Review of reintroductions to Ngambaa Nature Reserve

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Executive Summary

This report reviews a list of species that have been identified as potential candidates for reintroduction into sites in Ngambaa Nature Reserve, New South Wales. It also suggests additional species that had their historic ranges in these sites, but which may no longer persist there. For the indicative list of species, we estimated the population size that could be supported, as well as suggested measures that could increase these population sizes. The following species were considered likely to have included the Ngambaa reintroduction site within their historic ranges based on specimen records, sightings, distribution modelling, reports or other accounts and knowledge of their ecological requirements (see Table 1).

Table 1. Species being considered in this review and their conservation status. * is the status assigned to Australian mammals in the action plan (Woinarski et al. 2012). Blank = not listed, N/A = not applicable, LC = least concern, NT = near threatened, VU = vulnerable to extinction, EN = endangered, EX = extinct.

No.	Common name	Scientific name	EPBC Act	BC Act	MAP*
Suitable for reintroduction (locally extinct)					
1	Eastern bettong	*Bettongia gaimardi*	EX	EX	EX
2	Eastern quoll	*Dasyurus viverrinus*	EN	EN	EN
3	Bush stone-curlew	*Burhinus grallarius*		EN	N/A
4	Eastern bristlebird	*Dasyornis brachypterus*	EN	EN	N/A
Suitable for reintroduction (not likely already present)					
5	Rufous bettong	*Aepyprymnus rufescens*		VU	LC
6	Eastern pygmy-possum	*Cercartetus nanus*		VU	LC
7	Parma wallaby	*Macropus parma*		VU	NT
8	Northern long-nosed potoroo	*Potoroo tridactylus* ssp. *tridactylus*	VU	VU	VU
9	Eastern chestnut mouse	*Pseudomys gracilicaudatus*		VU	LC
10	New Holland mouse	*Pseudomys novaehollandiae*		VU	VU
11	Red-legged pademelon	*Thylogale stigmatica*		VU	LC
Species that are of conservation concern and occur at the site that may benefit					
12	Brush-tailed phascogale	*Phascogale tapoatafa*		VU	NT
13	Glossy black cockatoo	*Calyptorhynchus lathami*		VU	N/A
14	Varied sittella	*Daphoenositta chrysoptera*		VU	N/A
15	Square-tailed kite	*Lophoictinia isura*		VU	N/A
16	Barking owl	*Ninox connivens*		VU	N/A
17	Powerful owl	*Ninox strenua*		VU	N/A
18	Olive whistler	*Pachycephala olivacea*		VU	N/A
19	Masked owl	*Tyto novaehollandiae*		VU	N/A
20	Greater sooty owl	*Tyto tenebrosica*		VU	N/A
21	Spotted-tailed quoll	*Dasyurus maculatus*	EN		VU
22	Northern brown bandicoot	*Isoodon macrourus*			LC
23	Stephen’s banded snake	*Hoplocephalus stephensii*		VU	N/A
24	Green-thighed frog	*Litoria brevipalmata*		VU	N/A
25	Giant barred frog	*Mixophyes iteratus*	EN	EN	N/A
26	Yellow-bellied glider	*Petaurus australis*		VU	NT
27	Koala	*Phascolarctos cinereus*	VU	VU	VU
Potentially suitable but uncertainty exists					
28	Rufous scrub-bird	*Atrichornis rufescens*		VU	N/A
29	Black-striped wallaby	*Macropus dorsalis*		EN	LC
30	Hastings River mouse	*Pseudomys oralis*	EN	EN	VU
Not suitable for reintroduction					
31	Common planigale	*Planigale maculata*		VU	LC
32	Tasmanian Bettong	*Thylogale billardierii*			LC

Population estimates for each of the species on the indicative list were made based on population density estimates for these species from other areas. The resulting estimates show considerable variation between species and are subject to numerous caveats, including the extent of management that might be available to augment habitat quality if reintroductions were to proceed. There are seven species that require preliminary monitoring to gain evidence whether they occupy the site. This underscores the importance of preliminary baseline monitoring before conservation actions take place.

Background

The National Parks and Wildlife Service (NPWS) is developing plans to establish several feral predator free areas across New South Wales (see Fig. 1). Ngambaa Nature Reserve (Ngambaa) is one of the sites proposed for this action. This area is situated in the mid-north coast of New South Wales (NSW). The project will seek to reintroduce threatened and declining animal species, and restore ecosystem function and processes. This project extends the existing safe haven projects that NPWS has initiated at Mallee Cliffs and Pilliga National Parks to introduce more representation into the safe haven estate addressing earlier concerns of the situation of these areas (Ringma et al. 2019).

Figure 1. Preferred area for siting of the introduced predator-free area and plant community types.

Ngambaa is located between Port Macquarie and Coffs Harbour, and is about 20 km inland from South West Rocks (Fig. 1). The project brief involving this report from NPWS was to review an indicative list of 32 species identified by NPWS, that are mainly of conservation concern and could be reintroduced to Ngambaa, along with advice on whether Ngambaa lies within the historic distribution/range for that species, and determine whether any other species had its historic range within the site but is no longer likely to be present. For each species identified as historically occurring at the site, we were to estimate the likely population size (or population range) for that species based on the available habitat, and identifying any habitat augmentation that might enable a greater population size to be supported at Ngambaa.

Methods

Site details

The proposed fenced area at Ngambaa contains 13 plant community types, composed of five vegetation classes consisting of three vegetation forms (Table 2). Ngambaa lies within a contiguous tract of bushland of about 50,700 hectares. The area proposed for the predator-proof fence site is 2574.6 hectares, and the vegetation communities within the fenced area are shown in Fig. 1. The reserve has a history of logging and weed invasion so there are some areas that were historically disturbed (New South Wales National Parks and Wildlife Service 2004).

Table 2. Vegetation summary.

Plant Community Type	Vegetation Class	Size (ha)
Dry Blackbutt	Dry Sclerophyll	690
Brushbox	Wet Sclerophyll	634
Grey Gum – Grey Ironbark – White Mahogany – Spotted Gum (sub type A)	Semi-mesic grassy forest	1398
Grey Gum – Grey Ironbark – White Mahogany (sub type B)	Semi-mesic grassy forest	34
Rainforest	Rainforest	2

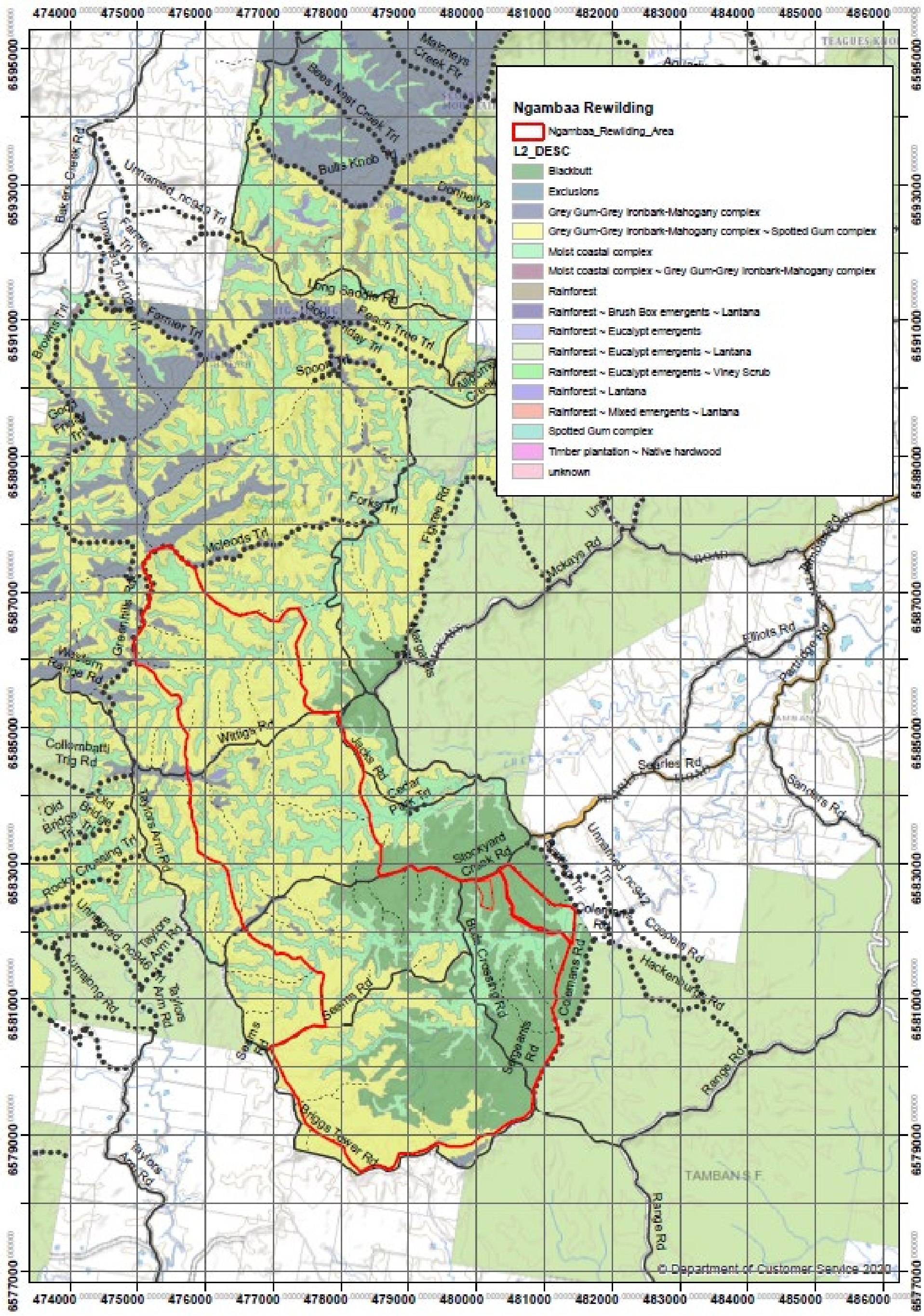

Figure 1. Preferred area for siting of the introduced predator-free area and plant community types.

