Title: Can post-mining revegetation create habitat for a threatened mammal?

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Running head: Revegetation for threatened mammal

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Pages:	24 (including tables and figures)
Tables:	1
Figures:	3
Word count:	5527

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Summary

Most revegetation conducted for biodiversity conservation aims to mimic reference ecosystems present pre-disturbance. However, revegetation can overshoot or undershoot targets, particularly in the early stages of a recovery process, resulting in conditions different from the reference model. Revegetation that has, as yet, failed to fully meet revegetation targets may, nonetheless, provide habitat for threatened species not present in reference ecosystems. To investigate this possibility, we surveyed Quokka (Setonix brachyurus), a threatened macropod, in a mining landscape in southwestern Australia. We established four sites in each of riparian forest, which is the preferred habitat of quokkas but is not mined, mid-slope forest, which is the pre-mining reference ecosystem but is not suitable habitat for quokkas, and revegetated forest on mine-pits 16 to 21 years post-mining. We recorded quokkas in all riparian forest sites and two revegetated forest sites but not in any mid-slope forest sites. Occupied revegetated sites had greater cover between 0 and 2 m and were spatially closer to riparian forest than unoccupied revegetated sites, suggesting predation pressure was likely influencing which mine-pits were occupied. Our study demonstrated post-mining revegetation can provide new habitat for a threatened species and suggested that

revegetating a small proportion of sites to provide new habitat for threatened species could be considered as a management option in some scenarios. This could improve landscape connectivity and increase both the area of available habitat and between-site heterogeneity, which could all potentially increase the ability of revegetation to conserve biodiversity.

Keywords: restoration, goals, quokka, jarrah forest, connectivity, understorey, heterogeneity

Introduction

Revegetation is becoming an increasingly important tool in the fight to conserve global biodiversity (Hobbs and Harris 2001). Most revegetation designed for biodiversity conservation aims to mimic the natural, or reference, communities present before habitat disturbance or degradation (Clewell and Aronson 2006). This is because, if revegetated areas support similar plant and animal communities as reference habitats, revegetation can increase populations of locally occurring species, including threatened species, by increasing both the area of available habitat and population connectivity. However, revegetation outcomes can be highly variable in quality and may differ structurally, or floristically, from reference communities (e.g. Mossman, et al. 2012). Given the importance of forest structure and floristics in influencing faunal communities (e.g. Jayapal, et al. 2009), it is likely that revegetated areas that differ structurally and floristically from reference communities, will also support different faunal communities including species not present in reference ecosystems. These differences may result in the development of potential habitat for threatened species that occur close to, but not in, reference ecosystems, although we are unaware of this possibility being explicitly considered as part of any post-mining revegetation strategy.

Since European settlement, Australia has suffered more mammal extinctions than any other country and many extant species remain threatened with extinction (Burbidge, et al. 2008). Many of these threatened species are restricted to southwestern Australia, a global biodiversity hotspot (Myers, et al. 2000), which now supports the only naturally occurring populations of eight threatened mammal species (Burbidge and McKenzie 1989). However, southwestern Australia is a relatively densely populated part of Australia rich in mineral resources; hence many regions in southwestern Australia are multiple-use regions where extractive activities, that potentially conflict with biodiversity conservation, are practiced (Prober and Smith 2009). One such region is the jarrah forest, a dry sclerophyll forest type

dominated by Jarrah (Eucalyptus marginata), which is endemic to southwestern Australia and supports populations of five threatened mammals. The jarrah forest is managed for multiple uses including water catchment, mining and logging, as well as conservation (Havel 1989). Several minerals, including gold and bauxite, are mined in the northern jarrah forest, with mining companies required to revegetate mine-pits to the pre-mining vegetation community (Bartle and Slessar 1989). One aim of this revegetation is to ensure that mining does not negatively affect the conservation status of threatened mammals (see McGregor, et al. 2014).

Alcoa of Australia have mined bauxite in the northern jarrah (Eucalyptus marginata) forest since 1963 and revegetated their mine-pits since 1966 (Koch 2007). Revegetation practices have improved over time and, post-1988, mine-pits have been revegetated with a mixture of local overstorey and understorey plant species (Grant and Koch 2007). However, many mine-pits revegetated between 1988 and 2000 have very high densities of canopy eucalypts and an overabundance of tall legumes (Grant 2006). This leads to these pits having dense over and understories (Grant 2006) that, structurally, more resembles vegetation found along streams and around swamps than the open jarrah forest typical of upland mid-slopes and ridgetops (Havel 1975). Given this different vegetation structure in revegetated mine-pits, there is the potential for mine-pits to provide habitat for faunal species that do not naturally occur in mid-slope jarrah forest, such as White-breasted Robins (Eopsaltria georgiana) (Craig, et al. 2015). There are previous examples of rock-inhabiting mammals colonising artificial rock piles (Schulz, et al. 2012), but we are unaware of any examples of mammals colonising revegetation in areas where they do not occur naturally (but see Nichols and Nichols 2003).

The Quokka (Setonix brachyurus) is a small (average male body weight is 3.6 kg while average female body weight is 2.9 kg) macropod endemic to southwestern Australia that has declined in range by >50% due to habitat fragmentation and predation by the introduced European Red Fox (Vulpes vulpes) (Hayward, et al. 2005a; Hayward, et al. 2005b; see Appendix A) and is considered Vulnerable under the EPBC Act (Department of Environment and Conservation 2013). In the northern jarrah forest it is almost entirely restricted to riparian forests where the understorey is dominated by Swamp Peppermint (Taxandria linearifolia), the most important food species (although it feeds on 29 understorey species), and it spends the day resting within these riparian forests. Quokkas are typically absent from the open mid-slope and ridgetop forest where mining takes place (Hayward, et al. 2005b). However, sightings from extensively mined areas led us to suspect that some revegetated mine-pits,

whose vegetation structure closely resembles riparian forests, might be occupied by quokkas. To determine whether revegetated mine-pits were occupied by quokkas, we assessed the presence of quokkas in 12 sites in unmined riparian and mid-slope jarrah forest and revegetated sites and asked the following questions: (1) Are revegetated sites occupied by quokkas?; (2) If revegetated sites are occupied, what factors influence occupation?; and (3) If revegetated sites are occupied, what are the implications for the management of revegetated sites?

