Suitability Index model (United States Fish and Wildlife Service, 1980) are three possible approaches. However, such techniques are often limited by an apparent lack of quantitative data and a need to make assumptions or simplifications (Burgman & Lindenmayer, 1998). While the use of robust, objective data is preferable, many programs instead utilize 'expert' knowledge (i.e. educated guesses from knowledgeable researchers) as a surrogate where such data are unavailable (Reading *et al*., 1996). Expert knowledge is commonly used in natural resource management, where decision-making is required despite a lack of data (Cowling *et al*., 2003). In these situations, expert knowledge is often combined with existing empirical evidence to model the best available information (Bock & Salski, 1998; MacMillan & Marshall, 2006). However, guidelines rarely exist to specify how to select 'experts' or how to use expert knowledge to guide reintroduction programs.

Since the emergence of habitat modelling techniques, the overall success rate of reintroduction programs seems to have altered little. Roloff & Kernohan (1999) identified inadequate consideration of parameter variability, inappropriate spatial scales, too narrow a range of model values, and population data collected over insufficient temporal scales as common deficiencies in these processes. Very few studies have attempted to use this knowledge to enhance the criteria by which they define habitat quality for a recovery program (Burgman & Lindenmayer, 1998), especially those based largely on assumptions, simplifications and subjective information. Here, we examine the extent to which field validation of habitat quality information, based primarily on expert knowledge, can improve our understanding of the habitat requirements of a threatened species, the eastern barred bandicoot *Perameles gunnii* in Victoria, Australia.

Perameles gunnii is a small to medium-sized (500 g) ground-dwelling marsupial (Family Peramelidae). It was originally widespread across the grasslands and grassy woodlands of the western Victorian plains (Seebeck, 1979; Kemper, 1990). After European settlement it suffered a 99% reduction in range (Seebeck, 1979), a decline closely allied to that of native grasslands in Victoria (Kirkpatrick *et al.*, 1995). Although viable eastern barred bandicoot populations exist in Tasmania, by the 1970s only one population remained on mainland Australia: in the town outskirts and rubbish tip at Hamilton in western Victoria (Seebeck, 1979).

In 1988 a recovery program was established, involving captive breeding and an initial release of bandicoots at three sites within their former range: Hamilton Community Parklands, Woodlands Historic Park and a farming property, 'Mooramong' (Fig. 1). While a population of around 400 animals was briefly achieved at one site, difficulties in establishing self-sustaining populations led to the development of a habitat suitability model for the species (Watson & Halley, 2000). The habitat model was based on subjective interpretation by experts of the available quantitative data, to determine the most important habitat variables to include and how these variables should be scored (Reading *et al.*, 1996). The model used five criteria thought to determine bandicoot habitat: (1) site size; (2) habitat structure; (3) predation level (by introduced carnivores, predominantly red fox *Vulpes vulpes* and cats *Felis catus*); (4) site shape; (5) long-term site security. A further four reintroduction sites were then selected: 'Lanark,' Lake Goldsmith Nature Reserve, Floating Islands Nature Reserve and Cobra Killuc Wildlife Reserve. However, by 2000, the situation for mainland *P. gunnii* had not greatly improved; populations were low or absent at all sites despite additional releases, suggesting a needed to reevaluate the view of suitable habitat.

Our study aimed to examine whether reevaluating habitat preferences might provide new insight into appropriate reintroduction site selection for conservation programs. We use the eastern barred bandicoot as a case study and compare bandicoot populations with a variety of habitat variables across the reintroduction sites to identify those associated with the presence of bandicoots. We then use multivariate analysis to develop a model that can provide a new understanding of habitat suitability, allow current release sites to be rationalized and inform future site selection.

Methods

Study areas

We surveyed all seven reintroduction sites established within the former range of *P. gunnii* in Victoria (Fig. 1). Animals had been released at these sites over the preceding 10 years (Table 1).

The release sites differ in vegetation structure and land management history. Lanark and Mooramong are working farms with associated grassland and grassy woodland components into which bandicoots have been released. The 111 ha release site at Lanark is a mixture of low, closed woodland, open grassland, permanent and ephemeral wetlands. The 200 ha release area at Mooramong comprises

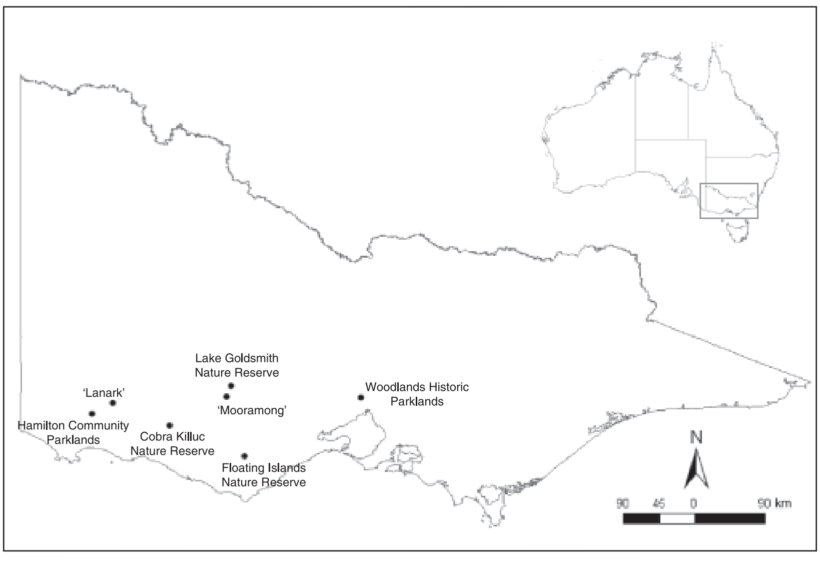

Figure 1 A map of reintroduction sites for the eastern barred bandicoot *Perameles gunnii* on mainland Australia. An additional reintroduction site was established at Mount Rothwell (not shown) subsequent to this study.