Determining presence and reintroduction suitability

Several sources were used to determine if species were likely to occur or would be suitable candidates for reintroduction into Ngambaa, including primary literature, historic records and the Atlas of Living Australia (ALA) database. Species distribution modelling using ALA records was carried out in the ALA's Spatial Portal to improve estimates of the probability that locally extinct species could occur in the fenced area. The variables used in the species distribution models included the following (along with associate variable code name): enhanced vegetation index (e11079), fraction cover – bare soil (e11072), fractional cover – persistent vegetation (e11073), normalized difference vegetation index (e11078), net primary productivity mean (e1676), soil pedality (e1841), soil depth (e1816), pH – soil (e1663), carbon – organic (e1665), aspect (e1680), elevation (e1674) and topographic slope (%). We only included verified and accurate records as identified by the record keeping database of ALA. The vegetation communities of the proposed predator-proof fence site were also used as a guide to inform the potential occurrence of species that had no recent records within the area and were also used as an addition guide to determine the suitably of reintroductions of locally extinct species.

Population size calculations

The literature was reviewed for density estimates for each species in nearby populations or within comparable habitat. Calculations of population size were made by multiplying these density estimates by the available area of suitable habitat at Ngambaa. Upper and lower density estimates were used to calculate upper and lower estimates of population size.

Species review

(1) Eastern bettong (*Bettongia gaimardi*)

The nominate form of the eastern bettong disappeared from the mainland of Australia at the beginning of the 20th century (Short 1998). An analysis of bounty's paid for agricultural pest control showed that the eastern bettong disappeared from the northern parts of its range, around Ngambaa, by about 1915 and this coincided with the northwards extension of the introduced European red fox (*Vulpes vulpes*) (Short 1998). There are no nearby records of eastern bettongs, the closest are in Pines Forest, Qld, which lies 369 km north of the site and in Campbelltown, Sydney, which is 392 km south.

The Tasmanian subspecies of the Eastern Bettong still persists and is widespread and patchily common in central and eastern parts of Tasmania (Woinarski et al. 2014). Combining the information from Woinarski et al. (2014) and the species distribution model (see Fig. 3), it appears that the mainland eastern bettong occupied coastal and sub-coastal localities along the east coast of New South Wales and into Victoria, although the paucity of reliable records and coarseness of the mapping preclude detailed insight into which habitats this subspecies preferred. Using historical records and inferences drawn from observations of the Tasmanian form, the mainland eastern bettong occupied woodland and open forest, and occurred in north-east and south-east regions of New South Wales (Wakefield 1967, Lunney et al. 2000). In Tasmania, the bulk of their diet consists of hypogeous fungi (Taylor 1992), however they also consume plant material and invertebrates (Taylor 1992). However, a reintroduced population occurring in Mulligans Flat consumes more plant material than previously recognised (A. Manning, *unpublish. data.*).

The species distribution model produced by Haouchar et al. (2016) suggested previous occupancy of this species within Ngambaa. Based on the habitat and estimated pre-European distribution, it is likely that this species occurred within or near Ngambaa. Due to recent success of reintroduction of this species into Mulligans Flat (Batson et al. 2016), and their adaptibilty of persist in this reserve, it is likely that reintroduction of this species into the Ngambaa predator-proof fence site will be successful.

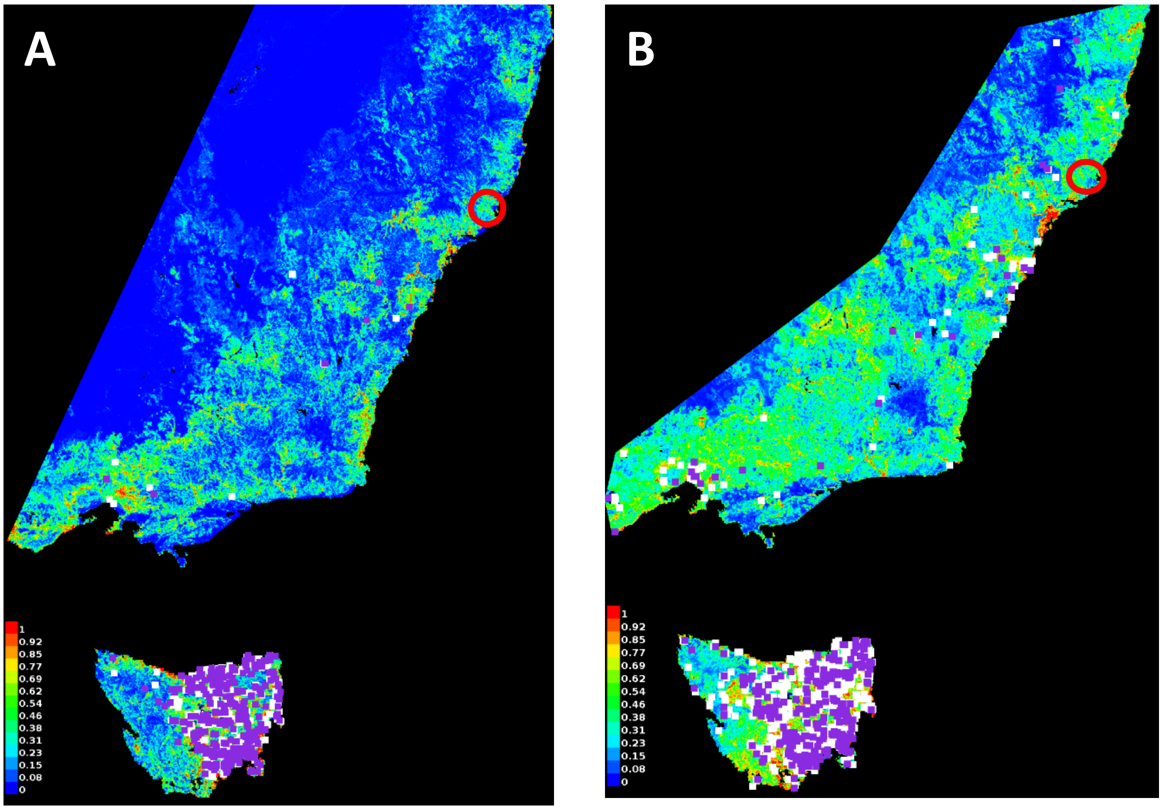

Figure 2. Species distribution models created using the Atlas of Living Australia spatial portal. 1A: eastern bettong, 1B: eastern quoll. Red circle indicates the area of the proposed predator-proof fence.

(2) Eastern quoll (*Dasyurus viverinus*)

This species was formerly widespread in eastern New South Wales but suffered sharp declines during the late 1800s and early 1900s (Peacock and Abbott 2014). Most populations have gone extinct, however there is some evidence of persistence in some populations inhabiting the Great Dividing Range dating to as late as 2006 (Frankham et al. 2016, Hope et al. 2019). Recent follow-up surveys were conducted in an attempt to validate anecdotal records but were not successful in detecting any new records of the eastern quoll (Hope et al. 2021). It is likely that the eastern quoll is extinct on the mainland. The decline is hypothesised to be a result of non-specific disease such as Canine Distemper Virus and predation-induced declines caused by the European red fox (Peacock and Abbott 2014, Woinarski et al. 2014). The closest confirmed historic records to Ngambaa occur about 17 km south (Australian Museum catalogue number: M.1477).

This species used to be widespread in eastern New South Wales, occupying a variety of woodland habitats in coastal and sub-coastal areas (Woinarski et al. 2014). It still occurs in Tasmania where it occupies open woodland, dry eucalypt forest and grasslands (Jones and Barmuta 2000). This quoll has been released into unfenced areas at Booderee NP on the south coast of New South Wales in the last three years where breeding and short-term persistence was achieved (Robinson et al. 2020b). These quolls had a varied diet that was similar to Tasmanian eastern quolls, consisting mainly of macropods, birds, invertebrates and some herpetofauna and plant matter (Robinson et al. 2020a).

Recent species distribution models have been produced for the eastern quoll to examine how their predicted distribution may change under different climate change scenarios (Barlow et al. 2021). These models indicated a low suitability (10 – 20%) in Ngambaa under all scenarios. However, these models were constructed only with climate variables that would be biased towards Tasmanian eastern quoll records, which are more numerous than mainland records. A previous study has demonstrated that only using climate data can lead to under-predictions in species distribution models (Zangiabadi et al. 2021). The historically low numbers of records on the mainland are likely due to disease and red fox incursion rather than changes in weather patterns. This would favour the models predicting the optimal weather conditions for quolls to be much cooler than what would have been true in the pre-European distribution of the eastern quoll. When repeating the species distribution model with environmental variables but without climate covariates, we found that the habitat suitability for eastern quolls in Ngambaa ranged between 40% - 70% (see Figure 2A). In the absence of the agent of their decline (foxes) that are likely to constrain habitat use (Caughley 1994), we suspect this suitability would be even higher in the Ngambaa safe haven.

Given this information of nearby records, its ability to exploit a wide variety of habitats and food resources, and distribution modelling using the ALA spatial portal, we believe that Ngambaa was within the historic range of *D. viverrinus*. Combined with the knowledge that this species has been successfully reintroduced into dry sclerophyll forests of Mulligans Flat (Wilson et al. 2020), this species makes a viable candidate for reintroduction.

(3) Bush stone-curlew (*Burhinus grallarius*)

The bush stone-curlew has declined across southern Australia and are now considered endangered in Victoria, New South Wales and South Australia (Garnett et al. 2011). Only some population strongholds remain in this region (Price et al. 2018). Sharp declines in bush stone-curlews are occurring in Brisbane populations despite originally being considered stable in urban parks (Joyce et al. 2018, Fulton et al. 2020). The main cause of decline is land use change, particularly related to agricultural intensification (Johnson et al. 1993). The species spends much time on the ground, and so is very likely susceptible to introduced predators (Carter 2010). However, vehicle collisions are another source of mortality (Price et al. 2018). There are three records dating from the 1990s – 2000s that are within 14 km of the site. More recent records are more distant, for example, there is a record from Valla Beach, which is 36.8 km north of the site, and consistent records noting breeding related behaviours as late as 2015 at Maria River Road adjacent to Limeburners Creek National Park (NP), which is 44.6 km south.