Methods

Study site

This study was conducted on Alcoa of Australia's Huntly minesite (32° 34'S 116° 06'E) in the northern jarrah forest, approximately 10km N of Dwellingup. Huntly minesite has a Mediterranean climate, with hot dry summers and warm wet winters and rainfall at Dwellingup averages 1236 mm yr^{-1} with >75% falling between May and September. The minesite consists of an undulating laterised landscape with numerous riparian valleys, where the laterite has been eroded by rivers and streams (Churchward and Dimmock 1989), that support dense forest, and lateritic mid-slopes and ridgetops that support an open forest. Following mining, ~40% of the landscape has been revegetated (Triska, et al. 2016) with mid-slopes and ridgetops consist of a mosaic of unmined forest and revegetated mine-pits, while riparian areas remain unmined (Koch 2007). Revegetation practices involve re-seeding Jarrah and Marri (C. calophylla) and 76 to 111 local understorey species (Koch 2007). Understories (between 0 to 2 m) in riparian forests are dense and dominated by Swamp Teatree (Pericalymma ellipticum), Swamp Peppermint (T. linearifolia) and Lepidosperma angustatum. Understories in mid-slope and ridgetop forests are variable but typically open and include over 100 species of which Water Bush (Bossiaea aquifolium), Yellow Buttercups (Hibbertia hypericoides), Free Flowering Lasiopetalum (Lasiopetalum floribundum) and Zamia (Macrozamia riedleii) are often common. Understories in revegetated mine-pits are often dense and dominated by Water Bush (Bossiaea aquifolium), Prickly Moses (A. pulchella), Holly-leaved Mirbelia (Mirbelia dilitata), Winged Wattle (Acacia alata), Glowing Wattle (A. celastrifolia) and Wonnich (Callistachys lanceolata). For more details of the study area and revegetation methods, see Appendix B.

Experimental design, quokka and vegetation sampling

To assess which habitats quokkas occupied, we sampled for quokkas in three different forest types: swamps and streamzones (hereafter 'riparian forest'), unmined mid-slope forest (hereafter 'mid-slope forest') and revegetated mine-pits (hereafter 'revegetated forest'). We sampled four sites of each forest type, giving a total of 12 sites, with one site of each forest type located in each of four blocks (Fig. 1; Table T1). Distances between sites within each block averaged 267 ± 47 m whereas distances between blocks averaged 5020 ± 347 m. As quokka home ranges average 5.42 ha (Hayward, et al. 2004), quokkas may have moved between sites within blocks but not between blocks, especially considering the two revegetated sites where quokkas were recorded (see Results) were 4203 m apart. Of the four revegetated forest sites, three were revegetated in 1991 (i.e. 21 years old) and one in 1996 (i.e. 16 years old).

At each of the 12 sites, we sampled for quokkas using a single motion-sensitive camera left active for 21 nights between 14th August and 4th September 2012. As a six-week pilot study showed no detections with unbaited cameras, even in riparian forest where scats and tracks showed quokkas were present, we baited cameras, a method shown to be effective for detecting quokkas (Dundas, et al. 2014), by scattering apples and universal bait (oats, honey and peanut butter) in front of the cameras. Bait was replaced daily as required but was not replaced after the 14th night to collect data for another study examining sampling techniques (DAW, unpubl. data). We used nine Reconyx Hyperfire HC500 and three Bushnell Trophy motion-sensitive cameras, with different models randomly distributed between blocks and forest types. Cameras were strapped around tree trunks at ~0.3 m in height and angled slightly below horizontal. They were set to take, when triggered, three consecutive photographs with a 12 second delay between photographs and a 60 second delay between triggers. All riparian forest cameras were placed in the middle of streamzones, while mid-slope and revegetated forest cameras were placed ≥70 m from other forest types to minimise the probability of attracting quokkas from these areas. We obtained a measure of relative activity of quokkas at each site by summing the number of nights when quokkas were detected.

To identify which forest structural variables were related to quokka activity, we estimated structural variables on one 25 x 12 m plot centred on the camera at each site. Plots were rectangular to fit entirely within streamzone vegetation at riparian forest sites. Within these plots, vegetation cover between 0 to 1, 1 to 2 and >2 m and litter and fine woody debris (FWD: woody debris <20 cm average diameter) cover were visually estimated to the nearest

5% on 12 1x1 m quadrats located 4 m apart on three transects that ran through the camera location and 12.5 m on either side. We also counted the number of pieces of coarse woody debris (CWD: woody debris >20 cm average diameter) in each quadrat. We used ArcMap 9.3.1 to obtain four landscape variables: area of riparian and revegetated forest within 100 and 250 m of each camera location. These distances were chosen to match the area of quokka core and total home ranges respectively (Hayward, et al. 2004).

Statistical analyses

To analyse how quokka activity varied between forest types, we used a Kruskal-Wallis test using the number of nights (out of 21) a quokka was recorded at each site as the dependent variable and forest type as the predictor variable. We tested, post-hoc, which treatments differed significantly using Mann-Whitney U-tests. As this method did not account for differences in detectability between forest types, we also analysed the data using occupancy models that accounted for detectability (see Appendix C). However, as the results were essentially identical to our Kruskal-Wallis analyses, we present only the former for ease of interpretation and, because where data are sparse, ignoring non-detection can be better than accounting for it (Welsh, et al. 2013).

To determine whether overall vegetation structure differed between forest types, we first normalised all six structural variables (cover from 0-1, 1-2 and >2 m, litter cover, FWD and CWD; transforming the last three using $\ln[x+1]$), and used these normalised variables to create a between-site resemblance matrix using a Euclidean distance measure. We visually represented these differences using Principle Coordinates Analysis and determined whether overall vegetation structure differed between forest types using permutational Multivariate ANOVA (Anderson, et al. 2008) with forest type as a fixed factor. We also determined whether individual structural variables differed between forest types using a one-way ANOVA, with forest type as the predictor variable, and determined which means were different using Scheffe's test (Day and Quinn 1989).

