1   **Status of mammals on Groote Eylandt: safe haven or slow burn?**

2   Jaime Heiniger [a], Hugh F Davies [b] and Graeme Gillespie [a, c]

3   [a] Flora and Fauna Division, Department of Environment and Natural Resources, Northern

4   Territory Government, Berrimah NT 0828, Australia

5   [b] NESP Threatened Species Recovery Hub, Research Institute for the Environment and

6   Livelihoods, Charles Darwin University, Casuarina Northern Territory 0909, Australia

7   [c] School of Biosciences, The University of Melbourne, Parkville Victoria 3010, Australia

8   * Corresponding author: jaime.heiniger@uqconnect.edu.au

9

10   **Abstract**

11   Native mammals across northern Australia have suffered severe decline, with feral cats *(Felis*

12   *catus)*, introduced herbivores and changed fire regimes being implicated as drivers. However,

13   uncertainty surrounding the relative contribution of each of these threats, and the interactions

14   between them, is limiting the development of effective management strategies. The absence of

15   introduced herbivores and cane toads (*Rhinella marina)* on Groote Eylandt, Northern Territory,

16   provides an opportunity to evaluate some hypothesised threats in isolation of others. We used

17   camera traps to investigate the correlates of native mammal distribution and abundance at 112

18   lowland savanna sites across Groote Eylandt. Two large grids of camera traps were also

19   deployed to obtain estimates of feral cat density. We hypothesised that native mammal

20   populations would be negatively associated with feral cat occupancy as well as frequent, large

21   fires. Native mammal site-occupancy on Groote Eylandt was generally higher compared to

22   mainland Northern Territory. Feral cats were infrequently detected, precluding both an estimate

23 of feral cat density, or an evaluation of the relationship between feral cats and native mammals.

24 We found no evidence that native mammal site-occupancy or relative abundance are negatively

25 associated with frequent, large fires. The relatively healthy state of native mammal populations

26 on Groote Eylandt is likely due to the low density of feral cats, the benign fire regime and the

27 absence of large introduced herbivores and cane toads. However, due to a lack of historical

28 data, the current state of mammals should not be taken as evidence that these populations are

29 safe from decline. This study highlights that the apparent resilience of mammal populations is

30 a result of complex interactions between factors that vary substantially across the landscape.

31 Caution is therefore required when making broad inferences about the drivers of mammal

32 decline from studies that are spatially and temporally limited.

33

34 **Keywords:** islands, feral cats, fire, declines, threatened mammals.

35

36 **Introduction**

37 In the 230 years since European arrival, Australia has lost >10% of its terrestrial mammal fauna

38 and contributed more than 30% of global mammal extinctions recorded since 1500 (Woinarski

39 *et al*. 2015). In contrast to other continents, where large-bodied mammals near areas with high

40 human population density have been most prone to extinction (Cardillo et al., 2005), Australian

41 species with body sizes between 35 – 5500 g (henceforth referred to as the critical weight range:

42 CWR), in remote and relatively unmodified areas, have suffered the highest extinction rates

43 (Woinarski et al., 2015). Worryingly, this trend has continued in recent decades, with

44 widespread, significant declines of CWR mammals recorded across Australia's northern

45 savanna landscapes (Woinarski et al., 2010, Ziembicki et al., 2015, Fisher et al., 2014, Davies

46 et al., 2018).

47 Factors implicated in the current mammal decline across northern Australia include the higher

48 incidence of large-scale, intense fires occurring since the breakdown of traditional Aboriginal

49 burning practices (Russell-Smith and Edwards, 2006, Russell-Smith et al., 2003), habitat

50 degradation by large, feral herbivores and pigs *(Sus scrofa)* (Legge et al., 2011a, Kutt and

51 Woinarski, 2007), and predation by feral cats *(Felis catus)* (Ziembicki et al., 2015, Fisher et

52 al., 2014, Woinarski et al., 2011a, Frank et al., 2014). Due to a lack of empirical data, we do

53 not know if the initiation of mammal population declines in northern Australia coincided with

54 the establishment of any one of these hypothesised threats. Although northern Australian

55 mammal declines were quantitatively documented only relatively recently, it is plausible that

56 mammal populations have been declining over a much longer period. The uncertainty regarding

57 the relative contribution of each threat to northern mammal declines is limiting our ability to

58 develop and implement effective mitigation strategies.

59 Further complicating our ability to understand the factors implicated in declines of native

60 mammal populations across northern Australia are potentially complex interactions between

61 them. For example, frequent fire and feral herbivores may exacerbate the impact of feral cat

62 predation by simplifying vegetation structure (McGregor et al., 2014, McGregor et al., 2015,

63 Leahy et al., 2016, Davies et al., *in press*), while fire concurrently degrades the availability of

64 critical resources such as tree hollows (Woolley et al., 2018), perennial grasses and a fleshy

65 fruit-bearing shrub layer (Vigilante and Bowman, 2004). However, not only do fire regimes

66 vary significantly in frequency and intensity across northern Australian savanna landscapes, so

67 too does the occurrence and density of feral herbivores. Superimposed on these are a range of

68 environmental conditions (such as rainfall, climate, productivity and landscape complexity)

69 and land-use histories (i.e. indigenous management vs. pastoral use) that also influence the

70 resilience of mammal populations. Resolving the contribution of these factors in such a vast,

71 complex landscape is crucial to developing informed and effective management.