Bush-stone curlews can occur in a diverse range of habitats, including open woodland, urban and peri-urban environments (e.g., parks and golf courses), saltmarsh, sparse arid shrubland, mangroves, grasslands with belts of timber, coastal she-oak forest and partially cleared farmland (Gates and Paton 2005, Cannard and Milton 2012, Murialdo et al. 2015). Bush stone-curlews have two key habitat needs: roosting and foraging, which can overlap but are often distinct. They are often associated close to permanent water sources (Gates and Paton 2005, Cannard and Milton 2012). There is some evidence that they prefer sites with high coverages of leaf litter (Cannard and Milton 2012). In addition, there was an association between bush stone-curlews and the presence of grasses at 5-15 cm tall and shrub cover 10-30% (Cannard and Milton 2012). They have a generalist diet consisting of invertebrates, vegetation, seeds, small mammals and eggs (Marchant et al. 2006).

Due to the distance of recent records, it seems unlikely that this species currently occurs within the site. However, dry sclerophyll sections of the reserve may provide appropriate habitat. With predator proofing, the conditions are likely to be favourable for recolonization of this bird. Their dispersive tendencies may enable natural colonisation but it is unlikely because (predicted) former panmitic population structure are now broken into severely dislocated populations (A. Manning, *pers. comm.*). Translocation of adults is a viable option (Price et al. 2018), and eight translocations of this species has taken place. The reintroduction attempt to Scotia Sanctuary failed when all founder birds dispersed, however site fidelity of introduced bush stone-curlews can be enhanced by clipping the wings of introduced birds so that they remain within the site for a year (A. Manning, *pers. comm.*).

(4) Eastern bristlebird (*Dasyornis brachypterus*)

The distribution of the eastern bristlebird previously extended up the coast of Australia from far east Victoria to south-east Queensland, however it has contracted to three disjunct areas (Baker 1997). The three main populations occur at (1) southern Queensland/northern New South Wales, (2) Barren Grounds Nature Reserve (NR), Budderoo NR, the Woronora Plateau, Jervis Bay NP, Booderee NP and Beecroft Peninsula and (3) Nadgee NR and Croajingalong NP close to the New South Wales/Victorian border. The main threats attributed to the decline of eastern bristlebirds are fire, habitat removal and predation from introduced predators (Lindenmayer et al. 2009). The northern population once extended as far south as at least Dorrigo. The closest record to Ngambaa lies ~30 km north-east and is from 1983. There are numerous records from Dorrigo and surrounding areas that date back to between the 1890s-1940s.

The habitat between the central and southern populations are different to the northern population. The habitat for the former includes dense, low vegetation including wet and dry heath, and open woodland with a heathy understorey (Baker 1977). In northern New South Wales, the primary habitat the birds use is open forest with dense tussocky grass understorey and sparse mid-storey near the rainforest ecotone (Baker 1997). This species depends on cover as it is mainly ground dwelling and hence requires refuge from predators (Baker 2000).

It is unlikely that eastern bristlebirds presently occur within Ngambaa as bird records are more often reported compared to other groups due to the abundance and passion of twitchers. This species has been translocated in several instances in the past and extensive accounts of reasons for success, failures and costs have been documented (Baker et al. 2012). There is strong genetic differentiation within the species, so the closest nearby populations will need to be targeted as founders (Roberts et al. 2011). This would most likely require capture and transportation of individuals from the northern population.

(5) Rufous bettong (*Aepyprymnus rufescens*)

Severe range contractions of the rufous bettong have occurred such that now this species exists in small, isolated populations in northern New South Wales (Schlager 1982, Short 1998). Threats associated with declines are interchangeable for other rat-kangaroos and include harvesting for pest-management by early Europeans on a massive scale, and the invasion of the European red fox and cats *Felis catus* (Short 1998). Recent records within the last decade occur close to Ngambaa. A cluster of records occurs 14.4 km north-west in Thumb Creek and another cluster lies 36.6 km south-west in Willi Willi NP. An older record dated from 1995 lies closer still, at 6.6 km to the east.

The rufous bettong is flexible in its habitat preferences being found in wet sclerophyll forests to open woodlands with a tussock grass understorey (Schlager 1982). They require tall native grasses where they construct cone-shaped nests for a diurnal refuge (Schlager 1982). We suspect these habitat requirements would be broader in the absence of the agent of their decline (Caughley 1994). The rufous bettong has a more varied diet than the eastern bettong, and includes roots, grasses, forbs and fungi (Schlager 1982, McIlwee and Johnson 1998).

Due to the close proximity of records, presence of suitable habitat, and the diverse diet of this species, it presents a valid case for reintroduction. However, since this species is cryptic, it is plausible that it may still exist within Ngambaa. A robust monitoring protocol would be beneficial to gain evidence of the occupancy status of the rufous bettong in Ngambaa before resources are invested into a translocation program.

(6) Eastern pygmy-possum (*Cercartetus nanus*)

This possum occurs from Tasmania and up the east coast of the mainland from Victoria into south-east Queensland (Bowen and Goldingay 2000, Harris and Goldingay 2005b, Harris et al. 2007). Its population trajectory in New South Wales is unclear but there is evidence for declines associated with land-clearing (Bladon et al. 2002). There are sparse records of this species throughout northern New South Wales (Bowen and Goldingay 2000) and survey effort is needed to produce detections (Bladon et al. 2002). There are recent wildlife rehabilitation records of juveniles that were obtained near Kempsey, about 16 km south of the site. Otherwise there are sparse records that occur more distantly (>40 km) throughout the large reserves systems of the Great Dividing Range north and south of the site. Additionally, there are distant coastal records both north and south of the site.

The eastern pygmy possum is found in a range of habitats, but is restricted to mainly rainforest in the north-eastern portions of its distribution (Bowen and Goldingay 2000). They mainly feed on nectar producing plants, such as *Banksia* sp. (Bladon et al. 2002). In the north of their range, where they occur primarily in rainforests, they use epiphytic ferns, burrows and logs as nesting sites (Bladon et al. 2002). The removal of foxes and cats is likely to benefit them. They are a food source for many native predators of conservation concern, many of which are under consideration in this review, including the spotted tail-quoll, Stephen's banded snake, greater sooty owl and the Australian masked owl (Law et al. 2013).

Due to the large amount of survey effort required to detect pygmy possums in the northern part of their range, it is difficult to rule out if they are present or not at the site or nearby. Presence of nectar producing Proteaceae may indicate habitat suitability. It would be beneficial to conduct surveys before any reintroductions are attempted. Nest boxes targeting eastern pygmy possums can provide an additional means of detection while also providing supplementary habitat (Harris and Goldingay 2005a, Rueegger et al. 2013). The high risk of predation from numerous naturally occurring predators (some which may also be considered for reintroduction), may make reintroductions of the pygmy possum difficult. However, this also highlights their importance in the ecosystem. Supplementary planting of nectar producing plants and increasing coarse woody debris, if these features are lacking, may enhance the habitat for this species. Eastern pygmy possums can readily negotiate the predator-proof fencing likely to be employed at Ngambaa, and so it may naturally recolonise without further assistance.

(7) Parma wallaby (*Macropus parma*)

The parma wallaby once occurred from southern Queensland to the Bega area in the south-east of New South Wales, however it suffered declines and was considered extinct for 30 years until an introduced population was discovered in New Zealand in the 1960s (Wodzicki and Flux 1967). This spurred surveys in New South Wales that led to the rediscovery of populations within its former distribution along the Great Dividing Range (Maynes 1977). Its present range consists of isolated patches along the coast and ranges of central and northern New South Wales, from the Central Coast to the Northern Rivers. The primary threat to parma wallabies is likely to be foxes as they are on the upper edge of the critical weight range of mammals that are at most risk of extinction from introduced species (Burbidge and McKenzie 1989). Indeed, fox predation has led to the failure of all attempted reintroductions of this species (Short et al. 1992). There are no records within close proximity to Ngambaa, however there are many records spanning the last two decades from New

England NP and adjoining reserves to the north-west of the site. This is also true for Willi Willi NP and adjoining reserves to the south-west of the site.

The parma wallaby occurs in wet sclerophyll forest that contains a mosaic of thick shrubby understorey interspersed with patches of blady grass areas (Maynes 1977). This mosaic appears important as they typically feed at night in grassy patches or on the edge of thick shrubland, and during the day they use dense cover as a shelter site. Like other species affected by introduced predators (Caughley 1994), we suspect the habitat requirements of parma wallabies will relax with the removal of foxes and cats.

It is doubtful that the parma wallaby still occurs within Ngambaa NR due to their restricted distribution and the lack of records. However, it is cryptic and difficult to distinguish from similar sized macropods and hence current occupancy cannot be ruled out. A targeted monitoring program is recommended before initiating reintroduction. If they are absent from the site, reintroduction into the predator-proof fenced area is appropriate given that there is suitable habitat present and the primary sources of previous reintroduction attempt failures have been addressed (i.e. predation from foxes) (Short et al. 1992).

(8) Northern long-nosed potoroo (*Potorous tridactylus* ssp. *tridactylus*)

As with the other rat-kangaroos, this species suffered initial extensive range contractions that coincided with Australia's colonisation by introduced predators (Claridge et al. 2007). The nominate subspecies, *P. t. tridactylus* has been impacted more severely that the other subspecies (*P. t. apicalis*). There is evidence of on-going declines in some populations (Andren et al. 2018, Milledge et al. 2021). Declines are associated with habitat removal and predation from the European fox and cats (Andren et al. 2018). There is a nearby record to Ngambaa at Deep Creek, about 9.4 km west, and numerous recent records (some as late as 2020) in the surrounding reserves, including Willi Willi NP to the south-west and Thumb Creek to the north-west.

The long-nosed potoroo uses a variety of different habitats across its range, including rainforest, dry and wet sclerophyll forests, open woodland and heathlands (Claridge et al. 2007). There is evidence that they require a mosaic of multiple vegetation communities for persistence as a variety of structural components appear important for different uses (Bennett 1993, Norton et al. 2011). Areas with a thick understory is likely to provide important refuge from predators, while open areas are used for foraging (Bennett 1993, Norton et al. 2011, McHugh et al. 2019). Their diet shifts from mostly fungi and seeds in autumn and winter to arthropods and vegetation in spring and summer (Bennett and Baxter 1989, Claridge and Cork 1994).