Table 1 The number of animals released at each reintroduction site for the seven reintroduction sites for the eastern barred bandicoot *Perameles gunnii* as at 2001

Site	Number of animals released	Period of release	Site size (ha)	Number released per hectare
Lake Goldsmith Reserve	80	1994–1998	100	0.80
'Mooramong'	85	1992–1995	110	0.77
Hamilton Community Parklands	75	1990–1995	100	0.75
'Lanark'	55	1994–1997	88	0.63
Floating Islands Reserve	50	1994–1997	85	0.59
Woodlands Historic Park	129	1998–1999	400	0.32
Cobra Killuc Reserve	105	1998–1999	500	0.21

open grassland and shrubland together with ephemeral and permanent wetlands, and the adjacent homestead buildings and garden. Four sites – Floating Islands Nature Reserve, Cobra Killuc Wildlife Reserve, Lake Goldsmith Nature Reserve and Woodlands Historic Park – are public land, managed as protected areas. Floating Islands is 85 ha of grassland, open grassy woodland and low open forest around a mixture of ephemeral wetlands. Cobra Killuc is a 500 ha grassland and grassy woodland remnant that includes former non-endemic eucalypt plantations. Lake Goldsmith is a 150 ha grassy woodland and native and exotic grassland around a shallow, ephemeral lake. The Woodlands reserve is a 400 ha complex of grassland, grassy woodland and some closed woodland. The site includes an area of anti-predator fence, mainly to exclude *V. vulpes* and *F. catus*. Hamilton parklands is a locally managed 100 ha complex of native grassland and grassy woodland surrounded by an anti-predator fence, also targeted at *V. vulpes* and *F. catus*.

We collected data on bandicoot abundance and habitat components primarily from quadrats within each reintroduction site, using 10 randomly distributed 5 m^2 quadrats for seven sites and two sampling periods. Owing to logistic constraints, we applied the same sampling strategy to all sites, each of which was sampled twice: once during spring (September–November) 2000 and once during summer (December–January) 2000–2001. This provided a balanced design less sensitive to the violation of assumptions.

Bandicoot abundance

In comparing habitat attributes with abundance, we used an indirect index of bandicoot abundance; the quantities of fresh diggings and fresh scats in a quadrat. Eastern barred bandicoot diggings – distinctive conical holes – have been shown to be an indicator of abundance for this species (Mallick, Driessen & Hocking, 1997). We recorded only fresh diggings, denoted by a moist soil pile and an absence of debris in the hole. Dig abundance was measured on an ordinal scale ('0' denoting no diggings and '3' very abundant diggings). An ordinal scale was considered to be both a more conservative estimate of abundance (to avoid over-estimating abundance where diggings are aggregated) and to provide a more efficient measure. Using a similar scale, we also recorded the abundance of fresh scats. As *P. gunnii* is the only bandicoot species that occurs in this region, we were confident that scats could be reliably identified. The scat and diggings measures were then combined to provide a per-quadrat index (0–6) of bandicoot abundance and a presence/absence datum.

Habitat components

The Reading *et al.* (1996) habitat model used the five criteria summarized in Table 2. However, many studies conducted on the Hamilton and Tasmanian populations (e.g. Seebeck, 1979; Dufty, 1994*a*) have suggested habitat components potentially important to bandicoots that were not included in the model. We did not repeat the model computations, but instead measured numerous habitat parameters identified as potentially important, omitting the criterion related to long-term management issues because it could not be evaluated adequately.

The reintroduction sites for *P. gunnii* differ widely in size (85–500 ha). While this can influence carrying capacity and long-term viability of reintroduced populations, the relationship between site area and numbers of individuals released may also be important if Allee effects prevent the population from establishing (Burgman, Ferson & Akcakaya, 1992). We therefore included measures of the size (ha) of reintroduction sites in our analyses, along with the reintroduction history (number of bandicoots and number per ha).

Climatic factors may be important to reintroduction success; breeding will cease in *P. gunnii* populations during periods of prolonged drought (Watson & Halley, 2000). We included rainfall for the previous year and previous 4 years (mm) and mean temperature maxima and minima (°C) in our analyses.