72 Islands provide important refugia for biota because factors responsible for mainland declines

73 are often absent or less intense (Legge et al., 2018, Ziembicki et al., 2015). However, as island

74 populations are generally small, with limited potential for recolonisation, they are often more

75 vulnerable to local extinction if threats do arrive (Woinarski et al., 2011b, Ziembicki et al.,

76 2015, Burbidge and Manly, 2002). For example, northern quolls (*Dasyurus hallucatus)* were

77 extirpated from Vanderlin Island in the Gulf of Carpentaria following the establishment of cane

78 toads (*Rhinella marina*), and native mammals declined to undetectable levels on West Island

79 following the introduction of feral cats (Paltridge et al., 2016, Woinarski et al., 2011b). Until

80 recently, Melville Island was also considered an important refuge for threatened mammal

81 species, however, Davies et al. (2017, 2018) reported evidence of population declines. This

82 pattern may indicate that declines may simply be lagging on these large northern Australian

83 islands compared to the mainland.

84 Groote Eylandt is Australia's fourth largest island and has high conservation values (Northern

85 Territory Government, 2017). It supports most of the global population of the northern

86 hopping-mouse (*Notomys aquilo)*; one of few extant populations of the brush-tailed rabbit-rat

87 (*Conilurus penicillatus*); is an important refuge for the northern quoll; and several other small

88 and medium sized mammal species occur at seemingly much higher densities than on the

89 mainland (Diete et al., 2016, Mahney et al., 2009, Taylor and ALC, 2016). The relatively intact

90 state of the mammal fauna may be due to the absence, or low levels, of threatening processes

91 operating on Groote Eylandt. While feral cats are present, the island remains free of cane toads,

92 has no feral herbivores or pigs, and retains a relatively benign fire regime compared to the

93 mainland of northern Australia (Northern Territory Government, 2017, NAFI, 2017).

94 However, recent research suggests that some mammal declines may have occurred on Groote

95　Eylandt (Diete et al., 2016), and the presence of feral cats poses a potentially major threat to

96　CWR mammals. In light of recent findings on Melville Island (Davies et al., 2017), mammals

97　on Groote Eylandt may also be vulnerable to some of the processes driving declines elsewhere

98　across northern Australia.

99　Understanding the responses of species to different threats under varying environmental

100　conditions is crucial to optimising management strategies in different ecological settings.

101　Groote Eylandt provides a useful opportunity to evaluate some factors implicated in the decline

102　of native mammal species in isolation of other potentially interacting threats. We conducted a

103　survey of lowland forest and woodland habitats on Groote Eylandt to ascertain if frequent,

104　large fires and feral cats, are influencing native mammal distribution and abundance.

105

106　**Methods**

107　*Study site*

108　Groote Eylandt is in the Gulf of Carpentaria, approximately 50 km from the Northern Territory

109　mainland (Fig. 1). The island is of low relief (<220 m above sea level) and has a monsoonal

110　tropical climate (Bureau of Meteorology, 2017b). It is characterised by laterite plains and

111　sandstone plateaux, with eucalypt woodland the dominant vegetation. Several other habitats

112　are present, including mangroves, monsoon vine thickets, coastal grasslands and shrublands.

113　The Groote archipelago is an Indigenous Protected Area and is managed by the Anindilyakwa

114　Land Council. There is a large manganese mine (Groote Eylandt Mining Company: GEMCO)

115　with an associated port and settlement (Alyangula) on the western side of Groote Eylandt,

116　which has been a significant driver of economic and infrastructure development. Two other

117　settlements are present on the island; Umbakumba and Angurugu (Fig. 1).

118   *Mammal survey*

119   We surveyed 112 lowland woodland sites across Groote Eylandt between April and September

120   2016 (Fig. 1). Lowland woodlands comprise a complex of vegetation associations that

121   collectively dominate the non-plateaux areas of Groote Eylandt. They are the preferred habitats

122   for the brush-tailed rabbit-rat and support a range of other CWR mammals, including the

123   northern hopping-mouse (Firth et al., 2006b, Diete et al., 2016). Surveying a broader range of

124   habitat types would have diluted our sampling effort and we were interested in investigating

125   environmental determinants of mammal species distribution rather than broad habitat

126   associations. Survey sites were stratified across land systems with lowland woodland (Lynch

127   and Wilson, 1998) by fire history (time since last fire; NAFI, 2017). All sites were a minimum

128   of 1 km apart and located within 2 km of vehicle access tracks.