It is likely that this species is at the site due to the habitat present at Ngambaa, the valid recent records in close proximity to the site, and the cryptic nature of rat-kangaroos. In the absence of records on-site after a dedicated survey, this would be an ideal species to reintroduce as it should respond well in introduced predator-free areas.

(9) Eastern chestnut mouse (*Pseudomys gracilicaudatus*)

This species occupies mainly a coastal distribution in the south of its range, from the Central Coast up to the Northern Rivers, where it expands from the coast to more inland regions up to Townsville in Queensland (Borsboom 1975, Menkhorst and Knight 2011). The threats of the eastern chestnut mouse are likely similar to that of the New Holland mouse. The closest record of this species to Ngambaa lies about 31 km north-east in Nambucca Heads and is dated to 2016. There are numerous recent and historic records from coastal heathland around Port Macquarie. The next closest record to the north is in Bellingen from 2018. There are no records in the reserve systems of the Great

Dividing Range nearby to Ngambaa - the closest being three records from Ellis State Forest (~78 km north).

This species can occupy a variety of habitats including woodland, heath and swamps, and the commonalities among habitat types include close proximity to water, fire prone vegetation communities and a thick grass ground cover layer (Borsboom 1975, Fox 1982). It has been found in habitats in northern New South Wales described as woodland with thick grass cover occurring on an undulating landscape with hills and gullies (Watts and Tweedie 1993). There is evidence of intraspecific competition for habitat and diet between this species and other native and introduced rodents, which can be facilitated by fire (Higgs and Fox 1993, Luo and Fox 1995). This eastern chestnut mouse has been previously considered an early successional species after fire where abundances peak before other rodents (Fox 1982). However, more recent evidence suggests that habitat including dead shrubs and rock cover are more important in determining site occupancy compared to fire succession (Pereoglou et al. 2016). The eastern chestnut mouse is considered a generalist herbivore, and appears to have a similar but more opportunistic diet in comparison to the New Holland Mouse (Luo et al. 1994).

Interpolating known records, the historical distribution of eastern chestnut mouse could have included the proposed reintroduction site. This species is relatively cryptic and can have irruptive population responses to resource fluctuations, so there may be a case for targeted monitoring to gain more evidence. It is possible that some of the grassy woodland habitat is suitable, and hence reintroduction might be valid. The success of a reintroduction is difficult to foresee given all the potential intraspecific competition and predation interactions that could eventuate with native co-occurring species. However, as for the New Holland mouse, there is the possibility that this species may colonise during post-fire vegetation succession are able to cross the fence boundary. Sourcing founders may be extremely difficult for this species (A. Manning, *pers. comm.*).

(10) New Holland mouse (*Pseudomys novaehollandiae*)

This mouse occurs in disjunct populations in Tasmania and Victoria, but extends up the eastern coast of New South Wales from Jervis Bay to Toowoomba in south-east Queensland (Menkhorst and Knight 2011). It occurs both coastally and along the Great Dividing Range. Declines in Victoria and Tasmania are severe and ongoing, which are associated with habitat degradation, predation from introduced mesopredators and possibly also genetic inbreeding and disease (Burns 2020). Its status in New South Wales has not been thoroughly assessed. It was also one of the 20 mammal species requiring urgent management interventions following the 2019-2020 black summer bushfires. The original distribution of this species is difficult to ascertain given that it is infrequently recorded and may be easily confused with other co-occurring rodents. Populations can also be irruptive and unpredictable (Wilson et al. 2005, Crowther et al. 2018). The closest records of this species occur in reserve systems of the Great Dividing Range that lie to the west and south-east of Ngambaa, in Werrikimbe NP, Oxley Rivers NP and Carrai NP.

The New Holland mouse occurs in heathland, dune woodland, open forest, and swamp habitats on sandy, rocky or loam-dominated soils (Fox 1982, Lazenby et al. 2007, Menkhorst and Knight 2011). It has been described as an early successional post-fire specialist (Fox 1982), but more recent evidence indicates it is more associated with structural habitat features rather than fire (Wilson et al. 2005). This species appears to be associated with low dense ground cover and shrubs (Wilson 1991). The New Holland mouse has a varied diet, consisting of vegetation, fungi, seeds and invertebrates (Wilson and Bradtke 1999).

Despite the lack of records of the New Holland mouse, interpolating known records leads us to conclude that it is likely that this species occurred at Ngambaa (Abicair et al. 2020). There is dry sclerophyll woodland available on-site that has been colonised successfully by New Holland mice in a reintroduction in Mulligans Flat in (Abicair et al. 2020). The presence of this species needs to be determined before introduction is considered. There have been many additional recent records of this species emerging after the 2019-2020 black summer bushfires across forests of the Great Dividing Range (B. Law, *pers. comm.*). Hence targeted surveys for this species should be conducted immediately, especially targeting northern areas of Ngambaa that burned during the fires. However, there is the possibility that this species may colonise during post-fire vegetation succession as they would be easily able to cross the fence boundary, provided a nearby population persists as dispersal capacity of this species is limited. Sourcing founders for reintroduction of this species is also likely to be a challenge (A. Manning *pers. comm.*).

(11) Red-legged pademelon (*Thylogale stigmatica*)

The red-legged pademelon has a patchy distribution along coastal and subcoastal eastern Australia from the Watagans Mountains, just south of the Hunter Valley, New South Wales, to Cape York, Queensland (Menkhorst and Knight 2011). Primary threats are habitat removal, predation from introduced mesopredators and inappropriate fire management. There are no records of this species within Ngambaa or the adjoining forests. There are records in the reserve systems to the north-west and south-west, however most of these are from two decades ago, with the most recent being in 2014.

The red-legged pademelon occurs mostly in rainforest, but can persist in wet sclerophyll forest and dry-vine forest (Menkhorst and Knight 2011). This species inhabits dense forest during the day, and moves to forage along forest edges and in open areas at night (Vernes et al. 1995). There is evidence that red-necked and red-legged pademelons exhibit spatial and temporal niche partitioning that can result in co-occurrence (Smith 2019). The red-legged pademelon has a diet consisting of fruit, seeds, leaves, grasses and hypogeous fungi (Vernes 1995, Vernes and Trappe 2007).

It is possible that this species may have been overlooked at Ngambaa NR, due to its cryptic nature and similarity with the red-necked pademelon. Targeted monitoring may be needed to gain additional evidence. If red-legged pademelons are truly absent, they would make a good candidate for reintroduction due to their small home-range size and ability to niche partition with the red-necked pademelon, which is likely present. Additionally, they occur in similar low-lying rainforest habitats, such as in the remnants of the Big Scrub Rainforest (C Beranek, *pers. obs.*).

(12) Brush-tailed phascogale (*Phascogale tapoatafa*)

The brush-tailed phascogale has a patchy distribution in all states of Australia excluding Tasmania, but the nominate form, *P. t. tapoatafa*, is the only subspecies that occurs in New South Wales (Menkhorst and Knight 2011). In its New South Wales distribution, it resides along either side of the Great Dividing Range. Habitat clearance and predation from the introduced European red fox has likely contributed to the declines observed in this species (Menkhorst and Knight 2011). There are seven records within 5 km of the proposed predatory-proof fence boundary dating from 1992 – 2015. There are numerous records further afield, many from within the last five years.

This arboreal mammal can occur in dry sclerophyll, open woodland, heath, swamps, rainforest and wet sclerophyll forest (van der Ree et al. 2001). In Victoria, greater rates of phascogale occupancy are associated with increased stem densities of rough-barked eucalypts, such as the red box Eucalyptus polyanthemos, which may provide enhanced food resources (Goldingay et al. 2020).

Their diet consists primarily of invertebrates but they also feed on nectar when this resource is available (Scarff et al. 1998). Caution is warranted in extrapolating habitat use information from other areas to New South Wales, and there is a paucity of studies investigating the ecology of the brush-tailed phascogale in the north of its range (Edwards 2018).

Recent camera trap monitoring confirmed that brush-tailed phascogale occur within the site (T. Leary, pers. comm.). It is likely that this species will be able to cross the predator-proof fence. However, an additional step of including canopy-crossing structures such as rope bridges across and above the fence may enhance connectivity for the brush-tailed phascogale (Soanes and van der Ree 2010) and other arboreal species (including eastern pygmy possums), while retaining exclusion of introduced mesopredators. If there is a lack of tree hollows on the site, nesting habitat can be supplemented with artificial chain-saw hollows (Terry et al. 2021).

(13) Glossy black cockatoo (*Calyptorhynchus lathami*)

This bird occurs across much of eastern Australia, from the coast to as far inland as Griffith, New South Wales, with scattered records occurring up the coast of Queensland to the wet tropics (Morcombe 2003). Threats for this species include wildfires, habitat removal, competition for nesting hollows and climate change (Cameron 2009). There are numerous recent records of this species within a 5 km radius from the site and further afield.

This species inhabits wet and dry sclerophyll forests of the coast and the Great Dividing Range where stands of she-oak occur (Cameron 2006). The primary food sources for this species are *Allocasuarina* sp., especially black she-oak (*Allocasuarina littoralis*) and forest she-oak (*A. torulosa*) (Chapman 2007).

Due to the dispersive nature of this bird and the large numbers of recent records both within Ngambaa and in surrounding reserve systems, this species does not require reintroduction. It is likely to colonise as a wide-ranging species. This species will benefit from appropriate fire management in Ngambaa as a large portion of their distribution was impacted by the 2019/2020 fires, however we do not know how fire affects seed production rates of its key *Allocasuarina* food resources. There is an opportunity to manage Ngamabaa NR as a fire refuge for dispersive species, such as the glossy black cockatoo. Provisional planting of locally occurring *Allocasuarina* spp., if there is a lack of this resource, may assist populations.