Trapping studies have shown that trap success can be related to high floristic diversity and structural complexity of the habitat (Dufty, 1994*a*) and so these factors were considered potentially important. We measured floristic diversity by recording the percentage cover of all plant species, including trees and shrubs, within the 5 m^2 quadrat using the Braun–Blanquet scale. We assessed the structural complexity of the habitat within a 50 m radius around each quadrat, this being the furthest distance *P. gunnii* is known

Table 2 The five criteria (indices) used by Reading *et al*. (1996) to produce a habitat suitability model for the eastern barred bandicoot *Perameles gunnii*

Criteria	Description
Site size	Sites are ranked based on their proximity to 2000 ha, the size determined from available density estimates (1.5 animals ha^{-1}) as large enough to support a genetically viable population *Site score*: 50 ha = 0.1–2000 ha = 0.99
Habitat structure	Based around 'cover,' a factor defined as including both living and dead vegetation and artificial shelter. Both the distance to cover and the amount of cover are combined in this factor *Distance score*: 70% >50 m from cover – 100% within 50 m of cover. *Amount score*: 2% cover = 0.1–100% cover = 1.0
Predator control	Combines the potential of a site for effective predator control (targeting *Vulpes vulpes* and *Felix catus*) with the distance to the nearest town *Potential score*: poor potential for control = 1 – greatest potential for control = 5. *Distance score*: <1 km = 1 – >10 km = 5
Site shape	Allows for edge effects and is measured as a function of site area to site perimeter. Shape score = (Axis ratio index × Site shape)$^{1/2}$
Site security	A management variable that incorporates information on fire potential, land tenure, the degree of local support or opposition, on-site personnel, equipment and support and site availability *Security score* = ([FP/5] + [LT/5] + [LS/5] + [OS/5] + [AV/5])/5

to forage away from cover (Dufty, 1994*b*). Structural complexity estimates included measuring the abundances of tree and shrub cover, short grass, dense long grass, fallen timber, surface rock, artificial cover and surface water to this distance on 0–3 ordinal scales. Cover is vital to *P. gunnii* (Dufty, 1994*b*), and artificial cover was included because the persistence of a wild population in a rubbish tip at Hamilton (Seebeck, 1979) suggested that this species does not discriminate between vegetative and artificial cover.

Vertical and lateral vegetative covers were used as surrogates for the visibility of bandicoots to avian and ground-dwelling predators, respectively. We determined vertical cover using Barkman's (1988) variant point quadrat method. We counted the number of times vegetation contacted a 1.6-m-long, 8-mm-diameter vertical pole, fixed 10 m from the center of each quadrat, at the eight evenly spaced compass bearings (Fig. 2). We measured lateral cover by evaluating the amount of vegetation obstructing an observer's view from the quadrat centre, from roughly 40 cm above ground, the approximate height of the red fox *V. vulpes*. Bandicoot-sized placards [based on Backhouse & Crosthwaite (1996)], marked with a contrasting black-and-white chequered pattern, were mounted on wooden stakes at these points and the number of placards visible recorded, as in Cox, Garrot & Cary (1997).

Victorian Department of Sustainability and Environment (DSE) trapping records show that some bandicoots suffer dramatic weight loss post-release (DSE, unpubl. data). This may be attributable to the largely unnatural diet of bandicoots within the captive breeding program (Myroniuk, 1995), but food limitation in the wild may also be important. We defined food availability as the invertebrate material available to bandicoots on the ground surface and in soil to a depth of 12 cm. To assess surface invertebrates, three 225 mL pitfall traps containing 70% ethanol were placed at each quadrat (Fig. 2). Traps were set at dusk each night for

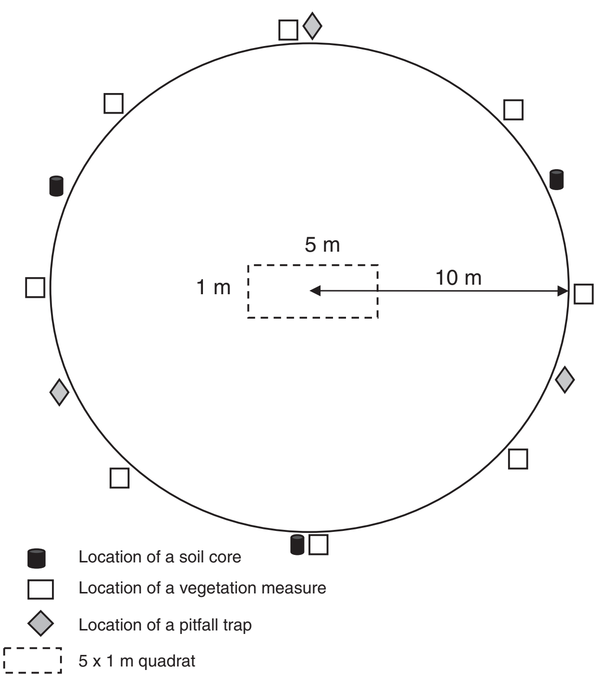

Figure 2 Diagram of the quadrat arrangement used for data collection at each of the reintroduction sites for the eastern barred bandicoot *Perameles gunnii.*

three nights, cleared each morning and any invertebrates 1 cm or greater in length were identified to Order level. Soil invertebrates were sampled using three soil cores of 530 cm^3 also taken from around each quadrat (Fig. 2). These were then sieved and any invertebrates over 1 cm identified similarly.