129   At each site, six Reconyx Hyperfire camera traps (Reconyx Inc., Holmen, WI, USA) were

130   deployed for a minimum of 35 consecutive nights. Five cameras were deployed in a diamond

131   configuration with four cameras deployed approximately 50 m away from the centre camera.

132   Cameras were deployed facing south with a near-horizontal field of view; with the top of the

133   camera 35 cm above ground level and facing the base of a bait station 150 cm away containing

134   a mixture of peanut butter, honey and oats. Cameras comprised four white flash (either

135   Reconyx PC850 or HC550) and one infrared flash camera (either Reconyx HC500 or HC600).

136   This combination reflected camera availability and was the same at each site to control for

137   operational variability amongst models (Heiniger and Gillespie, 2018). This camera

138   configuration has been shown to adequately detect feral cats for occupancy modelling in

139   northern Australian (Stokeld et al., 2015), and to achieve high detection probabilities for a suite

140   of native species (Gillespie et al., 2015).

141 A sixth white flash camera was deployed within the diamond with a setup modified from

142 Welbourne (2013). A Reconyx PC850 camera modified for higher sensitivity and with a focal

143 length reduced to 90 cm (henceforth referred to as modified PC850) was deployed 65 cm from

144 ground level and angled downwards at 45° to face the middle of a corkboard 90 cm away. The

145 30 x 30 cm corkboard was used to improve small mammal detections in the tropical

146 environment by creating a substrate with a more homogenous temperature, therefore increasing

147 contrast with the thermal signature of small animals (Welbourne, 2013). A bait station was

148 placed at the centre of the rear edge of the corkboard, and drift fences extended 4 m either side

149 of the corkboard to direct mammals towards the detection zone.

150 All camera deployments were undertaken with high precision, using tape measures for

151 distances and test photos to ensure alignment of camera sensor zones with target areas.

152 Vegetation within each camera's field of view was cleared to increase detections, reduce false

153 triggers and risk of camera damage from fires. All cameras operated continuously and were set

154 to take three photos per trigger, with a one-second delay between each photo. Camera

155 sensitivities were set to high, with no quiet period.

156 *Feral cat density*

157 To further evaluate the relationships between feral cats and native mammal occurrence, we

158 investigated feral cat population density in two areas; (i) a northern site with high brush-tailed

159 rabbit-rat occupancy, and (ii) a southern site with no brush-tailed rabbit-rat detections (Fig. 1).

160 At each location camera trap arrays were established for a minimum of eight weeks from mid-

161 September to mid-November 2016, using methods modified from McGregor (2015). Each

162 array overlaid a vehicle track and comprised 70 camera traps arranged in 5 rows of 14 stations

163 spaced between 400 – 600 m apart. Each array contained the same combination of three white

164 flash camera models (PC850, modified PC850 and HC550), which were assigned randomly to

165   stations. Cameras were positioned locally to maximise feral cat detections, for example along

166   dry creek-lines, bush tracks or naturally open areas. Cameras were deployed at a height of 45

167   cm to the top of the casing, facing south and focused on an attractant 2.5 m away. The attractant

168   consisted of a non-toxic feral cat bait (kangaroo and chicken meat sausage; Scientec Research

169   Pty Ltd, Warrandyte, Australia) and a stick soaked in cat urine. Camera operating settings were

170   the same as the mammal survey.

171   *Data analysis*

172   We chose a set of environmental predictor variables that best described the fire regime, site-

173   specific habitat characteristics, landscape attributes and anthropogenic disturbances, based on

174   current knowledge of factors influencing northern Australian mammal declines (Kutt and

175   Woinarski, 2007, Woinarski et al., 2011a, Woinarski et al., 2010, Davies et al., 2017). It was

176   not possible to incorporate the occupancy of large mammalian predators (feral cats or dingoes)

177   as predictor variables in analyses because detections of these species during the survey were

178   too sparse.

179   Following Lawes et al. (2015), we derived multiple variables to quantify the fire regime at each

180   site, including the proportion of surrounding area burnt, patchiness, fire frequency and time

181   since fire. However, preliminary investigation of these fire variables demonstrated significant

182   collinearity. Therefore, we constrained our analyses to include one fire variable: the proportion

183   of the circular area (radius of 3200 m or 250 m depending on the species home range)

184   surrounding each site that is burnt each year, averaged over the five years preceding the

185   mammal surveys.

186   All predictor variables were centred and standardised by subtracting means and dividing by

187   standard deviations (Gelman and Hill, 2006). Collinearity was assessed amongst all variable

188   pairs and the most biologically relevant variable was retained for analysis when the Pearson

189   correlation coefficient was greater than 0.5 (positively or negatively). The resultant set of

190   predictor variables (Table 1) was used to develop occupancy models for species with adequate

191   detection probabilities (MacKenzie et al., 2002) and to evaluate patterns of species diversity

192   (Shannon diversity index). Occupancy modelling was conducted using the "unmarked"

193   package in R (Fiske and Chandler, 2011, R Core Team, 2017).