To explore which structural and landscape variables were most related to quokka activity, we used best subset modelling and ran two analyses with quokka activity as the dependent variable and either structural or landscape variables as predictor variables. We used generalised linear models with a negative binomial distribution and a log link function to model all possible subsets of predictor variables. We ranked all models using AIC values corrected for small sample sizes (AIC_c) and calculated the weight (ω_i) of each model, which

is the probability that that model is the best model. We considered all models with a ΔAIC_c of <2 from the best model to be plausible and considered all models with a ω_i >0.1 to be well-supported. We further explored relationships between habitat structure and quokka activity by summing model weights for all models containing each predictor variable. The predictor variable with the largest weight was considered the most important. We considered all variables with summed model weights >0.4 to be well-supported (Converse, et al. 2006). To account for model overfitting, we also ran analyses on each variable separately, but these showed similar results to the best subset modelling analyses (see Appendix D), so we did not present those data.

Results

We recorded quokkas at six sites, which were all four riparian forest sites and two restored forest sites. Variability in overall vegetation structure was similar between riparian, revegetated and mid-slope forest (Fig. 2), although some vegetation variables were more variable in some forest types than others (Fig. 3; Fig. F1).

Overall vegetation structure differed significantly between forest types (Fig. 2), with riparian forest differing significantly from both revegetated and mid-slope forest and the latter two not differing significantly from each other (Table T2). Neither cover >2 m, litter cover, FWD nor CWD differed significantly between forest types (Table T2; Fig. F1). Cover from 1 to 2 m differed significantly between forest types (Fig. 3) with riparian forest having greater cover than mid-slope forest, although differences from revegetated forest were marginally non-significant and there was no difference between mid-slope and revegetated forest (Table T2). Cover from 0 to 1 m also differed significantly between forest types (Fig. 3) being significantly higher in riparian forest than either mid-slope or revegetated forest, but not differing significantly between the latter two (Table T2).

Quokka activity differed significantly between forest types ($\chi_2^2 = 6.31$, P = 0.043: Figure 3), with significantly more activity in riparian than mid-slope forest (P = 0.021) and activity in revegetated forest being intermediate (P = 0.186 and 0.309 with riparian and mid-slope forest respectively).

All well-supported habitat models for quokka activity included estimates of cover in the three strata (Table 1), although summed variable weights showed that cover from 1 to 2 m ($\Sigma\omega_i$ = 0.60, P = 0.028) was the only well-supported variable. Cover >2 and from 0 to 1 m were not

well-supported ($\Sigma\omega_i = 0.38$, P = 0.108 and $\Sigma\omega_i = 0.36$, P = 0.076 respectively) and neither were the remaining variables (CWD: $\Sigma\omega_i = 0.11$, P = 0.433; FWD: $\Sigma\omega_i = 0.07$, P = 0.945; litter cover: $\Sigma\omega_i = 0.07$, P = 0.946). Revegetated forest sites where quokkas were recorded had less cover >2 m (40.3 ± 8.5 vs 64.0 ± 6.5 %) and more cover between 0 to 1 and 1 to 2 m (33.0 ± 0.2 vs 24.2 ± 6.1 and 19.1 ± 3.6 vs 5.4 ± 4.3 % respectively) than those where quokkas were not recorded, although differences were not statistically significantly ($t_2 = 2.22$, 1.44 and 2.45, P = 0.156, 0.287 and 0.134 respectively), probably due to our small sample sizes.

Well-supported landscape models for quokka activity included area of riparian and revegetated forest within 100 m (Table 1) but only area of riparian forest within 100 m ($\Sigma\omega_i = 0.90$, P = 0.049) was a well-supported variable, with area of revegetated forest within 100 ($\Sigma\omega_i = 0.32$, P = 0.599) and 250 m ($\Sigma\omega_i = 0.16$, P = 0.767) and area of riparian forest within 250 m ($\Sigma\omega_i = 0.14$, P = 0.827) not well-supported. Camera locations in revegetated forest where quokkas were recorded were no closer to pit edges than those where they were not recorded ($t_2 = -0.87$, P = 0.496), but pits where quokkas were recorded were significantly closer to riparian forest than pits where they were not (38.5 ± 3.5 vs 72.0 ± 2.0 m: $t_2 = 8.31$, P = 0.027).

Discussion

Our study confirmed revegetated forest can provide new habitat for quokkas, as quokkas were present in two revegetated sites sampled yet absent from the mid-slope forest that was present in these areas pre-mining. As far as we are aware, our study is the first to show the occupancy of revegetated forests by a threatened species that does not naturally occur in habitat cleared for mining. The presence of quokkas in all riparian forest sites, and their absence from all mid-slope forest sites, confirmed previously recorded patterns of habitat use in the northern jarrah forest. Previous studies have shown that quokka home ranges are located almost entirely in riparian forest with mid-slope forest rarely used and, even then, only immediately adjacent to riparian forest (Hayward, et al. 2004; Hayward, et al. 2005b). Quokkas were recorded in one revegetated site on only a single night, suggesting that individual may have been dispersing, or that the site was visited by an individual whose home range was located primarily in adjacent riparian forest. However, the other revegetated site where quokkas were present recorded quokkas on 18 nights, suggesting this site formed part of the home range of at least one quokka. It also suggests that some revegetated sites are,