Soil properties may also influence the availability and accessibility of fossorial prey and so can be influenced by soil type and level of compaction (King & Hutchinson, 1983). To understand food accessibility, we classified the soil type within each quadrat through a textural method (Reynolds, 1971), ranked the soils by increasing clay content, and assessed soil compaction using a penetrometer.

To investigate food preferences, bandicoot scats were collected from the quadrats and their contents identified to Order level. To determine the composition of the scat, the relative proportions of each invertebrate Order were determined by obtaining the dry weight of the undigested fragments. This was then compared with the availability of invertebrates derived from pitfall trapping and soil cores.

Two other ground-dwelling mammals, the European rabbit *Oryctolagus cuniculus* and the swamp rat *Rattus lutreolus* were also sufficiently abundant at some sites to warrant consideration. Their abundance was measured based on an overall assessment of the abundance of scats and the abundance of species-specific marking of the soil (scratching for *O. cuniculus* and tunnelling activity for *R. lutreolus*) on an ordinal scale ('0' denoting no soil markings or scats and '3' very abundant soil markings and/or scats).

Although several predators are known to prey on eastern barred bandicoots, the red fox was considered most important at the reintroduction sites. We used DSE predator control records to determine and rate the intensity of predator control at each site on a 0–5 ordinal scale (Table 3). Additionally, we assessed relative fox abundance around each quadrat using a combination of sightings, scats, tracks or remains of kills, and combined these into an ordinal ranking, with '0' denoting absence through to '3' denoting very abundant fox activity.

Data analysis

We used a Kruskal–Wallis test to compare bandicoot abundance data across the seven reintroduction sites and a Mann–Whitney test to compare their abundances at different times of year. To examine the performance of the habitat suitability parameters, we compared sites differing in bandicoot abundance by Kruskal–Wallis tests, and quadrats with and without bandicoots by Mann–Whitney tests. To identify any significant associations between our bandicoot abundance index and habitat components, we used the non-parametric correlation analysis, Kendall's τ, with Holm–Bonferroni correction for multiple comparisons ($\alpha = 0.02$); variables failing to show any relationship were excluded from subsequent analyses. The remaining variables were subjected to a standard principal components analysis (PCA) to select the most important variable associated with each component and to exclude correlated variables.

To obtain a predictive measure of bandicoot presence within reintroduction sites, we used sequential logistic regression analysis to regress bandicoot presence/absence data from the individual quadrats against the variables selected through PCA. The threshold probability used to discriminate between presence and absence was 0.5. Logistic regression was chosen because it is as accurate as other multiple linear regression models but subject to lower variation than using abundance data (Pearce & Ferrier, 2001). Modelling was carried out on the combined datasets for spring and summer, as statistical testing found no difference in the abundance of bandicoots between the sampling periods (seasons) (see 'Results'). Finally, the predicted bandicoot probabilities were sorted by reintroduction site and mean habitat suitability scores were calculated for each.

Results

Bandicoots were detected at five of the seven reintroduction sites, consistent with the trapping records for these sites (DSE, unpubl. data). Their detection rate varied considerably between reintroduction sites, with bandicoots detected at between 0 and 70% of quadrats. We found significant differences in bandicoot abundance between reintroduction sites (K–W, $\chi^2 = 44.4$; d.f. = 6; $P < 0.001$) (Fig. 3) but no difference between sampling seasons (M–W, $U = 2303.5$; $Z = -7.60$; $P = 0.447$).

While the original model suggested that large reintroduction sites are preferable, Kendall's τ correlation analysis showed no relationship between site size and whether bandicoots were present ($\tau = -0.12$; $n = 140$; $P = 0.09$). While the number of bandicoots released at a site was negatively correlated with bandicoot abundance ($\tau = -0.15$; $n = 140$; $P = 0.03$), no relationship was observed when the number bandicoots released was corrected for site size ($\tau = 0.13$; $n = 140$; $P = 0.06$).

Table 3 The feral predator control scale used to rank the intensity of control measures used at the different reintroduction sites for the eastern barred bandicoot *Perameles gunnii*

Predator management ranking	Description
0	No predator control used
1	One method of control used regularly, for example regular poison baiting
2	Two methods of control used regularly, for example poisoning and shooting
3	Three methods of control used regularly, for example poisoning, shooting and den destruction
4	Three methods of control used plus additional protection provided by an on-site presence or predator control fence
5	Three methods of control used in a strategic manner, by a profession operator, plus additional protection provided by an on-site presence or predator control fence

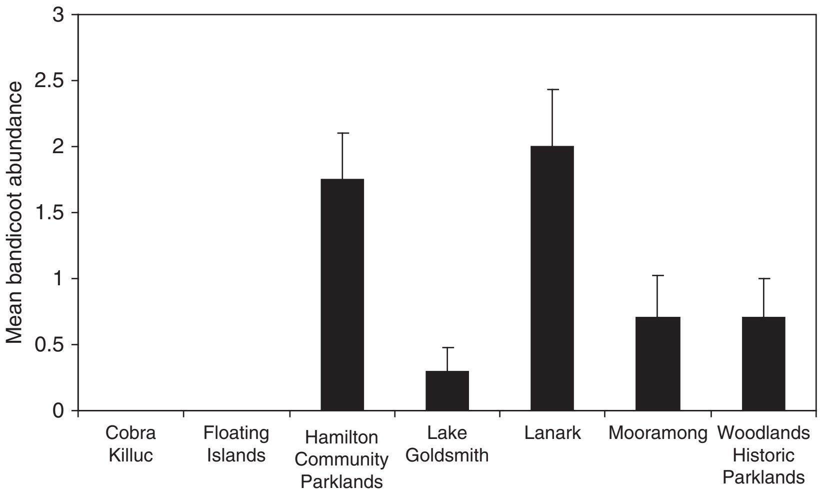

Figure 3 The mean (± standard error) bandicoot abundance recorded for each of the reintroduction sites for *Perameles gunnii*.