194   Single-season occupancy modelling was applied in two steps due to the large number of

195   predictor variables. First, the variables that best explained the detectability of each species were

196   determined by running all combinations of the eight predictor variables (i.e. 256 models) which

197   could potentially influence detectability (Table 1). This was done with occupancy constrained

198   to a saturated model of the eight occupancy variables. Model selection based on Akaike

199   Information Criteria (AIC) was then used to select the most parsimonious detectability model

200   in the candidate set. Second, all combinations of the eight occupancy variables were run (256

201   models) with detectability constrained to the most parsimonious model identified in step one.

202   Model selection based on AIC was used again to determine the best occupancy model. Where

203   no single model explaining the occurrence of the species emerged as superior (ΔAIC < 2),

204   model averaging was performed to provide parameter estimates (Burnham and Anderson,

205   2002). The detection probability over the survey period was determined for the best and null

206   model using the formula: $1-((1-p)^n)$, where p is the nightly detection probability and n is the

207   number of survey nights. We assessed the goodness of fit of the most saturated model for each

208   species using three tests based on parametric bootstrapping: Pearson's chi-square statistic, sum

209   of squared errors and Freeman-Tukey chi-square statistic. These tests repeatedly simulate

210   datasets based on the fitted model and then evaluate the probability that the observed history

211   of outcomes has a reasonable chance of happening (MacKenzie and Bailey, 2004).

212   For species with highly saturated occupancies, Royle-Nichols occupancy modelling was

213   applied, which assumes that heterogeneity in detection frequency is indicative of variability in

214   species abundance (Royle and Nichols, 2003). Modelling was otherwise conducted using the

215   same process as above.

216   Linear regression models were used to investigate the relationship between species diversity

217   and environmental parameters. Species diversity was modelled using the Shannon diversity

218   index (H), which takes into account the number of different species at a site and how even their

219   abundances are (Shannon, 1948). Because individuals of a species generally cannot be

220   distinguished using camera traps, we used the number of nights each species was detected as a

221   proxy for relative abundance. Model selection based on AIC was used to determine the best

222   model. Where no single model emerged as superior (i.e. ΔAIC < 2), model averaging was

223   performed to provide parameter estimates (Burnham and Anderson, 2002).

224

225   **Results**

226   *Mammal survey*

227   Twelve native mammal species were detected (Table 2). Naïve occupancy rates varied from

228   2% for short-eared rock wallabies (*Petrogale brachyotis*) to 100% for northern brown

229   bandicoots (*Isoodon macrourus*). Brush-tailed rabbit-rats were detected at 17% of survey sites,

230   mostly clustered on the northern peninsula (Appendix S1) and northern hopping-mice were

231   detected at only 3% of sites (Appendix S1). Feral cats were detected on nine occasions at eight

232   highly dispersed sites (Appendix S2), resulting in low overall detection probability (0.22).

233   *Correlates of species occupancy and relative abundance*

234   Data were adequate for single-season occupancy modelling for the delicate mouse (*Pseudomys*

235   *delicatulus*), grassland melomys (*Melomys burtoni)*, and the brush-tailed rabbit-rat. Nightly

236   detectability from the best model ranged from 0.13 for the grassland melomys to 0.18 for the

237   brush-tailed rabbit-rat (Table 3). Due to the length of time cameras were deployed (minimum

238   of 35 nights) the overall detectability of each species was very high (>0.99; Table 3). No single

239   occupancy model arose as superior for any of the three species (i.e. ΔAIC < 2; Appendix S3),

240   so model averaging was conducted to calculate parameter coefficients. Distance from mine or

241   town was the strongest predictor variable for the brush-tailed rabbit-rat (Fig. 2a), with the

242   species occupying sites further from these features. Occurrence of brush-tailed rabbit-rat and

243   grassland melomys were also positively associated with shrubs (Fig. 2). In contrast, these

244   variables had only marginal influences on delicate mouse occurrence, which was positively

245   associated with proportion of area burnt (Fig. 2b).

246   Northern brown bandicoots were detected at every site and northern quolls at all but two.

247   Royle-Nichols occupancy modelling did not produce a single superior model for either species

248   (i.e. ΔAIC < 2; Table 4 & Appendix S4) and model averaging was conducted. Northern brown

249   bandicoots had greater relative abundance at sites with more logs and shrubs, and a greater

250   proportion of the site burnt (Fig. 3a). Relative abundance of northern quolls increased with

251   ruggedness and proximity to major anthropogenic disturbance and roads (Fig. 3b).

252   Eight mammal species were included in species diversity analyses: northern quoll, northern

253   brown bandicoot, northern hopping-mouse, brush-tailed rabbit-rat, delicate mouse, grassland

254   melomys, common planigale (*Planigale maculata*) and short-beaked echidna (*Tachyglossus*

255   *aculeatus*). No single model of species diversity was superior and model averaging did not

256   identify any significant relationships between Shannon's diversity indices and predictor

257   variables.