possibly, suitable habitat, although studies examining whether revegetation provides all the resources required by quokkas, plus long-term survivorship and demography, would be required to accurately assess habitat suitability. While we cannot exclude the possibility that the bait placed in front of our cameras attracted quokkas from riparian forest into unsuitable vegetation in revegetated forest, we consider it unlikely for four reasons. Firstly, if bait attracted quokkas into unsuitable habitat then we would have expected to record quokkas in mid-slope forest, which is unsuitable habitat (Hayward, et al. 2004; Hayward, et al. 2005b), but we did not. Secondly, the quokka recorded on the single night in one revegetated site was recorded 5 nights after the last bait replenishment and the quokka recorded frequently in the revegetated site was recorded 3, 4 and 7 nights after the last bait replenishment. As a pilot study demonstrated that negligible bait remained after 48 hours, probably due to rainfall, ants and other mammals, it was highly unlikely any bait remained to attract quokkas at this time. Thirdly, one revegetated site recorded quokkas on more nights than any riparian forest site, yet these latter sites are known to be suitable habitat. It is extremely unlikely that bait would attract quokkas into unsuitable habitat more than it would attract them within suitable habitat, particularly given the intense predation pressure on quokkas (Hayward, et al. 2005a; Hayward, et al. 2004; Hayward, et al. 2005b). Lastly, there is a probable ecological mechanism explaining why some revegetated sites might provide suitable habitat for quokkas. Previous research has shown that quokkas prefer high vegetation cover close to ground level to protect them from predation (Hayward, et al. 2005a; Hayward, et al. 2005b). Revegetated sites where quokkas were recorded had greater cover below 2 m (19.1 ± 3.6 and $33.0 \pm 0.2\%$ between 1 to 2 and 0 to 1 m respectively) than sites where quokkas were not recorded (5.4 ± 4.3 and $24.2 \pm 6.1\%$ between 1 to 2 and 0 to 1 m respectively). These levels of cover below 2 m in revegetated sites where quokkas were recorded are greater than levels of cover in mid-slope forest, potentially affording quokkas the protection from predation they require. Taken in combination, these four lines of reasoning make it highly unlikely the bait attracted quokkas into revegetated forests that were unsuitable habitat. The most parsimonious explanation is that some revegetated sites provide, at least, suitable foraging habitat where cover is dense enough to greatly reduce predation risk. However, whether quokkas spend the day in revegetated forests, or retreat to riparian forests during the day, is unclear and, as stated previously, long-term studies would be required to more accurately assess the habitat suitability of revegetated forests both year round and inter-annually.

So what factors influenced utilisation of revegetated forests by quokkas? Our models suggested that cover in the three strata may influence habitat selection by quokkas. Although cover >2 m varied randomly between sites where quokkas were recorded and not recorded, all sites where quokkas were recorded had estimated cover (±5%) of >32.5% between 0 to 1 m and >13% between 1 to 2 m, whereas all sites where quokkas were not recorded had estimated cover levels below those values. Predation is a major influence determining quokka habitat use (Hayward, et al. 2005b; Hayward, et al. 2003) and, hence, it seems likely that sites with more cover below 2 m have sufficient cover to protect quokkas from predation. However, whether there is a threshold cover value below which habitat becomes unsuitable for quokkas, and whether that value is around the values identified in this study would need to be determined by further research. Our models also suggested revegetated sites where quokkas were recorded were more likely to have more riparian and revegetated forest within 100 m. However, revegetated sites where quokkas were recorded did not consistently have greater areas of revegetated forest within 100 m than sites where they were not, but were significantly closer to riparian forest, suggesting the proximity of riparian forest may influence quokka occurrence in revegetated forests. The decline of quokkas on mainland southwestern Australia, and its preference for swamps and streamzones, is likely the result of predation pressure from red foxes (Hayward, et al. 2005b; White 1952). Therefore, it is logical that quokkas likely suffer high predation pressure when dispersing through open mid-slope forest, as supported by Hayward et al. (2005a) recording all predated quokka from mid-slope forest. This study suggests revegetated forests are more likely to be occupied if they are <45 m (Table T1) from riparian forests. Overall, our results suggest that revegetated forests are most likely to provide suitable habitat for quokkas if they have high understorey cover and are close to swamps or riparian forest, probably because this reduces predation risk. While we cannot exclude other factors influencing quokka use of revegetated forests, it cannot be their presence in adjoining riparian areas because all riparian areas adjacent to the revegetation we studied contained quokkas. We also consider it unlikely to be due to a lack of food because Water Bush, the third most favoured food plant, was the most common understorey plant species in revegetation, while Holly-leaved Mirbelia, the second most favoured food plant (Hayward 2005), was also common in revegetation (MDC, unpubl. data). Hence, we consider it likely that predation is the main factor influencing quokka use of revegetated forests.

Management options

Our study suggested that, in some revegetated sites, cover below 2 m was denser than in mid-slope forest and mimicked dense cover found in riparian forests creating habitat that enabled quokkas to evade predation. Normally, mid-slope forest grows over a bauxite-containing duricrust layer, which is lacking beneath swamps and riparian vegetation (Koch 2007), but mining removes the duricrust and effectively increases water availability for jarrah forest plants (Kew, et al. 2007) leading to denser vegetation that resembles riparian forest. While we are not advocating that revegetation should always, or primarily, be targeted towards threatened species, revegetation can sometimes overshoot or undershoot targets, resulting in a structure that differs from the reference community. Our study suggests that managing a small proportion of these sites to provide habitat for threatened species can add value to revegetation outcomes that might not exactly match the sites' original revegetation goals, even if resulting habitats were different from reference habitats (see e.g. Pethiyagoda and Manamendra-Arachchi 2012). This would also increase both between and within-site heterogeneity in revegetated sites, which is often reduced compared to reference ecosystems (Brooks, et al. 2005). For example, our study suggested revegetated sites <45 m from riparian forest were most likely to be occupied by quokkas and, as <5% of revegetated mine-pits are located this close to riparian forest, this small proportion of sites could be managed to provide habitat for quokkas and increase between-site heterogeneity. Further research should investigate the quality of habitat provided by revegetated forests for quokkas and whether these forests are habitat 'sinks' (Pulliam 1988). However, regardless of habitat quality in revegetated forests, the greatest benefit to the long-term population persistence of quokkas would be obtained by appropriately revegetating areas that increased connectivity between natural populations and helped restore metapopulation structure (Hayward, et al. 2003).