Habitat components

For habitat structure, no association was found between lateral ($\tau = -0.03$; $n = 140$; $P = 0.71$) or vertical cover ($\tau = -0.03$; $n = 140$; $P = 0.71$) and the presence of bandicoots. We found that fox activity tended to be lower at quadrats where bandicoots were present (M–W, $U = 1649.5$; $Z = -1.90$; $P = 0.058$), whereas the intensity of predator control was higher at these sites (M–W, $U = 839.5$; $Z = -5.77$; $P < 0.001$). Ninety-four plant species were identified within the quadrats across all sites though no difference in floristic diversity was detected ($\tau = -0.01$; $n = 140$; $P = 0.926$). European rabbits *O. cuniculus* were present at all sites, while swamp rats *R. lutreolus* and their tunnels were abundant in two sites where bandicoots were also abundant (Lanark, Hamilton Community Parklands). Members of 19 invertebrate Orders were found in pitfall traps and 18 in soil cores, 14 of which were represented in bandicoot scats.

Non-parametric correlation analysis identified only eight of the variables tested as significantly correlated with bandicoot abundance after Holm–Bonferroni correction: swamp rat cover abundance ($\tau = 0.47$; $n = 140$; $P < 0.001$), predator management intensity ($\tau = 0.43$; $n = 140$; $P < 0.01$), artificial cover abundance ($\tau = 0.31$; $n = 140$; $P < 0.001$), sweet vernal-grass *Anthoxanthum odoratum* cover ($\tau = 0.31$; $n = 140$; $P < 0.001$), soil clay content ($\tau = 0.29$; $n = 140$; $P < 0.001$) and the presence of *Isopoda* and arachnids in bandicoot scats ($\tau = 0.39$; $n = 140$; $P < 0.001$ and $\tau = 0.31$; $n = 140$; $P < 0.001$) were positively associated with bandicoot abundance. Surface rock abundance was negatively associated ($\tau = -0.31$; $n = 140$; $P < 0.01$) with bandicoot abundance across the sites. There was no clear evidence that other habitat variables measured were significantly related to bandicoot abundance within the reintroduction sites (Table 4). After PCA, we retained just four potential predictor variables: predator management, *A. odoratum* cover, amount of surface rock and artificial cover abundance.

A new habitat model

Sequential logistic regression analysis showed that the probability that bandicoots would be present could be predicted accurately by two variables, predator management intensity ($B = 1.55$; SE $= 0.36$; $P < 0.001$) and surface rock abundance ($B = -0.79$; SE $= 0.30$; $P = 0.008$), together with a constant (-5.61; SE $= 1.34$; $P < 0.001$), in 80% of cases. The model was highly reliable and predicted false positives in only 9% of cases, and false negatives in only 10% of cases. An examination of the distribution of residuals by graphical analyses showed no evidence of departure from a normal distribution or any measurement sequence trends, supporting the adequacy of the model. Further validation of the model involved cross validation by randomly selecting half of the full dataset. For each dataset, the contribution of the four potential predictor variables previously identified were assessed in a logistic regression analysis by simultaneous direct entry using the Wald test; the only variables to meet the Hosmer–Lemeshow criteria were the two already identified using the full dataset, together with a constant. These two variables were used to compute two-predictor models by logistic regression using sequential direct entry and the process was repeated 10 times. The results from the cross validation datasets were similar to one another and to the model from the full dataset, supporting the model's validity. Including site differences failed to add to predictability; if locality was entered into the sequential logistic regression as a third predictor it did not improve the outcome.

This model was then reapplied to the dataset and the mean probability of detecting *P. gunnii* calculated for each reintroduction site. This enabled the sites to be ranked according to their suitability as reintroduction sites for this species. Table 5 outlines the rank order of reintroduction sites as described by our predictive model, together with mean scores of the habitat attributes significantly associated with bandicoot abundance.