(14) Varied sittella (*Daphoenositta chrysoptera*)

This species occurs across most of Australia with the exception of arid regions and Tasmania (Morcombe 2003). This species is uncommonly encountered and is in decline is some areas of New South Wales and this mirrors a decline in open woodland habitat (Seddon et al. 2003, Newman 2015). There are 15 records of this species within a 5 km radius of the site, mostly dating to the early 2000s. There are widely distributed records within surrounding reserves with more recent sightings.

This species uses a variety of habitats, including open woodlands, dry and wet sclerophyll forests, semi-arid woodland, coastal heath, urban parklands, but tend to avoid dense rainforest (Morcombe 2003). They forage for invertebrates on rough-barked eucalypts (Morcombe 2003).

This species is dispersive and will not require reintroduction. It is likely they will disperse naturally to the reserve for foraging.

(15) Square-tailed kite (*Lophoictinia isura*)

The square-tailed kite occurs in coastal and subcoastal areas in all mainland states of Australia (Debus and Czechura 1989). Habitat removal and illegal collecting of eggs are threatening processes for this species. There are 12 records within a 15 km radius of Ngambaa, where about half were recorded between 2015 – 2021.

This raptor has a preference for eucalypt forests and open woodlands (Debus and Czechura 1989, Debus et al. 1993). Field observations of a breeding pair that occurred at Wauchope (~60 km south) demonstrates that they will nest in *Eucalyptus pilularis* in wet sclerophyll woodland (Griffiths et al. 2002). Its diet primarily consists of nestling passerine birds, which are obtained by raiding nests (Brown et al. 2000, Griffiths et al. 2002). There is evidence that it shifts to a primarily insectivorous diet outside the breeding season of passerine birds, and it has been reported to raid paper wasp nests to consume their larvae (Debus et al. 1993, Brown et al. 2000).

Due to the numerous close and recent records and the availability of intact forest at the site and in the surrounding landscape, it is likely that the square-tailed kite will occur within the proposed predator-proof fence area. There is a potentiality for this species to nest within the site, but more likely the site will be used as foraging ground for existing territories. Higher densities of prey species associated with the removal of cats and foxes are likely to increase the carrying capacity of all predators, including square-tailed kites.

(16) Barking owl (*Ninox connivens*)

The barking owl is sparsely distributed along the eastern coast and semi-arid areas of the interior of mainland Australia and becomes more common in the tropical north (Kavanagh 2002a). Threats to this species include, habitat removal, inappropriate fire management, competition with introduced predators and secondary poisoning by rodenticides. There are six records within 15 km of Ngambaa that were recorded between 1998 and 2003. This includes one record that was recorded on the eastern boundary of the proposed predator-proof fence site in 2002.

The primary habitat of this owl is open woodland and partially cleared land, where they are often found near creeks and waterways (Kavanagh et al. 1995). They can be flexible in their diet but appear to prefer arboreal mammalian prey including sugar gliders (*Petaurus breviceps*) and squirrel gliders (*Petaurus norfolkensis*) (Debus et al. 2005). However, when arboreal mammals are scarce or absent, they can shift their diet towards scansorial mammals, insects and avian prey (Stanton 2011, Palmer and Caton 2016).

The forest present in Ngambaa is likely marginal habitat for the barking owl. There is a possibility that this species may visit as a ‘rare vagrant’ to the site since there are sparse records of this species within the area. Reintroduction is not suitable due to the mismatch in habitat present compared to what this species requires, but increases in prey species may result in this species becoming more common.

(17) Powerful owl (*Ninox strenua*)

This owl is distributed across much of the south-east coast from Victoria, extending up into New South Wales and as far north as Mackay, Queensland (Kavanagh 2002a). Threats include habitat removal, secondary poisoning, road mortality, inappropriate fire management and predation of fledgelings by introduced predators. There are two records of this species within Ngambaa’s proposed fence boundary dated in the year 2014. Additionally there are 10 records dated between

1992-2019 within 5 km of the fence boundary. There are numerous recent and historic records further afield and in adjoining reserve systems.

The powerful owl inhabits a diversity of habitats from open woodland, dry and wet sclerophyll, and rainforest (Kavanagh 2002a). This species requires large tracts of forest but can occur in fragmented landscapes (Kavanagh and Stanton 2002). While this species most often hunts arboreal prey, it is a generalist in the sense that it will target which ever medium-sized arboreal mammals are the most locally abundant (Cooke et al. 2006). Species that make up large portions of their diet in most populations are the sugar glider, greater glider, ring-tailed possums, brush-tailed possums and the grey-headed flying fox (Pavey et al. 1994, Schulz 1997, Kavanagh 2002b, Cooke et al. 2006).

This species does not require reintroduction as it is highly likely they are currently present at the site or will freely disperse in and out. Potential increases in food resources brought about by the predator-proof fence may result in increases abundance of this species within the area. If there is a lack of tree hollows, nest boxes or artificial hollows may provide an alternative. There is a record of a powerful owl using a nest box in an urban area. This nest box was designed with the same measurements as natural nests and had a floor dimension of 550 x 500 mm, a back wall of 800 mm, a front wall of 700 m to allow runoff with a 200 mm diameter entry hole (see Fig. 3 and see McNabb and Greenwood 2011 for more details).

Figure 4. Powerful owl using a nest box with photos of the design, photos by E. McNabb (McNabb and Greenwood 2011).

(18) Olive whistler (*Pachycephala olivacea*)

This bird is distributed from Tasmania, up through Victoria and in high elevation zones of the Great Dividing Range in New South Wales and the very south of south-east Queensland. This species has low detection rates during surveys (Lindenmayer et al. 2003), which may be due to the superficial similarity of its calls to common co-occurring species such as the eastern whipbird (*Psophodes olivaceus*) (White 1987). This species is threatened by habitat removal, inappropriate fire regimes, predation by exotic predators, competition with overabundant dominant birds and climate change.

There is a historic record 800 m away from the proposed boundary of the predator-proof fence. Most other records are from large reserve systems to the north and south at greater altitudes. Some exceptions include sparse coastal records, one of them being relatively recent recorded in autumn 2014, located about 21 km northeast of the site.

This species predominately inhabits rainforests and wet sclerophyll woodland above 500 m (Marchant et al. 2006). This species undergoes seasonal dispersal movements from high altitude during warm seasons to low altitude during cold seasons (Chan 2001).

Since this species has seasonal altitudinal dispersal patterns, reintroduction is not needed. Ensuring enhanced connectivity of the surrounding landscape and suitable conditions within the site would enable a higher chance of colonization of this bird during winter. This site may be useful for this species as a winter retreat where it is free from predation pressure of introduced predators.

(19) Masked owl (*Tyto novaehollandiae*)

Australian masked owls are distributed along a broad coastal band in all states of Australia with some populations reaching inland New South Wales, South Australia and Tasmania (Morcombe 2003). This species is listed as vulnerable to extinction in New South Wales. The primary threats to Australian masked owl are habitat clearance, introduced mesopredators, wildfire, drought and compounding impacts from climate change. Australian masked owl are scarcely seen in highly fragmented areas due mainly to urbanisation and agriculture (Kavanagh and Stanton 2002). However, there are some instances where they can form home ranges on the boundaries of large reserve systems, adjacent to fragmented habitat (Kavanagh and Murray 1996, Kavanagh and Stanton 2002). Native ground mammals that masked owl rely on for prey can be severely impacted by cats and foxes (Risbey et al. 2000), so the removal of these species is likely to increase the carrying capacity of many native predators. There were regular records of this species in 2003-2004 at Skillion Flat about 10 km to the south-west. Otherwise there are sporadic records in surrounding reserve systems, some more recent and others historic, which aligns with this species cryptic and dispersive nature.

This owl occupies large home ranges, which can include a variety of habitats. They roost in dense vegetation, tree hollows, and sometimes caves (Young et al. 2021). This species uses ecotones and open habitat for foraging (Kavanagh and Murray 1996, Young et al. 2020).

Due to the rarity and difficulty in detecting the Australian masked owl, and their dispersive tendencies (Kavanagh 1996, Debus 2001), reintroduction is not needed. Colonisation is more likely to occur with increases of prey resources resulting from the eradication of foxes and cats within predator-proof fence. The Australian masked owl in south-eastern Australia primarily feed on native scansorial mammals, such as *Antechinus* sp. and the bush rat (*Rattus fuscipes*), but will occasionally prey on arboreal mammals, such as the sugar glider (*Petaurus breviceps*) and the common ringtail possum (*Pseudocheirus perigrinus*) (Kavanagh 1996, Kavanagh 2002b).

A predator-proof fence enclosure will likely lead to increases in prey for masked owl. The areas around the fence will likely be kept clear and hence produce an ecotone that may provide optimum foraging habitat. Additional habit augmentation can occur if there are limit resources such as hollow bearing trees for prey and for roosting. Chain-sawed hollows (dimensions: 40 – 100 cm long, 20 – 40 cm wide, with an entrance diameter of 4 cm) would be suitable to increase habitat for arboreal and semi-arboreal prey species such as the sugar glider and Antechinus (Griffiths et al. 2020, Terry et al. 2021).

If an Australian masked owl nest is discovered within the site, and if hollow bearing trees are limited, provision of additional artificial nesting sites may assist the population. For example, Thomson (2006) found that large nest boxes (dimensions: ~117 cm tall x ~30 cm wide with a 25 x 25 cm entrance, see Fig. 4 for design depiction) placed in close proximity to an active nest were used as roost sites by juvenile masked owls.