Conclusion

Our study demonstrated that post-mining revegetation can inadvertently provide new and unexpected habitat for a threatened species. Under some scenarios, it may be important to provided habitat for threatened species, even on sites where the faunal species did not originally occur, and tailored revegetation could achieve this and, consequently, increase both between-site heterogeneity and biodiversity across the landscape (Bell, et al. 1997). Revegetation is becoming increasingly important in the fight to save global biodiversity. While this is typically best achieved by aiming to restore the reference ecosystem present pre-disturbance (Cristescu, et al. 2012; Munro, et al. 2007), considering the creation of additional

habitats for threatened species as a management option could potentially increase the ability of revegetation to help conserve biodiversity.

Acknowledgements

References

Anderson M. J., Gorley R. N. and Clarke K. R. (2008) PERMANOVA+ for Primer. Primer-E, Plymouth.

Bartle J. and Slessar G. C. (1989) Mining and rehabilitation. In: The Jarrah Forest: A Complex Mediterranean Ecosystem (eds B. Dell, J. J. Havel and N. Malajczuk) pp. 357-377. Kluwer Academic Publishers, Dordrecht.

Bell S. S., Fonseca M. S. and Motten L. B. (1997) Linking restoration and landscape ecology. Restoration Ecology **5**, 318-323.

Brooks R. P., Wardrop D. H., Cole C. A. and Campbell D. A. (2005) Are we purveyors of wetland homogeneity? A model of degradation and restoration to improve wetland mitigation performance. Ecological Engineering **24**, 331-340.

Burbidge A. A. and McKenzie N. L. (1989) Patterns in the modern decline of western Australia's vertebrate fauna: Causes and conservation implications. Biological Conservation **50**, 143-198.

Burbidge A. A., McKenzie N. L., Brennan K. E. C., Woinarski J. C. Z., Dickman C. R., Baynes A., Gordon G., Menkhorst P. W. and Robinson A. C. (2008) Conservation status and biogeography of Australia's terrestrial mammals. Australian Journal of Zoology **56**, 411-422.

Churchward H. M. and Dimmock G. M. (1989) The soils and landforms of the northern jarrah forest. In: The Jarrah Forest: A Complex Mediterranean Ecosystem (eds B. Dell, J. J. Havel and N. Malajczuk) pp. 13-21. Kluwer Academic Publishers, Dordrecht.

Clewell A. F. and Aronson J. (2006) Motivations for the restoration of ecosystems. Conservation Biology **20**, 420-428.

Converse S. J., White G. C. and Block W. M. (2006) Small mammal responses to thinning and wildfire in ponderosa pine-dominated forests of the southwestern United States. Journal of Wildlife Management **70**, 1711-1722.

Craig M. D., Stokes V. L., Fontaine J. B., Hardy G. E. S. J., Grigg A. H. and Hobbs R. J. (2015) Do state-and-transition models derived from vegetation succession also represent avian succession in restored mine pits? Ecological Applications **25**, 1790-1806.

Cristescu R. H., Frère C. and Banks P. B. (2012) A review of fauna in mine rehabilitation in Australia: Current state and future directions. Biological Conservation **149**, 60-72.

Day R. W. and Quinn G. P. (1989) Comparisons of treatments after an analysis of variance in ecology. Ecological Monographs **59**, 433-463.

Department of Environment and Conservation (2013) Quokka (Setonix brachyurus) Recovery Plan. Report No. 56, Department of Environment and Conservation, Perth.

Dundas S. J., Adams P. J. and Fleming P. A. (2014) First in, first served: Uptake of 1080 poison fox baits in south-west Western Australia. Wildlife Research **41**, 117-126.

Grant C. and Koch J. (2007) Decommissioning Western Australia's first bauxite mine: co-evolving vegetation restoration techniques and targets. Ecological Management and Restoration **8**, 92-105.

Grant C. D. (2006) State-and-transition successional model for bauxite mining rehabilitation in the jarrah forest of Western Australia. Restoration Ecology **14**, 28-37.

Havel J. J. (1975) Site-vegetalion mapping in tbe northem jarrah forest (Darling Range). I. Definition of site-vegetation types. Bulletin No. 86, Forests Department, Perth.

Havel J. J. (1989) Land use conflicts and the emergence of multiple land use. In: The Jarrah Forest: A Complex Mediterranean Ecosystem (eds B. Dell, J. J. Havel and N. Malajczuk) pp. 281-314. Kluwer Academic Publishers, Dordrecht.

Hayward M. W. (2005) Diet of the quokka (Setonix brachyurus) (Macropodidae:Marsupialia) in the northern jarrah forest of Western Australia. Wildlife Research **32**, 15-22.

Hayward M. W., De Tores P. J., Augee M. L. and Banks P. B. (2005a) Mortality and survivorship of the quokka (Setonix brachyurus) (Macropodidae : Marsupialia) in the northern jarrah forest of Western Australia. Wildlife Research **32**, 715-722.

Hayward M. W., De Tores P. J., Augee M. L., Fox B. J. and Banks P. B. (2004) Home range and movements of the quokka Setonix brachyurus (Macropodidae: Marsupialia), and its impact on the viability of the metapopulation on the Australian mainland. Journal of Zoology **263**, 219-228.

Hayward M. W., De Tores P. J. and Banks P. B. (2005b) Habitat use of the quokka, Setonix brachyurus (Macropodidae: Marsupialia), in the northern jarrah forest of Australia. Journal of Mammalogy **86**, 683-688.

Hayward M. W., De Tores P. J., Dillon M. J. and Fox B. J. (2003) Local population structure of a naturally occurring metapopulation of the quokka (Setonix brachyurus Macropodidae: Marsupialia). Biological Conservation **110**, 343-355.

Hobbs R. J. and Harris J. A. (2001) Restoration ecology: Repairing the Earth's ecosystems in the new millennium. Restoration Ecology **9**, 239-246.