Table 4 Non-parametric correlation analyses for eastern barred bandicoot *Perameles gunnii* abundance against the habitat components measured

Variable	Kendall's τ correlation coefficient[a]	Significance[b]	Number of samples	Holm–Bonferroni correction
Site size (ha)	−0.12	0.090	140	–
Number of bandicoots released	−0.12	0.098	140	–
Number of bandicoots released per hectare	0.13	0.065	140	–
Annual rainfall (mm)	−0.03	0.709	140	–
Mean maximum air temperature (°C)	0.18	0.032	100	–
Mean minimum air temperature (°C)	−0.19	0.022	100	–
Season	−0.06	0.447	140	–
Structural complexity	0.07	0.332	140	–
Tree abundance	0.04	0.644	140	–
Shrub abundance	0.02	0.819	140	–
Short grass abundance	0.18	0.019[b]	140	–
Dense long grass abundance	0.05	0.491	140	–
Fallen timber abundance	−0.09	0.204	140	–
Rock abundance	−0.31[a]	<0.001[b]	140	<0.001
Artificial cover abundance	0.31[a]	<0.001[b]	140	<0.001
Water abundance	0.16	0.036	140	–
Floristic diversity	−0.06	0.926	140	–
Acacia mearnsii	0.10	0.194	140	–
Acacia melanoxylon	0.19	0.018[b]	140	–
Acacia paradoxa	0.19	0.018[b]	140	–
Acacia salicina	−0.05	0.528	140	–
Acaena agnipila	−0.05	0.528	140	–
Acaena anserinifolia	−0.09	0.271	140	–
Acaena echinata	−0.07	0.370	140	–
Agrostis tenuis	−0.05	0.528	140	–
Aira cupaniana	0.08	0.321	140	–
Anthoxanthum odoratum	0.31[a]	<0.001[b]	140	<0.001
Arctotheca calendula	0.09	0.230	140	–
Arthropdium strictum	−0.17	0.034	140	–
Austrodanthonia carphoides	0.06	0.422	140	–
Avena barbata	−0.15	0.051	140	–
Avena fatua	−0.12	0.103	140	–
Briza maxima	0.09	0.229	140	–
Briza minor	0.01	0.901	140	–
Bromus diandrus	−0.06	0.457	140	–
Bromus mollis	0.06	0.441	140	–
Bromus rubens	−0.09	0.271	140	–
Brunonia australis	−0.07	0.370	140	–
Burchardia umbellata	0.13	0.096	140	–
Bursaria spinosa	−0.05	0.528	140	–
Calaphalus sonderi	−0.06	0.528	140	–
Carduus pycnocephalus	−0.10	0.205	140	–
Carduus tenuiflorus	−0.07	0.370	140	–
Casuarina stricta	0.01	0.950	140	–
Cenchrus longispinus	0.08	0.300	140	–
Chrysocephalum apiculatum	−0.07	0.370	140	–
Cirsium vulgare	−0.05	0.528	140	–
Convolvulus erubescens	−0.07	0.370	140	–
Cynodon dactylon	−0.05	0.528	140	–
Dactylis glomerata	−0.05	0.457	140	–
Danthonia caespitosa	−0.14	0.088	140	–
Danthonia setaceae	−0.08	0.338	140	–
Danthonia spp.	0.03	0.714	140	–
Dichelachne crinita	−0.05	0.528	140	–
Dichondra repens	−0.09	0.271	140	–
Drosera peltata	0.18	0.027	140	–

Table 4. Continued.

Variable	Kendall's τ correlation coefficient[a]	Significance[b]	Number of samples	Holm–Bonferroni correction
Elymus scaber	−0.09	0.170	140	–
Erodium botrys	−0.07	0.384	140	–
Eucalyptus botryoides	−0.15	0.067	140	–
Eucalyptus camaldulensis	0.09	0.288	140	–
Eucalyptus cladocalyx	−0.11	0.152	140	–
Eucalyptus maculata	0.17	0.032	140	–
Eucalyptus melliodora	0.07	0.370	140	–
Eucalyptus microcarpa	−0.11	0.152	140	–
Eucalyptus ovata	−0.07	0.370	140	–
Eucalyptus viminalis	0.06	0.471	140	–
Geranium dissectum	−0.07	0.370	140	–
Holcus lanatus	−0.12	0.101	140	–
Hordeum leporinum	−0.09	0.271	140	–
Hydrocotyle laxiflora	−0.09	0.271	140	–
Hypericum gramineum	0.06	0.370	140	–
Hypochoeris radicata	0.23	0.002[b]	140	–
Isolepis australiensis	−0.02	0.801	140	–
Isolepis marginata	−0.05	0.528	140	–
Lobelia pratioides	0.11	0.162	140	–
Lolium perenne	0.09	0.241	140	–
Lolium rigidum	0.04	0.607	140	–
L. rigidum	−0.07	0.370	140	–
Lomandra effusa	−0.07	0.489	140	–
Lomandra filiformis	−0.14	0.088	140	–
Microtis parviflora	−0.09	0.271	140	–
Moss	−0.19	0.012[b]	140	–
Myriocephalus stuartii	−0.05	0.528	140	–
Nassella nassiana	−0.01	0.913	140	–
Nassella trichotoma	0.11	0.528	140	–
Oxalis percaprae	0.02	0.848	140	–
Pelargonium rodneyanum	−0.05	0.528	140	–
Phalaris aquatica	−0.02	0.804	140	–
Phalaris minor	−0.05	0.528	140	–
Phalaris paradoxa	−0.05	0.528	140	–
Pimelea glauca	−0.05	0.528	140	–
Pinus radiata	−0.05	0.528	140	–
Plantago lanceolata	0.19	0.014[b]	140	–
Poa labillardieri	0.06	0.459	140	–
Polypogon monspeliensis	−0.07	0.370	140	–
Pteridium esculentum	−0.16	0.039	140	–
Ptilotus erubescens	−0.07	0.368	139	–
Romulea rosea	0.18	0.023	139	–
Rumex arctosella	0.06	0.454	140	–
Schoenus apogon	−0.06	0.455	140	–
Stellaria media	0.02	0.816	140	–
Stipa scabra	−0.05	0.524	140	–
Themeda triandra	0.15	0.049	140	–
Tricoryne elatior	0.10	0.213	140	–
Trifolium balansae	−0.05	0.528	140	–
Trifolium subterraneum	−0.02	0.812	140	–
Vicia sativa	−0.09	0.271	140	–
Vulpia bromoides	0.09	0.238	140	–
Vulpia myuros	−0.05	0.528	140	–
Lateral cover	−0.03	0.711	140	–
Vertical cover	−0.03	0.705	140	–
Pitfall trap totals	0.06	0.351	140	–
Amphipoda (pitfall)	−0.17	0.030	140	–