258   *Feral cat density*

259   There were no feral cat detections on the northern camera array and only seven detections, at

260   four different camera stations, on the southern array (Appendix S2). Based upon distinct pelage

261 patterns, these detections comprised four individual cats, but detections were too sparse to

262 estimate density.

263

264 **Discussion**

265 Native mammal populations have suffered widespread contraction across mainland northern

266 Australia (Woinarski et al., 2001, Woinarski et al., 2010). Research suggests that mammal

267 populations on some large islands are now also exhibiting similar signs of decline to those

268 recorded earlier on the mainland (Davies et al. 2017; 2018). On Groote Eylandt, naïve

269 occupancy estimates of all CWR native mammals were higher compared to lowland savanna

270 woodland habitats across mainland Northern Territory (Table 2; Stokeld et al., 2016). This

271 reaffirms that, although fewer mammal species occur on Groote Eylandt (as with other large

272 Northern Territory islands), their populations remain healthier than those on the mainland. This

273 is likely related to several interrelated factors, including the low density of feral cats, the

274 relatively benign fire regime and the absence of large introduced herbivores and cane toads.

275 Feral cats have been implicated as a major driver in the decline of mammals across northern

276 Australia. Although widely distributed across Groote Eylandt, naïve occupancy and detection

277 probability of feral cats were low compared to similar habitats on mainland northern Australia

278 and the Tiwi Islands (Table 2). This precluded both an evaluation of environmental patterns of

279 feral cat occurrence, as well as our ability to investigate the relationship between feral cats and

280 the distribution/abundance of native mammals. The number of cat detections on camera grids

281 was also too low to estimate population density. In contrast, feral cat density estimates from

282 mainland Northern Territory using similar methods have ranged from 0.09 - 0.5 cats per km^2

283 (Stokeld et al., 2016; Northern Territory Government, unpublished data, 2019) and McGregor

284 (2015) estimated a mean density of 0.18 cats per km^2 in woodland habitats in the Kimberley,

285 Western Australia. The detection of zero and four individual cats on our two camera grids,

286 combined with the low overall naïve occupancy, suggests that cat densities on Groote Eylandt

287 are far lower than those on the mainland. This finding contrasts with recent modelling

288 predicting that feral cat densities should be higher on islands than on the mainland (Legge et

289 al., 2017).

290 Interestingly, the density of feral cats remains low on Groote Eylandt despite well-established

291 populations of cats in two towns and one out-station (Anindilyakwa Land Council, unpublished

292 data, 2018), as well as high densities of their preferred prey (as inferred from the high

293 occupancy rates of CWR mammals presented in this study). This suggests that the population

294 dynamics of feral cats on Groote Eylandt is not simply influenced by prey availability. Due to

295 a range of factors (i.e. high prey availability, the absence of both cane toads and dingo control),

296 Groote Eylandt supports healthy populations of native predators that are known to prey on cats,

297 including dingoes (Kennedy et al., 2012, Moseby et al., 2012, Allen and Leung, 2012,

298 Paltridge, 2002, Wysong et al., 2019), olive pythons (H. McGregor, unpublished data, 2018)

299 and sand goannas (T. Jessop, unpublished data, 2018). It is plausible that the healthy

300 assemblage of native predators may suppress feral cat populations through both direct

301 predation and interference competition. The combination of a relatively benign fire regime and

302 the absence of large feral herbivores, may also have maintained the vegetation in a state that

303 reduces feral cat hunting success (McGregor et al., 2015, McGregor et al., 2017, McGregor et

304 al., 2016), although how these constraints may be offset by high prey availabilty is unknown.

305 The contrasting densities of feral cats on Groote Eylandt and the Northern Territory mainland

306 highlights our limited understanding of the factors that regulate their population dynamics.

307 In northern Australian savannas, frequent, high-intensity fires reduce the availability of critical

308 resources for native mammals, such as tree hollows (Woolley et al., 2018), fleshy-fruit bearing

309 shrubs (Vigilante and Bowman, 2004) and perennial grasses (Crowley, 2008). We found no

310   significant negative association between the occupancy or abundance of native mammals and

311   frequent, large fires on Groote Eylandt. In fact, we found a significant positive association

312   between our fire variable and northern brown bandicoot abundance and delicate mouse

313   occurrence. There are several potential explanations for these results. First, Groote Eylandt is

314   characterised as having a relatively benign fire regime compared to mainland northern

315   Australia. For example, the average annual area burnt on Groote Eylandt between 2000 and

316   2018 was 18.1% (ranging from as low as 1.5% to 40.1%; Department of Environment and

317   Natural Resources and Anindilyakwa Land Council, 2019), compared to Kakadu National Park

318   where around 50% is burnt annually (Russell-Smith et al., 1997). As such, native mammals