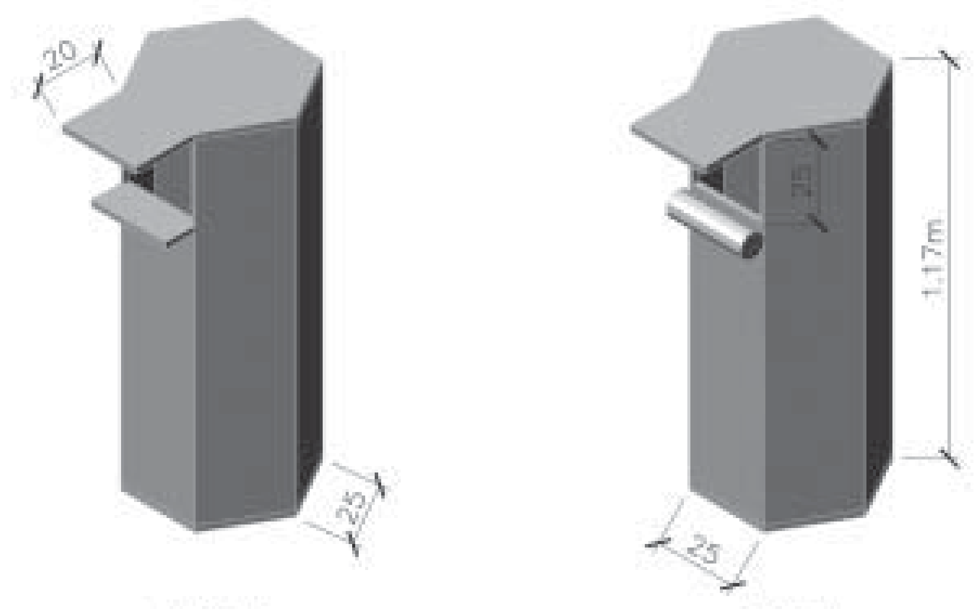

Figure 4. Design of Australian masked owl nest box by Thomson (2006).

(20) Greater sooty owl (*Tyto tenebricosa*)

The distribution of this owl extends along the eastern coast of Australia from the Dandenong Ranges in Victoria to the Conondale Ranges in Queensland (Kavanagh 2002a). Threats for this species include habitat removal, secondary poisoning from rodenticides, and impacts of inappropriate fire management on prey species. There are two records from 2019 that are within 800 m of the proposed boundary of the predator proof fence. There are numerous additional records within 10 km of the surrounding area, both recent and historic.

This owl occurs in rainforest and wet sclerophyll woodland (Kavanagh 2002a). There is evidence from sub-fossil owl pellets that the greater sooty owl regularly predated on several species that are proposed to be reintroduced or are targeted for population enhancement in Ngambaa, including the eastern quoll, eastern chestnut mouse, eastern bettong, eastern pygmy possum, brush-tailed phascogale and New Holland mouse (Bilney et al. 2010, Bilney 2020).

This species is likely to see an increase in carrying capacity through a relaxation of predation pressure of introduced mesopredators on potential prey species (Bilney et al. 2007, Law et al. 2013). This may result in increased abundances of other avian predators, as their diets overlap more in areas where there are less ground-dwelling prey species (Bilney et al. 2006). Hence, with increases of ground-dwelling mammals due to introduced predator eradication, there may be less dietary competition between the greater sooty and powerful owl (Bilney et al. 2006).

Due to the immediate proximity of recent records and the presence of suitable habitat, it is highly likely that a population already exists either within the proposed site or in the immediate area. Reintroduction is not needed and this species can disperse inside and outside the site readily. There is no published account of this species using artificial hollows or nest boxes, however structures mimicking the requirements of natural hollow sites in use could be used to inform design.

(21) Spotted-tailed Quoll (*Dasyurus maculatus*)

This quoll occurs in Tasmania, Victoria and up the coast of New South Wales and south-east Queensland with a disjunct population that represents a different subspecies located in the wet tropics of far north Queensland (Uzqueda et al. 2020). Unlike the eastern quoll, the spotted-tailed quoll persists on mainland Australia but its post-European colonization distribution has been reduced by about 40-60% (Belcher 2004). There are numerous records of this species within 5 km of the proposed predator-proof fence site and six of these are from the last decade.

The spotted-tailed quoll inhabits a diverse range of habitats from subalpine woodland, rainforests, wet sclerophyll and some dry sclerophyll woodland (Belcher 2004). A commonality between the habitats that it occurs in is the presence of varying topography composed of deeply dissected gullies and hilly terrain (Belcher and Darrant 2006, Glen and Dickman 2006b), although this might be an artefact of competition avoidance with introduced predators. This quoll can use a variety of microhabitats for den sites including hollow logs, rock crevices, burrows, tree hollows and artificial structures (Glen and Dickman 2006b). The diet of northern New South Wales populations consists mainly of small and medium-sized mammals (Glen and Dickman 2006a), including species that are considered for reintroduction in this review, including the long-nosed potoroo, parma wallaby, northern brown bandicoot and the yellow-bellied glider.

Due to the prevalence of recent records (despite the cryptic nature of the quoll), and the presence of appropriate habitat, it is highly likely that spotted-tailed quolls still persist within the proposed site of the predator-proof fence or otherwise is persists within the general area. Reintroduction is unlikely to be needed for this species. The presence of this species within a predator-proof fenced site may make reintroduction of potential prey species more difficult. Caution and consideration of the role of this predator in the survival probability of reintroduced prey species needs to be taken. Based on density estimates of nearby results is it likely that 8 spotted-tailed quolls live in the site proposed for fencing. The viability of this population, if it becomes isolated, is questionable. Genetic connectivity can be maintained through either (1) creating barrier crossing structures that are permeable to the quoll but impermeable to introduced predators or (2) on-going genetic management of the population through metapopulation management (Davies-Mostert et al. 2009).

(22) Northern brown bandicoot (*Isoodon macrourus*)

The northern brown bandicoot is found across much of northern and eastern Australia and is also present in New Guinea (Menkhorst and Knight 2011). In New South Wales, it is found along the northern coast and the Great Dividing Range from just north of Sydney. There is a record from 2019 within 5 km of the proposed fence site and there are another six nearby records that are older. There are numerous records from further afield in the large reserve systems of the Great Dividing Range and many that are located in fragmented coastal bushland. This species is threatened by predation from introduce predators, road mortality and habitat removal (although less so that other species as it appears more tolerant to disturbance).

This species occupies dry sclerophyll woodland, heathland, grasslands, but it can also persist in urban reserves and degraded bushland (Fox 1982, Gott 1998, Fitzgibbon et al. 2011). The northern brown bandicoot has a high fecundity and hence may be able to quickly population areas after recolonisation (FitzGibbon 2015) or in response to the removal of introduced predators.

It seems unlikely that this species has gone undetected in Ngambaa NR as it is unmistakeable in appearance, behaviour and the evidence of its presence that it leaves via its diggings. The reason for its potential absence is unclear, given that this species is ecologically flexible and adaptable in habitat use. This would be an excellent candidate for reintroduction.

(23) Stephen's banded snakes (*Hoplocephalus stephensii*)

This snake is cryptic and has a limited distribution extending north of Sydney to south-east Queensland. Threatening processes affecting Stephen's banded snakes include habitat fragmentation, forestry and poaching, none of which are likely to be an issue in the Ngambaa safe haven. Despite the cryptic nature of this species, it has been recently recorded nearby, about 2.7 km to the east of the site. Several other records exist in nearby reserve systems, both recent and historic.

This snake is flexible in the vegetation communities it exists in, being found in rainforest, wet sclerophyll and coastal woodlands (Fitzgerald et al. 2005). Despite this flexibility in vegetation communities, it requires large tracts of contiguous habitat for persistence (Fitzgerald et al. 2002). Additionally, it requires hollow bearing trees, which are used for refuge and foraging (Fitzgerald et al. 2002).

Due to the recent nearby records and the habitat present, it is likely this species persists within Ngambaa. This snake disperses along the ground (Fitzgerald et al. 2002) and is small enough to fit through fence mesh gaps, and hence it has the ability to freely move inside and outside the site despite predator-proof fencing. Primary prey species, such as the eastern pygmy possum, may benefit from introduced mesopredators predation-pressure release, which in turn may result in more resources for the snake (Fitzgerald et al. 2004). Taken together, it is likely that this species does not require reintroduction, and is already present in the proposed predator-proof fencing site, or will colonise the site unassisted.

(24) Green-thighed frog (*Litoria brevipalmata*)

This frog is distributed mostly on the east of the Great Dividing Range from the north of Sydney to south-east Queensland (Anstis 2013). It is unclear if this species is in decline as their cryptic nature precludes a detailed investigation into population trajectory or occupancy status (Mahony 1993, Lemckert et al. 2006). This species is threatened by habitat removal. It is unclear if additional threats such as chytrid-induced disease or altered weather patterns from climate change threaten this frog. There are numerous nearby records that date back to almost two decades and are as close as 2.7 km to Ngambaa. Recent records that date to about 4 years ago are located 30 km to the north and south of the reserve.

This species occurs in rainforest, wet and dry sclerophyll woodland and heathland (Murphy and Turbill 1999). It strictly breeds in ephemeral water bodies that refill after heavy rainfall (Anstis 2013) and seeks shelter primarily in leaf litter and low dense vegetation (Lemckert and Slatyer 2002).

Males call for one night for breeding, which can make this a difficult species to survey (Lemckert et al. 2006). Hence, this species persistence at the site is uncertain. The first step to determine if this species occurs at the site is to identify any ephemeral lentic water bodies that may be suitable breeding habitat. Then surveys can be undertaken at these water bodies during rainfall events of >80 mm during late spring or summer. If ephemeral waterbodies are limited on the site, and there is no evidence of the species persisting within or nearby, then habitat creation and reintroduction might be suitable. However, caution should be taken when constructing ephemeral wetlands, as hydroperiod can be difficult to predict and may not always be related to wetland depth as per hydrological models (Beranek et al. 2020a).

(25) Giant barred frog (*Mixophyes iteratus*)

Sudden declines of this species coincided with the period when the chytrid pathogen *Batachochytrium dendrobatidis* was thought to have spread across Australia (Mahony 1993, Mahony et al. 1997). However, despite impacts from chytrid-induced disease and climatic perturbations, many populations have persisted and some appear to be recovering (Goldingay et al. 1999, Lollback et al. 2021). Other threats include habitat removal and breeding site degradation. There are numerous recent records as late as 2017 from rainforest waterways within 5 km of the site, and records within the site dating to 1992 at Cedar Creek.