Table 4. Continued.

Variable	Kendall's τ correlation coefficient[a]	Significance[b]	Number of samples	Holm–Bonferroni correction
Araneae (pitfall)	<0.00	0.953	140	–
Blattodea (pitfall)	−0.11	0.182	140	–
Chilopoda (pitfall)	0.01	0.872	140	–
Coleoptera (pitfall)	0.12	0.088	140	–
Coleopteran larvae (pitfall)	−0.09	0.271	140	–
Dermaptera (pitfall)	0.07	0.364	140	–
Diotocardia (pitfall)	0.17	0.032	140	–
Diplopoda (pitfall)	−0.01	0.862	140	–
Diplura (pitfall)	0.16	0.023	140	–
Diptera (pitfall)	−0.04	0.579	140	–
Dipteran larvae (pitfall)	0.11	0.170	140	–
Hemiptera (pitfall)	0.07	0.400	140	–
Hymenoptera (pitfall)	−0.07	0.317	140	–
Isopoda (pitfall)	0.02	0.756	140	–
Lepidoptera (pitfall)	0.12	0.137	140	–
Lepidopteran larvae (pitfall)	−0.04	0.599	140	–
Oligochaeta (pitfall)	0.02	0.808	140	–
Orthoptera (pitfall)	−0.08	0.314	140	–
Scorpionida (pitfall)	−0.15	0.051	140	–
Sigmuretha (pitfall)	<0.00	0.972	140	–
Tricladida (pitfall)	0.10	0.223	140	–
Total soil cores	0.16	0.022	140	–
Amphipoda (cores)	−0.05	0.528	140	–
Araneae (cores)	−0.08	0.331	140	–
Blattodea (cores)	0.03	0.673	140	–
Chilopoda (cores)	−0.01	0.866	140	–
Coleoptera (cores)	−0.11	0.172	140	–
Coleopteran larvae (cores)	0.06	0.429	140	–
Diotocardia (cores)	−0.07	0.370	140	–
Diplopoda (cores)	−0.13	0.084	140	–
Diplura (cores)	0.08	0.316	140	–
Diptera (soil cores)	−0.04	0.579	140	–
Dipteran larvae (cores)	0.04	0.569	140	–
Hemiptera (cores)	0.04	0.629	140	–
Hymenoptera (cores)	−0.06	0.462	140	–
Isopoda (cores)	−0.09	0.282	140	–
Lepidoptera (cores)	0.12	0.137	140	–
Lepidopteran larvae (cores)	0.06	0.456	140	–
Oligochaeta (cores)	0.10	0.184	140	–
Orthoptera (cores)	0.13	0.184	140	–
Sigmuretha (cores)	−0.07	0.370	140	–
Tricladida (cores)	0.09	0.288	140	–
Total invertebrates (pitfalls and soil cores)	0.04	0.538	140	–
Soil clay content	0.29[a]	<0.001[b]	140	<0.001
Soil compaction	0.12	0.076	140	–
Scat totals	−0.02	0.823	140	–
Aranaeae proportion (scat)	−0.31[a]	0.040	30	<0.001
Blattodea proportion (scat)	0.12	0.452	30	–
Bone proportion (scat)	−0.20	0.232	30	–
Coleoptera proportion (scat)	0.10	0.493	30	–
Coleopteran larvae proportion (scat)	0.01	0.935	30	–
Diocardia proportion (scat)	0.14	0.403	30	–
Diplopoda proportion (scat)	0.25[a]	0.089	30	–
Diplura proportion (scat)	0.11	0.463	30	–
Diptera proportion (scat)	−0.16	0.303	30	–
Dipteran larvae proportion (scat)	−0.03	0.864	30	–
Fungi proportion (scat)	0.04	0.826	30	–

Table 4. Continued.

Variable	Kendall's τ correlation coefficient[a]	Significance[b]	Number of samples	Holm–Bonferroni correction
Hair proportion (scat)	−0.29[a]	0.083	30	–
Hemiptera proportion (scat)	0.21	0.167	30	–
Hymenoptera proportion (scat)	0.18	0.227	30	–
Isopoda proportion (scat)	0.39[a]	0.013[b]	30	<0.001
Lepidoptera proportion (scat)	−0.06	0.706	30	–
Lepidopteran larvae proportion (scat)	0.14	0.357	30	–
Orthoptera proportion (scat)	0.06	0.704	30	–
Plant proportion (scat)	−0.13	0.364	30	–
Soil proportion (scat)	−0.10	0.471	31	–
Rabbit abundance	−0.08	0.286	140	–
Swamp rat abundance	0.47[a]	<0.001[b]	140	<0.001
Predator control intensity	0.43[a]	<0.001[b]	140	<0.001
Fox abundance	−0.13	0.095	140	–

[a]Values that meet the Holm–Bonferroni correction criteria of ±0.25.
[b]Probability of <0.02.