Jayapal R., Qureshi Q. and Chellam R. (2009) Importance of forest structure versus floristics to composition of avian assemblages in tropical deciduous forests of Central Highlands, India. Forest Ecology and Management **257**, 2287-2295.

Kew G. A., Mengler F. C. and Gilkes R. J. (2007) Regolith strength, water retention, and implications for ripping and plant root growth in bauxite mine restoration. Restoration Ecology **15**, S54-S64.

Koch J. M. (2007) Alcoa's mining and restoration process in south Western Australia. Restoration Ecology **15**, S11-S16.

McGregor R. A., Stokes V. L. and Craig M. D. (2014) Does forest restoration in fragmented landscapes provide habitat for a wide-ranging carnivore? Animal Conservation **17**, 467-475.

Mossman H. L., Brown M. J. H., Davy A. J. and Grant A. (2012) Constraints on salt marsh development following managed coastal realignment: Dispersal limitation or environmental tolerance? Restoration Ecology **20**, 65-75.

Munro N. T., Lindenmayer D. and Fischer J. (2007) Faunal response to revegetation in agricultural areas of Australia: A review. Ecological Management and Restoration **8**, 199-207.

Myers N., Mittermeier R. A., Mittermeler C. G., Da Fonseca G. A. B. and Kent J. (2000) Biodiversity hotspots for conservation priorities. Nature **403**, 853-858.

Nichols O. G. and Nichols F. M. (2003) Long-term trends in faunal recolonization after bauxite mining in the jarrah forest of southwestern Australia. Restoration Ecology **11**, 261-272.

Pethiyagoda R. S. and Manamendra-Arachchi K. (2012) Endangered anurans in a novel forest in the highlands of Sri Lanka. Wildlife Research **39**, 641-648.

Prober S. M. and Smith F. P. (2009) Enhancing biodiversity persistence in intensively used agricultural landscapes: A synthesis of 30 years of research in the Western Australian wheatbelt. Agriculture, Ecosystems and Environment **132**, 173-191.

Pulliam H. R. (1988) Sources, sinks and population regulation. American Naturalist **132**, 652-661.

Schulz M., Wilks G. and Broome L. (2012) Occupancy of spoil dumps by the Mountain Pygmy-possum Burramys parvus in Kosciuszko National Park. Ecological Management and Restoration **13**, 290-296.

Triska M. D., Craig M. D., Stokes V. L., Pech R. P. and Hobbs R. J. (2016) The relative influence of in situ and neighborhood factors on reptile recolonization in post-mining restoration sites. Restoration Ecology **24**, 517-527.

Welsh A. H., Lindenmayer D. B. and Donnelly C. F. (2013) Fitting and interpreting occupancy models. PLoS ONE **8**, e52015.

White S. R. (1952) The occurence of the Quokka in the south-west. Western Australian Naturalist **3**, 101-103.

FIGURE CAPTIONS

Figure headings

Fig. 1. Location of motion-sensitive cameras in each of the three forest types within the Huntly landscape. Also shown is the landscape context of riparian (streams), mid-slope (unmined) and revegetated (restoration) forest.

Fig. 2. Principle Coordinates Analysis (PCO) of overall vegetation structure in the 12 sampling sites showing streamzone vegetation (○), mid-slope forest (□) and revegetated forest (Δ). Sites where quokkas were recorded are shown with solid symbols whereas sites where quokkas were not are shown as open symbols. PCO1 (x-axis) and PCO2 (y-axis) explained 33.9 and 24.6% of the variation respectively.

Fig. 3. Relative quokka activity and vegetation cover between 0 to 1 and 1 to 2 m in the three forest types. Letters above standard errors bars indicate significantly different means.

Supporting Information

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

Table T1. Details of the 12 sites where motion-sensitive cameras were deployed to detect quokkas. Distances to streamzone and pit edge were measured from the camera locations.

Table T2. Results of statistical tests on differences in overall vegetation structure and individual structural variables between forest types and the results of the post-hoc tests examining differences between forest types.

Figure F1. Mean ($\pm$SE) of the four habitat variables not shown in Fig. 3, to show the degree of variability of each variable with each forest type.

Appendix A. Information on the ecology and conservation status of the Quokka (Setonix brachyurus)

Appendix B. Detailed description of study area and associated vegetation types.

Appendix C. Results of occupancy modelling accounting for differing detectability between forest types.

Appendix D. Results of univariate analyses to verify lack of bias on results from overfitting models.

1   **Table 1.** Results of best subset modelling showing the three well-supported habitat models

2   and two well-supported landscape models that best explained quokka activity. The AIC_c of

3   each model, its weight (ω_i) and the relationship with quokka activity (+ve or –ve) are also
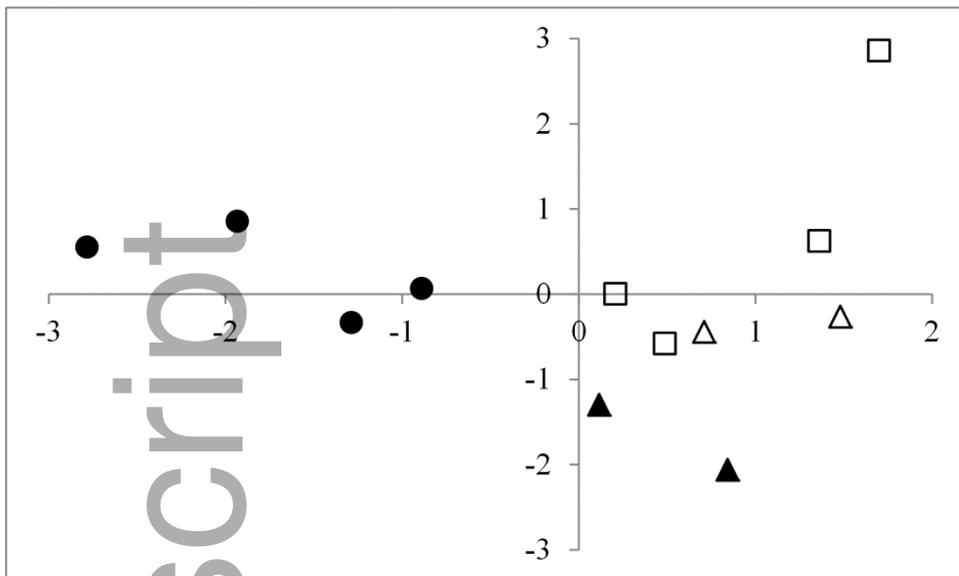
4   shown.