The giant barred frog occurs in rainforest and wet sclerophyll forest where they mostly breed in permanent streams (Lemckert and Morse 1999). A study of their movement indicates that adults do not move more than 20 m away from these waterways (Lemckert and Brassil 2000). Some important microhabitat features that have been found to correlate with occupancy include a positive relationship with the length of undercutting in the bank and a negative relationship with riffle length (Lollback et al. 2021). Another study found that increased erosion and sedimentation may cause negative impacts to populations (Lewis and Rohweder 2005).

It is likely that this species persists in permanent streams within rainforest sections of the site. Hence, reintroduction is probably not needed. Given the frequency with which frogs are killed by feral cats (Woinarski et al. 2020), it is likely that larger frogs present in Ngambaa will benefit from the removal of introduced predators. Monitoring should be conducted to update knowledge regarding the distribution of this frog in Ngambaa. Additionally, habitat management may enable improved conditions, such as preventing erosion with plants that have sturdy root systems, which may enhance undercutting banks and reduce sedimentation.

(26) Yellow-bellied glider (*Petaurus australis*)

This arboreal marsupial extends from Victoria, along the eastern coastline of New South Wales, into south-east Queensland with disjunct populations occurring in the far north wet tropics (Menkhorst and Knight 2011). There are >30 records within 5 km of the proposed site for the predator-proof fence within the last decade, many of which are from the last three years. There are more recent and historic records from further afield.

The yellow-bellied glider inhabits a variety of forests and woodlands in eastern Australia and is found at a range of altitudes from sea level to 1400 metres (Menkhorst and Knight 2011). Site occupancy of this species is higher with increases in the basal area of sap resource trees and decreases with increasing habitat fragmentation (Eyre 2007). This species has a large home-range size, which is between 46-53 hectares (Goldingay and Kavanagh 1993). Population viability analyses indicate that they require forest extents of between 18,000 – 35,000 hectares to remain viable (Goldingay and Possingham 1995).

Based on the recent records and the large extents of suitable habitat at the site, it is highly likely that the yellow-bellied glider occurs within Ngambaa. The reserve system that Ngambaa resides in is >50,000 hectares and hence it can support a viable population (Goldingay and Possingham 1995). Since this species has large home-ranges, and requires a large reserve system to maintain a viable population, it is advised that connectivity should be maintained for dispersal inside and outside the predator-proof fence site without compromising the exclusion measures of introduced mesopredators. It is possible that the gliders may be able to move freely inside and outside the fenced area by gliding over the boundary fence from tree to tree if the horizontal distance between the trees is less than 25 m (Goldingay 2014). This may be enhanced by the construction of gliding poles in the boundary area to reduce the length required for the gliders to jump. This has worked to facilitate movement of this species across highways (Taylor and Rohweder 2020).

(27) Koala (*Phascolarctos cinereus*)

Koalas are distributed across much of Victoria, New South Wales and Queensland, and historically occurred throughout these states except in semi-arid/arid areas. The distribution of koalas has decreased over time, with local extinctions occurring in many inland populations (C. Beranek, *unpubl. data*; Reckless et al. 2018), while even some coastal populations are showing evidence of decline (e.g., Port Stephens, C. Beranek, *unpubl. data*). Koalas are threatened by habitat removal, road mortality, disease (especially chlamydia), wildfire, inappropriate fire management, and droughts and heatwaves caused by climate change (Lunney et al. 2014, McAlpine et al. 2015, Phillips et al. 2021). There are numerous recent and historical records within 5 km (~50) of the proposed predator-proof fence boundary. Some occur within the site boundary.

Koalas are arboreal folivores that are mostly associated with eucalypt forests throughout their range, however their diet is regionally specific (Menkhorst and Knight 2011). In the mid coast-north coast portion of their range, there are several habitats that koalas occur in, including coastal swamp mahogany forest, open red gum woodland and other types of denser eucalypt woodland. From these habitats, they prefer to forage on foliage from tallow-wood (*Eucalyptus microcorys*), grey gum (*Eucalyptus propinqua*), swamp mahogany (*Eucalyptus robusta*), forest red gum (*Eucalyptus tereticornis*) and grey box (*Eucalyptus moluccana*) (Phillips et al. 2021).

Recent camera trapping surveys detected this species within the proposed fenced area (T. Leary, *pers. comm.*). Carrying capacity and density of the population can be confirmed with drone based thermal imagery (Beranek et al. 2020b, Witt et al. 2020). Surveys implementing this method will be used in Ngambaa in autumn 2022 with the primary goal of determining koala density throughout the reserve (A Roff, *pers. comm.*). The viability of this population needs to be considered. It can be maintained through either (1) creating barrier crossing structures that are permeable to the quoll but impermeable to introduced predators or (2) on-going genetic management of the population through metapopulation management. Population viability modelling based on known densities can be undertaken to determine the frequency required for supplementing the population with more genetic stock.

(28) Rufous scrub-bird (*Atrichornis rufescens*)

This species occurs in elevations greater than 600 m along the Great Dividing Range from Barrington Tops to Lamington NP in south-east Queensland (Newman et al. 2014). They were thought to be formerly distributed in lowland rainforests in northern New South Wales but may have suffered declines due to habitat clearance and logging (Ferrier 1984, Garnett et al. 2011), while its almost ground-dwelling ecology suggests introduced predators may also have been a threat. Fires are also thought to be a threatening process as the rufous scrub-bird appears to be a late successional species post-fire, presumably due to the removal of the undergrowth vegetation during burns that they rely on for cover (Stuart and Newman 2018). Distant records exist in the large reserve systems to the south-west and south-east of the site (>40 km) in areas of greater elevation (Andren 2016).

Rufous scrub-birds occur in warm and cold temperate rainforests and wet sclerophyll woodland (Ferrier 1985). They mainly use dense habitat consisting of rainforest undergrowth and woody debris (Ferrier 1985, Stuart and Newman 2018). They are considered to be insectivorous, but no specific knowledge of their diet exists as far as we are aware. It is not clear what habitat features drive occupancy patterns of this species (Stuart and Newman 2018).

While there are no historic records within proximity, it is likely that this species pre-European distribution extended to the study site (Ferrier 1984). Furthermore, it is likely that this species is locally extinct as bird records are more often reported than other groups and this species has an obvious call, even if it is cryptic in nature and only calls during summer. There are no known reintroduction attempts of the rufous scrub-bird and a comprehensive understanding of their occupancy patterns is lacking. However, the broad vegetation communities that this species requires are present on the site, so from this perspective, this species would make an interesting case for a pioneering reintroduction attempt. The current distribution of the rufous scrub-bird resides at somewhat higher elevation than Ngambaa (600 m compared to 200 m), so this adds another shroud of uncertainty. Caution must be warranted for collecting stock, as populations are small and the species is exceedingly difficult to capture (A. Stuart *pers. comm.*; Stuart and Newman 2018). However, in the absence of feral cats and red foxes, this species may do very well when reintroduced. If reintroduction were to occur, strategic fire management practices are needed to ensure mosaics of dense undergrowth remains unburnt. That said, it seems risky to take scrub-birds from known populations that are already tiny to reintroduce to Ngambaa when we lack an understanding in the specifics of their habitat requirements (A. Stuart, *pers. comm.*), and when there is no captive stock available to act as founders.

(29) Black-striped wallaby (*Macropus dorsalis*)

This species is mainly distributed in the north-eastern portion of Australia, from Townsville, Queensland, to northern New South Wales where it occurs on both sides of the Great Divide (Menkhorst and Knight 2011). On the north-west slopes of New South Wales it occurs in Brigalow remnants that extend to the south of Narrabri. On the north coast, it is confined to the upper catchments of the Clarence and Richmond Rivers. Agricultural incursion, road mortality, illegal killing and extreme weather events threaten this species (Woinarski et al. 2014). It is unlikely that this species has ever occurred previously at Ngambaa however there is some suggestion that it is extending its distribution to the south (P. Fleming, *pers. comm*.).

The habitat of this species is composed of dense shrubby vegetation that occurs near more open, grassy areas to provide suitable feeding habitat (Menkhorst and Knight 2011). It is closely associated with dry rainforest in the north coast of New South Wales but also occurs in moist eucalypt forest with a rainforest understorey or a dense shrub layer (Menkhorst and Knight 2011).

Since this species is previously not known from Ngambaa, its introduction here would be considered an assisted colonisation. It is unknown how this species may respond to competition with the extant macropod species, and the possibility of competitive exclusion should be examined. Assisted colonisation may be considered on the basis of providing a climate refuge and an insurance population for this declining species.

(30) Hastings River mouse (*Pseudomys oralis*)

This mouse has a scattered distribution where populations occur at elevations above 500 m along the Great Dividing Range from Mount Royal National Park, New South Wales to the Bunya Mountains in southern Queensland (Pyke and Read 2002). The main threats are habitat removal, predation from introduced predators and inappropriate fire regimes, however some disturbance may benefit this species (Law et al. 2016). Subfossil remains indicate that this species historically occupied lower elevations (Pyke and Read 2002), hence it is possible that this species may have occupied broader areas, in the absence of foxes and cats, to extend close to or include Ngambaa. The closest Hastings River mouse populations to Ngambaa exist in the Carrai Plateau and Styx River, which are located about 30 - 40 km to west.

This Hastings River mouse occurs in open forests that have dense, low ground cover and a diverse mixture of ferns, grass, sedges and herbs (Pyke and Read 2002, Law et al. 2016). Microhabitat refuges, such as logs and shrubs, are important for refuges (Graham 2005).

While there is no concrete evidence that this species used to occur within the area of the proposed predator-proof fence site, the habitat here includes dry grassy open forest, which would indicate that it is suitable. Reintroduction may be considered, except it is unknown how this species would interact with the other native rodents proposed to be reintroduced (New Holland mouse and eastern chestnut mouse). Furthermore, there is evidence of negative effects of presently occurring native rodents, such as the bush rat *Rattus fuscipes* (Law et al. 2016). This species can natural recolonise the fenced area by moving through the predator-proof fence.

(31) Common planigale (*Planigale maculata*)

This species is distributed north from the Hunter Valley in New South Wales, up the coast to the tip of Cape York in Queensland and along northern Australia, into the Northern Territory and the north of Western Australia (Menkhorst and Knight 2011). However, what is currently considered *Planigale maculata* consists of several cryptic species (Westerman et al. 2016). There are only four records of this species within a 25 km radius of Ngambaa. The most recent record is from 2010 and occurred to the east of the site. There are more records of this species when extending the radius to 50 km, however most are from coastal situations and with a few occurring in low elevations adjacent to reserves of the Great Dividing Range.