319   may simply not be strongly impacted by the fire regime on Groote Eylandt. Second, fire

320   influences fauna indirectly, via effects on vegetation structure and composition, including

321   complex interactions with other exacerbating factors such as predation (McGregor et al., 2017,

322   McGregor et al., 2015) and grazing by large introduced herbivores (Legge et al., 2019).

323   Therefore, given the absence of large introduced herbivores, as well as the low density of feral

324   cats on Groote Eylandt, such positive associations with fire are not necessarily surprising

325   (Radford, 2012). Third, while our fire variable, the proportion of area burnt averaged over the

326   preceding five years, captured the spatial and temporal variation in fire regimes on Groote

327   Eylandt, we do not have a firm understanding of how this variable encapsulates other

328   characteristics of the fire regime, such as intensity and patchiness. Due to issues of collinearity,

329   we did not investigate the influence of these other fire variables. In reality, the absence of strong

330   negative fire effects in our analyses likely reflects a combination of these explanations. We

331   suggest that future research should aim to elucidate how different fire characteristics influence

332   biodiversity on Groote Eylandt.

333   The high naïve occupancy rates of native mammals on Groote Eylandt suggest that populations

334   are relatively healthy when compared to mainland northern Australia (Table 2). However, due

335   to a lack of standardised historical data on Groote Eylandt, we cannot properly evaluate the

336   trajectory of native mammal populations. As such, the current state of native mammals on

337   Groote Eylandt should not be taken as evidence that these populations are safe from decline,

338   or that they are not currently declining. Despite the availability of seemingly suitable habitat,

339   the distribution of the northern hopping-mouse was highly restricted. This suggests that

340   populations of this species may have suffered declines on Groote Eylandt. However, the

341   reasons for this remain unclear.

342   Although less restricted, the brush-tailed rabbit-rat was absent from a large proportion of sites

343   in seemingly suitable lowland woodland. As with the findings of Davies et al. (2017), site-

344   occupancy of the brush-tailed rabbit-rat was positively associated with shrub density, which

345   has been hypothesised to reflect a greater availability of resources for native mammals (Friend,

346   1987, Davies et al., 2017), as well as shelter from predators (Briani et al., 2004). Our results

347   may also reflect a combination of these factors; however further research is required to

348   ascertain what factors may be limiting the distribution of this species on Groote Eylandt and

349   how they contrast with those operating on other extant populations.

350   Despite the seemingly low densities, cats could be impacting native mammal populations if

351   predation rates exceed recruitment and recolonisation (Ceballos and Cruzado 2004, Frank et

352   al., 2014, Short, 2016, Moseby et al., 2015). Both threatened rodent species may have relatively

353   limited dispersal capabilities (Firth et al., 2006a, Diete et al., 2016), and the brush-tailed rabbit-

354   rat has low fecundity, giving birth to one to three young each year (Taylor and Horner, 1971).

355   These species may therefore be limited in their ability to recolonise areas following disturbance

356   such as fire. Furthermore, studies have shown that cats selectively target CWR mammal species

357   (Kutt, 2012, Wysong et al., 2019), and that even at low densities, cats can exert significant

358   predation pressure on mammal populations (Frank et al., 2014). As such, despite their low

359   density on Groote Eylandt, feral cats could still be posing a significant threat to populations of

360   the brush-tailed rabbit-rat and northern hopping-mouse.

361   This research has identified important knowledge gaps that warrant further investigation to

362   better inform management:

363   • Greater understanding of the factors that regulate feral cat population densities in

364   northern Australia; in particular, to what extent are they limited by native predators;

365   • The impact of feral cats at varying densities on CWR mammal populations, and in

366   different environmental settings; and

367   • How fine-scale fire regimes are changing over time on Groote Eylandt and other

368   offshore islands, and what impacts these are having on biodiversity.

369   The greater resilience of mammal populations on Groote Eylandt compared to the northern

370   Australian mainland is likely due to the low density of feral cats, the relatively benign fire

371   regime and the absence of large introduced herbivores and cane toads. These factors have likely

372   benefited mammal populations by maintaining the availability of critical resources, while

373   concurrently reducing predation pressure. The resilience of mammal populations on Groote

374   Eylandt appears comparable to that of mammal populations on the Tiwi Islands. However, the

375   sources of resilience may differ between these areas. Compared to Groote Eylandt, there is a

376   greater number of threatening processes present on the Tiwi Islands, including a higher naïve

377   occupancy rate of cats, higher fire frequency, large introduced herbivores (buffalo and horse)

378   and feral pigs. While the Tiwi Islands have substantially higher rainfall (with some areas

379   receiving more than 2000 mm of annual rainfall compared to 1300 mm on Groote Eylandt),

380   they lack the rugged sandstone escarpment habitat that likely provide important refugia for

381   some native mammals on Groote Eylandt (Radford et al., 2014). As such, mammal populations

382   on Groote Eylandt may be more resilient to decline due to higher overall ecological integrity,

383   whereas mammal populations on the Tiwi Islands may be more resilient primarily due to higher

384   productivity. This contrast highlights that the way in which native mammal population declines

385   manifest themselves (in terms of severity or apparent resilience) is influenced by complex

386   interactions of factors that vary substantially across the landscape. Caution is therefore required

387   in making broad inferences about primary factors driving mammal species declines from a few

388   studies that are spatially and temporally limited. Management responses to conserve these

389   species need to be developed at appropriate ecological and management scales. They also need

390   to be adaptive to respond to new insights into threats and management responses as they

391   emerge.