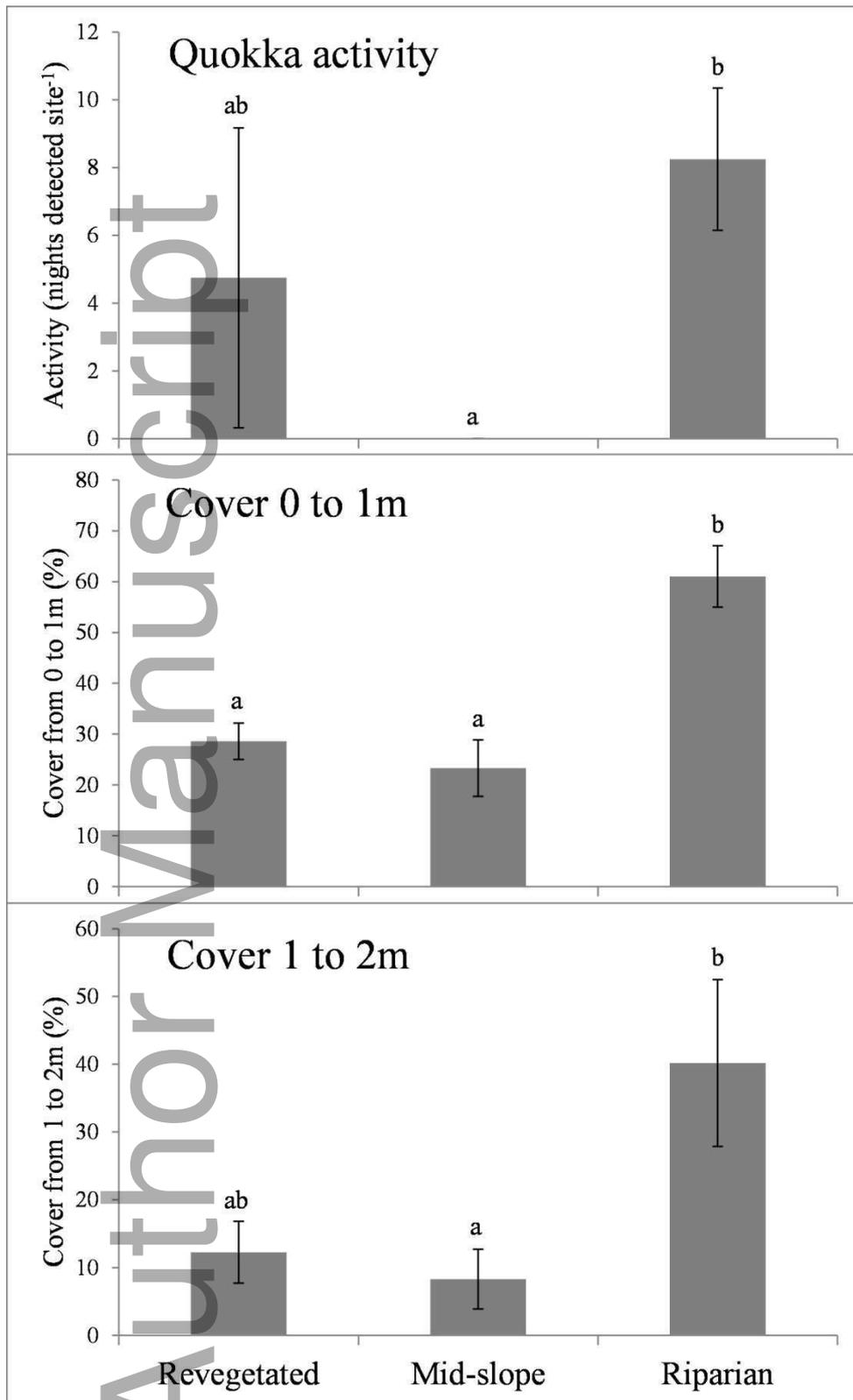| Predictor variable(s)	AIC_c	ΔAIC_c	ω_i
Habitat			
Cover 1 to 2m (+ve)	38.34	0.00	0.23
Cover 1 to 2m (+ve) + Cover >2m (-ve)	38.98	0.64	0.17
Cover 0 to 1m (+ve)	39.32	0.98	0.14
Landscape			
Streamzone <100m (+ve)	38.49	0.00	0.40
Streamzone <100m (+ve) + Revegetation <100m (+ve)	39.31	0.82	0.26

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Fig. 1. Location of motion-sensitive cameras in each of the three forest types within the Huntly landscape. Also shown is the landscape context of riparian (streams), mid-slope (unmined) and revegetated (restoration) forest.

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Fig. 2. Principle Coordinates Analysis (PCO) of overall vegetation structure in the 12

sampling sites showing streamzone vegetation (○), mid-slope forest (□) and revegetated

forest (Δ). Sites where quokkas were recorded are shown with solid symbols whereas sites