Table 5 The reintroduction site suitability scores and rankings according to our model, together with scores on the habitat attributes significantly associated with eastern barred bandicoot *Perameles gunnii* abundance after multiple comparisons correction. Scores from all 'Lanark' quadrats were identical

Site	Mean habitat suitability score	Site suitability ranking	Number of quadrats	Predator control index	Swamp rat abundance	Soil clay content	*Anthoxanthum odoratum*	Artificial cover score	Surface rock score
Hamilton Community Parklands	0.636 ± 0.010	1	20	4	2.05	2.6	17.85	0.85	0.05
'Lanark'	0.645	1	20	4	1.05	2.6	5	1.65	0.05
'Mooramong'	0.395 ± 0.031	3	20	4	0	2.15	1.25	0.9	1.35
Woodlands Historic Park	0.215 ± 0.019	4	20	3	0	1.48	0.05	0.65	0.55
Lake Goldsmith Reserve	0.084 ± 0.009	5	20	3	0	2.2	0	0	1.95
Cobra Killuc Reserve	0.073 ± 0.002	6	20	2	0	1.28	0	0.45	0.05
Floating Islands Reserve	0.001 ± 0.000	7	20	0	0	1.7	0.75	0.65	2.05
Possible range of values				0–5	0–3	0–3	0–100	0–3	0–3

Discussion

The value of review

The original habitat suitability model developed to guide reintroduction site selection for the eastern barred bandicoot in Victoria was based on the available empirical data and the judgment of experts. The model represented the best knowledge available after almost 10 years of intensive management of the species, an enviable position compared with some conservation programs. Almost 5 years on, that knowledge had not translated into any self-sustaining bandicoot populations (DSE, unpubl. data). Our results show that a reevaluation of suitable habitat in light of quantitative, field data can yield some valuable insights that may assist a conservation program to manage existing reintroduction sites and select new sites.

Of the six habitat attributes significantly associated with eastern barred bandicoot abundance, effective predator control emerged as the major positive influence. Artificial cover and the characteristically dense clumps of *Anthoxanthum* provide bandicoots with shelter; and soils with moderate clay content are likely to provide bandicoots with a suitable foraging area, whereas the presence of abundant surface rocks inhibits foraging. The association with swamp rats may reflect similar habitat preferences to bandicoots (especially predator control) rather than a determinant of the bandicoot population itself.

The predictive model yielded only two predictive variables: predator control and surface rock abundance. While the original model (Reading *et al*., 1996) had selected criteria considered relevant to selecting successful reintroduction sites over a long timeframe, our results suggest that habitat attributes that predict whether bandicoots will persist at a site are far simpler. Reading *et al*. (1996) recognized the need for predator control, especially to allow a population to establish at a site, and our results show that its importance cannot be underestimated. Satisfying the ability to achieve effective predator control may require compromising other criteria such as site size, for larger sites will be more expensive to equip with a predator exclusion fence and achieve sufficiently intensive predator baiting.

Habitat structure has been emphasized in several studies on this species (Seebeck, 1979; Dufty, 1991, 1994*a*). We found that while artificial cover and a tussock-forming grass were positively associated with bandicoots at reintroduction sites, habitat structure was not a significant predictor in the model. This may indicate that, while bandicoots show a preference for sites with good cover available, particularly artificial cover that does not vary seasonally, this cannot be traded for effective predator control. The appearance of surface rock in the model may indicate that the use of an indirect measure of abundance has a bias towards forging habitat. The presence of rock is likely to make foraging difficult and bandicoots appear to avoid these areas.

Reducing and strengthening reintroduction sites

We were able to use the predictive model based primarily on effective predator control and low surface rock to rank the quality of the existing reintroduction sites for this species. This process has allowed the threatened species recovery team to reassess their priorities and direct their resources towards the four sites ranked highest by our model. A new reintroduction site has been recently added that includes an anti-predator fence: Mount Rothwell (DSE, pers. comm.). However, all sites have experienced prolonged drought and population levels are still very low.

We hope that the predictive model will be regularly updated using quantitative evidence, to ensure that it reflects the best available knowledge about habitat quality for this species. A more intensive sampling regime than used in this study may be beneficial.

Conservation science is a crisis discipline (Soule, 1985) and decisions must often be made regardless of whether rigorous evidence is available (Fazey, Fischer & Lindenmayer, 2005). Our study shows that reassessing habitat quality for an endangered species using quantitative, field data can provide valuable new information. Modelling habitat preferences with logistic regression can provide equal accuracy to direct abundance models and be relatively cheap and easy to apply (Pearce & Ferrier, 2001). We contend that reevaluation of habitat models provide an efficient and effective way to update knowledge and set new directions within a conservation program.

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