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Tables

Table 1. Description of landscape and site-specific environmental variables used in analyses.

Variable	Description	Rationale
Proportion burnt	Proportion of area surrounding each site that is burnt each year, averaged over the preceding five years. Area used was scaled to reflect different species home range sizes: 250 m radius for brush-tailed rabbit-rat, delicate mouse and grassland melomys (Firth et al., 2006a), and 3200 m radius for northern quoll, northern brown bandicoot and species diversity (Lawes et al., 2015b). Derived from	Too frequent and/or intense fires have been implicated in CWR mammal declines across northern Australia (Lawes et al., 2015a, Lawes et al., 2015b, Legge et al., 2011b, Andersen et al., 2005).

fine-scale (30 m x 30 m) LANDSAT imagery
following Lawes *et al.* (2015b). This variable
was correlated with fire frequency, total
ground cover (perennial, annual, sedge, forb
and herb cover) and perennial grass cover.

Ruggedness	Standard deviation in elevation (Geoscience Australia, 2017) estimated over 250 m and 3200 m radius, as per proportion burnt. Ruggedness was correlated with rainfall.	Rugged habitats provide refugia for CWR mammal species (Hohnen et al., 2016, Braithwaite and Griffiths, 1994).
Shrubs	The presence of shrubs under 1.5 m was recorded every metre along two edges of a 50 x 50 m quadrat.	Shrubs provide shelter from predators and food resources for CWR mammal species (McGregor et al., 2015), and predicted brush-tailed rabbit-rat occupancy on Melville Island (Davies et al., 2017).

Logs	Number of logs (diameter ≥5 cm) crossing the perimeter of a 50 x 50 m quadrat.	Logs provide den sites and shelter for some CWR mammal species (Firth et al., 2006b).
Distance to anthropogenic disturbance	Minimum distance from each site to GEMCO mining operations or one of the three towns (Alyangula, Angurugu or Umbakumba). This covariate was not used to predict mammal detectability.	Disturbances from human settlements and roads may have pervasive effects on biodiversity proximal and distal from these sources (Sodhi et al., 2010, Laurance et al., 2009, Karanth et al., 2006, de Fretes, 1992).
Distance to roads	Minimum distance to a mapped road. This covariate was not used to predict mammal detectability.	
Distance to water	Distance to the closest permanent waterbody, defined by presence of water for more than 50% of all Landsat 5 and Landsat 7 satellite observations (Geoscience Australia, 2015). Permanent water body distribution was determined using the Australian Hydrological Geospatial Fabric (Bureau of Meteorology, 2017a).	Proximity to watercourses can influence the occurrence of CWR mammal species (Firth et al., 2006b).

Basal area	Calculated from the circumferences of all trees (diameter at 1.3 m > 5 cm) within a 5 m belt around the 50 x 50 m quadrat. Correlated with number of large trees (diameter at 1.3 m > 40 cm) within 5 m of the 50 x 50 m quadrat.	Tree basal area can influence the occurrence of CWR mammal species (Firth et al., 2006b).
Number of cameras operating	An observation-level covariate of the number of cameras operating at each site for each sampling day. Used as a covariate to estimate mammal detectability only.	Variation in detectability can arise from uneven numbers of cameras operating at sites due to camera malfunction (Stokeld et al., 2015).
Julian day	Date that sampling started at each site. Used as a covariate to estimate mammal detectability only.	Seasonal differences can influence mammal detectability (Einoder et al., 2018)

699

700 **Table 2.** Naïve occupancy estimates of mammal species detected during camera trapping
701 surveys in lowland woodland across the Northern Territory. Groote Eylandt estimates are from
702 the current survey, Tiwi Island estimates are derived from surveys conducted on Melville

703　(2015) and Bathurst Island (2014), and mainland estimates are derived from surveys conducted

704　at Cobourg peninsula (2014), Fish River Station (2014), Kakadu National Park (2015),

705　Litchfield National Park (2016), Nitmiluk National Park (2015), peri-urban areas of Darwin

706　(2014) and Warddeken IPA (2013 – 2015).

707　NA Species is known to occur in the area, but naïve occupancy estimate is not available.

708　- Species not known to occur in the area.

709　+ Estimate from Melville Island only as the species is not known to occur on Bathurst Island.