where quokkas were not are shown as open symbols. PCO1 (*x*-axis) and PCO2 (*y*-axis)
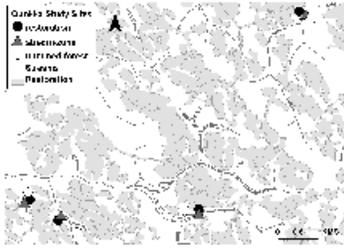
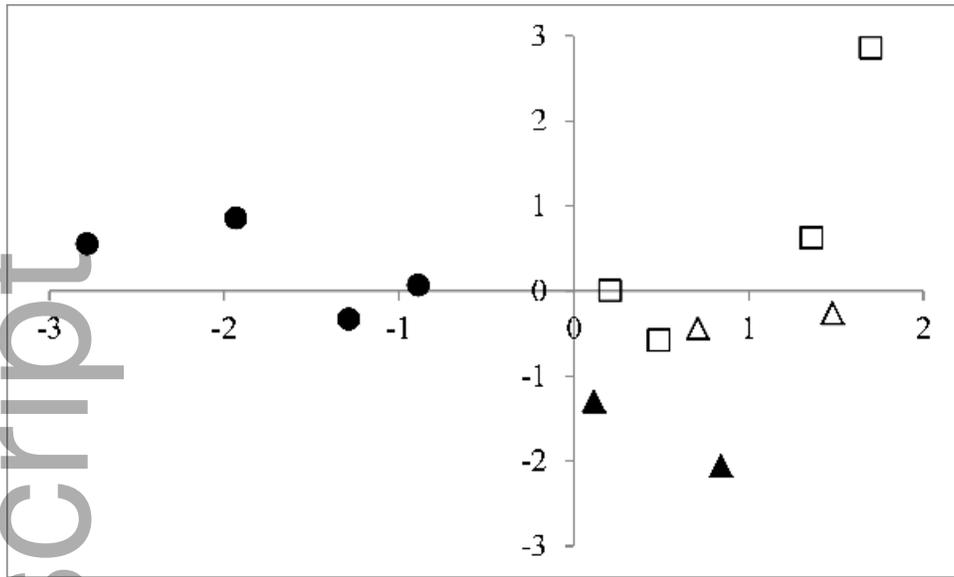explained 33.9 and 24.6% of the variation respectively.

483 **Fig. 3.** Relative quokka activity and vegetation cover between 0 to 1 and 1 to 2 m in the three

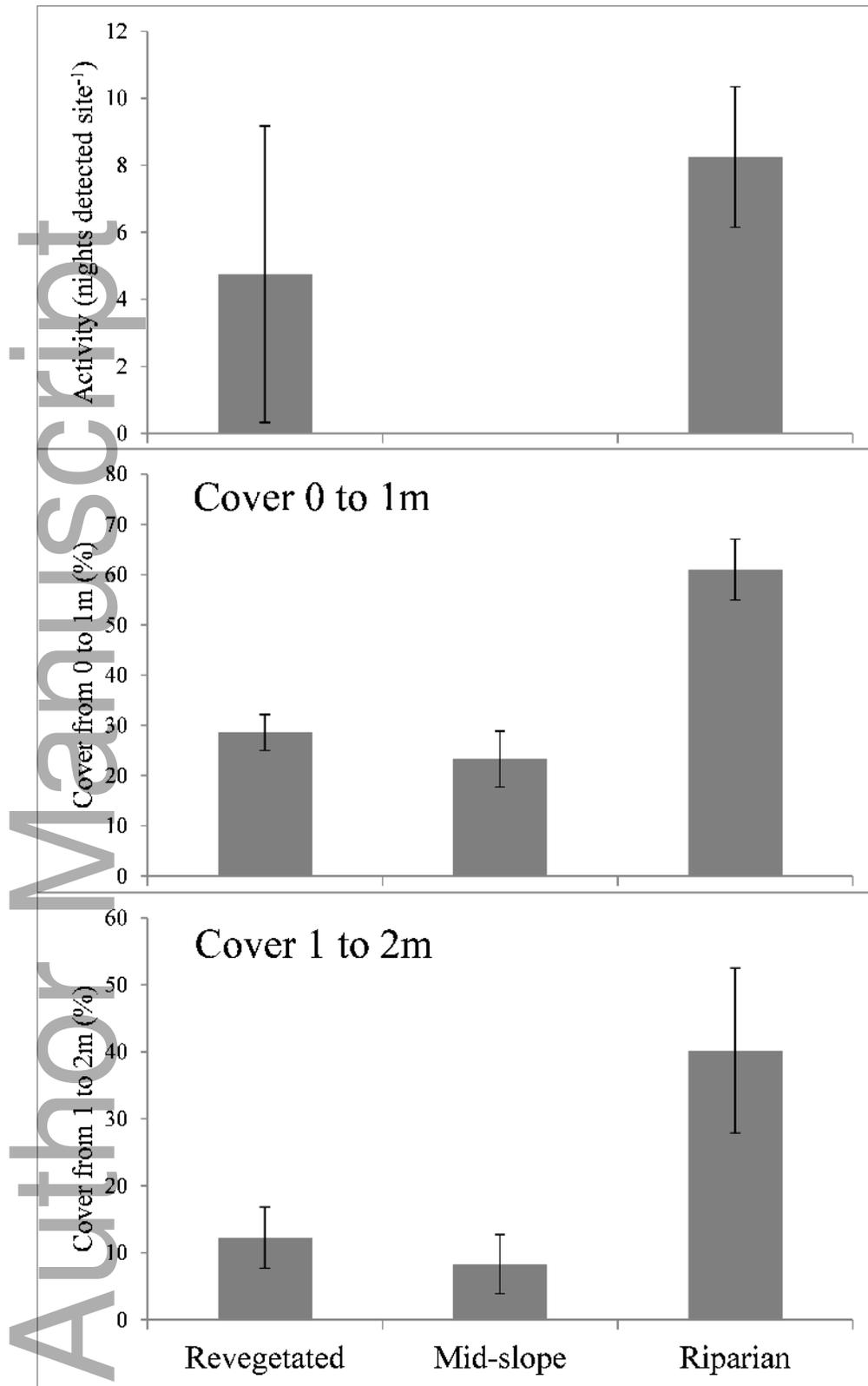484 forest types. Letters above standard errors bars indicate significantly different means.

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emr_12258_f2.tif

Activity (nights detected site⁻¹)

Cover from 0 to 1m (%)

Cover from 1 to 2m (%)

Cover 0 to 1m

Cover 1 to 2m

Revegetated   Mid-slope   Riparian

emr_12258_f3.tif