Common Planigales are flexible in habitat use and can occur in rainforest, eucalypt forest, heathland, marshland, grassland and rocky areas (Menkhorst and Knight 2011). Indeed, a close relative of *Planigale maculata* has been demonstrated to be a habitat generalist – requiring black soil plains, grassland and semi-arid woodland in the Kimberly region (Legge et al. 2011). However, in the south of their range in New South Wales, they appear to be restricted to open woodlands and coastal habitats. There are a paucity of records in forests of the Great Dividing Range.

It seems unlikely that this species occurs within Ngambaa. If it currently or historically occupied the reserve, it is likely that they would have occurred in low numbers since the habitat seems marginal for this species in this part of their range. The drivers behind why this species is excluded from forests of the Great Dividing Range is unclear. Depending on the design of the predator-proof fence, it is likely this species can move freely in and out due to its small size.

(32) Tasmanian pademelon (*Thylogale billardieri*)

This species is presumed extinct on the mainland of Australia and its present distribution is confined to Tasmania and surrounding offshore islands (Menkhorst and Knight 2011). The last records of it on the mainland occurred in the 1800s (Calaby 1971). Its historic mainland distribution was apparently confined to Victoria (Rose and Rose 2018). There is also evidence that these populations colonised Victoria when there was a connective land bridge between Tasmania and the mainland, and hence it evolved in Tasmania (Rose and Rose 2018). On this basis, this species should not be reintroduced into Ngambaa.

Population estimates

Estimates of population size have been produced for species that are likely restricted to the predator-proof fence area so that their population viability can be assessed (Table 3). The population sizes of dispersive species have not been estimated. The population size of the spotted-tailed quoll has been estimated to inform potential genetic management.

Table 3. Population estimates.

Species	Habitat assumed	Density (individuals/ha)	Estimate (up. – lo.)	References
Bettongia gaimardi	Dry sclerophyll	0.25 (0.18-0.43)	171 (124-299)	Manning et al. (2019)
Dasyurus vivverinus	Dry sclerophyll	0.1 (0.03-0.21)	72 (24-144)	Godsell (1983)
Dasyornis brachypterus	Semi-mesic grassy forest	2.7 (1.16-4.7)	3866 (1661-6730)	Bain et al. (2008)
Aepyprymnus rufescens	Semi-mesic grassy forest and wet sclerophyll	0.18 (0.03-0.37)	372 (62-764)	Pope et al. (2005)
Cercartetus nanus	Rainforest and wet sclerophyll	5 (2.5-7)	3180 (1590-4452)	Bladon et al. (2002)
Macropus parma	Wet sclerophyll and rainforest	0.04 (0-0.4)	25 (3-254)	[1]
Potoroo tridactylus	Wet sclerophyll and rainforest	0.26 (0.1-0.4)	165 (64-254)	Mason (1998)
Pseudomys novaehollandiae	Dry sclerophyll and semi-mesic grassy forest	3.4 (1-24)	7215 (2122-50928)	Wilson (1991)
Pseudomys gracilicaudatus	Dry sclerophyll and semi-mesic grassy forest	1 (0.14-2.29)	2122 (303-4850)	Fox (1982), Wilson et al. (2005)
Thylogale stigmatica	Rainforest	2.4 (1.9-2.9)	5 (4-6)	Vernes (unpublished data)
Atrichornis rufescens	Rainforest and Wet sclerophyll	0.03 (0.02-0.04)	21 (13-27)	Stuart and Newman (2018)
Macropus dorsalis	Dry sclerophyll			
Pseudomys oralis	Wet Sclerophyll	2 (0.1-4)	1268 (63-2536)	Pyke and Read (2002)
Dasyurus maculatus	Entire area	0 (0-0.01)	6 (2-14)	Glen (2008)
Isoodon macrourus	Semi-mesic grassy forest	1.4 (0.02-2.6)	888 (13-1648)	Fitzgibbon et al. (2011)

1 Based on personal observations of 5 individuals detected over 500 m long transects at Gibraltar Range National Park, but with an order of magnitude range.

There are limitations to the method of population size estimation used and caveats in interpreting the results. Firstly, these numbers are based on assumptions of habitat type use with density estimates of the closest habitat type from the literature. This also does not consider the stochastic population fluctuations that many of these species are known to undergo. Additionally, there may be unforeseen factors that governed population sizes of the species within the predator-proof fence area. Indeed, some of these factors will be entirely novel. For example, eastern quolls have not existed in forests of northern New South Wales for possibly over a century, and there has been no examinations of their populations in such habitat. Therefore, it is difficult to predict how their populations will respond and even what habitat types they will utilise. This is especially so when considering novel species interactions.

The bottom-up factors that constrain the densities of wildlife in Australia are also unknown – largely due to insufficient research on our wildlife across different regions. This stands in stark contrast to Africa where the densities of herbivores can be predicted by a combination of rainfall and soil type (East 1984), and predators by the biomass of their preferred prey species (Hayward et al. 2007). The density of Australian species undoubtedly varies across rainfall and soil productivity gradients, but our monitoring to date has provided insufficient evidence at present to predict how these affect population densities.

It is possible to forecast population size increases and fluctuations in response to novel species interactions with modelling techniques such as ensemble ecosystem modelling (Baker et al. 2017). These models can include expert knowledge of potential interactions with previous or existing monitoring data to produce accurate forecasts of potential population trajectories under different reintroduction scenarios (Baker et al. 2019). This allows inference to which strategy is likely to lead to success and which species are going to be the most difficult to establish (Peterson et al. 2021).

Habitat enhancement

Augmentation of habitat or other resources will be most successful if key resources that limit each species' population size can be targeted and increased. Clearly the creation of the fence and removal of introduced predators and competitors will offer major benefits to critical weight range fauna given the impacts foxes and cats have on this group (Burbidge and McKenzie 1989). Other potential habitat enhancement features that may increase the carrying capacity of Ngambaa's safe haven are discussed below.

Barrier crossing structures – A predator-proof fence in Ngambaa is unique, as it will occur in a forest with numerous arboreal species where most other predator-proof safe havens exist in more arid environments. In this situation, barrier-crossing structures could be considered as a feature to maintain connectivity of arboreal species while retaining exclusion of introduced mesopredators. This will be depend on the area of cleared vegetation on either side of the fence, but if it exceeds 25 m these structures might be beneficial. Structures that are typically used for maintaining connectivity for arboreal species across highways can be implemented in this situation. Rope bridges can be suspended from tree to tree above the fence. To date, intensive monitoring of several rope bridges had not led to the detection of a cat or a fox utilising these structures (B Taylor, *pers. comm.*). Glider poles may also be used to promote connectivity of gliders, although it is likely they would still be able to glide in and out depending on the tree heights and the height of the fence. The mesh size of the fence may also permit connectivity of small ground mammals, amphibians and reptiles, or one-way gates could be installed (Crisp and Moseby 2010). If one-way gates are used, it would be important for ongoing management of introduced predators in the surrounding landscape so that target species can establish populations beyond the predator-proof fence.

Coarse woody debris supplementation – It is presumed that the construction of the predator-proof fence will require some tree removal along the fence corridor. The trees that are removed can be salvaged as coarse woody debris habitat. Targeting grassy woodland areas that are lacking in coarse woody debris would be the most beneficial strategy, especially for ground-dwelling mammals such as the eastern quoll, New Holland mouse, and the eastern chestnut mouse. This would also be beneficial for the eastern bettong as they use fallen trees and logs for nesting sites (Taylor 1993).

The structure of debris placement is important for bush stone-curlews, which prefer lower branches of debris for visibility of potential predators (A. Manning *pers. comm.*).

Nest-boxes or artificial chainsaw hollows – Additional hollow resources should be considered if these are lacking. Young regrowth forest may be lacking suitable hollows and hence should be the target of this intervention. Nest boxes and artificial chainsaw hollows can be designed to target specific species such as the eastern pygmy possum, brush-tailed phascogale, the yellow-bellied glider and the greater glider. This will likely lead to increases in recruitment of these species.

Ephemeral wetland construction – The green-thighed frog is dependent on ephemeral wetlands and, if there is no suitable habitat and this species is desired on the site, provision of extra ephemeral wetlands should be considered. This may also benefit populations of other ephemeral breeding frog species that are likely to be present within the reserve, including the sandpaper frog (*Lechriodus fletcheri*), red-eyed tree frog (*Litoria chloris*), bleating tree frog (*Litoria dentata*) and the green tree frog (*Litoria caerulea*).

Cultural burns - Some species are dependent on grassland and hence care should be taken to ensure the habitat does not become overgrown. Small-scale cultural burns may lead to enhanced habitat and resources for these species. This may be especially important for proposed early fire successional species such as the eastern chestnut mouse and the New Holland mouse. However, caution needs to be taken to ensure not all habitat is burnt, as some species require old growth unburnt forest with a dense understory (e.g., the rufous scrub-bird).

Other issues

The creation of fenced reserves is of immense value to the conservation of Australia's wildlife (Ringma et al. 2017), however it also raises some unique management issues (Hayward and Kerley 2009). Knowledge of the carrying capacity of species at a site is critical in constrained areas (Hayward et al. 2007, Hayward and Kerley 2009). Once carrying capacity is reached, it is important to reduce the risk of overpopulation, which can occur via one-way gates (as occurs in Arid Recovery; Crisp and Moseby 2010) or via metapopulation management of moving individuals of dispersing age between reserves (as occurs for large African predators; Davies-Mostert et al. 2009, Lindsey et al. 2011). So adequate recognition of the management actions necessary to ensure population viability into the long-term is essential when managing a fenced safe haven.

To date, our knowledge and understanding of the ecology of biodiversity within fenced areas and the impacts of those fences largely stems from single site studies. Much greater advances would occur if research projects were replicated across the full suite of safe havens being created by the New South Wales government, rather than occurring in isolation.

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