Common name	Scientific name	Naïve occupancy		
		Groote Eylandt (n=112)	Tiwi Islands (n=110)	Mainland (n=131)
Brush-tailed rabbit-rat	*Conilurus penicillatus*	0.17	0.318	0.128
Dingo	*Canis lupus dingo*	0.36	0.46	0.5
Northern quoll	*Dasyurus hallucatus*	0.98	-	0.015
Feral cat	*Felis catus*	0.07	0.24	0.23
Northern brown bandicoot	*Isoodon macrourus*	1.00	0.918	0.471
Agile wallaby	*Macropus agilis*	0.85	0.855	0.572
Grassland melomys	*Melomys burtoni*	0.08	0.091	0.056
Northern hopping-mouse	*Notomys aquilo*	0.03	-	NA

Sugar glider	*Petaurus breviceps*	0.05	NA	NA
Short-eared rock-wallaby	*Petrogale brachyotis*	0.02	-	NA
Common planigale	*Planigale maculata*	0.03	-	NA
Delicate mouse	*Pseudomys delicatulus*	0.47	0.027	NA
Short-beaked echidna	*Tachyglossus aculeatus*	0.39	-	0.05
Fawn antechinus	*Antechinus bellus*	-	-	NA
Water rat	*Hydromys chrysogaster*	NA	NA	0.03
Antilopine wallaroo	*Macropus antilopinus*	-	-	0.122
Black wallaroo	*Macropus bernardus*	-	-	0.036
Common wallaroo	*Macropus robustus*	-	-	0.121
Black-footed tree-rat	*Mesembriomys gouldii*	-	0.636[+]	0.315
Northern nail-tail wallaby	*Onychogalea unguifera*	-	-	0.007
Western chestnut mouse	*Pseudomys nanus*	-	0.009	0.013
Black rat	*Rattus rattus*	NA	NA	0.039
Pale field rat	*Rattus tunneyi*	NA	0.109	0.006
Butler's dunnart	*Sminthopsis butleri*	-	0.064	-
Red-cheeked dunnart	*Sminthopsis virginiae*	-	0.018	0.013
	Trichosurus vulpecula	-	0.393	0.05

Common brushtail
possum

710

711   **Table 3.** Comparison of ΔAIC, occupancy, nightly and overall detection probabilities (over 37

712   nights) for the null model (where occupancy and detectability parameters are assumed to be

713   constant across all survey sites), and the most parsimonious single-season occupancy model

714   for each species.

Species	Model	ΔAIC	Occupancy (Ψ) (±SE)	Nightly detection probability (p) (± SE)	Overall detection probability
Brush-tailed rabbit-rat	Null model	199.58	0.17 (0.04)	0.24 (0.02)	0.99
	Best model	0.0	0.20 (0.06)	0.18 (0.03)	0.99
Grassland melomys	Null model	18.33	0.09 (0.03)	0.07 (0.02)	0.94
	Best model	0.0	0.11 (0.05)	0.13 (0.06)	0.99
Delicate mouse	Null model	0.0	0.52 (0.08)	0.13 (0.02)	0.99
	Best model	129.04	0.47 (0.05)	0.15 (0.01)	0.99

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716

717 **Table 4.** Comparison of ΔAIC, relative abundance, nightly probability of detection and the

718 overall probability of detection (over 37 nights) for the null model (where relative abundance

719 and detectability parameters are assumed to be constant across all survey sites), and the most

720 parsimonious Royle-Nichols occupancy model.

Species	Model	ΔAIC	Relative abundance (λ) ($\pm$SE)	Nightly detection probability (p) ($\pm$ SE)	Overall detection probability
Northern brown bandicoot	Null model	55.24	3.76 (0.38)	0.14 (0.01)	0.99
	Best model	0	5.22 (1.15)	0.11 (0.02)	0.99
Northern quoll	Null model	34.99	3.06 (0.37)	0.09 (0.01)	0.96
	Best model	0	4.35 (1.27)	0.07 (0.01)	0.92

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727 **Figure legends**

728   **Figure 1.** Groote Eylandt and its location in Australia. Location of the 112 survey sites (black

729   circles) and cat density grids (black open rectangles). Light grey – lowland forest and

730   woodlands; clear – sandstone outcrop and coastal grassland and shrubland complex; dark grey

731   – GEMCO mining operation area; dashed lines – roads.

732   **Figure 2.** Model averaged regression coefficient estimates for (a) the brush-tailed rabbit-rat,

733   (b) the delicate mouse and (c) the grassland melomys. Error bars indicate 95% confidence

734   intervals and asterisks indicate significant parameters where confidence intervals do not

735   overlap zero.

736   **Figure 3.** Model averaged regression coefficient estimates of relative abundance for (a) the

737   northern brown bandicoot and (b) the northern quoll. Error bars indicate 95% confidence

738   intervals and asterisks indicate significant parameters where confidence intervals do not

739   overlap zero.